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Ecological Restoration

Moving Forward Using Lessons Learned

Springer

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Foreword

Assume for a moment that you were recently promoted to the senior administration of a multinational institution engaged in natural resources preservation and management. Your training and experience are in public lands administration. Your board of directors has just assigned you to plan, initiate, and oversee a large ecological restoration program of truly global significance. Your acquaintance with ecological restoration mainly consists of having read review articles in the periodical literature. One realization pervaded these reviews: no two restoration programs are alike, and no 'cookbook' or checklist can be employed in the design and implementation of ecological restoration.

What is your first move? My advice is that you read this book.

The book embodies the essential wisdom of ecological restoration which has accumulated since the late 1970s. That was the time when the restoration discipline coalesced from a few isolated attempts at degraded ecosystem recovery and began stimulating the imaginations of many environmentalists. Some chapters describe how major biomes are restored, such as forests, wetlands, and grasslands. Other chapters treat topics that are common to any restoration program, such as the genetics of seed stocks, degree of reliance on natural regeneration, and the importance of input from stakeholders and local communities.

Each chapter is authored by an experienced restorationist, often a scientist with practitioner experience, who characterizes his/her topical area in a consistent format that allows ease in cross-referencing. This summation is followed by case histories of exemplary restoration projects, again presented in a standardized format. Each project is described, key outcomes are presented, lessons learned are given detailing what went right and what didn't, and management implications are specified.

These chapters and their case histories will give you knowledge from the direct experience of those who have already designed and administered restoration programs. The characterizations and case histories bring restoration to life in a way that can't be derived from a cookbook or checklist. The text is ample, so that you can grasp both principles and nuances of restoration project experience but sufficiently succinct to keep you from getting lost in technical details.

Absorbing the knowledge in this book will allow you to converse intelligently with the practitioners and scientists on whom you will depend for designing and implementing your restoration program. You will also be able to explain and defend your program to funding authorities and government regulatory officials. Defending your program is particularly challenging, since much of the effort and need for funding comes in the design phase and in the early implementation efforts. Long-term funding also needs to be defended, particularly for monitoring to determine when active restoration activities can be safely terminated.

What if you are not engaged in designing a global restoration program for a multinational institution? Recent graduates with a bachelor's degree who are looking for a challenging career in environmental science will also acquire insight from this book and so will college professors who want to add a course or seminar in ecological restoration to departmental offerings. Experienced restoration practitioners will benefit from this book by reading about different facets of the restoration discipline than those with which they are intimately familiar from their own work.

This book is a celebration of the coming-of-age of the emerging profession of ecological restoration. Long-time restorationists will be amazed at the depth and degree of sophistication that has developed in the past decade in restoration projects. In years past, ecological restoration was essentially an art and craft that was practiced on a trial-and-error basis. Subsequently, the profession has become much more of a science with predictable results, as will become apparent from reading the case histories in this book. Older restorationists will be awestruck by the level of sophistication to which our beloved discipline has arisen. We will all be heartened by these advances which are nicely encapsulated in this book.

Past-president Andre Clewell
Society for Ecological Restoration,
Washington, DC, USA

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About the Editors

Singarayer Florentine is a restoration and invasive species ecologist, with more than 20 years of experience in research and higher education teaching. He has been specialising in work on ecologically diverse habitats, and has had experience in three different countries. Florentine earned his PhD from Curtin University in Western Australia, and then moved to Queensland where he worked as Weed Scientist with the Tropical Weeds Research Centre. While he was with the School for Field Studies at the Centre for Rainforest Studies, Queensland, he was involved in several tropical rainforest restoration projects and began to conduct research into weed invasion in fragmented landscapes.

Linda Broadhurst recently retired as Director of the Centre for Australian National Biodiversity Research, a joint venture between CSIRO National Research Collections Australia and the Director of National Parks. During Linda's research career, her research interests included conservation and restoration genetics and improving seed collection practices for better restoration outcomes. She also spent many years translating scientific findings to help improve on ground seed collection and use practices to ensure that plantings has a broad genetic base.

Paul Gibson-Roy is a restoration ecologist specialising in grassland and grassy woodland communities. Paul has worked with various groups including universities, NGOs, governments and private businesses to increase the focus on the need to protect and restore these communities. His work has included detailed research and field-scale implementation of practice. In addition, he has also been keenly involved in better understanding the dynamics and capacity of the broader seed and restoration sectors, in particular around seed production and seed supply chains, to provide effective restoration services. He has spent many years communicating findings from research and practice and advocating for better restoration policy, programmes and outcomes.

Kingsley Dixon is a restoration ecologist and conservation biologist, with 40 years of global experience focused on landscape-scale restoration and conservation through community-led practices. He holds positions on international and national boards and commissions including President of the international Society for Ecological Restoration and member of the Task Force on Best Practices for the UN Decade on Ecosystem Restoration. Kingsley was awarded the Linnean Medal in 2013 for his significant contribution to the science of natural history.

Chapter 1
Ecological Restoration: Moving Forward Using Lessons Learned

Introduction

Singarayer Florentine, Linda Broadhurst, Paul Gibson-Roy, and Kingsley Dixon

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Background

As we move into the second decade of the twentieth-first century, it is impossible to deny that the ecosphere of our planet is in dire trouble from a range of biotic and abiotic pressures, many of which are widely accepted to continue into the indefinite future. While human activity has unarguably altered the natural world for millennia,

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the past 150 years or so have seen a significant increase in the natural resources required to support our way of life, the extraction of which has been to the detriment of our natural world. It is also increasingly apparent that human health and well-being are inextricably linked to having a healthy natural world, and that restoring and repairing our planet is therefore of critical and urgent importance. In addition, it is apparent that despite the increasing mobilization and utilization of our natural resources, we still face serious national and international challenges and inequalities. The United Nations now identifies 23 issues of pressing global concern (https://www.un.org/en/global-issues), including population growth, climate change and food security issues. Of the global population of 7 billion people, there are currently some 736 million people who exist in extreme poverty (World Bank, https://blogs.worldbank.org/opendata/half-world-s-poor-live-just-5-countries?cid=ECR_TT_worldbank_EN_EXT), with many more millions struggling with poor sanitation, lack of potable water, limited education and little food security.

Whilst it is recognized that global actions are in place to lift people out of extreme poverty and meaningfully improve their health and education, the additional resources required to support an estimated 8–10 billion people by 2050 will further exacerbate biodiversity decline. In addition, the Intergovernmental Panel on Climate Change (IPCC) has been warning us since 1988 of the profound consequences which will affect humans due to climate change. This concern has catalyzed action at all levels of government, science, industry and community to reduce carbon emissions in order to avoid the most extreme climate change predictions.

In parallel with these actions, most of which have been focused on human well-being, the Intergovernmental Science-Policy Platform for Biodiversity and Ecosystem Services (IPBES) has since 2012 been documenting the dramatic decline of the natural world and of the critical role biosystems play in the maintenance of peoples' health and well-being (Díaz et al. 2018). Indeed, it is now widely understood that the recovery of global biodiversity will have many benefits beyond improved ecological outcomes for vegetative species. There will be significant advantages, for example, through increased purity of water and atmosphere, for many aspects of human health, and for general socio-economic well-being (Aronson et al. 2020). Exemplifying the importance that is being placed on recovering nature is that on 1 March 2019 the United Nations General Assembly declared 2021–2030 to be the United Nations Decade on Ecosystem Restoration. The aim of this declaration was to help ensure that all nations contribute to actions designed to increase the betterment of nature and the enhancement of the well-being of peoples of all countries. It is anticipated that, through the immediate prevention of further degradation of the planet's natural ecosystems, and by repairing the damage which has already been done, natural resilience and restoration activities will be able to re-establish healthy biospheres.

However, while this global commitment by the UN indicates that there is a clear collective importance and urgency for us to come together as one global community to address this biodiversity crisis, the task ahead is large and complex. It is recognized that *undertaking global-scale restoration will require a global commitment.* It will require supportive and finely tuned government policies and programs, the highest level of scientific input and significant investment of resources and skill and time from industry and involved citizens. Equally, at the local level, it is important to recognize that restoration is most likely to be successful and enduring if it is influenced and molded by positive social, environmental and economic factors, such as when financial commitments and investments are deployed in equitable ways to achieve the highest level of restoration outcomes for both nature and the surrounding community.

The Focus of This Book

The need to learn from previous restoration activities to avoid the confidence-eroding effects of restoration failures has been long called for (Michener 1997). Proper documentation of restoration actions and their outcomes will strengthen the body of knowledge at our disposal to improve future practice. Unfortunately, little documentary evidence has historically been gathered from across different parts of the world, leaving a dearth of benchmarks against which restoration success can be judged. Limited documentation and assessment have, in part, been due to a lack of standardized national and international guidelines which are now becoming available. In addition, there are few published experimental methodologies, a resource that could be useful in developing locally focused restoration strategies. In this regard, there are some examples which might serve as exemplars, with the Florabank Guidelines outlining the best practice approaches for native seed collection and use (https://www.florabank.org.au/guidelines/) and the International Principles & Standards for the Practice of Ecological Restoration (Gann et al. 2019) which takes a wider perspective on strategic planning in a range of specific climate situations. Experimental methodologies of a more general nature have been discussed by Prober et al. (2018), but the paucity of information available has led to a general failure to learn from experience – a significant impediment to informing current and future restoration practice.

As with many practices that have merit, the lack of documentation in restoration has, in large part, arisen because funding agencies rarely provide adequate resources to enable projects to systematically appraise or evaluate restoration outcomes over long time periods. This has meant that there are few documented lessons for practitioners to learn from. The consequence of this situation is that knowledge improvement, which is so crucial for refining and increasing the scale and complexity of restoration strategies, remains elusive. In this respect, the types of questions that could guide the appraisal of historic restorations might include:

(i) Have the restoration activities been documented?
(ii) What form of documentation and evaluation took place?
(iii) Was the documentation undertaken at an accepted standard?
(iv) What documented projects succeeded or failed?

(v) Was there creditable science behind documented projects, be they successes or failures, and, if so, what type or scientific method was applied?
(vi) What lessons have been learned from the documentation and review of past restoration projects which can be used to inform future planning in ecological, technical and social areas?
(vii) What is the likelihood that proven procedures in one context, such as landscape scale, specific vegetation type and defined ecosystem, might be suitable in other systems?
(viii) In future, can successful restoration approaches carried out in the past be undertaken at lower cost without prejudicing restorative outcomes?

Questions of this type may help us to identify and learn from past projects. At the same time, projects which are designed to apply accepted national and global standards for ecological restoration will be more likely to document their restoration strategies and assessment methods. This reflective action will ensure that gains are made to restoration knowledge and practice, providing exemplars which are globally optimized, socially endorsed and community supported, thus ensuring their enduring value for people and nature.

Who Should Read This Book?

While restoration ecology is a relatively new discipline, it has gained significant prominence in recent decades as the pervasive and damaging impacts of humans on nature have become more widely discussed. Consequently, there have been a range of high-quality books focusing on relevant issues and related topics, and these have been supported by a burgeoning of peer-reviewed journal literature. In this context, the key feature of this book is its focus on restorations of the recent past, emphasizing aspects of what worked and what didn't. To meet this aim, the book brings together the accumulated knowledge and experiences of eminent researchers and practitioners from across the globe. The chapters have been systematically separated into a range of topics that encompass the broad remit of global ecological restoration. Some chapters have focused on specific ecosystems or biomes, whilst others examine aspects that could underpin restoration practice. Each chapter provides an introduction and discussion of a particular topic area, then delves much more deeply into the relevant field of practice through detailed and informative case studies that illustrate the implementation of ecological restoration across a range of scenarios. The authors' experiences in these case studies highlight elements of the practice of ecological restoration in considerable detail, focusing on project goals, key features and strategies, obstacles and barriers faced and major achievements. Perhaps more importantly, these case studies reflect on what did and did not work providing readers with information to evaluate their own projects.

If the lofty goals which have often been set for repairing global degradation are to be achieved, it is highly likely that a close and continual examination of restoration outcomes will be critical. In addition, learnings from the experience need to be rapidly and widely disseminated to ensure ongoing improvement in related projects. As editors of this book, we hope this addition to the literature will play some part in more firmly establishing a paradigm of ‘reflecting on the past to inform future practice’. Indeed, we believe the information and experiences presented in these Chapters will be of broad interest and applicability to all those engaged in the field of restoration ecology, be they restoration practitioners, land managers, farmers, traditional owners, researchers, students, environmental consultants, government agencies, not-for-profit organizations and, of course, to the wider community. Perhaps most, we hope the book will provide all readers with the inspiration, courage and confidence to undertake ecological restoration, which is most urgently needed (at whatever level and scenario), to preserve our natural world.

Book Structure

To assist the reader to locate information of particular relevance or interest, this book is organized into four focal Themes. The first Theme, ‘Restoring ecosystems and species’ (Chaps. 2–8), provides lessons learned from experiences with different vegetation communities and species using case studies from around the world. Chapter 2 provides a comprehensive account of ecological restoration that has taken place in grassy communities with four case studies selected to illustrate both the success stories as well as the challenges and pitfalls when restoring this class of plant community. This chapter is essential reading for those involved in grassland or grassy woodland/savannah ecological restoration being suitable for on-ground practitioners, researchers and natural resource management agencies. It highlights restoration of communities in the northern and southern hemispheres, including temperate and tropical locations. The authors of Chap. 3 focus on a range of long-term projects (case studies) sourced from tropical Australia, Asia and Central America. These projects provide a comprehensive account of the ecological restoration benefits of the different approaches used. Chapter 4 also deals with tropical rainforest restoration, but here the authors have chosen to illuminate the lessons learned from the tropical wet Asian context. Case Studies have consequently been selected from long-term study sites in the biodiversity hotspots of South and Southeast Asia, which include regions in Borneo, India, the Philippines and Sri Lanka. Chapter 5 is dedicated to an examination of the ecological restoration activities that have taken place in temperate forests. Because restoration in these different forest environments is unique, the authors use three case studies to specifically emphasize the key issues in a temperate forest restoration context. In Chap. 6, the authors have provided a range of success stories involved in the restoration of wetlands and riparian zones using six case studies to clearly illustrate lessons learned, especially the importance of hydrology in wetland ecological restorations. This set of case studies should prove to be very useful for a range of stakeholders, including catchment management authorities and researchers. The final two chapters in this book focus on other aspects of ecological restoration. Chapter 7 focuses on landscape resilience and assisted regeneration since it is widely recognized that the practice of ecological restoration is diverse and complex and requires practitioners to interpret and act on varied environmental and landscape cues in deciding which approaches to take in a new restoration project. Chapter 8, the final chapter in Theme 1, addresses various strategies and techniques that can be applied in the discipline of rare and threatened plant conservation and translocation. This unique area of ecological restoration is often viewed to be at the cutting edge of the discipline, and this chapter illustrates some of these advances via case studies.

Chapters 9, 10, and 11 present Theme 2, 'Highly human-modified landscapes', and address three major areas. In Chap. 9, US-based researchers provide a comprehensive overview of the program used to restore roadsides with native plants. These experienced practitioners and researchers have elaborated on what worked and what did not work in relation to several major projects undertaken on roadside locations in the Pacific Northwest of the United States. Chapter 10 deals with the many and varied issues embedded in the restoration of urban landscapes. The authors have identified different goals, scales and approaches used in four illustrative case studies centered on restoring native nature to urban environments. Finally, in Chap. 11, the authors have focused and elaborated on the many complexities associated with ecological restoration in the context of mine rehabilitation. This is a highly scrutinized area, which garners much media and public attention. Importantly, these authors highlight the recent development of international standards for mine site restoration and describe other aspects of practice that are important to successfully undertaking restoration of severely disturbed post-mined landscapes.

Theme 3 deals with other areas important to the success of ecological restoration. Chapter 12 addresses the essential issue of strengthening the global native seed supply chain which is so necessary for ecological restoration to meet global goals. Its case studies center on different approaches used to secure seed supply and focus on areas including seed production, community collection networks and the need for improved standards of testing, and handling native seed. Finally, Chap. 13 provides a discourse on lessons learned from the unique field of plant genetics. The case studies which focus on the relevance of genetic considerations to restoration outcomes should be of interest and relevance to all managers, practitioners and researchers involved in the practice ecological restoration.

To consolidate the book, Chaps. 14 and 15 provide a distilled focus on the social and economic dimensions (Theme 4) of ecological restoration. Chapter 14 demonstrates that ecological restoration is very much a social practice, involving both the local meaning-making and the messiness of real life. It highlights how restoration itself is affected by historical and current human interactions, discourses, and equity all of which impact on the degree of involvement and support shown towards the practice of ecological restoration. Importantly this Chapter emphasizes how with appropriate goodwill and determination, these human aspects can be celebrated and harnessed for achieving ecological goals. Chapter 15 focuses on the economic and financial dynamics that underpin environmental policy and programs and how these affect the practice of ecological restoration. It discusses factors that drive environmental economics, including of ecological benefits analysis and cost-effectiveness analysis, which are used to devise programs that offset the loss of nature (through development) with conservation and restoration or those that incentivize the restoration of degraded nature on lands previously used for agriculture or other uses.

Finally, the editors wish to state they deeply recognize the important contributions that Indigenous peoples make to the protection and conservation of nature, both historically and during current times. We also acknowledge the important work done by Indigenous peoples in the field of ecological restoration across the planet and the important knowledge they possess that would benefit others engaged in its practice. Therefore, it is with great sadness that we were unable to engage with Indigenous authors who could lead a specific chapter on this topic. Despite this, we are pleased that various chapters refer to Indigenous led programs or ways of approach. Chapter 14, in particular, provides a clear focus on the importance of Indigenous knowledge and participation to the recovery of global biodiversity. Chapter 16 provides a final synthesis of the book where the editors draw together the various themes and learnings to distill take-home messages developed by the chapters' authors, which we hope will allow the reader to begin on their journey using ecological restoration towards a lasting and positive effect for nature and peoples.

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Part I
Restoring Ecosystems and Species

Chapter 2
Grassy Community Restoration

Paul Gibson-Roy, Chris Heltzer, Sandrine Godefroid, Thibaut Goret, Maïké Dellicour, and Fernando A. O. Silveira

Contents

Aust Case Study: Paul Gibson-Roy (1). US Case Study, Chris Heltzer (2); Belgian Case Study: Sandrine Godefroid (3); Thibaut Goret (3); Maïké Dellicour (3). Brazilian Case Study: Fernando A. O. Silveira (4)

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Summary and Key Lessons
Each case study presented shows how grassy community restoration can be undertaken to various degrees. While there are some differences in the approaches used, or in how they are structured, resourced and timed, all demonstrate that restoration can be used to recover or reintroduce species and grassy communities to lands where they are absent or degraded. This is a very important message given the threat these communities face and should give hope that there is the knowledge and the tools available to halt and even reverse grassy community loss should peoples choose to do so and are properly resourced.

They also give insights into various issues that make grassy community restoration difficult to undertake or undermine its chances of success. Several such as seed supply (e.g., it's accessing, quantity, quality and cost) or unsuitable site conditions (e.g., high weed or nutrient loads) are consistent across case studies, while others such as being able to secure land for restoration, creating opportunities for local communities (e.g., via seed networks), providing incentives for landholders or others to undertake restoration (e.g., funded programs) and improving sector capacity (e.g., skills, training, technology) are more pertinent to specific situations or settings.

While these Case Studies show that we have the knowledge to undertake grassy community restoration, for this to occur at landscape and global scales will require well-tailored support from governments to create the legislative structures and opportunities that support human communities to achieve these outcomes. Indeed, restoration at the scale needed to repair anthropogenic damage done over millennia will take a commensurate effort in terms of time, resourcing and commitment from countries, jurisdictions and their peoples. Some countries are more advanced in this

than others; however, this should create the opportunity for knowledge sharing and even of cross-jurisdictional support. If native grassy ecosystems can be better integrated into the fabric of our landscapes, be they farm-scapes, urban regions, transport corridors or others, then humans and a rich natural biota will benefit. For this to occur, purposeful decisions and goals, clear pathways and concrete actions must occur so that at times ad-hoc and intermittent successes of the past are turned into purposeful strategies and widespread global advances of the future.

Management Implications

Grassy communities can be restored to agricultural, forested, urban and other landscapes using regenerative and reintroduction approaches.

- Grassy community restoration can achieve high levels of species and functional diversity as well as temporal resilience.
- Restored grassy communities create a myriad of biodiversity, ecological and ecosystem service benefits.
- Restored communities must be purposefully managed and maintained over time to preserve their structural and compositional integrity.
- Effective seed supply chains and delivering seed in quantity, quality, price and ethically are critical to successful restoration.
- Creating better employment opportunities for individuals and communities, increasing technical skills and training and improving infrastructure and technology are all crucial to overcoming barriers to increasing the effectiveness and scale of restoration.
- Landscape-scale grassy community restoration will rely on the formulation of insightful and finely crafted government strategies and policies that create the settings, frameworks and coordination required to build markets, improve sector capacity and meet ambitious grassy community restoration targets.

Introduction

Grassy communities, including grasslands, prairies, steppes, meadows, grassy woodlands, savannahs and grassy-forest complexes, are present on all continents except Antarctica and cover an estimated 40.5% of these landmasses (White et al., 2000). They occur across many disparate regions, from tropical to tundra and alpine areas and from arid to temperate zones (Gibson, 2009; Squires et al., 2018; Wilsey, 2018). Grassy communities are dominated by ground layer vegetation, primarily of grasses and forbs, whilst within grassy woodlands and savannahs, trees and shrubs are key functional components giving a sparse open upper stratum (Wilsey, 2018; Raghurama & Sankaran, 2021). A common feature of all these different community types is that they support high plant diversity, even within relatively small-scale pockets (Partel et al., 2005; Morgan & Williams, 2015).

Continual vegetation disturbance due to grazing herbivores has been instrumental in the formation and maintenance of grassy community structure, where conditions such as soils and climate might have otherwise been suited to the development of forests (Nerlekar & Veldman, 2020). Other factors such as fire, aridity and cold are also known to aid in the restriction of woody dominance and the consequent retention of grassy communities (Gibson, 2009; White et al., 2000). These areas, maintained by such factors over long periods, are referred to as 'old-growth' grasslands (Buisson et al., 2019, 2021a, b; Nerlekar & Veldman, 2020; Silveira et al., 2020), whilst 'new-growth', 'derived' or 'semi-natural' grasslands are terms given to those areas formed more recently through human-mediated disturbance to other vegetation types, such as through forest clearing.

The rise and spread of modern humans (*Homo sapiens sapiens*) saw humans also become intimately connected to the formation and maintenance of grassy communities. This was done through their use of fire primarily to meet food, cultural, spiritual and other needs (Bird et al., 2008; Gibson, 2009; Gammage, 2010). Grassy communities provided direct food sources for hunter-gatherers, such as seeds and tubers, as well as being a source of animal fodder, which attracted game. Over time, this also provided reliable feed for domesticated livestock (Gott et al., 2015; Gammage, 2010, White et al., 2000). However, the transition by many human societies from nomadic to sedentary agricultural lifestyles resulted in increasing degradation and loss of native grassy communities, especially after the domestication of a narrow group of plant species, including corn, millet, rice, rye, sorghum and wheat, which led to the conversion of native grassy communities to agricultural land supporting annual cultivated crops.

As human populations grew, areas of land transformed to croplands to feed societies increased and the areas occupied by native grasslands consequently decreased, even noting that in some areas forest clearing had led to localised increases of semi-natural grasslands. It is thought that approximately 40% of temperate grassy communities have now been converted to cropland or other forms of intensive agriculture. In some countries and regions, the degree of loss is much higher. Indeed, in Australia, temperate grasslands have been reduced to less than 1% of their once extensive range (Kirkpatrick et al., 1995), while in the United States, tall-grass prairies have been reduced to only 3% of their previous cover (Samson et al., 1998); likewise, in Europe, Schutyser and Condé (2009) reported continuing and substantial decreases in grasslands of ~260,000 ha between 1990 and 2000. And while in some parts of Europe grasslands still occupy substantial areas of the landscape, overall their quality has declined and more than 75% have been classified in unfavourable conservation status' condition (Silva et al., 2008). This situation of loss, degradation and fragmentation of grassland communities has been further exacerbated by the cessation of traditional cultural management practices, ongoing clearing for the development of towns and cities, agricultural expansion, conversion to forestry, invasive species, climate change and other anthropogenic factors (White et al., 2000; Gibson, 2009; Valko et al., 2016; Torok et al., 2021).

Grassy community loss is a tragic occurrence on many levels. First, such communities represent a vast number of endemic plant species, and loss will mean many unique forms of biodiversity will be lost (White et al., 2000). Not only are they floristically and functionally diverse, but they also provide habitat and resources for a vast number of organisms from different trophic levels, including animals, birds, fungi and bacteria. Together, these form intricate webs of existence that should be valued and preserved for their own sake. Beyond these attributes, grassy communities provide an array of ecosystem services which continue to benefit humans. These include the provision of food and raw materials (Valko et al., 2016), regulating erosion and soil loss (Cain & Lovejoy, 2004), improving water quality by reducing nitrogen and phosphorus run off (Vilsack, 2016), reducing flood risks (Johnson et al., 2016), sequestering of soil carbon (Gebhart et al., 1994, Yang et al., 2019), improving pollinator services for agriculture (Kremen & M'Gonigle, 2015; White et al., 2017, McMinn-Sauder et al., 2020) and improving air quality (Johnson et al., 2016). Yet, despite these many attributes, the formal protection of grassy communities across the globe is extremely poor, with percentages of protected areas ranging from as little as 1% to 3% in grassland regions (Henwood, 2010), highlighting again the urgent need for their restoration.

Grassy Community Restoration

Given the degree of loss of grassland across the world, it is important that where these native grassy communities are still present, their conservation, protection and maintenance should be among key environmental goals for societies and governments. However, given that in many countries there are few areas of native grassy community left to retain, their restoration through active or passive means must become a key environmental goal. Over recent times, ambitious global restoration targets to remediate the impacts of human-induced land degradation (in the order of 350 million hectares by 2030) have been set under programs such as the Bonn Challenge (https://www.bonnchallenge.org) and the United Nations Decade on Ecosystem Restoration (About the UN Decade| UN Decade on Restoration). These goals should naturally include targets related to the restoration of grassy communities since these always represent an affected vegetation type (Dudley et al., 2020; Tolgyesi et al., 2022).

Common goals for grassy community restoration are to increase their extent, range, quality and connectedness and to reinstate as high a proportion as possible of native species and functional diversity representing a desired or reference community (Prober & Thiele, 2005; Gibson-Roy & Delpratt, 2015; Valko et al., 2016; Barr et al., 2017). In this respect, the extent of species diversity attained in restorations often can depend upon the degree to which appropriate species are either available in the broader landscape as collectable seed, retained in soil seed or bud banks, or available as colonizing propagules (Price et al., 2021; Gibson-Roy, 2022). The relative contribution from each may depend on the restoration strategy used, such as restoration through reintroduction (seeding or planting), or restoration through assisted natural regeneration (spontaneous recovery), or combinations of both these approaches.

Where diverse and plentiful native seed/bud banks exist and/or there are local colonizing sources remaining near a restoration site, the probability of spontaneous natural recovery ranges from likely in temperate grasslands (Valko et al., 2016) to remaining unlikely in tropical edaphic grasslands due to dispersal limitation, low seed quality and slow seedling growth (Nerlekar & Veldman, 2020). And while the process of natural recovery is typically slow and long term (Partel et al., 2005), it can be promoted and accelerated by management interventions that mimic natural disturbances such as stock grazing, controlled burns, hay baling or the careful use of herbicides (even in tropical settings). These treatments can limit the buildup of biomass and/or litter, thereby creating suitable niches for seedling recruitment and/or restricting competition from mature plants (natives and exotics) for emerging or colonizing natives (McDonald, 2000; Prober et al., 2005). Conversely, in highly disturbed locations such as agricultural landscapes or along transport corridors, where land clearing, long-term cultivation, ongoing herbicide application or other forms of disturbance mean that native vegetation has been absent for long periods and soil-borne or colonizing propagule sources are largely depleted, here restoration through species reintroduction by direct seeding, seed hay or plantings is typically required (Barr et al., 2017; Gibson-Roy & Delpratt, 2015; Kiss et al., 2021). Encouragingly, both regenerative and reintroduction approaches have been shown to be effective in restoring grassy communities when used to their best effect (Valko et al., 2016).

There are several key factors that act as constraints to restoration. Foremost among them is gaining access to native seed in the quantity, quality, diversity and timing required (Gibson-Roy & Delpratt 2013; Delpratt & Gibson-Roy, 2015; Ladouceur et al., 2018; León-Lobos et al. 2018; Pedrini et al., 2020; Torok et al., 2021; Zinnen et al., 2021). In many countries and regions seed is limited due to the rarity of grassy communities themselves, whilst in other situations, grassy communities are still present to some degrees, but there are poor levels of training and workforce capacity that limit the effectiveness, quality and quantity of wild collections (Peppin et al., 2010; de Urzedo et al., 2019; Tangren & Toth, 2020; Gibson-Roy et al., 2021a, b). In some countries, such as North America and parts of Europe, well developed and structured markets for restoration mean that while rarity and access to remaining wild communities may still constrain supply of this vital resource, the value of the market has meant that it has been supported and supplemented by cultivated seed production. (De Vitis et al., 2017; Gibson-Roy, 2018 – see Seed Chapter X). In many other countries, the size of restoration markets remains small and so does not support well-developed seed production capacity, leaving seed supply as a major threat to project success (Hancock et al., 2020; Schmidt et al., 2019a).

Apart from the restrictions imposed by seed supply issues, other factors can and do constrain outcomes. Many can change spatially and temporally depending on the site, region or country and include excessive weed loads (soil stored and standing vegetation), unsuitable nutrient settings (because of prior agricultural practices), limitations of training and/or workforce capacity and poorly developed restoration policies or markets (De Vitis et al., 2017; Gibson Roy et al., 2021a, b; Cortina-Segarra et al., 2021). However, despite this, continued advances in knowledge gained from research and practice give a clearer understanding of the pathways towards successful grassy community restoration (Torok et al., 2021).

The following Case Studies give examples of grassy community restoration and allow us to consider work completed in recent decades. Each gives a picture of how restoration has developed and what it has been achieved over this period - including successes and persisting challenges. These projects highlight work done to restore grassy community types in different parts of the world and so allow us to expand on the approaches, philosophies and techniques used by those involved.

Case Study 1: Grassy Community Restoration in the United States[1]

Project Rationale(s) and Strategy(ies)

The Platte River Prairie Restoration project (PRPRP) is in the region between Grand Island and Kearney in Nebraska and was overseen by the Nature Conservancy USA. The main rationale for this project's instigation was based on the Conservancy's desire to assist in the conservation of America's once extensive native prairies (Fig. 2.1). To help achieve this aim, the PRPRP worked to restore native prairies located in ex-agricultural landscapes where remnant prairies are today severely fragmented and degraded. The project's goal was to convert several thousand acres of Conservancy-owned former crop fields back to species-rich native prairie habitat (and in some areas, to wetland) and in doing so to also reconnect various small and isolated remnant areas of native prairie. It was anticipated that successfully achieving this aim would have the effect of increasing the vigour and extent of these areas and thereby provide higher quality habitat for faunal species impacted by the continued loss of native prairies. A secondary aim of the project was to promote and disseminate the techniques and approaches which were specifically developed to

Fig. 2.1 Restored Sandhill Prairie with Large Beardtongue (*Penstemon grandiflorus*) in foreground

achieve these outcomes so that they might be taken up and used by others to achieve comparable results across various regions and states.

Major Project Concerns and Barriers

One of the prime constraints faced by those undertaking this type of work in the United States, related to strong cultural resistance from people living and working in rural regions to the belief that restoration takes land out of 'productive use'. Many rural Americans feel a 'moral obligation' of land ownership that land should be made useful to its full productive extent, and for the farming community, this meant the land should produce an economic or agronomic output, such as crops or livestock. Restoring productive farmland to native prairie was at odds with this objective, because there was no visible economic or agronomic output from this action. Because of this widespread view, there is limited interest in restoring prairie outside of land owned and managed by conservation organizations and 'recreational land' owned by people interested in outdoor recreation more than agriculture.

A second local issue, in addition to widespread cultural resistance, was the issue of farmland taxes. In the State of Nebraska, these taxes are based on the government-assessed value of the land's highest productive potential. If, under this system, a landholder is not able to reach reasonable productive or economic potential from their land, taxes are fixed to its 'assessed' potential. Beyond the obvious implications to farm livelihoods of such a situation, this land tax model creates serious negative outcomes for anyone wanting to turn farmland back to native vegetation, given there is no productive output from doing so and that land would thus continue to attract land tax at its assessed productive potential.

Beyond the cultural resistance and tax implications, there were other obstacles to be faced. For example, at the time of the initiation of this project, many among those in the conservation sector did not believe that resilient species-rich prairies could be recreated by direct seeding ex-crop fields with wild native seed mixtures. Indeed, these views were in some ways well-founded. For instance, securing the seed resources required to undertake this type of high diversity restoration, with up to 200 species being involved was a major challenge because in the region (and the State of Nebraska more broadly), there was not a well-developed native seed industry (Oldfield, 2019). Even if that were not the case, the cost of buying native seed (which is much higher than comparable pasture species) in the quantities required to restore thousands of acres of land would have been prohibitive for an environmental NGO. This meant all seeds had to be sourced locally from wild remnants, which created difficulties such as where the seeds and extant species could be located, in addition to the task of locating appropriate people and training them in the techniques required to harvest and process these seeds efficiently and effectively.

Accessing funds to undertake such an ambitious restoration program was also a considerable and ongoing project challenge. Beyond whatever resources, the Conservancy itself could garner through donations; project managers worked with various government agencies that ran conservation-focussed programs to identify any potential funding sources. Interestingly, the Conservation Reserve Program (CRP), which is the most established operational US farm environmental support program, was not a prime funding source for the PRPRP. This was because the CRP set a limit as to how much land a single landholder could hold to be eligible to enter their program, and because the Conservancy owned several properties across the country, it had exceeded the limit. However, project managers were able to identify other government programs that were able to provide suitable pathways that subsidized project costs.

Another seed-related issue arose once seeding itself came into focus. This centred on the nature of actual restoration seeding rates of native grasses. Whilst this would seem to be straightforward, because much of the project's work was linked to government-funded programs, this meant having to align project seeding rates with their established program rules and stipulations. Experience with on-farm seeding programs showed that most projects sought to rapidly establish native grasses as grazing pasture or for weed control, not, as in our case, for recreating diverse communities. This meant that government regulations were framed on the assumption that grasses would be drill seeded; therefore, they set high seeding rates. However, as the PRPRP's goal was to establish a complex species mix of grasses and forbs and that they intended to surface broadcast the seed because many species were not suitable for use with drill seeders, the regulations stipulated even higher seeding rates since it was assumed that this technique would be less effective. This regulatory approach would have created serious issues for seed supply, in terms of both quantities and costs. To manage this situation, the project coordinators worked closely with funding authorities to explain the approach and to show why relatively low seeding rates and the broadcasting of diverse seed mixes would be appropriate and successful. This friction between rapid grass introduction versus high diversity restoration remains an issue in the sector, and while the PRPRP has made progress with changing attitudes toward restoration practice in their area of influence, in many other regions and states, it is still the case that seeding grasses remains the main goal of native species establishment and programs continue to resist requests to support high diversity restoration.

The project also faced technical issues, especially relating to equipment, given there was almost no off-the-shelf, purpose-built machinery for restoring native prairies. This meant that important implements such as harvesters, seed cleaners and seeders all had to be adapted or modified from agricultural machinery. This also presented issues with high costs for equipment purchase and around finding people suitably trained in their use and maintenance. On this point, tight budgets meant that project managers had limited staff numbers. Very few were full-time, and most were students employed as seasonal technicians. This meant working with an enthusiastic but transient and untrained volunteer work force with high annual turnover. Volunteers certainly gained a unique set of experiences, opportunities, and knowledge that they took away when they left, but it also meant it was difficult for project managers to develop a long-term local work force with skills and knowledge around plant identification, seed harvesting and handling, seeding or site management. This is still an on-going problem in this area.

Key Project Features

Limitations and constraints aside, there were several features that defined the essence of this project. Foremost among them was its ongoing and faithful focus on restoring species-rich prairie. Regarding this point, the project received critical early support from Bill Whitney and his Prairie Plains Resource Institute, which had pioneered early prairie restoration methods in Nebraska during 1980s. Whitney helped to guide the PRPRP during its formative planning stages in the mid and late 1990s and gave valuable advice and perspectives to the project managers.

As indicated earlier, the long-term goal for restoring large-scale areas of prairie habitat required significant quantities and varieties of native seeds, and this needed to be sourced from the local region. Knowledge of local remnant areas was built up over time, and seed and propagule collections were planned and undertaken on an annual basis. Most of the seed harvesting was done by hand, with mechanical strippers or combines used to harvest key grass and forb species where possible (Fig. 2.2a–d). Collectors aimed to source and harvest as many species as possible,

Fig. 2.2 (**a**) Hand harvest (*Top left*); (**b**) combine harvest (*Top right*); (**c**) brush harvest (*Bottom left*); (**d**) seed mixture (*Bottom right*)

and estimates showed that there were over 200 species eventually used in restoration seed mixes. Even with limited staff and predominantly hand harvest, the project was able to secure enough seed to sow up to two hundred acres per year. In terms of seed handling and processing, a simple and straightforward approach was adopted by project managers. Seed was only cleaned to a basic level, focusing mainly on breaking seeds apart from each other and removing them from pods/stems so that they could be effectively spread when broadcast. Collections were also run through a hammermill cleaner where various screens separated the bulk of seed from chaff, after which it was stored in sacks or open buckets until sowings.

Each year, different farm locations were identified and prepared for seeding. Ideally, these fields had been under crops for up to a decade, which meant long-term herbicide programs controlling crop weeds had depleted invasive species' seed banks. Stubble from the most recent crop was mechanically hoed back into the soil, ensuring there was a levelled and workable bed. Because soils in the project area were primarily sandy loams, after stubble incorporation, no further harrowing or soil preparation was required. Seeding was then undertaken in the fall or winter. Seed was sown as a high diversity mixture onto the bare surface or onto a lightly snow-covered soil (which helped draw the seed back into the soil as the snow thawed and melted), by an EZ-Flow drop spreader or by hand (Fig. 2.3a).

Following sowings and during the early years of establishment, little was done to the sown fields other than to monitor and control any problematic weeds that might become issues. Of main concern were tree regrowth or tree colonisation, particularly of cottonwood (*Populus deltoides*) (Fig. 2.4a) and siberian elm (*Ulmus pumila*), and/or the establishment of dominant exotic perennial grasses, such as smooth brome grass (*Bromus inermis*), reed canary grass (*Phalaris arundinacea*) and Kentucky bluegrass (*Poa pratensis*). Fortunately, annual weeds were not a large issue over extended time periods, as the sown perennial prairies became established and competed more strongly for resources. Once the prairie species were established, other than for periodic weed control (Fig. 2.4b), management primarily focussed on restricting grass biomass by prescribed burning and strategic grazing to preserve forb diversity. Importantly, these older restored prairies became important seed resources for future restoration.

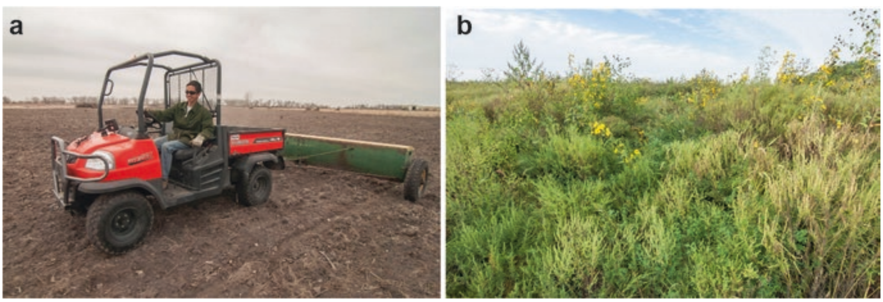

Fig. 2.3 (**a**) Broadcast seeding (*Right*); (**b**) early successional weeds (*Left*)

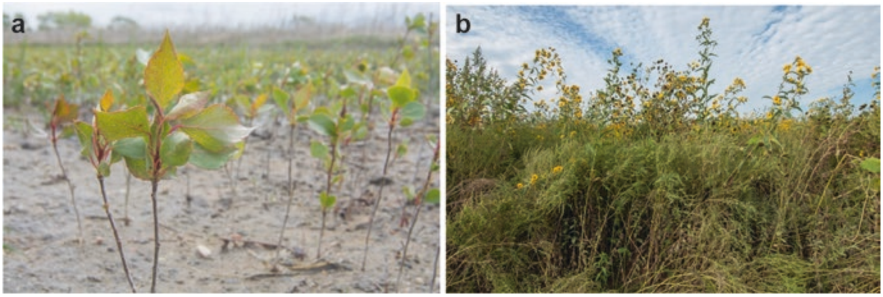

Fig. 2.4 (**a**) Cotton Wood seedlings (*Left*); (**b**) Herbaceous weeds (*Right*)

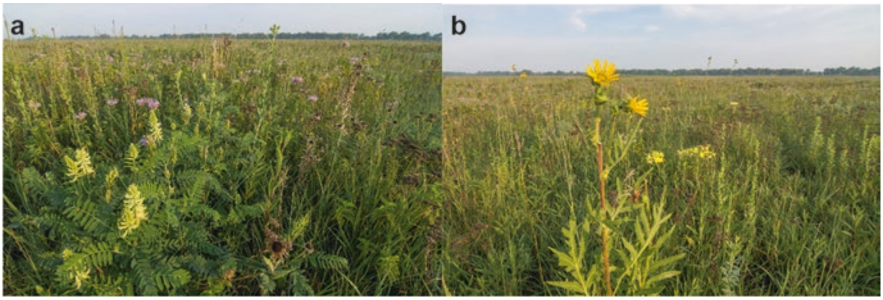

Fig. 2.5 (**a** and **b**) Mature native-dominated restorations

Major Project Outcomes

The PRPRP has been a great success. Since the 1990s it has restored more than 1500 acres of rare native prairie in the Platte River region of Nebraska, where it was once extensive (Fig. 2.5a, b). Importantly, this has been done using large numbers of native species, ensuring that the restorations are functionally complex and resilient. Together they have created a chain of prairies and wetlands representing a corridor of native habitat in an otherwise biologically depleted agricultural landscape. These restored prairies have also become 'working laboratories' for continued development of innovative techniques and knowledge related to prairie restoration and management. Most importantly, they are full of life and include birds, mammals, reptiles and amphibians, invertebrates and plants with animals moving from local fragmented remnants into and through these restored prairie lands.

Remarkably, ongoing monitoring and assessment have shown that almost every species used in sowings has become established, although experience has shown that some have proved more difficult to introduce or maintain than others. Perhaps even more importantly, monitoring has shown that these restored plant communities, established and managed over as long as 20 years, have maintained their ecological integrity and resilience. This is an important outcome given the unpredictable future faced given the likelihood of major climate change. Using a combination of approaches, including field days, site visitations, tours for government program managers and social media, project staff have tried to communicate to the broader conservation sector that it is possible to restore prairie landscapes using relatively simple techniques around seed harvest, handling, seeding and management. In doing so they hope to influence prevailing beliefs about the feasibility of prairie restoration locally, across Nebraska and at a national level.

What About the Project Worked, What Did Not Work and Why?

Despite these many successes, adoption of high diversity prairie restoration is still relatively uncommon across the United States. There are likely to be many contributing factors, but prime among them remain cultural resistance (i.e., productive land) and high financial costs (i.e., expense). It remains too expensive for most landholders or farmers to undertake such complex restoration on a larges cale, and government programs to date are not yet able to provide sufficient funding to prompt broader uptake. This is not to diminish the importance of agri-environmental programs such as the Conservation Reserve Program the Grassland Reserve Program, the Wetland Reserve Program or the Monarch Butterfly Program, each of which go some way to encouraging and incentivizing landholders to restore native vegetation on parts of their lands and thereby promoting growth and capacity of the seed and restoration sector more broadly. However, most of these initiatives do not result in permanent protection of restored landscapes, leading some to question the investment of public funds in these transient projects.

In the United States, there has been a long history of farm support programs, even in the case of very marginal lands. This could mean that governments and society now need to seriously consider alternative decisions about how and where farm support monies could or should be spent. It is possible that instead of programs trying to support increased farm productivity in very marginal landscapes, they might be better utilized by financially supporting these farmers to restore and manage, into perpetuity those marginal lands back to their native habitat. This will increase the ecological integrity and biological diversity of such fringe landscapes. Studies of restored Conservation Reserve Program lands have shown that there are many flow-on benefits to farmers from restoring native vegetation. These include improved water quality, reduced soil loss, carbon capture and potential alternative income streams. For these practices to be seriously and widely adopted (in marginal or other landscapes), governments would also have to significantly revise their approaches and metrics to collecting farmland tax.

It is also important to recognise that the goal of the PRPRP was not to reverse historical agricultural progress or to convert a sizeable percentage of currently cropped lands back to native prairie. Rather, it aimed to demonstrate that prairies can be returned to strategic parts of agricultural landscapes to increase native biodiversity and natural beauty, whilst at the same time providing other important ecosystem services. For over two decades now, the PRPRP has worked towards restoring up to 200 acres of prairie per year, engaging with local communities, training future generations of conservationists and spreading a message of hope that these activities can be replicated elsewhere. To that end, the Conservancy has initiated similar projects in other states and hopes that one day works of this nature are commonplace across the United States.

Case Study 2: Grassy Community Restoration in Australia[2]

Project Rationale(s) and Strategy(ies)

Prior to European settlement in the continent of Australia, temperate native grassy landscapes were managed and maintained by indigenous peoples (Gott et al., 2015). With the arrival of European settlers, there was a sharp cessation of indigenous cultural and management practices and the introduction of Northern hemisphere-based agricultural approaches, which, together, had a dramatic and disruptive impact on native grassy communities (Williams & Morgan, 2015). During the 1970s, 1980s and 1990s, there was much focus turned towards grassland conservation, and this was supported through legislative protection and regulation from Federal and State governments. However, despite this signal, agriculturally linked factors (in addition to others associated with human activities) continued to degrade native grasslands and grassy woodlands, leaving them among Australia's most threatened communities (Fig. 2.6a, b, Kirkpatrick et al., 1995).

The Grassy Groundcover Restoration Project (GGRP) was initiated to respond to this alarming situation at a time when there was little confidence from conservationists, ecologists or researchers that native grasslands and grassy woodlands could be reinstated by restoration. The project began as a collaboration between the University of Melbourne and Greening Australia (an environmental non-government organisation), and its underpinning centred on promising recently completed doctoral studies focused on the feasibility of grassland restoration (Gibson-Roy, 2004). The project aimed to further explore and develop findings from these and other early

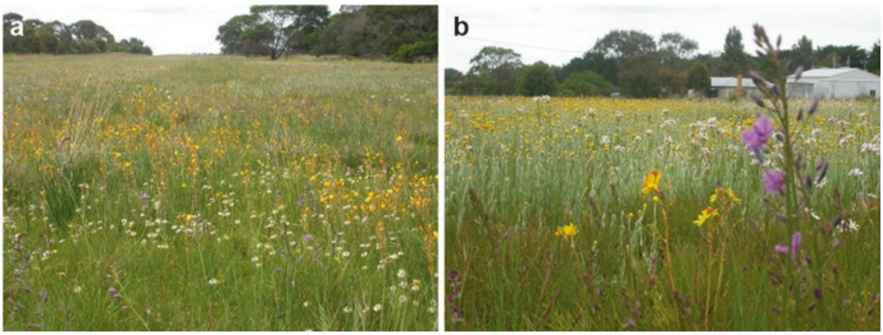

Fig. 2.6 (**a** & **b**) High-quality remnant grassy communities

studies under 'real world' conditions (Gibson-Roy, 2005). The project began in 2004 and continued in various forms until 2019, eventually leaving a legacy of grassy restoration sites across Southeastern Australia and inspiring other groups to take up these learnings and practices (Gibson-Roy, 2022).

Major Project Concerns and Barriers

Sadly, the early 2000s saw little appetite or formal support from government or their agencies for undertaking grassy restoration. Indeed, following the Federal government's 1989 commitment to replant one billion trees nationally, most restoration programs were almost totally focused on the woody strata. Also, the extreme degradation faced by grassy communities meant that many conservationists questioned activities that might further negatively impact on them, including seed-based restoration, which was viewed as a well-intentioned but inappropriate use of critically rare native seed. Under these settings, grassy community research and endeavour tended to focus on their ecology and management rather than on their restoration. Assessments of the few small-scale restorations undertaken, which were typically by hand plantings, showed only limited success due to harsh climatic conditions and herbivore impacts, but most commonly due to weed competition (Berkeley & Cross, 1986; Scarlett & Parsons, 1992; Shears, 1998; Delpratt, 1999; Morgan, 1999; Gibson-Roy, 2000; Smallbone et al., 2007) (Fig. 2.7).

Key Project Features

Experimentation

Over its life, the GGRP maintained a focus on the restoration of native grassy communities in the context of disturbed landscapes such as ex-agricultural land, urban development areas and transport corridors. Core elements included (i) a focus on

Fig. 2.7 (**a** & **b**) Woody tree and shrub plantings typical of government-funded programs

high diversity restoration; (ii) the management of elevated nutrient levels; (iii) the manipulation of weed-dominated soil seed banks; (iv) the refinement of seed production, seed harvesting, seed processing and direct seeding technologies and techniques; and (v) post-establishment management.

The early years of the project were strongly focused on experimentation and capacity development (Gibson-Roy, 2005, 2010, 2012, 2013; Gibson-Roy et al., 2010a, b; Taylor et al., 2013; Gibson-Roy, 2014a; Gibson-Roy et al., 2014), whilst in the latter years, this moved to applying or refining initial learnings on a larger scale or under different conditions and settings (Gibson-Roy, 2014b; Gibson-Roy & Denham, 2014; Gibson-Roy & McDonald, 2014; Delpratt & Gibson-Roy, 2015; Gibson-Roy & Delpratt, 2015; White et al., 2017; Morris & Gibson-Roy, 2018; Cuneo et al., 2018; Morris & Gibson-Roy, 2019a, b; Schmidt et al. 2020).

For the early experimental phase of the project, a steering committee was established to facilitate good governance and a technical panel to advise on the design and undertaking of experiments. To promote the project and its aims and to garner interest from landholders prepared to host one-hectare experimental sites, public presentations were held across the central and southwestern parts of Victoria. Landholder interest was overwhelming, and 11 locations (from over 50 offered) representing a wide range of land tenures, including farms, roadsides and public reserves, were chosen. These landholders agreed to preserve and manage restored sites in consultation with the project managers and to provide access for long-term monitoring, which was estimated to be 10 plus years.

Experimental treatments were developed under the guidance of the technical panel and applied across these 11 sites. These aimed to address the issues of excessive nutrification and weed-dominated seed banks by either exhausting or physically removing weeds and nutrient loads. ‘Exhaustion’ plots were treated with either 1, 2 or 3 years of fallowing by herbicide treatment (four per year), with each application preceded by shallow cultivation to stimulate weed banks before spraying. These were compared to ‘removal’ plots, which were treated by topsoil removal through mechanical stripping to a depth of 10 cm (Fig. 2.8).

Fig. 2.8 Topsoil manipulation. (**a**) small-scale excavator (*Left*); (**b**) large-scale grader and scraper (*Right*)

Seed Resourcing – Collection

Due to a lack of markets for their restoration, commercial seed supplies were largely unavailable. To meet this fundamental requirement, seeds were initially sourced through field collections from remnant areas and later supplemented through cultivated seed production techniques. Not surprisingly, locating remnant populations in highly fragmented landscapes was a great challenge, and collection zones were defined in relation to each sowing site, taking into consideration their current and historical distributions and elements of past connectivity between species and local populations. The aim of collection boundaries was to minimize the risk of creating inbreeding populations (in both restored and seed production sites), increase the diversity of species and the amounts of seed available for restoration and improve the adaptive potential within the restored communities and to preserve regional identities (Broadhurst et al., 2008; Bischoff et al., 2010).

Within seed zones, collectors aimed to match source population and sowing site conditions based on soil type and topography (Cole et al., 1999). Most of the seeds were collected within 50 km of each seeding site, with collectors targeting all relevant species which had been located and which were producing seeds. Hand collections aimed to take seeds from 50+ plants per species per population and to avoid conscious or unconscious selection to reduce the potential for relatedness. For mechanical collections (e.g., brush harvesters) population size routinely exceeded 10,000 individuals. Seeds were harvested from multiple source populations within a collection zone over the entire span of a ripening season. Project staff worked closely with seed collectors and seed production area (SPA) growers across all regions to improve their species recognition, and harvesting and processing skills and processing skills using regular project forums, workshops and technical newsletters. Ultimately the project established a highly proficient and dedicated group of collectors and growers across the regions in which it operated (Fig. 2.9).

Seed Resourcing – Seed Production

Wild collections only provided enough seed for a limited range of species (primarily grasses). Therefore, the project began to cultivate species in seed production areas (SPAs) to supplement wild sources. SPAs established early in the project were each linked to a growing region and provided seeds for one or more restoration sites (up to three). SPAs were usually set up in association with local native plant nurseries and grew plants in simple containerized settings (typically foam boxes) as high-density irrigated crops where competition from weeds and herbivores was minimized. Over time, in the latter stages of the project, these approaches were refined, modified and expanded and a smaller number of SPAs were developed at larger scales to service the seed needs of bigger restorations using more advanced growing systems that included weed-mat-covered in-ground beds, open field beds, raised covered beds and vertical trellis beds (Fig. 2.10).

Fig. 2.9 (**a**) Roadside grassy remnant (*Top left*); (**b**) brush harvested seed (*Top right*); (**c** & **d**) teams hand harvesting (*Bottom left* and *right*)

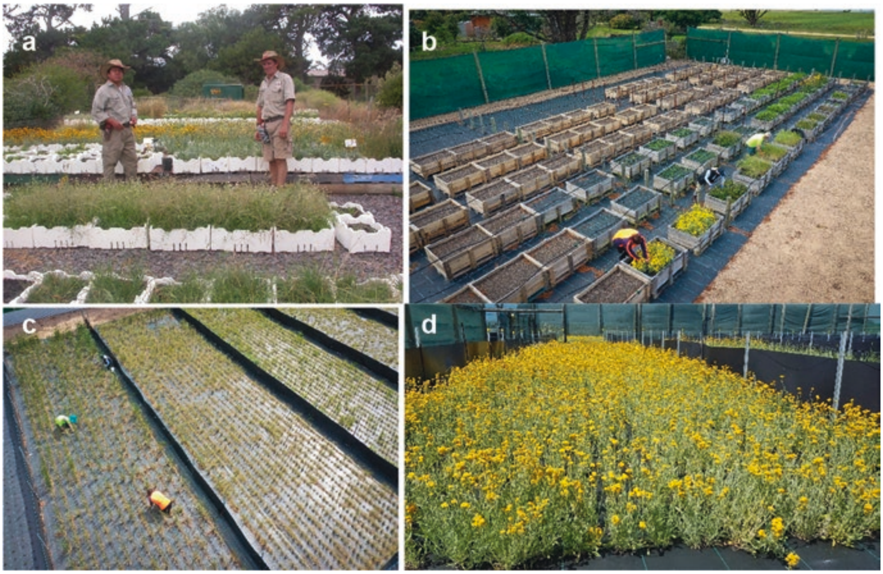

Fig. 2.10 (**a** & **b**) Containerised production systems (*Top left* and *right*); (**c** & **d**) weed mat production systems (*Bottom left* and *right*)

SPAs proved critical to the project's success. Through these facilities, large quantities of high-quality seed from numerous species indigenous to the restoration region were produced. Furthermore, these SPAs relied on only small amounts of wild-collected seeds to establish crops, which reduced impacts on remnant communities. Importantly, while collection protocols for remnant populations aimed to capture a broad range of genetic traits, similar protocols were established to increase the likelihood that these traits would be preserved through the seed production phase. In practice, this meant appropriate mixing and sub-sampling of wild seedlots when propagating production crops, avoidance of selection bias when pricking-out seedlings for bed plantings and harvesting seed from production crops over the whole fruiting period. SPA populations for a given species also contained as many individuals as possible (given space and resource considerations) but were typically composed of between hundreds and several thousands of individuals. In many cases, this meant that SPA crops were actually much larger than the source population/s. To further lessen the potential for genetic bottlenecks, crops were maintained for only two harvest seasons before new genetic material was introduced from wild populations.

Many hundreds of species were grown in seed production areas by the project over its life. Most species were readily propagated from seed and suited to some form of cultivated production system. Seed production enabled the project to grow sowable quantities of seed from many threatened species that would otherwise not have been available for use in restoration (Gibson-Roy & Carland, 2023; Gibson-Roy, 2010). These ex-situ populations of rare species also afforded them some protection from localized extinction where in situ populations may have been further impacted or even destroyed by some form of human disturbance. Another key feature of SPAs was that they represented large collections of species growing in centralised locations as weed-free monocultures, and this dramatically simplified collection in comparison to wild harvest. No longer did collectors have to spend long months travelling large distances, often in harsh conditions to locate and harvest seeds. SPA crops were maintained for ease of harvest and produced more reliable quantities of seed at times when source populations were often severely impacted by harsh climatic conditions such as drought, storms and other events, including uncontrolled fires, grazing and predation. In addition, most species cultivated in SPAs produced seeds over much extended periods in comparison to those in the wild.

Seed Resourcing – Seed Quality Characterisation

Seeds used in sowings (wild or production) were assessed for quality characteristics using purity and germination tests. Seed mixtures were also sampled at the time of sowing and germinated under nursery conditions, and where possible, in germination cabinets to gauge germination and emergence potential at the time of sowing (Gibson-Roy et al., 2010a). These approaches enabled important understanding of the seed's characteristics, both post-harvest and at the time of sowing, making it

Fig. 2.11 (**a**) SPA seed drying (*Top left*); (**b**) seed mixing (*Top right*); (**c**) seed testing under temperature- and light-controlled conditions (*Bottom left*); (**d**) seed storage under temperature 15 C and 15% relative humidity conditions (*Bottom right*)

possible to reliably consider seed quality in the post-analysis of field emergence patterns rather than attributing good or poor field outcomes solely to post-sowing factors such as soils, rainfall, temperature or predation (Fig. 2.11).

Site Preparation and Seeding

GGRP sites had various land use histories. Therefore, at each, soil testing was undertaken to determine the key soil characteristics, including soil texture, colour, pH, nitrogen, phosphorus and electrical conductivity. Some sites were located on cropping land, and these exhibited elevated nutrient levels and, because of long cultivation histories, had deep weed-dominated soil seed banks. Others had a history of pasture grazing, with lower fertilization and minimal cultivation, which left shallower weed seed banks and less nutrified soils. A small number was located on road verges where vegetation was typically dominated by colonising pasture grasses from nearby farms, together with broadleaf weeds. These had been exposed to regular soil and vegetation disturbance by road managers such as using slashers or graders or herbicide spray machinery, often meaning they were volatile in terms of weed loads and soil health. At all sites, native herbaceous species were largely absent or possibly represented by a few common species in very low numbers.

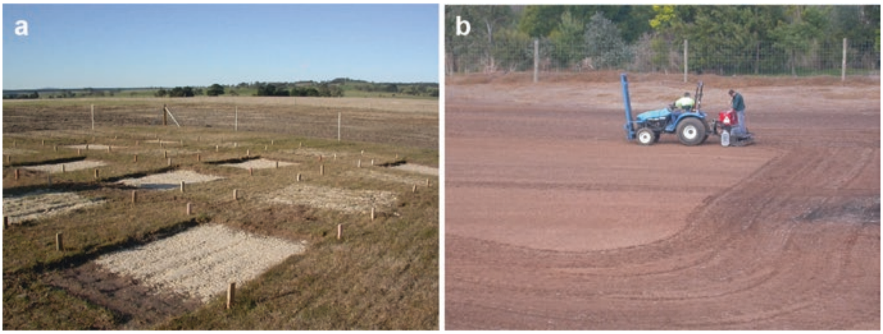

Fig. 2.12 (**a**) Small experimental plots following seeding (*Left*); (**b**) large-scale plots being mechanically seeded (*Right*)

In the first year of experimentation, treatments were applied to 2 × 2 m plots prior to being seeded by hand. As larger quantities of seed became available from SPAs in the following 2 years, plot size increased to 2000 m^{-2}. In later years (beyond the experimental phase), sites ranged in size up to 16 ha and almost exclusively used soil scalping, soil inversion or subsoil capping as the core nutrient and weed seed/bud bank treatment methods. All 2000 m^{-2} and larger plots were machine seeded using oscillating rotating tines to develop lightly tilled friable soils, while modifications to the seeding box allowed the seed mix (and sand carrier) to fall as a curtain onto the prepared seed bed with a mounted rake and roller lightly covering and pressing seed into the soil. Seed flow and tractor speed could be adjusted to achieve accurate sowing rates. This machine proved very effective on a broad range of soil types and conditions and for all native seeds used (ground layer, shrub or tree) (Fig. 2.12).

Sowing mixes contained up to 100 species. Grasses represented approximately 60–70% of seed mixes by mass, dominant forbs 10–15%, with sub-dominant forbs (or other functional groups) making up the remainder. These sub-dominant species represented the bulk of species diversity in the mix. Sowing rates varied from site to site and from year to year, being linked to seed availability, seed quality and restoration goals, but, in general, rates of between 40 to 50 kg per ha (representing pure seed and chaff) were used.

Major Project Outcomes

Approximately 230 species were established in the first 3 years of experimental sowings. These included 20 grass genera, 74 forb genera and 10 sub-shrub genera, showing that a wide range of ground layer species could be established by direct seeding (Gibson-Roy & Delpratt, 2015). A great many more species were used in later sowings across different regions of Victoria and into other States over the following decade and a half (e.g., Morris & Gibson-Roy, 2018; Cuneo et al., 2018).

In terms of experimental treatments, the most outstanding finding related to the differences in establishment success between soil stimulation plus herbicide-treated plots ($n = 96 - 2 \times 2$ m plots & $n = 22 - 2000$ m^{-2} plots) and soil-removed plots ($n = 96 - 2 \times 2$ m plots & $n = 22 - 2000$ m^{-2} plots). Here, monitoring revealed that species diversity, plant densities and structural composition were significantly higher or better on soil-removed plots compared to the long-term cultivation and herbicide-treated plots (Table 2.1). In many cases, this left them comparable in quality and composition with reference remnant communities, and importantly, these characteristics were largely maintained over the following decade and a half since (Gibson-Roy & Carland, 2023).

Underpinning this outcome was the effect of soil removal in restricting key nutrifying elements in topsoil-removed plots compared to non-removed. Soil testing of plots revealed that following soil removal, phosphorus levels declined to an average of 14 mg/kg, making them like those observed in reference communities (<20 mg/kg – Gibson-Roy et al., 2010a, b). Likewise, nitrogen levels were also reduced by half or more in comparison to non-soil removed plots. Analysis of vegetation data revealed a strong relationship between low P levels and higher native diversity and density, while low nitrogen levels corresponded to a reduced dominance of all grasses (exotic and native), which further aided sub-dominant native species' persistence (Fig. 2.13).

Conversely, diversity and structural complexity on cultivated and herbicide-treated plots was lower, regardless of the duration of treatment application (1–3 years). Whilst this treatment removed all standing vegetation at the time of sowings, monitoring revealed that large numbers of weeds continued to reemerge from soil seed and bud banks to compete with the sown natives, and because soil P and N remained at agronomic levels, these nutrients helped exotic species grow to a

Table 2.1 Comparisons of differences in mean values from measurements taken on topsoil removed and soil stimulation plus herbicide-treated plots located at 11 sites in western Victoria. Diversity = species number per plot; plant counts = plant number m^{-2}; vegetative cover = percentage cover m^{-2}

		Scalped	Non-scalped
Measure	Category	Mean	Mean
Diversity	Native	38	13
Plant counts	Total – native	58	24
	Native grass	48	22
	Native forb	10	2
	Total – weed	70	125
	Exotic grass	40	97
	Exotic forb	30	28
Vegetative cover	% native	56	34
	% weed	16	61
	% bare earth	29	5

Note: This dataset is derived from the monitoring of 2006 and 2007 sowings at five (2006 sowing) and four (2007 sowing) years post seeding. Treatments plots were all 2000 m^{-2} in size

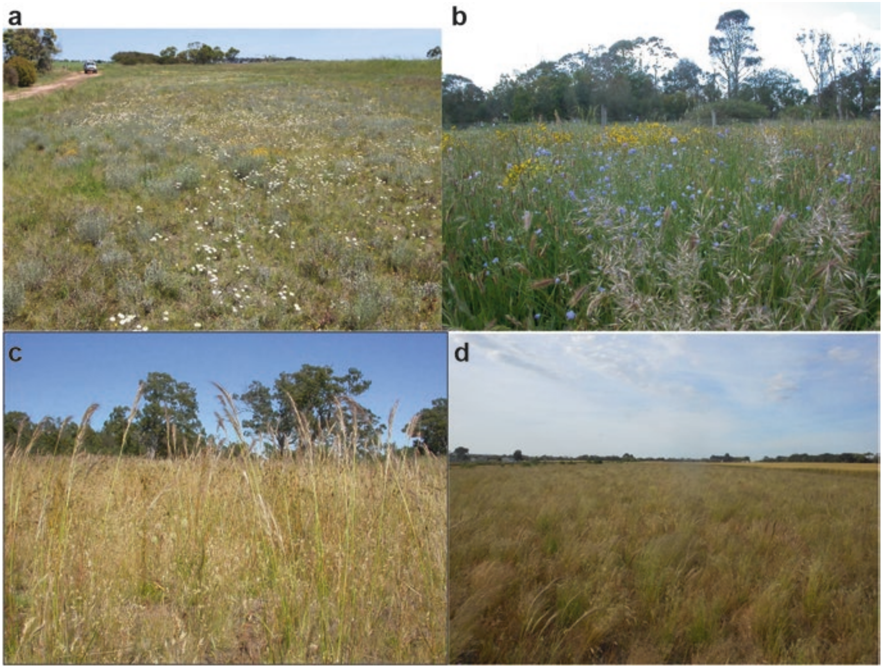

Fig. 2.13 (**a** & **b**) Seeded restorations on topsoil-removed sites. (**a** & **b**) wildflower rich restorations (*Top*). (**c** & **d**) grassy-dominated restorations (*Bottom*)

larger size or greater percentage vegetative cover than co-occurring natives. This finding clearly showed that weed-dominated agronomic soil seed banks were not exhausted despite long-term cultivation and herbicide treatment and continued to produce large numbers of emergent weeds which would compete vigorously with colonising, sown or planted natives. This provided a stark reminder of the degree of alteration that agriculture has had on soils and plant communities in these landscapes. It also highlighted another positive feature of soil removal which was that, as well as treating nutrient-laden layers, the process also removes weed seed and bud banks.

Among the species established by restorations were those that were locally, regionally and nationally threatened. Indeed, several GGRP restorations represented new populations of endangered species and some featured populations which greatly exceeded the size of wild source populations (Cuneo et al., 2018; Gibson-Roy, 2010; Gibson-Roy & Carland, 2023). This was made possible largely through the combination of nutrient and weed seed bank reductions, the utilization of seed production approaches to increase seed supply from these species for sowing and purpose-designed seeding equipment. Ongoing seedling recruitment from restored species was also verified by seedling emergence close to mature adults and from seedlings appearing in unsown areas such as adjoining walkways and large bare 'recruitment zones' left adjoining sowing areas. In later years monitoring

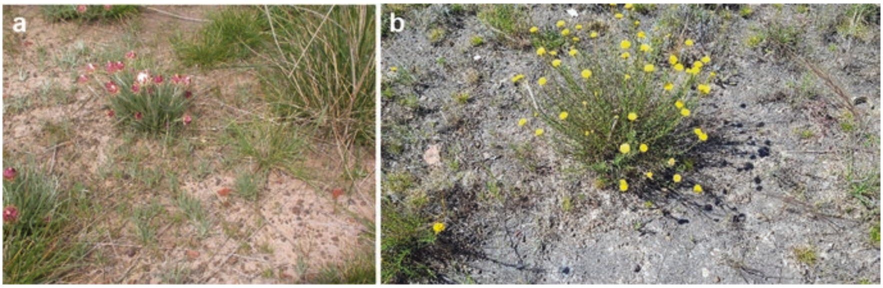

Fig. 2.14 Seeded listed threatened species established in restorations with recruiting seedlings surrounding mature adults; (**a**) hoary sunray (*Leucochrysum albicans* subsp. *albicans* var. tricolor) (*Left*); (**b**) button wrinklewort (*Rutidosis leptorrhynchoides*) (*Right*)

highlighted second, third and fourth generations of plants dispersed over considerable distances of up to several hundred metres from the initial planting (Gibson-Roy et al., 2010a; Gibson-Roy & Carland, 2023) (Fig. 2.14).

Another important feature of GGRP restorations was their high levels of colonization by other native animals and plants, indicating increasing functionality at multiple trophic levels. Insects, birds, mammals, amphibians and reptiles were routinely observed feeding, sheltering or nesting within restorations, where they had not been present before restoration (Gibson-Roy & Delpratt, 2015) and also within seed production areas (White et al., 2017, Schmidt et al., 2020). Native trees (eucalypts and acacias) commonly reappeared within restored areas where a nearby tree canopy provided a seed source. In other situations, unsown native ground layer species emerged from seed or bud banks or from colonisation from outside restored sites (especially where they adjoined remnants) Gibson-Roy & Carland (2023). Importantly, investigations of native plant roots from several restorations (scalped and non-scalped) as well as from reference areas also showed functioning arbuscular mycorrhiza at similar levels across all (Gibson-Roy et al., 2014) (Fig. 2.15).

What About the Project Worked, What Did Not Work and Why?

Experience from the GGRP over many years has highlighted (i) the need for good planning and goal setting; (ii) the importance of re-establishing complexity and function in restored communities; (iii) the value of the application of horticultural and agricultural principles, as well as ecological understandings in restorations; (iv) the success of seed production in addressing seed and species limitations; (v) the importance of the development and use of specialised restoration technology; (vi) the worth of embedding (where possible) experimentation within projects; (vii) the need to monitor and quantify outcomes to inform future practice and (viii) the value in purposeful involvement of stakeholders, communities and others in projects. The extended period over which the GGRP operated and the packaging of these key

Fig. 2.15 Examples of fauna colonising restored sites. (**a**) Native spider on grasses (*Top left*). (**b**) growling grass frog (*Litoria raniformis*) (*Top right*). (**c**) chequered copper butterfly (*Lucia limbaria*) (*Bottom left*); (**d**) little whip snake (*Suta flagellum*) (Bottom right)

points into a single project enabled it to clearly demonstrate the feasibility of grassland and grassy woodland restoration in disturbed landscapes.

The early experimental phase of GGRP was critical to its success. It was during this period that various replicated, field-applied experimental treatments were tested and where outcomes were monitored, evaluated and verified. These findings provided strong evidence for the efficacy of soil removal in treating elevated nutrient levels and weed seed and bud banks. These two factors are typically fundamental constraints to ensuring native ground layer species can establish and persist in restorations (Gibson Roy et al., 2010a, b). Based on this knowledge, the project confidently undertook many other larger restorations across several states (Cuneo et al., 2018; Morris & Gibson-Roy, 2019a, b; Gibson-Roy & Carland, 2023). An important application and expansion of these approaches occurred between 2013 and 2018, when after developing regional-scale seed production capacity, the GGRP undertook a series of restorations totalling nearly 50 ha in the urban matrix of Sydney, which is Australia's largest city. Mostly undertaken in western Sydney on the Cumberland Plain, the project restored nationally threatened grassy woodlands in parklands, council reserves and in national parks, where before, none had been successfully restored (Cuneo et al., 2018). This showed that after a decade of demonstrating the efficacy of these methods in rural landscapes, these same approaches

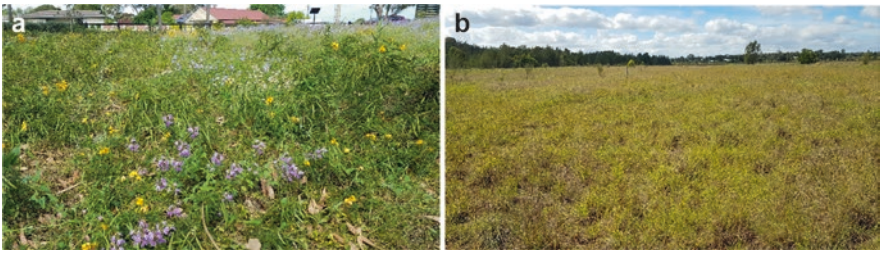

Fig. 2.16 Urban restorations. (**a**) wildflower-rich council reserve (*Left*); (**b**) grassy sward on public parkland (*Right*)

could be used to restore grassy communities in the urban context. In the intervening years, other groups, informed by these techniques and led by their own goals and motivations, have taken and, in many cases, further refined these approaches to implement successful grassy community restoration projects of their own (Gibson-Roy, 2022) (Fig. 2.16).

Globally, many people now recognise that full 'ecological reconstruction' is crucial to re-integrating native grassy communities into landscapes highly fragmented and degraded by agriculture, and this Case Study has demonstrated examples of pathways to success under Australian conditions. However, this Case Study does not represent the end of this quest. An emerging understanding of the Australian experience is that despite the best efforts of many committed people the prognosis for complex ecological restoration of grassy communities in Australia remains bleak (Gibson-Roy et al., 2021a, b; Gibson-Roy, 2022). Australian governments, and their agencies, together with the academics and researchers to whom they turn to as authorities, continue to overwhelmingly focus on conservation rather than a balance between conservation and restoration. This situation persists even though native grassy communities continue to be lost to agriculture and other forms of urban development. In parallel, the almost total bias of many decades towards tree and shrub plantings (primarily for carbon or functional outcomes) has also been maintained. Indeed, there are effectively no legislative or regulatory incentives for the uptake of complex grassy community restoration across arable landscapes even though the need has been identified (Mappin et al., 2021). Without strong and reliable markets for native seed and restoration services, restoration at a significant landscape-scale has not occurred, leaving the restoration sector small, incapacitated and dysfunctional. Despite clear and long-term evidence from the GGRP and other groups that complex restoration is indeed feasible and there being examples from other parts of the world showing governments can provide the right mix of policy and regulatory mandates to create an environment for seed and restoration markets to develop, for sector capacity to increase and for restoration at landscape scales to occur – in Australia, we remain steadfast in our reluctance to pursue such opportunities (Gibson-Roy, 2022).

On the positive side, Federal government parties of all political hues continue to support a national Environmental Protection Biodiversity Act, which in effect states that Australia's biodiversity should not be allowed to disappear. It is in this context that the GGRP, and the various small-scale projects that have since followed, have been able to provide a base of evidence, knowledge and experience which can and should be built upon and supported by governments and their agencies. Such a situation could see landholders, land managers and communities joining to restore complex native vegetation across vast areas of arable land and to long distances of road and rail corridors from which it has almost disappeared and where its restoration is most needed. If in the years and decades to come this does not occur, it will not be because it was not possible, but rather it will be because we were timid or too self-centred to ensure it did.

Case Study 3: Grassy Community Restoration in Belgium[3]

Project Rationale(s) and Strategy(ies)

As a result of a thousand years of mixed agro-pastoral practices, permanent grasslands on the European continent have complex structures and do not have easily identified natural states. For example, the grassland ecosystems of particular interest in this case study, in the southern half of Belgium (called Wallonia), are among the most species-rich vegetation types, sometimes exceeding 60 plant species per m^2 (Merunková et al., 2012), but these species are liable to represent many phases of invasion assisted by animals, wind, water and human seed transfer.

As a result, notwithstanding this rich vegetation environment, the question of the 'restoration' of these grasslands to their natural state is still somewhat contested. In this respect, in more recently colonised countries such as the United States or Australia, there are still remnant natural grassland communities which can act as benchmarks for restoration in many climatic and regional areas. In these instances, the term 'restoration' has a specific and criterion-based meaning. By comparison, in Europe, it has been estimated that about 20% of all European grasslands still have a so-called favourable conservation status within the meaning of Article 1 of the Habitat Directive (European Environment Agency, 2020),

Three conservation status categories can be assigned to European grasslands (A, B, C) (European Environment Agency, 2020). Recent assessments have shown that 76% are now in conditions that meet the 'unfavourable conservation status' category, indicating the extreme pressures these landscapes have been under. The conservation status 'goal' for our project restorations was to achieve an A (favourable) category. To this end, project managers sought to assist the recovery of degraded, damaged or destroyed grasslands at numerous sites and settings.

Furthermore, in Wallonia, only 5% of regional grasslands are currently in a favourable conservation status, and it is known that between 1955 and 2009, the area of permanent grasslands in Belgium was reduced by a third (Belgian Federal Government – https://statbel.fgov.be/). The major causes of reduction were

Fig. 2.17 (**a**) Low diversity degraded grassland (*Right*); (**b**) image showing grassland meadow divided into two plots 1 year after restoration actions have taken place (*Right*). Left section of this image shows restored plot area with forbs establishing while right image section shows unrestored plot area. Photo credits: (**a**) Maïké Dellicour; (**b**) Patrick Lighezzolo

urbanization, conversion to cropland, abandonment, and plantations of exotic species (particularly to Norway spruce, *Picea abies*). The remaining grassland areas in Wallonia were highly intensified, especially from 1955, through the use of chemical fertilizers, the earliness and frequency of successive mowing and from increases in livestock density. In addition to the gradual loss of species-rich grasslands, essential elements of the landscape, such as ponds, hedges and orchards, have also disappeared. This represents a major cause of local extinction of many animal species living in these environments, such as insects, birds, bats and amphibians (Fig. 2.17).

Today, ecosystem restoration is recognized as a priority (UNEP & FAO, 2020). A growing number of scientists are convinced that it is necessary to intervene by reintroducing or reinforcing plant communities for the purpose of nature conservation. This option was adopted by the partners of several ongoing LIFE projects co-financed by the European Commission and in particular the 'LIFE Bocage Meadows' project (LIFE11NAT/BE/001059), led by Natagora, a nature conservation NGO (Goret et al., 2020).

Major Project Concerns and Barriers

When trying to restore habitats, (i) it is difficult to decide the nature of the technique which will be chosen for implementation; (ii) budgets are often limited, while restoration is expensive; and (iii) it is often not easy to predict species recovery trajectories. We, therefore, wanted to improve the cost-efficiency of our restoration strategies and ensure that appropriate action plans were developed. For grasslands such as those defined by the European Union (see European Commission, 2013), information exists on the results of previous conservation efforts published through technical notes, detailed action plans and scholarly scientific articles. However, this information is somewhat scattered and provides only approximate indications of the target habitat criteria or the environmental conditions under which restoration has taken place. It was, therefore, sometimes difficult for us to know if, in our cases, it was relevant to apply a technique recommended in other (unknown) contexts. It became clear that any sort of tool proposing restoration measures adapted to each type of European grassland was sorely lacking before the implementation of our project. We, therefore, decided to develop a knowledge and literature-based decision tree to facilitate the adoption of the most appropriate restoration techniques for the Wallonia area (Goret et al., 2021).

Key Project Features

Within the framework of the 'LIFE Bocage Meadows' project, we restored around 200 ha of lowland hay meadows (*Arrhenatherion* community) between 2012 and 2020, following the methodology developed in our decision-making tool (Goret et al., 2021). The first step consisted in carrying out exhaustive floristic surveys in the meadows to be restored. These inventories made it possible to identify the phytosociological alliance of the meadows and to verify whether they were already in the *Arrhenatherion* target habitat. We subsequently determined their conservation status, according to the methodology outlined in the Habitats Notebook of the Department of Natural and Agricultural Environment Studies. This considered the presence and abundance of characteristic species in the area to be restored. Meadows to be restored are classified into three categories. Meadows in good conservation status are classified as A and thus do not require any further intervention. Medium-level conservation status is classified as B; and poor conservation status is classified as C. Finally, classification X refers to areas where target habitat is absent and can evidence bare soil after deforestation or degraded meadow of the *Cynosurion* type.

Regarding the techniques that were employed in this work, provided that a threshold of 5 mg of available phosphorus per 100 g dry soil (Janssens et al., 1998) was not exceeded and that the main threats to conservation such as fertilization and agricultural management have disappeared, we can summarize our approach as follows:

(i) If the conservation status of the habitat is C or X and it was not A or B less than 5 years ago (based on the average lifespan of the soil seed bank of characteristic species), then we carried out population reinforcement by two seeding techniques after having exposed 50% of the soil:

– **Sowing** of harvested seeds from source meadows in good conservation status as close as possible to the target meadow.
– **Green hay transfer** from source meadows in good conservation status as close as possible to the target meadow.

(ii) If the conservation status of the habitat is B (or it was A or B less than 5 years ago), we considered that the seed bank of species characteristic of the habitat was still present in the soil and that it was necessary to promote their germination and

Fig. 2.18 (**a**) Mechanical seed harvest (*Left*); (**b**) collected seed (*Right*). (Photo credits: (**a** & **b**): Xavier Janssens)

Fig. 2.19 Seed hay application. (Photo credit: Thibaut Goret)

the development of seedlings. After total cessation of fertilization, the restoration consisted of mowing twice a year, the first time from June 15 and again in September. This mowing regime allowed nitrogen to begin to be exported (starting a process that takes 10 to 15 years), and it enabled the plant cover to be as low as possible at the time of germination, namely between April and October. Grass that grows back after mowing can be treated again between August and October (Figs. 2.18 and 2.19).

In each case, post-restoration management had to be adapted and controlled for a number of years before being readjusted to recurrent management. This was judged to be appropriate when the conservation status of the area had improved significantly. After seeding, it is therefore essential to carry out two to three mowing sessions per year. In the following years, it is possible to move to two mowing per year, maintained until the meadow moves into a good conservation status and can therefore be managed by a single mowing per year in July (late mowing). To ensure the return of the entomofauna, it is essential in all cases to maintain sufficient refuge areas for insects, and we sequestered at least 10% of the surface on which we were working. Our sowing operations were carried out in September, which was the best period for seed germination of characteristic species and to promote the development of the seedlings. We deliberately avoided the summer droughts season, which can cause significant suffering to seedlings.

Major Project Outcomes

To measure the success of the project, we created a transition matrix showing the evolution of the conservation status of treated areas (Table 2.2). While 71% of the grasslands were initially in a poor conservation status (C or X), after our treatment, 87.6% of these grasslands are now in a good or medium-level conservation status (A or B). Eighty-six percent of the treated area has seen its conservation status improved throughout the project. It is noted that the improvement was not always an increase in a single level of conservation status. Rather, over the 6-year monitoring period, 34.2% of the area improved by 2 conservation status levels and 9.2% improved by 3 levels. Inversely, the reason why some treatments did not result in a conservation status improvement is that most of the work was done only 1 or 2 years before the final monitoring, and thus, the time elapsed was too short to observe significant improvement. The second reason is that the seed bank was probably missing and that restoration by only changing the mowing regime was thus not enough to overcome this constraint.

Species richness significantly increased with time for the three restoration techniques (Fig. 2.20). The number of species gained per year was equivalent for all techniques with on average 2.4 additional species being found each year. After 6 years, the mean species richness after mowing and sowing reached that of the reference meadows. This was not the case for fresh hay transfer where an increase to 33 species was found compared to 47 species in the reference meadows. This difference is explained by the fact that initial states of meadows, and thus the species richness and composition, differed among the restoration techniques. Fresh hay transfer had a significantly lower initial species richness than mowing and sowing (23 species compared to 27 and 30 species for sowing and mowing respectively). This result indicates that time is a major factor for the botanical restoration of

Table 2.2 Transition matrix showing the evolution of the conservation status of restored meadows

		Final conservation status				
		A	B	C	X	Total
Initial conservation status	**B**	19.8%	9.2%	/	/	29.0%
	C	27.7%	14.5%	3.4%	/	45.6%
	X	9.2%	6.5%	7.3%	1.7%	24.7%
	Wooded area	0.0%	0.7%	/	/	0.7%
	Total	56.7%	30.9%	10.7%	1.7%	100.0%

Note: Values in the table are percentages of treated areas

Fig. 2.20 (**a** & **b**) Species-rich restored grasslands. Photo credits (**a** & **b**): Maïké Dellicour

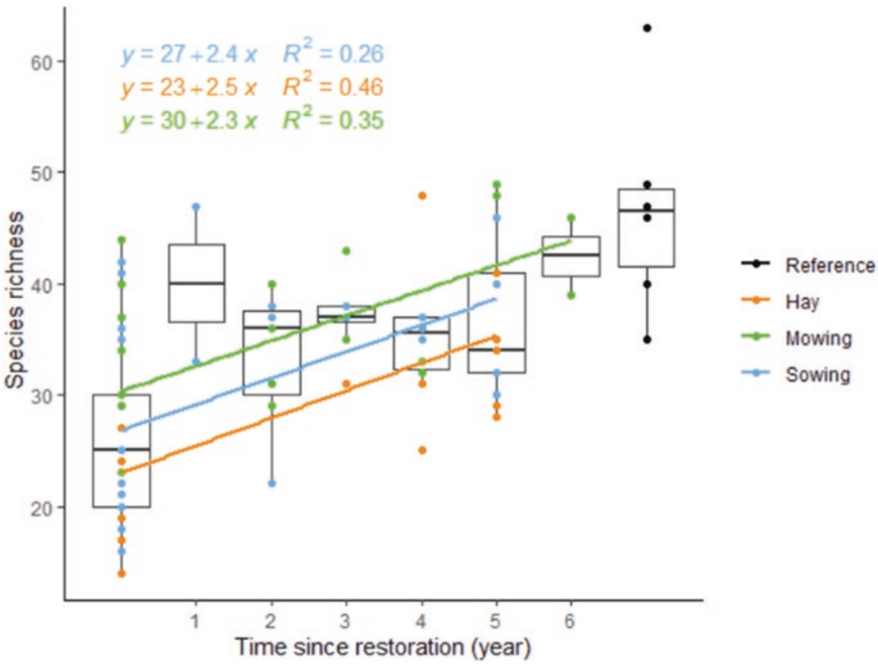

Fig. 2.21 Linear regression models showing significant positive relationships between species richness and time since restoration (mowing: $F = 13.94$, $p < 0.001$, sowing: $F = 10.49$, $p = 0.003$, hay: $F = 19.1$, $p < 0.001$). (Note: boxplots show the 25th percentile, median and 75th percentile)

meadows. Indeed, in the case of mowing, 6 years were needed to observe a successful species richness recovery. Five years were necessary for sowing, and 7 years would probably have been necessary to observe complete success of species richness recovery with fresh hay transfer (Fig. 2.21).

The effect of these treatments on plant community composition was also evaluated and showed very encouraging results. Similarity between species composition

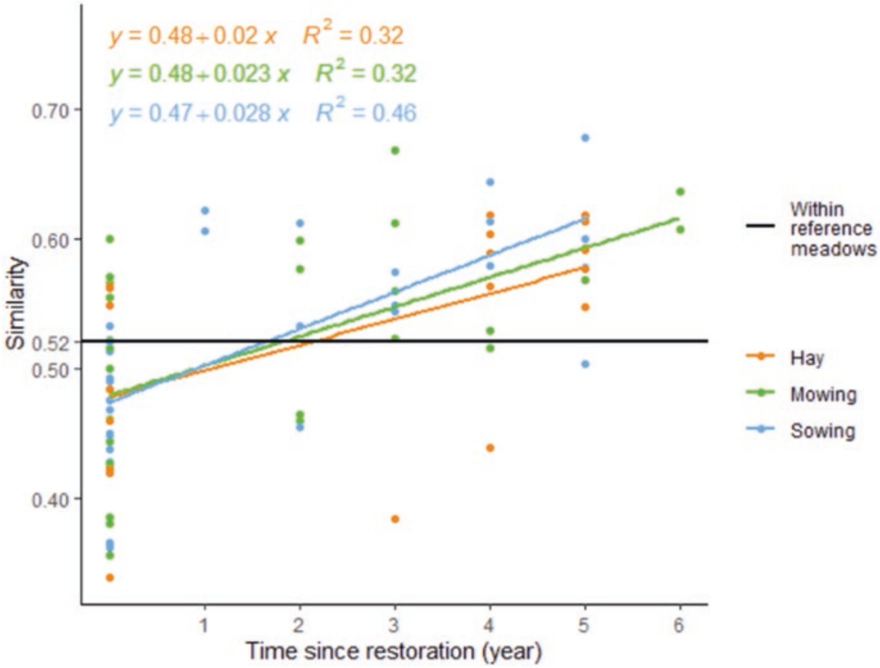

Fig. 2.22 Linear regression models showing significant positive relationships between similarity with the average species composition of reference meadows and time since restoration (mowing: $F = 12.18$, $p = 0.002$, sowing: $F = 25.86$, $p < 0.001$, hay: $F = 10.17$, $p = 0.004$). Note: boxplots show the 25th percentile, median and 75th percentile

of treated and reference meadows was calculated, and results clearly showed that similarity of restored plant communities with reference meadows significantly increased with time for the three treatments. On average 2.4% of similarity with reference meadows was gained each year (Fig. 2.22). Remarkably, it took only 2 years for the similarity between the treated and the reference meadows to be equivalent to the similarity observed between alternative reference meadows. This result indicates a restoration success since the recovery of species composition attained the level of similarity found within reference meadows.

What About the Project Worked, and What Did Not Work?

Our investigations have shown that sowing, hay transfer and mowing, all led to equivalent significant changes each year in species richness and species composition compared to the reference meadows, which were chosen to represent conservation status A. It is noted that the final species richness of areas treated by fresh hay transfer did not reach that of reference meadows because initial states differed between techniques, and fresh hay transfer had a lower initial species richness. Following the decision-making tool of Goret et al. (2021), mowing was favoured for less degraded meadows and active population enhancement (sowing) was reserved for highly degraded meadows. Thus, having started with a lower number of species, meadows restored by fresh hay transfer will necessarily take more time to reach target species richness. All techniques showed successful regeneration of species composition. These results demonstrate that adapting restoration technique depending on the initial degradation state and the direct vicinity of the reference meadow is a relevant factor in successful conservation.

The outcomes of this project highlight the importance of soil preparation and transfer of seed-containing plant material in more impaired sites. This is consistent with the outcomes of several studies that have tested the effectiveness of species introduction to restore lowland hay meadows or alluvial meadows in Europe (Edwards et al., 2007; Schmiede et al., 2012; Baasch et al., 2016; Harvolk-Schöning et al., 2020). Success of mowing also attests to the efficiency of management extensification as a means of restoring slightly degraded meadows, based on the assumption that target species possibly remain in the seed bank. It is congruent with the results of previous studies which reported a positive effect on species richness after cessation of fertilization and implementation of extensive management through mowing or grazing (Pallett et al., 2016; Van Vooren et al., 2018).

In slightly altered landscapes providing seed sources and stopping disturbances are valuable tools for conserving valuable grassland habitats and achieve restoration goals (Ruprecht, 2006). Postponing mowing from spring to summer was demonstrated to be effective in promoting plant and invertebrate diversity in European meadows (Humbert et al., 2012). Similarly, a twice-a-year defoliation frequency was shown to be efficient in enhancing plant and insect richness and increasing export of potassium in agricultural lands (Uchida & Ushimaru, 2014; Piqueray et al., 2019).

Before any type of treatment, it is recommended that the local ecological conditions should be explored to decide which type of restoration action is more likely to succeed (Prach et al., 2020; Goret et al., 2021). Mowing should be favoured in mildly impaired sites, where there is low environmental stress and evidence of intermediate productivity, which are usually located in more well-preserved landscapes (Prach et al., 2020). When selecting a treatment, financial and practical factors must also be considered. Passive recovery naturally requires lower costs than seed transfer, while fresh hay spreading additionally imposes organisational constraints. Fresh hay must be transferred immediately to the receptor site after cutting, as storage would compromise seed viability due to rapid fermentation (Blakesley & Buckley, 2016). The large volume of fresh hay which needs to be transferred also requires close proximity between donor and receptor sites (Blakesley & Buckley, 2016). Notwithstanding these precursory conditions, compared to sowing, green haymaking is less time-consuming, requires commonly available machinery (Blakesley & Buckley, 2016) and produces a very more efficient seed harvest yield (Scotton & Ševčíková, 2017). The residual hay layer left on the receptor site can also favour seedling establishment (Loydi et al., 2013).

In conclusion, the 'LIFE Bocage Meadows' project shows that meadow restoration can be a great success if the treatment is adapted to the local conditions. These depend on the ecological context, which includes initial levels of degradation, presence of a seed bank and an adjacent well-preserved meadow. In addition, there are financial and practical factors that must be factored in depending on the environment. Decisions on the appropriate treatment can be difficult to make given the multiple factors that must be carefully considered. To this end, the recently published dichotomous key which was used in this project should assist practitioners to make appropriate choices for a successful restoration process (Goret et al., 2021).

Case Study 4: Grassy Community Restoration in Brazil[4]

Project Rationale(s) and Strategy(ies)

Tropical grasslands are essential global ecospheres, being home to much unique biodiversity, providing key ecosystem services and sustaining the livelihoods of hundreds of millions of people. However, notwithstanding these remarkable attributes, they are amongst some of the most misunderstood, neglected, mismanaged and threatened ecosystems worldwide. In Brazil, the disproportionate focus on forest ecology and its restoration, coupled with the economic interests of colonial legacies, have created widespread misconceptions on the ecology of old-growth grasslands which has had many detrimental, long-standing ramifications for our understanding of grassland restoration (Overbeck et al., 2015; Silveira et al., 2022).

Restoration programs in Brazilian open area biomes, which include grasslands, savannas and shrublands are currently in their infancy, and, in addition, the paucity and geographically limited nature of the studies which have been undertaken have hindered us from learning transferable lessons. However, the last decade has witnessed an upsurge in theoretical and empirical papers addressing grassland restoration which, collectively, have led to an improved conceptual framework tailored for grasslands and savanna restoration (Buisson et al., 2019). In this regard, the major sources of degradation of tropical grasslands include (i) land conversion for crops and pastures, (ii) quarrying, (iii) mining and (iv) afforestation (tree planting in former non-forested sites).

Fortunately, tropical grassland restoration practice and science are now becoming more common, and in this case study attention is given to three independent initiatives: (i) restoration of Cerrado grasslands (the world's most biodiverse savanna) which has been degraded by conversion to pasture, (ii) restoration of Cerrado grasslands that are degraded by afforestation and (iii) restoration of nutrient-impoverished megadiverse montane, which include the *campo rupestre* grasslands in south-eastern Brazil (Silveira et al., 2016) which have been degraded by mining and quarrying. These three grassland types are floristically, climatically and edaphically distinct, and each has been impacted by different degrading factors to varying degrees. Consequently, restoration efforts have been necessarily localised. The goal of this case study is to provide a brief overview of restoration programs established in these three areas, then address the specific learned lessons, rather than to attempt a comprehensive review of all grassland restoration initiatives in Brazil.

Major Project Concerns and Barriers

The Cerrado, the largest Neotropical savanna, originally covered more than two million km^2. This area mostly involved seasonal climates and was found in the nutrient-poor soils of central Brazil. Its original distribution covered 20 degrees of latitude, with elevations ranging from 100 to almost 2000 metres above sea level (Borghetti et al., 2020). Cerrado vegetation has been found to be extremely heterogeneous, and variations are is driven chiefly by local fire regimes, water availability and soil fertility (Bueno et al., 2018). Open, fire-prone formations have a continuous biodiversity layer of herbaceous strata composed of grasses, graminoids, forbs and sub-shrubs and a discontinuous woody layer formed by scattered, small-sized shrubs and trees (Fig. 2.23a). For decades open grasslands in the core area of the Cerrado have been converted to intensive cattle raising using fertilized pastures composed of invasive African grasses. These pastures dominated by African grasses require intense and constant NPK fertilization, and decade-old fertilization regimes have produced negative legacies that have had drastic consequences for natural communities and ecosystems. Consequently, the major challenges for the restoration of open grasslands are (i) returning soil fertility parameters to pre-disturbance conditions and (ii) removing or suppressing the invasive grasses. In contrast to forests, native herbaceous species of savannas are light-demanding, and shade cannot be used to control or restrict the exotic species (Sampaio et al., 2019). Although challenging, returning sites to natural low soil fertility is expected to provide multiple benefits, including the prevention of invasion by African grasses (Giles et al., 2021), restoring the natural soil microbiota (Wolfsdorf et al., 2021) and the shifting of plant communities from fast-growing to a more appropriate slow-growing functional signature (Giles et al., 2021).

In the State of São Paulo, representing the southern portion of the Cerrado, grasslands establish in places with more fertile soils, higher precipitation, and are located close to semi-deciduous forests. In addition to conversion to pastures, these grasslands have been degraded by replacement with or because of ingress from, pine tree plantations and general woody encroachment. Afforestation and encroachment pose major threats to the biodiversity and ecosystem services provided by these grasslands (Honda & Durigan, 2016, Haddad et al., 2020). Ironically, when these areas are targeted for restoration, they are commonly ‘restored’ by tree planting, which in fact represents inadequate understanding of the aim of the intervention, and has several negative consequences, leading to their degradation instead of restoration. Clearly, if restoration goals include the recovery of old-growth savanna biodiversity

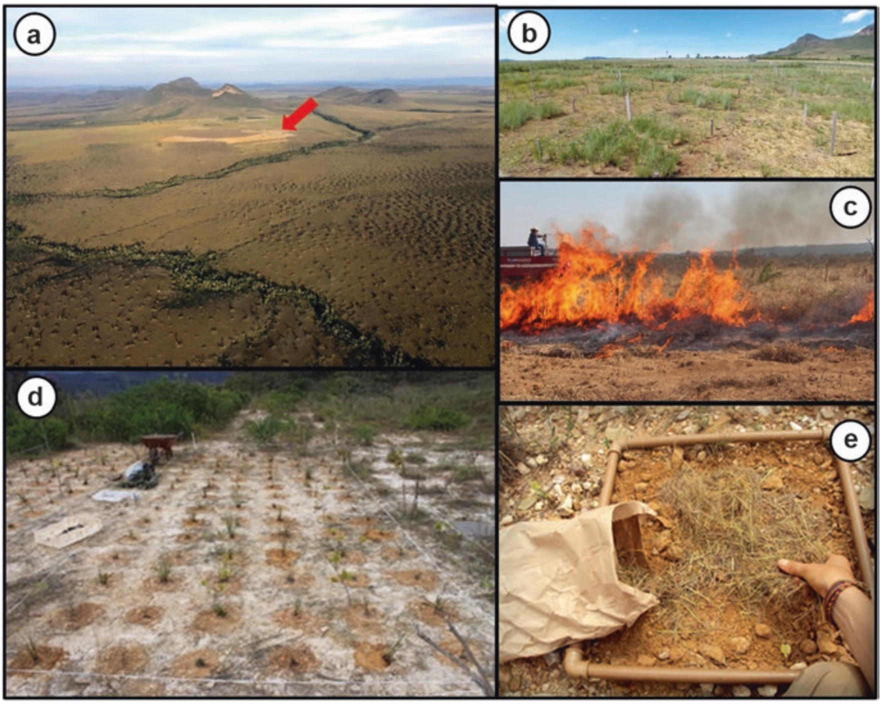

Fig. 2.23 Aerial view of the Chapada dos Veadeiros National Park in Central Brazil, where a large-scale restoration experiment has been implemented in a forest-grassland mosaic. (**a**) the red arrow points an area where multiple seed-based restoration treatments are being conducted, including seed sowing; (**b**) seedling planting in the Cerrado; (**c**) pescribed fires have been used in a nearby area to restrict invasive African grasses; (**d**) seedling planting in campo rupestre (**e**) and hay transfer have been experimentally tested to restore *campo rupestre* grasslands in Southeastern Brazil. (Photo credits: (**a**). Fernanda Barros; (**b**). Guilherme Mazzochinia; (**c**). Alessandra Fidelis; (**d**–**e**). Soizig Le Stradic)

and structure, interventions are required to prevent woody encroachment and reintroduce a broad suite of native grasses, forbs and shrubs (Cava et al., 2018).

In southeastern Brazil, significant areas of the *campo rupestre* grasslands have been lost due to opencast iron ore mining and quartzite quarrying. *Campo rupestre* is characterized by extremely impoverished soils, outstanding plant endemis and communities dominated by slow-growing and seed-limited species. This combination results in negligible natural regeneration after soil disturbance (Le Stradic et al., 2018, Onésimo et al., 2021). As a consequence, major concerns include (i) sourcing large amounts of high-quality seeds required for specific local revegetation (Dayrell et al., 2016), (ii) developing appropriate protocols for contributing species' propagation (Machado et al., 2013; Figueiredo et al., 2018a, b), (iii) finding suitable strategies for species' reintroduction and undertaking long-term monitoring (Le Stradic et al., 2014; Gomes et al., 2018; Figueiredo et al., 2021) and (iv) recovering ecosystem function in sites where topsoil's and sometimes subsoils, which have been entirely removed.

Key Project Features

Experimental Focus

Given the pronounced knowledge gaps in basic biological aspects of grassland species involved in these areas, initial studies have focused on an understanding of seed germination requirements, dormancy-breaking mechanisms, optimum conditions for seedling establishment and monitoring diversity after seed sowing or seedling planting (Pellizzaro et al., 2017). Although initially focused on a few woody species, researchers inevitably turned their attention to herbaceous species, which represent the bulk of diversity and which are essential for promoting soil vegetative cover (e.g., Figueiredo et al., 2020; Oliveira et al., 2021). In these early restoration attempts, seed sowing and seedling planting were understandably based on a trial-and-error approach, owing to virtually absent theoretical frameworks on grassland restoration.

More successful restoration of the open grasslands in Central Brazil was implemented through community-based networks that supply native seeds and seedlings for projects in the Cerrado area (Schmidt et al., 2019b). Many seed-sowing experiments were conducted in these projects, and a series of treatments were attempted to control invasive grasses (Fig. 2.23b, c). Direct seed-sowing experiments tested the survival of 75 native herbaceous and woody species for up to 2.5 years (Pellizzaro et al., 2017). Silva & Vieira (2017) evaluated the effects of seed burial, comparing surface exposure to buried seed and mulching (with no-mulch, 5-cm straw mulch and 10-cm straw mulch). The emergence, survival and growth of 16 woody trees of native Cerrado tree species with variable seed sizes and shapes and seedling type were also involved in this trial. In a second investigation, Coutinho et al. (2019) sowed seeds of 54 native grasses, shrubs and trees in order to test the effects of initial functional-group composition on assembly trajectory. Finally, Sampaio et al. (2019) tested whether seeding density affected native plant cover and whether soil ploughing is effective in controlling invasive grasses. These experiments were done in different soil types and with the different plant guilds of grasses, shrubs and forbs.

In one post-afforestation experiment (Haddad et al., 2020), the composition of herbaceous communities was compared across several treatments, which included (i) a burned and abandoned pine plantation, (ii) a burned and pine harvested site and (iii) an area planted with 82 native species which was previously used as a pine plantation. Haddad et al. (2021) later carried out an experiment consisting of a comparison of plant communities and vegetation structure in (i) abandoned pine plantation stands, (ii) areas open to passive restoration (natural regeneration after pine clearcutting) and (iii) native tree planting, where native tree seedlings were planted at high densities after pine clearcutting. The reference site was a Cerrado location which had never been exposed to tree planting. It has been noted that in previously afforested sites, the pine needle layer may prevent native regeneration after abandonment and cutting of the forest. In this respect, an experiment tested topsoil translocation, plant transplantation, direct seeding, topsoil translocation plus direct seeding and needle layer removal in both dry and wet grasslands (Pilon et al., 2018). Topsoil translocation involved the uprooting of plants and then extracting a 5-cm-deep layer of topsoil, where most seeds were concentrated.

For the *campo rupestre* project example, small-scale restoration experiments included monitoring the outcome of planting native shrub species (Fig. 2.23d), hay transfer (Fig. 2.23e) and direct seeding in experimentally degraded sites where the soil had been entirely removed. In the shrub planting experiments, Gomes et al. (2018) monitored survival, growth and recruitment of 10 shrubs 8.5 years after transplantation. The hay transfer experiment (Le Stradic et al., 2014) consisted of the spreading of seed hay (collected from pristine areas) at degraded sites and estimating seedling emergence after 14 months. The direct seeding experiment (Figueiredo et al., 2021) tested whether the addition of plant material (litter) improved seedling established of 14 native species. Finally, topsoil transfer was tested as a strategy to overcome the physical, chemical and biological filters of degraded ironstone *campo rupestre* by monitoring natural regeneration after spreading a 30 cm-depth layer of topsoil on bare soil in degraded areas (Onésimo et al., 2021).

Seed Networks

Since 2012, a partnership between the Brazilian Protected Areas agency (ICMBio), the University of Brasilia and Embrapa (the Brazilian Agriculture and Animal Husbandry Research Enterprise) has performed grassland restoration experiments in Central Brazil. These experiments aimed to develop efficient low-cost, seed-based techniques for restoring grasslands and savannas at a landscape scale (Pellizzaro et al., 2017; Coutinho et al., 2018). Direct seeding experiments also considered the effectiveness of the involvement of local communities and the use of agricultural machinery to reduce restoration costs. Three hectares were restored in 2012 and 2013, and a fruitful partnership with the Cerrado Seeds Network allowed the restoration of a further seven hectares in 2014. Seed collection, preparation and storage techniques were adapted or developed by that group using local ecological knowledge and available scientific information. People from local communities were trained and performed all restoration stages, from seed collection to seed sowing. In 2015, a power line company (Norte Brasil) established a pioneer agreement with ICMBio and local seed collectors to restore 95 hectares inside a protected area through mechanized direct seeding. The experiment sponsored by the power line company significantly increased the demand for native seeds and generated more than US$ 60,000 of income for more than 60 families within local rural communities in 2015–2016 (Schmidt et al., 2019b).

Understanding that seed collection for restoration can result in income generation and livelihood improvement, seed collectors funded a community association, named Standing Cerrado (*Cerrado de Pé* in Portuguese) that, in partnership with the Cerrado Seeds Network, now sells native seed for restoration projects in central Brazil. This cooperative has now become self-sustaining and is currently generating revenue to improve local livelihoods. It has also indirectly led to a decreased rate of local vegetation conversion, such as that involving pasture areas or other non-native use because conserved areas outside the park became important seed sources, generating income for local dwellers through seed collection and sale. The direct seeding restoration methods developed within this initiative cost less than US$ 3500 per ha compared to US$ 7000 per ha for tree seedling planting (Schmidt et al., 2019b). The cost of establishing a 1-year-old seedling by direct seeding has been shown to be cheaper than an equivalent approach through nurse-seedling planting for 56 of 57 species (Raupp et al., 2020).

Seed price for each species is established collaboratively among collectors, this value being based on the species density in natural areas coupled with the labour, time and equipment required for seed collection and seed processing. The Cerrado Seeds Network holds a National Register of Seeds and Seedlings and, in partnership with research institutions and universities, tests collections for seed quality, according to legal requirements. However, it is worth noting that because the use of native species is still incipient in grassland restoration, there are as yet in Brazil no established seed quality regulatory parameters for these species. Therefore, the Cerrado Seeds Network and partner institutions are developing and proposing parameters for seed quality tests for several species (Schmidt et al., 2019b).

Seed sourcing for large-scale restoration in *campo rupestre* is extremely challenging due to multiple reasons. First, most native and endemics are locally rare, have irregular fruiting seasons and have limited fruit production (Dayrell et al., 2016). Second, germination requirements and dormancy-breaking mechanisms are largely unknown. Third, large percentages of embryo-less and unviable seeds result in low-quality seed lots. Collectively, these factors make seed collection and seed quality unpredictable, consequently hampering extensive species propagation and restricting the establishment of seed production areas. Despite these initial challenges, intensive sampling in natural areas allowed for seed collection in sufficient amounts to make laboratory experiments, possible, together with greenhouse, trial-and-error seedling production (Dayrell et al., 2017).

Site Preparation and Seeding

In Central Brazil, sites were prepared in multiple ways before sowing seed. In the experiment by Pellizzaro et al. (2017), soil was ploughed prior to seeding to decrease dominance by invasive grasses and soil compaction one or two times during the dry season (May–October). In Coutinho et al. (2019), soils were prepared by repeated harrowing aiming for decompaction, uprooting invasive grasses and destruction of invasive grass seedlings that had germinated from the soil seed bank. In 2014, a controlled burn was conducted before soil harrowing to reduce invasive grass biomass. This approach facilitated the effectiveness and ease of soil harrowing and led to the removal of invasive grass seeds held in vegetation above the soil surface. Silva & Vieira (2017) spread seeds on the soil surface or buried at 3–5 cm depth and tested the effect of 5-cm and 10-cm straw mulching.

In Haddad et al. (2021), there was no soil tillage for native tree planting, so the underground structures of previously existing native species were preserved and therefore could resprout. Additionally, exotic grasses were controlled with glyphosate herbicide before native tree planting and for over 2 years afterwards. Pilon et al. (2018) and Haddad et al. (2020) did not perform site preparation in their experiments, except for what is mentioned above.

Most restoration experiments in *campo rupestre* did not involve any site preparation. This was most probably due to the assumption that native species adapted to low-fertility soils would be outcompeted by invasive species following increased soil fertility. However, in mined and quarried sites, iron ore or quartz extraction removed both soil and topsoil, so reconstructing the physical substrate remained a challenge that needed to be addressed before the seeding and planting stages. To reinstate native ecosystems post-mining, the development of a substrate similar to the iron-rich cap rock is necessary (Levett et al., 2021). To achieve this outcome, accelerating microbial iron cycling, dissolution and recrystallization of goethite catalysed by root exudates and bacteria and slope stabilization using biocrusts (complex association between soil, microorganisms and extracellular polymeric substances), have been tested as solutions to create environmental conditions suitable for the reintroduction of native species from ferrugineous *campo rupestre*. Nevertheless, a key challenge remains upscaling such biotechnologies to the landscape-level, which would lead to significant advances in mine-site restoration in Brazil (Levett et al., 2021).

A few studies examined the role of litter addition and topsoil transfer in the establishment of target species in post-mined sites. The establishment of native species was evaluated under four different conditions: (i) seeding on the degraded substrate, (ii) seeding covered by 1 cm degraded substrate layer, (iii) seeding on 1 cm topsoil layer and iv) seeding covered by 1 cm topsoil layer (Figueiredo et al., 2021). Another experiment established permanent plots to monitor floristic and life-form spectra in post-mined sites 4 years after topsoil transfer (Onésimo et al., 2021).

Major Project Outcomes

Under field conditions in the first rainy season after planting, Pellizzaro et al. (2017) found that 62 out of 85 species of trees, shrubs and grasses produced seedlings, of which 30 of them reached at least 20% survival rate. After the first year, 41 species had above 60% of survival, some with an astounding 80% survival rate. A separate study found that seed burial did not affect the emergence of species with round seeds, but negatively affected species with flat seeds (Silva & Vieira, 2017). Another found that harrowing and fire failed to eliminate the seed bank of invasive grasses which subsequently were able to re-establish, while short-lived shrubs and annual grasses lost dominance primarily to invasive species or perennial grasses. Most low-coverage plots shifted to invasive grass dominance after 2 years (Coutinho et al., 2019). Silva & Vieira (2017) showed that despite straw mulching reducing the emergence of native species with flat seed shape, it increased soil moisture and strongly reduced emergence of the invasive *Urochloa decumbens*, resulting in a higher growth rate of tree seedlings up to 1 year for five species. Encouragingly, results of these various seed-sowing experiments do indicate the feasibility of reintroducing a considerable number of native species from different functional groups in Cerrado restoration (Sampaio et al., 2019), but that controlling invasive grasses remains as a major challenge given, they have been shown to eventually regenerate.

Results from Haddad et al. (2020) showed that herbaceous plant communities of all three post-afforestation sites, regardless of management and fire history, were very different from the old-growth savannas that were destroyed to establish pine plantations five decades ago. Consistently, Haddad et al. (2021) showed that both passive restoration and native tree planting restored the structure, richness, and composition of the woody layer, reaching values like the reference ecosystem, but in all treatments, the herbaceous layer lacked the sub-communities of shrubs and herbs typical of undisturbed savannas even 15 years after passive restoration. Thus, these results are consistent with a growing body of evidence that shows that the species-diverse herbaceous communities of tropical savannas are unable to recover rapidly after afforestation and fire exclusion.

Even after 8 years post gravel extraction degraded *campo rupestre* sites were characterized by altered soil properties, and plant communities with impoverished seed banks. Species composition was still very different from that at reference sites (Le Stradic et al., 2018). Unfortunately, this result suggests that relying on natural regeneration is not a feasible strategy. Even more disappointing were results from the previous hay transfer experiments which showed that few seedlings emerged following the spreading of this material despite the large number of seeds contained in the hay (Le Stradic et al., 2014). This outcome indicates that hay transfer may not be as useful a method for restoring degraded areas of *campo rupestre* as envisaged. In this respect, a hypothesis that remains to be tested is whether mechanical seeding on lightly cultivated or slotted soils might create seed niches and constitute a viable restoration alternative.

Despite these findings, other positive outcomes have arisen in studies on ferrugineous *campo rupestre*. The mineralization of the biocrusts have been found to have led to the formation of biocemented aggregates that mechanically stabilized the crushed iron-rich waste material suggesting the potential of synthetic biocement as a long-term stabilization strategy for waste rock stockpiles, engineered slopes and mine remediation requiring the reformation of iron-rich duricrust (Paz et al., 2021). Another promising result is the finding that root exudates in this iron-rich substrate contributed indirectly to iron dissolution, particularly during phosphate solubilization, and the resulting surplus iron not taken up by the plants was redeposited, promoting the cementation of the residual minerals (Paz et al., 2020). Another study found that litter addition to the first 20 cm of the substrate plus seed sowing promoted the establishment of herbaceous and woody species (Figueiredo et al., 2021). In a further topsoil transfer experiment, 105 species were subsequently identified, and community composition and life-form spectra progressively resembled the reference areas (Onésimo et al., 2021). Unexpectedly, weed presence did not prevent the regeneration of native species. Altogether these results indicate that a combination of the appropriate reconstruction method for the physical environment and the correct site preparation appear to be promising restoration strategies in post-mined sites.

What About the Project Worked, What Did Not Work and Why?

These various direct seeding experiments in Central Brazil have suggested promising strategies for some types of grassland restoration. They indicate that when developed in an inclusive social-economic context, direct seeding has the potential to increase biodiversity, overcome the prohibitive costs of seedling planting and generate income for local communities. Nevertheless, seed sowing still appears to have a limited role in controlling invasive non-native grasses, and for the use of slow-growing species which remain less represented in seed-sowing programs because of their relative lower fecundity when compared to fast-growing species. Notwithstanding the innate problems which are apparent, the success of the Cerrado Seeds Network has been now established, and this is likely to increase following increased legal flexibility in terms of a relaxation of seed testing and commercialization rules (Schmidt et al., 2019b).

Grassland restoration after afforestation has been rarely studied but current evidence suggests that tree cutting and managing appropriate fire regimes have positive effects in re-establishing plant communities. However, the development of more effective strategies for the regeneration of the herbaceous communities is needed (Haddad et al., 2020). Already, prescribed fires have been shown to reduce the biomass of invasive species (Damasceno & Fidelis, 2020) and to help restore post-afforestation Cerrado sites (Zanzarini et al., 2019). Also, there is some suggestion that the combination of fire and ploughing may be an effective method to remove or at least restrict invasive grasses, but this treatment may need to be applied several times and/or for years. Fire should be used carefully as a restoration tool, and the fire regime needs to be managed considering natural fire frequency and the management context of specific vegetation types (Schmidt et al., 2019b; Haddad et al., 2020). Ploughing, which was shown to be effective in decompaction, uprooting of invasive grasses and destruction of invasive grass seedlings, may jeopardize regeneration of natives in bud banks by destroying or damaging underground storage organs.

Scepticism towards the feasibility of *campo rupestre* restoration has arisen owing to the repeated lack of spontaneous natural regeneration (Le Stradic et al., 2018), failures in hay transfer experiments (i.e., Le Stradic et al., 2014), the impoverished nature of native seed banks (Medina & Fernandes, 2007), overall low seed quantity/quality produced by native species (Dayrell et al., 2017) and low germination under field conditions (Figueiredo et al., 2021). These results unambiguously indicate that seed-based techniques would prove unviable as restoration strategies. Nevertheless, despite the low percentage of establishment, other studies suggest (i) a significant biotechnological potential for biocrust reconstruction, (ii) moderate to high seedling survival and growth across a different range of substrates (Machado et al., 2013, Figueiredo et al., 2018b), (iii) substantial success in sexual and vegetative propagation, (iv) moderate persistence and recruitment of planted individuals (Gomes et al., 2018), (v) a positive effect of the incorporation of plant litter and (vi) topsoil transfer in revegetation of post-mined sites with viable cost (Figueiredo et al., 2021). Taken collectively, the current prospect for *campo rupestre* restoration is more positive than previously thought a decade ago.

Grasslands and savannas that have been subjected to medium or high-intensity disturbance are typically composed of low-resilience grasses, short-lived herbs, and shrubs that have shallow roots and bud bank organs (Schmidt et al., 2019b). Nevertheless, regeneration after endogenous disturbance is chiefly driven by the resprouting of underground storage organs (Buisson et al., 2019). Therefore, transplanting underground storage organs may be a cost-effective strategy to enhance resilience in degraded grassland restoration, by increasing resprouting particularly for slow-growing species which can increase biodiversity. This hypothesis does, however, remain to be tested under field conditions.

The recently growing body of evidence on tropical grassland restoration indicates that multiple strategies may turn out to be feasible as large-scale alternatives to seedling planting (e.g., Buisson et al., 2019). Regarding this issue, a better understanding on the ecology of tropical grassy biomes, improved recognition of their value to mitigate climate change and better resourcing of restoration projects (Silveira et al., 2022), are expected to provide additional support for the improved restoration policy and higher standards of practice. These improvements are much needed so that tropical grassland restoration science can reach maturity. As such, in addition to improvements in knowledge and practice the importance of developing appropriate and multidimensional indicators of grassland restoration success is likely to emerge as an immediate challenge for the restoration sector in the coming years.

Chapter Synthesis

These four case studies focus on different grassy community types, and each shows that grassy community conservation and restoration is difficult, but feasible. Similar examples of success have been shown in other countries (Buisson et al., 2018; Puthod et al., 2020; Wagner et al., 2020, Freitag et al., 2021). While there were specific factors guiding the planning, approaches and goals set, each case study shows that successful outcomes are possible and the process of undertaking ecological restoration provided important learning experiences. Whilst the differences between regions and countries in these case studies are instructive, there are some key areas of similarity suggesting there may be fundamental factors or principles underlying the approaches taken in grassy community restoration which, when adjusted to suit specific local conditions or settings, will broadly lead to success (e.g., Goret et al., 2021). This is a very important message given the threat that these communities face. It should also give hope that in the future increasing knowledge, capacity and technologies and engagement by people and communities will allow us to halt and even reverse grassy community loss at local, landscape and perhaps even global scales.

For grassy community restoration to occur at the global scale required to repair anthropogenic damage over millennia it will take a commensurate effort in terms of time, resourcing and commitment from countries, jurisdictions, and their communities. Whilst some countries are clearly more advanced in this area than others, this can create the opportunity for knowledge sharing and cross-jurisdictional support. We assert that grassy communities can be, and should be, better integrated into the fabric of our landscapes, be they as farm-scapes, abandoned forestry, urban regions, transport corridors or protected remnant areas. For this to occur, purposeful decision-making and goal-setting, guided by clear pathways and concrete actions that meaningfully involve people, communities and practitioners, must be put in place so that ad-hoc actions and intermittent successes of the past are turned into purposeful strategies and widespread global advances of the future.

Ten key implications that have arisen from our reflections on the four Case Studies presented. These are:

Implications

1. Grassy communities can be restored using careful regenerative and reintroduction approaches into agricultural, post-forested, urban and other landscapes where they once existed or where they are now the desired community type.
2. Grassy communities can be conserved and enhanced through restorative management approaches.
3. Grassy community restoration can achieve high levels of species and functional diversity as well as temporal resilience.
4. Restored grassy communities create a myriad of biodiversity, ecological and ecosystem service benefits.
5. Restored communities must be purposefully managed and maintained over time to preserve their structural and compositional integrity.
6. Effective and inclusive (e.g., to local communities) seed supply chains, delivering seed in quantity, quality, price and ethically are critical to successful restoration.
7. Seed production approaches are likely to be critical to supplementing seed supply for restoration, especially for rare or uncommon species.
8. Developing overall sector capacity can improve training, technical skills and knowledge and lead to greater employment and career opportunities for people and communities involved in restoration.
9. Improved infrastructure and technology development will be crucial to increasing the effectiveness and scale of restoration undertaken.
10. Landscape-scale grassy community restoration relies on the formulation of insightful and finely crafted government strategies and policy to create the settings, frameworks and coordination required to build markets, improve sector capacity and meet ambitious grassy community restoration targets.

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Chapter 3
Restoring Tropical Forests: Lessons Learned from Case Studies on Three Continents

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Introduction

Complexity of structure, high species diversity, niche abundance and a myriad of ecological interactions combine to challenge the very human notion that tropical rainforests could ever be truly restored to their natural condition. Despite this, our understanding of the ongoing loss of tropical biodiversity, particularly the widespread intolerance of obligate forest species to fragmentation and loss of forest cover, has spurred global attempts to reverse the decline. Evidence of this support is seen in the Bonn Challenge, the UN Decade of Ecosystem Restoration and various 'trillion tree planting' initiatives (Brancalion & Holl, 2020). Tropical restoration is underpinned by ecological succession and community reassembly theory, where ecosystem recovery is largely driven by interactions between animals and the plants on which they depend (e.g., Howe, 2016). In this scenario, succession is neither uniform nor predictable (Norden et al., 2015), but in an ecosystem restoration context it provides a means to test traditional notions of sequential replacement and a framework to monitor development of ecosystem processes and function (Hobbs & Norton, 1996).

Typically, restoration interventions have been dichotomised as either 'passive' or 'active', the former meaning reliance solely on natural regeneration and the latter involving tree planting (DellaSala et al., 2003). However, an either/or approach is overly simplistic; anthropogenically modified landscapes impose both biological and socio-economic constraints, and restoration requires nuanced approaches that consider landscape context and prior land-use, as well as biophysical factors (Holl & Aide, 2011). Restoration ecologists respond by using various techniques, ranging from manipulation of natural regeneration through to planting increasingly diverse mixtures of species, and various terms describe these techniques, as discussed by McDonald et al. (Chap. 7, this volume).

Tropical restoration is mostly conducted in developing nations, on lands where agriculture provides the primary livelihood. This means that loss of agricultural land (which can be regarded as an opportunity cost) to forest restoration for the provision of global ecosystem services exposes lower socio-economic societies to additional economic stress, unless such services are fairly valued and paid for. Recognising this, restoration may embrace economically or culturally valuable species to encourage uptake, but this and other trade-offs also require a nuanced approach. As such, large-scale global restoration initiatives test the ability of restoration ecologists to ensure potential biodiversity benefits are realised, livelihoods are protected and appropriate restoration techniques are applied (Di Sacco et al., 2021).

In this chapter, we provide a brief overview of the ecological and socio-economic factors that influence tropical forest recovery, illustrating how these have been addressed under various ecological and socio-economic settings. We have used three long-term restoration case studies carried out in tropical Australia, Asia and Central America. Our studies encompass various levels of intervention used to achieve restoration outcomes that are relevant to both the level of degradation and landscape context. Despite inherent differences, common problems and challenges can be seen to emerge. We close by detailing key unifying lessons distilled from these case studies.

Key Constraints

Ecological Factors

Following human disturbance, autogenic recovery rates in tropical forest ecosystems vary tremendously. In some cases, biomass and species composition recover within a couple of decades (Marín-Spiotta et al., 2008; Letcher & Chazdon, 2009). Elsewhere land may remain in a state of arrested succession due to highly degraded soils or competition with aggressive ruderal species (Chazdon, 2003; Lamb et al., 2005). Rates of natural regeneration depend on a combination of the type, intensity, duration and sequencing of past disturbances, the ecology of the specific forest type (Holl & Aide, 2011) and crucially, the density and composition of incoming seed rain. Since the seeds of most tropical forest species are recalcitrant, few retain viability in the soil seed bank beyond 2–3 years post-clearing (Vázquez-Yanes & Orozco-Segovia, 1993). Consequently, long-term recovery of tree species richness and its accompanying biodiversity depends mostly on seed rain. This in turn is dependent, firstly, on the presence of seed sources near restoration sites and, secondly, on viable populations of seed-dispersing animals, given that 70–90% of wet forest tropical tree species are dispersed by animals (Howe & Smallwood, 1982).

Many studies demonstrate that animal-mediated dispersal is often a primary factor limiting tropical forest recovery (reviewed in Holl, 2007). Regeneration may also occur vegetatively from seedlings and/or re-sprouts from stumps, roots or stems already present when land was abandoned. The contribution of different modes of regeneration to ecosystem recovery depends on the nature an intensity of prior disturbance (e.g., low-intensity agriculture or selective logging) providing these modes of regeneration remain after human disturbance.

After seed arrival, several other factors may limit seed germination and seedling survival, as well as time to reproductive maturity. These include seed/seedling predation, competition from aggressive under-storey vegetation, stressful microclimatic conditions, limited availability of soil nutrients and diseases (Holl, 2012, Fig. 3.1). Seed/seedling predation by insects and mammals can be a major obstacle to the recovery of certain species on agricultural lands. On former pasture lands, aggressive exotic grasses (e.g., *Imperata cylindrica, Urochloa* spp., *Megathyrsus* spp., *Pennisetum* spp., *Saccharum spontaneum*) often form a monoculture which out-competes tree seedlings and elevates fire risk. Ferns (e.g., *Dicranopteris* spp., *Pteridium* spp.), shrubs and vines can rapidly overwhelm disturbed sites and impede the establishment and growth of forest trees (Zimmerman et al., 2007). Invasive species of both plants and animals are a particular obstacle to recovery in island ecosystems (Cordell et al., 2009).

Stressful microclimatic conditions may also limit seed germination and seedling survival and growth, particularly in seasonally dry forests (Vieira & Scariot, 2006). Light levels and air and soil temperatures are commonly much higher and humidity and soil moisture levels much lower in agricultural lands than in forests. Moreover, drier conditions in pastures and high grass biomass provide ideal fuel for fire, which

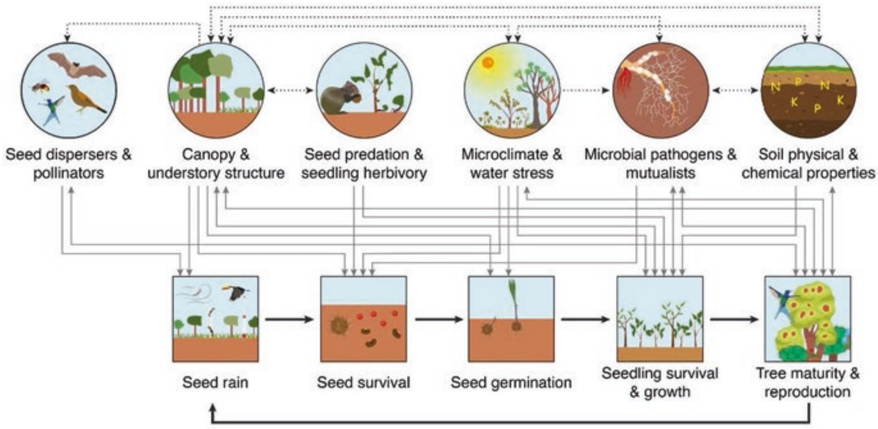

Fig. 3.1 Ecological factors affecting the rate of forest recovery. Square boxes illustrate stages in the dispersal, establishment and reproduction of vegetation. Circles illustrate ecological factors that affect the rate of transitions between the stages. (Holl et al., 2000)

kills seeds and seedlings of wet forest species, as most are not well adapted to fire (Janzen, 2002; Nepstad et al., 2008). Fires are becoming increasingly important with rising temperatures and more variable rainfall resulting from climate change, in addition to anthropogenic disturbances (Armenteras et al., 2021), and in some cases may lead to a transition towards savanna vegetation dominated by fire-tolerant species.

Soil nutrients and structure vary greatly across the tropics and as a function of land use history. In the large areas of the tropics covered by oxisols and ultisols, seedling growth is often limited by low nutrient levels, although the extent of nutrient limitation and the primary limiting nutrient vary by soil type and extent of degradation (Powers & Marín-Spiotta, 2017). After intensive human use, soils may become highly compacted, which impedes root growth and water-holding capacity. Many tropical trees form mycorrhizal associations, which facilitate phosphorus uptake, but agricultural land uses may substantially alter microbial communities (Carpenter et al., 2001; Allen et al., 2005), in turn affecting nutrient cycling.

The relative importance of each particular factor (Fig. 3.1) varies greatly from site to site depending on local-, landscape- and regional-scale factors. Surrounding land uses affect not only the abundance and composition of native flora and fauna that arrive at a site but also the abundance of potential seed and seedling predators, invasive plants and pathogens and the risk of fire spreading from adjacent land uses. If remnant trees are intentionally retained within agricultural lands, such as shade trees for coffee, cacao or for grazing animals in pastures, they can facilitate recovery (Guevara et al., 1986; Ramos et al., 2020). Higher within-site tree cover plays an important role in facilitating natural recovery by attracting seed-dispersing animals, ameliorating stressful microclimate conditions and shading out light-demanding vegetation (Holl, 2012). Recovery also tends to be faster in relatively warmer and wetter lower-elevation areas, which generally favour more rapid growth (Zarin et al., 2001).

Socio-economic Factors

Restoring tropical forest ecosystems involves both direct and indirect costs, regardless of the particular methods employed. Whilst ecologists have delivered the *technical* means to restore self-sustaining ecosystems, the long-term *socio-economic* sustainability of restored ecosystems is assured only when the value of their benefits outweigh restoration costs and where restoration outcomes are valued higher than alternative land uses. Although there is a growing body of literature showing this to be true in many situations (Abram et al., 2016; Bradbury et al., 2021; Mappin et al., 2021), no reliable mechanisms exist to convert benefit values into cash incentives for local people, who bear the brunt of the financial and social costs of restoration.

The level of cost depends on the extent of restoration intervention needed and the needs of the local economy. Even where restoration is achieved solely through natural regeneration, costs remain for site protection, including fire prevention, livestock exclusion, prevention of logging and for assisting regeneration by weeding, fertilizer application and mulching. Where natural regeneration potential is insufficient, tree planting becomes necessary, and this requires seed collection and tree nurseries, in addition to planting, maintenance and monitoring costs. If start-up funds are obtained as loans, interest payments must be added to the ancillary overheads, which also include costs for planning, training, legal services and verification. Finally, lost opportunity costs (defined as income forgone from the most likely alternative land-uses) must also be considered. Labour is the greatest cost component (Brancalion et al., 2019), and since labour costs depend on the local economy, total restoration cost varies enormously among countries.

As expected, assessing the economic value of ecosystem services is challenging. The Economics of Ecosystems and Biodiversity study (TEEB 2009) determined that the average value of tropical forest ecosystem services was $US6,120 ha/year ($US7732/ha/year today, adjusted for inflation, based on 109 studies). Watershed-related services accounted for 38.8% of that value, climate regulation (mostly carbon storage) 35.9% and forest products 11.2%, with cultural services and genetic resources comprising the remainder. All these values depend on biomass accumulation and biodiversity recovery—both core goals of restoration.

Economists and social scientists have made minimal progress with realizing these benefits to local communities in financial terms. For estimating the value of carbon sequestration, the UN's REDD+[1] scheme offers some hope in this respect. However, the scheme has been criticized for subverting local forest management practices to meet global demands and for failing to deliver adequate income to local people. Furthermore, forest-related CO_2 emissions in most of the countries where REDD+ has been implemented have not declined as expected (Duchelle et al., 2018; Elliott, 2018). Although forest-carbon value often exceeds revenue from the main

[1] Policies and incentives, developed under the UN Framework Convention on Climate Change, to finance restoration and conservation of forests as carbon sinks.

drivers of deforestation (Abram et al., 2016; Mappin et al., 2021; Jantawong et al., 2022), REDD+ has largely failed to provide financial incentives for restoration due to cultural factors, inadequate governance and unfavorable socio-economic conditions. Another problem is the unpredictability of fluctuations in carbon credit prices, relative to the profitability of alternative land uses, which constitutes a considerable financial risk.

Although the value of non-timber forest products (NTFPs) in restored forests is low compared with other benefits, start-up investment is often not needed and local people can directly market the products to customers (de Souza et al., 2016). NTFPs also provide security when other income sources decline (Guariguata et al., 2009), and their diversity provides a buffer against fluctuating market prices. However, sustainable harvesting is essential for the long-term viability of slow-growing species, and such a strategy requires careful monitoring. If yields begin to decline, community-level agreement on self-regulation needs to be introduced.

Financial realization of watershed services is also problematic. Efforts in this respect mostly consist of estimating the cost of 'avoided detrimental impacts', such as preventing flooding, landslides or mitigating declines in agricultural productivity arising from drought or siltation of irrigation infrastructure. These issues are mostly unpredictable in time and place. Furthermore, inhabitants of upper catchments often bear the brunt of restoration costs, whereas many of the water-related benefits accrue to downstream users. This suggests that watershed services should be regarded as a public good rather than a commodity, and in this respect, payments for them have increasingly been derived by governments through taxation, with successful schemes well-documented in Latin America and China (Porras et al., 2008).

Attainment of all these income streams from forest restoration depends on competent governance, particularly as it relates to land tenure, taxation and the absence of corruption (Mansourian, 2020). Another key requirement is extensive capacity-building, to mentor stakeholders in the skills, initiative and integrity, needed to implement these financial mechanisms. Innovative marketing will also be essential, because both investors and the public are largely unfamiliar with environmental services, and assistance will be needed in assembling support to sustain these services.

Case Studies

Strategies to overcome the ecological and socio-economic constraints associated with tropical forest restoration are detailed in the following case studies. They illustrate three different approaches, with each based on a specific landscape and social context. Whilst these studies all involve various levels of active restoration, each ultimately relies on natural regeneration to complete the recovery of forest structural complexity, biodiversity and ecological function. Each example also demonstrates that successful restoration involves meaningful engagement with all stakeholders and instituting a concomitant obligation to ensure that community and landholder needs and expectations are met. Importantly, each Case Study details projects established over 20+ years ago, allowing comprehensive insight into the processes of community reassembly, external support, on-going financial issues and the challenges which have been faced.

Case Study 1: Applied Nucleation in Costa Rica

Achieving restoration at scale is a major challenge for practitioners, and a key factor is cost, particularly for active restoration methods (Holl & Aide, 2011). Developing active restoration approaches that facilitate forest recovery, as much as or more than plantation-style planting, while reducing implementation costs is the key to scaling up tropical forest restoration. Trees can be planted in spatial patterns (Shaw et al., 2020), such as strip-planting (i.e., rows of seedlings between unplanted areas allowed to regenerate naturally), planting patches of trees in applied nucleation (Corbin & Holl, 2012) or focusing plantation-style planting in areas where regeneration is impeded.

The Islas Project, established between 2004 and 2006 in southern Costa Rica (Holl et al., 2020), is the longest-running tropical restoration experiment designed to directly compare the effectiveness of applied nucleation to plantation-style planting and natural regeneration. Forests in this region are at the boundary between Tropical Premontane Wet and Rain Forest zones. They range in elevation from 1100 to 1430 m and receive a mean annual rainfall of 3500–4000 mm, with a dry season from December to March. Restoration treatments were replicated across 15 ~ 1-ha sites, each separated by >700 m and spread across a >100-km^2 area (Fig. 3.2). At each replicate site, three 0.25-ha (50 × 50 m) plots were established, receiving one of three treatments: natural regeneration, applied nucleation or plantation. Plantation treatments were uniformly planted with trees, whereas applied nucleation plots were planted with six patches of trees of three different sizes: two each of 4 × 4, 8 × 8 and 12 × 12 m (Fig. 3.3). No planting was done in the natural regeneration plots. Four tree species, widely used in a range of agroforestry practices in the region, were inter-planted in alternating rows, each separated by 2.8 m. Species included two later-successional species, *Terminalia amazonia* (Combretaceae) and *Vochysia guatemalensis* (Vochysiaceae), and two fast-growing N-fixing species, *Erythrina poeppigiana* and *Inga edulis* (Fabaceae). A range of surrounding forest cover (~8–80% within 500 m) was integrated into the experimental design. Results from this study are detailed in more than 55 publications to date (http://www.holl-lab.com/islas-project.html) as well as in a recent synthesis paper (Holl et al., 2020). Here we highlight core findings that are most relevant to the theme of this chapter.

Applied nucleation, where only a quarter of the number of seedlings was planted as compared to plantations, is effective in restoring a range of ecological metrics but costs less than more extensive planting and promotes ecological heterogeneity over time (Holl et al., 2020). Most floral and faunal groups quantified had similar abundance and/or species richness by the end of the first decade of recovery in both

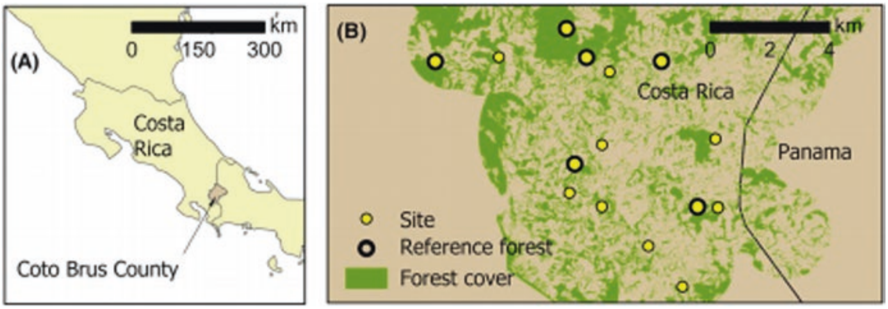

Fig. 3.2 Study area (**a**) and the 15 study sites from which data were collected in southern Costa Rica (**b**). (Forest cover data are from Mendenhall et al., 2011)

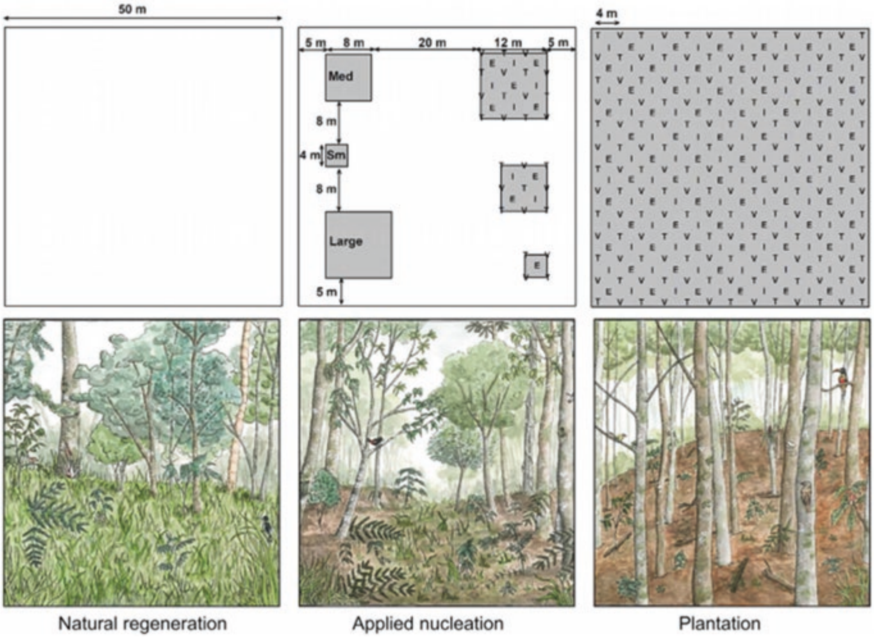

Fig. 3.3 Top panels show planting design and bottom panels illustrate the plots after 15 years, showing both planted and naturally recruited vegetation. In top panels, grey areas were planted with *E Erythrina poeppigiana*. *I Inga edulis*, *T Terminalia amazonia* and *V Vochysia guatemalensis*. *Sm* small, *Med* medium. (Artist credit: Michelle Pastor)

applied nucleation and plantation restoration treatments (Fig. 3.4). Applied nucleation and plantation treatments attracted similar abundances of seed-dispersing birds and bats (Fig. 3.4a, b, Reid et al., 2014, 2015b), resulting in similar abundance and species richness measures of animal-dispersed seed deposition and germination and seedling recruitment (Fig. 3.4c, d, Reid et al., 2015a; Holl et al., 2017; Werden et al., 2020). Furthermore, both active restoration treatments had consistently greater recovery compared with natural regeneration (Holl et al., 2020). A critical

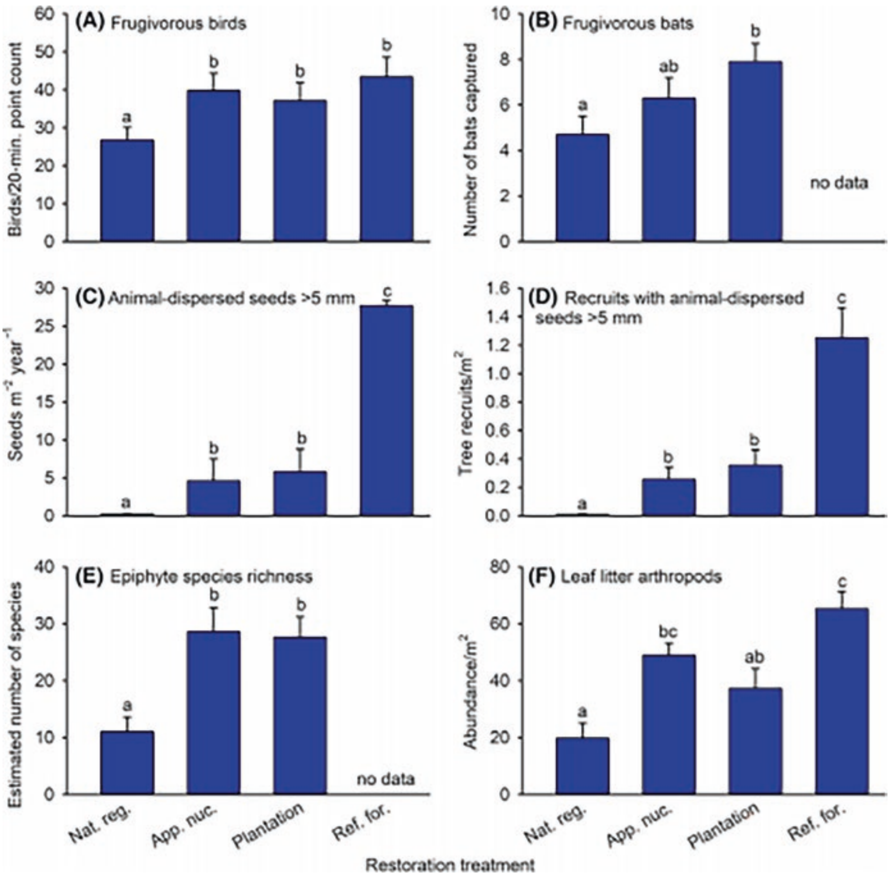

Fig. 3.4 Responses of ecological variables to forest restoration treatments. (**a**) Frugivorous bird abundance in 2016 (n = 11 sites); (**b**) frugivorous bat abundance in 2009 and 2012 (n = 10, Reid et al., 2015b); (**c**) abundance of animal-dispersal seeds >5 mm in 2012–2013 (n = 10, Reid et al., 2015a); (**d**) abundance of recruits with animal-dispersal seeds >5 mm in 2015 (n = 13, Holl et al., 2017); (**e**) estimated species richness of epiphytes in 2015 based on sample-based accumulation curves (n = 13, Reid et al., 2016); and (**f**) leaf litter arthropods in 2012 (n = 4, Cole et al., 2016). Values are means ±1 se. Means with the same letter do not differ significantly using Tukey's multiple-comparison test among treatments

factor in the vegetation recovery of both active restoration treatments was probably the role played by large-seeded dispersers such as toucans (Ramphastidae), in the recruitment of late-successional species (Reid et al., 2021); active restoration treatments overall had close to two-fold the proportion of large-seeded species arriving into treatments compared with natural regeneration sites (Werden et al., 2021).

Applied nucleation costs less to implement than plantation-style planting because of the lower cost of planting and the maintenance of fewer trees (in this case 27% of trees planted in plantations). This is a key benefit that enables its use for scaling-up restoration, to achieve ambitious global targets (Wilson et al., 2021). Applied nucleation also promotes heterogeneity. This applies in the structural sense, as there is a gradient of canopy cover from the interior of planted tree nuclei to natural regeneration areas (Holl et al., 2013). It also applies to seed dispersal, as vertebrate-dispersed seeds were more heterogeneous in applied nucleation than in the plantation treatments (Werden et al., 2021). Moreover, growth of later-successional saplings was 39% higher in applied nucleation plots than in plantations, probably due to greater light availability (Kulikowski et al., 2023). As such, applied nucleation promotes recovery that more closely mimics natural regeneration, but at an accelerated rate.

Over the 17 years of this case study, recovery patterns have been seen to shift rapidly. For example, *Inga edulis* quickly became the dominant planted tree across all sites in the first few years of the study (Holl et al., 2011), but it has since been displaced by *Vochysia guatemalensis* (Lanuza et al., 2018). In turn, while it is not surprising that aboveground biomass was greater in the plantation treatments after a decade of recovery, litterfall rates at the onset of the second decade were comparable in plantation and applied nucleation restoration strategies (Lanuza et al., 2018). This indicates that productivity in applied nucleation can reach similar levels to the more expensive plantation option within just a decade. Successional dynamics of recruiting species have also shifted rapidly, from dominance by early-successional species in the first few years to a marked increase of later-successional species in active restoration treatments (but not in natural regeneration), during the second decade (Holl et al., 2017; Werden et al., 2021). Such rapid changes underscore the importance of long-term monitoring of recovering systems to increase understanding of the implications of different restoration interventions.

Recovery across sites was highly variable, consistent with most multi-site restoration studies. For example, above-ground biomass varied ~10-fold across sites (Holl & Zahawi, 2014). Whereas near complete canopy cover in plantation plots was established at some sites within 3–5 years, other plantation plots still have only partial canopy cover, even after >15 years. We have determined that a few important baseline measures can help predict whether or not a site is going to recover rapidly. First, we found a strong positive relationship between the initial rate of change in planted tree height in the first 2 years and the above-ground biomass of those same sites 6–8 years later (Holl & Zahawi, 2014). As such, early indicators of growth represent benchmarks for regeneration capacity. Second, we found that the number of tree recruits that establish within the first year and a half and their canopy cover are good indicators of the number of recruits and canopy over upcoming years (Holl et al., 2018). As such, leaving a targeted restoration site for a year or two to document natural recovery, before deciding whether to enact active restoration measures, is strategic. Finally, while it is important to quantify general trends that guide our ability to implement restoration at scale, a key management lesson is that selection of restoration strategies must be tailored to local site conditions.

Local restoration strategy was consistently more important, in the first few years, than was the percentage of surrounding forest cover, in driving recruitment patterns. Whereas surrounding percent forest cover was consistently a weak predictor of recovery (Reid et al., 2015a; Holl et al., 2017), the composition of surrounding remnants was key (Zahawi et al., 2021). Presence of a potential 'mother tree' within

100 m of a restoration plot resulted in a 10-fold increase in conspecific recruit abundance on average; proximity of adult 'mother trees' was also important, as was their abundance, where abundance of 'mother trees' was strongly correlated with the amount of forest cover. As such, results underscore the importance of assessing not only the amount of surrounding forest cover to predict the potential for recovery but also the species composition of that forest.

Whereas the ecological and economic advantages to less-intensive restoration approaches (such as applied nucleation and natural regeneration) are clear, limitations and social obstacles exist (Zahawi et al., 2014; Holl et al., 2020). First, most practitioners are accustomed to the widespread practice of plantation-style tree planting. Furthermore, many funding agencies measure success as numbers of trees planted. Whereas there are clearly other factors to consider as metrics of success, moving ingrained perceptions away from the 'need' for monoculture plantations will be difficult. Second, less uniform approaches tend to look 'messy' or 'unkempt'. This transitional successional stage is a hindrance; local residents and investors may perceive it as abandoned land or project failure. As such, clear guidelines and sharing of information about proposed restoration approaches are essential to overcome both of these factors. Spatially patterned approaches are likely to be most appropriate where large land holdings are being restored with limited resources (Holl et al., 2020; Wilson et al., 2021).

Finally, central to restoration success is the assumption that what is set aside for recovery can persist in a regenerating forested state for a prolonged period of time (i.e., for several decades and ideally longer). However, experiences from our study, as well as assessments of the longevity of secondary forest patches in the region and elsewhere in Latin America (Reid et al., 2019; Schwartz et al., 2020), paint a somewhat challenging picture. Even within the confines of a formal study framework with year-round vigilance and monitoring, incursions and damage to some of our plots have occurred multiple times. This low-grade degradation can come in the form of livestock (e.g., cattle or goats released to graze in plots), opportune harvesting of trees for firewood or other timber purposes, and, in the worst situation, the wholesale conversion of land-use by an owner who no longer wished to participate in the project. Such incidents underscore the importance of understanding local socio-economic drivers of deforestation and developing effective socio-economic tools to counteract them (Brancalion & Holl, 2020; Di Sacco et al., 2021). To not do so increases the probability for long-term project failure and squanders the limited financial resources that are available for restoration.

Case Study 2: Testing the Framework Species Method of Forest Restoration in Northern Thailand

Chiang Mai University's Forest Restoration Research Unit (FORRU-CMU) was established in 1994 to develop effective techniques to restore the tropical forest ecosystems of northern Thailand, with a particular focus on biodiversity conservation and environmental protection. At that time, colonial-era logging and subsequent agricultural conversion had left remnants of both primary and degraded forests scattered across landscapes, which were consequently undergoing rapid deforestation. This was exacerbated by continued disturbance including (i) fire, (ii) hunting of large seed-dispersing animals and (iii) land use encroachment. A national logging ban in 1989 prompted the instigation of a policy to convert many cancelled logging concessions into protected areas. This created a demand to restore forest ecosystems to near-natural conditions, encouraging the return of maximal biomass, structural complexity, biodiversity and ecological functioning by means of harnessing regenerative potential at both landscape and site levels. This trend towards restoration, primarily for conservation, was boosted substantially in 1993 when the *Plook Pah Chalermphrakiat* project was launched to celebrate His Majesty King Bhumibol Adulyadej's Golden Jubilee. The goal was to plant diverse mixtures of native tree species on sites totaling more than 8000 km^2 nationwide.

One restoration technique in line with the above criteria, which had emerged at that time, was the framework species method (FSM) (Fig. 3.5), which was first conceived to restore forest to degraded sites in the Wet Tropics of Queensland, Australia (Goosem & Tucker, 2013). This method complements natural regeneration by densely planting open sites, close to natural forest, with woody species characteristic of the reference ecosystem (*sensu*, Gann et al., 2019) selected for their ability to accelerate ecological succession. To begin the process, a rapid site assessment first determines the existing density of natural regeneration, based on a count of saplings >50-cm tall, live stumps or remnant mature trees. Framework species are then planted to raise the stocking density to that capable of closing the canopy within 2–3 years (3100 trees per ha in upland northern Thailand).

Framework tree species are selected from the indigenous tree flora of the reference forest for their high survival and growth rates on exposed sites, ability to inhibit herbaceous weeds and attractiveness to seed-dispersing wildlife. Only a small fraction of species from the reference-forest ecosystem are established, but planted trees attract frugivorous birds and mammals, dispersing seeds of many other tree species from nearby remnants. Planted trees also create suitable ground-level microclimate and weed-free conditions, which support establishment of the seedlings that germinate from the incoming seeds by providing a moist, shaded microclimate, free of weed competition (Fig. 3.5).

Following training with the originator of the technique, Nigel Tucker, at Lake Eacham National Park, Queensland, FORRU-CMU staff, adapted and tested the concept to restore upland evergreen forest in northern Thailand as the first reference-forest target (EGF, sensu Maxwell, 2001).

A survey of evergreen forest (EGF) trees in Doi Suthep-Pui National Park recorded more than 250 species (FORRU-CMU, 2005). Thereafter, multiple individuals of 100 identified species were tagged for a phenology study to determine optimal seed-collection times. Nursery experiments were performed to develop effective germination techniques, which produce potted saplings (30–50 cm tall) by the start of the rainy season—the optimum planting time (Blakesley et al., 2002). These experiments led to the development of production schedules, detailing the

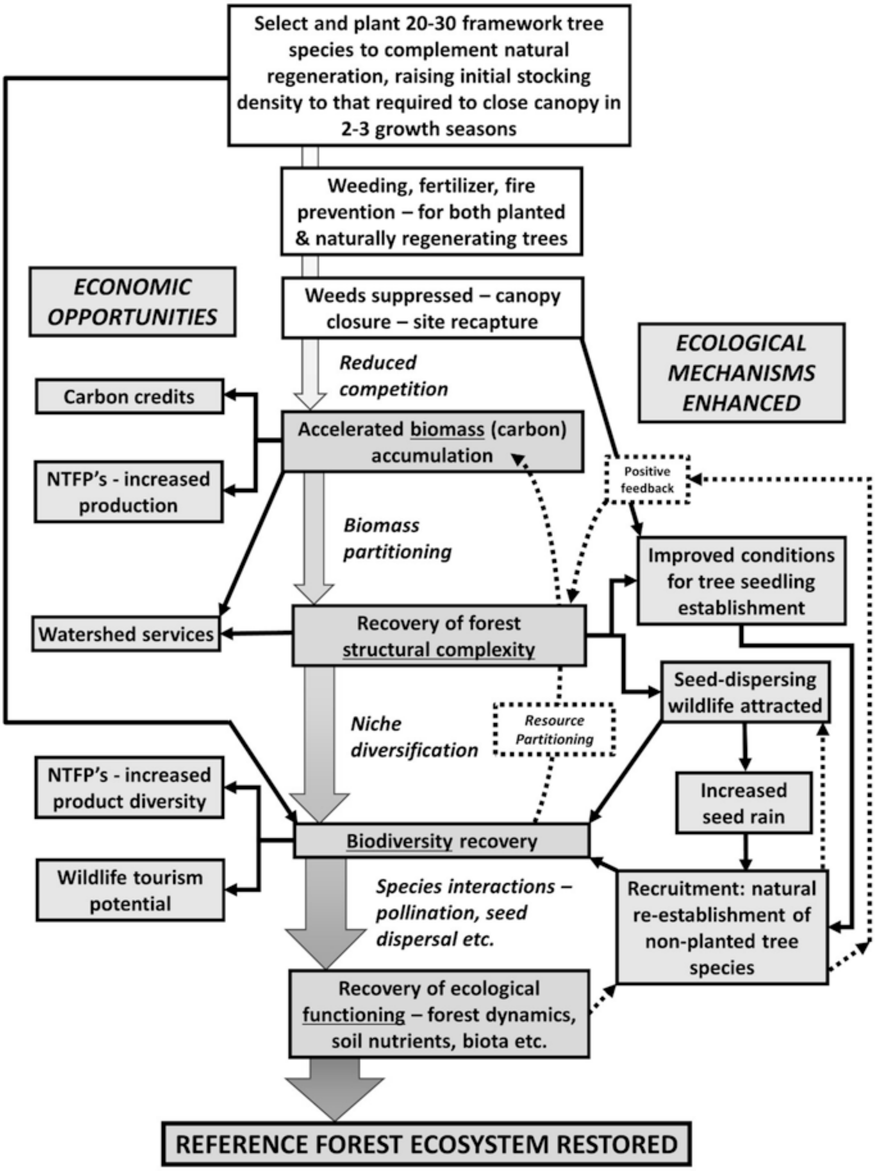

Fig. 3.5 How the framework species method works

treatments and timings required for efficient planting-stock production of each species (Elliott et al., 2002; Elliott & Kuaraksa, 2008).

Field trials were established annually (1996–2013), forming a chronosequence and a wildlife corridor, covering 33 ha of a watershed 1200–1325 m above sea level. The plot system overlooked the Hmong community of Ban Mae Sa Mai, in the

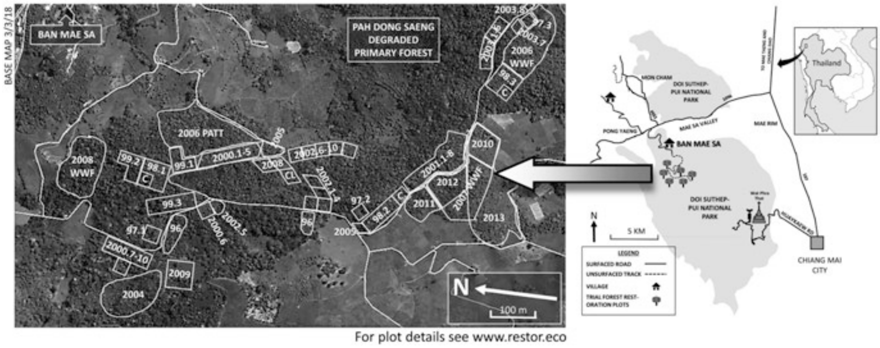

Fig. 3.6 A chronosequence of trial plots, planted annually from 1996 to 2013. Numbers indicate year of planting

upper Mae Sa Valley of Doi Suthep-Pui National Park (DSPNP), Chiang Mai Province (Fig. 3.6, 18°51′46.62″ N, 98°50′58.81″ E). Plot details and locations may be viewed at www.restor.eco.

Plots ranging in size from 0.48 to 6.4 ha, were planted annually with various mixes of 20–30 framework species. The initial density of planted trees was calculated as 3100 per ha, less the estimated density of pre-existing natural regeneration, the latter being determined by surveys using circular sample plots of 5 m radius. Weeds were cut 6 weeks before planting, followed by a single application of glyphosate, which provided the planted trees with weed-free conditions for 6–8 weeks, before further hand-weeding became necessary. Planting stock comprised saplings 30–50 cm tall, grown from locally collected seeds. Trees were grown in plastic bags (22.8 × 6.3 cm) in a medium of forest soil, peanut husk and coconut husk in the ratio of 50:25:25.

Planting spots were marked with bamboo canes, spaced 1.8 m apart. A triplicated field trial in 1999, which compared spacings of 1.5–2.5 m on subsequent species recruitment (Sinhaseni, 2008), confirmed that a spacing of 1.8 m was optimal. Wider spacing delayed canopy closure, which resulted in weed persistence and fire. Closer spacing resulted in higher tree mortality and lower tree species recruitment. Approximately 50–100 g of fertilizer (NPK 15:15:15) was added around the base of each tree stem (Elliott et al., 2000), along with a mulch of cut weeds or corrugated cardboard.

This basic planting protocol was varied each year, in order to test the relative performance of different tree species and the effectiveness of various silvicultural treatments, including spacing, fertilizer types and dosages, weeding frequency, pre-plant pruning, bare-rooted vs. potted plant stock and cardboard mulch mats.

Hand weeding and fertiliser application, applied to both planted trees and natural regeneration, were performed three times in both the first and second rainy seasons after planting. Fire breaks were cut in mid-January at the start of the hot-dry season. Subseqeuntly, until mid-April (the start of the rainy season), fire prevention teams manned observation points to detect and extinguish any fires approaching the study area.

Standard data-collection protocols were developed, specifically to determine which tree species met framework-species-qualifying criteria. We monitored survival and growth 2 weeks after planting and at the end of the first, second and, sometimes, third rainy seasons. Survival and relative growth-rate data were combined into a relative performance index, allowing comparison among both the species and the treatments tested each year (see Elliott et al., 2013). Plots planted in 1998, 1999 and 2000 were also monitored over 6 years to determine first flowering and fruiting events and to assess their attractiveness to wildlife.

A key outcome of this work was an effective FSM for restoring evergreen forest on Stage-3 degraded land (sensu Elliott et al., 2013) (Fig. 3.7). Top-performing framework tree species were identified (Elliott et al., 2003) and the silvicultural treatments that maximized post-planting performance were determined (Elliott et al., 2000).

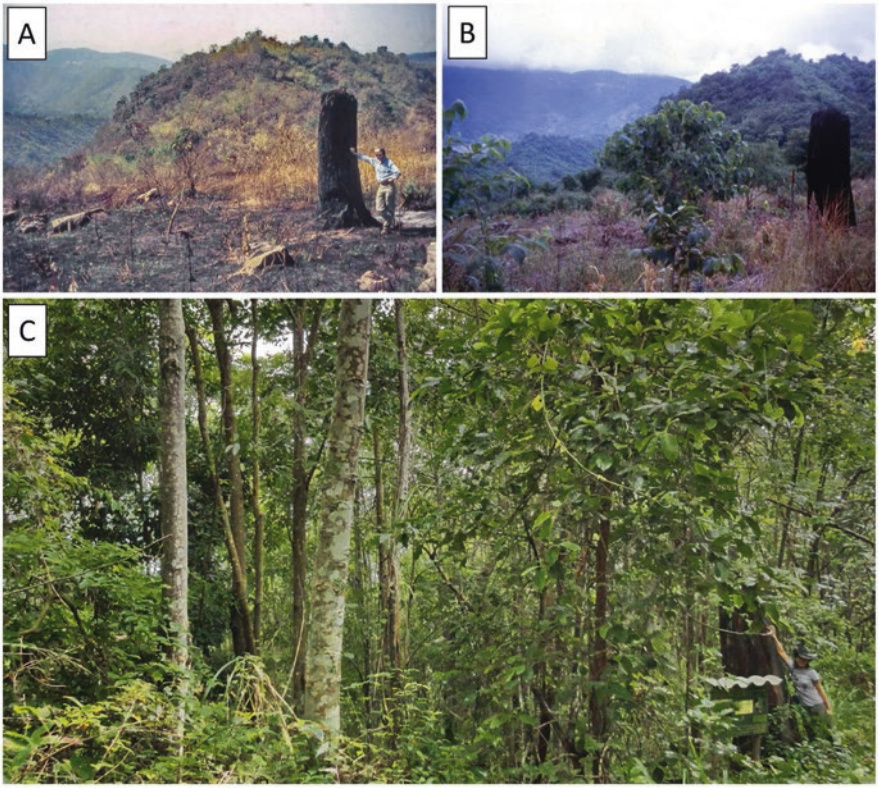

Fig. 3.7 Deforested, over-cultivated and repeatedly burnt, this site in the upper Mae Sa Valley supported very little natural regeneration (**a**). Within 1 year of planting framework species in 2000, several of the planted trees over-topped weeds and began site recapture (**b**). By 2012, a structurally complex and biodiverse forest had re-established, with many trees germinating from incoming animal-dispersed seeds (**c**)

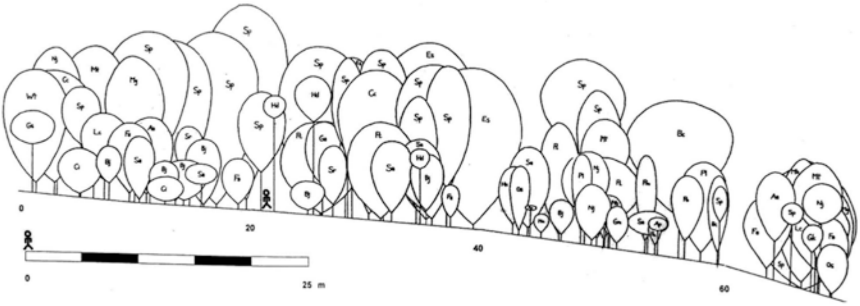

Fig. 3.8 Profile diagram (6 m wide) showing the multilayered canopy achieved by the framework species method 6 years after planting

Rapid biomass and carbon accumulation were achieved by design, since framework species were deliberately selected for high survival and rapid growth and planted to achieve high initial stocking density. Jantawong et al. (2017) reported that tree-carbon stocks in the FSM plot system exceeded those of nearby old-growth forest remnants after 16–17 years. Above-ground tree-carbon accumulation was 106 ton C/ha over 14 years—almost double the pan-tropical average for natural forest regeneration (58 ton C/ha) over 20 years (Silver et al., 2000) and substantially higher than that achieved by 17-year-old teak plantations in western Thailand (16-ton C/ha) (Chayaporn et al., 2021).

Partitioning of the accumulating biomass resulted in rapid recovery of forest structural complexity (Fig. 3.8). Using the best-performing species and maintenance regimes, canopy closure can now be achieved routinely within 2–3 years. After 6 years, pioneer species form an upper canopy of 16–18 m above ground, with planted climax tree species creating a dense under-story 8–10 m high (Fig. 3.8). Tree seedlings and saplings form a dense ground layer growing in a deep layer of leaf litter, with litterfall reaching rates typical of old growth forest in 14–16 years (Kavinchan et al., 2015). The last structural components to return were vascular epiphytes and woody climbers, which appeared 18–20 years after tree planting.

Structural complexity created the niches required for biodiversity recovery. Species richness of the bird community increased from about 30 before planting to 88 after 6 years, representing about 54% of bird species recorded in nearby mature forest (Toktang, 2005). Sinhaseni (2008) documented 73 species of non-planted trees re-colonizing the plot system (0.46 ha sampled) within 8–9 years, most having germinated from seeds dispersed from nearby forest by birds (particularly bulbuls), fruit bats and civets. Species richness of mycorrhizal fungi (Nandakwang et al., 2008), lichens (Phongchiewboon, 2006) and bryophytes (Chawengkul, 2019) also increased, often exceeding the levels found in natural forest.

Recovery of ecological functioning, particularly those plant-animal interactions that enable pollination and seed dispersal, led to the return of natural forest dynamics, as evidenced by the density and diversity of regeneration ready to replace the planted trees (Sangsupan et al., 2018), particularly pioneers, which live for only 20–25 years.

In addition to an effective restoration procedure for EGF in northern Thailand, the project generated information for generic research methodologies, which were needed to devise framework species approaches suited to the ecological and social circumstances of almost any tropical forest type. This culminated in the publication of a guide for research students in 2008 (Forest Restoration Research Unit, 2008). The manual included standardised protocols for nursery and field experiments, data collection, presentation and analysis. Building on lessons learned from the EGF plots, FORRU-CMU devised equally effective FSMs for lowland deciduous forest in northern Thailand, bamboo-deciduous forest in Kanchanburi Province (Sapanthuphong et al., 2011) and lowland evergreen forest in Krabi Province (Elliott et al., 2008).

From the outset, education and outreach, based on the research outputs of the project, were essential activities of the unit. Educational events were implemented for school children and their teachers, workshops for NGO's, government officers and community groups and training courses for professionals. Text books were produced in multiple languages, enabling outreach to extend to most south-east Asian countries. Units, based on the FORRU-CMU model, were replicated in China (Weyerhaeuser & Kahrl, 2006) and Cambodia (Kim, 2012), assisting forest authorities in those countries to interpret and establish FSMs, suited to their local forest types and socio-economic conditions.

Since FORRU-CMU is in a science faculty, our primary role has been to overcome the *technical* barriers to forest restoration. However, when establishing field trials, close collaboration with local communities was essential, inevitably involving us in addressing socio-economic aspects of restoration. Consequently, we developed procedures to perform participatory site surveys, project planning and collaborative costing and management protocols to run community-based tree nurseries (Table 3.1). This led to FORRU-CMU's subsequent involvement in managing tree planting for Thailand's first model PES (Payments for Ecological Services) project, which linked restoration financing to private-sector bottled water production (Elliott et al., 2018).

Case Study 3: Ecological Function in a Restored Wildlife Corridor

Whilst Australia has a well-developed economy, tropical forest restoration is subject to the same ecological and socioeconomic constraints present throughout the tropics. North Queensland's Wet Tropics, the anthropogenic fragments of a Gondwanan remnant, have been impacted by typical patterns of human settlement, resulting in loss of habitat concentrated in areas of favourable climate, topography and high soil fertility. Some rainforest communities having been reduced to 2% of their pre-European colonisation extent. Lowland forests have been largely cleared for sugar cane and banana production, with only narrow strips of riparian forest remaining along major rivers. Most habitat below 40 m asl is limited to swamp and mangrove

Table 3.1 Breakdown of restoration costs for northern Thailand, at various levels of initial degradation using the framework species method and/or ANR (Elliott et al., 2013), for a 10-ha site

Field establishment costs	100% tree planting			Tree planting: ANR 50:50			100% ANR		
By budget items	Y1	Y2	Total	Y1	Y2	Total	Y1	Y2	Total
Planting stock	1838	0	1838	919	0	919	0	0	0
Materials and equipment	315	129	444	254	129	383	192	129	321
Transportation	146	24	169	100	24	123	54	24	78
Labour	1033	549	1582	874	547	1421	715	544	1259
Quantifiable transaction costs – Planning training etc.	54	21	75	54	21	75	54	21	75
Total field costs by budget item	**3387**	**723**	**4109**	**2201**	**720**	**2921**	**1015**	**718**	**1733**
By task									
Pre-planting site survey	13	0	13	13	0	13	13	0	13
Site preparation	297	0	297	244	0	244	191	0	191
Tree planting (+initial ANR tasks)	2346	0	2346	1219	0	1219	91	0	91
Maintenance (weeding, fertilizer) – 2 years	694	704	1398	694	704	1398	694	704	1398
Monitoring – 2 years	36	18	54	31	16	47	26	13	39
Total field costs by task	**3387**	**723**	**4109**	**2201**	**720**	**2921**	**1015**	**718**	**1733**
10% contingency for unanticipated transaction costs	**339**	**72**	**411**	**220**	**72**	**292**	**101**	**72**	**173**
Subtotal	**3725**	**795**	**4520**	**2421**	**792**	**3213**	**1116**	**789**	**1906**
Interest			**1399**			**821**			**371**
Grand total			**5919**			**4034**			**2277**
Costs per ton C sequestered (US$/ton C)			**10.78**			**7.34**			**4.15**

Data from Chiang Mai University's Forest Restoration Research (FORRU-CMU), August 2021
ANR assisted natural regeneration

complexes, whereas upland fragments above 400 m asl, including many protected areas, are surrounded by a highly modified cropping and grazing matrix of private lands which imposes strict boundaries on the movements of many obligate rainforest species.

Donaghy's Corridor was established to address this movement by restoring habitat to reconnect a 489-ha isolated reserve at Lake Barrine to the 80,000-ha block at Wooroonooran. These two National Park reserves were previously separated by ca.1-km of privately owned grazing lands. Intervening cattle pasture had been in place since the 1940s, and the banks of Toohey Creek, which flows through the property from Barrine into Wooroonooran, were severely eroded and compacted by livestock. In addition, large vertebrates such as the endangered southern cassowary (*Casuarius casuarius johnsonii*), a key species in the dispersal of fruits greater than 30-mm diameter, are now absent from Barrine and the overall loss of genetic variability has been documented in ubiquitous, but rainforest-dependent species at Barrine (Campbell, 1995).

A baseline survey recorded all vascular plants and mammals on the site before treatment, including vegetation along the creek, isolated paddock trees and other vegetation within 100-m of re-planted areas, but excluding forests at either end of the site. Permanent stock-exclusion fencing was erected around existing riparian vegetation and intervening pasture areas were re-planted. This ensured protection of higher-quality habitat resources so they could continue to attract seed-dispersing wildlife from adjacent areas. The corridor was established in four blocks over a period of 4 years, planting around 1.2 ha per year, with stems 1.7-m apart. Between 1995 and 1998, 16,800 selected seedlings from 100 reference ecosystem species (McDonald et al., 2016) were planted. Monitoring commenced on completion, focusing on colonisation by plants, reptiles and small mammals. Project design, establishment and monitoring parameters are discussed in Tucker (2000), and early post-establishment outcomes are detailed in Tucker and Simmons (2009) and Paetkau et al. (2009). Utilising both genetic and mark-recapture techniques, these studies demonstrated that within 5 years, planted areas functioned as both a movement conduit and habitat for some small mammals.

This restoration procedure at Donaghy's Corridor was designed to establish a complex forest structure which would encourage rapid faunal utilisation and movement. In comparison to case studies 1 and 2, this project used a larger number of species, of about 55 on average per year, with selection based on functional traits. In addition to 30–40 framework species (Goosem & Tucker, 2013), narrow endemics, threatened species, large-fruited species and food plants of targeted vertebrates such as cassowaries were also planted. Such diverse plantings may be considered 'maximum diversity' approaches (Goosem & Tucker, 2013; Florentine et al., 2016), to be used where ecological connectivity is a primary goal of restoration. In this case study, we document changes in vegetation composition and structure that occurred over a 26-year period, and their relationship to ecosystem function, as seen through the prism of vertebrate seed dispersal.

Species Diversity and Composition

Baseline surveys recorded 132 native plants existing on site prior to treatment. These occurred along the creek and within 100-m of the corridor edge; trees, vines and shrubs, mostly concentrated in two small fragments totaling 1.75 ha. In 2000, 3 years after planting had been completed, transect surveys of naturally regenerating species revealed 115 native plants (4472 records from 180 × 5 m × 3 m plots), 25 of these being sourced from forests outside the corridor. In 2021, a re-survey of these 180 plots revealed 153 regenerating native species (4501 records), where coincidentally, 25 were again sourced from forests outside the corridor. Eleven regenerating species had disappeared in the intervening period. In 2000, average diversity of regenerating species at ground level was 6.9 per 15 m^2. By 2021, average diversity at ground level foe species with stems less than 1 m, had increased to 15.4 per 15 m^2.

Regenerating species comprised 59 plant families. Lauraceae was the most common (17 species), followed by Sapindaceae (13 species). Both are characteristic families in well-developed rainforests, producing fleshy fruits which are dispersed by many birds and mammals. Families of other basal lineages, for example, Annonaceae, Aristolochiaceae, Monimiaceae, Myristicaceae and Piperaceae, were represented by a number of regionally endemic species such as *Galbulimima baccata* – Himantandraceae, both inside and outside transects.

Changes in average seed size were less apparent. The number of large-seeded species (>30 mm diameter) increased from 7 to 14. Species diversity increased slightly but consistently across all fruit sizes.

Regenerating species were an admixture of pre-existing and planted species. However, 16% of regenerating species were neither planted nor pre-existing, and they had clearly originated from elsewhere. Many species have multiple dispersal vectors (Tucker & Murphy, 1997), but birds are responsible for most dispersal (89% of species). At the same time, species dispersed by both mammals and birds accounted for 31% whilst 25% were wind-dispersed.

This percentage of wind-dispersed species was largely attributable to reproduction of planted genera from Rutaceae and Proteaceae. Other species are dispersed by water, insects and gravity. Of 14 large-seeded species present in 2021, two of these (*Gardenia ovularis* seed dimensions 40 mm × 20–38 mm, and *Beilschmiedia bancroftii* 65–70 mm × 50–60 mm) were neither pre-existing nor planted. They have been introduced from outside the immediate vegetation matrix, emphasizing the potential diversity effect of vegetation proximity and presence of vertebrate dispersers.

Over a 20-year period, shifts occurred in the typical successional stage of regenerating vegetation (Fig. 3.9) and by 2021, late-successional and gap phase species occurred in equal proportions. In the intervening period, gap-phase species marginally declined, intermediate and late-intermediate groups remained relatively stable, but numbers of late-successional species increased.

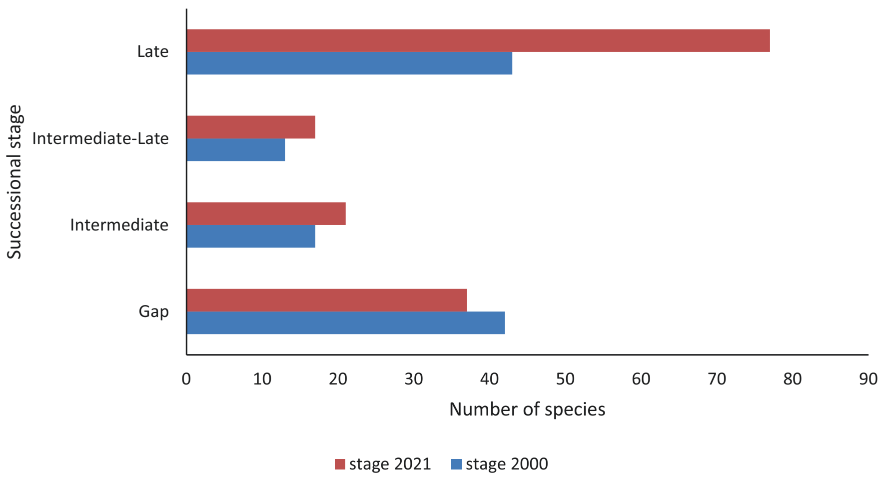

Fig. 3.9 Successional stage of regenerating vegetation

Structural Development

In 2000, plantings displayed an even canopy of 3–5 m in height, whilst regenerating seedling heights were 25–150 mm, but this simple structure was sufficient to suppress weed growth and attract seed-dispersing wildlife. By 2021, a taller canopy (up to 32 m) with under-storey elements was in place, in addition to a diverse ground storey. In this community, regeneration was composed of a range of life forms. Canopy trees (40 spp.) and under-storey trees (42 spp.) were dominant, but vines, scramblers and rattans (25 spp.) were conspicuous elements of the under-storey, in some instances reaching canopy level.

Planted trees largely comprised the canopy and under-storey layers, even though regenerated vegetation was increasingly conspicuous in the under-storey. Buttresses were common on canopy trees which also hosted small numbers of epiphytic orchids and ferns. Figure 3.10 compares the forest structure in a 26-year section of the corridor and a forest reference site at Barrine. Whilst the number of stems greater than 1 cm diameter at breast height and the number of individuals is similar in the two sites, basal area in the reference site is higher than the corridor transect.

Vertebrate Dispersal

The observed mechanisms of effective dispersal, germination and persistence indicate the existence of suitable plant niches within the forest and the presence of vertebrates capable of moving variously sized fruits. In this instance, dispersal of large-fruited Lauraceae such as *Endiandra insignis* (50–90 mm × 50–100 mm) and *Beilschmiedia bancroftii* almost certainly resulted from scatter-hoarding behaviour by giant white-tailed rats (*Uromys caudimaculatus*). Other Lauraceae appearing since 2000, including *Beilschmiedia tooram* and *Endiandra sankeyana* bear fruits of 35–55 mm dia. It is probable that white-tailed rats were also responsible for their dispersal, generally highlighting the important role of rodents in dispersal (Jansen et al., 2012). These plants and animals are all Wet Tropics endemics, characteristic of well-developed upland rainforests.

Thirty-one bird species were recorded in the corridor in 2021 (Tucker and Freeman, unpublished data). Twelve birds were mixed forest species and 19 were rainforest-dependent species, including four Wet Tropics endemics – the grey-headed robin (*Heteromyias cinereifrons*), Victoria's riflebird (*Lophorina victoriae*), pied monarch (*Arses kaupi*) and tooth-billed bowerbird (*Scenopoeetes dentirostris*). In addition, six obligate frugivores were present – the Australasian figbird (*Sphecotheres viridis*), the black-eared catbird (*Ailuroedus melanotis*), the wompoo fruit-dove (*Ptilinopus magnificus*), the topknot pigeon (*Lopholaimus antarcticus*), the tooth-billed bowerbird and the migratory channel-billed cuckoo (*Scythrops novaehollandiae*). Other species, including Lewin's honeyeater (*Meliphaga lewinii*), have mixed diets and are also important seed-dispersers in regenerating forest. Gape widths of all these species vary between 10 and 30 mm, accommodating the most common seed-size classes of regenerating vegetation.

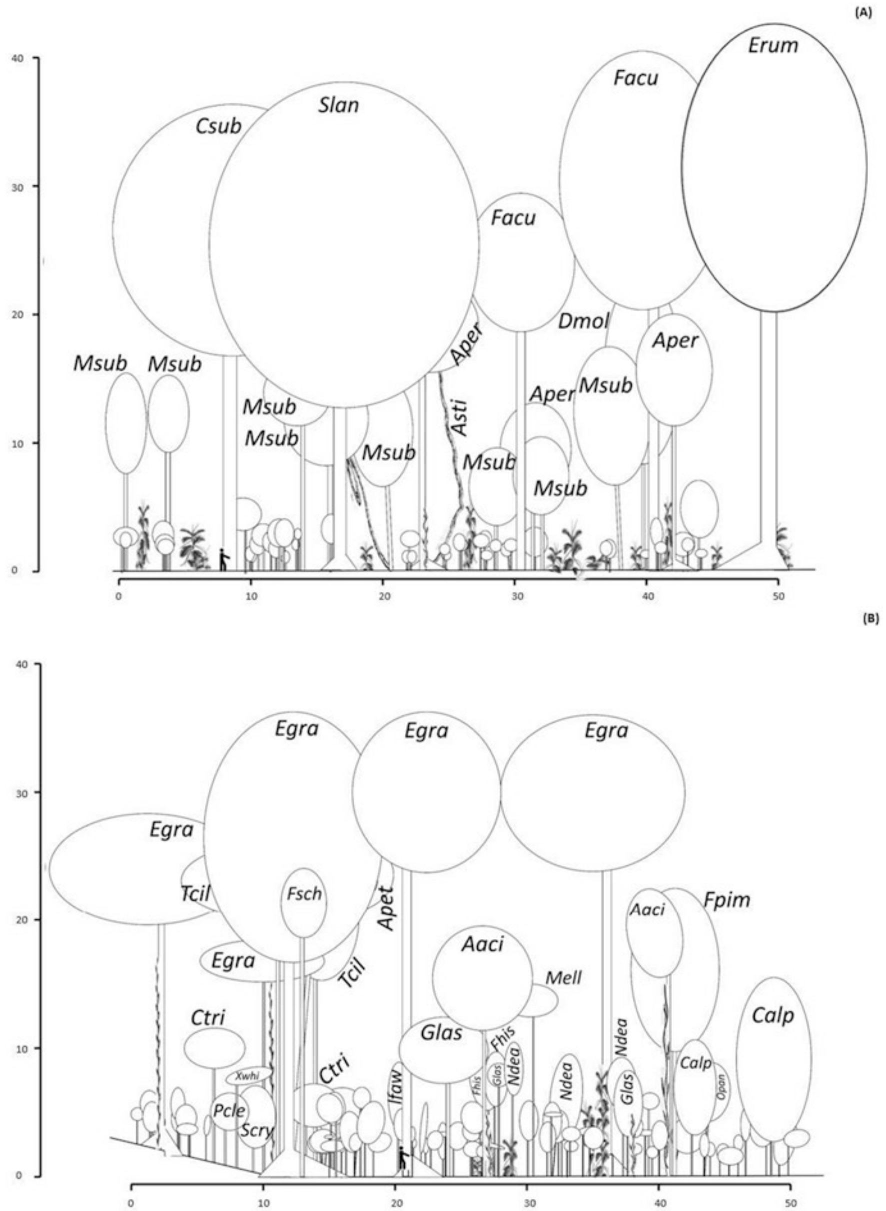

Fig. 3.10 Profile diagrams of (**a**) reference forest site at Barrine and (**b**) Donaghy's Corridor. Key: *Aaci Acronychia acidula, Aper Argyrodendron peralatum, Apet Alphitonia petriei, Asti Austrosteenisia stipularis, Calp Castanospora alphandii, Csub Cardwellia sublimis, Ctri Cryptocarya triplinervis, Dmol Dysoxylum mollissimum*, *Egra Elaeocarpus grandis*, *Erum E. ruminatus*, *Fhis Ficus hispida*, *Fpim Flindersia pimenteliana*, *Fsch Flindersia schottiana*, *Glas Guioa lasioneura*, *Lfaw Litsea fawcettiana*, *Mell Melicope elleryana*, *Msub Macaranga subdentata*, *Ndea Neolitsea dealbata*, *Opan Olea paniculata*, *Pcle Phaleria clerodendron*, *Scry Syzygium cryptophlebium*, *Slan Sloanea langii*, *Tcil Toona ciliata*; *Xwhi Xanthostemon whitei*

Whilst figbirds, pigeons and channel-billed cuckoos are characteristically nomadic, other birds such as grey-headed robins and black-eared catbirds are confined to smaller territories, suggesting that the corridor contained resources that are used by both sedentary and wider-ranging species. Similarly, white-tailed rats are large and highly mobile rainforest rodents and have been recorded moving through, and residing within, the corridor after 3 years (Tucker & Simmons, 2009). Remote camera surveys in 2021 recorded white-tailed rats throughout the corridor. Dispersal of large fruits from within and outside the plantings confirmed their dispersal abilities and suggested that for this species, restored vegetation is used as both habitat and a movement conduit. In a study of bush rats (*Rattus fuscipes*), genetic exchange occurred in the corridor within 3 years (Paetkau et al., 2009), demonstrating that the corridor helped to overcome prior genetic isolation in the Barrine fragment.

Socio-economic Context

Locally, high land prices generally preclude the availability of large areas of cleared land for restoration by any method, especially in productive agricultural areas where native vegetation cover is very low and connectivity is most needed. Conversion of agricultural land to rainforest therefore requires targeted use of private lands to create such corridors, carried out with considerable community support. Donaghy's Corridor typifies this situation since close contact and open negotiation with the Donaghy family (the main landholder) were the key factors influencing project outcomes. In this instance, the landholder wished to increase shade cover for grazing cattle to reduce heat loads during humid summer months (Lees et al., 2019). The establishment of the corridor vegetation provided significant shade, but a 3-row shelter-belt of hoop pine (*Araucaria cunninghamiana*) was additionally established outside the corridor to supply extra shade and also to provide an additional source of farm income. Hoop pine is an indigenous species commonly established in commercial timber plantations, and thus has significant value. Most hoop pines are now of equal height to corridor canopy vegetation, and the rows are favoured resting areas for stock (Fig. 3.11).

Community volunteers raised all seedlings and completed all plantings to establish the corridor; funding for fencing and off-stream stock watering points was provided by State and Commonwealth agencies. Initial monitoring was done by scientists and staff from the Queensland Parks and Wildlife Service's Lake Eacham Nursery and a number of academic institutions, which demonstrated a cooperative effort across a range of stakeholders. Of particular importance is the community expectation that such public investment on private land was protected from future disturbance. Stakeholder engagement was therefore critical from conception to completion, and when Donaghy's Corridor was ultimately protected under a Nature Refuge Agreement, it was provided with the same level of legislative protection as the adjacent National Park, thus securing its long-term future.

Fig. 3.11 Donaghy's Corridor joins Lake Barrine (foreground) to Wooroonooran. Note the row plantings of hoop pine (*Araucaria cunninghamiana*) outside the fenced corridor area. (Photo: T. Holt)

Reflection and Key Summary Points

Whilst these foregoing case studies are from three continents with markedly different social, economic and cultural settings, some common unifying threads are evident. Below we discuss six key points, derived from these threads, which we consider essential to the success of any tropical rainforest restoration project. Although it is obvious that no two restoration sites are the same, we feel that most projects would benefit by incorporating these general concepts into their project planning and implementation.

Prioritize Protection of Existing Forest

Despite the importance of tropical forests for conserving biodiversity, sequestering carbon, maintaining hydrologic cycling and supporting human wellbeing, rapid tropical deforestation continues with forest losses exceeding gains in many regions

(Sloan et al., 2019). These three case studies clearly illustrate that although active restoration can accelerate tropical forest recovery, it is impossible to precisely recreate the diverse forests that were originally cleared. Substantial differences in species composition between the restoration sites and their respective reference forest ecosystems remained for all the case studies, (even after 15–26 years) and significant regeneration input. In particular, late-successional and large-seeded tree species were poorly represented. Hence, it is clear that the first priority must be to protect existing forests (Brancalion & Holl, 2020; Di Sacco et al., 2021) which means that restoration practitioners must accurately address the most important drivers of forest loss and degradation, which vary greatly depending on the socioeconomic and political context.

Preventing forest clearance and sensitively managing existing forest fragments are the most cost-effective forest conservation strategies. Moreover, undamaged existing forests provide significant contemporary benefits. Recovering habitats take many years for biomass to accumulate and for biodiversity to recover to the point of yielding substantial ecosystems services and forest products (Moreno-Mateos et al., 2017). Finally, our case studies illustrate that even small fragments of forest in agricultural landscapes, if managed well, can serve as reference ecosystems for establishing restoration goals and are therefore important biological reservoirs for the recolonization of restoration sites.

Match Management with Degradation Level

The intensity of degradation, the distance to remnant forests and the availability of seed dispersers are issues which are directly correlated with the nature of the restoration approach. Elliott et al. (2013) outlined five stages of forest degradation for which levels of restoration intensity and cost correspondingly increase (Fig. 3.12 and Table 3.2).

Degradation Stage 1 is exemplified by selective logging. In such cases, sources of natural regeneration at a site remain varied and dense. If the site is protected from agricultural and exotic vegetation encroachment, wildfire and livestock grazing, it is likely the forest will recover without any further intervention. This is known as spontaneous or natural regeneration (Chazdon & Guariguata, 2016).

Degradation Stage 2 is similar, except that tree removal has been more intense, and reduced canopy cover allows weeds to colonise and suppress regeneration. In consequence, and in addition to the protective measures described above, other interventions are needed to tip the competitive balance in favour of regeneration, including weed control and fertiliser application. This is known as assisted natural regeneration (ANR) (FAO, 2019), and at Stages 1 and 2, the density of regenerating woody plants is sufficient to rapidly close the canopy, usually within 2–3 years.

Degradation Stage 3 occurs where sapling density falls below that needed to achieve canopy closure within a reasonable desired time frame. At this point, protective measures and ANR must be complemented by tree planting, with obvious

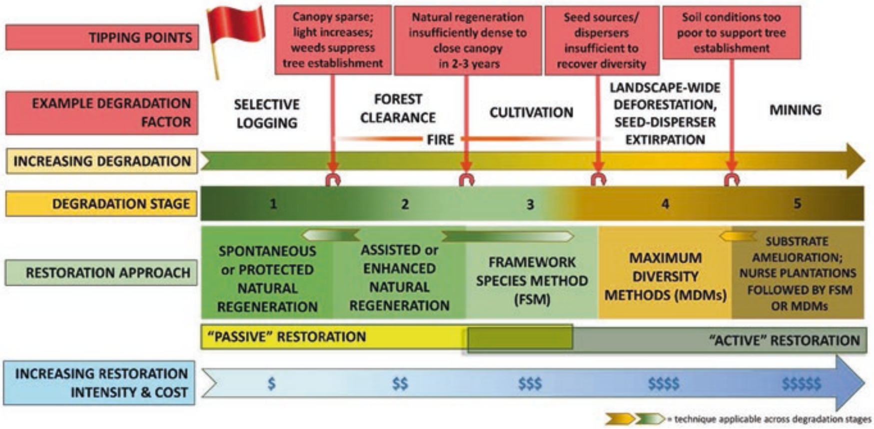

Fig. 3.12 Matching restoration approach with level of degradation. (Adapted from Elliott et al., 2013)

Table 3.2 Examples of forest ecosystem restoration implementation costs arranged from least to most degraded site conditions

Degradation Stage[a]	Restoration method	Country	Costs (US$/ha)[b]	Note
Stage 1	Spontaneous/ protected natural regeneration	Brazil	51	NR without fences (Brancalion et al., 2019)
		Thailand	340–395	Fire breaks, patrols & suppression
Stage 2	Assisted/enhanced Natural regeneration, ANR	Malaysia	82–117	Vine cutting, selective liberation of economic species. Degraded forest (Ong, 2011)
		Brazil	360	Assisted natural regeneration (Brancalion et al., 2019)
		Philippines	715	Fire prevention, weed pressing. 500 regenerants/ha. Open weedy sites (Bagong Pagasa Foundation, 2011)
		Cambodia	985	Fire prevention, vine cutting. 6950 regenerants/ha. Dense scrub (FAO, 2014)
		Thailand	2090	Fire prevention, ring-weeding. 974 regenerants per ha. Open weedy sites (FAO, 2014)
		Lao PDR	2135	Fire prevention, vine cutting. 5000 regenerants/ha. Dense scrub (FAO, 2014)
		Thailand	2276	Fire prevention, weeding, fertilizer application & monitoring. >3100 regenerants/ha. Open, weedy sites (case study 3)

(continued)

Table 3.2 (continued)

Degradation Stage[a]	Restoration method	Country	Costs (US$/ha)[b]	Note
Stage 3	Framework species Method, FSM	Brazil	825	Enrichment planting (Brancalion et al., 2019)
		Indonesia	880	Planting 400 trees/ha (Swinfield et al., 2016)
		Thailand	2276–5700	FORRU-CMU current costs. Planting (up to 3100 trees/ha), weeding, fertilizer, fire prevention, monitoring (case study 3)
		Australia	8720–12,280	Termed 'enhancement'. Planting with weed control (Catterall & Harrison, 2006)
Stage 4	Maximum diversity Method MDM	Brazil	821–1706	Direct seeding. 5000 trees/ha. 57 species (Raupp et al., 2020)
			2436	Seedling planting (Brancalion et al., 2019)
			3976	Tree planting. 2500 trees/ha. 57 species (Raupp et al., 2020)
			4350	80–100 species 2500 trees/ha, with deep ripping, added top soil on bauxite mine (Parrotta et al., 1997)
		Thailand	11,030	High density, 43 tree species, with some substrate amelioration (Miyawaki method) (Toyata pers. comm.)
		Australia	17,550–26,280	Termed "reinstatement". High density and diversity of native rainforest tree seedlings (Catterall & Harrison, 2006)
Stage 5	Site amelioration/nurse plantation, then FMS or MDM, as appropriate	Thailand	15,970	Rehabilitation of open cast limestone quarry. Site amelioration + framework species method. 3100 trees/ha (Siam cement group, pers. comm.)

[a] Elliott et al. (2013)
[b]Adjusted for inflation to 2021 values

cost increases, as seed collection programs and tree nurseries become necessary (Table 3.2). Where restoration sites are close to forest remnants, the framework species method works well. Framework tree species may be planted to complement ANR in small nuclei (case study 1), in larger plots (case study 2) or to form wildlife corridors (case study 3), depending on local ecological and economic conditions. They are selected specifically to enhance regeneration through weed suppression and animal seed-dispersal from nearby intact forest (Fig. 3.12).

Degradation Stage 4 occurs when seed-dispersal at the landscape level is insufficient to achieve acceptable rates of regeneration because forest remnants are too distant, seed-dispersing animals have been extirpated or ecological connectivity is required at more rapid temporal scales. Under such conditions, forest ecosystem restoration can only be achieved by planting most of the characteristic tree species of the reference ecosystem. This is the "maximum diversity" approach to forest restoration discussed in case study 3.

Degradation Stage 5 is reached when soil and microclimatic conditions have deteriorated beyond the point at which tree seedlings can establish without substrate amelioration. This is characteristic of open cut mined surfaces. Necessary procedures to improve the substrate's physical structure can include topsoil addition, deep ripping and mounding to improve drainage and aeration. Adding fertilizer, organic materials and green mulching can improve nutrient status and promote recovery of soil fauna and microbiota (Sansupa et al., 2021). Planting *Ficus* spp. and legumes as nurse trees can also improve soil structure and nutrient status respectively. Once the soil conditions have been improved, applied nucleation, the framework species method, or maximum diversity approaches can be implemented, depending on distances to seed sources and disperser availability.

Determining which level of degradation has been reached need not be complicated. A rapid site-assessment protocol is available, using simple participatory techniques to measure pre-existing natural regeneration, weed cover and soil conditions to guide stakeholders towards the most appropriate restoration strategy (Elliott et al., 2013). To assist in this work, several online tools are now available to advise on species selection for sites at Degradation Stages 3–5 that require tree planting (Fremout et al., 2022).

Encourage Dispersal

Conservation of large frugivores is essential for effective seed dispersal, just as seed dispersal is crucial to maintain diversity, connectivity and colonisation. Since large frugivores depend on mature forest, conserving this forest is a necessary precursor to maintain their dispersal services. Dispersal is also conditional on the configuration and composition of remnant forest patches and individual trees across the landscape and the behavioural responses of different dispersers (González-Varo et al., 2017). Many large frugivores do not cross open areas between forest patches. Moreover, species such as primates, tapirs, fruit bats, hornbills and cassowaries are rare, threatened or in decline throughout the tropics and this has myriad negative effects (Galetti et al., 2013; Boissier et al., 2020). In this context, restoration potentially plays a dual role.

First, by re-establishing habitat islands between fragments, it can enhance the mobility of large frugivores and the likelihood of maintaining dispersal at the landscape scale. This is the so-called stepping stone concept. The case studies in this chapter demonstrate the catalytic effect of such habitat establishment and vertebrate-mediated dispersal on ecosystem recovery at a range of scales. Clearly, where and how habitats are restored will depend on site- and species-specific parameters and objectives (McDonald et al. Chap. 7, this volume) but site-patch-to-remnant proximity, and the size and composition of both remnant and restored sites are additional key factors affecting dispersal success (Zahawi et al., 2021).

Second, habitat restoration provides additional resources that sustain frugivore populations and increases the likelihood of their persistence. Maintaining frugivore populations and the dispersal services they provide is essential to restore the structural complexity, species diversity and ecological functioning that typify mature tropical forests. As such, a more heterogeneous and ecologically connected landscape favours large frugivore persistence and the probability that dispersal will continue to aid natural development of functionality and resilience in restored areas. Restored forest may not closely resemble intact forest for decades or centuries, but these case studies show that strategic placement of restoration sites, as well as their species composition, can rapidly encourage effective dispersal across landscapes (Fig. 3.13).

Selecting which tree species to plant should be based on previously recorded performance, or on the functional traits that predict performance, to maximise ecological and social benefits (Rodrigues et al., 2009; Meli et al., 2014). Survival of planted trees is paramount, and using local species from the reference ecosystem confers the benefits of local adaptation. Many rainforest species, including shade-tolerant late-successional species, grow well in open, degraded sites. This ecological plasticity allows for direct establishment of late-successional species, circumventing existing barriers to establishment and the time lag associated with natural seed dispersal. Where degradation is severe, Leguminosae or other N-fixing groups should be planted to improve soil condition and fertility. This includes fast-growing species to shade out weeds, since it is key to reducing competition with newly established trees.

Because seed dispersal is crucial, selecting species that attract seed-dispersing wildlife is immensely beneficial. Fleshy fruits or arillate seeds with a 3–10-mm diameter attract many bird species with various gape sizes. Pioneer trees which fruit within a few years of plantings are important in this regard (Camargo et al., 2020). Furthermore, their early mortality (often within 20–30 years) creates light gaps and provides coarse woody debris, both of which to habitat structure and biodiversity recovery.

Across the tropics, several plant families are consistently associated with frugivorous seed dispersal. Some of these are Annonaceae, Arecaceae, Burseraceae, Lauraceae, Moraceae, Sapotaceae and Sapindaceae. Species from these families are likely to attract many frugivore guilds. Similarly, including a suite of local *Ficus* increases food availability for frugivores during seasonal scarcity, contributing to the continuity of dispersal and regeneration throughout the year (Zahawi & Reid, 2018).

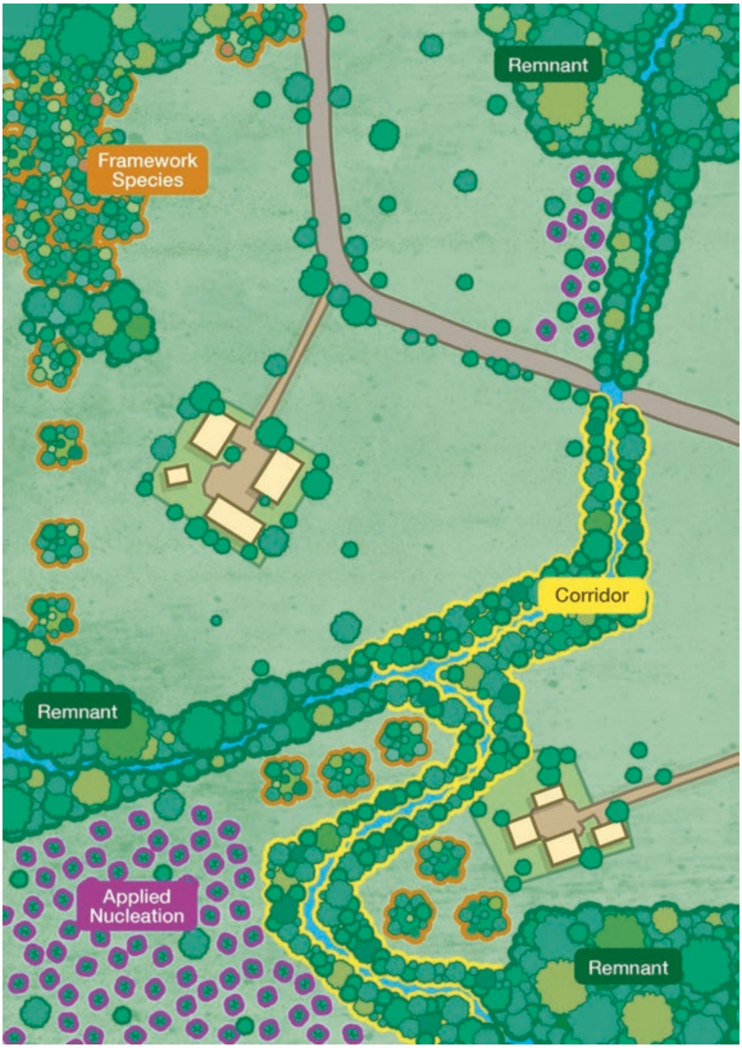

Fig. 3.13 Selection of restoration strategies tailored to local site conditions. The size, composition and location of remnant vegetation affect restoration strategy choice. In sites proximal to remnant forest (green outline) and some scattered trees, applied nucleation (purple outline) is an effective strategy to foster regeneration and recover large areas when it is compatible with stakeholder restoration goals. In larger open areas, frugivore-attracting framework species (brown outline) can be planted adjacent to remnant forest or as patches that form stepping stones between remnants. Reconnecting patches of remnant vegetation through corridors (gold outline) permits the flow of genetic material across the landscape. Using riparian zones to re-establish ecological connectivity confers additional benefits to soil stability and water quality. (Illustrator: Tim Parker)

Design Trials to Learn from Experience

These case studies demonstrate the value of using trials to assess the effectiveness of proposed restoration techniques and species choices locally, and draw attention to the considerable length of time needed for trials to yield sound advice. Therefore, attending to pre-existing knowledge is important for project initiation. Data from regular monitoring, accumulated as projects mature, is used later for 'adaptive management' – a central tenet of ecological restoration (Gilmour, 2007). International standards can provide broad guidance (Pedrini & Dixon, 2020), but surveys of reference forest and restoration sites involving all local stakeholders are essential to yield locally relevant information. Indigenous and local knowledge is invaluable for identifying the tree species that thrive on deforested sites, for locating seed trees, and for selecting species that local stakeholders value (Wangpakapattanawong et al., 2010).

We recommend that monitoring be carried out in three locations. These are: (i) the *origin or control* (part of the degraded site where no restoration interventions are applied), (ii) the *treatment* (where restoration interventions are applied) and (iii) the *target* (usually a nearby remnant of the reference ecosystem). Before any restoration interventions are applied, starting site conditions (baseline data) should be measured at permanent sampling points across all three locations (Viani et al., 2018), and measures should be repeated annually, at least until canopy closure. Comparing monitoring steps (i) and (ii) determines the effectiveness of restoration interventions relative to natural regeneration. Comparing steps (ii) and (iii) determines the extent of progress towards restoration goals and how restoration practices can be improved (Viani et al., 2018).

Variables recorded should relate to the fundamental restoration goals of maximizing the recovery of biomass, increasing forest structural complexity, recovering biodiversity and achieving sustained ecological functioning (Elliott et al., 2013). As we have previously indicated, these goals should also be consistent with social variables indicating improved human livelihoods (Viani et al., 2017). To establish a data bank, the size and condition of each tree should be recorded. Simple confidence limits can then be applied to estimate changes in tree density and size over time, with biomass and carbon accumulation derived from allometric equations (Pothong et al., 2021). For this task, drones now offer cost-effective and non-intrusive alternatives to conventional, labour-intensive field work to monitor tree survival and growth and canopy closure (de Almeida et al., 2020).

Encourage Stakeholder Participation Throughout the Restoration Process

Successful restoration depends on involving stakeholders at all stages, from planning and implementation to maintenance and monitoring (Mansourian & Vallauri, 2014; Holl, 2017). It is important to understand that restoration often fails because planted trees are not maintained, local people convert the land back to agricultural production, or less frequently, clear trees as a political protest (Brancalion & Holl, 2020). In order to avoids such pitfalls, the inclusion of all stakeholders (including those likely to legally or illegally use the land for other purposes) in the setting of project aims and its subsequent development, together with clarification of land tenure and usufruct rights, will increase the likelihood of long-term socio-economic sustainability of restoration projects (Guariguata & Brancalion, 2014; Chang & Andersson, 2019). These community stakeholders should be involved in planning to ensure that the project is transparently designed to address their needs and concerns. They should also be meaningfully engaged throughout the implementation, maintenance and monitoring phases of projects (Holl, 2020).

In some cases, restoration projects can be undertaken on publicly owned lands, but to meet the ambitious restoration targets of the Bonn Challenge and the UN Decade on Ecosystem Restoration, it is likely that most of them will have to occur on private lands. Since most landowners depend on income from their land, restoration projects must monetize the benefits of restoration. This may take the form of cash payments for environmental services to encourage landowner participation (Pirard et al., 2014). It also means that restoration projects must be designed to meet community needs, such as selecting tree species that will ultimately provide shade, timber, honey, firewood or other product with values for the community (Meli et al., 2014) or, alternatively, choosing a plantation-style planting design to accommodate landowner aesthetic preferences (Zahawi et al., 2014).

Equally important to successful restoration is incorporating local and indigenous knowledge into the project and ensuring that landowners are trained in best practices for restoration and site maintenance, to provide extra technical capacity. Moreover, giving landowners management responsibility over the project is a key to project success (Gregorio et al., 2020; Hagazi et al., 2020) and engaging stakeholders in participatory monitoring is a powerful way to encourage social learning and to promote adaptive management (Case study 2, Evans et al., 2018).

Our case studies focused largely on the ecological aspects of forest restoration, given our expertise as ecologists and our focus on tropical forest restoration for biodiversity conservation. We close by reiterating that achieving the ambitious forest restoration targets proposed internationally will require undertaking forest restoration for a range of reasons, including improvement of ecosystem functions and human livelihoods (Brancalion & Holl, 2020; Di Sacco et al., 2021). Key to the success of these efforts will be in (i) clearly stating the goals of specific projects, (ii) tailoring restoration approaches to be consistent with the stated goals and with local ecological and economic conditions, (iii) carefully monitoring whether goals have been achieved and, (iv) engaging stakeholder participation and support of the project (Brancalion & Holl, 2020).

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Chapter 4
Ecological Approaches to Forest Restoration: Lessons Learned from Tropical Wet Asia

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Summary and Key Lessons
In this chapter, we present examples of restoration methods that have successfully returned forest cover to a selected set of study sites in biodiversity hotspots of south and southeast Asia. These examples, which focus on ecological restoration of tropical lowland and lower montane rainforests, hold the promise of better-advised and more widespread application for similar landscapes in need of restoration.

In the mixed dipterocarp forest regions of the lowlands and hills of southwest Sri Lanka, we planted rainforest tree species across a range of site and shade conditions. This planting was beneath the canopy of a *Pinus caribaea* plantation, a non-native tree used widely for reforestation in the wet zone of Sri Lanka. Our results

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demonstrated that many native tree, shrub, liana, and herbaceous species of economic and conservation value performed better both in survival and growth under intermediate shade conditions than beneath a closed canopy pine plantation or in fully open habitats. We also demonstrated that, in the absence of fire, many rainforest pioneer and site generalist species can naturally recruit beneath pine plantations.

In both the lower montane forest of Western Ghats in India and the Knuckles region of Sri Lanka, we also demonstrated that planting mixtures of native tree species and actively controlling invasive species is a more effective approach to restoring forest cover than reliance on natural recruitment and release mechanisms in open areas. This result presumably reflects the fact that many rainforest trees of this forest type are dispersal-limited and sensitive to fire and grazing, which means that, for successful restoration, they must be planted and protected during their establishment.

The "Rainforestation" method, originally practiced in Leyte and now widespread throughout the mixed dipterocarp forest regions of the lowland and hills of the Philippines, also reveals the importance of planting as compared to natural regeneration. Here, the planting of eclectic mixtures of native trees on community and small private lands, based on owner preferences, has shown high survival rates combined with utility value to the landowners.

Lastly, in East Kalimantan, Borneo, within the same mixed dipterocarp forest type, the Samboja Lestari restoration project demonstrated that reforestation on *Imperata* grassland can succeed through mixed methods by (i) assisting natural regeneration where and when appropriate; (ii) using successional agroforestry, where crops are initially cultivated then relinquish their growing space to planted trees, and (iii) direct planting, at the same time, of both native and non-native trees.

All these approaches have demonstrated that successful restoration can be achieved across an assortment of differing socio-economic and ecological environments. These attempts range from specific site-based restoration treatments facilitating similar composition and structure to the original rainforest, such as in Sri Lanka and Western Ghats, to more general strategies which establish tree cover to support more general economic and conservation values as seen in the Philippines and east Kalimantan.

Statement of Implications

Our studies across wet environments in tropical Asia indicate that, in most cases, tree planting is necessary to overcome dispersal limitation and that high diversity mixed-native species plantings can lead to significant forest recovery. Late successional tree species of the original rainforest with conservation value typically need to be planted on appropriate sites and need some degree of partial shade and protection from fire and herbivory for their successful establishment. This can be achieved by planting both native site generalist and restricted tree species beneath a thinned canopy of native or naturalized non-native trees that are tolerant of fire, high light, and low nutrient soil conditions, acting as nurse tree stands. The plantings of non-native species of economic value mixed with native species can also be a viable reforestation option for enhancing local livelihoods. Finally, in some circumstances, rudimentary second growth forest can be established without active planting when fire and other disturbances such as herbivory are excluded.

General Introduction

The United Nations' Decade on Ecosystem Restoration (2021–2030) is emerging as a unified global strategy toward conserving threatened biological diversity, mitigating climate change, and curbing desertification. Enabled by the three international conventions, namely the Convention on Biological Diversity [UNCBD], the UN Framework Convention on Climate Change [UNFCCC], and the UN Convention to Combat Desertification [UNCCD], and enacted in conjunction with other multilateral agreements such as the Bonn Challenge and the New York Declaration on Forests (which aims to restore 350 million hectares of degraded landscapes by 2030), these calls to action have mobilized a level of political commitment and are acting as a potential accelerator of ecosystem restoration efforts around the world (https://wedocs.unep.org/bitstream/handle/20.500.11822/30919/UNDecade.pdf). Forest Landscape Restoration (FLR) has emerged as a key approach to fulfilling these formidable restoration goals, with the overall objective of achieving a more optimal balance between economic and ecological criteria for reversing deforestation and land degradation.

Of particular relevance to this chapter is the notion that concentrating restoration initiatives in regions that are biodiversity hotspots appears to provide an opportunity to make significant contributions toward the goals of the UN Decade on Ecosystem Restoration (https://forestdeclaration.org/goals/goal-5) whilst, at the same time, conserving native biological diversity. Today's 36 terrestrial biodiversity hotspots represent only 2.4% of earth's surface area (15.4% of its land area), yet they collectively harbor no less than 50% of the world's vascular plant species and nearly 43% of terrestrial vertebrate species, which are amphibians, birds, mammals, and reptiles (Hrdina & Romportl, 2017; Mittermeier et al., 2011). Hotspots by definition have lost at least 70% of their original habitat and many are threatened by continued deforestation and land conversion, making them some of the most endangered terrestrial ecoregions of the world (Brancalion et al., 2019; Cunningham & Beazley, 2018; Mittermeier et al., 2004).

The south and southeastern tropical Asian region is an important area for restoration because it is home to five global biodiversity hotspots, these being in Western Ghats and Sri Lanka, Indo-Burma, Sundaland, the Philippines, and Wallacea. A common feature across this broad region is an endemic-rich, hyper-diverse tropical rainforest flora that displays a degree of habitat specialization through niche partitioning and habitat filtering (Smith et al., 2018). Such spatial patterning of tree species at landscape level results from variations in topography and site conditions, such as hydrology, and soil conditions and external disturbances, which include drought, windfall, landslides, and temporary forest clearance for agriculture. These spatial patterning processes can play a significant role in shaping forest structure, species composition, and diversity as is evident from studies involving large-scale Forest Dynamics Plots. Such work has been done in Sinharaja (Sri Lanka), Lambir (Sarawak), and the Danum Valley (Sabah), Malaysian Borneo, Gunung Palung National Park (west Kalimantan in Indonesian Borneo), and, to a limited extent in Palanan, the Philippines, and Mudumalai in the Western Ghats, India (Co et al., 2006; Davies, 2001; Gunatilleke et al., 2006; Paoli et al., 2006; Pulla et al., 2017; Punchi-Manage et al., 2013; Russo et al., 2008; Smith et al., 2018; Webb & Peart, 2000; Yamada et al., 2007). It has also been noticed that habitat preferences are reinforced by species' differential demographic responses to supra-annual variations in climate such as droughts during El Niño events, which affects seedling recruitment, growth, and mortality of plant species (Yamada et al., 2007).

The forests of the south and southeast Asia region have suffered from widespread deforestation and forest degradation, making it a major target for forest landscape restoration. This chapter focuses on restoration approaches employed in five landscapes spread across three biodiversity hotspots of the region: two study sites in Sri Lanka, and one each in the Western Ghats of India, the Philippines, and Indonesia. The case studies include small-scale experimental plot trials in Sinharaja and the Knuckles regions in Sri Lanka, and landscape-level trials in Anamalai Hills in Western Ghats, Leyte Province in the Philippines, and the Samboja Lestari restoration project in East Kalimantan, Indonesia (Table 4.1). These case studies demonstrate that there are a variety of different approaches that can be undertaken to restore the forests, and these reflect the differing social and ecological contexts of the sites and management objectives of the restoration proponents. These restoration strategies can help such countries to achieve the significant restoration targets that they have set for themselves for the 2021–2030 period (Table 4.1).

Case Studies

Study 1: Sinharaja (Sri Lanka)

Background

Our forest restoration research in Sri Lanka was undertaken in the ever-wet Mixed Dipterocarp Forest (MDF) formation of the southwestern part of the island. The MDF formation is biologically the richest forest type in this region and has a strong biogeographic affinity with MDFs in Sundaland and the Philippines (Ashton, 2014). The MDF-dominated landscape in SW Sri Lanka exhibits a series of parallel hill ranges running in SE–NW direction with steep-sided V-shaped valleys (Erb, 1984, Annex 1a).

Table 4.1 Salient features of the selected case study regions in south and SE Asia

Restoration case study sites/region	**Sinharaja World Heritage Site (WHS), Sri Lanka**	**Knuckles Forest (Central Highlands WHS), Sri Lanka**	**Valparai Plateau/ Anamalai Hills, Western Ghats, India**	**Leyte Province, Philippines**	**Samboja Lestari study site/ Balikpapan, Indonesia**
Climate of the case study region	Perhumid/Everwet tropical	Perhumid/Everwet tropical	Humid/wet tropical	Perhumid/ Everwet tropical	Perhumid/ Everwet tropical
Reference forest type	Lowland Mixed Dipterocarp Forest	Lower Montane Mixed Species Evergreen Forest	Mid-elevation Tropical Wet Evergreen Forest	Lowland Mixed Dipterocarp Forest	Lowland Mixed Dipterocarp Forest
Elevational range at study sites (m)	400–500	1058–1157	800–1350	80–140	10–100
Mean annual rainfall (mm)	5016	4830	2400	2400	2250
Major soil type	Ultisols/Red Yellow Podzols	Ultisols and Mountain Regosols	Alfisols	Andisols	Acrisols
Land extent of the case study	03 ha demonstration site, later replicated in two other sites	2.5 ha Demonstration site, Later extended to 5 ha	100 ha (actively restored) 1075 ha (passively restored)	Initially a 2.4 ha site, later extended to 28 small demonstration sites	1850 ha
Year of initiation of the project	1991	2003	2001	1992	1999
Forest restoration/rehabilitation method/s used	Relay Floristic Successional Method using *Pinus caribaea* as nurse trees	Succession-based mixed species planting using native and naturalized species as nurse trees	Maximum diversity mixed native species planting protocol with weed removal/passive natural regeneration	Rainforestation Farming – Native species with food crop species	Assisted natural regeneration/ Agroforestry/ Buffer zone restoration
National restoration targets/ pledges	**Sri Lanka**		**India**	**Philippines**	**Indonesia**
Nationally determined contribution/ Bonn challenge pledges for 2021–2030[a]	200,000 ha		58,433,270 ha	7,250,000 ha	12,000,000 ha
Population Density (persons/km^2)	**341**		**464**	**368**	**151**

[a]https://www.pbl.nl/en/publications/goals-and-commitments-for-the-restoration-decade

Large areas of the forest have been lost to small landholder cultivation, with tea and other plantation crops that have subsequently been abandoned. These areas have been colonized by shade-intolerant and fast-growing grasses, shrubs, vines, and ferns. Seed sources are available within the surrounding forest landscape, but natural regeneration of late successional tree species has been slow (Ashton et al., 2001a).

As elsewhere in the humid tropics, efforts by the Forest Department of Sri Lanka since the mid-twentieth century to establish native forest plantations on degraded areas have met with little success (Vivekanandan, 1989). The primary reason for the low success of reforestation is that the species selected for planting were late successional canopy species, which are ecologically ill-adapted to establishment on open, eroded, nutrient-poor, and fire-prone sites. In the 1960–1990s, the forest department switched to large-scale planting of introduced trees such as *Pinus caribaea* as single species plantations that performed better under these conditions. The Forest Department was successful in establishing around 15,000–18,000 ha of mature *P. caribaea* plantations in the lowland districts of southwest Sri Lanka by the turn of the twentieth century (Bandaratilake, 1989). However, these monoculture plantations which clothe the hill crests and upper slopes came under frequent criticisms from the environmentalists and local village communities for (i) supporting frequent anthropogenic fires during dry periods, followed by (ii) heavy soil erosion and landslides during monsoonal rains, leading to (iii) poor regeneration of native species, (iv) reduced groundwater recharge and rapid drying up of water courses during the dry seasons, particularly during *El Nino* years, and (v) lack of tangible benefits to local communities who have been traditionally dependent on a range of timber and non-timber products and other ecosystem services from natural forests (Perera, 1989).

Rationale and Goals

We made use of basic socio-ecological findings gathered since the 1970s from the natural reference forest of Sinharaja, a UNESCO World Heritage Site (WHS) and International Biosphere Reserve (IBR), to experimentally manipulate the canopy of a *Pinus caribaea* nurse tree plantation in its buffer zone (Ashton et al. 2001a, b, 2014). The primary goal was to facilitate transformation of the buffer zone to a mixture of trees aimed at meeting both conservation goals and the livelihood needs of the local people, particularly in respect of their traditional artisanal and dietary use of forest products for basketry, health food, medicines, and beverages.

To achieve this goal, our study had two objectives: (i) to identify the optimal environmental conditions within *Pinus* plantations which would encourage establishment of native rainforest tree species based on their known functional traits and habitat affinities (Figs. 4.1 and 4.2) and (ii) to evaluate growth and yield of species producing timber and non-timber forest products of rural economic value.

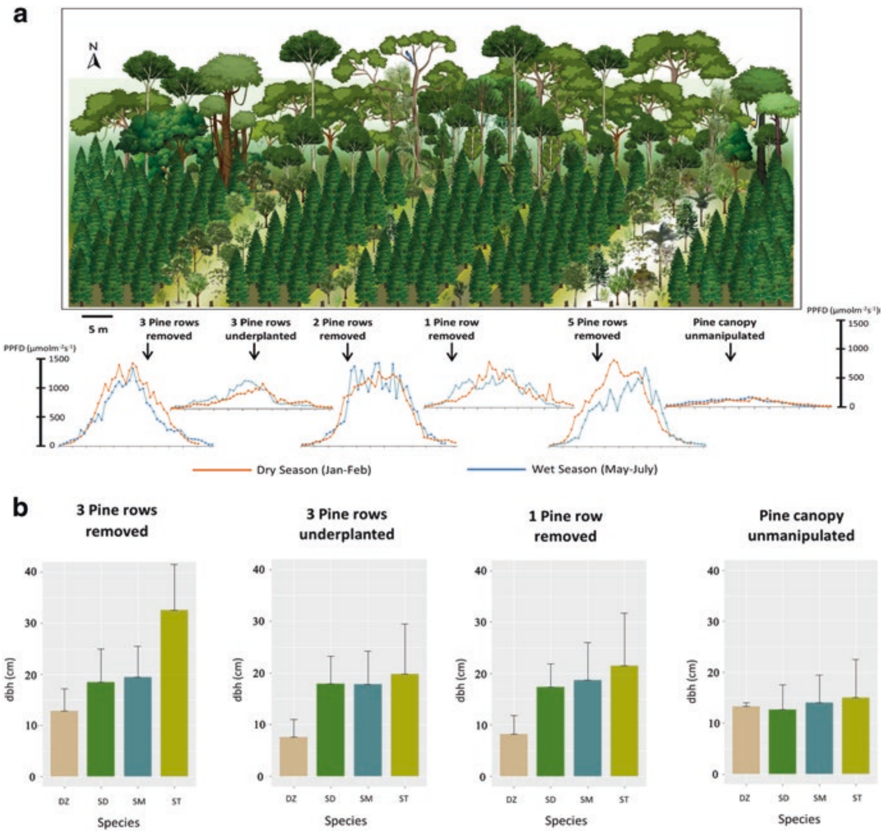

Fig. 4.1 (**a**) Experimental design for introducing site generalist and restricted native and naturalized species of utility value under the manipulated nurse canopy of a *Pinus caribaea* plantation creating different diurnal light regimes, in the buffer zone of the Sinharaja MAB reserve, Sri Lanka. The daily average Photosynthetic Photon Flux Density (PPFD) measured from 09:00 to 15:00 above the canopy of the planted saplings (12 years after establishment) during wet and dry seasons is plotted beneath each pine canopy removal treatment. (**b**) Diameter growth performance of four late successional native tree species under different pine canopy manipulated treatments, after 31 years (1991–2022). *DZ Dipterocarpus zeylanicus, SD Shorea disticha, SM Shorea megistophylla, ST* Shorea *trapezifolia* (all of the family Dipterocarpaceae and endemic to Sri Lanka.)

Key Strategies Used

First, we conducted comparative floristic surveys of the (i) reference natural forest, (ii) selectively logged forests, and (iii) *Pinus* plantations and fernlands of the buffer zone of the Sinharaja WHS/IBR to select candidate species for restoration (De Zoysa et al., 1991; Gunatilleke & Gunatilleke, 1985; Shibayama et al., 2006; Tomimura et al., 2012).

Fig. 4.2 Three pine-row removal treatments creating canopy gaps (width 10 m) along a N-S direction and planting of native species of utility value. This photographic sequence shows the growth performance of site generalist- and restricted species planted in pine canopy gaps over a period of 31 years

Second, seedling ecophysiology of over 50 tree species was studied using shade house experiments and plantings in natural forest canopy gaps to determine their survival, growth, and site adaptations (Ashton & Gunatilleke 1995; Ashton et al. 2001a, 2006, 2011, 2014; Ediriweeera et al., 2008; Goodale et al., 2012; Gunatilleke et al., 1998).

Third, species distributional patterns in the 25 ha Sinharaja CTFS ForestGEO Plot, together with another set of 100 plots (totaling 25 ha) yielded information on tree community structure and habitat affinities of about 140 species across the ridge-slope-valley landscapes that are typical of lowland MDF forests of SW Sri Lanka (Annex 1a). These patterns were indicative of 'site–species matching' (Gunatilleke and Gunatilleke 1985; Gunatilleke et al. 2004, 2005a, 2006; Ashton et al. 2011, 2014; Punchi-Manage et al. 2013).

Fourth, we conducted a series of genetic diversity studies in the reference forest (The Sinharaja Rainforest Complex) which estimated the rates of gene flow and the degree of genetic differentiation among populations of selected canopy tree species (Gunatilleke & Gunatilleke, 2013).

Fifth, a series of socio-economic studies were carried out on the use of non-timber forest resources by villagers living around the Sinharaja WHS with a view to incorporate these species in restoration programs (De Zoysa, 1992; Everett, 1995; McDermott et al., 1990).

Project Management

Using the data thus generated from previous studies, researchers from the Universities of Peradeniya and Sri Jayewardenepura (Sri Lanka), and Yale University (USA), with logistic support from the Forest Department of Sri Lanka, started an initial experiment in 1991 to transform an 11-year-old *P. caribaea* plantation in a sloping landscape typical of this region into a native forest. Treatments included canopy openings that varied in size arranged along a slope in order to reflect a range of light conditions found in the rainforest (Fig. 4.1a).

Transplanted species included the natural forest canopy dominant species *Dipterocarpus zeylanicus*, *Shorea megistophylla*, *S. trapezifolia* and *S. disticha* (all of Dipterocarpaceae), and *Mesua ferrea* (now *M. thwaitesii* of Calophyllaceae). These species associated with valleys, slopes, and ridges were selected as framework native species for this landscape investigation (Gunatilleke et al., 2005b). In addition, the native utility tree species *S. stipularis* (Dipterocarpaceae), *Diospyros quaesita* (Ebenaceae), *Pericopsis mooniana* (Fabaceae), and *Caryota urens* (Arecaceae) each having a strong affinity to a site along the valley-mid slope-ridge continuum were also selected for planting (Ashton et al., 1997). The naturalized non-native site generalist species *Swietenia macrophylla* (Meliaceae), a much sought-after timber species among small landholders in this region, was also included for comparison.

Moreover included in this experiment were *Coscinium fenestratum* (Menispermaceae) a medicinal vine, *Arundina graminifolia* (Orchidaceae) an ornamental ground orchid, *Calamus ovoideus* (Arecaceae) rattan, and *Elettaria cardamomum* var. *majo*r (Zingiberaceae) which is wild cardamom. These non-timber forest species are of significant local economic value. The experiment was established as a two-factor (light, species) factorial design comprising three replicates for each treatment (Ashton et al. 1997, 1998, 2001a, b, 2014). All planting materials, whether raised in local nurseries and as wildlings (non-timber species), were sourced from local provenances giving due consideration to their spatial genetic structuring (Murawski et al., 1994a, b; Stacy, 2001). Growth rates of tree species were monitored over 12 years and that of non-timber species over 9 years (Gunatilleke et al., 2005b; Kathriarachchi et al., 2004) (Fig. 4.2) and statistically analyzed to estimate the effects of opening size and slope (Ashton et al., 1997; Gunatilleke et al., 2005b). Diameter growth of four canopy-dominant species after 31 years (using the most recent census data collected in 2022) is given in Fig. 4.1b.

It was found that all tree species performed better in canopy openings than either in the closed canopy or fully open conditions outside of the pine plantation. By year 12, the best performing mid-slope specialist, *S. trapezifolia,* which is a relatively shade-intolerant species, had grown to 14 m in height in the three pine rows removed canopy opening treatment providing 50% of the full sunlight environment (Ashton et al., 2014; Fig. 4.2). This was almost the same height as the *P. caribaea* trees in the surrounding stand. *Mesua thwaitesii,* a shade-tolerant and stream-associated species, showed about half the growth rate (both height and diameter) of the best-performing Dipterocarps across all canopy opening treatments (Ashton et al., 1997, 1998, 2014; Gunatilleke et al. 2005b). The 2022 census data provided further evidence of nuanced diameter growth performance (Fig. 4.1b). Among the NTFP species, the shade-intolerant rattan, the orchid, and the medicinal vine, all of which are forest fringe specialists, grew better within openings than beneath the canopy of the *Pinus*. However, wild cardamom, a shade-loving understory herb, grew better in both partial and full shade conditions (Gunatilleke et al., 2005b; Kathriarachchi et al., 2004).

Integration of population genetic parameters of species is an important element in the design and implementation of restoration projects (Nef et al., 2021). Studies involving species in different geographical ranges and disturbance regimes in the Sinharaja rainforest complex have indicated a strong potential for biparental inbreeding depression within forest tree populations and for partial reproductive isolation. These conditions have led to outbreeding depression among fragmented populations across the landscape. The optimal outcrossing for two canopy species of MDFs examined occurred over a range between one to several kilometers (Stacy, 2001; Stacy et al., 2001). Selective logging resulted in elevated levels of inbreeding in canopy dipterocarp species (Murawski et al., 1994a, b). Furthermore, genetic diversity studies carried out in 10 subpopulations of the canopy species *Shorea trapezifolia* along the altitudinal range from west to eastern Sinharaja rainforest complex has shown that small forest fragments have already begun to show genetic differentiation due to limited gene flow as a result of long-term isolation leading to genetic drift (Dayanandan, 1996; Gunatilleke & Gunatilleke, 2013). These findings underscore the importance of integrating intraspecific genetic information in restoration planning, including provenance-based seed sourcing for rainforest restoration projects and setting benchmarks in genetic differentiation over time (Gunatilleke & Gunatilleke, 2013).

Challenges

A major challenge was communicating the message to forestry officials and policy makers that there are socio-ecological benefits in establishing mixed-species plantations comprising mostly native trees as opposed to plantations of *P. caribaea* in critical watersheds and sites close to protected forests. Foresters are generally trained in plantation silviculture, and therefore convincing them how a mixed species native forest plantation could be established as an alternative in this degraded landscape is a problem that needs to be addressed (Gunasena et al., 1989).

Indeed, there is a dearth of sound ecological and silvicultural knowledge on the native tree species for purposeful reforestation, and this information had to be obtained before wider communication could be achieved with potential partners. In addition, invasion of exotic weeds, fire, and damage from wild animals to planted seedlings, particularly wild cardamom and sugar palm, had to be successfully controlled during the initial period with harmonious assistance from local communities.

Major Outcomes

This native species planting trial for conservation and utility value at Sinharaja now serves as a demonstration site to promote its replication and upscaling. The ecological, ecophysiological, and population genetic information on tree species made available from prior studies has been successfully incorporated into site-species guides for propagation and planting using *P. caribaea* plantations as a facilitatory successional mechanism (Ashton et al., 2011, 2014; Gunatilleke et al., 2006; Gunatilleke & Gunatilleke, 2013; Punchi-Manage et al., 2013).

Incorporating the lessons learnt from this site, two other restoration trails have been established in the lowland rainforest region of Sri Lanka (Geekiyanage et al., 2021; Jayawardhane & Gunaratne, 2020). In one of these trials, a biological corridor linking two rainforest fragments in the Greater Sinharaja Rainforest complex is being established in partnership with Dilmah Ceylon Tea Company PLC, a leading private sector company (https://www.dilmahconservation.org/initiatives/sustainability/biodiversity-corridor-endana.html). Of great assistance here was that a grant from the Fondation Franklinia to improve the conservation status of globally threatened rainforest tree species listed on the IUCN Red List was awarded to continue our ongoing studies (https://fondationfranklinia.org/en/conservation-rainforest-southwest-sri-lanka/).

Key Learnings

Establishment of fast-growing generalist species, such as *P. caribaea* in degraded landscapes, can catalyze active forest landscape restoration across the variable topography which is so typical of SW Sri Lanka through appropriate light manipulation of nurse stands. The Forest Department of Sri Lanka is now working with relevant stakeholders to convert *P. caribaea* plantations into broadleaf mixed-species stands in critical watersheds elsewhere in this climatic region.

We note that most of the late successional species are site-specific in the ridge-slope-valley landscape which is typical of SW Sri Lanka (Annex 1a), necessitating an understanding of their ecology to ensure site–species matching for planting and tending.

Successful introduction of several non-timber forest species of socio-economic value into the mixed species tree plantings in the buffer zone restoration project has laid the foundation to replicate this protocol through community participation in both buffer and transition zones of protected areas. At least two such studies using lessons learned have been applied elsewhere in SW Sri Lanka (Geekiyanage et al., 2021; Jayawardhane & Gunaratne, 2020).

Alstonia macrophylla (Apocynaceae), another non-native and naturalized tree species with invasive tendencies and, at the same time, of considerable timber value to rural communities has the potential to be used in a similar manner. Some native species have already established under their shade and judicious manipulation of the canopy of these *A. macrophylla* trees with community participation (taking a cue from the present *P. caribaea* study) could lead to the scaling up of the establishment of mixed native species forest stands.

Study 2: Knuckles (Sri Lanka)

Background

The lower montane rainforests of the Knuckles region are a northern extension of the Central Highland massif (Annex 1b). Constituting less than 1% of Sri Lanka's forests (Premakantha et al., 2021), these forests are critical for safeguarding biodiversity and ecosystem services such as soil conservation, carbon sequestration, and the provision of water for hydro-electricity generation and downstream agriculture. Presently, lower montane forests in the Knuckles Conservation Forest (KCF) are located as patches in a mosaic of other land uses that include tea and non-native tree plantations, grasslands, and scrublands (Fig. 4.3a). Annual burns that are frequent in the grasslands during the dry season have led to reduced carbon stocks, increased soil erosion, and downstream flash-flooding in response to extreme weather events. The region has similar landscape heterogeneity, physiognomy, and associated socio-ecological issues to those of Anamalai forests of the Western Ghats, India (Gunaratne et al., 2014; Muthuramkumar et al., 2006).

Rationale and Goals

The project aimed to identify ecologically and socially acceptable restoration prescriptions to accelerate natural forest recovery in the degraded grasslands of the Knuckles Conservation Forest.

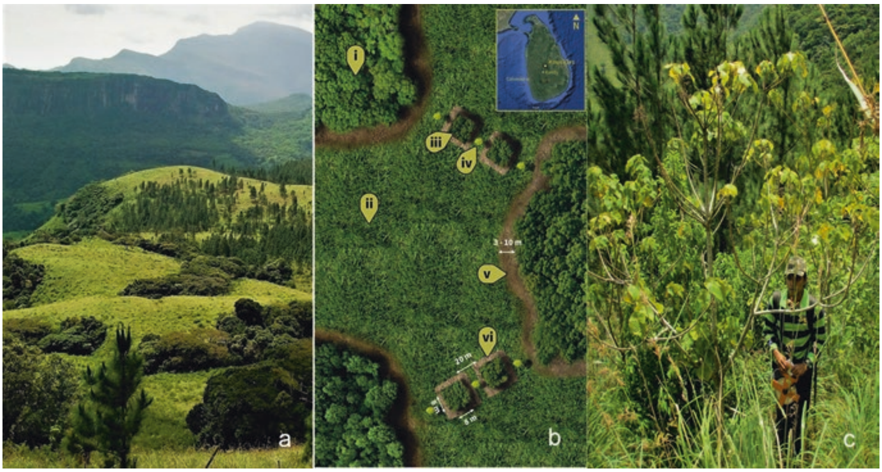

Fig. 4.3 (**a**) Landscape mosaic on eastern slopes of KCF; (**b**) Restoration model for KCF (i): Protection of remnant forest fragments (ii): Degraded grasslands (iii): Establishment of tree islands with *Gliricidia sepium* as a nurse plant and early successional native tree species with application of biofilmed biofertilizer (iv): Early successional tree/shrub species (v) Grass removal and tilled up to 10 cm at forest edges (vi): Fire belts (2 m) (Designed by Risiru Hemage); and (**c**) Well-grown *Macaranga indica* tree in a tree island (8 × 8 m) established in grasslands 4 years after establishment

Key Strategies Used

The project was conducted in four phases. Phase 1 (2003–2006) determined the site-specific biotic factors that could be impediments to tree colonization in the grasslands. The biotic constraints to colonization addressed by this research were limitations induced by seed dispersal into grassland, persistence of an antagonistic soil seed bank, and effects of herbivory and competition with the existing sward on native seedling emergence and survival. Abiotic factors that prevent tree colonization, including fire, micro-climatic conditions, soil nutrients, water availability, and disturbance, were also examined. In this phase, two early successional species, *Macaranga indica* and *Symplocos cochinchinensis*, and mid-late successional species *Dimocarpus longan* and *Syzygium spathulatum* were planted inside the forest and degraded grasslands to determine their potential use for restoration. Phase 2 (2006–2009) investigated the performance of the two native tree species (*M. indica* and *S. cochinchinensis' mis-spelt*) with *G. sepium* as a nurse plant along with the addition of cow dung as an organic fertiliser. Phase 3 (2011–2015) included the introduction of *Gliricidia sepium* as a nurse plant and application of biofilmed biofertilizer to test the performance of two early successional (*M. indica* and *S. cochinchinensis*) and mid-late successional (*Bhesa ceylanica* and *Eugenia bracteata*) tree species established in different sized tree islands (2 × 2 m, 4 × 4 m, and 8 × 8 m). These were carried out under four treatments in combination with and without *G. sepium* and with and without biofilmed biofertilizer. The attitudes of local villagers toward our restoration goals were also recorded. In Phase 4 (2017–2021), 20 native species (across nearly 5 ha) were transplanted between forest fragments. These sites are used to train undergraduates, civil society members, and forestry practitioners in the key aspects of restoration ecology.

Project Management

An initial dialogue was held with private landowners in the KCF and the forest officers administering the area. Local communities were encouraged to participate in the project through a series of awareness-building meetings. Native tree species for planting in degraded sites were selected, these being based on their known successional status and the availability of seeds and seedlings. The local community supported the planning, field establishment, and construction of the nursery and shade houses located at the field site. An outreach program was initiated to disseminate research findings in partnership with the Forest Department of Sri Lanka, Noritake Lanka Porcelain (Pvt) Ltd., and the local community to restore degraded lands in the KCF (https://www.noritake.lk/csr.php).

Seeds and seedlings of native trees from forest edges or along roadsides were collected from the KCF (ensuring that <10% of seeds/seedlings were collected from a single mother plant). Seedlings were raised in polythene bags in forest topsoil and kept in fenced nurseries to protect them from predators, with no special seed treatment or chemical application during the first phase. In the third phase, biofilmed biofertilizer (produced using bacterial and fungal strains isolated from the rhizosphere of native species) was tested for enhancement of the growth and survival of native tree species in the nursery (Gunasekera, 2022).

Fire belts were established around all plots to protect them from dry season fires (May to September). Invasive plant species (*Austroeupatorium inulifolium*) were removed by cutting at ground level, while the weedy grass species *Cymbopogon nardus* was manually excavated. All removed plant material was piled along contour lines of the plots to reduce erosion and care was taken to retain all naturally regenerating forest species within the site during the removal of invasive species. Plots were fenced using barbed wire and mesh to protect plots from domestic cattle and wild herbivores such as sambar deer and elephants. Seedlings were introduced to the site during the main rainy season of the region (October to December) at a planting density of 4 seedlings/m^2. Plots were maintained by weeding (wet season) and clearing fire belts around plots (dry season) for 2 years after planting. The restoration plots were monitored for seedling performance for 18 months in Phase 1, for 12 months in Phase 2, and for 24 months in Phase 3. An outreach program initiated in 2017 will continue until 2026 under Phase 4.

Challenges

The main challenges of the project were associated with conflicts over land tenure. Even though degraded grasslands are located within the KCF, some lands are still privately owned. Additional problems included a high incidence of herbivory on transplants and limited fund allocation by local funding agencies for long-term monitoring of restoration programs.

Major Outcomes

The research findings were used to develop an instruction guide for linking lower montane forest fragments (Gunaratne et al., 2010, 2011, 2014) (Fig. 4.3b). During the project period (2003–2015), the frequency of anthropogenic fires in the dry seasons declined drastically due to awareness among the local community of the restoration work. An outreach program organized by the University of Peradeniya in partnership with the stakeholders was used to train a generation of students in ecological restoration.

Key Learnings

Natural regeneration on degraded grassland is constrained by limited dispersal of woody plants from residual forest fragments. Competition from non-native grasses reduces the growth and survival of planted seedlings, which can be offset by use of *Gliricidia sepium* as a nurse plant and the addition of biofilmed biofertilizer

(Gunasekera, 2022). Large tree islands of native tree species can be established in grasslands to link lower montane forests (Fig. 4.3).

Study 3: Western Ghats (India)

Background

The Western Ghats range running along India's west coast (8–21°N; 73–77°E; 160,000 km^2) is recognized as a global biodiversity hotspot together with Sri Lanka (Mittermeier et al., 2004; Annex 1c). Deforestation since the early nineteenth century which was carried out in order to establish commodity plantations of coffee, tea, cinchona, spices, and timber has resulted in loss of natural ecosystems and biodiversity. This land use change also led to land degradation. Records show that by the early 1900s, large tracts of Valparai Plateau in the Anamalai Hills were under intensive tea or coffee plantations after deforestation of the natural forests (Mudappa et al., 2014; Mudappa & Raman, 2007). Furthermore, between 1985 and 2018, the Western Ghats region suffered a decrease in evergreen forest cover from 16.2% to 11.3%, along with loss of 12% of interior (contiguous) forest cover, primarily due to increase in built-up area and destructive developmental activities such as mining, and land conversion to agriculture and plantations (Ramachandra & Bharath, 2019).

Rationale and Goals

Our work in the Anamalai Hills of the Western Ghats aimed to restore the ecology and native biodiversity in degraded tropical rainforests. We focused on large tracts of mature rainforests protected within the 958 km^2 Anamalai Tiger Reserve and over 45 rainforest fragments (having areas of 0.3 to over 300 ha) embedded within the tea and coffee plantations on the adjoining 220 km^2 Valparai Plateau. Since 2001, our work has continued to study and conserve the larger rainforest tracts, ecologically restore degraded areas, and extend conservation efforts into the surrounding landscape which has been modified for human use (Mudappa et al., 2014, Mudappa & Raman, 2007).

Over the last 25 years, research in the Anamalai Hills (Raman et al., 2018) has addressed the impacts of forest fragmentation and degradation on plants (Muthuramkumar et al., 2006) and many animal taxa such as spiders (Kapoor, 2008), amphibians (Karthick, 2019), reptiles (Harikrishnan et al., 2018), birds (Sidhu et al., 2010), and mammals (Mudappa et al., 2007; Sridhar et al., 2008; Wordley et al., 2018). A large fraction of rainforest biodiversity persists in fragments, with patches between 1 and 10 ha having conservation value as biodiversity refuges and animal corridors (Kumar et al., 2010; Sridhar et al., 2008). While the area of the fragment exerts a small influence, rainforest animal communities are more strongly influenced by the degree to which the extant habitat structure and native plant diversity resemble relatively undisturbed mature rainforests. Degraded sites inevitably have greater prevalence of secondary successional or deciduous plant species and animal taxa typical of more open country or drier forests. These studies suggested that if such degradation could be reversed and fragments brought to resemble mature forests, it is likely to benefit conservation of native rainforest plant and animal species. Furthermore, rainforest bird species also benefit if surrounding tea and coffee plantations use native tree species as shade trees for their crop (Raman, 2006; Raman et al., 2021). By fostering the use of native rainforest tree species as plantation shade trees, the conservation value of production landscapes can be significantly enhanced.

Challenges

Factors responsible for degradation of rainforest fragments that needed to be addressed include (i) past logging; (ii) conversion to coffee, cardamom, or vanilla plantations followed by abandonment; (iii) hard, exposed edges created along boundaries with tea plantations and hydroelectric reservoirs; (iv) chronic disturbances due to local firewood collection; and (v) effects of linear intrusions such as roads and power lines. Compared to mature rainforests, degraded fragments are characterized by open canopies, fewer large trees, lower plant density, reduced species richness, and basal area of trees and woody plants, especially those of mature rainforest tree species. Natural regeneration was limited or affected by invasive alien plant species, particularly *Lantana camara*, *Chromolaena odorata*, *Mikania micrantha*, *Sphagneticola trilobata,* and the African shade tree *Maesopsis eminii* (Joshi et al., 2015; Muthuramkumar et al., 2006). Robusta coffee (*Coffea canephora*), a shade-tolerant crop species, was seen to invade from adjoining plantations while the invading plants extended over 200 m into rainforests. This is likely to affect the regeneration (Joshi et al., 2009). In fragments, recruitment of seedlings through seed dispersal was reduced compared to recruitment in contiguous forests and appeared to depend more on the diversity of remnant overhead trees (Osuri et al., 2017), canopy cover, and restoration with mixed native species plantings (Osuri et al., 2021). Most rainforest fragments on the Valparai Plateau are on private land belonging to plantation companies, who are primarily concerned with their areas under tea or coffee production. The fragments, therefore, receive little attention or protection and were at risk of being cleared or converted to commercial plantations, especially where they are recorded as plantation lands rather than forests in government revenue records.

In the light of the above comments, to proceed with ecological restoration, the main concerns and barriers that needed to be addressed were (i) establishment of partnership with private landowners to recognize, protect, and restore the rainforests within their estates; (ii) removal of invasive alien weeds and maintenance of cleared sites; (iii) overcoming regeneration barriers through targeted planting of native rainforest species; and (iv) engage and involve local people in the protection of restored sites and to reduce tree cutting for firewood and other purposes.

Project Management

Our organization, the Nature Conservation Foundation (NCF), a non-profit conservation research organization, had established a Rainforest Research Station in the Anamalais and built a rapport with local plantation companies through a series of regular dialogues. As a first step, to establish partnerships with plantation companies within whose estates rainforest fragments remained, NCF signed memoranda of understanding (MoUs) between 2002 and 2005 with Hindustan Unilever Limited (now Tea Estates India Ltd), Parry Agro Industries Limited, and Tata Coffee Limited. The companies were motivated by their own corporate social responsibility and environment policies, sustainability certification for their crop and the innate interest of top managers in nature conservation. The MoUs recognized about 1075 ha of rainforest in 35 rainforest fragments in their estates for rainforest research, restoration, and wildlife conservation. The MoUs with the latter two companies were then periodically renewed at 3–5-year intervals and are still ongoing.

NCF established a local rainforest plant nursery currently located at Varattuparai on land generously provided by Tata Coffee (Fig. 4.4a). On a year-round basis, our team collects seeds of native trees and lianas from the edges of fragments or from roadsides through the forests, ensuring that no disturbance to forest interiors is caused. Seeds are raised in black polybags in locally collected soil mixed with organic compost, fenced from seed predators if required, with no special seed treatment or chemical application. Over the years, around 170 native species have been raised in the fully organic nursery, which presently stocks around 40,000 seedlings and has saplings of about 90 species.

The most degraded parts within the identified rainforest fragments were chosen for restoration on the basis of careful preliminary vegetation surveys. Multiple restoration plots of 0.25–1 ha area were demarcated in each site, totaling to a maximum of about 1–5 ha of land restored per year. The restored area is dependent on logistics, funding, and the number of saplings available. Restoration plantings were established in a range of site conditions ranging from dense shade to open grassy meadow. The sites included densely weed-infested areas with a disturbed canopy dominated by native trees or sites under non-native trees such as *Eucalyptus* sp., silver oak, *Grevillea robusta*, *Spathodea campanulata*, and *Maesopsis eminii*. Our restoration protocol involved the following steps (Mudappa & Raman, 2007, 2010; Osuri et al., 2019; Raman et al., 2009):

(i) Demarcation of the site and erection of temporary fencing (if livestock grazing was an issue).
(ii) Careful removal of invasive alien plant species, especially *Lantana camara. Lantana* and shrubby invasive alien plants were cut with a billhook, the rootstock removed with a mattock, and the woody material placed outside the site to be used as fuelwood by local people.
(iii) Retention of all naturally regenerating native plants, especially pioneers, such as *Clerodendrum infortunatum*, during weed removal.
(iv) Ensuring a maximum diversity of mixed native species during the planting protocol (27–82 native species per plot).

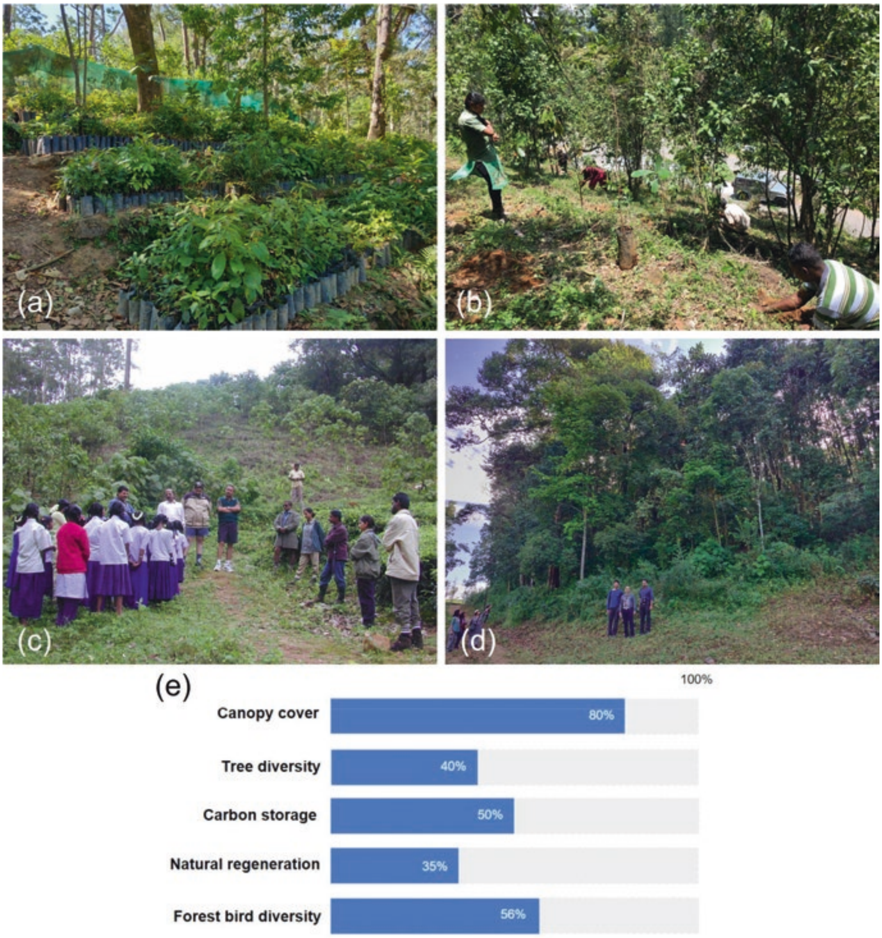

Fig. 4.4 Rainforest restoration in the Anamalai Hills, India, showing: (**a**) The rainforest plant nursery in Varattuparai; (**b**) A field team planting rainforest saplings in a degraded rainforest fragment restoration site; (**c**) The Stanmore restoration site in 2002, prior to tree planting by volunteers from a nearby school; (**d**) The same restoration site in 2020 showing a taller, dense canopy of young rainforest trees.; and (**e**) Forest recovery (after 7–15 year) and bird community recovery (after 9–17 year) following rainforest restoration on the scale from baseline values in passively restored sites left as-is to target values in benchmark forests (0–100%)

(v) Planting 2- to 4-year-old saplings (after hardening) between late May and July, so as to receive rain from both the southwest monsoon (June to August) and northeast monsoon (October to December), at an average planting density of 1099 saplings/ha (1 SE = 154 saplings/ha).
(vi) Maintenance of plots through weeding for 1–2 years after planting.
(vii) Monitoring sapling survival (up to 6 years in some plots) followed by overall forest recovery and carbon storage monitoring after ~15 years, in 2017.

Major Outcomes

Over the last two decades, besides overseeing passive restoration of the 1075 ha, we have actively restored around 100 ha through activities such as weed removal and mixed native species planting. Our monitoring (Raman et al., 2009) indicated that restoration with a high native plant diversity (>100 species) can be achieved in degraded sites with high sapling survival (>60%) after 2 years, including in open areas, *Lantana camara* infested sites (~75% survival), and under a canopy of non-native trees such as *Eucalyptus grandis* and *Maesopsis eminii* (67–76%).

When 7–15 years had elapsed after planting, comparisons with benchmark rainforests indicated that active restoration led to better recovery than passive restoration in comparable sites (Osuri et al., 2019), (Fig. 4.4). On the scale from baseline values in passively restored forests to target values in benchmark forests (0–100%), active restoration increased canopy cover (82%), tree (stems ≥10 cm girth at breast height) density (69%), species density (49%), late successional species density (42%), and compositional similarity to benchmark forests (14%). Likewise, among the saplings, stem density, species density, and late successional species density also recovered consistently by 51%, 52%, and 34%, respectively. Aboveground carbon storage recovered by 47% in actively restored forests (with a mean value of 143.9 Mg/ha C), relative to the difference between naturally regenerating sites (49.0 Mg/ha) and benchmark rainforests (287.6 Mg/ha). Recent research also indicates that actively restored sites support greater diversity of rainforest birds (and fewer open country bird species) and greater similarity in bird community composition to benchmark rainforests than comparable passive restoration sites (Hariharan & Raman, 2021).

In recent years, we have tried to share our findings and promote better ecological restoration protocols through dissemination of research findings, workshops and training events, direct outreach to visitors at our nursery and the Anamalai Nature Information Centre, and the creation of an informal network of restoration practitioners in India.

Key Learnings

Our restoration efforts have had significant local success but, as yet, have had little wider impact, since conservation and biodiversity-friendly land use practices are yet to be adopted into mainstream activities by the plantation sector. Currently, only limited commercial benefits for private businesses exist through sustainability certification, and incentives for natural ecosystem protection and ecological restoration are non-existent. Consequently, private lands remain at risk of being ignored or converted to commercial uses. The model of ecological restoration using a high diversity of mixed native species suitable to local ecosystems has also not been significantly adopted by government agencies, which still carry out large-scale afforestation using monocultures or small numbers of mostly inappropriate species. This results in the planting of ecologically unsuitable species in ecosystems at risk.

Consequently, within the Western Ghats, state forest departments are yet to adopt better protocols to restore many degraded areas within their jurisdiction.

In comparison, our restoration efforts have led to substantial recovery in forest structure and carbon storage in previously degraded systems (Osuri et al., 2019), but we have not found ways to reduce on-going pressure on sensitive lands due to people's understandable dependence on forests for fuelwood and timber. Furthermore, as a result of a focus on trees, other plant life forms, including understorey plants, herbs, and epiphytes, remain poorly represented in restored sites. Whilst naturally regenerating seedling density and species density in benchmark and restored sites are significantly higher than in unrestored, naturally recovering sites (Osuri et al., 2021), limited regeneration of old growth plant species suggests that unresolved barriers to full rainforest recovery still remain.

Study 4: Leyte, The Philippines

Background

The Philippines, which is an archipelago of over 6000 islands, is recognized to be one of the most megadiverse countries in the world. Endemism is so high on a hectare-for-hectare basis that the country has been referred to as "the Galapagos times ten" (Heaney & Regalado Jr, 1998). The country used to be almost entirely forested with 12 different forest formations but has suffered from widespread deforestation and forest degradation (Fernando et al., 2008). The first major episode of clearance to the Philippine forests came during the Spanish colonial period when large areas of the lowland forest were cleared for the establishment of large-scale plantations of sugar cane, tobacco, cotton, and other commodities. Some specific islands were heavily deforested, but forest cover, overall, remained at around 70%. The second major wave of deforestation came during the American colonial period when large-scale industrial logging was introduced. This was a major industry that continued into the post-colonial period, during which the Philippines became the largest global exporter of hardwood timber. Whilst the degraded forests might have been able to regenerate if left unattended, the logging road networks gave access to landless farmers who used the areas for destructive forms of slash-and-burn agriculture (*kaingin*). Other significant drivers of deforestation have been mining and infrastructure expansion such as road construction, hydropower dams, and tourism facilities construction (Rebugio et al., 2007). At present, only 3% of primary forest cover remains, making the Philippines one of the hottest of the biodiversity hotspots, and there is consequently a high probability of species extinction (Heaney & Mittermeier, 1997; Myers et al., 2000).

To address the loss of forest cover, the Philippines government has embarked on numerous reforestation programs over the last century (Chokkalingam et al., 2006). The conventional reforestation approach relies on the use of fast-growing non-native species, such as *Acacia mangium*, *Gmelina arborea,* and *Swietenia* *macrophylla*. These trees have been widely promoted by the Philippines Department of Environment and Natural Resources (DENR) for reforestation, because they can grow well in the prevailing harsh, open conditions. In some cases, these trees have succeeded in reestablishing tree cover, but they do not provide the same level of ecosystem services as do mixed stands of native trees. Indeed, in many cases, they do nothing to preserve traditional Philippine biodiversity or to meet the growing demand for native hardwoods. They have also been found to perform poorly when struck by typhoons, which enter the Philippines Area of Responsibility on an average of 20 times per year (Santos, 2021).

Rationale and Goals

Dr. Paciencia Milan of Visayas State College of Agriculture, now the Visayas State University (VSU), and Dr. Josef Margraf of the German Technical Cooperation Agency, now the German Corporation for International Cooperation (GIZ), developed a closed canopy and high diversity forest farming system known as Rainforestation Farming, or RF, as an alternative to conventional reforestation (Margraf & Milan, 2006; Milan, 2020). Initiated in 1992, RF was designed to conserve biodiversity and enhance ecosystem services by planting native trees, while simultaneously meeting the economic needs of local community members.

Key Strategies Used

RF was conceptualized as an agroforestry system that would provide multiple economic benefits by integrating ground crops, fruit trees, and native timber trees. RF was particularly introduced to replace the practice of slash-and-burn agriculture *(kaingin)*. RF also has the potential to be a system that could enhance or replace the small landholder coconut monocultures, which is a major land use on the island of Leyte where the Visayas State University is located (Göltenboth et al., 2003; Fig. 4.5; Annex 1d).

Fig. 4.5 Cienda-San Vicente Farmer's Association's rainforestation site in 1996 (left) and 2022 (right). (Courtesy of VSU-ITEEM)

The key innovations underlying rainforestation were two-fold: ecological and social.

Ecologically, the proponents emphasized the use of native tree species, particularly, but not exclusively, dipterocarps. The conventional wisdom at the time was that dipterocarps could not be used for reforestation because seeds were frequently not available since Dipterocarp seeds are only available during infrequent masting events. In addition, they were considered to be slow growing. The use of dipterocarps in RF was made possible by (i) using wildings rather than seeds, (ii) employing the use of a recovery chamber to boost the survival of the wildings extracted from the forest from about 50% to nearly 100%, and (iii) learning which of the dipterocarps and other natives were truly late successional species, since there are fast-growing species that can be planted in more open conditions.

Socially, RF was designed to enhance rural livelihoods, a prerequisite for successful reforestation given the Philippines' dense, rural population. Conventional reforestation has been top-down and target-driven, where local communities were hired to plant the tree seedlings provided by the DENR. Typically, the tree species planted were chosen without input from local community members, so that they had little real interest in ensuring that the trees survived. Moreover, there was a perverse incentive that if the trees died, the community could benefit from the wage-labor stemming from a future reforestation project. The shift embodied by RF was to emphasize the importance of the trees in providing a broad array of ecosystem services and by working with the local land managers to choose the tree species that the community members wanted. With this shift, the community members could see value in the trees themselves, which they then helped plant and maintain without relying on direct payments. A lot of work was also done in establishing and promoting community organizations, introducing secure tenure, working out equitable distribution of benefits, and the development of a "community family" approach.

Project Management

The first RF site was developed on a *kaingin* farm of 2.4 ha, which was later expanded to 6 ha. This was located within the VSU forest reserve (Annex 1d), and planting started in 1992, where much initial experimentation took place. From 1995 to 2000, VSU established 28 small-scale demonstration sites, ranging in size from 2.4 ha to 5.44 ha, across the Leyte province in conjunction with two local communities and 26 private landowners (Nguyen et al., 2014; Ota et al., 2018; Schneider & Pohnan, 2012). MoUs were signed between the landowners, VSU, and the local government units. Training, planting materials, and technical assistance were provided by VSU, while the landowners were responsible for maintaining the sites. Early prescriptions entailed planting 2500 seedlings per hectare of early successional trees, followed 2 years later by 2500 late successional trees (Göltenboth & Hutter, 2004; Milan et al., 1998). The system was enriched with food crops such as pineapple, okra, sweet potato, cassava, climbing yams, and fruit trees which included durian, lanzones, rambutan, and mangosteens. Approximately 100 tree species were used across the sites with each site having as many as 40 different species. Each site, however, had an individual mixture of tree species depending on what was available in the nursery at the time (Nguyen et al., 2014). Non-native species were also found in some of these plantations (Nguyen et al., 2014; Ota et al., 2018), but those trees generally preexisted in the sites before the establishment of RF.

Challenges

Developing a new agroforestry approach that integrates native forest tree species inevitably faces numerous challenges and obstacles. These included convincing farmers to adopt a new system, strengthening people's organizations, overcoming tenurial problems, conducting research to strengthen the scientific basis of the system, and maintaining funding. With the DENR actively promoting these fast-growing exotics, finding the planting materials needed for RF was also a challenge until mother trees were identified and local community nurseries established.

One specific challenge that has to be faced in all rainforestation sites stems from the fact that a national logging ban was issued in 2011 outlawing the harvest of native timber from natural forest areas. This blanket prohibition clearly affects the harvest of planted native trees, so the DENR has instituted a process through which RF adopters can register their newly planted native trees, allowing them to be eventually harvested. However, this requires interactions with the DENR and therefore there is the possibility of increasing transaction costs. This is one significant obstacle to the planting of native species, which is not faced by those choosing to plant and harvest exotic species.

Major Outcomes

The reported conditions of the early rainforestation sites range from "highly successful to completely abandoned" (Ota et al., 2018). Several surveys have been undertaken by external scholars to look at the social impacts, tree growth, and stand dynamics across the sites (Nguyen et al., 2014; Ota et al., 2018; Schneider et al., 2013; Schneider & Pohnan, 2012). Other studies have highlighted the positive effect on soil quality (Asio & Milan, 2002), community empowerment (Asio & Bande, 2005; Compendio & Bande, 2017), provision of stable income to farmer adopters (Ahrens et al., 2004; Voyeux, 2003), and high carbon sequestration potential which will contribute to the mitigation of global warming (Bande et al., 2014).

The VSU Institute of Tropical Ecology and Environmental Management continues to carry out the work of promoting and further developing RF. With strong advocacy from the Haribon Foundation, RF was also adopted by the DENR as an official reforestation strategy (DENR, 2004). Despite the formal acceptance of rainforestation by the DENR, the use of fast-growing exotics has remained fairly well-entrenched. The situation is slowly changing with a growing role for the use of native species in such government programs as the National Greening Program (NGP) of 2011–2016, which targeted the reforestation of 1.5 million hectares, together with the enhanced NGP of 2016–2028, which aims to reforest an additional 7.1 million hectares (Republic of the Philippines, 2011, 2015; Table 4.1). These changes in DENR policy and practice have not come easily but have required the ongoing engagement of rainforestation supporters, many of whom have come together as members of a voluntary network, known as the Rain Forest Restoration Initiative (RFRI). RFRI members have also been active in the restoration of a large number of RF sites throughout the Philippines.

Key Learnings

On the technical side, many lessons have been learned from the early Rainforestation pilot sites and have been integrated into subsequent rainforestation initiatives. The maximum tree planting density, for example, is now 2500 trees/hectare in completely open areas and it is recognized that fruit trees, which need more light, should be planted in areas where they will not be overly shaded by timber trees. Research underlying the individual growth characteristics of native trees has also progressed significantly. For example, it is now known that a number of dipterocarp species, including *Shorea contorta, Parashorea malaanonan, Hopea plagata, Hopea philippinensis*, and *Dipterocarpus alatus* are mid-successional species that grow quite well in open conditions. As such, these trees can be planted independently or simultaneously with early successional species, rather than relying on a two-stage planting as previously recommended.

On the social side, several studies by Schneider and Pohnan (2012) and Ota et al. (2018) have suggested that the original RF sites did not generate as much cash income as the adopters expected, though they did provide products such as fruits, timber, seedlings, and firewood that could be harvested for domestic consumption. Nevertheless, the RF implementers were generally pleased with the program and benefited in many other ways, including enhanced environmental services, greater resilience to disturbances like typhoons, access to other income-generating opportunities, and greater community self-confidence from working with outside experts. The two people's organizations also benefited greatly from their nursery operations and increased access to other government reforestation projects (Ota et al., 2018; Schneider & Pohnan, 2012). These findings have important implications for the way in which Rainforestation will be promoted in the future.

Study 5: East Kalimantan Indonesia

Background

The restoration site in Indonesia is located in the lowlands of East Kalimantan, which is part of the Indonesian portion of Borneo (Kalimantan). This comprises about 70% of the island. East Kalimantan has experienced widespread deforestation and forest degradation primarily as a result of industrial-scale logging, conversion for plantations, and mining (Gaveau et al., 2014). The mixed dipterocarp forests were particularly hard hit by commercial exploitation, which started in the early 1970s, because the trees provide extremely valuable timber. They can constitute as much as 60% of the total basal area of the lowland forest, where harvest rates in the region ranged from 80 to 100 cubic meters/hectare compared to 30–50 cubic meters/hectare elsewhere in the tropics (Kartawinata et al., 2008).

As a result of consistent logging, the forests have become more susceptible to fires, particularly during prolonged dry seasons associated with the *El Niño*-Southern Oscillation. The danger presented by fires became particularly evident in 1982–1983, when fires burned approximately five million hectares of primary and secondary forests across Borneo (Goldammer & Siebert, 1990; Siegert et al., 2001). Given the frequency of fire events, many previously forested areas are now dominated by pyrophytic *Imperata cylindrica* grasslands, which now cover an area of over 2.2 million hectares in East Kalimantan. This area has been the focus of many reforestation efforts and attempts to use it for smallholder agriculture across Indonesia, but these have been mostly unsuccessful (Garrity et al., 1997).

Rationale and Goals

Samboja Lestari was established in 1991 by the Balikpapan Orangutan Society now known as the Borneo Orangutan Survival Foundation (BOSF; Annex 1e). This is an Indonesian non-profit organization dedicated to the conservation of the Bornean orangutan (*Pongo pygmaeus*) and its habitat. The program's aim was specifically to rescue, rehabilitate, and, where possible, release orphaned or misused orangutans rescued from areas of habitat loss and the wildlife trade. In the early years, however, BOSF only had access to 3 hectares of forest for their orangutan rehabilitation and reintroduction efforts. Other forest areas nearby were undergoing deforestation or were being used for research so were not available since orangutans tend to damage the trees. Thus, BOSF launched a forest restoration program in 1999 to convert approximately 1850 hectares of land dominated by *Imperata cylindrica* into a young forest and wildlife sanctuary.

Key Strategies Used

There were several key strategies in developing Samboja Lestari, which were as follows:

1. To clarify the legal status of the area: Land for the project was purchased by BOSF to minimize the chance of tenurial conflicts. A significant part of the land was a former transmigration site, where the Indonesian government had attempted to resettle people from Java and other more densely populated areas of the country. As a transmigration site, it had proven largely unsuccessful, so the land was purchased from the former inhabitants for a reasonable price. Other

areas of the site were purchased from members of older, more established communities.

2. To create the necessary infrastructure: BOSF built its offices on the site, together with an Ecolodge, which was established with the idea of housing volunteers and other supporters. It also developed a road system to improve access throughout the site for planting, to monitor plant growth and to facilitate fire suppression. A fire tower was established to aid in the monitoring of fire outbreaks, and a large pool and small ponds of water were also created in anticipation of firefighting needs.
3. To develop a partnership with local community members: The program was specifically designed to give multiple benefits to local community members. In this respect, during the restoration process, some farmers were given permission to cultivate the land between the growing trees in certain areas, and locally grown fruits were purchased at a premium from farmer groups by BOSF for the orangutans. Alternative livelihoods were also provided through sugar palm tapping and various types of carvings and handicrafts. Locals were also employed for fire prevention, nursery management, security, and tree planting, with others offered positions as staff in the office and the Ecolodge. BOSF also supported farmer field schools to teach agroforestry and provide environmental education for students.
4. To ground the project on the latest advances in scientific knowledge: The restoration approaches used at Samboja Lestari drew heavily on the Tropenbos Kalimantan Project, a long-term research project based at the nearby Wanariset-Samboja area. This project ran from 1985 until about 1997 and focused on facilitating the restoration of dipterocarp forests through research on mycorrhizae, species selection, soil and site classification, planting stock production, and growth and yield studies (Effendi et al., 2001). Dr. Willie Smits, founder of BOSF and the driving force behind Samboja Lestari, had formerly served as team leader of that project.

Project Management

As indicated, the restoration of Samboja Lestari significantly benefited from the legacy of the Tropenbos Kalimantan Project at nearby Wanariset-Samboja in terms of the facilities that had been developed, including a large nursery and one of the best herbaria in Indonesia. Plant propagation for the restoration efforts was initially carried out at Wanariset-Samboja, but a nursery was later built at Samboja Lestari in conjunction with a large site for creating compost out of orangutan feces and other organic waste, which was subsequently used for fertilizing tree planting.

Restoration of the site then focused on three different zones:

(a) The Sanctuary Zone, which was located in the interior. This roughly 300 ha zone contains the Ecolodge, an area for orangutan and sun bear cages, and islands where orangutans that cannot be returned to the wild can live. Trees were planted in this zone, but reforestation also relied on natural regeneration, some of which was facilitated by transferring seed-bearing soil from secondary forest areas.

(b) The Reforestation Zone, where an array of different techniques were used. An arboretum was established on 82 ha with 281 tree species, focusing especially on species native to Kalimantan, particularly with an endemic nature. In some open areas, line plantings of trees using *Shorea balangeran*, *Vitex pinnata, Aquilaria mollucana*, *Durio* spp., and *Alstonia* spp. was carried out. These trees quickly shaded out the exotic *Imperata cylindrica* facilitating establishment of natural regeneration. An agroforestry approach was used in some areas with the trees being interplanted with agricultural crops, including pineapples, beans, corn, and ginger, as well as bananas and papayas. The crops reduced competition for the trees and the fertilizers applied to the crops helped promote tree growth. Finally, in large areas of Samboja Lestari, Assisted Natural Regeneration was applied where adequate numbers of species appropriate for natural regeneration was present.

(c) The Buffer zone, within which a 100 m ring of sugar palm (*Arenga pinnata*), was partially planted around the site to suppress fire and provide resources for local livelihoods. Sugar palm has very dense foliage, which not only kills off *Imperata cylindrica* but also provides thatch, ethyl alcohol, natural medicines, fibre, and many other useful products for community use.

A guiding philosophy underlying this entire restoration effort was finding "synergy with nature," in an attempt to accelerate natural succession. In order to do this, fast-growing local tree species that were attractive to wildlife in terms of fruit, nectar, and nesting sites were planted to entice wildlife to re-enter the area. It was anticipated that wildlife would bring seeds from the nearby forest areas such as Bukit Soeharto and Wanariset-Samboja into the site, thus enhancing species diversity. Silvicultural techniques were applied not only to the trees that had been planted but also to those trees that had emerged through natural dispersal. In general, this approach worked well, although the transition required that some traditional approaches needed to be modified. Field staff needed constant reminding that certain trees, which had previously been cut down as weeds, now needed to be treated and protected similar to those trees that had been intentionally planted (Neidel et al., 2012).

Challenges

The greatest challenge to restoring the *Imperata* grassland area was due to the flammability of the grass, increasing the risk of fire during the dry season. To address this problem, a 35 m observation tower was built to make sure that fires were quickly detected. In addition, a fire break of sugar palm was planted around much of the site, and dispatch teams were formed to quell fires before too much damage could be done. BOSF also had a fire truck and other fire-fighting equipment on hand. Of equal importance was that the Samboja Lestari project was developed in conjunction with a consistent recognition of the local communities' economic needs, in order that the surrounding village members were disposed to help control threats to the site. These threats include illegal logging, clearance for conversion of land for agriculture, and the indiscriminate use of fire for clearing.

Major Outcomes

The Samboja Lestari project determined that areas dominated by *Imperata* grassland need to be maintained in the face of anthropogenic fire events. If fires are quickly terminated, the land will undergo natural succession, allowing it to be restored to a high diversity secondary forest (Yassir et al., 2010). Over the course of this project, biodiversity recovery of the site was quite rapid, with 1221 plant species, 57 bird species, and 18 species of mammals having been recorded on the site by 2008–2009 (unpublished data). To some observers, Samboja is one of the most successful tropical forest restoration sites anywhere in the world (see, for example, Little, 2008). However, Samboja Lestari is not without its detractors. Meijaard (2009), for example, questioned a number of claims that Smits made about Samboja Lestari in his 2009 TED Talk. It has also been suggested that the funds would have been better used for the conservation of intact forest (Little, 2008; Thompson, 2010).

Key Learnings

One of the key learnings is that maintaining local community support, even after active restoration activities are over, is extremely important. The Samboja Lestari project was designed to make sure that the local community members received economic benefits from the project in return for their continuing support. A major change in leadership at BOSF, however, led to a halt to the restoration work, a distancing from former staff who had been involved in those efforts, and a discontinuation of many of the community programs. The danger of losing community support became apparent in 2015, when outsiders started clearing part of Samboja Lestari for agriculture. The local community leaders, who would have previously intervened, did not act to stop the encroachment. Later when called upon to assist with the suppression of a fire that had broken out in the site, local community members were also unwilling to respond. As a result, approximately 300 hectares burned before outside help could be mobilized from a local mining company to suppress the fire. For a forest restoration site originally built on such strong social principles, this was a very unfortunate setback (Fig. 4.6).

Fig. 4.6 Samboja Lestari in 2002 and 2016. (Courtesy of Dr. Ishak Yassir)

Discussion

The case studies of this chapter show that active restoration with a high diversity of native species can lead to successful rainforest recovery. However, it should be understood that the long-term effects of these programs can be transient, which emphasizes that the retention and protection of existing mature, undisturbed rainforest sites from future disturbances will always remain the top conservation priority (Di Sacco et al., 2021). Ecological restoration is not a complete substitute for habitat and landscape conservation, and the pledges of restoration elsewhere should not be used to justify existing forest conversion to other land uses in critical habitat areas.

Our studies have thus underscored the unequivocal need to protect all rainforest fragments, large and small, including those that occur outside protected reserves, especially in global biodiversity hotspots. These fragments act to conserve rich biodiversity and contribute to the delivery of important ecosystem services. With respect to future restoration activities, these fragmentary areas can provide seeds and seedlings and serve as reference forests that can enhance our understanding of the structure, function, and composition of local forests which are the target of restoration efforts. Forest fragments can also be incorporated as key nodes in the establishment of forest corridors, as has been shown in the Western Ghats case study.

The case studies of this chapter have showcased how the ecological knowledge of native plant species in nearby reference forests and species trials can be successfully employed in restoring degraded heterogeneous landscapes typical of south and southeast Asian lowland and lower montane rainforests. This is an important outcome since lack of sufficient ecological information about native forest species of conservation and utility value have contributed to restoration failures in the past (Vivekanandan, 1989). For example, the identification of the successional gradients of species in terms of their needs for direct sunlight or differing amounts of shade is a key consideration, and in some cases, this research has brought about a much more nuanced understanding of the needs of different tree species. Conventional wisdom in Indonesia and the Philippines, for example, held that all dipterocarps were late successional species requiring significant shade, whereas we now know that a number of species have high survival and growth in open conditions. Indeed, one species in East Kalimantan, *Shorea belangeran,* has even performed well in a completely open and highly degraded coal mine rehabilitation site (Yassir & Adman, 2015).

For other species, these case studies have underscored the need to provide appropriate partial shade conditions for late successional species. In Sri Lankan rain forests, *Shorea trapezifolia* stand out as highly successful mid-successional native tree species and is emerging as among the best candidate native species for incorporation into landscape restoration activities in lowland rain forest areas (Kathriarachchi et al., 2004; Fig. 4.1b). Suitable plantations or secondary forest stands of pioneer/early successional trees, which have already successfully replaced invasive grasses and ferns, are often available in the landscape as readymade nurse tree stands for restoration planting of mid-late successional species. In Sri Lanka, *P. caribaea* plantations and mixed stands of *Alstonia macrophylla, Macaranga peltata, Trema orientalis, Symplocos* spp. provide good examples of mixed nurse stands that are often available outside protected areas (personal observations). In Indonesia and the Philippines, existing plantations of *Acacia mangium* and *Paraserianthes falcataria* can play a similar role (Otsama, 2000). A cautionary note is that some species, such as *Acacia mangium* and *Alstonia macrophylla,* have proven highly invasive in some circumstances. This means that special care needs to be taken when planting new plantations for this purpose (Koutika & Richardson, 2019).

In addition to concerns with the differential effects of light environments, species chosen for restoration have to be well-suited to site conditions in terms of belowground resource availability. This concern is linked to the available parent material, the topographic position, and the extent of soil degradation. For example, in Sinharaja, Sri Lanka, incorporating the topographical affinities of different species into the selections for enrichment planting of *P. caribaea* plantations played an important role in the success of this restoration strategy (Ashton et al., 1997, 2011). Notwithstanding this understanding, in other areas such as the Philippines, where forest restoration has moved from research to broader scale implementation, some of these affinities are known, but more random patterns of tree planting are commonly practiced.

It is known that symbiotic root-inhabiting mycorrhizae play an important role in tropical forests by enhancing trees' ability to take in water and nutrients (Hodge, 2017). Thus in almost all of our case studies, native forest soils have been used in nursery work to ensure the introduction of native microbial inoculum from the earliest stages for the healthy growth of seedlings. In the Knuckles forest case study in Sri Lanka, biofilmed biofertilizer application has had an additional benefit for enhanced survival and relative growth rate of the seedlings of the native tree species in the nursery and under field conditions (Gunasekera, 2022; Seneviratne et al., 2011). Whether some of the established plantations (such as with the *P. caribaea* trees in Sri Lanka) facilitate the establishment of the newly planted dipterocarps, through mycorrhizal connections, is currently unknown. The role of root-associated mutualists in enhancing the success of restoration programs is therefore an important area of future research.

Studies involving isolated small populations of canopy tree species in Sri Lankan rainforest fragments have shown that an understanding of the initial pool of genetic variation seems to be a critical element in the design and implementation of an FLR project (Gunatilleke & Gunatilleke, 2013; Nef et al., 2021). Genetic diversity is tightly linked to species' reproductive ecology, fitness, and adaptive potential, which are often correlated with life-history traits characteristic of divergent subpopulations of localized species (Nef et al., 2021; Richards et al., 2016; Smith et al., 2018). Integration of intraspecific genetic diversity attributes in ecogeographic scale restoration planning is potentially significant for the longer-term sustainability of forest restoration initiatives. These population genetic considerations will help capture intraspecific ecoregional diversity and conserve their nuanced adaptive variations in climate-resilient populations, especially in biodiversity hotspots (Gaisberger et al., 2022). This feature warrants greater emphasis in broader, regional-scale spatially explicit conservation *cum* restoration planning in the face of continuing threats of genetic erosion in a changing climate.

These five case studies illustrate the critical importance of co-developing restoration programs with local communities. In addition to the need for community help in protecting and maintaining the site, one benefit of doing so is that these communities possess indigenous knowledge on species' traits, habitat affinities, and utility values that might take a lengthy research program to otherwise uncover. Integration of such knowledge at the early stages of restoration planning in Sri Lanka and the Philippines contributed to their early success. Working with local village communities also allows an avenue for local demonstration of successful approaches and an opportunity to encourage rapid uptake of new techniques for management of their private land, for example, in analog forests and agroforestry gardens.

Our experience suggests that uptake will be enhanced if the restoration program adopts species that deliver products that have been harvested traditionally from the natural forest (Ashton et al., 2014). Planting a mixture of utility species and native forest species is integral to the rainforestation farming initiatives in the Philippine case study, and this attribute of the system contributes to improvements in food security through climate resilient agroforestry (Milan & Margraf, 1994; Veridiano et al., 2020). A further step is to actively promote and develop cottage industries based on the products of restoration plantings, for example, sugar palm, rattans, medicinal plants, honey, and health foods. This approach has been taken in partnership with tea plantation companies in Sri Lanka that are incentivized by opportunities for the tea companies to diversify into agro-ecotourism (https://www.dilmahconservation.org/initiatives/sustainability/biodiversity-corridor-endana.html). In India, large plantation companies could be successfully engaged as partners in restoration if they are willing to set aside areas for protection and restoration, but the lack of firewood resources for workers who live alongside these forest patches poses a continued challenge for protecting restored sites from future degradation.

Additional demonstration projects with evidence-based site-specific prescriptive guidelines and strong rural socio-economic underpinnings are urgently needed for implementation of large-scale restoration programs across the multitude of landscapes that exist across wet tropical Asia. Demonstrating the economic viability of these programs enhances the likelihood that they would be taken up at larger scales. For example, we have compared the relative economic merits of restoring pine-clad hilltops and upper slopes in the wet lowlands of Sri Lanka using native species with conversion to small-holder tea gardens (Ashton et al., 2014). The financial analysis demonstrated that planting a native forest for timber and non-timber products may be more profitable than cultivation of tea smallholdings. This financial analysis excluded any payments for ecosystem services, which, if available, would have undoubtedly strengthened the financial incentive for ecosystem restoration. One example of a successful Payment for Environmental Services (PES) scheme was the Environmental Protection Fee introduced by Bago Local Government Unit (LGU) in Negros Island, Philippines, which has been levied on all city water users since 2016. This revenue has been used to fund the conservation of forest and biodiversity initiatives, forest protection measures to ensure sustained water flows, and alternative livelihood programs for forest communities (Global Forest Goals Report, 2021). Similar reward schemes could be developed for conversion of non-native plantations into mixed-species forests in critical watersheds for ecosystem restoration and rehabilitation activities using the methods which have been proven to be successful in our case studies.

A study somewhat akin to rainforestation in the Philippines, but using a mixture of native species and fast-growing non-native eucalypt spp. in the Atlantic Forest regions in Brazil, has yielded promising results in offsetting restoration implementation costs without undermining the ecological outcomes of restoration within a period of only 7 years since planting. The income from eucalypt wood production has offset 44–75% of restoration implementation costs (Brancalion et al., 2019). Likewise, non-native early successional species, such as those of *Acacia, Alstonia, Eucalyptus, Macaranga, Paraserianthes*, *Melia,* and *Pinus,* that are already established in degraded forests in Sri Lanka could be used either as mixed nurse tree populations in a relay floristic method or growing simultaneously with native species. Some of these species, such as *A. macrophylla* and *Melia azedarach* which were introduced to the island more than a century ago are now naturalized to a greater extent and emerging as potential nurse tree stands for native species establishment beneath them in degraded lowland rain forest landscapes. Furthermore, these non-natives are utility timber species widely popular among local communities. Experimental trials need to be initiated for silviculturally managing these stands in facilitating native species restoration for their ecological and socio-economic feasibility. This approach has the potential to be used in community-led forest landscape restoration programs, particularly in small-holder-dominated economies off-setting at least a part of the restoration cost, but only in areas where adequate infrastructure for transport, marketing, and utilization already exist or where some sort of PES scheme is feasible.

Certification of agricultural products can also provide an economic incentive for improvements in land management, which can include investments in restoring degraded forest patches that do not contribute to the land area under cultivation. For example, larger tea plantations in the lower montane Knuckles region and Valparai

Plateau in Western Ghats, India, contain embedded forest fragments that could be restored under a certification scheme that promotes sustainable management of biodiversity and ecosystem services in production landscapes (Chowdhury et al., 2021), such as those that exist for certification of sustainable palm oil (Brandi et al., 2015). This scheme could be designed in a way that member organizations are mandated to conduct a land use planning exercise that identifies lands that are marginal for tea production and promote restoration initiatives that engage plantation communities with enhanced livelihood opportunities.

Similarly, Payment for Ecosystem Services that prioritize biodiversity, soil and water conservation, preventing floods and landslides, provision of timber, fuelwood and non-timber resources, and promotion of ecotourism may leverage tea plantation companies to diversify their investments in ways that support increased tree cover. For example, multiple crop certification systems focusing on landscapes rather than just a single end product (such as tea) may trigger diversification from monoculture tea plantations into a mosaic of land uses that support greater native biodiversity, ecosystem services, and the livelihoods of people dependent on them (Chowdhury et al., 2021). Small landholder tea producers who provide the majority of tea exported by Sri Lanka are more amenable to this type of multiple crop analog-forest landscape certification system including mosaics of homegardens as well as smallholder tea plots. These high-biodiversity agroecosystems, reinforced with elements of nature-based and sustainable land management practices, may find value-added niche markets for such products (FAO/INRA, 2016; Padulosi et al., 2012).

Conclusions

The case studies presented in this chapter illustrate how the goals of the UN Decade on Ecosystem Restoration could be achieved in practice across multiple settings in wet tropical Asian biodiversity hotspots, where conservation of biodiversity is a priority. Development of the concept of ecosystem services over the last two decades, with its emphasis on the market valuation of the wide array of goods and services provided by natural ecosystems, has the potential to generate a paradigm shift in forest restoration. Government natural resource management agencies throughout the region, whether it be the DENR in the Philippines, the Forest Departments in Sri Lanka and India, or the Ministry of Environment and Forestry in Indonesia, have persistently practiced reforestation using only a very narrow suite of mostly non-native trees, frequently planted as monocultures. These trees, while usually performing well from a narrow production forestry perspective, have failed to achieve other management objectives, such as enhancing biodiversity, supporting local community livelihoods, or improving local hydrology. Fortunately, a broader diversity of restoration approaches, like those discussed in this chapter, are becoming increasingly popular. Overcoming the inertia and vested interests in conventional reforestation, however, remains a major challenge.

Instituting innovative payment schemes for ecosystem services as a way to finance forest restoration is a pragmatic way forward. The lack of a mechanism to generate regional data on the economic value of ecosystem services provided by restoring landscapes has been a constant barrier to progress. This is so important in geographically and socio-culturally similar regions like the south and SE Asian region. To address this need, the Economics of Ecosystem Restoration (TEER) initiative, a standard framework to assess the cost and benefits of restoration projects, was recently developed under the aegis of the UN Decade on Ecosystem Restoration and could be a key tool for mobilizing donors, investors, project implementers, governments, and other stakeholders (Bodin et al., 2021). Coupling these types of initiatives with enabling national policy environments for their larger-scale implementation based on ecologically sound field trials may create investment opportunities for forest landscape restoration (Lamb, 2018). Green or Climate Bond marketing portfolios which support a wide range of nature-based solutions toward sustainable land management are an emerging opportunity for natural capital investments that include forest landscape restoration especially suited for Global Biodiversity Hotspot regions of the world, e.g. Rainforest Impact Bond program in Indonesia (Innovative Rainforest Bond Structure Unveiled at Indonesia (globenewswire.com), (https://www.globallandscapesforum.org/wp-content/uploads/2020/10/How-can-Green-Bonds-catalyse-investments-in-biodiversity-and-sustainable-land-use-projects-v12_Final.pdf).

As the pace of innovation in forest and landscape restoration increases, there is also a need for establishing national and regional networks for sharing practical experience (WRI, 2021), and the joining together of communities, governments, financial supporters, and research agencies to overcome barriers inhibiting planning, financing, and policy reform. Only by working together to develop robust ecological, social, and economic foundations for forest restoration and empowering people to make these changes will we be able to realistically meet the Bonn Challenge targets of our respective regions. This is especially relevant for endemic-rich Global Biodiversity Hotspots threatened with rapid forest degradation and deforestation, especially during this UN Decade of Ecosystem Restoration (https://www.bonnchallenge.org/about).

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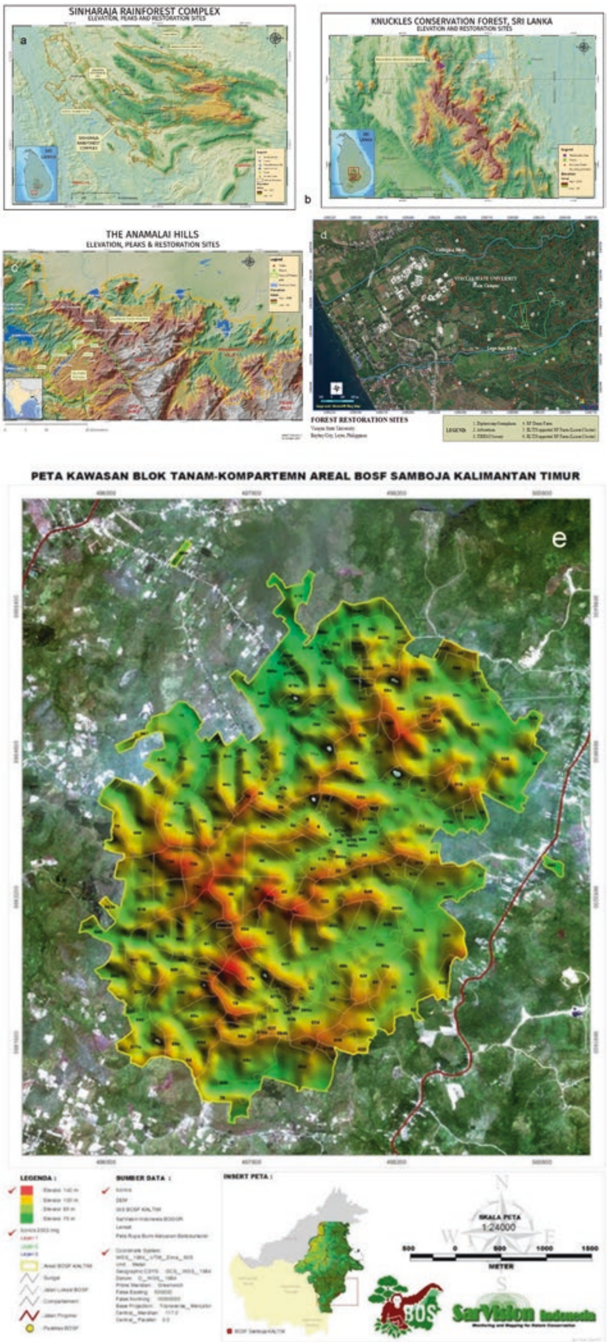
Annex 1



Location of each restoration study site in relation to the elevation and relief of the broader landscape in (**a**) Sinharaja WHS, Sri Lanka, (**b**) Knuckles region of the Central Highlands WHS, Sri Lanka, (**c**) Valparai Plateau, Western Ghats,India (**d**) Leyte, Philippines, and (**e**) Kalimantan, Indonesia

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Chapter 5
Temperate Forest Restoration

Nick Reid, Yvette Dickinson, Rhiannon Smith, Michael Taylor, and David Norton

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Summary and Key Lessons
Eight thousand years ago, temperate and boreal forests covered 3200 million hectares, or a quarter of the earth's land surface. They have been reduced to approximately half that amount over the last two millennia to make way for agriculture and human settlement. In recent decades, however, this trend has reversed, with net gains in temperate and boreal tree cover of over 3 million ha annually between 1990 and 2015. Because social and economic drivers determine whether forests shrink or expand in the modern era, this chapter profiles three contrasting examples of temperate forest restoration or rehabilitation that are driven by very different motivations. The first, Tiramoana Bush in Te Wai Pounamu South Island, Aotearoa New Zealand, is 407 ha of former pasture land, which is being restored to native temperate forest as an offset for a nearby landfill development. The second case study concerns Colorado's Front Range, which is 1.7 million ha of predominantly wildlands with 2 million residents, who have suffered devastating wildfires over a 30-year period. The Front Range Roundtable, representing multiple agencies, organisations, and community, negotiated the restoration of 13,000 ha of lower-montane dense coniferous forest, to emulate pre-settlement grassy old-growth woodlands and reduce the threat of catastrophic fire. The final case study describes the reforestation of 'Taylors Run', a 750-ha farm in northern New South Wales, by two generations of the Taylor family, after nearly all of the natural eucalypt cover was lost to 'New England dieback' between the 1950s and 1970s. The rehabilitation programme featuring exotic and native trees and shrubs to withstand the dieback caused by defoliating insects has restored shade and shelter for livestock as well as biodiversity and amenity, generated a net positive carbon balance, and created new business opportunities. The long timeframes, high costs, and complex social dynamics associated with temperate forest restoration and rehabilitation require innovative inter-generational policy, funding, and business solutions, together with careful consideration of monitoring and evaluation processes and social understanding to ensure the success of multi-decadal and multi-century projects.

Management Implications
Climate change and the material needs of the world's growing population will ensure that temperate forest restoration and rehabilitation will remain important societal goals well into the next century.

- Passive regeneration of temperate and boreal forests is already widespread in the northern hemisphere, due to the abandonment of farmland, a trend that will likely continue due to socio-economic and political drivers.
- Temperate and boreal forest restoration, on the other hand, could become increasingly difficult and expensive, if human modification of such forests increases.
- Degradation of the biophysical environment, plant and animal pests and pathogens, and lack of propagules are important barriers to temperate forest restoration.
- In fire-prone ecosystems, hazard reduction burning that maintains forests in short-term cycles of recovery may increase rather than reduce the likelihood of destructive wildfire over planning horizons of 50–100 years and longer.
- Some temperate and boreal forests are characterised by replacement sequences where early successional species give way to stands of self-replacing species over the course of centuries.
- The monitoring and evaluation (M&E) criteria for the restoration of such forests should reflect the lengthy trajectory of successional states. Innovative long-term funding arrangements will be required to help achieve these outcomes.
- Given multi-century-long timeframes, and despite the best of intentions, however, it is unlikely that temperate and boreal forest stands will remain undisturbed by human or natural agencies. M&E criteria for the restoration of these forests should therefore accommodate the likelihood of unplanned events.
- Given the historic destruction and modification of temperate and boreal forests, climate change, and humanity's growing needs, investment in novel forest ecosystems such as timber plantations and agroforestry (ecological rehabilitation) will increase in future.
- Success in restoration is unusual; practitioners should not be discouraged by failure, but encouraged to learn from their mistakes and to manage adaptively.

Introduction

The world's temperate and boreal forests occur in the mid and higher latitudes between the sub-tropics, arid steppes, and deserts, on the one hand, and sub-Arctic and sub-Antarctic regions on the other. Eight thousand years ago, temperate and boreal forests, which in this chapter include woodlands with an interrupted tree canopy, covered an estimated 3200 million ha, or 24% of the earth's land surface (Shvidenko et al., 2005). The vast majority of this forest cover (96%) occurred in the northern hemisphere (Dinerstein et al., 2017) across Europe, the former Soviet Union, Asia, and North America. Temperate forests in the southern hemisphere were of much smaller extent but remain dominant components of the vegetation of Australia, New Zealand, and Chile.

Temperate and boreal forests are globally significant in terms of their biology, biogeochemical functioning, and ecosystem service provision (Silander, 2001). They contain the largest and oldest plants in the world and are humanity's major source of timber and wood products. The biomass of at least some temperate forests exceeds that of any other terrestrial biome. Like all forest ecosystems, these are pivotal to the functioning of the biosphere and are important refuges for terrestrial biodiversity, particularly fungi, mosses, and lichens. In addition, temperate and boreal forests provide essential ecosystem services and afford a window into the biogeography and evolution of life on Earth (Newton & Featherstone, 2005; Shvidenko et al., 2005). Northern hemisphere temperate and boreal forests consist predominantly of three types: (i) coniferous forest (dominated by species of *Pinus*, *Picea*, *Larix,* and *Abies*); (ii) angiosperm (broadleaved) forest, which is dominated by many genera, including *Populus*, *Betula*, *Quercus*, *Fraxinus, Ulmus,* and *Tilia;* and (iii) mixed conifer and broadleaved forest. Southern hemisphere temperate forests are distinct floristically and phylogenetically and are dominated by trees of Gondwanic origin, such as *Nothofagus*, *Eucalyptus,* and austral conifers in the families, Araucariaceae and Podocarpaceae. Ecosystem services provided by temperate and boreal forests include (i) the provisioning of resources such as timber, fuelwood, water, game, and medicines; (ii) regulation of the environment through services such as regional climatic amelioration, hydrological services, and catchment protection; (iii) supporting services such as nutrient cycling, primary production, and habitat provision; and (iv) cultural and recreation benefits (Gamfeldt et al., 2013; Shvidenko et al., 2005; Li et al., 2016; Millar & Stephenson, 2015; Silander, 2001). In addition to these critical services, the importance of temperate and boreal forests as carbon sinks in mitigating global warming is difficult to understate (Silander, 2001).

Notwithstanding the global significance of temperate and boreal forests, human civilisation has been made possible by converting forests to other uses, particularly agriculture. Europe lost 50–70% of its original forest cover, mostly during the early Middle Ages, and North America lost about 30%, mostly in the nineteenth century (Shvidenko et al., 2005). Australia lost 44% of its forests and woodlands in the 226 years after European settlement (Metcalfe & Bui, 2016), and New Zealand lost 71% of its original forest cover in the past 700 years since the arrival of humans (Ewers et al., 2006). Estimates of the remaining extent of temperate and boreal forests in 2015–2020 vary between 1708 and 1910 million ha. This is almost half (44–48%) of global forest cover and constitutes 12–13% of the earth's land surface (Keenan et al., 2015; FAO, 2020).

In the late twentieth and early twenty-first centuries, recovery of part of the former extent of temperate and boreal forests has been an ecological restoration success story. Economic development and forest policies in formerly forested boreal regions during the last 40 years have led to natural reforestation and expansion of forests, due to their regeneration potential and the suppression of fire from the 1960s to the mid-1990s (Shvidenko & Nilsson, 2002). Active temperate forest restoration and plantation development in Europe, North and South America, and Australasia have also played an important role. Between 1990 and 2000, temperate and boreal forest increased in extent by 2.9 million ha/year, of which 1.2 million ha were forest plantations and 1.7 million ha were expansion of natural forests (Shvidenko et al., 2005). This increase accelerated in the first 15 years of the twenty-first century, with temperate forest cover growing at 3.8 million ha/year and boreal forests at 0.5 million ha/year (Keenan et al., 2015). Although part of the increase was due to the passive regeneration of abandoned farmland, policy and regulatory initiatives have been important drivers in forest recovery. These actions have reduced agricultural overproduction in Europe, increased timber production, and sequestered carbon to offset greenhouse gas emissions (Dudley, 2005; Mansourian & Regato, 2005; Schuyt, 2005; Shvidenko et al., 2005). In this way, the increase in temperate and boreal forest cover has partially offset the worldwide dramatic loss of tropical and subtropical forests over the same period.

Notwithstanding the positive overall global trend in forest extent in temperate regions in recent decades, the condition of temperate and boreal forests has not necessarily followed suit. Air pollution, damage by vertebrate, insect, plant and fungal pests, periodic catastrophic fire, and climate change have all contributed to the deterioration in temperate and boreal forests (Lorenz et al., 2010; Millar & Stephenson, 2015; Godfree et al., 2021). Furthermore, studies have revealed the complex nature of forest decline, suggesting that the poor condition of forests has been due not only to the toxic impacts of pollutants and pests on tree health but is a result of contributory effects of soil eutrophication, acidification, and climate change (Nelleman & Thomsen, 2001). Also, whilst warming climates have been implicated in the expansion of boreal forests northwards and of temperate forest tree lines upwards in elevation, there have been increasingly severe outbreaks of pest insects in North America, Europe, and Siberia and a global increase in the frequency, intensity, and extent of forest fires (Locatelli et al., 2010; Millar & Stephenson, 2015; Shvidenko et al., 2005). Indeed, a wide variety of future impacts associated with climate change are predicted for temperate and boreal forests. However, the many interactions amongst the changing climate, storm intensity and frequency, wildfire, invasive species, and pathogens and the diversity of biotic responses amongst species to these pressures mean that specific predictions are generally highly uncertain (Locatelli et al., 2010; Park et al., 2014).

According to the Society for Ecological Restoration's (SER) *International Principles and Standards for the Practice of Ecological Restoration* (2nd edn; Gann et al., 2019), ecological restoration is the process of assisting native ecosystems that have been degraded, damaged, or destroyed to recover their ecological integrity and to facilitate as close a resemblance as possible to the ecosystems that would have occurred had they not been disturbed. This is just one of a set of restorative activities designed to conserve biodiversity, recover the ecological integrity and resilience of ecosystems, improve the quality and quantity of ecosystem services, and transform the way societies interact with nature (Gann et al., 2019). Temperate and boreal forest ecosystems have demonstrated remarkable resilience in the past century with the passive regeneration of temperate and boreal forests through natural seed dispersal on abandoned farmland in North America, Eurasia, and elsewhere (Shvidenko et al., 2005; Keenan et al., 2015; Chazdon et al., 2020). However, according to the SER standards, since ecological restoration is an intentional goal-oriented activity (Gann et al., 2019), the unplanned regeneration of abandoned farmland is better considered to be 'passive recovery' rather than active restoration. Moreover, the considerable expansion of temperate forest cover in recent decades due to afforestation, and estimated to be 150 million ha between 1990 and 2015 (Keenan et al., 2015), is better described as 'rehabilitation' rather than restoration. This accords with the standards, since the establishment of plantations of exotic species and native monocultures is inconsistent with the restoration of biodiverse native reference ecosystems.

It is relevant to note that forests are human-dominated (i.e. managed) ecosystems (Stanturf, 2005; Noble & Dirzo, 1997) and ecological restoration is a goal-oriented activity (Gann et al., 2019; Stanturf, 2005). This anthropomorphic dimension of restoration underlines the importance of community intentions, since these are the key driver of if, where, and what forest restoration occurs. People's motivations also define the outcomes of successful restoration. In this regard, the need to prioritise socio-economic considerations in restoration projects is well-recognised (Castillo-Mandujano & Smith-Ramírez, 2022): restoration is more likely to succeed when a thorough assessment of landowner or land manager and stakeholder needs and priorities inform restoration targets and when social and economic criteria are part of monitoring and project evaluation procedures. For example, in developing countries, project success is dependent on achieving worthwhile socio-economic development (Aronson et al., 2006), whilst in developed countries, economically profitable projects can be viewed as business management decisions and assessed accordingly. Alternatively, where profit is not the object, projects are driven by landowner or community visions and goals. It has been noted, however, that very few studies seeking to identify priority ecosystems for restoration, actually take social or economic criteria into account. Of 64 studies reviewed by Castillo-Mandujano and Smith-Ramírez (2022), 70% did not incorporate stakeholder considerations, and of the 88% of studies that had biodiversity conservation as their main goal, only 11% included social goals and 9% economic goals.

The human construct of ecological restoration, reflected in the SER standards, has another important implication, given that one or more centuries will be required for restoration plantings to achieve the mature or old-growth status of many reference temperate forest ecosystems worldwide (Petrokas et al., 2020; Anyomi et al., 2022). These long timeframes are complicated by forest succession processes, which produce multiple, not necessarily convergent trajectories that are dependent on the interplay between the life-history characteristics of a range of potentially dominant species and multiple disturbances at different spatio-temporal scales, all set against a variable backdrop of site conditions, climate, and management (Christensen, 2014; Arroyo-Rodríguez et al., 2017; Anyomi et al., 2022). In forests, these processes often result in sequences of seral vegetation states, involving the replacement of one assemblage by another (e.g. Noble & Slatyer, 1980; Santana et al., 2010; Anyomi et al., 2022). In such circumstances, a temporally variable schedule of criteria for monitoring and evaluating (M&E) forest restoration will be required to reflect the desired initial and subsequent successional states. Later stage criteria may not be relevant for some decades or even a century or more.

In view of these considerations, this chapter offers three very different case studies. The first two are temperate forest restoration case studies, whereas the third is an example of ecosystem rehabilitation, and each comes from a separate country.

The case studies highlight the importance of each project's socio-economic goals, as well as biophysical and biodiversity goals, and relate them to the success of each project. In turn, the diversity of goals amongst the projects explains the broad range of outcomes. In more detail, the three case studies are:

Case Study 1. This study examines the restoration of 407 ha of former grazed farmland on the east coast of Te Wai Pounamu South Island, Aotearoa New Zealand,[1] as an offset for a nearby landfill (rubbish dump) development. The project is funded by Transwaste Canterbury Ltd., a public–private partnership company that owns the landfill. The 200-year goal is to restore the area to native reference ecosystems of coastal broadleaved, mixed podocarp–broadleaved, and black beech (*Fuscospora solandri*) forest and associated wetland. A successional approach was adopted on commencement in 2004 with initial plantings of early successional native shrubs and trees and assisted natural regeneration.

Case Study 2. This case study documents the successful 10-year collaboration between diverse stakeholders and landholders on the Colorado Front Range Fuels Treatment Partnership Roundtable. The roundtable represents state and federal agencies, local governments, environmental organisations, scientists, and a community of >2 million people living in or adjacent to the 1.7 million ha of Colorado's Front Range in the eastern Southern Rocky Mountains. After devastating wildfires over the previous 30 years, in 2010 the Roundtable agreed to treat 13,000 ha of dense lower-montane forests of ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) to reduce the risk of catastrophic wildfire. They agreed to restore the pre-settlement grassy open woodland, characterised by a complex mosaic of stands of diverse structure and age classes with old-growth attributes, by a combination of mechanical thinning, cutting by chainsaw, prescribed burning, and weed control.

Case Study 3. The final case study describes a 43-year project to reforest the 750-ha sheep-grazing property, 'Taylors Run', in northern New South Wales, after basically all of the native timber was killed by Christmas beetles (*Anoplognathus* spp.) and other defoliating insects (New England dieback) between the 1950s and 1970s. Graziers, Jon and Vicki Taylor, self-funded their tree planting commencing in 1979, and their son, Michael, and his wife, Milly, continue to plant a wide variety of exotic and native trees and shrubs to provide shade and shelter for livestock, restore biodiversity and ecological balance to the farm ecosystem, manage their commercial radiata pine (*Pinus radiata*) resource, enhance amenity and the property's capital value, and achieve sustainable agroforestry and carbon-positive farming.

[1] This use reflects the bicultural nature of Aotearoa/New Zealand as derived from the 1840 Treaty of Waitangi.

Case Study 1: Restoration of Ex-pastoral Farming Landscapes in Aotearoa, New Zealand

Project Rationale and Strategy

Restoration of land that was previously used for pastoral farming is common in Aotearoa New Zealand (Norton et al., 2018). Such restoration addresses the substantial habitat loss associated with human settlement starting about 1280 AD (McGlone, 1983; Ewers et al., 2006) and enhances connectivity between remaining forest remnants in these landscapes (Zhang et al., 2021). Ranging from small riparian plantings to large landscape-scale projects, these restoration plantings are undertaken by groups as diverse as individual landowners, commercial companies, iwi (Indigenous tribal) incorporations, community groups, government agencies, philanthropists, and charitable trusts. Most of Aotearoa New Zealand below the climatic tree line was forested in historical times, so ex-pastoral farming sites are well suited for re-establishing native forest. This case study utilises one such restoration project as an example of how this approach to temperate forest restoration is applied in Aotearoa New Zealand.

Commencing in 2004, the 407-ha Tiromoana Bush restoration project arose as part of the mitigation for the establishment of the Canterbury Regional Landfill at Kate Valley (Norton, 2009). The site was used for sheep and beef farming up until 2004 and is located between 0 and 360 m above sea level on the east coast of Te Wai Pounamu (South Island). The site experiences an annual rainfall of 920 mm, falling mainly in the winter season. Prior to restoration, the vegetation was predominantly pasture, with some areas of kānuka (*Kunzea robusta*), a widespread and common early-successional tree of mixed-species native shrubland and low forest. The site was heavily grazed by sheep and cattle. Historically the area would have been forest, which was likely cleared 500–700 years ago as a result of fires associated with early Māori settlement (McWethy et al., 2009).

The vision for the project is to restore the site to native forest in 200 years' time. This will include establishing species such as coastal angiosperms, mixed podocarp–angiosperm, and black beech (*Fuscospora solandri*) forest, accompanied by small areas of wetlands. Six 35-year outcomes were identified. If these outcomes can be shown to have been achieved by 2039, it will indicate that the restoration is progressing towards the long-term vision. These six outcomes are:

1. Vigorous regeneration is occurring within the existing areas of shrubland and forest, at a sufficient rate to ensure that natural successional processes are leading towards the development of mature lowland forest.
2. The existing Korimako (Bellbird *Anthornis melanura*, a honeyeater) population has expanded and Kereru (*Hemiphaga novaeseelandiae*, a native pigeon) are now residing within the area. In addition, the species richness and abundance of native water birds have both been enhanced.

3. The area of black beech forest has increased with at least one additional black beech population established.
4. Restoration plantings together with natural regeneration have enhanced connectivity between existing forest patches.
5. Restoration plantings have re-established locally rare vegetation types.
6. The area is being actively used for recreational, educational, and scientific purposes.

Transwaste Canterbury Ltd. is a public–private partnership company that owns the landfill and has been active in its public support for the restoration project and in promoting a broader conservation agenda in the region. Shareholders of the partnership company are Waste Management NZ Ltd., Christchurch City Council and Waimakariri, Hurunui, Selwyn, and Ashburton District Councils. Day-to-day management is guided by a 5-year management plan and annual work plans. The management plan provides an overview of the restoration approach taken, while annual work plans provide specific on-ground restoration actions that will be undertaken to implement the management plan. Contractors undertake the majority of the work, which includes active planting, plant and animal pest control, walkway management, and ecological monitoring.

Major Project Concerns and Barriers

The major constraints to the project being successful related to the long history of habitat loss associated with human settlement and the fact that the site was a sheep and beef farm for the better part of a century prior to restoration commencing, with a dominant exotic grass sward over much of the site. Other concerns were the potential impact of domestic and feral grazing animals and weeds and the lack of seed sources of many native forest species close by. All these concerns were addressed through the management actions described in the next section and through the provision of a guaranteed funding stream (about NZ$350/ha/annum) from Transwaste Canterbury Ltd. as part of the resource consent requirements to operate the landfill.

Key Project Features and Major Project Outcomes

The main management actions undertaken and outcomes achieved to date have included:

1. The gazetting of an Open Space Covenant on the title of the property in July 2006 through the QEII National Trust (www.qeiinationaltrust.org.nz). This legal action has provided in-perpetuity protection of the site irrespective of future ownership.

Fig. 5.1 Dense native regeneration in the understorey of seral kānuka forest after removal of sheep and cattle grazing

2. Exclusion of browsing by cattle and sheep at the outset of the project through upgrading existing fences and construction of new fences. A 16-km deer fence has been built, which together with intensive animal control work by ground-based hunters, has eradicated red deer (*Cervus elaphus*) and has helped to reduce damage caused by feral pigs (*Sus scrofa domesticus*). This has allowed the understorey of existing shrubland and forest to recover (Fig. 5.1) and has also facilitated the expansion of natural regeneration into pasture areas as a result of natural seed dispersal (Fig. 5.2).
3. Strategic restoration plantings, which have been undertaken annually to increase the area of native woody (Fig. 5.3) and wetland vegetation. A weir was also established to create a larger wetland area, as well as provide food and nesting resources for native birds. A key focus of the plantings has been on enhancing linkages between existing areas of regenerating forest and re-establishing rare ecosystem types such as wetland and coastal forest. The oldest plantings now have natural regeneration of native species occurring in the understorey. While many of the planted species have been shorter-lived early successional species, there is now an increasing focus on interplanting with long-lived mature forest canopy-forming species such as the conifer, Tōtara (*Podocarpus totara*).
4. The undertaking of annual weed control focusing on species that (i) are likely to alter successional development, such as wilding conifers (mainly *Pinus radiata*) and willows (*Salix cinerea* and *S. fragilis*) or (ii) like Old man's beard (*Clematis vitalba*) have the potential to smother native regeneration. Gorse (*Ulex europaeus*) and Broom (*Cytisus scoparius*) are not controlled as these species have been shown to act as nurse plants for native forest regeneration and will eventually

Fig. 5.2 Expansion of predominantly native woody vegetation on the hillsides to the left of restoration plantings in the valley over time. (**a**) 2005 (top) and (**b**) 2021 (bottom). The radiata pines in the background are outside the boundary of the Tiramoana Bush restoration site

be overtopped. The cost and collateral damage associated with their control would outweigh native biodiversity benefits.

5. Establishment of a public walking track, undertaken early in the project, which has been subsequently enhanced and extended, with new interpretation material being included. The walkway goes through the heart of the restoration project down to the beach on the Pacific Ocean, before returning to the car park. It mainly follows old farm tracks and takes about 3.5 h to complete. Public access has been seen as a core component of the project from the outset so the public

Fig. 5.3 Ten-year-old restoration planting at Tiramoana Bush with naturally regenerating kānuka forest behind

can enjoy the restoration project and access a section of the coastline that is otherwise relatively inaccessible.

6. Part of the walkway upgrade included working closely with the local Māori tribe, Ngāi Tūāhuriri, who have mana whenua (customary ownership) over the area. They were commissioned to produce a pou ika (fisheries marker) at the walkway's coastal lookout (Fig. 5.4). The carvings on the pou reflect cultural values and relate to the importance of the area to Ngāi Tūāhuriri and especially values associated with mahinga kai (the resources that come from the area) in the coastal environment.
7. Regular monitoring has included assessments of bird, vegetation, and landscape characteristics, with additional one-off assessments of invertebrates and animal pests. The Tiromoana Bush restoration project has also been used as the basis for university student research projects.

What About the Project Worked, What Did Not Work and Why?

Important lessons learnt over the past 18 years have both shaped the approach to management at this site and have implications for the future management of other projects. These lessons include understandings such as:

1. Control of exotic browsing mammals, both domestic and feral, has been an essential factor in the success of this project. While domestic livestock were excluded at the outset of the project through fencing, feral red deer and pigs have

Fig. 5.4 The Pou Ika (Māori customary fisheries marker) on the walkway coastal lookout

been an ongoing problem and have compromised restoration outcomes, thus requiring additional management inputs such as specific deer fencing and culling.

2. Since removal of grazing, the dominant exotic pasture grasses, especially cocksfoot (*Dactylis glomerata*), now form tall dense swards. These swards severely restrict the ability of native woody plants to establish and thus herbicide control must be used both pre- and post-planting to overcome this factor. During dry summers (which are common), the grass sward is also a significant fuel source and the walkway is closed during periods of high fire risk in order to avoid accidental fires, which would decimate the restoration project.
3. Regular monitoring is important for assessing the biodiversity response to management. Annual photo-monitoring (now spanning 19 years) has been able to highlight significant changes in land cover across the site, while more detailed monitoring of plants and birds has strongly informed management actions. For example, 7 years of bird monitoring has indicated an ongoing decline in some native birds, a decline that is thought to be most likely due to predation (by cats, mustelids, rodents, and hedgehogs). As a result, a significant predator control programme commenced in 2019.
4. Simply removing grazing pressure from areas of existing regenerating native woody vegetation cannot be expected to result in the return of the pre-human forest because of the absence of seed sources. Permanent plots suggest that kānuka is likely to be replaced by māhoe (*Melicytus ramiflorus*), with few other tree species present. Gap creation and enrichment planting of a larger range of native species are therefore being used to speed up the development of a more diverse podocarp–angiosperm forest canopy (Tulod et al., 2019; Forbes et al., 2020).

The guaranteed funding stream from Transwaste Canterbury Ltd. for this project will continue for the life of the landfill (about 35 years). However, all restoration projects require funding for long time periods and even a project of this size will still require a degree of management after 35 years as it develops into the future, particularly in respect of plant and animal pest control. This has been addressed in part by Transwaste Canterbury Ltd. establishing exotic plantation forests on adjacent land they own (Fig. 5.2), which will provide a post-landfill income stream and provide potential land areas in which to expand the restoration project.

The project is guided by an annual 5-year management plan that sets goals and management actions. As we approach the 20-year milestone for the project, it is proposed to revisit the original management plan goals (which were part of the resource-consent conditions for operating the landfill) and based on the lessons learnt above, to develop a new 10-year management plan for the site. It is important to be adaptive in restoration management and to recognise that issues will change through time. The development of a longer-term plan is also important for managing project management succession, which is a particular challenge for long-term restoration projects.

Case Study 2: Collaborative Forest Landscape Restoration on Colorado's Front Range

Project Rationale and Strategy

Wildfire hazard has increased across much of the western United States resulting from historical land management coupled with global climate change. This case study describes one project that successfully sought to engage the community in ecological restoration at the landscape scale, which was specifically developed for both societal and ecological benefits.

Historically, prior to Euro-American settlement, the dry coniferous forests in the lower montane of Colorado's Front Range (1600–2850 m a.s.l.) were open woodlands dominated by ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) with a graminoid, forb, and shrub understorey maintained by low and mixed-severity wildfires with a 1–35 year return interval (Battaglia et al., 2018). In turn, the spatial complexity of the open woodland structure with a mix of individual trees, clumps of trees, and openings form a discontinuous distribution of wildfire fuels (Fig. 5.5) that maintained the low and mixed-severity wildfire regime (Larson & Churchill, 2012). Changes to forest management since the latter half of the nineteenth century, including removal of cattle and sheep grazing and fire suppression in these open woodland forests, has led to an increased density of trees and large extents of contiguous forest canopy (Fig. 5.6). The composition of Douglas-fir and other shade-tolerant and fire-susceptible species has increased, and this increased tree density combined with the drying effects of global climate change has led to increased fire risk.

Fig. 5.5 Prior to Euro-American settlement, the lower montane forests of the Front Range were open woodlands with individual trees, clumps of trees, and openings. This recently burned stand in Swan Valley in the Flathead National Forest in northwestern Montana is an example of the open woodland forest structures that were common in the Front Range. Dave Powell, USDA Forest Service (retired), Bugwood.org

Wildfire activity on the Front Range (the mountain range on the eastern side of the southern Rocky Mountains in central Colorado and southeastern Wyoming, adjacent to the cities of Colorado Springs, Denver, Boulder, and Fort Collins) has increased over time with numerous large, high-severity wildfires occurring over the last 30 years or so, including several that burned >4000 ha, destroyed homes, and adversely affected water reservoirs critical to the City of Denver (Addington et al., 2018). In recorded history, there have been three fires that were greater than 50,000 ha: the Cameron Peak fire (84,000 ha, 2020), East Troublesome fire (78,000 ha, 2020), and the Hayman fire (55,000 ha, 2002). All three were within the last two decades, and the two largest have been in the last 3 years (USDA Forest Service, 2022a). There are >2 million people living and working within 881 communities in the wildland–urban interface (WUI) of the Front Range (Front Range Roundtable, 2010), and the increased frequency of high-severity fires poses a risk to human life, homes, and infrastructure. Furthermore, these fires have also impacted critical habitat for a range of terrestrial and aquatic species, including the federally threatened Pawnee montane skipper (*Hesperia leonardus montana*) and the greenback cutthroat trout (*Oncorhynchus clarki stomias*) (Addington et al., 2018).

The increase in wildfire occurrence on the Front Range has catalysed action amongst a number of community collaborations. The Front Range Roundtable (formerly the Front Range Fuels Treatment Partnership Roundtable) is a coalition of state and federal agencies, local governments, environmental organisations, the scientific community, and the public, which formed following the Hayman fire in 2002. It aims to create a more resilient landscape through collaboration, sound land

Fig. 5.6 A typical lower montane forest stand on the front range prior to restoration. (Note the relatively closed canopy of similarly sized trees compared to Figs. 5.5 and 5.7)

management, and community engagement (Front Range Fuels Treatment Partnership Roundtable, 2006). In 2010, the Front Range Roundtable initiated the Colorado Front Range Landscape Restoration Initiative (CFRLRI) with US$18.8 million of funding over 10 years from the USDA Forest Service's Collaborative Forest Landscape Restoration Program (CFLRP). This initiative aimed to mitigate the increased fire hazard by restoring the forest structure and ecological processes of large areas of the lower montane forests of the Front Range and was just one of 23 projects across the United States funded through the CFLRP programme.

There are 1.7 million ha of forestland across the 10 counties that make up Colorado's Front Range. The Front Range Roundtable identified 325,000 ha of lower montane ponderosa pine and mixed conifer forests requiring ecological restoration. Of this total area, the 10-year CFRLRI sought to directly treat 13,000 ha within the Arapaho-Roosevelt and Pike San Isabel National Forests. The mission of the USDA Forest Service (2022b) is '… to sustain the health, diversity and productivity of the nation's forests and grasslands to meet the needs of present and future generations', and these national forests are managed for multiple purposes including (but not limited to) timber production, recreation, wildlife habitat, and watershed protection. This area was identified as high priority as its ecological restoration would also achieve community protection, watershed restoration, and habitat improvement goals.

The planned treatments aimed to shift the forest structure to an open woodland by:

1. At a stand scale, decreasing tree basal areas (the cross-sectional area/ha occupied by tree stems at a height of 1.35 m) and tree densities (the number of trees/ha), increasing quadratic mean diameter (quadratic mean of tree stem diameters at a height of 1.35 m), increasing fine-scale structural heterogeneity (sensu Larson & Churchill, 2012), decreasing litter and duff[2] depths, increasing the prevalence of ponderosa pine, increasing herb cover, maintaining wildlife use within treated stands, and protecting old-growth characteristics such as openings, snags, aged coarse woody debris, groups and clumps of trees, and large old trees.
2. Re-establishing a complex landscape mosaic, including increasing canopy gaps and openings, and enhancing forest structural and age-class diversity across the landscape.
3. Enabling more low-severity and mixed-severity fires, to reduce the likelihood of broad-scale forest loss due to high-severity, stand-replacing fire.

Major Project Concerns and Barriers

These efforts to mitigate wildfire hazard and restore historic forest structures and disturbance regimes have faced several major challenges. One of the biggest challenges has been the sheer scale of the restoration needed. While the Front Range Roundtable was able to identify 325,000 ha of the 1.7 million ha forested landscape needing ecological restoration, the CFRLRI had to strategically focus on restoring 13,000 ha over a 10-year period due to funding and logistical constraints. However, wildfire is a large landscape-scale process, and the area that the initiative is able to restore is dwarfed by the size of the wildfires that are becoming more common in the region.

The scale of the challenge is further exacerbated by the lack of a market-driven solution. Much of the restoration activities needed to focus on re-creating historic forest structures and reducing wildfire fuels through cutting trees; however, the steep terrain and lack of local markets for the small-dimension logs meant that restoration treatments were expensive with little opportunity to recuperate costs through product utilisation (e.g. timber, bark, and mulch for landscaping). In other regions, the presence of an active wood market utilising small-dimension wood may facilitate restoration by providing income to supplement restoration funding.

Lastly, there is concern about the need for social licence to operate (SLO), particularly given that the CFRLRI is focused on restoring public lands managed by federal agencies. While there is a large body of work indicating that the public is generally supportive of wildfire mitigation and forest restoration activities in the

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[2] The undecayed and decayed organic matter lying on top of the mineral soil.

southern Rockies (Kaval et al., 2007; Ostergren et al., 2008; McGrady et al., 2016), forest and fire managers across the western United States report encountering societal resistance to prescribed fire (Carroll et al., 2007; Hamilton & Salerno, 2020; Quinn-Davidson & Varner, 2012) and cutting of trees and concerns about a perceived lack of SLO persist. The short-term aesthetics, public familiarity, and perceived risks of restoration techniques such as tree cutting and prescribed fire are likely to be key factors in these challenges (Peterson & Vaske, 2016).

Key Project Features

A key feature of the Colorado Forest Restoration Initiative is the adoption of a collaborative adaptive management approach, with state and federal agencies, local governments, environmental organisations, the scientific community, and the public, having direct input into the initiative through the Front Range Roundtable (Aplet et al., 2014). This approach is exemplified by the ecological, social, and economic monitoring of treatments (Barrett et al., 2018, 2021; Clement & Brown, 2011). Effectiveness monitoring is an essential component of adaptive management, and members of the Front Range Roundtable were at the core of developing the desirable future conditions, writing the monitoring plan, and regularly reviewing monitoring outcomes. Furthermore, there were some components of the monitoring programme that were directly undertaken by members of the collaborative. Regular field trips for the collaborative partners to visit restoration sites were also a key part of this approach.

Major Project Outcomes

Between 2010 and 2019, the Colorado Front Range Landscape Restoration Initiative successfully reduced wildfire fuels and created desirable open woodland forest structures on more than 12,870 ha by cutting trees using either mechanical equipment on accessible terrain (Fig. 5.7) or manually using chainsaws on steeper terrain. This cutting of trees focused primarily on retaining older trees and ponderosa pine and the creation of complex forest structures by leaving a mix of individual trees, groups of trees, and openings. The primary goal of these activities was restoration and wildfire mitigation, with <5% of the area cut and the timber sold to augment restoration funding (Front Range Roundtable, 2019b). Fire disturbance regimes were returned to a further 2874 ha through the use of prescribed fires between 2016 and 2019 (Front Range Roundtable, 2019b). The risk of catastrophic wildland fire was reduced on 22,210 ha of the WUI by removing high-priority hazardous fuels. In addition, 15,170 ha of terrestrial habitat were restored, and 5580 ha were treated to manage noxious weeds and invasive plants (Front Range Roundtable, 2019b).

Fig. 5.7 A typical forest stand following mechanical treatment to restore historical forest structures. (Note the presence of clumps of trees with openings between. Ideally, these mechanical treatments will be followed by prescribed fire to reduce surface fuels, and mimic the natural disturbance regimes dominated by low-severity and mixed-severity wildfires)

In project areas, the basal area was reduced on average by 64% and tree density reduced on average by 78% (Front Range Roundtable, 2019a). In addition, the mean quadratic mean diameter increased following treatment, indicating that large trees were preferentially left as residuals in order to promote old-growth-like conditions (Barrett et al., 2021). In addition, the treatments were concentrated within watersheds, in order that 50% of the watershed was a mosaic of forested and non-forested patches of 4 ha or smaller and the remainder was a mosaic of forested and non-forested patches up to 10 ha (Front Range Roundtable, 2019a). However, the variation in tree density at fine scale (i.e. the patterns of individual trees, clumps of trees and small openings that contribute to local spatial complexity) was recalcitrant and did not increase following treatment (Barrett et al., 2021). These fine and landscape-scale mosaics are anticipated to result in a desirable mix of low and mixed-severity wildfires, which is characteristic of the historical fire regimes.

Songbird point count surveys indicate that bird species varied in how they were distributed with respect to treatments at a fine spatial scale. Both positive and negative relationships were observed as species were distributed unevenly between treated and untreated areas depending on time since treatment. However, while diversity at treated areas varied little from untreated areas at the fine spatial scale, species richness was higher across treated landscapes.

What About the Project Worked, What Did Not Work and Why?

1. The Colorado Front Range Landscape Restoration Initiative effectively used a collaborative adaptive management approach (Aplet et al., 2014; Barrett et al., 2018), capitalising on the existing partnerships of the Front Range Roundtable. Stakeholders were explicitly involved in the adaptive management cycle, particularly during planning and monitoring (Aplet et al., 2014). While much of the restoration was implemented by the U.S. Forest Service through stewardship contracts, stakeholders were collaboratively engaged in designing desirable conditions based on scientific reconstructions of historical and ecological information (Addington et al., 2018; Dickinson and Spatial Heterogeneity Subgroup of the Front Range Roundtable, 2014) to design monitoring protocols and to assess treatment outcomes (Barrett et al., 2021; Cannon et al., 2018). For example, historic ranges of variation (HRV) based on ecological reconstructions of the forest structure prior to extensive European colonisation in the late 1800s were used to describe the desirable conditions. This provided clear goals for restoration, while also allowing some flexibility to adjust goals to reflect predictions of future climate change. While HRV indicated a range of acceptable tree densities, values at the lower end of the range were selected in areas where climate change would be expected to increase the risk and severity of wildfire. Initially, these desirable conditions were largely qualitative but further research over time provided a basis for quantitative goals that could be used as monitoring benchmarks (Battaglia et al., 2018; Dickinson, 2014). Furthermore, initial monitoring of restoration outcomes indicated that there was a need to further reduce tree densities, increase gap sizes and frequencies, and to favour ponderosa pine (Cannon et al., 2018). The adaptive management approach resulted in improved outcomes for most monitoring benchmarks over time (Barrett et al., 2021).
2. In order to ensure positive outcomes from the collaborative adaptive management approach, it was important to regularly engage stakeholders through informal means such as quarterly meetings and field trips. However, there was also a need for more formal means of engagement to provide evidence that stakeholder recommendations were successfully incorporated into planning and implementation (Beeton et al., 2020; Cannon et al., 2018). Furthermore, regular engagement was important to ensure that relevant stakeholders were engaged at the right time and in an appropriate manner (Beeton et al., 2020). Unfortunately, wildlife monitoring of the restoration treatments only began several years into the 10-year project, due to lack of engagement with stakeholders interested in this aspect of the project.
3. Maintaining resources for the programme in the longer term will be challenging. The initial funding for the Colorado Front Range Landscape Restoration Initiative spanned 10 years, which could be considered a relatively long lifetime for a US federal programme (USDA Forest Service Washington Office CFLRP Staff, 2020). However, 10 years is relatively short for ecological restoration, particularly when considering the protracted National Environmental Policy Act

planning process on United States federal lands and the landscape scale of the project.

4. While the security of working on federal lands with long-term land tenure is an advantage and may offset the funding challenge, it also means that restoration activities must abide by other federal policies that may act as institutional barriers. For example, the creation of large openings was restricted to <0.1 ha in some areas due to pre-existing forest plans (Cannon et al., 2018). Furthermore, the relatively low use of prescribed fire in the first years of the project could be, in part, attributed to institutional barriers and the limited social licence to operate.
5. In addition to recreating ecological composition and structure, there is a need to reinstate ecological processes (Addington et al., 2018). While open woodland forest structures were restored to over 12,870 ha through mechanical or manual treatments, only 2874 ha were burnt with prescribed fire. Historically, these forests were maintained by a frequent, low to mixed-severity fire regime that created structurally complex open woodlands with relatively low densities of trees. Without the reinstatement of this ecological process through either prescribed fire or wildfire, it is likely that trees will quickly regenerate following treatment and return to pre-treatment conditions, particularly on more mesic sites (Fialko et al., 2020).
6. Working at scale is necessary if the aim is to influence ecological structure and function, given that wildfire is a landscape-scale process (Addington et al., 2018). Working at scale may also provide scope for greater efficiency. However, large spatial scales require longer timeframes given the logistics of applying treatments (USDA Forest Service Washington Office CFLRP Staff, 2020). At times, a 'go slow to go fast' approach is needed with long timeframes for adequate planning; but once planning is in place, treatments can be applied over large areas efficiently (USDA Forest Service Washington Office CFLRP Staff, 2020).

Case Study 3: Agroforestry at 'Taylors Run': Returning Tree Cover to a Grazing Property on the New England Tablelands, Northern New South Wales

Project Rationale and Strategy

Returning woody vegetation to farmland that was originally temperate forest or woodland often consists of establishing and developing agroforestry systems tailored to the commercial needs of the farm. Ideally, these novel ecosystems of trees and shrubs should (i) complement the existing agricultural enterprises, (ii) thrive in the farmland environment, and (iii) provide new income streams and important ecosystem services. Trying to restore the natural forest and woodland of the district is rarely the objective of farmers and could interfere with farm income and profits.

This case study of the rehabilitation of the 750-ha property, Taylor's Run, Kentucky, NSW, Australia, describes the former approach of developing a novel agroforestry system, which has proven successful for two generations of landowners.

The Taylor family have lived and farmed at Taylors Run for six generations, ever since their ancestor, William Tydd Taylor, purchased the 'Terrible Vale' run (including the portion now known as Taylors Run) shortly after European settlement in 1840. Elevation is 1020–1120 m a.s.l., and the climate is temperate with warm summers (mean January maximum, 27 °C), cool winters (mean July minimum, −0.7 °C), and a mean annual rainfall of 785 mm with a slight summer (October–March) dominance (62%). Michael and Milly Taylor took over management of Taylors Run in 2012 from Michael's parents, Jon and Vicki, who had managed the farm (then known as 'The Hill') since 1980, having taken over from Jon's father before them. The chief enterprise of the farm has always been fine wool, with stocking rates varying with the seasons between 4.6 and 6.1 sheep/ha over the past three decades. Wool cut per sheep has consistently averaged around 2.9–3.2 kg/head. Cattle constitute up to a quarter of the livestock (in terms of feed requirements) in good seasons, with up to 150 breeding cows, but these cattle are sold off in drought as necessary.

The main impetus for reforestation of Taylors Run was 'New England dieback', the chronic and severe defoliation of native eucalypt trees that started in the 1950s (Mackay et al., 1984; Landsberg & Wylie, 1988; Reid & Landsberg, 2000). Native tree cover had been deliberately retained over part of the property for livestock shade, shelter, and amenity (Taylor & Taylor, 2004). The original old-growth grassy eucalypt forests (≥30% tree cover) and woodlands (<30% tree cover) on the New England Tablelands were maintained by Aboriginal burning up to the time of European settlement. The first settlers ring-barked (rung) the timber to increase pasture growth for their sheep flocks. Because eucalypts typically regenerate quickly and thickly under such conditions, the regrowth forests (>30% tree cover) had to be periodically rung and the eucalypt seedlings 'sucker-bashed' to maintain areas of open pasture and woodland for livestock grazing. Progressive tree clearing and removal of regrowth timber occurred on Taylors Run, particularly in the periods 1860–1890 and 1910–1930, with about 25% of the farm retaining tree cover in the mid-1950s. Annual aerial fertiliser application of superphosphate began in 1956 in conjunction with seeding of introduced temperate legumes (white clover *Trifolium repens* and sub clover *T. subterraneum*) and later temperate pasture grasses such as cocksfoot (*Dactylis glomerata*) and ryegrass (*Lolium* spp.). Fertilised and sown pastures led to increased pasture biomass and stocking rates, with sheep numbers doubling over a 10-year period. This scenario, combined with previous tree clearing, led to an ecological imbalance resulting in one of the most severe tree declines worldwide (Mueller-Dombois, 1990). Jon remembers '… *walking through a grove of eucalypts one summer day in 1956 with* [his] *father and a CSIRO entomologist. The air was full of the humming of Christmas beetles* [*Anoplognathus* spp., Scarabaeidae, Fig. 5.8]*, which had suddenly appeared, eating the leaves of the trees. The trees were being defoliated continually,* [which was] *sickening and killing them. After we started aerially supering, most of the native trees were affected by dieback and died, including the small regrowth trees, which were between 15–20 years old.*

Fig. 5.8 Christmas beetles (*Anoplognathus* sp.) chewing young leaves of Fuzzy box (*Eucalyptus conica*)

A lot of trees died in the 1960s, being defoliated two or three times a year. The surviving trees were so sick, they didn't set seed. [However,] *if they had* [set seed]*, the seedlings would have been grazed off by the higher stock numbers. You don't see regrowth coming on or surviving in fertilised country.* [As a consequence,] *between 1967 and 1990, nothing came up anywhere, and the tree ecology all fell in a heap. It started noticeably in 1956 and the impact continued strongly until 1970 by which time there were not many trees left*'.

After dieback had denuded Taylors Run of native tree cover, Jon and Vicki commenced a tree planting programme in 1979. Their initial motivations were both aesthetic and pragmatic. The treeless landscape 'didn't look right anymore' and the increased exposure in the New England winters was hard on livestock, particularly young lambs and recently shorn sheep. Initially, the Taylors planted bare-rooted radiata pine (*Pinus radiata*) seedlings, which were cheap (AUS$0.15) and readily available, unlike seedlings of other exotic trees and native trees, which were difficult to source and cost AUS$2 per tube. Trees native to the northern hemisphere, like radiata pine, had been widely planted in gardens, windbreaks, and along driveways in the region and had the advantage of not being attacked by the many herbivorous insect taxa responsible for eucalypt dieback. However, by 1982, the Taylors were planting significant numbers of native and other exotic trees for diversity's sake and to avoid an overstorey monoculture. Nevertheless, for cost, reliability, and a commercial timber resource, radiata pine remained the mainstay of the planting programme. Today, when conditions permit, Michael continues to plant 1–2 ha of pine annually as well as another 1–2 ha of native and other exotic tree and shrub species. An estimated 250,000 trees have been planted on Taylors Run over the past 43 years, and 20% or more of the farm is now timbered, almost all of it established with the machine planter (pines) or hand planted (other species). Some oaks and acacias have been direct-seeded, as well.

Major Project Concerns and Barriers

Initially, stock shelter was the most pressing reason for planting trees, thus most of the plantings until 1990 were linear or block configurations along paddock fence lines or in mid-paddock situations, particularly on rises, to provide protection for livestock and pastures. Having established some initial shelter, Jon and Vicki experimented with other planting designs. Encouraged by Ron Watkins from 'Payneham Vale', Frankland, Western Australia (Norton & Reid, 2013, p. 187), they undertook their first whole-paddock contour planting in 1992. After planting lines and blocks of trees fenced from livestock for a decade or more, they wanted to encourage shelter and provide habitat for biodiversity throughout their paddocks in the same fashion as open woodland. They did this by selecting a 63-ha paddock and planting double rows of unfenced trees on the paddock contours (to intercept and use surface and subsurface flows of water) with an average distance of 60 m between the double rows. This was varied to a maximum of 130 m on flatter ground and as little as 30 m on steeper rises. After preparing the planting lines and grazing the paddock intensely to control weeds, the stock were removed and the entire paddock planted in double rows, one row of pines for commercial timber, and another of native trees and tall shrubs to provide shelter and biodiversity benefits after the pines were harvested. For the first couple of years, there were virtually no stock allowed in the paddock to protect the plantings, but from the first winter, small numbers of young cattle and lambs were introduced and allowed access for short periods to eat the grass. Jon and Michael learnt how to judge the appropriate length of grazing of livestock in whole-paddock plantings through observing how long a new flock of lambs or mob of young cattle remained eating the pasture before they raised their heads and started to browse the trees. This varied with the class of livestock, the season, and the amount and quality of feed in other paddocks. By the fifth or sixth year, the plantings were at least 3 m high, which allowed normal grazing to resume (Fig. 5.9). The Taylors have subsequently undertaken whole-paddock contour plantings in two other large paddocks and parts of three others.

Tree and shrub species selection was originally something of a headache for the Taylors. The adage of returning local native species to ensure that planted trees and shrubs thrived did not apply: most eucalypts were heavily attacked by Christmas beetles and many other defoliating insects (Heatwole & Lowman, 1986; Lowman & Heatwole, 1992), as well as being killed by frost or waterlogging in the lower parts of the landscape and in the open, due to the loss of the ameliorating tree cover (Fig. 5.10). Wattles (*Acacia* spp.) and casuarinas (*Casuarina* and *Allocasuarina*

Fig. 5.9 Contour paddock plantings in Top and Middle Sugarloaf Paddocks with Lower Sugarloaf Paddock beyond (top right with cloud beyond)

Fig. 5.10 A young Blakely's red gum (*Eucalyptus blakelyi*) planted low in the landscape near Salisbury Waters in Shugg Paddock, showing damage by frost and skeletoniser, lerp (Psyllidae), and chewing insects

spp.) also suffered from frost and waterlogging. Any shrubs <2 m in height, both native and exotic, were browsed and killed once livestock were let back into whole-paddock plantings. However, with a reduction in fertiliser application and the frequency of pasture sowing over time together with the gradual increase in tree cover, native tree health and predators and parasitoids of defoliating insects, such as insectivorous birds and wasps, scattered native paddock trees and native plantings have begun to thrive, particularly the large block plantings permanently fenced from livestock and the remnant paddock trees high in the landscape.

Taylors Run is dissected by St Helena Creek and the permanent Salisbury Waters. Management of the riparian zones required a different approach to the rest of the property because, if permitted, livestock naturally congregate and preferentially graze in riparian zones, overgrazing the pasture, exacerbating gully and sheet erosion in waterways, and fouling the water. Fencing the creek in a long narrow paddock (Long Frog Paddock, Fig. 5.11) allowed the riparian zones to be managed separately from the rest of the farm. This allowed the grass to regenerate and revegetate the river flats, providing good feed reserves when required and reducing erosion. The floodplain and banks were planted to trees, slowing down flood waters and allowing the sediment from upstream (free fertiliser) to be deposited across the

Fig. 5.11 Paddock plan for Taylors Run, showing paddock names and areas (in ha), and unstocked areas (reserves and dams)

Fig. 5.12 Deciduous tree species (poplars, elms, and willows) flourishing alongside Salisbury Waters in Long Frog Paddock with a native planting, Frog Forest, and six remnant New England peppermint (*Eucalyptus nova-anglica*) trees in the middle distance on the left. A block planting of radiata pine (Shaft Hill Reserve) can be seen in the (centre) background

creek flats. The Taylors found that the riparian zones on the farm are difficult to revegetate with native trees and shrubs due to the severity of the frosts, waterlogging, and insect damage. However, a variety of northern hemisphere deciduous trees thrived (Fig. 5.12) and encourage abundant pasture growth beneath. Pasture production is better around the base of some trees than others, especially beneath poplars, willows, and natives such as Rough-barked apple (*Angophora floribunda*).

Native mammalian herbivores have increased with the increasing tree cover in recent decades. Jon noted in 2003: *'Kangaroos are a constant problem with fence destruction mostly. When I first left school in the late* [19]*60s, there were only one or two kangaroos on the property. Nowadays, there's a resident population of 40–50 all year round … Before, we only used to have Eastern Grey Kangaroos* [*Macropus giganteus*]. *Now, there are also a lot of wallabies and black wallaroos* [*M. robustus*]. *The swamp wallaby* [*Wallabia bicolor*] *is a tree browser. They've come in since the Treefest site got going … The wallaroos come in around the Treefest site mostly, but now we are starting to see them in a lot of other areas … Some kangaroo culling has been required to alleviate the rapid build-up in numbers, especially in dry seasons when they flock to our destocked tree areas'*. These days, Michael employs a commercial harvester to control and utilise the expanding kangaroo numbers. The NSW Government issues licences for both the commercial harvest and non-commercial culling of kangaroos (DPE, 2022a).

Although weeds have not been a major issue with the exclusion of livestock from timber belts and block plantings, blackberry (*Rubus fruticosus* sp. agg.) and horehound (*Marrubium vulgare*) occasionally invade fenced enclosures and require control. The sheep suppress these species in grazed country. Although dense radiata pine plantings exclude most understorey growth, one of the advantages of thinning and high-pruning the pine blocks is that the extra light allows naturalised pastures to thrive, dominated by weeping rice grass (*Microlaena stipoides*), a valuable year-long green native species.

Key Project Features

An economic analysis of the benefits and costs of implementing whole-paddock contour plantings over 11% of the property in 10 years yielded a net present value (or profit) of AUS$113/ha, despite planting costs of AUS$145/ha and a discount rate of 5% (Thompson, 2006). The analysis assumed no overall loss of carrying capacity and gradual reductions in adult sheep mortality of 50% and increases in lambing percentage from 80% to 90% over 20 years, due to the beneficial effects of increased shade and shelter on livestock and pasture production. Indeed, wool production varied little over the decade during which up to 11% of the property was out of production due to tree planting (Taylor & Taylor, 2004). Moreover, during the 1994, 2002, 2013–2014, and 2017–2019 droughts, Jon and Michael were able to graze for brief periods the tree-planted paddocks, which contained large amounts of feed because the stock had been excluded. So, the Taylors' tree planting programme has paid for itself four times over.

A change in grazing management in recent years has led to natural regeneration of the few remaining native eucalypts that have persisted in paddocks since the first half of the twentieth century. Eucalypt regeneration is suppressed completely by commercial flocks of sheep that are set-stocked or grazed in long rotation, which means they are moved in rotation to a new paddock every few weeks or months. However, eucalypt regeneration often occurs abundantly in paddocks grazed by sheep in short rotation, which sees a paddock grazed perhaps just four times a year for a few days at most (Wright & Wright, 2004). Michael adopted this type of grazing management – high-intensity, short-duration, long-rest grazing – in 2012, and young eucalypt saplings are now starting to appear in paddocks around remnant paddock trees, despite periodic grazing. Particularly noteworthy is the regeneration of several stands of New England peppermint (*Eucalyptus nova-anglica*) and White or Ribbon gum (*E. viminalis*), which are being monitored by B. Vincent, T. Paine, and R. Andrew (pers. comm.). The former is a 'critically endangered' ecological community under both state and federal legislation (DPE, 2022b), and the latter is 'endangered' under state legislation (DPE, 2022c).

Major Project Outcomes

An intended longer-term aim of Jon and Vicki's tree planting programme was establishing an on-farm radiata pine timber resource and enterprise. This came to pass, but their interest in propagating tree species cheaply for planting on-farm resulted in additional commercial opportunities, commencing in the 1980s. These strengthened the socio-economic resilience of the farming business. The new enterprises included a contract tree-planting business: Jon developed a machine tree planter for planting bare-rooted pine seedlings at Taylors Run. Because dieback had left many other properties denuded of tree cover, Jon contracted to plant radiata pine windbreaks throughout the district in the 1980s. After a trip to the US researching tree-propagation and planting equipment, Jon and Vicki established Taylors Treeline Pty Ltd., a planting equipment company, and introduced the Swedish Hiko tray system into Australia in 1991. Hiko propagation trays and other planting-equipment technology revolutionised native plant propagation and revegetation across Australia, with Hiko tubes reducing costs to AUS$0.40 per seedling. It also allowed Jon and Vicki to experiment with a much wider variety of native and exotic trees and shrubs at Taylors Run, many of which they raised in their on-farm nursery. The decade of the 1990s saw the Australian wool industry in crisis due to low prices caused by the collapse of the federal government's wool reserve price scheme in 1991 (Abbott & Merrett, 2019). The Taylors were able to ride out this difficult period and retain their award-winning fine-wool enterprise through income diversification generated by their tree-related businesses. Jon and Vicki also commenced managing their radiata pine plantings for commercial timber production from an early stage, thinning and high-pruning the better stands to produce high-quality clearwood logs. They established an on-farm sawmill and purchased a log peeling machine to produce roundwood poles. They found a commercial market for the pine thinnings in a pole-impregnation plant in Tamworth, 80 km away, and Jon and subsequently Michael have been selling sawn timber at the farm gate at retail prices since 2007.

One of the most exciting outcomes of the Taylors' development of agroforestry is that the farm now has a net positive carbon balance, sequestering more carbon than the equivalent in greenhouse gas emissions, due to the growing agroforest. Vanguard Business Services undertook a natural resources audit of Taylors Run in 2021, quantifying emissions and sequestration across the farm. Net carbon sequestration was 212 t CO_2-e/year, representing the difference between total emissions of 960 t CO_2-e/year and estimated sequestration of 1172 t CO_2-e/year (Gardner et al., 2021). Total emissions were the sum of on-farm emissions (from fossil fuel use, fertiliser application, and ruminant livestock emissions), electricity use, and relevant pre-farm emissions (production and transport of fossil fuels and production and transport of purchased inputs including livestock, fodder, grain, and amendments). Carbon sequestration estimates were conservative, based on the above- and below-ground biomass of the trees as well as coarse woody debris, but none associated with the soil or pasture. Despite the potential revenue from the Australian Government's Carbon Farming Initiative (DAWE, 2022) and earlier programmes, the Taylors have not been tempted to sign long-term agreements that would limit their options and that of future generations for managing the farm, given the modest returns from the government's carbon farming schemes to date.

The tree planting programme has had a range of biophysical impacts, including positive effects on native biodiversity in addition to those noted earlier. Koalas (*Phascolarctos cinereus*) occurred historically in the native woodland and forest on

Taylors Run but fur trappers extinguished the local population in the 1860s. Jon's father had never seen a Koala on the property until 1996, when the first Koala for the better part of a century was seen in a 7-ha hilltop site with several large New England stringybarks (*E. caliginosa*). These had been fenced off and planted with native trees and shrubs in 1992 as part of the first national 'Treefest' expo (Southern New England Landcare, 2012). Since then, quite a few Koalas have been seen on the property, including young ones, both in radiata pines (where trees to be thinned have had to be left due to the presence of Koalas) and in planted eucalypts. Radiata pines in parks and gardens in the region are disproportionately attractive to Koalas (Carr et al., 2017) as roost trees for the dense cover they offer.

The presence of echidnas (*Tachyglossus aculeatus*) has also increased on the property. In 2003, Jon said *'We used to see one every couple of years, but for the past 3 years now, we have seen four or five per year, and we have seen families of two or three following each other nose to tail. There's lots of evidence of them in the fenced off areas – didn't use to see much evidence of them,* [but] *now it's easy to find their scratchings'*.

Native bird numbers fluctuated greatly in Jon's lifetime: *'Bird numbers hit a real low at the beginning of the* [19]*60s, just after we had lost a lot of trees. Everyone was using a lot of organochlorine (dieldrin) and organophosphates in those days for jetting* [which means applying insecticidal solutions to the fleece of] *sheep. There were hardly any birds left. Bird numbers were still pretty low when we started planting trees at the beginning of the* [19]*80s. Now there's 20 to 50 times more … The* [Australian] *magpie* [*Gymnorhina tibicen*] *numbers that got down to 30–40 on the place have increased enormously'*. Bird surveys conducted on 20 sheep properties in the region in the summer of 2002–2003 demonstrated the positive impact of reforestation on avian diversity and abundance. The windbreaks of introduced trees (predominantly radiata pine), block plantings of radiata pine and whole-paddock contour plantings at Taylors Run, contained three times more native bird species and four times more individual birds than open pastures without trees. These differences were statistically significant (Reid et al., 2006). Native block plantings at Taylors Run supported as many bird species and individuals as native forest and woodland on farms elsewhere in the region.

What About the Project Worked, What Did Not Work and Why?

The Taylors are cautious about exaggerating the financial benefits they have achieved from their transformation of Taylors Run through tree planting and their many other initiatives. These include the careful riparian zone and farm dam management carried out to ensure clean drinking water for livestock and the new business opportunities arising from the tree planting programme and radiata pine timber resource. They believe Taylors Run is capable of the same livestock production as it was before tree planting, despite the fact that 20% or more of Taylors Run is now timbered. In other words, the extra production resulting from the increase in shade, shelter, biodiversity, and water quality at least compensates for the land taken out of production with trees. Reforestation has also reduced risk by reducing the Taylors' exposure to extreme climatic events, diversifying tree cover to include a much wider variety of exotic and native species (to avoid another dieback disaster) and by establishing new tree-based enterprises. Michael estimates that over 400 species of tree and shrub have been planted on the farm. Over 300 native and exotic species alone were planted by many different commercial tree planting contractors and not-for-profits from across Australia at the Treefest site in 1992. In addition to radiata pine, the exotic species planted on the farm include a range of poplars and willows, which are easy to propagate from cuttings and to source locally, and are frost-hardy, but also elms, maples, and birches and even fruit trees along the creek. Exotics are the only species that can be reliably planted on flats and along creeks, owing to the lethal impact of frosts and waterlogging on native trees planted low in the landscape. Jon and Vicki collected over 80 species of oak in México in the 1990s, and Michael and Milly continue to harvest and sow acorns from the original plantings. Large-seeded native acacias also proved easy to propagate, and 15–20 species have been planted on the farm, but only successfully higher in the landscape.

The ecological and economic resilience of the farm was severely tested by the recent 2017–2019 drought, which was the driest 36-month January–December period on record in NSW (BOM, 2022). This was followed by two excessively wet years. The soils, vegetation, and landscape largely withstood the physical stress of these wild climatic swings. However, all the willows, about 5% of the radiata pines, many wattle-leaved peppermints (*Eucalyptus acaciiformis*) and snow gums (*E. pauciflora*) and some cypresses (*Cupressus* spp.) died at the height of the drought due to water stress. By the end of 2019, the drought had become so severe that Michael was just weeks away from selling all the remaining sheep (despite the decades of investment in award-winning Merino genetics). If this had occurred, the radiata pine timber business would have been their economic fall back.

Through their impressive work in environmental repair, plantation development, and the wool industry, the Taylors have won industry recognition in each area. In addition to an international Zegna wool award in 2002, they won the Royal Agricultural Society of NSW's Regional Ibis Award in 1996 for good conservation practice and the biennial Australian Forest Growers' National Farm Forestry Award in 2000, in recognition of their agroforestry achievements. More recently, in 2013, they won a Landcare Australia award for Innovation in Sustainable Farming Practices, and in 2022, Michael was named Australian Farmer of the Year in the Kondinin Group and ABC Rural annual awards (Burt, 2022).

Taylors Run is an excellent example of how development of novel agroforestry systems in a temperate forest environment through thoughtful, practical, entrepreneurial management can improve the triple bottom line. The Taylors understandably feel good about their achievements, as they have a comfortable lifestyle and a range of economic options, the farm environment is steadily improving, biodiversity has increased under their management, and their business is carbon-positive, despite the farm's reliance on fossil fuel-based energy inputs such as diesel and mains electricity.

Lessons Learnt

1. 'Improved' pastures – fertilised and sown pastures of exotic grasses and legumes – proved to be a two-edged sword in New England. While fertilised and sown pasture technology transformed southern Australia's livestock industries in the latter half of the twentieth century (e.g. McDonald, 1968), it was and is responsible for New England dieback on the Northern Tablelands of NSW.
2. Remnant native vegetation: despite losing almost all of their remnant native timber to New England dieback, the Taylors found that some paddock trees eventually improved in health if they were included in 'revegetation corridors' such as planted laneways or in fenced windbreaks and conservation zones, such as the Treefest site. Presumably defoliating insects were eventually controlled to a degree by the surrounding woody vegetation biomass and the harbour it afforded predators and parasitoids of the herbivorous insects. Jon and Vicki fenced one small rocky hillock with several native trees as a conservation initiative. To their delight they found it encouraged hundreds of volunteer seedlings of native trees and shrubs that did not have to be hand planted.
3. Whole-paddock plantings on the contour effectively combine productivity (timber and grazing), biodiversity, and shelter. Grazing these paddocks while the young tree and tall shrub canopies are within browsing height is possible with careful observation and management. Whole-paddock plantings that build up a pasture reserve while the trees are still young save stock and money during droughts. According to Michael, the paddocks also show better pasture growth, even after the trees have grown beyond stock browsing height and the paddocks are again regularly grazed, presumably due to the shelter and nutrient redistribution provided by the trees.
4. Shrub species selection: incorporating shrubs <3 m in height in unfenced plantings in grazed paddocks is ill-advised, as stock browse and kill them. Shrubs, however, can survive in 'special' fenced conservation or niche areas on farm, such as rocky knobs, wildlife corridors, and around farm dams that are permanently fenced from stock for reticulated clean drinking water. The clean water, which the stock prefer, is reticulated to troughs outside fenced dams.
5. Radiata pine and livestock shelter. Blocks and strips of radiata pine have been planted in most paddocks and have changed sheep grazing patterns. In summer, stock appreciate the shelter during the day and generally only come out to graze early and late in the day. Paddocks with lots of shelter by way of dense tree plantings are invaluable immediately after shearing and during lambing in late winter and early spring each year, when cold wet windy weather can cause heavy stock losses (Geytenbeek, 1962; Egan et al., 1972; Lynch et al., 1980; Donnelly, 1984).

Chapter Synthesis

Differences or Similarities of Approach Between These Case Studies and Elsewhere

These three case studies span a range of ecological restoration and rehabilitation objectives and issues. Case Studies 1 and 2 are examples of ecological restoration (sensu the SER standards; Gann et al., 2019), where the objectives are to restore native ecosystems according to local reference ecosystems as in the case of Tiromoana Bush (Case Study 1) or according to the pre-European settlement reference of frequently burnt, old-growth open-forests and woodlands in the case of the Colorado Front Range (Case Study 2). By contrast, Taylors Run (Case Study 3) is an example of ecological rehabilitation, where the objective was to build and sustain a novel model of sustainable agroforestry production. The Taylors' actions have improved the quality and quantity of the numerous ecosystem services that underpin the farm's agricultural and forestry enterprises, enhanced native biodiversity and recovered some of the lost local ecological integrity, strengthening the resilience and commercial profitability of the business, and demonstrating a transformative solution to the way in which Western agriculture traditionally interacts with nature.

There is nothing particularly unusual about these objectives. In relation to case study 1, restoration of forest on former farmland is now commonplace in New Zealand (Norton et al., 2018) and in other temperate zones around the world (Stanturf & Madsen, 2005; Stanturf, 2016), as well as in the tropics (Uhl et al., 1988; Parrotta et al., 1997; Holl et al., 2000; Lamb et al., 2005; Griscom & Ashton, 2011).

Regarding case study 2, the conceptual knowledge to restore extensive and uniformly managed forests whose resilience has declined over time, predisposing them to catastrophic decline or destruction by pests, pathogens, or wildfire, comes from panarchy theory and the adaptive cycle of ecosystems (Gunderson & Holling, 2002; Higgins & Duane, 2008; Egan, 2007). Given climate change and a warming planet, the frequency and intensity of weather that increases fire risk and the area burnt by wildfires has already increased (IPCC, 2022). The incidence of fire is likely to increase further, at least in arid, temperate, and boreal zones (Moritz et al., 2012). Hence, fire management will increasingly need to avoid the 'rigidity trap' associated with fire suppression policies and the pathological inability of some bureaucracies to innovate in wildland fire management (Butler & Goldstein, 2010). The evidences that forests with old-growth attributes are less likely to carry wildfire, as in the case of Colorado's Front Range, has parallels in other temperate forests (Zylstra et al., 2022). In southern and eastern Australia, for example, the shrub understorey fuel load in eucalypt forests and woodlands declines after 50 years or so (Croft et al., 2016), with a declining likelihood of wildfire in older forests (Zylstra et al., 2022). At Taylors Run, the Taylors are well aware of the risk of wildfire, particularly in the stands of pines and native trees. They consider the disconnected nature of the plantings, the intervening plantings of deciduous species of lower flammability, and the green *Microlaena* pastures beneath the managed stands of pine, all lower the risk of unmanageable fire.

In relation to case study 3, there is nothing surprising about the need for sustainable agriculture. Indeed, the FAO (2014, 2016) has argued that sustainable food and agricultural production is key to achieving all 17 of the United Nations' Sustainable Development Goals.

The issues and 'restoration barriers' that these case studies have had to deal with and overcome are also widespread and frequently encountered. In the case of Tiromoana Bush (Case Study 1), the need to control weeds that could otherwise dominate vegetation or block or divert recovery are frequent issues in ecological restoration worldwide (D'Antonio & Meyerson, 2002; Weidlich et al., 2020). Ungulates, whether they be domestic (such as cattle and sheep), feral, or native (e.g. in Europe and North America), can profoundly affect the dynamics and composition of temperate forest understoreys, particularly palatable woody species, worldwide (e.g. Hester et al., 2000; Lunt et al., 2007; Norton, 2009; Dodd et al., 2011; Bernes et al., 2018). Large native mammalian herbivores in Australia are similarly influential (Leigh & Holgate, 1979; Cummings et al., 2005; Nilar et al., 2019). In the case of the Colorado Front Range, the risk of catastrophic wildfire at the WUI is an escalating problem for societies in high-risk environments worldwide, given climate change, the growing human population, and expanding urbanisation (Enright & Fontaine, 2014; Ganteaume et al., 2021; Mahmoud & Chulahwat, 2020). With respect to the third case study, the combined issues of dieback and an altered or degraded environment hostile to restoring the pre-existing vegetation are not unusual, with many parallels worldwide in the case of both specific dieback syndromes (e.g. Allen et al., 2015; Koch, 2015; Pautasso et al., 2013; Martin et al., 2019) and environmental modification more generally, such as that caused by mining or agricultural degradation (e.g. Barrett-Lennard et al., 2016; Cross & Lambers, 2017; Navarro et al., 2017). Dieback and environmental degradation, however, are not necessarily pervasive or widespread in most regions.

What is most noteworthy about these three case studies is their achievement of their respective objectives, to date. Complete or partial failure of ecological restoration projects probably frequently occurs but is likely to be under-reported. An indication is provided by a national survey of ecological restoration projects in México (Méndez-Toribio et al., 2021). Of 75 projects, only a third of them achieved a high level of biodiversity recovery. In the case of 87 restoration projects reviewed by Lockwood and Pimm (2004), only 20% were completely successful in meeting their objectives, the remainder being unsuccessful or only partially successful. In the case of the farmland rehabilitation case study, Taylors Run, comparative farm data are lacking, but in the authors' experience, profitable carbon-positive farms that have restored and are continuing to enhance native biodiversity are rare.

What Key Advances or Actions in Practice, Technology, or Other Facilitated Success?

Each of the case studies reviewed here is employing a wide range of standard restoration practices to achieve their objectives. Rather than repeat these in detail, we point out particularly noteworthy practices and insights in each case.

At Tiromoana Bush, where the long-term objectives envisage restoring refence forest ecosystems in 200 years' time, an important strategy has been to hasten the development of dense early successional vegetation consisting of pioneer species such as kānuka and tolerance of certain exotics, particularly Gorse and Broom, that shade out herbaceous weeds but facilitate the ingress of late-successional future-dominant species beneath their canopy, through avian dispersal of fleshy fruits. The role of early successional trees and shrubs, including exotic species, in accelerating recruitment of later successional fleshy-fruited species is a well-known restoration strategy both in the temperate zone and tropics (Rodríguez, 2006; García et al., 2010; De la Peña-Domene et al., 2013). The strategy has the virtue of increasing the ecological resilience of the restoration project by promoting the self-sustaining natural regeneration of later successional species, at least of fleshy-fruited species. This is important given the long multi-century timeframe to establish old-growth forest reference communities and the possibility that project funding and active restoration could cease well before the endpoint is achieved. However, inclusion of some mature forest canopy species in plantings is important, especially when seed sources are distant or lacking (Forbes et al., 2020).

In the Colorado Front Range Case Study, the key ingredient was the collaboration of so many stakeholders and land managers in the Roundtable, which led to effective oversight of a large area of wildland–urban interface (WUI). The project had the advantage that the Roundtable was already in existence prior to the funding being received, so the ground rules for discussion of controversial issues and a measure of trust amongst participants had already been established. Nevertheless, social and economic perspectives are often overlooked in key aspects of ecological restoration (Castillo-Mandujano & Smith-Ramírez, 2022), and in such an emotive and contested space as wildfire in the WUI, the importance and difficulty of achieving effective stakeholder engagement and consensus in this situation should not be underestimated. Whole primers are required to guide restoration project managers, who are often trained in forestry or ecology rather than social science, negotiation, and conflict resolution, on how to organise such roundtables for contentious ecological restoration and environmental management programmes.

The third case study offers three salient insights. First, if biodiversity conservation is not the sole or overriding priority in an ecosystem recovery project, other priorities will usually dictate that reference native ecosystems are not the most appropriate target, especially where commercial priorities take precedence. In such situations, rehabilitation rather than restoration (sensu the SER Standards; Gann et al., 2019) will be the most appropriate response. Second, the Taylors' remarkable achievements underscore the rarity of commercial examples of sustainable agriculture in Australia, and possibly elsewhere. Very few farms known to the authors, along with Taylors Run, might qualify as such. We can only identify a handful of farming operations in Australia that likely qualify as 'sustainable' in terms of being profitable, carbon-positive, good for native biodiversity and building resilience (e.g. Landsberg et al., 1998; Wright & Wright, 2004; Williams, 2017; Reid, 2018), but the lack of sustainability verification mechanisms means this is speculative on our part. This speaks to the lack of incentives and accrediting schemes, and ultimately the lack of sound government policy (Campbell et al., 2017; Lockie, 2020) and industry leadership for encouraging and verifying sustainable agriculture in Australia. The situation is probably similar in most other countries, which points to a monumental failure of governance world-wide. Third, for practical entrepreneurial farmers, ecological restoration and rehabilitation can provide business opportunities to strengthen the economic resilience of traditional farm businesses, which often otherwise are completely dependent on just one or two enterprises (e.g. crops and livestock). We note that the benefits are likely to flow both ways. On the one hand, ecological restoration is expensive and not necessarily efficient, as evidenced by one and two-row mechanised seeders and planters (Freudenberger, 2018). On the other, given the right incentives, farmers and agricultural industries are quite capable of developing new and effective equipment to reduce costs and match the scale and complexity of the restoration problem at hand.

What Barriers Continue to Impede Success?

Currently, there are a few barriers impeding the success of the three case studies profiled here. Perhaps the most obvious is the 10-year funding horizon for the Colorado Front Range project, which ceased in 2020, and the question whether the roundtable can (i) secure further external funding, (ii) devise activities that are self-funding, or (iii) persuade participating land management agencies to adopt contributing roundtable-approved activities and land management treatments as part of their routine business. This begs the question with 200-year or even 50-year projects, what governance and funding or business frameworks can be established to ensure continued success long after the current restoration champions of the project have moved on. In settings such as family farms where inter-generational succession is functional and future generations are interested in pursuing sustainability, one generation of managers can bequeath the farm to equally passionate descendants. However, farm succession is often a vexed issue (Lobley, 2010; Zagata & Sutherland, 2015; Falkiner et al., 2017; Leonard et al., 2017; Nuthall & Old, 2017; Zou et al., 2018).

Another issue in case study 2 was the relatively limited extent of hazard reduction burning that was completed (2874 ha). Events frequently conspire to thwart hazard reduction burn plans, such as inappropriate weather, unplanned-for contingencies and emergencies (e.g. wildfire), and necessary changes in agencies' or property owners' short-term priorities or resources. Furthermore, the history of large catastrophic wildfires in the region and the perceived risks of prescribed fire have limited the social licence for its use and created institutional and policy barriers. However, implementation of hazard reduction burns in priority locations on a rotational basis will be required to achieve the roundtable's vision.

A point worth noting about all of these case studies is the long (often multi-century) timeframes associated with ecological restoration of forests or, at least timbered, ecosystems, and the requirement for continuing success, ad infinitum, given the goal of sustainability. However, given the importance of the adaptive management paradigm to ecological restoration (Murray & Marmorek, 2003), long-term success does not necessarily mean that restoration goals must be set in stone for evermore. Values, mores, and the external decision-making environment change and new knowledge is gained, and so a project's restoration goals often evolve. Adaptive management over long timescales must necessarily incorporate shifting goals, which does not necessarily mean that a restoration programme failed even though the original goals might not end up being met. Rather, it means that the programme kept up with the times and maintained relevance as the current custodians and stakeholders of the project wrestle with and process new knowledge, issues, values, priorities, and unplanned events. This is especially true in the face of global climate change.

In conclusion, the risks to civil society posed by climate change and the need to feed, clothe, and sustain all of humanity with dignity means that temperate and boreal forest restoration and rehabilitation will continue to be high priorities for ecosystem management. Given human ingenuity, the wide array of technologies already being deployed to restore and rehabilitate temperate biomes will doubtless be improved upon, but passive regeneration of these ecosystems is already accomplishing much of the task and will continue to do so. We must accept, however, that the task of restoration will be more difficult in landscapes the more they have been transformed (Arroyo-Rodríguez et al., 2017). Given the rapidly changing world and inevitable resource limitations, the challenge will be whether the novel forest and agroforest ecosystems that will emerge across temperate wildlands and farmscapes are capable of sustaining biodiversity and the quantity and quality of ecosystem services that will be required by the end of the century and beyond.

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Chapter 6
Just Add Water? Wetland and Riparian Restoration

Michelle T. Casanova, Mark Bachmann, Simonetta Bagella, Adam Bester, Lance Lloyd, and Lisa Farnsworth

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Summary and Key Lessons

For successful restoration of wetland and riparian systems, we need to recognise several key points:

1. The primary driver for the existence of wetlands and rivers is hydrology, which is governed by the amount and pattern of water availability in the landscape. When water is removed, flow is modified or groundwater is depleted; it changes wetland and riparian systems and their dependent flora and fauna. When the pattern of water availability is changed (typically less water but also sometimes more water), this can alter the duration or depth of flooding, it can change the length of dry-times, and it can remove characteristics of the natural pattern of flow. Consequently, the flora and fauna of the wetland or riparian area can often change in response (Casanova & Brock, 2000). Thus, the primary question in

restoration of these systems is 'What can or will characterise the hydrological regime in the future?'

2. Natural wetland systems have had the capacity to contract and expand in response to variable hydrology over time. Rivers can migrate across their floodplain when they are able, and inundation and flow, including extensive overland flows which manifest as floods, represent restorative events in wetland systems (Tiegs et al., 2005; Dixon et al., 2016; Casanova, 2015). It is now recognised that restoration efforts need to provide for the continued existence, movement, and variability of wetlands in space and time (Gell et al., 2016), and it is worth noting that seeking to define a fixed endpoint, or an ultimate state in this context, is often problematic (Rohr et al., 2018).
3. Natural wetland and riparian systems are inextricably linked to their surface and groundwater catchments (Lane et al., 2018). The catchment is where the water and nutrients come from, and many wetland species rely upon both wet and dry phases in their life cycles. For a restoration activity to be successful, the inputs from the catchment and the connectivity with other wet and dry systems need to be considered.
4. Wetlands have a high degree of inherent resilience. The plants and animals in them have adaptations such as germination cues and/or breeding triggers as responses to disturbances such as drought, high flows, physical changes, and grazing (Brock et al., 2003; Casanova, 2012). In many cases, it has been found that some disturbance is required to maintain biodiversity.
5. Restored wetlands usually need continuing management (Wolfenden et al., 2018; Galatowitsch & Bohnen, 2021). It is not likely to be sufficient at most damaged sites to simply undertake restoration activities and then cease active management, since weeds, nutrients, hydrology, and food webs will all need to be closely monitored to ensure long-term success.

Management Implications

- Wetlands and riparian zones can be restored when appropriate hydrology is returned; nutrients are either controlled or not a problem; and exotic flora and fauna can be managed. This often entails recognition of processes in the catchment, as well as *in situ*.
- Hydrological restoration can be sufficient for restoration when the only thing that has changed is the hydrology, but this is rarely the case.
- Restored wetlands can have high diversity, provide habitat for rare and migratory species, and can sequester carbon.
- A clear and appropriate goal and ongoing management are requirements for success.
- Wetlands are not alone in the environment, and provision of connectivity with other wet and dry-lands can facilitate success.
- Although there is a high degree of community and NGO input into wetland restoration world-wide, the places where it is most successful is where there is community engagement, appropriate, informed government policy, and funding that meets restoration needs. People need wetlands and for restoration to occur, wetlands need people.

Introduction

It is widely recognised that the areas designated as 'wetlands', which also includes riparian zones, are amongst the most threatened ecosystems across the world (McInnes et al., 2020). When these areas are wisely managed, they provide habitat for a suite of microorganisms, plants, and animals which together create the special character of wetland ecosystems. Wetlands contribute essential services and processes to the wider ecological community (Casanova & Powling, 2014; de Groot et al., 2018). However, these areas are being challenged by: (i) the erosion of their natural footprint and alteration of their function due to hydrological change, such as artificial drainage, diversions and levees, land reclamation, water regime modification, and flow regulation; (ii) eutrophication and pollution; arising from exogenous pressures on their internal systems; and (iii) the introduction of aggressive exotic organisms which outcompete the natural flora (Zedler & Kercher, 2004). In addition to these disturbances, there are the synergistic disturbances of climate change, agricultural intensification, and increased urbanisation associated with expanding human populations (Davis et al., 2015). The recognition of the damaging effects of these factors has contributed to the growing movement for wetland conservation world-wide (https://www.ramsar.org/). However, it is becoming clear that it is not enough to merely attempt to conserve wetlands that are currently in good condition. The extraordinary degree of global wetland loss, which is estimated to be as high as 87% since 1700 AD and 64% since 1900 AD (Davidson, 2014; Ramsar, 2015), indicates that work to restore wetlands will be critical for preventing species and ecosystem extinction. This work will need to focus on recovery of currently lost or compromised ecosystem services such as carbon sequestration and flood mitigation.

Fortunately, throughout the world, there are now initiatives that create incentives and provide resources for wetland restoration. For example, in Europe and the United Kingdom, the EU Water and Habitat Directives (https://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm) provide guiding incentives and targets for the maintenance of wetland water quality and ecosystems. The International Union for the Conservation of Nature have identified a range of endangered wetland species and have provided wetland community listings which recognise the threats and pathways to recovery of individual species and ecosystems (e.g. https://www.iucn.org/resources/conservation-tools/), and in the United States, the Environmental Protection Agency provides direction for wetland restoration (USEPA, 2000). Individual countries also have their own provision for protection of wetlands and incentives for wetland restoration, which are often championed by not-for-profit, non-government organisations (e.g. https://www.greeningaustralia.org.au/projects/wetland-restoration/).

Notwithstanding these initiatives, there still are major challenges to wetland and riparian restoration and they can be summarised as (i) recognition of the problem; (ii) recognition of the wetland type and development of an achievable goal or endpoint in a changing climate; (iii) securing financial resources and implementing appropriate 'works'; and (iv) determining whether the restoration activities have been successful (Roberts et al., 2017a) so we can learn from our mistakes and successes. In this chapter, we present several case studies where these central problems are being approached in different ways.

Case Study 1: Permanently Wet–Glenshera Swamp (South Australia)

Project Rationale and Strategy

Phases of human settlement in most growing societies progress from (i) the building of dwellings along rivers or near lakes, (ii) modification of the local drainage patterns, usually by draining swamps and changing the courses of streams, and (iii) increasing waterway and wetland pollution through their use as drains, sewers, or dumping grounds (Casanova, 2016). In many places, freshwater sources surrounded by urban and agricultural areas have suffered unchecked degradation until late in the twentieth century. At this time, as environmental consciousness rose, significant changes in social and political priorities gained traction. This case study illustrates the pressures which accrue on wetlands with increased population growth and describes the restorative work which has been carried out.

The wetland and riparian systems of the Fleurieu Peninsula in South Australia were managed by the Kaurna, Ngarrindjerri, Ramindjeri, and the Peramangk people for many centuries before invasion by European settlers. The land was productive, well-watered, and contained a diversity of wetland and robust riparian ecosystems. These were, in a sense ‘supermarkets’ for First Nations people and were the basis of spiritual life and connection to country. In terms of deliberate management, there is evidence of a fire-history contained in the sediments of these swamp systems (Bickford et al., 2008; Bickford & Gell, 2005) and early botanical explorers remarked with delight on the diversity of water plants associated with the wetlands (Grandison, 1996).

Major Concerns and Barriers

After management by First Nations people was interrupted in the mid-1800s, the character of the landscape began to change. The pattern and nature of this change was not uniform in location or time across the Fleurieu Peninsula, because of differences in the ease of settler access related to the topography, the density and composition of native vegetation types, and the underlying fertility of the soil. In general, the more easily accessed and open native vegetation types such as grasslands and grassy woodlands on more fertile mineral soils were altered first. This meant that despite their proximity to higher density settlement, many areas which supported areas of dense, scrubby woodland and heath habitats typical of Fleurieu Swamps, were seen as 'wastelands' and remained largely undeveloped until the 1890s (Farrington et al., 2017).

However, this didn't last, as during and after the 1890s, intensified settlement occurred in the region and the valley peatlands were targeted for drainage and clearance. Settlers sought out and exploited the natural fertility of these saturated sediments for enterprises such as market gardening. The pattern of this early development and the associated subdivision of the peatlands is still evident in the cadastral pattern of land parcels (i.e. small parcels of land dividing up the peatlands along the valley floor) and the extensive networks of artificial drains found across several catchments of the Fleurieu (Farrington et al., 2017). While most catchments were subject to *ad hoc* drainage works within the boundaries of each property, there were exceptions to this treatment such as the Nangkita settlement where a more comprehensive, larger, multi-parcel, intensive, and sophisticated drainage network was established (Farrington et al., 2017).

In some cases, fire regimes that mimicked a component of First Nations land management practices were continued by pastoralists to stimulate native vegetation growth for livestock, but this typically ended when the land was subdivided for closer settlement. Lands were then most-commonly cleared and sown with introduced pasture species. A second wave of development across the wider catchment, both in the uplands and the swamps, occurred from the 1920s, with the advent of new pasture establishment technology and mechanised clearance. This wave of clearance and drainage intensified after World War II but was largely complete by the 1970s (Farrington et al., 2017).

Somewhat remarkably, despite increasing settlement and drainage, some areas of Fleurieu Swamp have persisted as relics, which are valuable aids and reference points for restoration. It is noted that the dominant land use since the 1920s has been livestock grazing, which has been more recently replaced by irrigated horticulture through the expansion of wineries, olive production, and forestry (Casanova & Zhang, 2007), in parallel with a large increase in rural-residential properties. All these changes in land use are more intensive users of water, which in turn has an impact on the availability of water required to sustain the remaining relic wetlands (which require constant saturation with shallow groundwater to maintain their pristine condition).

Key Project Features

In the twenty-first century, based on improved scientific knowledge, there was recognition of the important cultural and biodiversity values of the swamps of the Fleurieu Peninsula (EPBC, 2013). What had been an extensive and unique system of peaty bogs, chains-of-ponds, and groundwater-fed streams in existence for thousands of years was now whittled back to a few, critically endangered residual pockets of habitat sheltering a suite of unique flora and fauna values (Murfet & Taplin, 2000).

The original extent of the swamps of the Fleurieu Peninsula is estimated to have exceeded 2000 hectares in area. Almost half of this has been lost through drainage and development, and those sites that now remain are small and highly fragmented. Given the high degree and intensity of human disturbance in this region, combined with ongoing and demonstrable threats, the swamps of the Fleurieu Peninsula were listed by the Commonwealth of Australia as a *critically endangered* ecological community under the *Environment Protection and Biodiversity Conservation Act 1999* (*EBPC Act*) in March 2003. In terms of the condition of these swamps, 53% are degraded, 21% are in moderate condition, and only 2% are in a near-pristine state (Bachmann & Farrington, 2016; Harding, 2005).

In response to this more recent realisation of their conservation and heritage value, these swamps have been the focus of significant, proactive conservation works, which include livestock exclusion and weed control. This has been implemented since the 1990s, often led by non-government organisations. However, until recently, these activities have largely overlooked underlying changes to hydrology caused by the preceding decades of artificial drainage.

Recent restoration works in broadly similar peat wetland systems in the Limestone Coast region of South Australia (Bachmann, 2016) and the Glenelg Hopkins region of Victoria (Bachmann, 2020) indicated that these ecosystems could be capable of responding favourably to hydrological restoration works. This is due to the reliable groundwater-driven base flows that are a prerequisite for successful peatland rehydration and restoration. One site that constitutes the largest and most intact remaining example of the Fleurieu Swamp community, the Glenshera Swamp, occurs within the Stipiturus Conservation Park (Fig. 6.1), situated approximately 6 km west of Mount Compass, South Australia. This park, which was proclaimed in 2006, provides habitat for one of the most significant swamp-based populations of the nationally endangered Mount Lofty Ranges Southern Emu-wren (*Stipiturus malachurus intermedius*) and 64% of native plants found in and around the swamp have regional and/or state conservation ratings.

Fig. 6.1 September 2014 aerial image of Glenshera Swamp, showing drains (black dashed lines) and the Stipiturus Conservation Park boundary (red line)

The listing of Fleurieu Swamps, together with some of their resident threatened flora and fauna, as matters of national environmental significance under the *EPBC Act* in 2003, provided fresh impetus and government resources to document their values and extent, and initiate restoration efforts. Consequently, Glenshera Swamp has been the focus of restoration planning and physical remedial works by a non-government organisation since 2015 (Bachmann, 2018).

Major Project Outcomes

The vision for Glenshera Swamp, as outlined in the park management plan (DEH, 2007), was to maintain a healthy wetland ecosystem that supports a flourishing population of Mount Lofty Ranges Southern Emu-wren and in addition that also provides important habitat for other species of conservation significance. Implicit in maintaining wetland health is the requirement to maintain a hydrological regime that supports a diverse and functional swamp community into the future. This is a particularly important consideration given declines in rainfall, runoff, and recharge due to the predicted impacts of climate change. These declines, combined with artificial drainage networks present throughout the site, may not provide sufficient water to meet the on-going needs of the swamp community.

An assessment was undertaken (Bachmann & Farrington, 2016) from September 2015 until January 2016, which found that although the site was supported by regional groundwater flows, it was also strongly influenced by seasonal rainfall and localised surface and groundwater catchment flows. Past drainage activities that had occurred over several decades, commencing in the 1940s, deliberately sought to divert inflows, dry the slopes, drain the swamp-bed, and increase downstream drawdown (Fig. 6.1). These activities were all aimed at significantly increasing the flow of water out of the system, irrespective of its source.

The hydrological restoration goal identified for the Glenshera Swamp in 2016 was to undertake remedial works that would significantly slow down and make better use of water within or passing through Glenshera Swamp to preserve and, if possible, enhance its ecological values. These works aimed to achieve this goal through increasing saturation of the peatland.

Addressing alterations to hydrology caused by the bypass drain constructed in the 1940s was the initial focus of work in the conservation park (Fig. 6.2).

The first works, funded by the South Australian state government and completed with the support of community volunteers, were intended to reinstate overground inflows to the swamp. Additional works included the installation of three weirs downstream of the original creek to slow flows and/or to prevent lateral drawdown of water from the swamp margin. The capacity to monitor progress towards the aims was incorporated into the restoration works. This was done using before-and-after photo-points and placement of a network of surface and groundwater-level data loggers across the site, as well as survey monitoring of species occurrence and abundance.

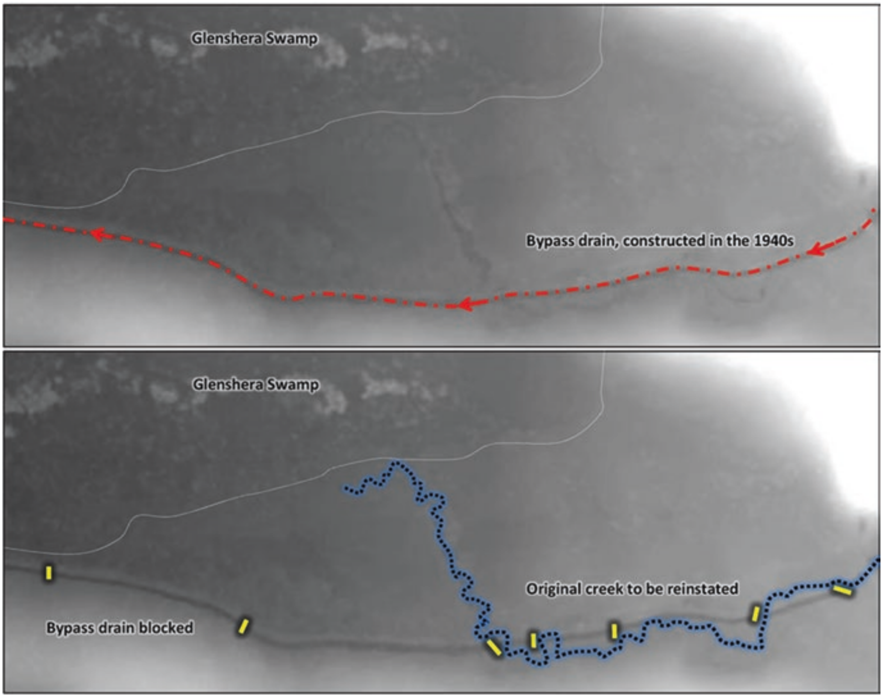

Fig. 6.2 Before and after reinstatement of the meanders in the creek and decommissioning of bypass drains

Building on those initial results and with the support of the downstream neighbours and community funding, more restoration work was undertaken on the more heavily drained and degraded portion of Glenshera Swamp outside the reserve boundary. Continuous drain backfilling to pack the channels was undertaken to counteract the continuous fall in gradient across the peat bed, utilising former spoil material left on-site from when the network of drains was first constructed (Fig. 6.3).

What Worked, What Did Not Work, and Why

This project was purposefully designed with SMART objectives in mind (Specific, Measurable, Achievable, Realistic, and Timely) (Bachmann & Farrington, 2016; Doran, 1981). As early as July 2017, for the first time in 70 years, the catchment started to generate sufficient runoff to reactivate the former creek channel, re-establish a narrow band of adjacent floodplain, and restore all low to moderate creek flows towards the main swamp. This success led to the drainage channel being more permanently back-filled (Fig. 6.3). Most drains in the network across privately owned peatland were also backfilled, enabling rehydration of the peat, from

Fig. 6.3 Looking west down a section of the diversion drain at the top of Glenshera Swamp, shown here before and after back-filling in 2020, and again in 2021, showing how the area is re-saturated, restoring flows to the swamp and also being actively revegetated with wetland species

Fig. 6.4 Glenshera Swamp, upstream of the western-most regulating structure. 1. Before restoration, introduced pasture grasses dominate the drained wetland; 2. After the restoration, water returns and pasture grasses are drowned; 3. One year after restoration, wetland vegetation begins to return; 4. Two years after restoration and re-saturating the peatland, wetland vegetation is thriving

groundwater discharges and seepage. The results were both immediate and visually dramatic (Fig. 6.4). In the area of private peatland along the park boundary, backfilling the channel also had an immediate impact, re-saturating the peat profile and triggering spontaneous recovery of native wetland vegetation. This is representative of the changes now taking place at the site in those areas that were subject to restoration works.

An analysis of data collected (Roberts, 2019) has indicated that the capacity of the peatland to retain moisture has increased since restoration. The rapid, flashy flows in the artificial drainage network that occurred after heavy rainfall events have been replaced by significantly attenuated flows, because of the slower water movement through the site and increased water retention and storage within the peatland. This shift has also been detected via an increase in both peatland water levels and summer and autumn base-flow discharge from the site, compared to pre-restoration conditions.

Importantly, the managers chose to minimise the physical disturbance to the site and maximise the natural regenerative capacity of the wetland, as well as incorporate a network of community groups and neighbours in consultation about, and implementation of, the works. The initial success of the first phase of works has resulted in further plans to continue with remedial works, to backfill the last remaining artificial drains at Glenshera Swamp within the Stipiturus Conservation Park.

Like the previous works, these next steps are proposed to be a partnership between a non-government organisation (The Nature Glenelg Trust), the local community (Friends of Parks), and the State Government (National Parks and Wildlife Service) in South Australia. A strong foundation built on (i) recognition of the problem at both government and community levels, (ii) trialling the works, (iii) communication and engagement with the broader community, including volunteers, funding organisations, and government, (iv) implementation of low-impact but effective physical works, and (v) on-going hydrological and photographic monitoring are the hallmarks of success in this project. For the project to be extended to other (privately owned) swamps, similar public input and cooperation would be required.

Summary

- A goal was set in the initial stages. It was feasible, accounted for by the future hydrology and hydrological management and there were no legacy issues concerning nutrients and water quality. The reason for restoration was to restore bird habitat, and this was thought to be achievable.
- The project was based on knowledge of the potential water regime required for the re-establishment of swamp vegetation as habitat.
- Most of the regenerative processes have been passive (i.e. occurred as a consequence of connectivity with sources of regeneration and seed banks), and the active works (filling drains) have been relatively low disturbance.
- The financial and governance resources were available for undertaking restoration (land tenure and government protections and incentives), and the community was knowledgeable and supportive.
- This site will require some degree of active management in the long-term, which is informed by monitoring.

Case Study 2: Permanently Wet: The Norfolk Broads (England)

Project Rationale and Strategies

Shallow freshwater lakes represent an example of a wetland system with which many people are familiar, and the examples described in this chapter on wetland restoration have deliberately included reference to these. Wetlands are often abundant in well-watered regions, with much of the Northern Hemisphere temperate region being a prime example. Because of their innate properties, these systems have been used for transport, the agricultural industry, and for leisure activities. As a result of these uses, many of these systems experienced significant declines in water and habitat quality from about 1900 onwards (Bennion et al., 2018). This decrease in habitat quality was mainly due to the synergistic effects of increasing industrialisation, urbanisation, and fertilised agriculture in their catchment areas. There are other examples of this progressive decline in wetland quality throughout the world in Madgwick et al. (2011), Tatrai et al. (2000), Havens and Gawlik (2005), Xie (2006), de Vicente et al. (2012), Ibelings et al. (2007), and Sondergaard et al. (2003). In many regions, shallow lakes have lost their submerged vegetation and experienced decline in the related ecosystem services, including storage of clean, clear water, sediment stabilisation, nutrient retention, and habitat provision for plants, fish, and invertebrates.

The Norfolk Broads (in the counties of Norfolk and Suffolk in England) provide an example of this widespread problem and the well-funded attempts made towards restoration. Concerns about these systems initiated targeted research into restoration processes and the development of complex restoration strategies to ameliorate wetland decline. It is relevant to note that, in contrast to many wetland ecosystems, shallow permanent lakes have the advantage of being appreciated by surrounding communities. Boating, fishing, and swimming are all widespread leisure activities on these systems, and so people have an ongoing appreciation for their condition. People also tend to notice when shallow lakes become degraded and are likely to support restoration efforts when they are implemented (Fig. 6.5).

Fig. 6.5 A scenic view of part of the Norfolk Broads (© Shutterstock), which, despite its aim for attractiveness, illustrates the highly managed landscape, and within the channel, algal abundance and a relative lack of water plants and vegetation

Major Concerns and Barriers

The Norfolk Broads was historically a vast area of fens, peaty, swampy land, and in medieval times, the fens were exploited for peat production on an almost industrial scale (Lambert et al., 1960). The excavation process resulted in a mosaic of shallow permanent lakes of about 2 m depth, interconnected waterways including natural rivers, extensive reed beds, and wet forest communities. From the 1880s onwards, this system of lakes, canals, and rivers became a favourite boating destination, and the value of the natural vegetation to the region was extensive. Of particular note in this respect were the water plants, including charophytes which are macroalgae of the family Characeae, and submerged angiosperms such as *Naijas marina* (Ayres et al., 2008; Madgwick et al., 2011). From the late 1880s to the early 1900s, at least nine species of *Chara* and *Nitellopsis,* many of them rare or restricted (e.g., *Chara canescens*)*,* were collected from the Broads (Groves & Bullock-Webster, 1924). However, during the middle of the twentieth century, runoff from agriculture and sewage from adjacent towns resulted in decreasing water quality and increased eutrophication. Consequently, between the 1920s and 1960s (Madgwick et al., 2011), there was a loss of species diversity as the charophytes were replaced by taller plants with floating leaves, which were then replaced by planktonic algae, especially blue–green algae. These changes had flow-on effects on invertebrate and vertebrate diversity as well. Fortunately, the area has been subject to extensive remediation and is now managed as a pseudo national park (the Broads Authority and Environment Agency).

It was recognised that a problem existed in the 1970s, when water quality decline and extensive macrophyte decline started to impact on macroinvertebrate and fish abundance and diversity, as well as boating and leisure activities in the Broads (Mason & Bryant, 1975; Osbourn & Moss, 1977). At the same time, there were extensive efforts to identify the causes and remedies for shallow lake eutrophication and waterplant decline in other places (Ganf, 1974; Sand-Jensen et al., 2008). Nutrient loading (Moss, 1983) was identified as a major issue, a process which, in simple terms, causes a set of self-sustaining conditions that favoured an ecologically stable state dominated by phytoplankton growth instead of a stable state dominated by submerged water plants (Fig. 6.6). It is appropriate to note that paleolimnological studies supported the historical existence of conditions conducive to high submerged plant diversity and abundance and gave evidence for the observed transitional vegetation changes that were reported in the 1970s (Sayer et al., 2016).

Key Project Features

Declining water quality due to human-induced eutrophication was so widespread in the Norfolk Broads that it led to a large number of groups working to determine the best restoration practices to remediate the situation. The main goal was to restore

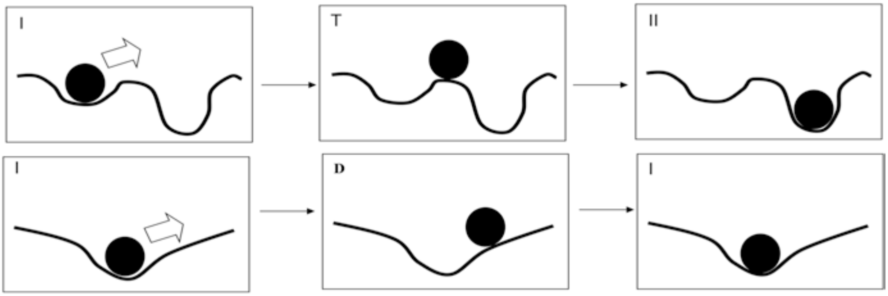

Fig. 6.6 The cup-and-ball analogy (after Laycock, 1991 in Briske, 2003). In the top row, according to state and transition models, (I) disturbance (the broad arrow) moves the community (ball) over a threshold (T) to a new stable state in the range of environmental conditions (II). The depth of the cup is related to the magnitude of disturbance required to cross a threshold. In the bottom row, according to equilibrium (successional theory), (I) disturbance moves the community to a new part of the range of environmental conditions (D), when the disturbance is removed, the system returns to the same stable condition (climax community, I)

the overall water quality of the area. Phytoplankton growth and macrophyte decline were directly related to increased nutrients, so initially the problem was seen to be related to an overabundance of phosphorus and nitrogen. This led to actions initially designed to prevent or lower nutrient inputs as an appropriate management strategy. However, limited understanding of the dynamics of plant and phytoplankton growth (Kalff & Knoechel, 1978) and the difficulty of removing large stores of nutrients already in bodies of standing water stimulated a rapid increase in the number of scientific studies to understand these processes more clearly.

The Broads cover 303 km^2, covering multiple land tenures and stakeholder parties, so that restoration or management goals cannot easily be realised without broad political and social support and agreement. To this end, an ambitious and fully costed recovery plan for the Norfolk Broads National Park (National Parks England, 2021) has been developed, which provides laudable strategic aims such as a reduction in point source and diffuse pollution, the development of stewardship partnerships with landholders, and targeted species recovery. However, the implementation of individual action plans for different aspects of restoration are currently the responsibility of a plethora of government and non-government organisations. The United Kingdom still adheres to the EU Water Framework Directive which aims for 'good status' surface and groundwater, and the EU Habitats, and Birds Directives which aim for the conservation of natural habitats, flora, and fauna. These directives articulate strategic goals, such as to achieve good ecological and chemical status, and the conservation of habitats, and oblige signatories to achieve them within a certain timeframe. The original target was for 2015, but additional targets have been set for 2021 and 2027.

A large proportion of the initial 'works' in the restoration of shallow eutrophic lakes related to (i) researching and understanding the range of 'desirable' conditions, (ii) identifying the mechanisms that maintain clear water, and (iii) characterising processes that result in decline of water quality. Importantly, it was also necessary to understand which nutrients were having the most critical negative impacts (with phosphorus considered to be the most likely cause). Equally important was to understand how and why water plants decline (possibly through shading and aggressive competition) and why phytoplankton had become abundant. Attention was also given to changed patterns of herbivory by invertebrates and fish and what actions might be required to restore these complex food webs and desired water plant dominance. In the latter case, the application of 'biomanipulation' or removal of fish stocks to allow invertebrates to increase in abundance, facilitate water plant recovery and result in improved water quality was a central concern (Scheffer, 1990).

Major Project Outcomes

Regeneration of characteristic submerged and floating plant communities is viewed as a general indication of successful restoration of such wetland ecosystems (Hilt et al., 2018; Madgwick et al., 2011). Although it seems that the concepts of 'biomanipulation' and 'alternative stable states' have always been present in the framework of restoration ecology, particularly in relation to shallow lakes, the development of management strategies to implement the theory was somewhat incremental in the 1980s and 1990s (e.g. Timms & Moss, 1984; Mills et al., 1987; Scheffer, 1990; Philllips, 1992). Nutrient reduction through nutrient absorption by chemicals or filters and the removal of sediments, together with detailed knowledge of the food web, increased understanding of the potential limitations to water-plant recovery. The research in the Broads led to important discoveries on the road to lake restoration. It is noted that for water plant recovery, knowledge of seed or propagule production, together with an appreciation of the effects of light, salinity, depth, and degree of disturbance, were essential to achieving successful outcomes.

What Worked, What Did Not Work, and Why

To date, various methods have been trialled to reduce phosphorus inputs into the system. These include (i) improving sewage treatment, (ii) causing phosphorus to be precipitated from the water through application of flocculants, (iii) implementing on-farm nutrient retention strategies, and (iv) redirecting high-nutrient inflows to alternative locations (Moss, 1983). Despite these attempts, nutrient release from sediments, referred to as endogenous nutrient cycling, remains a major influence (Sayer et al., 2016). Application of biomanipulation via manipulation of food webs was attempted in some of the Broads where nutrient concentrations were a problem (Ormesby Broad: Tomlinson et al., 2002; Cockshoot Broad: Moss et al., 1996). There was initial success in returning to a clear water phase.

Whereas there are some instances, especially in Europe, of successful short-term restoration of shallow lake water quality via biomanipulation (Jeppesen et al., 2007), in general, such approaches are not universally successful or long-term (Hilt et al. 2018). The absence of fish, natural non-biogenic turbidity, the absence of propagules or a germination stimulus, a lack of on-going maintenance of fish stocks, and unameliorated nutrient levels can prevent success. So now, after more than three decades of works to reduce external nutrient inputs, removal of nutrient-rich sediments and manipulation of fish stocks in various parts of the Broads, algal blooms still occur and waterplant communities have not recovered to historical abundance and diversity (Madgwick et al., 2011; Phillips et al., 2016; Sayer et al., 2019; Wagstaff et al., 2021).

In a review of the theory (regime shifts and stable-states Fig. 6.6) upon which biomanipulation is based, Capon et al. (2015) found little evidence for the importance of these concepts in most ecosystems, except for some instances in freshwater lakes. Notwithstanding this review, shifts between turbid phytoplankton-dominated states and clear-water, macrophyte-dominated states have been reported for shallow lakes in temperate climates (Jeppesen et al., 2007). Restoration of wetlands, particularly where changes have occurred over many decades and where nutrients cannot be removed remains problematic, and perhaps the biggest problem to restoration, in many situations, is the lack of development of an appropriate goal. It appears that the current management strategy for the Broads lake system aspires to their restoration to an unstable clear water 'lake-state', with abundant submerged vegetation that does not impede leisure activities such as boating and fishing in the system. This outcome or state is quite different to the original medieval system of a mosaic of shallow peaty swamps and water retention below the soil surface. It is provisionally suggested that, perhaps the problem should be recognised not as one of nutrient excess, but rather too much open-water habitat, too rapid a flow-through of water through the system, and a lack of upstream and onsite nutrient sequestration by vegetation and soil. In this case, an alternative, potentially achievable goal could be to slow the movement of water through the system and use the techniques being used in the Fleurieu Peninsula wetlands, to restore at least part of the boggy medieval landscape, rather than aspire to the 1880s–1970s model.

Summary

- It may be that the goal set for this wetland landscape was not realistic, in trying to return to an unstable system.
- The amount and speed of water flow through the system, along with legacy conditions of nutrient loading and vegetation removal inhibit a return to both the shallow-lake landscape and the peaty bog mosaic.
- A huge amount of active and passive restoration has taken place.
- The financial and governance resources are available for undertaking restoration (land tenure and government protections and incentives) but the community goal of a system of shallow lakes might be unrealistic.
- For restoration to a landscape of shallow lakes, this system will require a large degree of active management in the long term.

Case Study 3: Permanently Wet: The Glenelg River (Victoria, Australia)

Rationale

The Glenelg River (Bochara, Bugara, Pawur) rises in Gariwerd (the Grampians National Park) in western Victoria. Historically, the river originated and drained from a wide and diffuse system of swamps and bogs into a meandering permanently flowing river, consisting of a linear series of ponds, swamps, and billabong-strewn floodplains and ended in a swampy, episodically open and closed estuarine system close to the South Australian border. At most times, it was a free-flowing river, experiencing episodic overland flows during floods, and delivered nutrients to the productive coastal region whilst providing a conduit for the migration and breeding of native fish. It had, and still retains, important values for the indigenous Gundjtjmara, Wotjobaluk, and Boandik peoples. However, in 1953, the Rocklands Reservoir was created near its headwaters to retain water from the Glenelg for transfer to irrigated farming systems to the north, in the Wimmera-Mallee region, via inter-basin pipeline transfer.

Major Concerns or Barriers

Because all upper catchment flows were directed into another river system (Wimmera River) and with large-scale catchment changes, in the 1980s and 1990s, the Glenelg River was on the brink of ecological collapse. The abundance and diversity of river-dependent species had declined, and with every drought event, the river was reduced to isolated pools that were the only refuges for fish and the Glenelg Spiny Crayfish. Without flow to freshen the pools, they were exposed to increased salinity from groundwater, death of freshwater dependent flora and fauna, and subsequent anoxia. Where the river flowed through agricultural land in its central reaches, erosion created ‘sand-slugs’ that gradually moved through the river system, flattening the profile of the stream, filling in deep refuge pools, and reducing heterogeneity. Clearing of vegetation in the riparian zone was widespread at this time and the movement of domestic stock down riverbanks to and from the river presented potential point sources of nutrient pollution and erosion. However, despite the lack of flow, the lower reaches, in connection with the estuary and wetlands and largely surrounded by forest, were heritage-listed for their natural values. Several species that occur in the river were listed as threatened, including flowering plants, fish, molluscs, and invertebrates.

Fig. 6.7 A tributary of the Glenelg River where it passes through agricultural land. (**a**) a view in 2002 showing degraded riparian zone and lack of connections to the broader landscape, (**b**) revegetated riparian zone and creation of connecting plantations in 2015

Key Project Features

In the early 2000s, the Glenelg Hopkins Catchment Management Authority (a statutory body concerned with land and water management in the region) in conjunction with community groups and other government and non-government agencies, commenced the Glenelg River Restoration Project. This was an ambitious undertaking aiming for an integrated and long-term approach to restore the health of the river. The magnitude and diverse nature of the environmental problems required an integrated response involving varied actions to address problems in the upper catchments, as well as for in-stream issues. The long-term project vision was for a river system that could sustain both its ecological function and original flora and fauna. Since the river flows through in a highly utilised agricultural landscape (Fig. 6.7), it was important that it also provide stock water as well as a sustainable sand extraction industry to provide on-going sand-slug removal. In order to have an actively engaged and passionate community involved in managing and caring for the river, additional goals were that it be known nationally as a recreational destination for canoeing, fishing, and camping and sustain its economic value to the local community via tourism.

Major Project Outcomes

A whole-of-river approach was needed to initiate recovery in this system. Because the Glenelg has been dammed and its flow redirected and managed (i.e. a regulated river), changes at many levels of management were required to initiate its restoration. In 1994, the Council of Australian Governments recognised that more water was needed to maintain the health and viability of river systems nationally and that the environmental requirements of rivers should be met (COAG, 1994). This led to an assessment of the ecological requirements of the Glenelg River (SKM, 2003).

Fig. 6.8 A view of the Glenelg River at Coleraine (**a**) during low-flow before the return of environmental water illustrating extensive sand deposition, bank erosion, and a lack of riparian vegetation, (**b**) in 2016, after bank stabilisation and delivery of environmental water reinstating permanent pools and riparian vegetation

The river was found to be degraded, so after the establishment of a legal framework and creation of the Office of the Environmental Water Holder for managing and sharing flows, an allocation of water to the river was able to be made from the upstream reservoir in 2010. Conversion of open irrigation channels through the creation of the Northern Mallee Pipeline and the Wimmera Pipeline made 40 GL of water available annually to be shared between the Glenelg and Wimmera Rivers. Initially most of the water was directed to the Wimmera River, and delivery into the Glenelg River required significant changes to the structure of the water delivery structures. This included the construction of carp screens to prevent exotic carp from moving into the Glenelg River from the reservoir. At the same time, numerous large and small projects were coordinated by the CMA in collaboration with water authorities, industry groups, First Nations people, research institutions, anglers, individual river-side landholders, landcare groups, and private industry. These projects comprised an ambitious programme of revegetation, in-stream wood reinstatement, sand extraction to restore structure and deep pools, removal of stock access and weed control (Fig. 6.8). The total expenditure is estimated to be approximately $17,000,000 over 12 years. The most expensive items were erosion control structures, fishways, and water delivery structures.

What Worked, What Did Not Work, and Why

Each year monitoring has been undertaken along the Glenelg River (through the Victorian Environmental Flows Monitoring and Assessment Program, or VEFMAP) to determine the response of fish, water quality, flow, and physical habitat (Alluvium, 2013). Since water flows have been improved monitoring has verified improved river health conditions resulting in increased recruitment in native fish populations. Outcomes of this type have only been made possible through the additional activities of fish barrier removal (weirs and fords) and the creation of instream habitat (tree-trunks and constructed 'fish hotels'). After 10 years of environmental flows, there has been a significant increase in abundance (150%) and distribution (i.e. over 320 km) in the river of native tupong (*Pseudaphritis urvillii*) and estuary perch (*Macquaria colonorum*). These fish species are dependent on the connection between the estuary and freshwater reaches to complete their lifecycles and were previously absent from the freshwater reaches. Additionally, the improved connectivity and habitat values created through the management programme have seen the return of Australian grayling (*Protroctes maraena*) to the river, which had not been there for 122 years. Once abundant throughout the coastal rivers of south-eastern Australia, Australian grayling populations declined due to altered river flows, water extraction, and barriers to fish movement in the form of weirs.

Over 2000 km of fencing was completed, together with 756 km of waterway frontage revegetation (mostly via Landcare incentive payments where the materials are supplied by the funding body, and the landholder supplies the labour). This proved to be important to protect the gains made through environmental flows, sand extraction to remove sand-slugs and restore habitat, and the removal of fish barriers. Fencing and revegetation allowed instream plants to immobilise eroded soil and sand, preventing it from entering the river where it caused destruction to instream habitat. The restored riverside vegetation also filtered surface runoff, provided shade, habitat and food to instream biota, and there has been reformation of several waterholes and low flow channels (Sims & Rutherfurd, 2018). Fences prevented stock from damaging the physical form of the river channel, damaging riparian vegetation and fouling the water.

Although the Glenelg River is beginning to bounce back, there remains much work to do particularly with the escalating challenges and impacts from climate change. The changes have been favourably received by riverside landholders, people who fish the river and the general public. The public were kept informed of the benefits of the changes through targeted publicity, meetings with landholders, and via social media. Although it is unlikely that the Glenelg River will ever be returned to the ecological status that existed before river regulation and will never be completely connected with its entire floodplain again, it demonstrates that with sufficient background knowledge, goals, and resources, improvements can be made. Restoration of a river system can take generations and requires significant on-going maintenance to protect previous investment and successes and can only be done when there is an appropriate legal and funding framework and a coordinating organisation.

Summary

- A goal was set in the initial stages. It was feasible, accounted for the future hydrology and hydrological management. There are legacy issues concerning salinity and sand-slugs, but these are being managed. The reason for restoration was to restore fish habitat and human amenity, and this was thought to be achievable.
- The project was based on the knowledge of the potential water regime required for the reestablishment of fish habitat and to fulfil functional needs (e.g. migration, refuge).
- Most of the regenerative processes have been passive (i.e. occurred as a consequence of connectivity with sources of regeneration and seed banks), and the active works (bank stabilization, erosion control, delivery structures) have been well resourced.
- The financial and governance resources were available for undertaking restoration (land tenure and government protections, permissions, and incentives), and the community became knowledgeable and supportive.
- This river will require active management in the long-term, especially during times when there are low allocations of water, which is informed by monitoring.

Case Study 4: From Permanent to Temporary: Mokoan (Victoria Australia)

Project Rationale

Part of the process of worldwide wetland decline has been removal of wetland function through their transformation into lakes or reservoirs. Indeed, the previous case study on the Norfolk Broads is an example of this process. A more recent example can be seen in the creation of Lake Mokoan and the Winton Wetlands in central Victoria, Australia.

A large geographic area of central Victoria originally contained a complex mosaic of temporary and ephemeral wetlands, with intervening dry land that provided significant resources to indigenous humans and native flora and fauna, at a landscape level. The system was used and managed extensively by the Yorta Yorta people and other aboriginal groups for a meeting place and a site to harvest the bountiful resources in this mosaic of ecosystems. Mokoan was an early name used for the region, which is thought to have an indigenous origin. European settlement in the 1860s saw the indigenous people displaced and the lands used for livestock grazing (cattle and sheep). In the period since, the area has retained significant values for humans, and still represents a large repository for indigenous flora and fauna. The largest of the 33 wetlands in this interconnected mosaic became known as Winton Swamp.

In 1970, the entire wetland system was flooded from the nearby Broken River system to create a permanent lake (Lake Mokoan) to provide irrigation water for the farming region. The flooding resulted in the loss of 100,000 river red gums and the wetland bed grasses (Fig. 6.9). This new lake initially had high natural values,

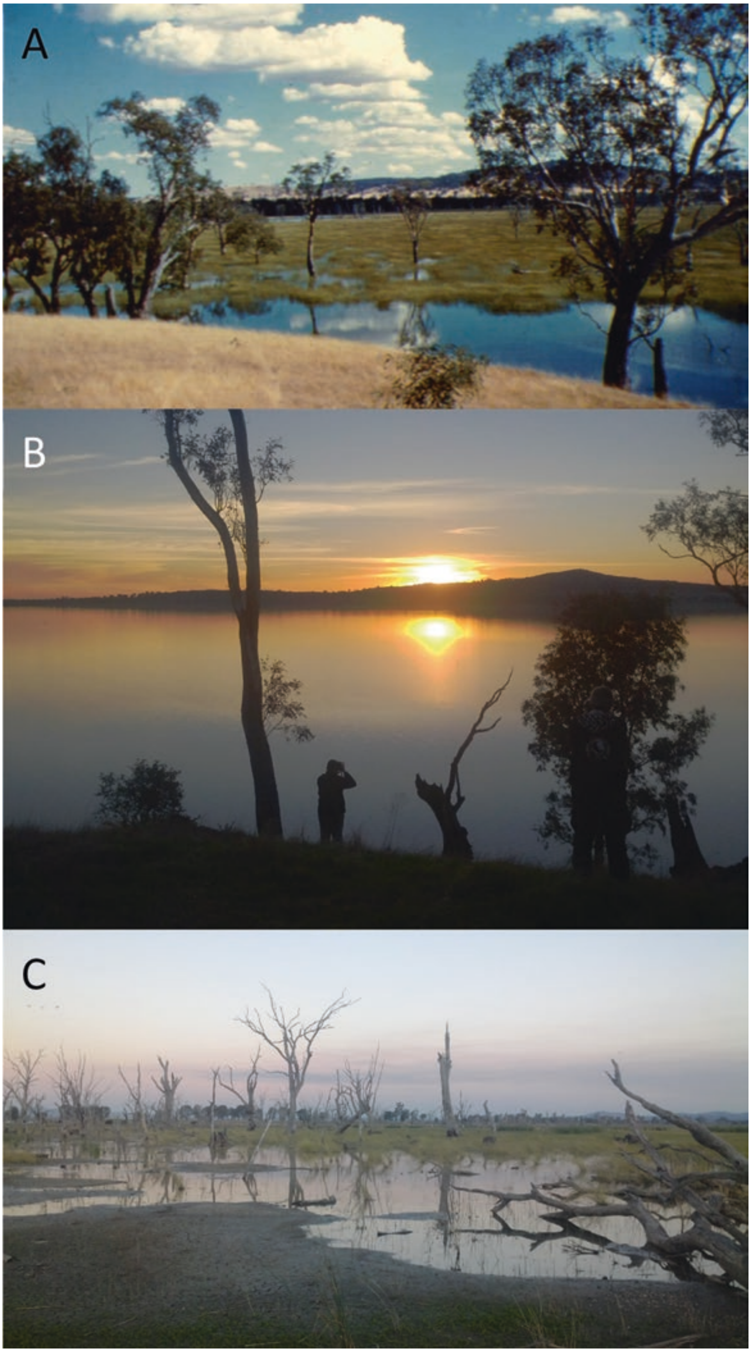

Fig. 6.9 The Mokoan/Winton Wetlands main swamp (a, 1959 and b, c 2017). (**a**) Overview from 'The Spit' with Southern Cane Grass understorey and river red gum overstorey. (**b**) Same locality as (**a**), illustrating the absence understory and decline in condition of overstorey. (**c**) View from 'The Spit' facing south, illustrating regeneration of Southern Cane Grass, but not river red gum. (Photograph A courtesy of Helen Aston)

with clear water, and records indicate aquatic macrophytes were abundant. It provided a new and valuable venue for recreation activities such as fishing, boating, swimming, and duck-shooting. Notwithstanding this encouraging start, in the late 1970s, the reservoir managers began to notice a gradual loss of the aquatic macrophytes. Eventually, after the 1982–1983 drought conditions led to the further decline in aquatic vegetation and degradation of water quality and the system experienced almost continuous toxic algal blooms (*Microcystis aeruginosa* – blue–green algae).

Major Concerns or Barriers

Although a perceived problem with Lake Mokoan was the decline in water quality from external source eutrophication leading to continuous algal blooms, the actual underlying issue was more complicated. Careful study of the system indicated that, after the initial filling, what had been an area of temporary wetlands appeared to be fulfilling all the requirements of a healthy shallow lake system. The drought in 1982 saw the system dry completely, and when it refilled, the formerly abundant water plants failed to re-establish (either from seed bank depletion or burial under sediment accumulation during permanent flooding or destruction while dry).

As a consequence, the water quality declined due to persistent inorganic turbidity exacerbated by the cycling of endogenous nutrients. At this time, wind shear was thought to be an important process in maintaining the mixing of sediment, with concomitant release of nutrients from the lakebed into the water column. While it is clear that the drought delivered the final blow to the storage lake as a functioning aquatic plant-dominated system, it was the changes in water management in the late 1970s prior to the drought, where the storage was kept near full-supply level, instead of fluctuating, that triggered the decline (O'Brien et al., 1996). This ecological decline resulted in algal blooms that reduced light penetration and prevented water plants from re-establishing, allowing further sediment mobilization and consolidating the change from the water plant-dominated stable state to the algal dominated state. It is noted that this significant change in environment is an example of where the 'alternative stable states' model (Scheffer, 1990; Fig. 6.6) can be used to effectively explain conditions. The decline in water quality prevented public use of the site and the water.

In 2010, after recognition of its inefficiency as a water storage because of the water quality problems, the retaining wall for Lake Mokan was breached and the fluctuating water regimes of the original wetland areas were reinstated by lake mangers. The system is now called 'Winton Wetlands' and is managed by a government-appointed Committee of Management.

Key Project Features

Drying or permanent drawdown of a lake system to restore conditions for a functional wetland system is not a common practice. There are few examples of this sort of restoration in the literature, with a useful discussion being given by van Wichelen et al. (2007), where a temporary drawdown was undertaken. Notwithstanding this paucity of reports, drawdown has been seen as a management tool for riparian restoration (Dahm et al., 1995; Toth, 1993). However, such a programme requires a large amount of both political will and public support (Whalen et al., 2002), and the Lake Mokoan Future Land Use Strategy (FLUS) was created with funding by the Victorian Department of Sustainability and Environment (DSE), with support from Goulburn-Murray Water (G-MW) and the Goulburn Broken Catchment Management Authority (GBCMA). The Lake Mokoan Future Land Use Steering Committee (LMFLUSC) assisted with development of the FLUS. The strategy established goals for the restoration works to decommission Lake Mokoan, and these were provided to the Winton Wetlands Committee of Management (WWCoM) when it was established in 2010.

The WWCoM facilitated the development of the Winton Wetlands Restoration and Monitoring Strategic Plan (Barlow, 2011). The project has been further overseen by a highly qualified Environmental Strategy Advisory Panel (ESAP: consisting of independent scientists and knowledgeable community members), whose terms of reference are to ensure that scientific knowledge is applied to the ecological renewal aspects of the project. ESAP, together with the WWCoM staff, provides continuous review of the objectives in Barlow (2011). The objectives have been used to provide specific planning documents to manage various aspects of the ecological renewal system including:

- A Pest Animal Management Strategy
- A Revegetation Strategy (reviewed on an annual basis)
- A Grazing Management Strategy
- A Fish and Carp Management Plan (Lloyd, 2015)
- An assessment of Water Quality and Aquatic Vegetation Management (Lloyd & WWCoM, 2015)

The aspirational goal for the wetlands area recognises the capacity for long-term natural ecological renewal at the site and its potential for functional change. The time-specific goals are:

- By 2036, the wetlands are thriving in a natural ecosystem achieved through the return of key flora and fauna communities. This will take into account understanding of pre-impact condition, recent land and water use change, pests, and climate changes including the needs of key threatened species in a well-managed landscape.
- By 2036, we will have a better understanding of future trajectory of the wetland and terrestrial ecosystems at Winton Wetlands.

A key feature of this plan is the adoption of the conditions prior to inundation as a reference or target and a commitment to comprehensive recovery. For this, previous site surveys from the 1950s and 1960s (Helen Aston pers. comm.) including site photographs (Fig. 6.9) and species lists provide some empirical evidence of previous conditions before the site was inundated. The conditions prior to European management were not well-documented, and further research needs to be undertaken to know what these were. However, before inundation the site was considered to be just one of the extensive natural wetland areas occupying central Victoria (H. Aston, pers. comm.).

Major Project Outcomes

The Winton wetland project has initiated a number of investigations and works in order to achieve the strategic goals (BECA, 2006) and targets (established in Barlow, 2011). These focused on understanding wetland values, addressing and reversing current threats, and on understanding the ecological processes that help to maintain the system.

Regular monitoring of birds, fish, plants, reptiles, frogs, and water quality is imperative to understand ecological values and measure success. Monitoring programmes have been carried out by staff and volunteers as well as through supported research (e.g. Roberts & Looby, 2019). Nest boxes have been created to facilitate bird, mammal, and marsupial presence and abundance. Camera trapping surveys are regularly undertaken to monitor the use and effect of these artificial habitat structures.

Revegetation of terrestrial features of the system was a major focus early in the project, since it was recognised that there were links between recovery of wetlands and their surrounding vegetation, especially through nutrient retention in the landscape (Sutcliffe, 2018). The objective was to establish structurally important species, with the view to adding structural and floristic diversity once the overstorey had established, and this stage is currently underway. Community and volunteer groups, such as the Regent Honey Eater Project, and the Friends of Winton Wetlands have been critical to the early success of these stages. Planting of aquatic plants in wetland areas and research by university students in the development of suitable techniques (e.g. Richter-Martin, 2017) have also been important in increasing the biodiversity in some of the smaller wetlands.

The impact of exotic and invasive species has been a significant disturbance issue at the Winton wetland. Feral animals predate native animals (such as foxes preying on turtles), and invasive introduced grasses such as Phalaris (*Phalaris aquatica*) compete strongly and often dominate native plants (wetland and terrestrial). To this end, management actions that focus on fox control must be continuous due to recolonisation from surrounding landscapes. Additionally, active weed management through herbicide spraying, strategic grazing, and hand removal have been ongoing. Research undertaken by university students has also provided support for restoration strategies (e.g. Sutcliff, 2018). European carp (*Cyprinus carpio*) and invasive introduced fish species dominated native fish populations in the former Lake Mokoan (and in many other inland water systems of eastern Australia). A comprehensive European carp control programme including the installation of carp screens, fish-downs, and habitat enhancement for native predatory fish has been implemented to assist in its control. This is in addition to targeted drying events during carp breeding to help control this exotic fish.

Integral to the recovery and management of the site is a thorough understanding of the hydrology of the wetlands and associated catchments. A water management strategy has been developed to address agricultural drainage of many of the smaller wetlands onsite and to provide understanding of the water balance of the system, which will eventually lead to a predictive management tool for system-wide water management.

What Worked, What Did Not Work, and Why

River red gum (*Eucalyptus camaldulensis*) is an important species in wetland structure and function and this species was an obvious casualty when the wetlands were flooded to create Lake Mokoan. Little natural regeneration of this important species has occurred. Onsite research (White, 2016; Armstrong, 2018) showed that the major driver of this poor river red gum regeneration was seed limitation (i.e. few remaining adult trees to produce 'seed rain'). In contrast to some of the early ideas such as seeding from a helicopter, the use of direct hand-seeding combined with the protection of saplings from macropod grazing (i.e. Kangaroos) has resulted in some successful tree restoration.

Turbidity in the standing water of wetlands has been greatly reduced compared to the lake system since the drawdown 2010. Lake turbidity levels in the mid-1990s were often between 120 and 250 NTU (Nephelometric Turbidity Units) and up to 800 NTU in some locations. Since the drawdown turbidity has fallen to <100 NTU near inflows and below 20 NTU within other wetland areas.

Regular fish surveys in the main wetlands pre- and post-drawdown have revealed that fish populations previously dominated by European carp are now dominated by small-bodied native fish such as carp gudgeons (*Hypseleotris* spp.). Also, breeding programmes focused on the endangered native Murray cod (*Maccullochella peelii*) have been instigated and new populations of this species have been established in a number of permanent water locations across the wetlands. Recent drying cycles have contributed to the removal of other exotic fish including goldfish (*Carassius auratus*) and carp-goldfish hybrids, and this has been assisted by significant predation from native piscivorous birds such as pelicans (*Pelecanus conspicillatus*) as the water levels fluctuate.

Other native birds have also been monitored returning to the wetland system. In 2017, the site experienced the first colonial breeding event for many years of the nankeen night heron (*Nycticorax caledonicus*) at Ashmeads Swamp, and the juvenile birds from that event were then sighted across many of the wetland sites during the following year. A pair of white-bellied sea eagles (a nationally listed threatened species: *Haliaeetus leucogaster*) have fledged chicks in sequential years.

Similarly, native marsupials such as Phascogales, Sugar-gliders, and Antechinus together with mammals like rakali (*Hydromys chrysogaster*), have returned to the site with the restoration of habitat and the creation of corridors between the surrounding landscape and the wetlands.

This on-going project required the injection of millions of dollars, thousands of hours of employee and volunteer time, continuous expert input and deliberate monitoring of progress. Although there have been many achievements, this is not a project that can be left to its own devices. Just as in this past the environment was managed over millennia by First Nation's people, it will be necessary that this wetland system be actively managed into the future to maximise its natural values and processes.

Summary

- A goal was set in the initial stages. It is feasible because it accounted for the future hydrology and hydrological management. There are legacy issues concerning exotic plants and animals and over-abundance of native animals, and controls are being undertaken. The reason for restoration was because the previous state did not function as desired, and subsequent goals (to restore natural habitat and provide human amenity) are thought to be achievable.
- The project was based on the knowledge of the potential water regime required for the reestablishment of swamp vegetation.
- Some of the regenerative processes have been passive (i.e. occurred as a consequence of connectivity with sources of regeneration and seed banks), and the active works (planting trees, restoring water movement) have been relatively low-disturbance.
- The financial and governance resources were available for undertaking restoration (land tenure and government funding) and although the community was not initially knowledgeable and supportive, this is improving.
- This site will require active management in the long-term, which is informed by monitoring.

Case Study 5: Temporary Wetland Challenges: Mediterranean Temporary Ponds (Sardinia)

Rationale for This Study

Mediterranean Temporary ponds (MTPs) occur in European and North African Mediterranean climates. They are similar to Australian Seasonal Herbaceous Wetlands (SHWs see following section) in hydrology, vegetation, flora and fauna, but occur across a variety of jurisdictions and continents, making their orchestrated conservation and restoration difficult. Exacerbating this issue is that recognition of their intrinsic values and concern about their current decline is a relatively recent occurrence.

Major Concerns and Barriers

MTPs have been historically neglected, degraded, and destroyed in the Mediterranean region, with their destruction accelerating during the last century (Bagella et al., 2016; Rhazi et al., 2012). The main pressures arise from agricultural intensification, agricultural abandonment (which removes grazing altogether), overgrazing, hydrological perturbations, excessive recreational use, introduction of exotic species (from America and Australia), and the effects of climate change (Fig. 6.10; Bagella & Caria, 2013; Bolpagni et al., 2019; Grillas et al., 2004; Zacharias et al., 2007). Under these pressures, thousands of MTPs, and even entire pond landscapes, have disappeared (Rhazi et al., 2012). Moreover, they have been frequently overlooked in studies due to their shallow water and small surface areas (Boix et al., 2017; Fois et al., 2021).

Fig. 6.10 Mediterranean Temporary Ponds on the island of Sardinia: (**a**) a temporary pond showing intact surrounding vegetation and emergent *Ranunculus* species flowering, (**b**) a pond with shallow edge vegetation, (**c**) a temporary pond subject to rubbish dumping and eutrophication, (**d**) a temporary pond subject to drainage, overgrazing, and soil disturbance

Negative human-induced hydrological perturbations typically relate to MTP drainage for control of mosquitoes, reclamation of land for urban development or agricultural activities, or for collecting of water for irrigation, watering of livestock and domestic use (Aponte et al., 2010; Zacharias & Zamparas, 2010). In addition, there has been increasing damage to MTPs from inappropriate recreational activities and uses, including horse riding, cross-country sport using SUV cars and motorbikes, or for the development of golf courses where they are in-filled or made more permanent (Bouahim et al., 2010; Grillas et al., 2004; Serrano et al., 2006; Zacharias & Zamparas, 2010). Furthermore, climate change, resulting in modified seasonal and annual temperatures together with altered precipitation patterns and amounts, negatively affect MTPs. Indeed, these climatic elements drive the critical hydrological processes which, in turn, control the ecology of MTPs. The modification of these systems, in the long term, puts at risk the long-term existence of many plant and animal species (Caria et al., 2021).

A focal area for MTPs in the western Mediterranean Region is the island of Sardinia. The prevalence of the toponym ’Paulis’, used in the island language to name MTPs, suggests that these habitats were formerly more abundant and must have had some social recognition at a local scale in the past (Bagella et al., 2016). Despite this, the lack of information on their ecology, biodiversity, temporal dynamics, and spatial distribution on a regional scale has made their inclusion in conservation programmes complicated.

Key Project Features

Two projects have recently focused on these neglected habitats, both implementing scientific and educational knowledge, supporting interventions for areas of conservation and filling gaps in the ecological networks. The ‘Paulis Project’ funded by the Region of Sardinia in 2012 had the aim of generating interest in MPTs through the implementation of scientific research and development of educational products. In addition, the ‘Cli-P-on’ project run from the University of Sassari was designed to evaluate the responses of vascular plants within MTPs to the variation in hydrological regimes, which are in turn influenced by climate change. These projects should provide opportunities for developing adaptive management strategies and recovering lost and degraded MTPs, an important outcome given that they can be considered suitable ‘sentinels of climate change’ (Céréghino et al., 2014; Williamson et al., 2009).

The goals of both The Paulis and Cli-P-on projects are intended to support and facilitate the restoration and conservation of MTPs. These habitat types are included in the European Community Natura 2000 network which is a series of protected areas covering Europe’s most valuable and threatened species and habitats. The MTPs represent some of the core breeding and nesting sites for rare and threatened avian species in the EU, and this network aims to ensure the long-term survival of

Europe's most valuable and threatened species and habitats, listed under both the Birds Directive and the Habitats Directive.

A return to traditional land use is a crucial factor for MTP conservation. Extensive human activities, in particular their utilisation for moderate grazing, have maintained plant diversity in these habitats over the centuries. In contrast, land abandonment has had adverse effects on plant assemblages (Bagella et al., 2010; Rhazi et al., 2001). The positive impacts of traditional grazing practices appear to be due to the inherent resilience of these temporary wetlands in response to soil and vegetation disturbance (Dovrat et al., 2014). It has been suggested that moderate disturbances by cattle enhance plant diversity (Bagella et al., 2010) because this mild disturbance controls the density of competitive shrubs and prevents colonization by opportunistic species from the surrounding areas (Ferchichi-Ben Jamaa et al., 2014). In the same way, moderate disturbances by wild boars, through the reshuffling of the seed bank and the creation of open spaces, favour the development of species typically found within MTPs (Caria et al., 2021). MTP responses to these disturbances mirror those which are adopted in the face of frequent disturbance generated by alternate flooded and dry stages (Carta, 2016). On the other hand, over-grazing by domestic animals can cause eutrophication, can permanently damage sensitive vegetation, and create excessive physical disturbance (Bouahim et al., 2014).

Major Project Outcomes

Promoting relevant practical activities for the conservation of MTPs requires effort to disseminate information and increase public and political awareness of the importance of this habitat type. During the last few years, policymakers responsible for managing protected areas and developing Natura 2000 have gained understanding and been informed about the conservation status of these habitats and the possible threats at the local scale. Indeed, the European Habitat Directive (92/43/EEC) includes MTPs amongst the priority habitats (Annex I) for which conservation actions are immediately necessary. Also, the contribution and engagement of the public is required for the conservation of MTPs through awareness and education campaigns. MTPs are ideal for engaging the public in practical actions since they provide opportunities to increase public knowledge of biodiversity. To assist with this, in the framework of the 'Paulis Project', an online interactive guide for plants growing in the MTPs has been created (http://dryades.units.it/stagnisardi_en/).

Evaluation of the achievement of the conservation targets in Europe is achieved by measuring the temporal trends in the status of species and habitats (Gigante et al., 2016). In Europe, habitat monitoring is mandatory every 6 years for every country, arising from Art. 11 and Art. 17 of the 92/43/EEC Habitats Directive. Every nation is required to assess conservation status and to evaluate if the EU biodiversity policy has been effective (Evans, 2012; Henle et al., 2013).

In Italy, a national methodological tool for Annex I habitats monitoring is available (Bonari et al., 2021; Gigante et al., 2016). The criteria considered to assign to a particular habitat a 'Favourable Conservation Status', include (i) its distribution, in terms of both its natural range and area, (ii) its structure and functions, and (iii) the conservation status of its 'typical' species.

The Paulis project resulted in an international conference and published proceedings and had a following on social media. Our knowledge of the functioning and value of this wetland type is improved. This provided improved awareness of the role and value of these systems. The progress towards the goals is published online regularly (e.g. https://epi.yale.edu/epi-results/2022/country/ita). However, to date, wetland loss still exceeds wetland restoration in most places. Although we know how to restore these wetlands, the lack of financial resources and public motivation often hinder progress.

Summary

- A broadscale goal has been set in the EU in relation to habitats and water quality. It might not account for the future hydrology (in a changing climate) and management of MTPs is diffuse and largely in private hands.
- The project was based on the knowledge of the potential water regime required for the reestablishment of habitat.
- The regenerative processes are largely passive (i.e. a consequence of connectivity with sources of regeneration and seed banks) and few active works (e.g. restoration of agricultural uses) are likely to be needed.
- There are governance resources available for undertaking restoration (government protections and incentives) but the community needs more knowledge and support.
- Restoration of MTPs will require more management and financial support in the long-term and monitoring of progress is undertaken.

Case Study 6: Temporary Wetland Challenges: Seasonal Herbaceous Wetlands (Victoria, Australia)

Project Rationale and Strategies

This case study does not illustrate restoration of an individual wetland, rather it is about ongoing efforts to restore a landscape of wetlands for the conservation of multiple species. This study started with documentation of the problem and recognition of the biological and ecological values of these wetlands. It continues with mobilisation of the available protections and restoration incentives, investigations into socio-economic drivers of degradation, and the development of best management practices. It is recognised, for this mosaic of spatially isolated, temporally variable wetlands, that conservation of a single wetland is merely creation of a zoo and that effective conservation of the ecological function of these systems entails a landscape-level approach. One example of the landscape value of these systems is their use by water birds. Brolgas (*Grus rubicunda*: Australian cranes) use clusters of wetlands for successful breeding; they typically nest in one area then move with their chicks to nearby wetlands to eat frogs, insects, tubers, and other prey (Veltheim et al., 2019). They also use clusters of wetlands, after the chicks fledge, as flocking sites for the summer season (King, 2008; Veltheim, 2018). Brolgas have been listed as *endangered* in Southern Australia, and as wetland habitat has been reduced and agriculture has expanded, numbers of these long-lived birds have declined. Extinction is possible if the landscape of wetlands is not restored.

Temporary wetland systems have been poorly valued in many cultures (Kingsford et al., 2016). For example, in contrast to the attitudes of the indigenous Djap Wurrung and Gunditjmara peoples of Western Victoria, early European settlers thought of wetland areas as unpleasant places and a hindrance to productivity and movement (Mitchell, 1839). Despite the recognition that large populations of water birds used wetlands for feeding and breeding, the seasonal herbaceous wetlands (SHW) of Western Victoria were often used by settlers as dumping grounds and sumps for wastewater, or alternatively, drained and converted to 'productive pasture'. These ongoing practices resulted in a systematic decline in the overall area of wetlands and to a decrease in the number of individual wetland systems. The introduction of large machinery late in the twentieth century, capable of large-scale clearing and crushing of rocks, digging of ditches and ploughing lands, intensified wetland loss enabled the conversion of large numbers of remnant wetlands to cropping land (Casanova & Casanova, 2016). Of particular concern is that these damaging process are ongoing across this region (Farrington et al., 2019) (Fig. 6.11).

Major Concerns and Barriers

The Seasonal Herbaceous Wetlands (SHWs) of Western Victoria, which are known as 'swamps', were largely unappreciated by landholders and regulators until it was realised that they represented a large and important proportion of the region's wetland systems (GHCMA, 2010). In addition, they contain significant areas of native vegetation in an otherwise highly utilised agricultural landscape (Willis, 1964). Recognition of their intrinsic ecological values (Casanova, 2012; Casanova & Powling, 2014) coincided with one of the longest droughts in the region's history; from 1998 to 2010. During this time, these areas were rarely filled with water and were often completely dry for long periods, and as a consequence, many were easily converted to cropping land (Casanova & Casanova, 2016).

In contrast to the situation in the Fleurieu Peninsula, where the wetlands were listed as *critically endangered* quite early (2003), SHWs are less recognised and

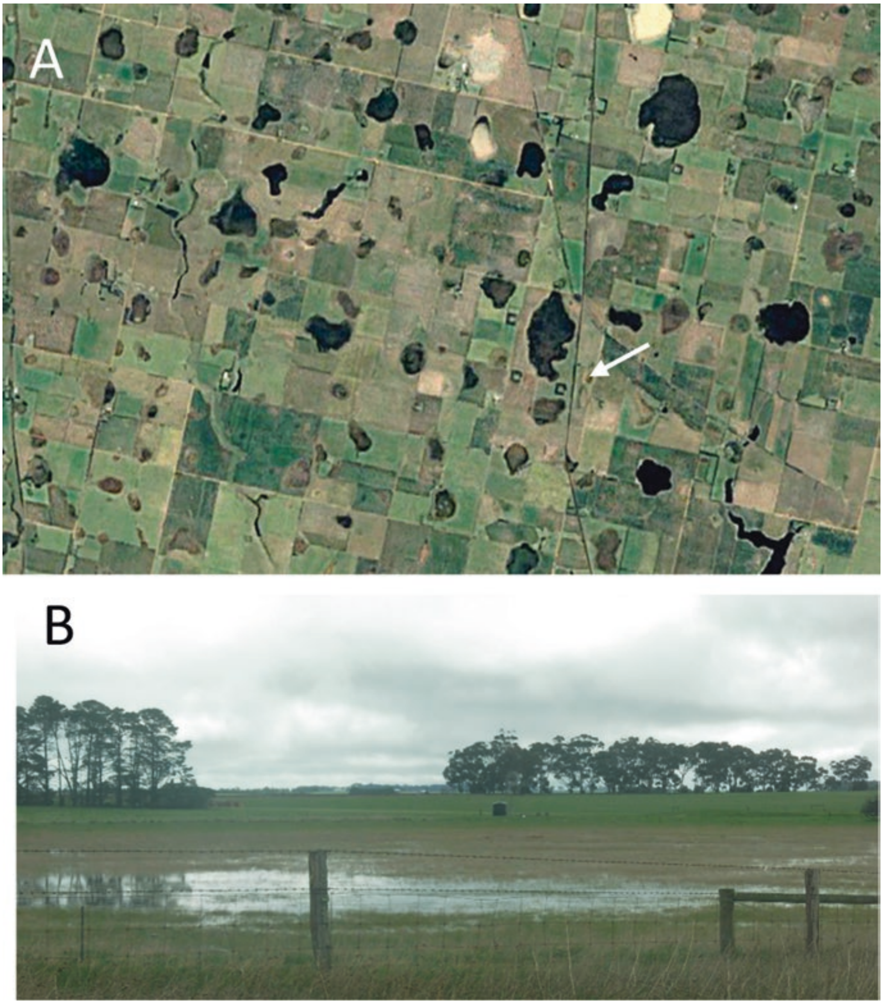

Fig. 6.11 (**a**) The landscape of Seasonal Herbaceous Wetlands (dark patches) in the vicinity of Westmere, Victoria (Image: Landsat; Copernicus Dec. 2011), the arrow indicates the position of Pines North Swamp. (**b**) Pines North Swamp from the ground in 2010. Pines North Swamp is relatively inconspicuous when compared with the size and depth of nearby wetlands, yet studies have shown that even such small wetlands have high biodiversity values

more poorly valued by many land managers (Curtis & Meis-Harris, 2020) only being listed as *critically endangered* ecosystems in Australia in 2012 under the *Environmental Protection and Biodiversity Conservation Act* (1999). Enforcement of their protection is rarely undertaken (https://www.awe.gov.au/environment/epbc/compliance-and-enforcement/compliance-outcomes 09/04/2021), and although they are protected by native vegetation clearing regulations that are enforceable by local government (https://www.environment.vic.gov.au/native-vegetation/native-vegetation), these protections do not appear to be implemented.

Today, these areas represent a management challenge for farmers, conservation agencies, and land managers. From an agricultural perspective, they are either too wet or too dry. They often have hostile soil conditions for cropping (high clay content), and these conditions also prevent traditional exotic pasture establishment and persistence. When grazing was the major agricultural pursuit in the region, this was not such a problem as sheep and cattle could graze the native vegetation in these areas when they were dry. From a restoration and conservation perspective, the temporary nature of these systems means that they are not always recognised as being wetlands, and their disconnection from sources of water, other than rainwater, provides few options for management of the water regime. However, it is now becoming clear that detailed understanding of the nature and functioning of these systems is urgently needed for their effective management so they can be preserved and continue to contribute to the natural environment.

Key Project Features

Given that the increasingly warming climate will continue to change the frequency and duration of flooding in these systems, restoration of the landscape of wetlands must focus on the maintenance of system resilience at a number of different scales. Appropriately resilient systems will have elements of diversity and connectivity that can allow them to adapt when conditions change. Recognised mechanisms that contribute to resilience include (i) high biodiversity (to facilitate species redundancy, so that there are a number of species with overlapping ecological roles), (ii) connected distribution of wetlands in the landscape (to allow migration of animals and dispersal of plants amongst individual wetlands), and (iii) the existence of healthy seed and tuber-banks (for the re-establishment of vegetation when an inundation occurs). Although many of these wetlands have been modified or drained, it is anticipated that a focus on conserving or restoring clusters of wetlands could significantly facilitate resilience across these systems. This approach could restore the mosaic of interconnected wetlands across the landscape which would provide critical habitat at the landscape-level for various fauna including migratory water-birds and other specialist wetland species such as frogs, algae, and plants.

Because SHWs are temporary systems filled mainly by rainwater and endorrheic catchment run-off in the winter and spring, the species found in them have adaptations to cope with prolonged dry periods. In addition, many of these specialised species require periods of dryness (e.g. when wetlands to dry out annually during summers). Moderate grazing by native or domesticated animals (sheep, cattle), together with the variable water regime, represent disturbances that can support the coexistence of a diverse range of plants and animals. Initial attempts to conserve and restore these wetland systems entailed scientific studies to document (i) their biological values (Casanova & Powling, 2014), (ii) the threats to their existence (Casanova, 2012; Casanova & Casanova, 2016), and (iii) their geographical distribution and their size (Papas & Moloney, 2012).

It is suggested that the three most important things for restoration of SHWs are (i) the return of an appropriate water regime through removal of drains, (ii) protection of the seed and tuber banks, and (iii) appropriate management. Apart from actions to prevent the drainage of additional wetlands, in many cases, there are no feasible economic options to deliver more water to damaged wetlands. However, blocking any existing drains to retain water in wetlands for longer periods represents an important first step in their restoration. After mitigating hydrological issues, control of disturbances created by cropping (e.g. physical disturbance, application of soil ameliorants and biocides) during dry periods is required. Regarding the protection of the seed bank, which is a major component of resilience, it has been suggested that it is more about what you don't do, than what you do (Casanova & Casanova, 2016). For example, avoidance of chemical pollutants to the area and protection from herbicides, insecticides, and biocides is often sufficient. It has been noted that current studies are focusing on finding the optimal range of grazing regimes that maintain biodiversity values (Morris et al., 2013). Investigations are therefore needed into which grazing species are appropriate for these sensitive areas, how often grazing should be allowed, and how intense these grazing regimes might be. In addition, further investigation into the role of fire as a vegetation management tool (to either stimulate germination or remove biomass) in wetlands is needed.

Major Project Outcomes

Progress has been made in the last 20 years. Temporary wetlands are now recognised as catchment 'assets' by the regional Catchment Management Authority; their decline is recognised as a matter of environmental significance by the Australian Government; they are mapped in publicly accessible geographic information systems; their biodiversity is being researched; land-holder attitudes are being assessed to facilitate communication and education; and management strategies are being developed in collaboration with landholders. There has been a focus on educating and creating awareness by highlighting their value to fauna, particularly frogs and birds. The major impediment to large-scale restoration is the fact that these wetlands exist largely on private land, managed by many individuals, whose primary interest is agricultural productivity. The way forward likely involves supplying incentives and modifying social drivers of agricultural expansion.

Summary

- No landscape-level goal has been formally articulated for SHWs even though they have been listed as *critically endangered*. When one is set, it should account for future hydrology (in a changing climate). Management of SHWs is diffuse and largely in private hands.

- Restoration activities are based on the knowledge of the potential water regime required for the reestablishment of habitat.
- The regenerative processes are largely passive (i.e. a consequence of connectivity with sources of regeneration and seed banks), and few active works (e.g. blocking drains, intermediate grazing, fire management) are likely to be needed.
- At the highest level (federal government) there are protections, but at the regional level (Local Council), there is little activity. At the catchment scale (Catchment Management Authorities), there is improved recognition of the issues and support for restoration. Although government protections and incentives exist, the community of stakeholders needs more knowledge and support.
- Restoration of SHWs will require more management and financial support in the long-term and monitoring of progress needs to be undertaken.

Chapter Synthesis

Overall, human-induced changes to wetlands and riparian systems have resulted in reduced global biodiversity and impaired environmental function across many landscapes. More recently, there is recognition of the close relationship between human existence and environmental function, and the role of traditional and First Nations management of wetlands for continued existence and biodiversity conservation (e.g. McLeod et al., 2018).

Several considerations become apparent in examination of these case studies (after Roberts et al., 2017b):

- The goal that is set in the initial stages is hugely important. It should be (i) feasible; (ii) account for the future hydrology and hydrological management; (iii) account for legacy issues concerning nutrients and water quality; (iv) articulate a reason for restoration and (v) be based on the range of achievable outcomes.
- The future water regime will dictate (i) the vegetation that is possible; (ii) how nutrients and other resources are cycled. Knowledge of the potential water regime or the capacity to control the water regime will influence the goal and success of restoration.
- Whether the regenerative processes that occur are passive (i.e. occur as a consequence of system resilience or process restoration (e.g. connectivity with sources of regeneration, seed banks, improved flow) or active (e.g. sediment removal, planting or structural works).
- The financial and governance resources available for undertaking restoration (e.g. legislative protections, facilitation, and incentives). This includes community capacity and knowledge.
- The realisation that there will always be some degree of active management informed by monitoring.

If, in the initial stages, a realistic and feasible goal is set that accounts for the legacy problems and consequences of change, that considers the process by which restoration will occur, is supported by good governance, community expectations, and resources and provides for on-going management and monitoring, then success is possible.

Success should be gauged by measurable parameters that indicate a functional change, such as water clarity or recovery of plant or animal populations. It has been suggested that setting up a monitoring regime to detect a positive trajectory of change in the short to medium term (Bachmann & Farrington, 2016), rather than expecting a stable state outcome quickly after works have been completed, can provide an alternative, effective, and easily communicated way of measuring restoration success. This is particularly important given the time required for most ecological transitions to occur. It is also arguable that, as supported by these case studies, overall success in wetland restoration might also be measured by an improved relationship between humans and the natural environment.

Implications

Coming decades will present both old and new challenges. The old ones will remain: modification of water regime and eutrophication brought about by changing land-use and intensification of human utilisation of ecosystems, and introductions of exotic organisms that change or impact on stable process are clearly going to be with us for a long time. New challenges are emerging: climate change may result in temporary wetlands becoming less frequently flooded, and having shorter durations of flooding; permanent wetlands as the end-point of polluted water might be seen because of competition for clean water with other users; riparian systems may face further water extraction and reduction in flows. In addition, there are still challenges in relation to our knowledgebase of wetland functioning, the importance of keystone species and the role of wetlands in the environment. Added to this is the relatively poor reputation of wetlands in the minds of the public and policy makers. Phrases like '*drain the swamp*' in relation to addressing corruption in politics (e.g. by Ronald Regan in the 1980s and Donald Trump in 2016) simply reinforce the perception of wetlands as unpleasant places.

These issues suggest the following strategies and opportunities: (i) we need to connect people with nature so that they understand about the role and value of heterogeneity and biodiversity in conservation; (ii) we need to set appropriate goals for restoration; (iii) we need to implement best management practice that is space and time appropriate; (iv) we need to provide space for migration and movement of wetlands and rivers and their dependent flora and fauna in a connected landscape; and (v) we need to protect and manage any remnant areas before they are damaged. From a practical land management perspective, it may be sufficient to 'just add water' for the restoration of wetlands, but more complex environmental and socio-political strategies, more advised responses and interventions are often also required.

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Chapter 7
Using Degree of Natural Regeneration Potential to Guide Selection of Plant Community Restoration Approaches at a Restoration Site

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Summary and Key Lessons
This chapter draws on global case study literature and eight case studies (from seven continents) to argue the benefits of adopting a resilience-based approach to plant community restoration site assessment and treatment prescription. We propose that identifying a range of restoration 'approaches' to returning plant species to a site (involving levels of natural regeneration and/or levels of reintroduction) can help practitioners better match their interventions to the site's degree of degradation.

Management Implications
A resilience-based framework to guide restoration site analysis and treatment prescription can assist restoration planners and practitioners to clearly identify locations where natural regeneration-based treatments may be most feasible and effective – and where, how, and to what degree it is necessary to reintroduce missing components of the ecosystem's plant community due to resilience depletion. We emphasise the value of treatments that mimic natural triggers for regeneration and outline implications for planning, drawing on conservation planning principles. Considering that all restoration projects seek to reinstate the ecosystem's resilience, we conclude that treatments that harness or rebuild that resilience as soon as possible can be a key to timely restoration success.

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Introduction

It is widely recognised that the global development of ecological restoration best practice has made substantial progress over recent decades across terrestrial, marine, and freshwater ecosystems (SER, 2004, Gann et al., 2019), but it is also recognised that we need to always be asking 'could we do better?' As a contribution to this discussion, this chapter presents a case for the more deliberate adoption of resilience-based conceptual frameworks to help guide effective and appropriate ecological restoration, irrespective of ecosystem type.

Based on our backgrounds as practitioners from a range of continents who largely work with harnessing natural regeneration to restore degraded plant communities, we jointly consider that the most useful framework to guide successful recovery is one where site assessment and treatment prescription are informed by the degree to which a restoration site retains (or does not) the capacity for plant[1] species to re-establish by self-organised natural recovery. This capacity is termed 'resilience' in ecology – i.e., referring to an ecosystem's intrinsic capacity to 'bounce back' to an approximation of its former state after a disturbance if it has not been pushed beyond a tipping point (Holling, 1973; Westman, 1978; Dell et al., 1986; Holl, 2012; Tongway & Ludwig, 2012; McDonald, 2000a; Standards Working Group SERA, 2021; Gann et al., 2019).

The eight case studies and literature cited in this chapter show (for plant communities) that (i) high biodiversity outcomes can be achieved, often at lower cost, in cases where natural regeneration potential persists and is harnessed and that (ii) resilience assessment will help refine understanding about what species are missing and guide effective and efficient strategies for reintroduction on sites where resilience is fully or partially depleted.

For over two decades, restoration approaches have been characterised in the literature as falling into either 'passive' or 'active' categories, with the passive category usually referring to spontaneous natural regeneration and the active category referring to broadscale reintroduction (DellaSala et al., 2013; Clewell & McDonald, 2009; Holl & Aide, 2011). While these approaches may be helpful for defining the two ends of a continuum (Prach & Hobbs, 2008), this binary view has more recently been a source of misinterpretation, leading to frustration amongst practitioners. The source of this frustration is due to: (i) the term 'passive' being at odds with the often highly active practices needed to facilitate natural recovery; (ii) lack of recognition of the existence of a more nuanced continuum of interventions between the two extremes; and (ii) understatement of the level of ecological analysis and insight

[1] We note that, while the term 'ecosystem' is of course inclusive of fauna and the points made in this chapter can apply equally to the spectrum of recovery or reintroduction of fauna (including marine ecosystems based on corals and shellfish), we focus this chapter on how an ecosystem's vegetation community can be reinstated. This is to avoid the confusion associated with the fact that the vast majority of ecological restoration projects focus on first re-establishing the habitat and subsequently largely rely on natural recovery for the return of a diversity of fauna, with faunal reintroductions being less common.

needed to plan and implement the reintroduction of complex biodiverse plant communities rather than characterizing this as simply 'active'. Conceptual frameworks based on a more realistic continuum of interventions are emerging and are now the subject of current global discourse amongst practitioners (see Chazdon et al., 2021). We hope this chapter will provide further information for consideration about potential frameworks.

In this chapter, we also share insights about the complex responses of native plant species to both natural and human-induced disturbances, and the ways adaptation to the former during a species' evolution may offer potential to recover after the latter. We note that, currently, many of these insights are not found in books or journal papers alone, since much practical insight is still emerging from the work of restoration practitioners. As can be seen in the case studies reviewed and featured below, there are many such groups pushing the envelope of theory and practice and coming up with not only new restoration solutions but also a deeper understanding of natural recovery potential and limits of ecosystems in the face of a range of contemporary impacts. These insights are leading to new ways to harness modes of natural recovery for rebuilding resilient ecosystems.

Rationale for a Spectrum of Restoration Approaches Along a Resilience Gradient

We propose that the plant community components of restoration projects – whether large or small in scale – often call for a range of intervention 'approaches' where each approach is well matched to the degree of resilience remaining at specific locations within the site (Box 7.1, Gann et al., 2019; Standards Reference Group SERA, 2021). We argue that these approaches logically align along a gradient of condition that can be spatial and/or temporal. Potential for natural regeneration is usually observed to be higher where: (i) the site is in close proximity to remnant areas (Dovciak et al., 2005; Kirmer et al., 2008; Prach et al., 2015; Crouzeilles et al., 2020); (ii) impacts have been more recent, shorter duration or lower intensity; and/or (iii) impacts are similar in nature to disturbances to which the local native species are adapted (McDonald, 2000; Blackham et al., 2014; Hoscilo et al., 2013). Conversely, more complex reintroductions are more likely to be needed with increasing distance from propagule sources and in cases of extreme or very long duration impacts, particularly where these are dissimilar to those to which the species are adapted.

Simplistic, linear gradients of condition do not always apply, however, as some species in a plant community may be more vulnerable to the same impact than others or may have higher or lower capacity for propagule store, resprouting or migration. Long-grazed grassy ecosystems, for example, may lose more palatable species earlier than non-palatable species even though the impact type and intensity is similar at site scale (Wang et al., 2019) and extensively cleared rainforests may lose mature phase dispersal-limited species earlier than more readily dispersed species (Brown & Lugo, 1994; Banerjee, 1995; Kooyman, 1996).

Complementary to arguments for categorising restoration interventions on the basis of degrees and types of interventions anticipated to be required to assist natural recovery (Chazdon et al., 2021), we find that a conceptual separation of approaches based on whether or to what degree they can foster natural regeneration is a useful strategy for informing restoration planning and treatment prescription. Such distinction allows a restoration planner to clearly examine the degree to which plants within a restoration site may: (i) regenerate after removing drivers of degradation alone; (ii) require more active facilitation to remove barriers to natural regeneration; (iii) require selective or partial reintroduction alongside regeneration; or (iv) require complete reintroduction as a means of returning species to a site (Holl, 2012; Standards Reference Group SERA, 2021). Each of these four approaches is illustrated in turn in the next section of this chapter using case studies from the literature as well as case studies provided by the authors. These are presented in a sequence commencing with cases requiring the lowest subsidy and progressing towards the highest subsidy with respect to human transport of propagules to a restoration site.

Box 7.1: Ecological Underpinnings

Using natural regeneration potential and limits as an organising principle for identifying appropriate restoration interventions is underpinned by ecological theory coupled with common sense. Ecosystems – whether aquatic or terrestrial – invariably have a capacity to recover after disturbances and fluxes in environmental resources. This biological resilience is an intrinsic capacity conferred by the individual species that make up the ecosystems, derived from the adaptations developed by individual species during exposure (over evolutionary timeframes) to environmental flux and disturbances (Holling, 1973; Westman, 1978).

Differences can also be distinguished between responses of species to lower and higher impacts. Recovery after impacts that do not remove species entirely is referred to in ecology as 'secondary succession', while recovery after impacts that *do* entirely remove species (including their dormant propagules) is referred to as 'primary succession'. That is, in secondary succession cases, impacts may damage much of the above-substrate vegetation in a terrestrial or marine ecosystem, but the site may still retain 'in-situ' resilience represented by retained plant fragments or seed that may then drive recovery (Meiners et al., 2015). In primary succession cases, these biota are effectively entirely removed by the disturbance, hence recovery is dependent on 'migratory' resilience via recolonisation from surrounding areas, whether terrestrial or marine cases (Grubb & Hopkins, 1986; Walker & del Moral, 2003).

Success at many restoration sites, where natural regeneration has been allowed or actively fostered, shows that this biological resilience can lend a capacity for recovery after human-induced impacts to the extent these impacts are similar to disturbances and disturbance regimes to which organisms have

Box 7.1 (continued)

adapted (Cairns et al., 1977; McDonald, 2000a; Chazdon, 2014). In turn, this assessment allows insight into the thresholds beyond which the potential for natural regeneration is partially or fully depleted (Luken, 1990; Hoscilo et al., 2013).

While there are broad similarities between ecosystems with respect to in-situ regeneration mechanisms (i.e. stored propagules or resprouting), the speed of migratory recovery (colonisation) can differ markedly between ecosystem types, largely based on dispersal mechanisms. For example, marine vegetation communities (such as seagrass, mangrove and saltmarsh), are highly migratory as propagules are freely dispersed in water, while coastal terrestrial ecosystems similarly benefit from the oceanic spread of organisms to intertidal zones. Some tropical and subtropical rainforests are also relatively rapid in their colonisation due to high proportions of trees that are dispersed by flying frugivores (Corlett, 1998), although the more dispersal-limited species in these systems are slower to colonise. Riparian and freshwater wetland ecosystems also have relatively high resilience due to dispersal by water flows and waterbirds; while many species in terrestrial herbaceous ecosystems (such as grasslands) have some facility to disperse by wind or in animal fur or faeces. Some of the least migratory ecosystems are arguably those dominated by fire-adapted vegetation that depends upon in-situ resilience mechanisms that are more cryptic and less readily dispersed (geosporus or bradysporous seed stores and cryptic bud banks) (e.g. Gill, 1981).

Case Studies

This section contains case studies from peer-reviewed and 'grey' literature for each of the four approaches, as well as illustrative case studies provided by the authors.

Examples of a Spontaneous Regeneration Approach

A 'spontaneous natural regeneration' approach (sometimes referred to as a 'natural regeneration' approach) is understood to involve the removal of any external degradation drivers (a step required for any restoration approach) with no additional interventions undertaken (Holl & Aide, 2011; Gann et al., 2019; Standards Reference Group SERA, 2021).

Because natural regeneration is commonly observed to occur spontaneously in nature, it is tempting to assume two things. First, it is sometimes assumed that where regeneration occurs spontaneously, it may not be valid to refer to it as a 'restoration' approach, given that restoration is defined as an intentional 'activity' (SER, 2004). Second, it is often assumed that spontaneous regeneration can only occur where degradation arising from human agency is low (Ghazoul & Chazdon, 2017).

With respect to the first of these assumptions, it is logical to propose that, for an approach to be considered ‘restoration’, at least some degree of degradation derived from human impacts would need to have occurred *and* some degree of restoration planning carried out or intended (Clewell & McDonald, 2009). With respect to the second of these assumptions, it can be observed that instances of spontaneous regeneration are not always confined to low-degradation cases. In-situ recovery potential and capacity for speedy recovery of plant communities is certainly likely to be higher in lower degradation cases where soil profiles or other substrates are not significantly altered. However, human perceptions of what constitutes ‘high degradation’ can be skewed by superficial indicators such as high levels of above ground weed cover relative to native cover. Furthermore, even where in-situ resilience is severely disrupted, migratory recovery (i.e. colonisation from outside a site) can still be seen to occur after appropriate management of the receiving site. This is particularly so where the site is close to sources of appropriate propagules (Crouzeilles et al., 2020) or if timeframes are sufficiently long (Prach et al., 2020). Indeed, the observation that recovery can occur after high levels of disruption in nature (e.g. volcanic eruptions, large landslips, etc.) suggests that it is not surprising that recovery could occur after larger-scale human impacts to the extent the latter are similar in effect to the range of variation found in nature (Chazdon, 2014).

Case Study 1 (Box 7.2) from Central Europe summarises the results of a recent review of vegetation recovery amongst a range of various heavily disturbed sites including abandoned mines and quarries. The study found that spontaneous natural regeneration could be relied upon even in the most heavily disturbed sites studied. The study found that natural regeneration could be especially favoured when the following circumstances were present: (i) the environmental site conditions were not extremely changed (e.g. the substratum had not become toxic or the soil fertility had not been reduced to extremely low levels); (ii) the biota of the ecosystems had been naturally adapted to the type of disturbance experienced; (iii) the area of a disturbed site was not very large; (iv) where robust natural ecosystems still occurred in the vicinity; and (v) alien invasive species were rare or absent (Prach et al., 2020).

Other important factors include the observation that sufficient time needs to be available for regeneration processes to take effect, although this is still impacted by other factors that can dictate a project’s success or failure. In a world-wide meta-analysis of effectiveness or success of spontaneous regeneration after a strong disturbance, 60% of studies reported ‘success’ when a trend to reaching a target vegetation was evident during a period of 100 years; in 33% of studies, spontaneous regeneration was deemed to be only partly successful; and only in 7% cases was it totally unsuccessful (Prach & Walker, 2020). Besides latitude, the degree of human-alteration of a landscape emerged as the most important factor determining the success of natural regeneration, but such conclusions should be interpreted with caution; specific local context and conditions must also be taken into account when relying on natural regeneration of vegetation communities in restoration projects.

Case Study 2 (Box 7.3) illustrates that extensive spontaneous regeneration has also been observed after the cessation of widespread cattle ranching on long-cleared

Box 7.2: Case Study 1. Overview of Spontaneous Regeneration at Long-Abandoned Sites in Central Europe

In ecological restoration, the question is often asked: Does spontaneous regeneration/ecological succession proceed towards a potential natural vegetation state (i.e. expected endpoint community for the site conditions)? This issue was investigated at a country scale in central Europe elaborating 17 particular successional series in various heavily disturbed habitats located in various parts of the Czech Republic (Prach et al., 2016). The habitats included extracted peatlands, corridors of the former 'iron curtain', artificial fishpond islands and barriers, sedimentary basins, various spoil heaps after mining, various stone quarries, forest clearings, burned-down forests, road verges, sand and gravel-sand pits, river gravel bars and abandoned arable fields (totaling 2602 vegetation samples) compared with vegetation records from corresponding natural vegetation (i.e. reference) communities.

The results demonstrate that the spontaneous successions generally proceed towards the respective (and expected) potential natural vegetation state (Fig. 7.1).

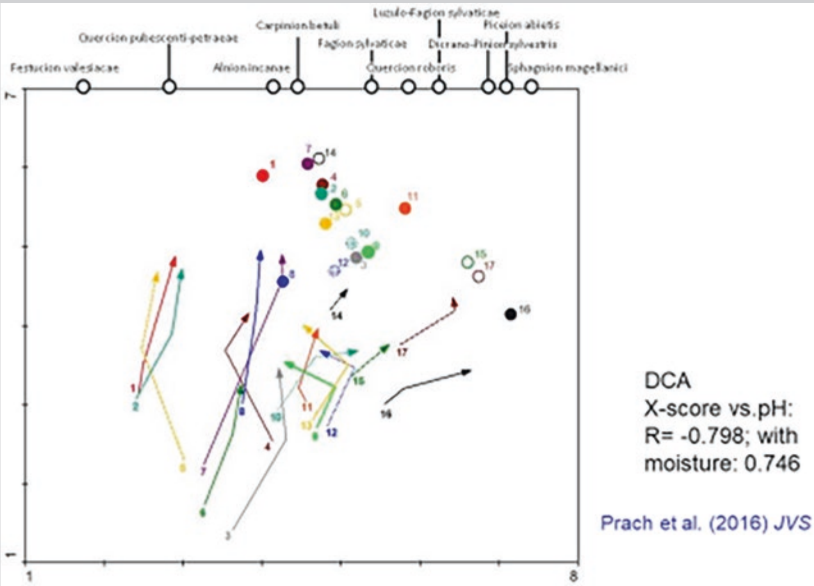

Fig. 7.1 Directions of recovery (through spontaneous ecological succession) towards the respective natural vegetation state (i.e. expected endpoint community for the site conditions). The diagram is based on dissimilarity in species composition amongst the samples (multivariate statistical analysis – the ordination method Detrended Correspondence Analysis) from successional series (arrows) and the corresponding natural vegetation state (points). The arrows connect average positions of early (1–10 year), intermediate (11–25 year), and late (>25 year old) successional stages. Above the upper line of the diagram, there are plotted positions of vegetation units of natural vegetation based on ca. 30,000 records from the whole country (adapted from Prach et al., 2016)

Box 7.2 (continued)

The several bent trajectories in the middle, not directly approaching the potential state are caused by (probably temporary) dominance of species that are not typical of the respective potential state. The average time needed to reach the potential state was between 200 and 250 years estimated by extrapolation. All species recorded in the reference communities (421) were also recorded in the recovering vegetation. It can be concluded that in the general view across the high number of examples of stages of recovery spread over the whole country, successions advanced in the direction of the corresponding expected state. However, in particular situations (such as toxic and extremely nutrient-poor sites, and sites with dominance of alien invasive species in late successional stages), spontaneous recovery may be less successful or completely ineffective. Such situations, however, occurred only very rarely. It should also be noted that early successional stages can be in some cases preferred in restoration projects over the late ones as they harbour higher species diversity and/or endangered species of various biota (Řehounková et al., 2016).

Box 7.3: Case Study 2. Spontaneous Natural Regeneration of Forests in Northwestern Costa Rica

In the 1960s, extensive areas of Costa Rica's Guanacaste province resembled an African savanna, dominated by species of African grasses that were imported to the region to form large cattle haciendas during the nineteenth century (Fig. 7.2). Most of the dry forests and seasonal moist forests of this

Fig. 7.2 In 2017, the previously cleared landscape surrounding the village of Hojancha in Guanacaste Province, Costa Rica, had recovered more than half of the original forest cover, largely through natural regeneration of forests on former pastures. (Photo Robin Chazdon)

Box 7.3 (continued)

10,140 km^2 region had been cleared over many decades and converted to ranch lands (Calvo-Alvarado et al., 2009). From 1960 to 1979, forest cover decreased even further from 37.8% to 23.6%, driven by a strong international export market for beef that also supported smallholder farmers.

A combination of factors at the end of the 1970s led to the widespread abandonment of grazing lands. These factors included a decline in global beef prices and the emergence of a novel 'payment for environmental service' programme for land owners (Calvo-Alvarado et al., 2019; Porras et al. 2018). In 1986, forest cover in Guanacaste reached a minimum of 23.1% and then began to increase. In 2005, forest cover reached 47.0%, even higher than reported for 1960 (37.8%), with an annual rate of forest regrowth of +1.26% (Calvo-Alvarado et al., 2009).

Forest gain was attributed to spontaneous natural regeneration in areas from irregular to very irregular topography, coinciding with the widespread abandonment of pastures in marginal areas with poor accessibility and low productivity due to steep slopes with shallow soils (Arroyo-Mora et al., 2005). From 1986 to 2000, the extent of forest cover within protected areas increased by 20% as a result of better fire control practices, primarily in tropical dry forest areas, such as Santa Rosa National Park (created in 1972) and Palo Verde National Park, created in 1978. A national 'payment for ecosystem services' programme (PSA) instituted in 1987 was initially not a strong driver of early forest regeneration in Guanacaste Province; the total area covered by PSA contracts in Guanacaste accounted for only 12% of the total natural forest cover of the province in 2009. But this situation changed; by 2015, PSA contracts for recovery of secondary forests accounted for more than 5000 ha (Calvo-Alvarado et al., 2019). From 2005 to 2012, forest cover in Guanacaste province continued to increase from 47.0% to 50.74%, despite substantial increases in international beef prices, suggesting that conservation and reforestation policies in Costa Rica (Forestry Law 7575 in 1997) effectively encouraged forest regrowth and prevented clearance of naturally regenerating forests (Calvo-Alvarado et al., 2019).

Across the Guanacaste landscape, forest cover is now a heterogenous mix of different successional stages of dry and moist tropical forest. Within the Guanacaste Conservation Area, at least 40,000 ha of former pastures are now covered with young regenerating forests due to protection from human-caused fires by local fire crews (Allen, 1988). Hulshof and Powers (2019) describe two dominant modes of secondary forest regeneration: via wind-dispersed seeds into large patches; or, via animal-dispersed seeds into smaller fragments that differ in deciduousness and habitat suitability for animals (Castillo-Nunez et al., 2011; Janzen, 1988). Today, Guanacaste's population is more urbanised, more economically focused on tourism and less invested in cattle production, an example of a forest transition driven by economic development and fuelled by both spontaneous and facilitated/assisted natural regeneration.

land in Costa Rica (Chazdon et al., 2020). Indeed, several studies from Latin America have documented extensive natural regeneration in dry and wet forest ecosystems and in montane regions (Lennox et al., 2018; Aide et al., 2019; Nanni et al., 2019), largely due to cessation of agricultural activity (cropping or grazing) and the associated burning of fields. Unfortunately, evidence of reversals of natural regeneration is common in many regions (Reid et al., 2019; Schwartz et al., 2020; Wang et al., 2020; Sloan, 2022), indicating that, in many contexts, regenerating ecosystems will require protection and economic incentives for landowners and farmers to ensure their long-term persistence (Case Study 2, Box 7.3, Chazdon et al., 2020). Such vulnerability to re-clearance indicates the high importance of addressing socio-economic and policy barriers for long-term restoration success (Chazdon et al., 2020; Smith et al., 2020).

Examples of a Facilitated Natural Regeneration Approach

Many projects from around the globe include interventions to stimulate, assist, or facilitate natural regeneration of vegetation in areas where, for one reason or another, natural regeneration cannot proceed spontaneously or would proceed too slowly with the simple removal of external degradation drivers (Standards Reference Group SERA, 2021; Gann et al., 2019; Allison & Murphy, 2017). These areas have been found to encompass entire restoration sites or, more commonly, smaller areas within a larger site containing areas of variable condition.

Case Study 3 (Box 7.4) from a riparian site in Canada shows how active removal of barriers or filters to natural regeneration – applied either as a one-time or repeated interventions – resulted in a succession of forest trees (Polster, 2017). Other active interventions that have resulted in native vegetation recovery around the globe include the removal of competition by undesirable species, whether native or non-native (Luken, 1990; Lamb, 1993); reinstating physical and chemical substrate conditions (Bradshaw, 1983; Clewell & Aronson, 2013); reinstating processes such as flooding and drying or fire to trigger germination of seed or hatching of eggs (White & Pickett, 1985); and/or, attracting seed-dispersing fauna using debris piles and perches (Ludwig & Tongway, 1996; McClanahan & Wolfe, 1993; Holl et al., 2000; Zimmerman et al., 2000). Migratory potential can also be facilitated by the installation of faunal attractants including water sources and shelter to increase visitation by dispersing fauna (Ludwig & Tongway, 1996; McClanahan & Wolfe, 1993; Holl et al., 2000; Zimmerman et al., 2000). These interventions are informed by, amongst other things, the likely natural conditions or processes to which the pre-existing species have become adapted over evolutionary timeframes and a knowledge of specific barriers that reduce the rate or quality of natural regeneration.

It is important to conceptually separate the facilitated restoration approach from other approaches because potential for natural regeneration after assistance is frequently underestimated, often because it is not easily observable. Sources of vegetation regeneration potential (seed or buds) are often hidden within substrates, and plant tissues and migratory propagule sources often go unnoticed. In addition, this potential can be obscured by competitive exclusion by non-native or highly invasive plants that colonise or recover quickly and dominate disturbed sites. Generalisations that natural regeneration is only possible in cases of low anthropogenic impact are often misinterpreted to mean that if natural regeneration does not happen spontaneously, then it cannot happen at all. There are many cases, however, that show that in situ natural regeneration capacity can extend much further into higher degradation zones than might be anticipated. i.e. where appropriate seed germination triggers are applied and regeneration niches are kept competition-free for a critical establishment period. For example, a 3-year restoration contract to trial the application of fire followed by rigorous weed management on a a small, retired grazing property in Australia resulted in widespread germination from the soil seed bank of 35 species of native forbs, sedges, and grasses, as well as recruitment from seed rain from nearby remnants of seven species of trees and shrubs. All the recruiting species were characteristic of the site's long-cleared forested wetland. This result was surprising as, prior to treatment. Most of the site was highly dominated by tall, exotic grasses and appeared to be in a condition that even regeneration specialists assumed would require mass planting (Coward, 2015).

Indeed, natural regeneration can be facilitated in sites showing relatively high degradation effects, even where both physical and chemical conditions have been altered. In areas with intensive agriculture, such as The Netherlands, for example, where (as in many other places) nutrient-enriched topsoil favours dominance of strong competitors including alien species, removal of that topsoil initiated natural regeneration and succession towards heathlands or wetlands (Klimkowska et al., 2007). This process was also reported from abandoned fields in Denmark (Ejrnaes et al., 2003). Liming of excessively acidic substrates has also facilitated natural regeneration and succession on spoil heaps created by sulphur mining (Melgar-Ramírez et al., 2012) and in acidified lakes (e.g. Rogora et al., 2016) and the recharging of a water table often facilitated natural regeneration in previously industrially harvested peatlands or other degraded wetlands (Joosten et al., 2017). Various other instances where interventions initiated natural regeneration have occurred in Europe. For example, the reinstatement of regular mowing into degraded hay meadows where a strong competitive generalist species had long-dominated the area was found to result in more diverse recovery (Poschlod & WallisDeVries, 2002) and selective, low-impact tree harvesting has been shown to initiate re-establishment of native forest species in forest monocultures (Jonášová et al., 2006).

Box 7.4: Case Study 3. Using Natural Processes for Restoration After the Removal of Heber Dam, British Columbia, Canada

The Heber River Diversion Dam (Heber Dam) on Vancouver Island, British Columbia, Canada, was removed in 2009 to return the flows in the Heber River to pre-dam conditions and restore the footprint of the dam and its 3 km pipeline. The restoration treatments were modelled on natural successional processes, harnessing natural colonisation. The recovery of dam and pipeline removal disturbances was initiated in 2012 with the fall season dispersal of seeds from mature pioneering species that formed a significant part of the nearby local undisturbed vegetation.

As the barriers to natural recovery were compaction of substrates and a lack of micro-sites, works involved creating a rough and loose soil surface, with an array of mounds and holes to promote infiltration (avoiding erosion) and creating varied micro-sites for a diversity of species to establish. Large woody debris was added at a rate of 100 m^3/ha. Commitments had been made to the Mowachaht/Muchalaht First Nation that they would be involved in the restoration work. A First Nation crew was hired to transplant about 1000 ferns which also brought in a host of soil micro-organisms.

Natural recovery processes led to an increase in plant species numbers from 32 to 84 species over five growing seasons (Polster, 2017) including five species of conifers, while the percent cover of the vegetation increased every year to 54% at the fifth year (Figs. 7.3 and 7.4). Red alder (*Alnus rubra)* trees

Fig. 7.3 July 2013. Dam area. Rough substrate of the disturbed area surrounded by stands of red alder (*Alnus rubra*). (Photo David Polster)

Box 7.4 (continued)

Fig. 7.4 June 2017. Same area showing prolific regeneration of red alder along with 80 other species. The alder trees are starting to thin out while other species move in. (Photo David Polster)

had exceeded the agreed stocking rate (4500 stems/ha) and conifers and fruit-bearing plants were found in 98% of the 50 sampling plots. Early colonisation of red alder fixed nitrogen and provided a deciduous cover over the slower-growing conifers that ensured their protection during summer, while providing sufficient light for them to photosynthesise in winter. The cost of the restoration work has been significantly less than the cost of traditional reclamation treatments, such as tree planting, and the diversity of established species has been far greater than with such traditional treatments.

The density of red alder will reduce over coming decades and the understory vegetation will shift as the overstory changes and soils build over time on the relatively inert gravels. By allowing species composition and cover to be dictated by natural processes, diverse ecosystems appear to be establishing on the disturbed sites with every indication that this process will continue.

In Australia, a method known as 'bush regeneration', which is based on releasing native plant species and their niches from weed competition, was initiated in Sydney in the late 1960s (Bradley, 1971; Buchanan, 1990). The simplicity and effectiveness of this method rapidly resulted in the formation of an industry based on the engagement of bush regeneration contractors and other groups to restore degraded urban remnant vegetation in Sydney from the late 1970s, spreading to some other cities and regions in the early 1980s and onwards (Gye & Thomas, 2007). The methodologies and underpinning theory were further developed and adapted over the last five decades, progressively becoming better informed by the resilience responses of species from the particular target ecosystem. In fire-adapted sclerophyll and grassy ecosystems, for example, it soon became clear that competition management alone was insufficient to trigger recovery of disturbance-adapted species and that applied disturbances, including fire or light soil disturbance, may also be important to stimulating optimal diversity levels (McDonald, 1995; McDonald et al., 2002). Similarly, flooding and drying was found to be beneficial for facilitating regeneration in freshwater (Nias et al. 2003) and saline wetlands (Alexander et al., 2008); while leaving poisoned tree weeds as bird perches was found to be highly important in facilitating colonisation in rainforests (Cockbain, 2016). We note that all these innovations mimic the characteristic natural triggers or facilitators for regeneration that have developed through adaptation over evolutionary timeframes in these ecosystem types. During the 1990s, the practice was locally referred to as an 'assisted natural regeneration' approach, with the term referring solely to interventions designed to trigger natural regeneration, with supplementary reintroduction considered a separate and complementary approach.

Around the same period, an important and highly influential system of tropical forest restoration and management emerged, which is now generally referred to as Assisted Natural Regeneration (ANR) (Shono et al., 2020). This is a method or management system applied largely for the benefit of traditional landholders and subsistence farmers who are managing declining tropical forests. The method involves the removal or reduction of barriers to natural regeneration as well as enabling a degree of enrichment planting, particularly for social benefit (Chokkalingam et al., 2018; FAO, 2019). One of the earliest published examples of ANR is the case of tropical forest regeneration in the Philippines, in which a method was described for facilitating rainforest seedling recovery through managing *Imperata* grass dominance (Shono et al., 2007). The ANR method was officially introduced in 1989 through a Department of Environment and Natural Resources (DENR) Memorandum Circular No. 17 and has become mainstreamed in reforestation programmes since that date (Castillo, 2017). Other examples of ANR in tropical ecosystems occur in Brazil (Crouzeilles et al., 2020; Alves et al., 2022), southeast Asia (Dugan et al., 2003), China (Liu et al., 2017), and India (Box 7.4. Venkataraman, pers. comm. 2021).

Case Studies 4, 5, and 6 provide examples where a predominantly facilitated natural regeneration approach has been applied in developing countries in Indonesia, South India, and Ethiopia, respectively. In the areas where natural regeneration was facilitated or assisted, higher value ecological outcomes were attained (at a lower cost) than would have been the case had recovery in those areas relied upon a reintroduction approach.

Case Study 4 (Box 7.5) illustrates how, in a tropical peatland in Kalimantan, Indonesia, tropical peatlands degraded through draining and excessive burning have been helped to recover by the reinstatement of the high-water table hydrological regime and preventing fire, which is not a natural part of the system; again using methodologies suited to and embraced by the local communities. Case Study 5 (Box 7.6) illustrates how a predominantly facilitated regeneration approach has reinstated important habitats in South India, tailored to suit both the ecosystem and the local communities.

Case Study 6 (Box 7.7) shows that a somewhat similar methodology, Farmer Managed Natural Regeneration (FMNR) (Rinaudo et al., 2021), has been developed in Africa to harness natural regeneration and thus counter extreme environmental degradation and improve the livelihoods of subsistence farmers. This methodology focuses on not only removing stresses upon surviving rootstocks to enable them to regenerate but also on reviving the traditional horticultural practices of pruning and coppicing trees to optimise their growth and productivity to benefit villagers.

A facilitated regeneration approach is also frequently applied in grassy ecosystems and rangelands in temperate zones of North America (Papanastasis, 2009, and see http://www.nagrasslands.org/category/beneficial-management-practices/grazing-management/). It is also practiced in grassy ecosystems in pastoral and conservation sectors in Australia where the techniques of managed grazing, soil disturbance, and weed control are variously applied to create niches and apply triggers for natural regeneration from soil-stored seed or seed rain (Lodge & Whalley, 1985; Eddy, 2005; Davidson & O'Shannessy, 2017; Shorthouse et al., 2012; MCMC, 2019 and see https://www.environment.act.gov.au/nature-conservation/conservation-and-ecological-communities/grasslands/grasslands-restoration-project). A facilitated regeneration method commonly known as 'waterponding' has also been applied to over 40,000 ha of severely degraded scalds in semi-arid rangelands in inland New South Wales, Australia. ('Scalds' are areas of severe wind erosion where sandy topsoil has been entirely removed leaving the clay subsoil. See https://site.emrprojectsummaries.org/2019/10/21/waterponding-the-marra-creek-nsw-rangelands-update-of-emr-feature/) This work resulted in high levels of cover from colonising native species after rainfall retention followed by drying created deep cracking of the previously impermeable claypan surfaces. The formation of these deep, moist cracks created niches for the lodgement and germination of wind-dispersed seed, leading to the recovery of vegetation on sites that had previously failed to spontaneously regenerate despite many decades after removing extreme grazing pressure (Thompson, 2008; Thompson & CWLLS, 2019). This approach is increasingly being applied in other states of Australia.

Considering the strong evidence that facilitated natural regeneration can often increase species diversity and retrieve otherwise lost genetic material (even at some apparently highly degraded sites) – and at the very least can provide insights into which species are missing – we recommend that this approach should be carefully considered prior to applying reintroduction interventions. Formal reviews of case studies and higher levels of experimentation relating to this approach to ecological restoration is highly warranted and urgent.

Box 7.5: Case Study 4. Facilitated Regeneration in Tropical Peatlands, Central Kalimantan, Indonesia

Tropical peatlands in Kalimantan, Indonesia, are forested swamps that extend tens of kilometers between rivers, with peat depths of up to 10 m below the forest. The forests contain rare and endangered fauna, such as the Borneo orangutan (*Pongo pygmaeus*), clouded leopards (*Neofelis nebulosa*), and gibbons (*Hylobates* spp.). Over the last few decades, logging, agricultural and industrial plantation land conversion has led to the drainage of vast extents of these ecosystems (Dohong et al., 2017). With drainage comes high risk of fire, leading, in part, to the Asian Haze Crisis. Currently, only 6% of tropical peat swamp forests are classified as intact (Miettinen et al., 2016). However, global interest in tropical peatland restoration has increased in recent years, due to greater focus on climate change and the vast carbon storage associated with tropical peatlands (Harrison et al., 2020).

Indonesia contains more than 50% of the world's tropical peatlands. In 2015, during a severe dry season, more than 2 million hectares of tropical peatlands burned, releasing vast volumes of toxic haze. In 2016, the Indonesian president formed the Badan Restorasi Gambut (BRG) – *Peatland Restoration Agency.* BRG priortises restoration through the three Rs of Rewetting, Revegetation, and Revitalisation of Livelihood (Giesen & Nirmala, 2018). Whilst returning vegetation cover is a priority, Indonesia acknowledges that simply reintroducing vegetation will not address the underlying barriers – namely, disturbed hydrology and community use of fire on the land. By focusing on alleviating these two main regeneration barriers whilst simultaneously supporting vegetation recovery, chances of long-lasting success increase.

The Mawas Programme of the Indonesian non-profit, the Borneo Orangutan Survival Foundation (BOSF) supports the goals of the Indonesian Provincial and District Governments in Central Kalimantan in protecting and restoring a 309,000 ha tropical peat dome, containing one of the largest remaining wild populations of orangutan. Due to historically intense drainage in the region, half of this area has lost its forest cover and burns regularly.

BOSF Mawas prioritises a holistic approach to forest recovery. Replanting occurs in the most severely degraded areas – working in partnership with local communities to establish community-led seedling nurseries and community-managed replanted plots. At these planting sites, surrounding canals are blocked, and fire-prevention teams are established and they work hard through the dry season to discourage use of burning and quickly extinguish any fires. At the same time, community engagement and education programmes focus on training in and facilitating the establishment of alternative livelihood activities, such as fishponds, honey farms, rubber and paper pulp smallholdings, and swiftlet nest houses (Mahyudi et al., 2014) (Figs. 7.5 and 7.6).

Box 7.5 (continued)

Fig. 7.5 Aerial view of one of the ex-Mega Rice Project Canals, in Central Kalimantan, Indonesia, being blocked by BOSF, to recover hydrology in the area, reducing fire risk and enhancing natural regeneration. (Photo Laura Graham)

Fig. 7.6 A team of local community and BOSF staff members at a replanting (reforestaion) training event, at the front of the BOSF Mawas Mantangai Camp, Kapuas District, Central Kalimantan, Indonesia. Reforestation efforts are targeted to areas where natural regeneration capacity is lowest. Community engagement and the assisted regeneration interventions hydrology rehabilitation and fire management go hand-in-hand with reforestation efforts in tropical peatlands, ensuring reforestation also enhances surrounding natural regeneration. (Photo Laura Graham)

Box 7.5 (continued)

Over the last 15 years, BOSF-Mawas has worked with local communities to replant nearly 4000 ha, with more planned. With over 100,000 ha of degraded peatlands, however, BOSF does not hope to replant it all. Reducing the use of fire and raising the water table in the area protects planted seedlings, but also initiates natural regeneration, with many dispersers coming back into the area and canopy height and sapling diversity increasing, despite its severity of degradation. This facilitated restoration approach has high relevance for scaling-up restoration efforts across Indonesia's degraded peatlands.

Box 7.6: Case Study 5. Assisting Natural Regeneration to Restore Forest Ecosystems in South India

Junglescapes, an Indian non-profit organisation, has been restoring degraded forest habitats in a tiger reserve (Bandipur Tiger Reserve) in South India since 2008. The restoration sites are dry deciduous and thorn scrub ecosystems, both of which typically have high plant and animal diversity. Degradation has arisen from forest wood collection and livestock grazing, moderate to high invasion by alien species such as lantana (*Lantana camara*) and inappropriate forest fires. The restoration objectives are two-fold: revival of plant diversity and return of animals.

Initial restoration design was based on reintroduction through planting of saplings. However, a combination of highly degraded site conditions and low rainfall levels of 600–800 mm a year concentrated over a few days resulted in poor survival and establishment of planted saplings. As a result, the approach was changed after the first 2 years to assisting natural regeneration, focusing on removing barriers to the revival of natural recovery processes.

Restoration methods were focused on arresting and reversing soil erosion, increasing water retention, removing invasive species, and accelerating revival of native grass and pioneer species. Grass re-emergence was crucial to alleviate degraded soil and to prevent weed re-invasion. Actions implemented were generally able to address more than one of these goals. Moisture improvement actions included constructing trenches, rock detention structures, and swales. Gully plugs and contour trenches helped reverse soil erosion. Improved moisture levels helped native grasses expand from pre-existing patches in most areas. (Grass seed dispersal was limited to weeded areas which had poor grass presence and is not considered natural regeneration per se.) Species gaps observed after 5–6 years were addressed through seed dibbling (i.e. dropping of seeds in small holes of one to two-inch depth made manually in the ground and covering these holes with loose soil). When this occurs, the approach would be considered a combined regeneration/reintroduction approach.

Restoration actions to date have been undertaken across around 3000 acres (1214 ha). Plots under restoration for over 5 years see vegetation recovery of around 60% of the reference in the case of tree species and over 75% in the case of shrub and grass species (Figs. 7.7 and 7.8). Return of almost all native

Box 7.6 (continued)

Fig. 7.7 Condition of thorn scrub habitat in Bandipur Tiger Reserve in South India, prior to restoration treatment. This habitat is usually typified by large, open grassy patches with short-statured trees and a diversity of shrub and other middle-storey plants. In this instance, however, the native species were reduced due to high presence of the non-native invasive plant lantana (*Lantana camara*). (Photo Ramesh Venkataraman)

Fig. 7.8 The same site about 4 years after facilitated natural regeneration treatment including removing invasive species and accelerating revival of grass and pioneer species. (Photo Ramesh Venkataraman)

Box 7.6 (continued)

faunal species has been observed, resulting in significant seed dispersal benefits. Surveys show plant species appearing without proximate seed sources, indicating natural migration potential is possible beyond the in-situ limitations, through long-range seed dispersers such as birds, sloth bear (*Melursus ursinus*), Asian elephant (*Elelphas maximus*), Indian bison (*Bos gaurus*), sambar deer (*Rusa unicolour*), and other species.

The biodiverse recovery in apparently severely degraded sites or those that have remained degraded for long periods of time indicates that the potential of propagule banks, as well as the potential for seed migration, are much higher than historically estimated. The outcomes also indicate that assisting natural regeneration provides additional benefits of rapidly rebuilding resilience and self-organization. Hence, in such sites, this approach could precede other treatments. Restoration costs were around a fourth that of plant re-introduction methods, and this enabled scaling-up of restoration with the same resources.

Box 7.7: Case Study 6. Community-Managed Reforestation in Humbo Ethiopia

Forest clearing by humans for fuel wood and charcoal production, followed by free-range grazing and continued harvest of coppicing stems growing from live tree stumps, contributed to extreme landscape degradation in Humbo, Ethiopia. Any intense rainfall event would lead to flash flooding, destroying roads and bridges, submerging crops, and causing extensive erosion. Large amounts of topsoil had been lost, reducing productivity in sloping areas and leaving large silt and rock deposits on farms in lower areas.

World Vision's Community Managed Humbo Reforestation project combines development gains for the local communities with benefits for the environment. The regeneration of 2724 hectares of degraded native forests with indigenous, bio-diverse species was achieved by a method called Farmer Managed Natural Regeneration (FMNR). This method was co-developed with farmers (and ultimately whole villages) as a distinct practice in 1983 in Niger Republic as a response to deforestation, the failure of conventional tree planting practices and deteriorating livelihood conditions of small holder farmers.

The method is built upon the observation that, contrary to perceptions, apparently tree-less landscapes often contain a vast repository of living tree stumps and seeds with the capacity to regenerate and grow rapidly with pruning – at low cost, quickly and simply (with low technology), and this can be done at scale. Normally, spontaneous regeneration is inhibited by constraints such as continuous grazing pressure, repeated burning, and removal of woody

Box 7.7 (continued)

biomass either for fuelwood collection or as a land-clearing practice for agriculture. This 'discovery' led to the realization that the main constraints to reforestation were not technical or financial, but social and policy related. All that is required is a change of behaviour/land management practices to encourage natural tree establishment through managing resprouting woody vegetation and germinating seeds. In Niger, it is estimated that FMNR spread across the country at a rate of 250,000 hectares per year over a 20-year period and that some 200 million trees were regenerated on farmland in that time. Average tree density rose from 4 trees/hectare in the early 1980s to 40 trees/hectare today (e.g. Fig. 7.9). FMNR would never have spread at the rate it has unless farmers were given the freedom to adapt the practice to their particular situation and objectives and the authority to benefit from their work (i.e. the right to harvest timber and non-timber forest products).

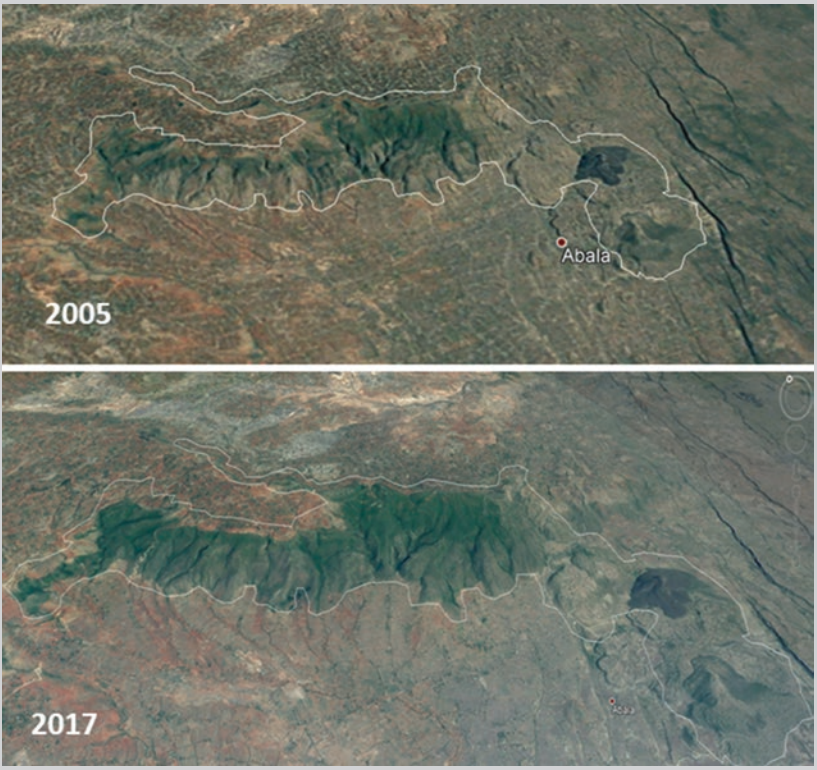

Fig. 7.9 Satellite images of Humbo restoration site taken in 2005 and 2017. Project activities commenced in 2006

Box 7.7 (continued)

The Humbo project (Fig. 7.10) now provides increased fodder, firewood, wild foods, and traditional medicines and a range of ecosystem services such as improved ground water and springs, decreased erosion and flooding, and increased biodiversity; and indirectly, by creating a new community-based income stream through the sales of carbon offset credits.

Fig. 7.10 Community members actively manage their forest, undertaking regular pruning and thinning activities to encourage rapid growth while providing direct benefits in the form of fuelwood, small poles and stakes, and fodder. (Photo Tony Rinaudo)

Examples of a Combined Regeneration/ Reintroduction Approach

A hybrid approach between regeneration and full reintroduction is applied where natural regeneration, whether spontaneous or facilitated, can only result in the return of a subset of the target ecosystem's plant species within the desirable timeframes, and there is a need and desire to supplement this recovery with reintroductions of missing species. Such missing species are usually those with lower in-situ or migratory resilience per se, and the problem emerges after a parent plant has been destroyed and no seed bank or migratory potential remains (e.g. Dell et al., 1986; Lunt, 1997; Prober & Theile, 2005). In an ecological restoration context, this approach may also be applied where reinforcement of genetic diversity of extant species is required to compensate for fragmentation effects or where a wider range of local species is needed to build climate adaptability (Prober et al., 2015).[2] There are two main modes for this combined approach which are described here; the selection of their use varies according to the timing of necessary reintroductions:

(a) *Cases where the project commences with interventions to facilitate natural regeneration – with reintroductions carried out later in the programme*

This mode is highly important in largely cleared sites where only some species are clearly persistent, but where the potential for missing plant species to colonise or recover from bud or soil seed banks may not be clear due to high weed presence and low native presence in the above-ground flora. In such cases, best practice would see a regeneration phase applied prior to any reintroduction phase to optimise natural recovery and to provide a diagnostic tool to identify which species, if any, are actually missing. This mode is also commonly used where threatened, underrepresented, or ecologically specialised species are found to be missing. For example, locally extirpated, threatened, or underrepresented plants – or large-fruited species (that are no longer naturally dispersed) – are often the subject of reintroduction while other species recover through spontaneous or assisted regeneration.

A combined approach is frequently used in tropical ecosystems where additional enhancement planting is combined with facilitated regeneration for ecological, social, or economic purposes (FAO, 2019; FORRU, 2005; Mahyudi et al., 2014). Interplanting of food crops is carried out on ANR sites in the Philippines (FAO, 2019) and agroforest development is undertaken through ANR in Indonesia (Burgers et al., 2014). Where these plantings are not local native species and may not be part of the ecological restoration, their deliberate integration illustrates how ecological restoration can be encouraged as a complementary pursuit to economic renewal to meet dual ecological and social needs (Souza et al., 2016).

(b) *Cases where the project commences with reintroductions with regeneration expected as a later phase*

This mode is best applied in fully cleared sites when and where it is apparent that in-situ regeneration potential is very low or non-existent, but where nearby colonisation potential is potentially very high and can be substantially enhanced by early reintroductions.

A highly successful example of a 'reintroduction first' combined approach is the Framework Species Method as applied in tropical and subtropical rainforest restoration where high proportions of species are naturally dispersed by flying frugivores (Goosem & Tucker, 1995, 2013; Elliott et al., 2003; Elliott & Kuaraksa, 2008). This method involves planting of short-lived early phase species to attract (and thereby facilitate) natural dispersal of less common, longer-lived species from nearby forests. Twelve case studies where the Framework Species Method have been applied

[2]Another motivation for reintroduction in a broader 'ecosystem restoration' (sensu FAO et al., 2021) rather than 'ecological restoration' approach (sensu Gann et al., 2019) can be to increase growth of timber and non-timber species that support local livelihoods and provide economic benefits for local people (Del Amo & Ramos, 1992; Paquette et al., 2009; Ricker et al., 1999; Souza et al., 2016).

in Asia, Latin America, and Madagascar are reviewed by Elliott et al., 2023. Chapter 3 of this book presents three case studies that reintroduce three species, 30–40 species, and 100 species, respectively, with results showing more species colonising with higher numbers of species initially planted higher numbers of (Tucker et al., 2023).

For example, Case Study 7 (Box 7.8) illustrates how – starting four decades ago – a small number of early successional trees were planted adjacent to the small subtropical rainforest remnant at Victoria Park Nature Reserve in northern New South Wales, Australia, and fostered colonisation by scores of later phase species from the remnant over time. This methodology is now used commonly to facilitate regeneration around remnants in that region and beyond. At Donaghy's Corridor in the adjacent state of Queensland, Australia, where the Framework Species Method was initiated, only two of 1300 short-lived pioneer bleeding heart (*Homalanthus novoguineensis*) seedlings planted in the tropical rainforest corridor remained after 20 years, but in their place were 153 other recruiting species, increasingly represented by mature phase plants (see Tucker et al., 2023).

In some cases, regeneration follows reintroduction not necessarily as an intentional activity but as a side effect of reintroductions. There are many examples where a range of desired plant species were sown or planted, but regeneration also occurred spontaneously or was facilitated by the removal of undesired, often alien species in degraded woodlands, grasslands, and various brownfields or abandoned arable land (Chazdon et al., 2021). A European example of this process is recorded by Prach et al. (2014), who provide a summary of landscape-scale restoration of approximately 600 ha of species-rich dry grasslands on former arable land where about one-third of target species were sown, one-third naturally regenerated, and one-third was still missing after 20 years since the initial sowing. Similarly, work carried out in Kalimantan, Indonesia (Case Study 5, Box 7.6), found that the exclusion of fire over a larger area not only protected the plantings but also facilitated natural regeneration. This showed that the increased effort to prepare for or protect reintroduced species can provide opportunities to remove regeneration barriers for other species in the planted zone.

Box 7.8: Case Study 7. Perches Facilitating Subtropical Rainforest Remnant Expansion, Big Scrub, NSW, Australia

By the 1970s, the former 75,000 ha 'Big Scrub' rainforest located on the north coast of NSW was reduced to just 1% of its former range. The remaining 800 ha were fragmented into many small and unviable remnants, now federally listed as critically endangered. At that time restoration of the rainforest became an increasing aspiration for communities and agencies, but practical options for revegetating such an immense area with high-quality outcomes was a daunting prospect.

In trial of potential for harnessing natural processes for recovery, plantings of a small number (>15) of early successional rainforest tree species were established in previously cleared and grazed paddocks around one of the

Box 7.8 (continued)

small remnants, Victoria Park Nature Reserve, in the late 1970s, continuing into the 1980s. Around the same time, six mature non-native camphor laurel (*Cinnamomum camphora*) trees at the site were poisoned and left standing. The purpose of the work was to (a) test potential for the plantings and dead trees to act as perches to attract dispersal of native rainforest trees from the adjacent rainforest remnant and (b) identify any missing species requiring supplementary planting.

Outcomes to date. By 15 years after planting, while only seven of the originally planted species persisted, a total of 68 tree and shrub species were recorded as having naturally recruited in 36 × 25m^2 quadrats at the site. This represented 72% of the 94 tree and shrub species occurring in the remnant. Of this total 45 were later successional species, 56 were frugivore-dispersed, and 8 were wind-dispersed. Recruitment of later successional (including mature

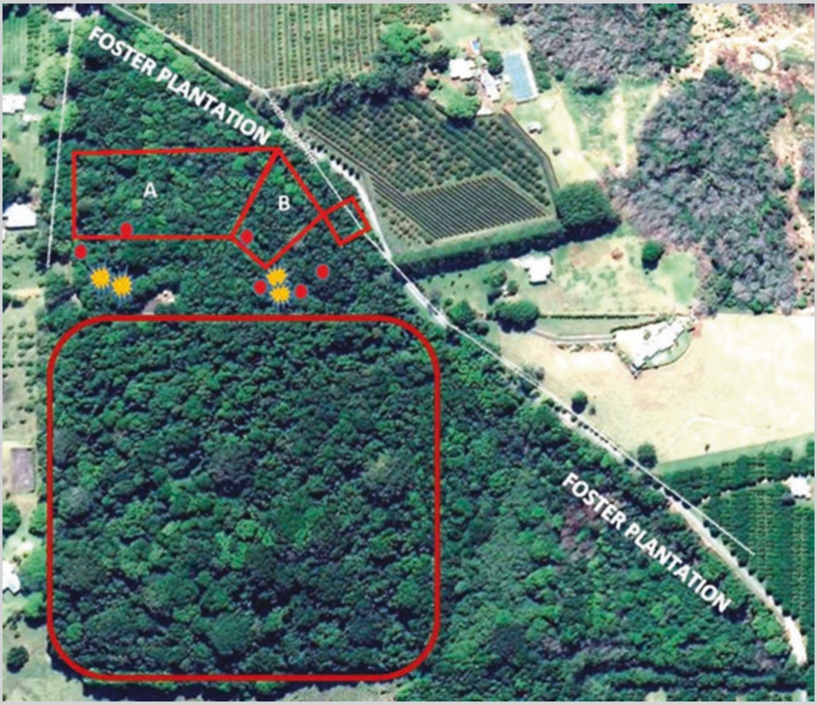

Fig. 7.11 The Victoria Park rainforest remnant (centre square) is one of less than 100 fragments remaining of the former 75,000 ha 'Big Scrub' subtropical rainforest. Plantings of a small number of early successional native species (plus poisoning of weed trees) provided perches that accelerated natural dispersal and recruitment of over 70% of the adjacent remnant's tree species within 15 years, with full recovery depending on reintroduction of the dispersal-limited missing species. (Diagram. Tein McDonald)

Box 7.8 (continued)

phase) tree species continued in the plantation over time (Fig. 7.11) but was substantially higher under tall, isolated native trees and poisoned camphor laurel. Species richness appeared to stabilise in the more successful sites by 24 years, although height and girth of specimens continue to increase. A nucleating pattern was strongly evident, with the nuclei expanding over time and ultimately coalescing (Fig. 7.12). A number of missing species including the ecosystem's dominant, white booyong *Heritiera trifoliolata*) were identified as dispersal -limited and requiring supplementary reintroduction.

The success of camphor laurel poisoning and the early successional plantings such as the one at Victoria Park – along with results of mature phase plantings elsewhere – has influenced restoration practice in the Big Scrub landscape. Two mixed-species planting models are promoted by Big Scrub Landcare: one with higher levels of early-phase species for sites closer to remnants and the other with higher levels of later-phase species at longer distances from such seed sources (Kooyman, 1996; Big Scrub Landcare, 2019). Planting

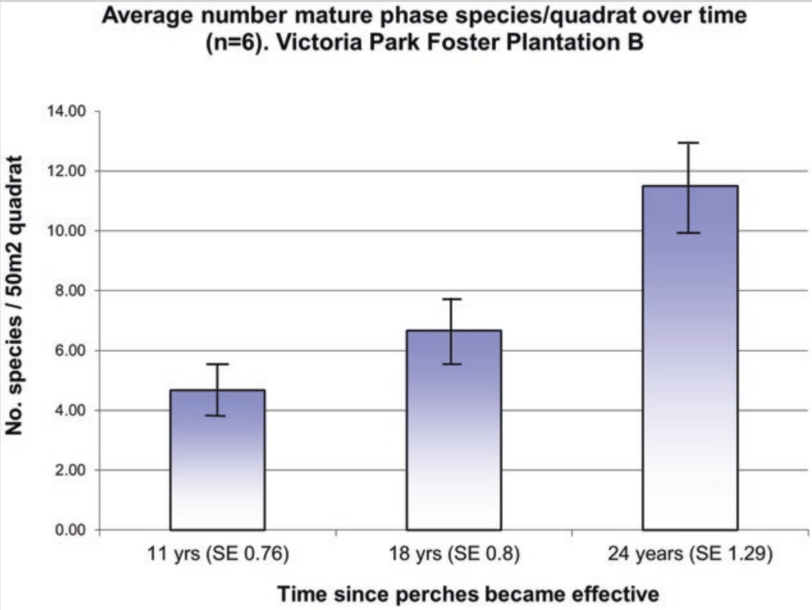

Fig. 7.12 Monitoring in the early successional plantings adjacent to Victoria Park rainforest remnant over 24 years found gradual increases in later successional species colonising and moving up into the higher height strata over time. By 15 years, 72% of the species had colonised the remnant, two-thirds of which were later successional (late secondary or mature phase) species

Box 7.8 (continued)
of hundreds of hectares of rainforest has now been successfully carried out by landholders across the former Big Scrub landscape (Parkes et al., 2012) and several hundred hectares of camphor laurel regrowth has been converted to recovering rainforest by poisoning weed species alone. Substantial work is ongoing into developing genetically diverse seed orchards for less well-represented later-phase species for future planting out across the landscape including into early successional regrowth stands (Parkes & McDonald, 2021).

Examples of an Approach that Relies Fully on Reintroduction

Reassembly of the vegetation of an ecosystem by means of reintroduction only (also referred to as 'reconstruction' Gann et al., 2019) is required when natural regeneration potential, either in-situ or migratory, has been entirely depleted for all but some ruderal species. In other words, a comprehensive reintroduction approach aims to substantially rebuild an ecosystem from an extremely low starting point. This approach is typically undertaken on long-cropped agricultural sites, decommissioned roads, or waste disposal areas, as well as on mine sites where all plant biota and subsets of soil microbiota require reinstatement, although in many cases, the aspiration is far lower than reintroducing all plant species of the community due to knowledge or resource limitations.

This approach, when applied to genuine ecological restoration rather than simply tree planting, is necessarily more challenging than regeneration approaches for a range of reasons. First, it can take more effort to identify the appropriate reference ecosystems where few if any plant species remain on the restoration site. This means that more intensive examination of the site's physical attributes, the surrounding landscape, historical records, and predicted climates is needed (Standards Reference Group SERA, 2021). Second, comprehensive reintroduction is all the harder in scenarios more distant from extant ecosystems where organisms needed for pollination, dispersal, and decomposition are less readily available. A third challenge is that availability of nursery stock of the desired species can be limited by seed supply and by difficulties propagating 'recalcitrant' species whose germination cues are unknown (Grant & Koch, 2007; Chazdon et al., 2021). These three constraints have, nonetheless, been overcome at a range of sites globally, including in the Gondwana Link Corridor in south-west Western Australia, where multiple reference communities are carefully identified prior to direct seeding of a high diversity of species (see Jonson, 2016). The additional effort undertaken to overcome these difficulties of reconstruction at such sites has resulted in substantial levels of reinstatement of a range of ecosystem functions.

A fourth challenge that is often present in very high degradation cases is that substantial knowledge and effort may be needed to reinstate the physical conditions at a site if degradation has pushed these outside the appropriate range for the desired ecosystem. That is, where impacts have caused a site to be extremely nutrient poor, excessively nutrient enriched, toxic, or hydrologically altered in contrast to their normal prior conditions, substantial intervention will be needed to return substrate and/or hydrological conditions to acceptable thresholds (Walker & del Moral, 2009; MacPhee & Wilks, 2013).[3]

A fifth challenge is to rebuild a resilient vegetation community able to reproduce and recover after disturbances. The task of reaching and testing this goal is more difficult in full reintroduction cases than regeneration cases where that capacity is already demonstrated during the restoration process itself. In full reintroduction cases, it is highly important that (i) the genetics of the reintroduced biota are of sufficient diversity and appropriate integrity to allow reproduction under current conditions and allow adaptation under future conditions (Prober et al., 2015) and that (ii) niches and conditions for recruitment are provided by the return of appropriate disturbances, including occurrences of natural mortality.

Practitioners of a full reintroduction approach sometimes seek to harness the property of resilience to help build their communities as soon as possible after the initial reintroductions, thus building and testing this property simultaneously. This is most practicable in ecosystems where the plant species have short life cycles and can reproduce and recruit rapidly, thus avoiding the need to entirely plant out or seed the site.

Case Study 8 (Box 7.9), for example, is a freshwater wetland reconstruction project in retired agricultural land in north central Victoria, Australia, that uses manipulation of available resources and ecological disturbances to take advantage of the resilient properties of the plant species. The project used irrigation infrastructure to schedule wetting and drying events in imitation of natural cycles to prompt a suite of planted and hand-broadcast species to flower and fruit as early as possible. This allowed the seed from the reintroduced species to recruit widely during the restoration phase itself. The selection, contract growing, and establishment methods for the species used in this project depended on a high level of restoration experience and ecological understanding, particularly with respect to the adaptive responses of the component species to natural disturbances.

While this case study has the unique advantage of a naturally highly dynamic and resilient ecosystem, it illustrates how techniques developed to enhance regeneration in a facilitated regeneration approach (such as management of competition and application of germination triggers) can also be applied when reintroduced species start to fruit and disperse their own propagules. Such techniques are likely to be important to accelerate the establishment of appropriate densities of desirable species and reduce densities of undesirable species.

A resilience-informed approach to rebuilding an ecosystem based almost entirely on reintroduction also needs to take into consideration not only species composition

[3] It is noted that return of these conditions may also result in subsequent natural colonization processes (Chazdon et al., 2021; Wilks & MacPhee, 2016).

but any need to sequence the introduction of some species ahead of others (Temperton et al., 2013). Any such need can often mean that reintroductions are not carried out in one pass. For ecosystems such as rainforests that exhibit distinct sequences of succession, for example, earlier phase species may need to be reintroduced ahead of (or at least at the same time as) later phase species to help ameliorate site conditions for the latter. Similarly, there may be a need to first establish healthy populations of disturbance-responsive shorter-lived species to allow them to build soil seed banks before they are overshadowed by longer-lived, higher-stature species (Grant & Koch, 2007). This staged approach can be particularly important where herbaceous species are a key target for establishment prior to later reintroduction of woody species (Gibson-Roy et al., 2010). In cases where the mature phase species would characteristically return immediately after natural disturbances or can tolerate the level of exposure involved, single pass operations where all species are introduced at once may be more desirable and more efficient (Rokich, 2016).

Box 7.9: Case Study 8. Muringa Wetlands – Reconstructing Habitat for the Australasian bittern *(Botaurus poiciloptilus)* and growling grass frog *(Litoria reniformis)*

The Muringa wetlands are reconstructed systems that cover 1.5 and 0.5 ha, respectively, and are located on the agricultural property 'Muringa' within the Lower Loddon catchment in north-central Victoria, Australia. The purpose of these relatively small wetlands is to provide breeding habitat for the endangered Australasian bittern (*Botaurus poiciloptilus*) (bitterns) and growling grass frog (*Litoria reniformis*). The Muringa wetlands form a complex with another 70 ha of wetlands on the immediately adjacent conservation property 'Wirra Lo', previously an irrigated dairy farm that is naturally regenerating after the removal of livestock and the reinstatement of wetting and drying regimes. Wirra-lo is known to support a resident population of growling grass frog, and bitterns have been recorded calling there.

The sites were previously 'borrow pits' from which clay was extracted to construct levees during the agricultural irrigation phase. These were reshaped at the start of the restoration phase to provide a diversity of habitats (Fig. 7.13). The ends of the wetlands closest to the irrigation pipe inlets were excavated to a depth of approx. 1.2 m to support dense tall reeds suitable for bittern nesting and to provide deep refuge pools to hold water throughout the region's hot and often dry summers to support the growling grass frog. The remainder of the wetlands was shaped to create shallow wetland meadow environments suitable for bittern foraging (Fig. 7.13).

Water is delivered to the wetlands through the previous farm irrigation infrastructure, timed to meet the ecological needs of the wetlands and the threatened species for which they have been designed. After a short initial wetting in late winter 2021 and planting in spring (November 2021), the site

Box 7.9 (continued)

was allowed to dry to allow good root development of the sedges and help the lower growing herbs to optimally flower, fruit, and set seed and germinate. Reflooding was carried out in May 2022 after a brief drying phase over summer 2021–2022 to enhance development and expansion of the wetland plants.

In November 2021 (spring), the wetlands were planted and seeded with around 4000 plants, representing 50 plant species characteristic of episodically inundated wetlands of the region, including some rare and threatened species to provide seed sources for their increased distribution to other wetlands by waterbirds. Focus has been placed on species that would not naturally arrive at the site due to isolation from seed or vegetative sources.

The deeper ends of the wetlands were planted with tall reeds, rushes, and sedges including broad-leaf cumbungi (*Typha orientalis*), tall spike-sedge (*Eleocharis sphacelata*), marsh club-sedge (*Bolboschoneus medianus*), and giant rush (*Juncus ingens*) to provide suitable nesting habitat for bitterns. The broad-leaf cumbungi was planted at a relatively low density of one plant per square metre as it can expand across at least 1 m in a year. The more extensive shallow areas of the basins were planted with low growing forbs, grasses, and sedges. Of these, nardoo (*Marselia drummondii*), water ribbons (C*ycnogeton*

Fig. 7.13 Looking from the shallower end of the wetland towards the deeper end at 8 months after planting when reflooding had commenced. Wetland meadow species are establishing well in the shallow areas to provide foraging areas for the bittern and taller growing reeds suitable for bittern nesting have established at the deeper (far) end of the wetland. Deeper refuge ponds have been created for the growling grass frog. (Photo Damien Cook)

Box 7.9 (continued)

Fig. 7.14 Temporarily netted wetland meadow species showing strong establishment and expansion of nardoo and water ribbons within 8 months after planting. Both these species are not only important forage plants for waterbirds but are also important food plants for local Indigenous people who are closely involved in guiding the project. (Photo Damien Cook)

multifructum), red pondweed (*Potomogeton cheesemanii*), and common spike-sedge (*Eleocharis acuta*) were planted at four plants per square meter within framed netting to deter browsing by wetland birds during the plant establishment phase (Fig. 7.14).

Other forbs, grasses, and sedges were planted in smaller numbers (with some seed hand broadcast) throughout the shallow meadow zone or at the high-water mark. Amongst these were the small red milfoil (*Myriophyllum verrucosum*) and the robust milfoil (*Myriophyllum papillosum*). Locally rare or declining species have also been planted including small monkey-flower (*Elacholoma prostrata*), starfruit (*Damasonium minus*), burr daisy (*Calotis scapigera*), and grey raspwort (*Haloragis glauca f. glauca*). Moira grass (*Pseudoraphus spinescens*), a very important wetland grass that has experienced massive decline in the floodplain of the Murray River, has also been planted.

Within 6 months from the date of planting, the vegetation had expanded vegetatively or produced seed. The taller reeds had consolidated to a high level of cover. nardoo had covered the entire plots in which they were planted and have spread beyond those plots. Water ribbons planted as tubers had

Box 7.9 (continued)
produced flowering spikes, and the hand-broadcast water ribbons seed had germinated, with tubers already forming under the mud surface. All the lower stature forbs and sedges had flourished and produced seed. In addition, a small amount of natural regeneration occurred for a range of species that have not been planted, including variable flat-sedge (*Cyperus difformis*), tall flat sedge (*Cyperus exaltatus*), hairy carpet weed (*Glinus lotoides*), and tangled lignum (*Duma florulenta*). Other species are likely to colonise from upstream channels and be dispersed into the wetland over time.

Chapter Synthesis

Why Have Mainstreaming Resilience-Based Approaches in Restoration Taken So Long?

This chapter has assembled literature and case studies to illustrate the proposition that benefit can be gained from using understanding of a site's levels of resilience to guide restoration. Insights into ecosystem resilience emerged in the field of ecology in the 1970s and has attracted increasing interest from researchers and practitioners over the past five decades (Holling, 1973; Westman, 1978; Fox & Fox, 1986; Reice et al., 1990; Lugo et al., 2002; Bengtsson et al., 2003). Harnessing resilience has increasingly proven to be a major plank of guidelines for tropical forest restoration over the past two decades (Chokkalingam et al., 2018; Crouzeilles et al., 2019; Shono et al., 2020). Yet reference to resilience assessment in generic restoration guidelines has remained surprisingly uncommon until relatively recently (McDonald et al., 2016; Gann et al., 2019; Standards Reference Group SERA, 2021).

While the slowness to include resilience-based concepts in generic restoration guidelines is puzzling, it may be explained by a combination of factors (Chazdon et al., 2021; McDonald, 2021). One factor, for example, may be the strength of the common assumption that problems caused by humans are best solved by human-based solutions rather than nature-based solutions, influenced by the predominance of agricultural, silvicultural, horticultural, and aquacultural traditions in the West. Another contributing factor is the mistaken view that natural regeneration potential, if spontaneous, cannot be considered as truly restoration, since restoration is necessarily an *activity* undertaken by humans rather than a deliberate choice to refrain from acting (Chazdon et al., 2021). A further factor is likely to be the conflation of the concepts of resilience and succession amongst academics and teachers, which may have led to the rejection of both concepts by some ecologists who are concerned that succession necessarily implies sequential or seral stages (which is not the case in all ecosystems) (McDonald, 2021). In addition, the earlier interpretation of the term ecosystem resilience as describing the degree, manner, and pace of recovery of ecosystems after disturbances (Westman, 1978) has been somewhat overshadowed by the more recent use (and popularization) of the term to describe the persistence of complex social–ecological systems in a changing world (Walker & Salt, 2006), whether due to resistance, recovery from or adaptation to a change. The most important contributing factor, however, may simply be that recovery capacity and its indicators are not easily observable and, as a consequence, not well understood or studied. Buried seed banks, bud banks, and distant seed sources are, by definition, not usually visible. These constraints to resilience analysis can also mean that the scale and heterogeneity of surrounding landscape patches that supply propagules for restoration areas are also poorly identified for many ecosystems. Anticipating recovery from such elements therefore depends upon greater sharing of experience and increased research to build a more comprehensive body of knowledge for restoration planning, implementation, and monitoring.

Management constraints also pose challenges. A sense of urgency to restore can also cause some to prefer accelerated approaches rather than waiting for natural regeneration, despite the potential for large areas to recover over time by regeneration approaches in some parts of the world (Zahawi et al., 2014). There is also often a lack of recognition and support of natural regeneration by government agencies responsible for managing native ecosystems, and there can be a lack of clarity on how to monitor outcomes of restoration through natural regeneration (without recourse to the usual practice of counting the number of tree seedlings planted and their survival rate). Slowly recovering naturally regenerating areas can also tend to be viewed by local communities as unproductive wastelands without active management. These areas are then more likely to face the risk of being cleared and replaced by other land uses (Zahawi et al., 2014; Chokkalingam et al., 2018).

While these factors might explain why a resilience-based framework for restoration has been slow to develop, we consider that the time is ripe for greater dissemination and uptake of such a framework (Box 7.10).

Box 7.10: Assessing Degree of Resilience

For a restoration project to optimally prioritise recovery, a detailed site assessment (usually undertaken at the 'inventory' stage of restoration planning) must be undertaken (SER, 2004; Holl & Aide, 2011; Chazdon et al., 2021). Identifying whether soil seed banks may be present or colonisation potential exists needs to be informed by an understanding of the recovery mechanisms of the site's individual component species that have evolved over evolutionary timeframes through adaptation to prevailing environmental resource limitations, stresses, and disturbances. For plants, this will particularly involve understanding whether they can resprout and/or store seed in the soil or colonise readily. For animals and other biological groups, it will similarly involve understanding of reproduction cycles, recovery mechanism, refugia, and colonisation potential.

Box 7.10 (continued)

Appropriately matching restoration approaches to sites or zones with differing potential and limits of recovery will also require knowledge of the barriers or filters to natural regeneration and the degree to which restoration techniques can overcome such barriers to harness any resilience that may be present (McDonald, 2000a, b; Polster, 2017; Chazdon et al., 2021). It is very easy to over- or under-estimate a site's resilience without careful identification of which approaches and treatments are going to provide the most appropriate outcome given the available resources. This means that effective site assessment requires restoration experience and knowledge. Site assessors should therefore strive to gain experience in interpreting indicators of potential or limits to natural or facilitated regeneration within the particular ecological community – experience derived from direct restoration observations at a wide range of sites. If this experience is not present, project managers need to be prepared to test techniques before commitment of substantial resources. Irrespective of degree of site assessment experience, continued and regular monitoring will allow project managers to adapt to unexpected biological responses to interventions that will inevitably occur and cannot be entirely planned for.

An important additional challenge for site assessment and treatment prescription is presented by emerging climate change, a challenge that cannot be underestimated and is largely unchartered at present. Assessors need to consider likely or potential adaptation and reassembly of biota as species naturally shift (or are predicted to shift) as temperature and moisture gradients change within a site or landscape (Standards Reference Group SERA, 2021).

Implications for Planning and Prioritisation

Social benefits and economic opportunities must play an important role in landscape planning (see Chaps. 14 and 15). However, the examples of resilience-based restoration approaches provided in this chapter show that knowledge about (i) resilience resources within landscapes and (ii) which restoration approaches are most appropriate for harnessing this resilience is also key to the planning process (Curran et al., 2012).

Spatial Implications

Conservation planning identifies priority locations for formal conservation based on the location and relative configuration of intact sites within a landscape, sites which provide the biological reservoirs that store local and regional biodiversity

(Conservation Measures Partnership, 2013). Restoration planning can be informed by similar principles, given that 'intactness' is a predictor of ecological resilience, and gradients of condition (resilience) across a landscape, occur in converse relationship to gradients of impact (McDonald, 2000b; Standards Reference Group SERA, 2021).

Indeed, the configuration (location, separation distance, or connectedness) of resilience resources in the landscape (individual plants, vegetation patches, etc.), are a key to restoration planning (Thackway & Lesslie, 2006; Davidson et al., 2011; Chazdon, 2017). These are sometimes referred to as 'ecological memory' (sensu Bengtsson et al., 2003; Sun et al., 2013). Relative location of a source sites to a receiving site has a very high influence on the rate and nature of colonisation (White et al., 2004; Graham et al., 2016; Blackham et al., 2014), while connectedness at larger scales is a primary factor in influencing recovery potential, particularly of fauna (Hanski, 1999; Tucker & Simmons, 2009).

An example of restoration harnessing spatial resilience is the strategy of Applied Nucleation which establishes rainforest 'starters' in strategic locations across a tropical forest landscape to more effectively distribute potential for sequential recovery over space and time (Holl et al., 2020; Wilson et al., 2021). This can equally apply to establishing seed source nuclei or nuclei that receive seed (Parkes & McDonald, 2021). Nucleating recovery patterns can be used to good effect in any ecosystem type (Yarranton & Morrison, 1974) and strategic focus on nuclei and their immediate surrounds can result in resilient 'islands' expanding and ultimately coalescecing where staged restoration is undertaken for resource limitation reasons (McDonald, 2000b; Vergés et al., 2020, and see Box 7.9).

Temporal Implications

This chapter's case studies show that trialing regeneration approaches prior to reintroduction makes sense where there is any potential for extant natives (or persistent propagules) to contribute to the recovery at a site (Fig. 7.1). Even where not all species recover (or where recovery is spatially patchy) such a preliminary treatment allows the identification of zones of weaker recovery, missing species, and inadequate genetic representation and can also allow stronger evidence of the pre-existing ecological community if other evidence is lacking.

Although rarely tested experimentally, it is logical to assume that harnessing extant resilience in a restoration project by deploying regeneration-based approaches may return a site to the identified recovery trajectory earlier than where restoration depends upon having to reassemble an ecosystem from scratch. This has implications for prioritisation where ecological outcomes are urgent (as distinct from situations where social goals may be more urgent). That is, whether at a landscape or site scale, ecological advantage is likely to be gained if the treatment of more resilient areas occurs prior to less resilient areas, particularly where higher condition sites are capable of subsequent expansion into the lower condition sites. There are circumstances, however, when a comprehensive reintroduction approach may be preferred from the start. These include circumstances where a fully depleted site provides important connectivity between two habitats, particularly under circumstances where (i) habitat increases are time-sensitive, (ii) where loss of genetic diversity is identified, or (iii) where localised plant extinctions are occurring (Tucker & Simmons 2009).

Conclusion

The proposal put forward in this chapter is that restoration is likely to be more successful if interventions are matched to the degree a site either retains or has lost its resilience, that is, its lever of remaining natural regeneration capacity. The case study literature referred to, together with the eight original case studies presented here, show that where regeneration capacity persists, efficiencies and often greater species diversity and fidelity may be gained by harnessing that capacity (using a range of methods). Efforts need to be made to consider hidden or stalled regeneration capacity and to consider ways to facilitate it prior to assuming reintroductions are needed. Where and when it is clear this capacity is depleted, however, or where critical habitat is needed to enhance high-quality recovery in landscapes, a resilience-based view can help in the design of assemblages that will develop their own capacity to reproduce and self-perpetuate as early as possible, providing a basis for ongoing evolution and adaptation (Ghazoul & Chazdon, 2017).

A resilience-based view of restoration is highly important under any circumstances, but this is particularly important at a time when the global imperative for restoration has reached a critical point (UNEP & FAO, 2020). The challenge of the United Nations Decade on Ecosystem Restoration, for example, requires us to pay greater attention to any opportunities to harness natural recovery. Not only is the area of restoration required across the globe far too large to depend upon reintroductions alone, but it would be entirely inappropriate to invest precious propagation resources where they are not needed. Restoration is more difficult and usually far more costly when no ‘head start’ is provided by extant natural recovery potential. This suggests that, where ecological goals are foremost, benefit can be gained from giving priority to sites where there is potential to harness natural processes of recovery, even if that needs to be undertaken simultaneously with partial reintroduction where it is proven to be required. Such prioritisation is likely to offer higher certainty for restoring the highest area possible for the lowest input and cost. A higher or sole focus on reintroduction, however, comes into its own (and needs to be properly resourced) where natural regeneration cannot be rapidly facilitated and a critical habitat link, stabilisation, buffer, or amenity is rapidly required. Such a focus on reintroduction should be designed in a manner to rebuild resilience that can become activated as soon as possible and provide new seed sources for further natural or facilitated expansion.

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Chapter 8
Rare and Threatened Plant Conservation Translocations: Lessons Learned and Future Directions

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Summary
The conservation and recovery of rare and threatened plant species is often challenging due to the small number of propagules, plants, and populations available. Here we document eleven lessons that have come to light during our work in this field.

1. Select a suitable recipient site.
2. Use a large number of founding plants.
3. Use genetically diverse foundation plants.
4. Promote germination and survival, and provide aftercare.
5. Monitor over a sufficient time to establish and document evidence of population sustainability.
6. Use appropriate benchmarks to measure conservation translocation success.
7. Establish continual monitoring of plants for disease and for disease prevention.
8. Understand that establishing a population via conservation translocation is a long-term process.
9. Conduct the conservation translocation as an experiment.
10. Follow a set of clear guidelines for the conservation translocation and publish results to benefit reintroduction science.
11. Engage policymakers, practitioners, and the public in conservation translocation initiatives.

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Glossary

Assisted migration: An attempt to establish a species, for the purpose of conservation, outside its indigenous range in what is considered to provide appropriate habitat for the species based on climate change or habitat change predictions. Such translocations are potentially high-risk projects with success often difficult to predict, and should only be carried out after an extensive risk assessment has been conducted. Also known as assisted colonization.

Benchmarks of success: standards or reference points by which the translocation can be measured to understand whether it is successful or on the trajectory to success. For example, comparing reproductive output of a translocated population to one or more wild populations can help gauge whether the translocated population is self-sustaining.

Conservation translocation: the intentional movement of a target organism for its conservation benefit. Plant conservation translocations usually entail an intermediary phase with a botanic institution. For example, seeds collected from wild populations are propagated in a nursery and then are transplanted out to the wild site.

Genetic diversity: total genetic variation for a population, taxon or other taxonomic rank.

Propagules: units of vegetative reproduction (includes seed, spores, or vegetative matter capable of independent growth e.g. cutting material).

Rare plant species: For purposes of this chapter, a rare plant species is one with few individuals and few populations living in the wild. Usually these species are associated with narrow geographic range in restricted habitats.

Recipient site: The location where the translocation planting is to occur.

Source site: The location where propagation material (for example seeds) is collected from, to undertake a translocation.

Microsite: a term used in ecology to describe a pocket within an environment with unique features, conditions or characteristics. Classifying different microsites may depend on temperature, humidity, sunlight, nutrient availability, soil physical characteristics, vegetation cover, etc.

Reintroduction: An attempt to establish a population of a taxon in a site or habitat type where it no longer occurs (locally extinct). This may be part of the process of restoration or reconstruction of a habitat where the species was previously known to occur. Also known as re-establishment.

Introduction: An attempt to establish a population in a site where it has not previously occurred but is within the known range of the species and provides similar habitat to known occurrences.

Introduction

The ultimate goal of rare and threatened plant conservation is to ensure that the rare taxon persists and evolves within a range of natural contexts (CPC, 2019). Rare and threatened plants are usually narrow-range endemics or near-endemics with a few populations, a few extant individuals, and hence have a high risk of extinction. The full suite of plant conservation actions from surveys and monitoring to ex situ collection and research necessarily should precede any efforts to restore rare species to natural habitats (CPC, 2019, Fig. 8.1). Conservation translocation, or the intentional movement of a target organism for its conservation benefit, is an essential step, but not the first step (IUCN/SSC, 2013; Fenu et al., 2019; CPC, 2019). Because of increasing threats to biodiversity from disease, habitat loss, land alteration, habitat fragmentation, invasive weeds, altered fire regimes, grazing, altered hydro-ecology, and climate change, we should expect to see conservation translocations increasingly used to improve the probability of the survival of rare plant species (Zimmer et al., 2019). However, because translocation can be a complex and expensive undertaking that requires long-term commitment, practitioners around the world are still investigating the practical approaches and strategies to increase the efficacy of this conservation tool (Albrecht et al., 2019; Fenu et al., 2019; Silcock et al., 2019; Zimmer et al., 2019). Disturbingly, the general findings of these worldwide reviews are that very few translocation attempts have resulted in evidentially documented populations that are capable of recruiting to the next generation. This low probability is likely to have arisen from a combination of interacting factors, including the need for species to have sufficient time and supportive environments to mature and reproduce successfully (Albrecht et al., 2011). Consequently, both biological and sociological barriers challenge the success of translocation. To reverse the current decline in biodiversity and preserve rare taxa, we must understand the nature of these challenges and develop conservation actions to enhance recovery of endangered species and preserve their habitats (Monks et al., 2019).

We believe that if we are to improve the practice of conservation translocation, there is a clear need to learn from past successes and failures (Albrecht et al., 2019; Berger-Tal et al., 2020), allowing conservation managers to introduce proven approaches to maximize the success of future programs (Ames et al., 2020). In this chapter, we (i) discuss some of the lessons learned from past conservation translocations (Fig. 8.1; Table 8.1), (ii) provide Case Studies illustrating the breadth of complexity of the practice of conservation translocation, and (iii) suggest how we may profitably change our practices in the future using the lessons learned.

Lessons Learned

Select a Suitable Recipient Site

Because most plants are literally rooted in the ground for most of their life cycle, the initial selection of appropriate sites and microsites is critically important for successful conservation translocation (Maschinski, Falk, et al., 2012a, Maschinski and Albrecht, 2023). Indeed, the majority of reviews of conservation translocation outcomes indicate that poor site and microhabitat selection is a major reason for failure (Silcock et al., 2019). Conservation translocations that have been rushed with little planning have low chances of success and waste scarce valuable resources and

BEFORE YOU BEGIN

READ GUIDELINES
- Follow steps outlined.

JUSTIFY THE NEED
- Evaluate whether conservation translocation is appropriate as an alternative to habitat management and threat abatement.

LOGISTICS
- Make a plan. Ensure clear goals & objectives.
- Develop experimental design.
- Review and follow laws.
- Collaborate with landowners and land managers.
- Ensure availability of resources and funding.

KNOW THE SPECIES
- Gather information about the species' biology, life history, ecology, habitat preferences & distribution.
- Consider genetics of existing and source populations.
- Assess a suitable recipient site.
- Evaluate threats.
- Consider future climate. Consider collaboration with a modeler.
- Consider mutualists and the habitat needs required for species' establishment success.
- Select and match source material to site.

IMPLEMENTATION

PREPARE THE SITE
- Weed, thin canopy, introduce prescribed fire, or deep rip, if necessary
- Plan for population growth.

GATHER and PREPARE THE PLANTS
- Collect propagation material or retrieve from seed bank.
- Allow sufficient time for plants to grow to appropriate size for transplanting.
- Label plants for long-term tracking.
- Begin with large numbers of genetically diverse, healthy founders.
- Plant in pattern and microsite conducive to good growth and pollination.

LOGISTICS
- Notify and involve land manager(s).
- Choose best season for transplanting or seeding.
- Organize and bring all necessary materials and equipment to the site.
- Enlist enough people to help prepare and install the plants.
- Bring snacks and water.

AFTERWARDS

AFTERCARE
- Water, weed, and protect plants from herbivores and vandalism to promote germination and survival.
- Treat disease and pests if necessary.

MONITORING PLAN
- Manage the long-term commitment required to evaluate population establishment.
- Monitor on regular basis, taking measurements according to the plan and the life history of the species.
- Analyze and report data.
- If necessary, devise new hypothesis, new plan, and repeat implementation and monitoring steps accordingly.
- Document activities.
- Publish results.
- Share successes with policy makers. Practitioners and the public.

Fig. 8.1 Overview of steps that should be undertaken before, during, and after a conservation translocation

Table 8.1 Lessons learned from worldwide reviews of plant conservation translocations

1.	Select a suitable site.
2.	Use large number of founders. If necessary, introduce propagules multiple times to get a population to establish.
3.	Use genetically diverse founders.
4.	Take measures to promote seed germination, seedling, and whole plant survival.
5.	Monitor over a long enough time to document sustainability.
6.	Use appropriate benchmark to measure success. Don't expect species with varying life histories to all behave the same, establish the same, or take a short time period to establish sustainable populations.
7.	Watch out for disease.
8.	Anticipate future change. Don't just plant and walk away. Continue to maintain the site. Control invasive species or your target plants will be toast.
9.	Conduct your conservation translocation as an experiment. Use an adaptive management approach.
10.	Follow guidelines and please publish your results for the benefit of reintroduction science around the world!

Godefroid et al. (2011), Dalrymple et al. (2012), Guerrant (2012), Albrecht et al. (2019), Fenu et al. (2019), Silcock et al. (2019), Diallo et al. (2021)

labour time (Falk et al., 1996). Consequently, it is recommended that informed planning and the use of guidelines built on previous experiences are needed to enhance the chances of improved conservation translocation outcomes (Commander et al., 2018, CPC, 2019).

In light of these reported findings, selecting recipient sites that have similar ecological conditions to locations with secure, healthy rare plant populations is prudent. In addition, noticing the presence of obvious threats to plant establishment, and avoiding sites with threats or controlling threats is advised. For example, a recipient site that has high spatial coverage of an invasive species should be either avoided or treated to remove the invasive species. In Western Australia, sites may be tested for presence of *Phytopthora* and avoided for restoration of rare taxa.

The lessons learned from previous works have indicated that the selection of suitable recipient sites may be accomplished by paying attention to several key factors, including:

(i) Assess environmental variables associated with sites where the target species has good growth, survival, and reproduction, and use a rank to compare potential recipient sites. For example, use the community composition of a site where the extant population is thriving and compare the same attribute across several recipient sites so that sites may be ranked (Maschinski, Falk, et al., 2012a).
(ii) Compare multiple environmental variables where the target species is present (occupied sites) and where it is absent (unoccupied sites) using multi-variate analysis such as principle components analysis (PCA) (Brzosko et al., 2018).
(iii) Use abiotic, biotic, and functional traits (e.g., specific leaf area, leaf dry matter content, leaf present nitrogen, leaf carbon to nitrogen ratio, and water use efficiency) of the recipient site community as bioindicators (Ames et al., 2020).
(iv) Assess the presence of appropriate pollinators (Reiter et al., 2017).
(v) Ensure there are appropriate microsites suitable for harbouring pollinators (Noe et al., 2019).
(vi) Establish and maintain a suitable soil microbial community necessary for plant growth and reproduction (Becknell et al., 2021).
(vii) Consider the future climate envelope (IUCN/SSC, 2013; Munt et al., 2016; Bellis et al., 2021).
(viii) Account for impending land use changes that may affect long-term site suitability.

When instituting a complex translocation program, it would be sensible to carry out preliminary assessments, with adequate sampling across the seasons to map the recipient site conditions fully. Assessments that may be needed include evaluating pollinator presence or sampling soils for laboratory analysis.

Because changing climate may alter conditions at a recipient site beyond the tolerances of the target species, consider collaborating with modelers to generate species distribution models (SDMs) in the planning process. SDMs provide predictions of sites with suitable environmental conditions under future conditions or the future climate envelope. For example, using SDMs, Bellis et al. (2021) assessed the suitability of four recipient sites in the United Kingdom and Ireland as stable conservation translocation sites for nine plant and four insect species. The potential recipient sites were suitable for nine species. One major caveat about SDMs is that these require good environmental data to generate sound predictions and sometimes these data are not available. It is also possible that the target species has extremely restricted habitats and is constrained by particular abiotic factors that have limited options in the future. In these cases, SDMs may not be helpful. Dalrymple et al. (2021) suggest that conservation translocations may be used as bioassays of the effects of global climate change, especially if measurements are paired with careful environmental assessments before installation and if plants are closely monitored following installation.

Use a Large Number of Founding Plants

Recent meta-analyses have confirmed that the strongest predictor of a successful conservation translocation is using a sufficient number of founders (Silcock et al., 2019). Fenu et al. (2019) found that 500 juvenile or adult plants increased the chances of success, while Albrecht et al. (2019) showed that using at least 50 plants as founders increased the chances of successful population establishment. It may be necessary, in some cases, to introduce propagules multiple times for a population to establish successfully (e.g., Duquesnel et al., 2017). Once established, these populations have documented evidence of next-generation recruitment within 20 years. When using seeds as founders, it will require thousands to tens of thousands for successful population establishment (Knight, 2012). Furthermore, because many conservation translocations require more than five years before recruitment occurs, confidently determining population establishment may require decades. This is an important consideration to be built into the planning and monitoring phases of a translocation.

Ex situ collections of plant species provide an important source of propagules (seed, plants, cuttings) for conservation translocations as seen in Case Study 1 based on *Banksia brownii* and Case Study 2 *Pilosocereus robinii*. During the planning phase of conservation translocations, building up plant and seed numbers in ex situ locations and in conservation seed bank holdings can help to ensure adequate numbers of founders are available for successful population establishment.

Use Genetically Diverse Foundation Plants

Genetically diverse founding populations have a greater chance of survival and adaptation than founders with low genetic diversity and they support the overarching goal of protecting biodiversity (Laikre et al., 2020). The seeds or plants used to initiate the conservation translocation should be (i) from a location with similar ecology to the recipient site, as these potentially hold similar adaptations, (ii) genetically diverse, and (iii) reproductively healthy (Basey et al., 2015; Commander et al., 2018; Godefroid et al., 2016; Hoban et al., 2018; Maschinski & Albrecht, 2017). Genetic factors have been shown to play a significant role in conservation translocation outcomes. Without careful consideration of source material and appropriate genetic management, conservation translocated populations may experience inbreeding or outbreeding depression, and they may generally lack adaptive potential and resilience to environmental change (Robichaux et al., 1997).

A clear understanding of the population genetic structure of the target taxon prior to conservation translocation can help to determine whether to use a single population or mix multiple populations as the founding population (Monks et al., 2021; Neale, 2012). Some reviews of rare plant conservation translocations indicated that short-term success was greater when using plants from mixed source populations rather than a single source (Dalrymple et al., 2012; Fant et al., 2013; Godefroid et al., 2011), whereas other studies found no advantage of mixing population sources (Liu et al., 2015).

Mixing source material can increase genetic diversity in the founder population and improve the chances of successful establishment; however, the ultimate genetic health of the conservation translocated population may be influenced by the effective population size of the source populations. Fant et al. (2013) found that mixed source conservation translocations had significantly higher inbreeding coefficients than the originating populations, possibly because of the small effective population sizes of the originating populations. It is noteworthy that census population size alone does not equate to genetic diversity and it may require genetic study to determine effective population size. Tracking the genetic health of the conservation translocation population is key to determining whether mixing sources truly conserves genetic diversity and consequently biodiversity. See Case study 3 *Arnica montana* (Van Rossum et al., 2020).

It is also worth noting that significantly higher inbreeding coefficients in admixed translocated populations compared with source populations can potentially be due to a Wahlund effect, driven by genetic sub-structuring (Fant et al., 2013; Monks et al., 2021). While basically an artefact of the admixture process, it is important to note that in artificially created populations these effects may be detected during the initial establishment phase but should not necessarily be viewed negatively in terms of long-term translocation success, as the genetic sub-structuring would be expected to disperse, and the high inbreeding coefficients decrease as plants from different source populations interbreed.

If genetic information is lacking, consider the nature of the breeding system when selecting source populations (Havens et al., 2015). For selfing species, using a few local populations that are best adapted to the translocation site is recommended to maximize evolutionary potential (Weeks et al., 2011). On the other hand, species that are self-incompatible or mixed mating but have high levels of outcrossing or long-distance gene flow such as wind-dispersed species, source populations can often be safely mixed for translocation (Weeks et al., 2011). For the rare outcrossing perennial herb *Centauria corymbosa*, mixing population sources and planting at high densities was required to ensure adequate numbers of compatible mates for seed production (Colas et al., 2008).

For species with little genetic variation or elevated levels of inbreeding, fitness can often be improved by bringing together genotypes from multiple populations to increase the genetic diversity. This is especially so if fragmentation has disrupted historical patterns of gene flow and resulted in genetic differentiation of the populations (Frankham, 2015; Maschinski et al., 2013). In these circumstances, applying the decision tree developed by Frankham et al. (2011) can help determine whether to mix source populations for translocation. Mixing multiple sources for small, inbred populations can infuse new genotypes and rescue inbred populations (Frankham, 2016). Understanding how a translocation may be able to rescue or improve genetic diversity is a fertile avenue for research. For more information on this issue, see Case Study 3 about *Arnica montana*.

Promote Germination and Survival and Provide Aftercare

According to reviews conducted in Australia, the United States, and Europe, most conservation translocations have been done with whole plants (Silcock et al., 2019; Guerrant, 2012; Dalrymple et al., 2012). In Australia, 9% of 859 conservation translocations used only seeds (Silcock et al., 2019), while 26% of 145 projects used only seeds in the United States (Guerrant, 2012), and 18% of 249 projects used only seeds in Europe (Dalrymple et al., 2012). In the review by Dalrymple and others (2012), of 47 conservation translocations that used seeds as founders, high numbers of propagules (5640 ± 2007) were introduced, yet only 5% survived in comparison to 249 projects that used juvenile or adult plants as founders, which had 65% and 99.8% survival respectively. Yet, 47% of the plants originating as seeds recruited next generation, in comparison to 5% and 21% of plants transplanted as juveniles or adults, respectively.

Seeds may be most appropriate for several conditions, including species with annual life histories, or species that live in very rocky habitats, or species that experience frost-heaving (See *Braya longii* Case Study 6). Alternatively, whole plants may be more appropriate as founding propagules for other conditions, including for long-lived species, for species that do not produce viable seed, and for species easily propagated from cuttings. Regardless of the propagule used, promoting germination of seeds, survival of the seedlings and whole plants, and providing aftercare is advised to help a new population become established (Commander et al., 2018; CPC, 2019).

Seeds have complex biology. Many have dormancy, which delays germination until seeds experience cues that release dormancy and allow germination (Baskin and Baskin 2014). To overcome dormancy pre-treating certain species' seeds by abrasion or fire may increase germination success in the field (Maschinski et al., 2018; Turner et al., 2013). To assure the survival of seedlings that emerge, practitioners may need to water seedlings until they establish.

In many cases, it may be necessary to provide supplemental water and protection from herbivores during the establishment phase (Commander et al., 2018; CPC, 2019). Ongoing site management for invasive removal, fire, or canopy thinning may be necessary to maintain conditions suitable to the target species' survival and establishment (Commander et al., 2018; CPC, 2019). Attention to factors such as these that promote rapid growth and reproduction during the early stages of translocation will often increase the chances of second-generation recruitment and long-term population persistence (Albrecht et al., 2019).

Monitor Over a Sufficient Time to Establish and Document Evidence of Population Sustainability

Meta-analyses of conservation translocation studies have clearly indicated that it may take decades to observe and document that a conservation translocation has been successfully established and produces seedlings that successfully survive and reproduce to establish subsequent generations (Albrecht et al., 2011). Instituting a careful, long-term monitoring protocol that is appropriate for the life history of the target species is essential (Maschinski et al., 2012b). An excellent example of this process is given by the monitoring of a reintroduction of the endangered species *Chloropyron maritimum* subsp. *maritimum* (salt marsh bird's beak). This work carried out in the United States over 26 years allowed Noe et al. (2019) to understand complex patterns of tidal range and oscillations confidently. The lunar nodal cycle, precipitation levels, and temperature variations influenced the observed 'boom' and 'bust' patterns of germination.

Use Appropriate Benchmarks to Measure Conservation Translocation Success

Conservation translocations may have projects and/or biological goals (Pavlik, 1996). Project goals may include community engagement, influencing conservation ordinances, or providing insights into ecosystem management techniques. Biological goals include quantifying the survival, growth, reproduction, next-generation recruitment, and population spatial expansion of the target species. These are also known as benchmarks of success (Albrecht et al., 2019). We recommend setting biological benchmarks that are appropriate to the target species' life history, considering the expected time to reproduction and production of next-generation offspring and adjusting monitoring intervals accordingly. While herbaceous species may be able to reproduce within three years or less, long-lived, slow-to-mature species may take decades to flower, and even longer to produce second-generation recruitment.

Examples of slow-to-mature species that require survival and growth as benchmarks of conservation translocation success in the early years following establishment include Sargent's cherry palm (*Pseudophoenix sargentii*), which first flowered 25 years following translocation (Possley et al., 2021), and *Cypripedium acaule,* which required eight to 10 years to flower (Hugron et al., 2020). Next-generation recruitment will require more than 2.5 decades for *Pseudophoenix sargentii,* whilst for *Cypripedium acaule*, a species prone to dormancy, will require more than 10 years. In contrast, the herbaceous hoary pea (*Tephrosia angustissima* var. *corallicola*) grown from cuttings and reintroduced to a pine rockland in South Florida, had flowers at the time of the installation and produced seedlings three months later (Wendelberger & Maschinski, 2016). The appropriate benchmarks, which were set for measuring the success of this reintroduction were survival, growth, flowering, and recruitment, with a monitoring frequency being monthly after the first seedlings had emerged.

Establish Continual Monitoring of Plants for Disease and for Disease Prevention

Worldwide, the spread of plant diseases is a significant threat to biodiversity (Corredor-Moreno & Saunders, 2020). For example, in Australia, two introduced pathogens, *Phytophthora* dieback (*Phytophthora cinnamomi*) and myrtle rust (*Puccinia psidii*) seriously threaten native flora and are very difficult to control (Broadhurst & Coates, 2017). An ocean away, on Hawai'i Island, the keystone forest tree species 'ohi'a lehua (*Metrodideros polymorpha*) has been dying across large areas due to two fungal pathogens (*Ceratocystis lukuohia* and *Ceratocystis huliohia*). These fungi cause Rapid 'Ōhi'a Death (ROD) disease (Fortini et al., 2019), and it is estimated that these large 'ōhi'a mortality events affect more than half the range of 63% of Hawai'i's endangered native plant habitats.

These lessons clearly suggest that conservation translocations must take preventive measures to minimize the impacts of disease. For example, in Western Australia, phytosanitary measures are mandatory in greenhouse production prior to seedlings being moved to the field with production nurseries requiring accreditation under the Nursery Industry Accreditation Scheme, Australia and adherence to the Australian nursery industry best management practice program (see https://nurseryproduction-fms.com.au/niasa-accreditation/). Of particular concern are specific pathogens that are already manifesting as known problems, and plants must be tested for pathogen infestation prior to conservation translocation to prevent failure and protect the environment in which the conservation translocation is proposed. Exogenous controls such as the pressure washing of vehicles before and after moving into sites known to have pathogens are essential, as is the fencing of the conservation translocation areas, including locked gates, to prevent casual entry of visitors not following sanitary protocols.

With regard to the *Phytophthora* example quoted above, in Australia, there are options available for the use of an appropriate fungicide, potassium phosphite (Barrett & Yates, 2015). However, chemical options may vary depending on local environmental regulations imposed at the conservation translocation site. In Hawai'i, excluding introduced feral ungulates with fencing has been found to reduce the prevalence of ROD disease, which is probably due to the reduction in physical damage to 'ōhi'a trees, a precondition for *Ceratocystis* infection. A committee on the islands has formulated an ROD Strategic Response Plan, which includes quarantine of the sensitive areas, together with 'do not transfer wood' protocols and statewide efforts to collect 'ōhi'a tree seeds across the archipelago (Walsh & Wolkis, 2021).

Understand That Establishing a Population via Conservation Translocation Is a Long-Term Process

In the preparation stage of a conservation translocation project (Fig. 8.1), it is important to anticipate future changes. It is not a real option to plant sensitive species, then walk away. Practitioners must continue to maintain the site in order to control invasive species and to maintain regular inspections of developing plants. Measuring how the target species is faring following planting or sowing, whilst documenting how important components in the ecosystem and abiotic elements change over time will help practitioners explain observed changes. A good monitoring plan incorporates notations of factors impacting the target species directly, such as its growth and survival, and may also include evidence of damage from herbivores, as well as soil moisture, temperature, precipitation, shading by competitors, and the presence of invasive species. Close collaboration with land managers is essential for implementing follow-up monitoring and management activities over a considerable time span, and in some locations, this will be a critical requirement for helping the target species population to establish. An example of long-term collaboration and ongoing intervention in a translocation site is Case Study 4, which recounts the regeneration of spiral fruited wattle (*Acacia cochlocarpa* subsp. *cochlocarpa*) using controlled fire regimes.

Conduct the Conservation Translocation as an Experiment

The difficulties evidenced to date with the low success of conservation translocation attempts suggest that it might be prudent for practitioners to focus on strengthening reintroduction science. When conservation translocations are conducted as experiments, practitioners formulate hypotheses during the planning stage, make careful observations, collect data, evaluate the findings and adjust. One way to accomplish this is to use an adaptive management approach, which follows a series of hypothesis-test-evaluate-adjust cycles (Allen et al., 2011; Holling, 1978). Alternatively, implementing conservation standards will help create a good road map and prepare the practitioner to expect results and future steps that may be needed to achieve a successfully established conservation translocation (CAML, 2021). The great need to document successes, justify expenditures, and increase the cost-effectiveness of restoration programs for the benefit of human well-being calls for practitioners to use experimental approaches and innovations that can encourage public and policy-maker support and improve outcomes (Prober et al., 2018). For more discussion, see Case Study 2 regarding the Key tree cactus. We are convinced that a well-designed experimental translocation can advance our understanding of how to conserve the target species, even in cases when the translocation initially has low success.

Follow a Set of Clear Guidelines for the Conservation Translocation, and Publish Results to Benefit Reintroduction Science

Despite nearly four decades of plant translocation practice, the science behind these activities is still in its infancy, and practitioners around the world are keenly seeking more information from managed processes (Abeli & Dixon, 2016). As we have previously indicated, instituting a conservation translocation is only one step in a process that should begin with a well-thought-out plan and involve long-term monitoring. However, it is important to realize that researchers, who are currently attempting to understand emerging patterns in reintroduction, comment that few studies have been published. Indeed, only 100 of the 1001 studies reviewed by Silcock et al. (2019) were published. A few of those reviewed by Dalrymple et al. (2012) were monitored for enough time to evaluate the success of the project confidently. We join our colleagues in encouraging all practitioners to (i) follow a set of well-planned guidelines, (ii) maintain sites and monitor for an adequate period, and (iii) publish (Commander et al., 2018, CPC, 2019, Note that E-versions are available https://www.anpc.asn.au/translocation/ and https://saveplants.org/best-practices/why-conserve-rare-plants/).

Engage Policymakers, Practitioners, and the Public in Conservation Translocation Initiatives

Fenu et al. (2019) note that scarce financial resources were an important factor limiting the implementation of conservation translocations. Because continued and repeated management of translocated populations is costly, adequate and ongoing funding is essential to achieve and maintain the successful establishment of conservation translocations (Fenu et al., 2019). If those involved with conservation translocation are to have continuing public support and be confident of the funding needed for monitoring, educating policymakers, funding agencies, and the general public is important (Broadhurst & Coates, 2017).

Conservation translocation requires long-term commitment and adequate long-term funding to support it. Outcomes therefore may not be clear in the short term, but only after decades. Recently developed training guides, available online through professional organizations (the Australian Network for Plant Conservation, the Center for Plant Conservation, and the Society of Ecological Restoration), seek to make expertise from professional conservationists available to a worldwide audience of the public, practitioners, policymakers, and potential financiers. These increased efforts to engage public attention and interest from policymakers are essential precursors for the development of significant biodiversity preservation programs.

Discussion

As a result of considering these lessons that we have learned, we have reflected on the difficult question of 'What do we think we will be doing differently in the next decade?' There are four statements that we would like to make in this respect, and we follow with some comments which we think will support these positions.

(i) *Conservation translocations will be made in areas predicted to have conditions that will be suitable for the species. Because of climate change effects, these areas may be substantially different from the habitat where a species currently occurs. Sometimes these locations may even be outside of the current range of the species*.

Determining a suitable future habitat is reliant on good-quality models generated from good-quality data, and we foresee new collaborative opportunities for data collection and sharing will be important. To illustrate the way in which this might occur, we note that models have been already developed to guide decisions about potential needs for assisted colonization in Italy. Casazza et al. (2021) generated models for endemic Italian plant taxa ranges under pessimistic and optimistic climate change scenarios. Simulations under a pessimistic climate change scenario showed that most species would lose more than 30% of their range and eight taxa were predicted to lose 95% of their current suitable range. Yet the good news is that 90% of the species that may experience loss of range would still have suitable sites in protected areas as optional future homes. The authors recommended that a network of protected areas should be employed to accommodate natural or assisted-range shifting of species affected by climate change.

Further to this issue, there is a growing realization that some of the currently identified habitats will not be suitable in the long term for some species. Coastal ecosystems, especially those with fresh water, are particularly vulnerable to climate change (Noe et al., 2019; Wendelberger, 2016). Under conservative climate change scenarios for sea level rise, most United States Pacific estuaries are projected to lose nearly all their extant habitat area by 2110 (Thorne et al., 2018). In addition, southern mainland Australia has already seen winter rainfall declines of 12–20% since the 1970s (Bureau of Meteorology and CSIRO, 2020). While some consideration has been given to climate change and plant conservation (Harris et al., 2013), Diallo et al. (2021) note that climate change does not appear to have been a major factor contributing to considerations related to plant conservation translocations conducted to date. These researchers strongly urge researchers to take climate change into account when choosing recipient sites for future conservation translocations.

(ii) *We will need ample amounts of seeds to establish populations successfully.*

In the past four decades, the integration of ex situ conservation with translocation and species recovery is a cause to celebrate (Heywood, 2017). Targeted ex situ collection initiatives for entire regions have been underway and have successfully stored seeds around the world, including the Millennium Seed Bank Partnership (https://www.kew.org/read-and-watch/20-facts-millennium-seed-bank-20th-anniversary), MedCARE (Fenu et al., 2020), the Australian Seed Bank Partnership (https://www.seedpartnership.org.au/) and the California Plant Rescue (www.caplantrescue.org). These initiatives reflect an understanding of the importance of having adequate seed supplies if we are to curb the loss of biodiversity. There is also a great need to increase the efficiencies of using seeds, reduce procurement costs and improve successful establishment from seeds (Cross et al., 2020).

Whilst banking seeds is an appropriate action for many taxa around the world, it is also recognized that seed orchards or field gene banks may also be needed to ensure adequate seed supply is available for future restoration (Broadhurst & Coates, 2017, CPC, 2019). The nature of these required techniques and resources is likely to vary with the life history of the target species. Furthermore, for species that cannot be banked or may not produce seeds, cryobiotechnologies will be required (Pence et al., 2020). New initiatives for unlocking secrets of storing exceptional plant species are also underway, such as those at the Exceptional Endangered Plant Conservation Network at the Cincinnati Zoo and Botanic Garden, which is also helping researchers communicate their successes and challenges in this area.

(iii) *Maximize and rescue genetic diversity in many different ways.*

Increasingly conservation translocations will be challenging and we will have to embrace an adaptive framework, involving the easing of restrictions about using single seed sources that are near a particular site. Such adaptations may include the conscious mixing of genes from different populations to maximize genetic diversity or enhancing gene flow across populations especially those impacted by fragmentation. This could involve moving germplasm from large, diverse populations to smaller ones. However, using single sources for conservation translocations of ecologically and genetically distinctive populations is advised (Kaulfuß & Reisch, 2017), noting that an exception to this generalization may need to be considered if the size of populations is not demographically viable.

(iv) *We will need to introduce rare species into restored land that may have been previously degraded and within urban matrices or at the wildland-urban interface.*

Some experiences have shown that restoration of a site prior to conservation translocation promises to improve success, such as in Case Study 6, the recovery of *Braya longii*. This action of restoring degraded sites may create refuges that benefit sensitive species, especially those that may lose their homes to sea level rise or changing precipitation patterns. Seeking suitable sites that are close to existing populations is a reasonable first step, and these will likely be at a wildland-urban interface. It should also be noted that, as droughts become more severe, restoring hydrological systems will be necessary to conserve rare species.

It is anticipated that new genomic data may enhance our abilities to detect and use unique genes for resistance to pathogens, as has been demonstrated with the American chestnut (Newhouse & Powell, 2021), or may allow us to see small differences between populations as in the work with Torrey pine (Steele et al., 2021).

This promising approach may be especially applicable if technological advances can reduce the cost of genomic testing.

Carefully conducted multi-generation genetic monitoring studies (See Case Study 3, *Arnica montana*) inspire us to consider genetic outcomes as an evaluation of translocation success (Albrecht & Edwards, 2020). It is clear that striving to have translocated plants capture as much genetic diversity as possible will help to ensure high adaptive potential (Godefroid et al., 2016).

Case Studies

Case Study 1: Disease Impact and Translocation of the Critically Endangered Feather-Leaved Banksia (Banksia brownii)

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Rationale for the Conservation Translocation

Feather-leaved banksia (*Banksia brownii*) is a long-lived shrub or small tree growing to 4 m, which is killed by fire and regenerates from seed that has been stored in the woody fruits in the plant canopy. Its large, conspicuous inflorescences are pollinated by nectar-feeding birds and small mammals, it is self-compatible and can have significant levels of selfing. Plants are known to grow within three geographically distinct areas of southwest Western Australia, over a range of approximately 90 km (Fig. 8.2). Population genetic studies show that significant genetic differentiation exists between these areas, and these isolated population groups also display significant ecological differences, occupying contrasting habitats with different substrates, associated vegetation, and climate. These populations are considered to be discrete conservation units important for the management and recovery of the species (Coates et al., 2015).

Feather-leaved banksia is highly susceptible to the introduced soil-borne pathogen *Phytophthora cinnamomi* and this constitutes the greatest threat to this species' ongoing persistence (Barrett & Yates, 2015). Of the 30 known populations, 12 are now extinct due to *Phytophthora* dieback. Genetic diversity studies based on material grown from ex situ collections from both extirpated populations and extant populations indicate that *Phytophthora* dieback reduced 38% of the feather-leaved banksia total genetic diversity (Coates et al., 2015). The species is ranked as

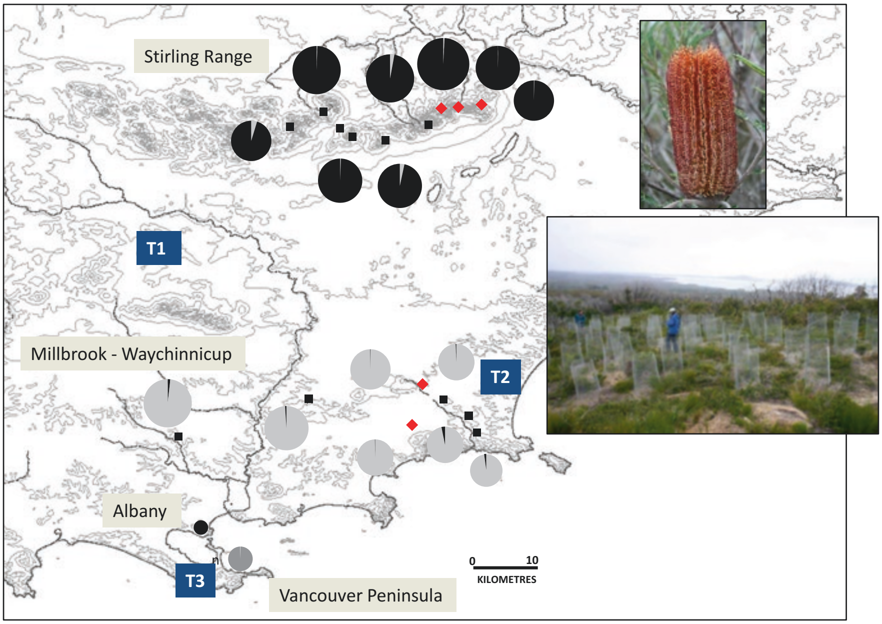

Fig. 8.2 Pie charts show mean q-matrix membership proportions of *Banksia brownii* populations when $K = 3$ from a STRUCTURE analysis (Coates et al., 2015). The size of pie charts is relative to the level of genetic diversity. Germinated seed from extinct populations was initially used to establish two separate translocated populations (T1 and T2) in disease-free areas. The translocated population T3 was established in a disease-free site with seed from the single Vancouver Peninsula population
Key: Extant populations ■ Germinated seed from extinct populations ♦.

Critically Endangered under the *International Union for Conservation of Nature's* Red List of Threatened Species (www.iucnredlist.org).

The Conservation Translocation

Three separate conservation translocations were planned to represent each Feather-leaved banksia conservation unit (Monks et al., 2019) (Fig. 8.2). Using a GIS-based search of conservation reserves in the vicinity of the known Feather-leaved banksia populations, we located suitable translocation sites in each of the three regions where this species occurs. Search criteria included similar soil and vegetation to the natural populations, secure tenure, absence of threats, and proximity to the known historical and extant populations. *Phytophthora* dieback was the most significant challenge in many otherwise suitable sites, including all potential sites covering the disjunct populations in the Stirling Range National Park. Using an experimental framework, we established each translocation using seeds collected from natural populations which had been stored, in some cases, for up to 20 years. A proportion of the seed used for two of the conservation translocations (Sites T1 and T2 in Fig. 8.2) included seed from populations that have since gone extinct in the wild. To ensure disease protection at each site, strict disease hygiene measures were applied, including (i) nursery soil being sterilized (ii) vehicles, footwear, and other work equipment cleaned of all soil and sprayed with fungicide prior to entry to seed orchards, and (iii) entry to the sites after planting was restricted to dry soil conditions to limit movement of soil when monitoring the sites.

Major Outcomes

- Potassium phosphite aerial applications for managing *P. cinnamomi* were successful in preventing further population loss and have stabilized population numbers despite the natural populations being infected by the pathogen*;*
- New populations representing three genetically distinct conservation units have been successfully established through conservation translocations to *Phytophthora*-free sites;
- Genetically diverse ex situ seed collections have proven to be critical for the conservation of Feather-leaved banksia and the establishment of conservation translocations. Some 60,000 seeds held in long-term ex situ storage represent the genetic diversity of the species including seed from now-extinct populations.
- Having access to seed from extinct populations when establishing translocations, highlights the importance of ex situ seed collections, not only for the conservation of this species, but many other threatened species in the region, as is noted in Case Study 5. There are now 65 seed collections from across 24 distinct Feather-leaved banksia populations. As a risk management strategy, 23 of these collections have been duplicated with the Royal Botanic Gardens Kew, in the Millennium Seed Bank.
- While conservation translocations and associated recovery actions are usually designed to be primarily beneficial to a single target threatened species, there can be significant benefits for co-dependant species. These benefits can be enhanced if management can be coordinated between the host and the dependent species. For example, *Banksia brownii,* which is a threatened species, is the sole host to the critically endangered herbivorous plant-louse *Trioza barretta.* A coordinated approach to prevent co-extinction has recently involved the successful translocation of the plant louse from the Vancouver Feather-leaved banksia population to the new translocated population at T3 (Fig. 8.2) (Moir et al., 2016).

What Worked to Ensure Euccessful Translocation and Minimize Disease Risk?

- Careful selection of disease-free sites was critical in establishing new populations as many otherwise suitable sites were infested with *Phytophthora cinnamomic.* These infested areas included all the obvious potential sites covering the disparate populations in the Stirling Range National Park.
- Strict hygiene protocols during the translocation process and during subsequent monitoring has continued to ensure that no disease has been introduced to date.
- Using an experimental framework has facilitated the collection of important information for long-term translocation success in species facing similar threats, as in Case Study 5, the Stirling Range Endemic Species in Seed Orchards Case Study.
- Summer watering significantly improved the survival of planted seedlings.

What Is Next?

- While all three translocated populations have flowered and produced seed from multiple plants, recruitment has only been observed at Site 2. Understanding how recruitment can be encouraged at the other sites and the potential use of fire as a stimulant will require further investigation.
- Detailed monitoring of all translocated populations has been carried out every 12 months and will continue. This includes assessing survival, reproductive state, and recruitment.
- After 10 years, survival at one site declined to 20%, despite an initial three years of good health and survival. Extended dry periods associated with a changing climate suggest the site has become unsuitable for this species. This decline has aided in developing more accurately defined site characteristics required for Feather-leaved banksia. Further conservation translocations at wetter sites may be required to conserve the Stirling Range population group.

Case Study 2: Using an Adaptive Management Process to Restore Key Tree Cactus (Pilosocereus robinii)

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Rationale for the Conservation Translocation

Key tree cactus (*Pilosocereus robinii)* is a North American endangered columnar cactus endemic to the Florida Keys, USA, northwest Cuba, and The Bahamas (Fig. 8.3a, b). Growing at elevations below 2.15 m in the Florida Keys, *Pilosocereus robinii* has experienced steep population declines since 1994 and is now extirpated from Lower Matecumbe Key (Lima & Adams, 1996; Possley et al., 2021). A closely related taxa, *Pilosocereus millspaughii,* has also been extirpated from Key Largo due to sea level rise (Possley et al., 2021). In the next 50 years, sea level rise projections range from 21 to 64 inches (53 to 163 cm) and in 100 years, rises of 40 to 175 inches (101 to 445 cm) may occur. These increases will gravely threaten all of the United States populations (Southeast Florida Regional Climate Change Work Group 2020). In addition to sea rises, the increased frequency and intensity of hurricanes has substantially and negatively impacted populations in the past two decades. Further, few plants now produce fruit in the wild, and pathogens of unknown origin are quickly killing individual species. Finally, high development pressure has left limited suitable protected habitat for the species within existing ranges in the USA.

Fig. 8.3 (**a**–**c**) Key tree cactus in cultivation at Fairchild Tropical Botanic Garden: (**a**) flower, (**b**) a ripe fruit that has split open (common in the species), and (**c**) several dozen seed-grown cacti. Photos: James Lange, Devon Powell, Brian Harding/FTBG

The Conservation Translocation

The US Fish and Wildlife Service and Florida Park Service have worked with Fairchild Tropical Botanical Garden's plant conservation team to recover a significant population of the Key tree cactus. Prior to the conservation translocation, many questions needed to be resolved so that appropriate sites could be selected, and we sought to understand factors associated with species mortality and population health.

Subsequently, we conducted a series of experimental tests using an adaptive management framework. These involved posing a series of hypotheses, rigorous testing, and critically evaluating findings to revise our ideas and create new hypotheses as we refined our understanding of the conservation needs of Key tree cactus. In this work, our preliminary findings allowed the Recovery Team to select confidently two government-owned locations as recipient sites. We designed our series of conservation translocations under experimental conditions to test the suitability of microsites hypothesizing that survival would vary with elevation and light availability. In 2012, we propagated plants from cuttings and reintroduced 36 plants into low-elevation hammock and 36 plants into a hammock/mangrove ecotone. In 2015, we transplanted 89 plants into a mature hammock, which had the highest elevation of the three sites.

Major Outcomes

- Prior to the conservation translocation, we measured soil salinity, elevation, and canopy cover near-live and recently dead cacti across two wild subpopulations located approximately 50 miles south of the translocation site at the National Key Deer Refuge. We found that soil salinity was 1.5-fold greater near dead (517 ± 96 ppm) than near-live (385 ± 71 ppm) plants, while elevation was 12 cm higher near the dead plants than live plants (Goodman et al., 2012).
- To test whether soil salinities measured in the field would kill Key tree cactus, we conducted a greenhouse experiment with 130 shoot tips of seedlings from each of two maternal lines that were watered with salt solutions ranging from 0 to 80 mM NaCl. At high salinity levels (40 mM and 80 mM NaCl), the plants from the National Key Deer Refuge lineage exhibited reduced growth and high sodium accumulation at high salinity, while plants from the Key Largo lineage exhibited vigorous growth, low sodium accumulation, and salt tolerance (Goodman et al., 2012). Subsequently, the Key Largo plants subsequently were reclassified to be the different taxon *Pilosocereus millspaughii* (Franck et al., 2019).
- Taking into account the experimental levels of salt tolerance, we assessed soil salinities in 2008 and 2011 across the eight extant populations and two potential reintroduction sites in the Florida Keys. In 2008, four sites had salinities greater than the salinity level we measured near dead plants, whereas in 2011 all sites showed low and tolerable levels of soil salinity. We verified that soil salinity levels at the potential reintroduction sites were low enough to support Key tree cactus.
- Soil salinity was not the only mortality factor for the reintroduced cacti. Regular monitoring showed that an unknown pathogen caused rapid decline and death in some cacti, while hurricanes damaged otherwise healthy plants, some of which slowly declined and died (Figs. 8.4 and 8.5). By Feb 2020, 12 of 36 cacti (33%) in the low-elevation hammock survived, 17 (47%) survived in the hammock/mangrove ecotone, and 19 (21%) survived in the high-elevation hammock. It is important to note, however, that initial planting size was a confounding factor, wherein cacti installed in the high-elevation hammock were significantly larger than those at the other two translocation sites.

What Worked?

While the precise reasons for this species' decline in the Florida Keys remain elusive, what is clear is the continued importance of maintaining ex situ collections of the Key tree cactus and continuing these reintroduction efforts. Our ex situ collection at Fairchild now holds approximately 50 potted plants from almost all wild populations, and nearly 10,000 banked seeds. Several dozen newly germinated seedlings in ex situ holdings will add immensely to the stability of our collection and provide material for future reintroductions. In total, the ex situ holdings at Fairchild represent more individuals than are living in the wild and include rooted cuttings from plants that are no longer alive in the wild;

Fig. 8.4 A larger Key tree cactus that appeared healthy when planted sometimes completely collapsed within a few months, for unknown reasons. The plant shown here was introduced to a preserve in the Upper Keys in 2015. It was reportedly still in fair health with a few dark spots in June 2016 but dead in January 2017. (Photo: James Lange/FTBG)

Fairchild Tropical Botanical Garden recently provided 92 seed-grown plants to Keys land managers for translocation to seven sites which were selected through a systematic process, in which all land managers participated (Figs. 8.3c and 8.6).

What Were the Challenges?

Because flowering and fruit set in the wild were rare, we propagated many plants from vegetative cuttings for the 2012 and 2015 reintroductions (Fig. 8.3 a, b). Whilst this allowed us to conserve individuals that were dying in the wild, we did observe however that these larger container-grown plants had weaker root systems, quickly became root-bound, and were easily knocked over in hurricanes when used in the translocation;

The high-elevation hammock had rocky substrate with little if any soil. At the land manager's request, we used only sterile sand and perlite to secure plants at the site, resulting in plants having a low survival rate;

Hand-pollinations must occur during a narrow flowering window. Even plants propagated from cuttings require several years to reach reproductive maturity;

Hand-pollinating the ex situ plants generated many viable fruits in captivity, making it possible for us to collect and bank seeds and grow more healthier plants for conservation translocations;

Fig. 8.5 The final collapse of what was once a magnificent colony of *Pilosocereus millspaughii* in the Upper Florida Keys, stretching more than 3 meters tall and several meters wide. This colony, discovered in the 1980s, began to decline in the 2000s. Hurricane Irma in 2017 (and its accompanying storm surge) seemed to have dealt the final blow. (Photo: Jennifer Possley/FTBG)

Despite sending tissues to state laboratories, we were not able to identify the pathogen infecting Key tree cactus. The conservation translocations certainly evidenced the rapid action of an unknown pathogen, with individuals that appeared well on monitoring period being completely mushy a week later. Our Australian colleagues' success with potassium phosphite (Case Study 1) is inspirational, but it is yet to be determined whether using this chemical would be effective or allowed on public lands in Florida.

What Is Next?

Fairchild Tropical Botanical Garden will continue to hand-pollinate the ex situ collection to increase the numbers of seeds that are available to conserve in ex situ seed banks and for propagation to conservation translocations. Our land manager partners' plan to continue to work with us to build up wild populations, either through augmentations or reintroductions to new locations. It has given us great hope that there are so many individuals committed to seeing that the Key tree cactus stays on the land for as long as the sea allows.

Fig. 8.6 Fairchild staff and volunteers reintroduced 36 seed-grown Key tree cacti to a previously-augmented preserve in the Upper Florida Keys in 2021. Selected sites were beneath natural canopy openings. (Photo: Rebecca Collins/Florida Park Service)

Case Study 3: Mountain Arnica (Arnica Montana)

Prepared by Joyce Maschinski, Center for Plant Conservation and San Diego Zoo Wildlife Alliance, based upon Van Rossum and others (2020)

Rationale for the Conservation Translocation

Mountain arnica *(Arnica montana)* is a perennial, self-incompatible, herbaceous plant species with a transient seed bank that grows in both the lowlands of western Europe and the mountainous areas of central Europe. Lowland populations have rapidly declined, despite continued ecological management. With the onset of changing climate effects, the potential for losing lowland genetic diversity spurred van Rossum and co-workers to investigate whether plant conservation translocations could restore genetically viable populations. They also examined whether genetic changes occurred during the translocation process, and if so, what were the nature of these changes.

The Conservation Translocation

Seeds collected from two large populations of Mountain arnica at Elsenborn and Lagland, Belgium, were propagated in a nursery to provide the translocation's source material. Van Rossum et al. (2020) transplanted 700 plants of 10-week-old Mountain arnica plants from the two mixed sources into each of three recipient sites within a fenced enclosure. In this translocation plot, existing vegetation had been scraped to reduce the competition from extant plants and to reduce the effect of introduced soil nutrients. This prior treatment served to create a condition of sparse covering flora and therefore little competition for the transplants. Prior to translocation and for each life stage (transplants, first-generation seedlings, and juveniles), the researchers determined estimates of within-population genetic variation at nine nuclear microsatellite loci for the three translocated populations and for the two seed source populations.

Major Outcomes

A number of useful outcomes emerged from this case study, which may be of assistance in future translocation planning. These outcomes were:

(i) Recruitment was observed to occur only two years after installation from sexual reproduction.
(ii) Translocated populations exhibited high contemporary pollen flow, substantial admixture between source populations, and low inbreeding in F1 offspring. There was no genetic differentiation between generations within and between the translocated populations, and no evidence of outbreeding depression in F1 offspring. This indicated that mixing source populations resulted in high genetic diversity.
(iii) Multigenerational genetic monitoring and fitness assessments were used successfully to evaluate the extent of experimental conservation translocations.

What Worked to Optimize Genetic Restoration?

There were three important observations arising from this work:

(i) Using a high number (700) of founding plants from two genetically diverse seed sources resulted in significant population stability.
(ii) The self-incompatibility system of Mountain arnica favoured disassortative mating and prevented inbreeding, as predicted by Luijten et al. (2000).
(iii) Intensive site preparation provided conditions of low competition from existing vegetation sources and the bare soil environment promoted flowering of the transplants, seed germination, and robust seedling establishment.

What Is Next?

The authors of this case study have recommended additional monitoring of the established population, because it is possible that the effects of outbreeding depression may be expressed later in the second and third generations. Such outbreeding may emerge after the disruption of co-adapted gene complexes (Frankham, 2015), unless reproductive isolation between differentiated populations already exists (Martin et al., 2017).

Case Study 4: Regenerating Declining Translocated Populations of the Spiral Fruited Wattle with Fire

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Rationale for the Conservation Translocation

Spiral fruited wattle (*Acacia cochlocarpa* subsp. *cochlocarpa*) is a perennial prostrate shrub that is endemic to the Mediterranean-type climate zone of south-west Western Australia. The subspecies recruits prolifically post-fire, reaches reproductive maturity within 2 years, then gradually the numbers decline until the subspecies persists primarily as a long-lived physically dormant soil-stored seed bank until the next fire event (Yates & Broadhurst, 2002). In the late 1990s, the subspecies was known from 51 individuals across two populations found on the side of a highway (Monks et al., 2018). Threatened by road maintenance activities, weed invasion, and chemical spray drift from adjoining agricultural land, the populations were steadily declining. The absence of fire to stimulate seedling recruitment, challenges of managing threats in the roadside habitat, and the limited remaining habitat led to the decision to establish new translocated populations to conserve the subspecies.

The Conservation Translocation

Over a period of two decades, four translocation sites were established primarily through planting ex situ propagated seedlings (Monks et al., 2018). Multi-year planting occurred at each site (between two and five plantings), and the need for summer irrigation and herbivore exclusion was investigated experimentally at two sites. Regular monitoring was undertaken of species' survival, growth, reproduction, and natural seedling recruitment. When no seedling recruitment was observed at the translocation sites after 15 years, prescribed fire was applied to one translocation site to stimulate regeneration.

Major Outcomes

- Four new populations of at least 180 mature plants each were established and have subsequently produced viable seed at a rate comparable to the wild population.
- Fire appears to be a major factor in ensuring recruitment and may likely need to be incorporated into future management of conservation translocations of this species.
- Whilst excluding herbivores had no effect in the year after planting, it did significantly improve survival and growth of translocated seedlings when assessed at 5 and 10 years post-planting, with almost double the number of plants surviving in

fenced areas at ten years post-panting compared to unfenced plants (29% compared to 55%; Dillon et al., 2018).

- Irrigating plants for the first summer, a period of 4–5 months of hot dry conditions, resulted in improved survival of translocated plants, but did not enhance growth over a 10-year period. However, results were confounded by significantly higher than average rainfall before and after planting (Dillon et al., 2018).
- Fire was applied to a 15-year-old conservation translocation site, where no recruitment had been observed. Significant recruitment occurred post fire, and several adults either escaped the fire or re-sprouted. More than 75% of seedlings survived the first summer and 25% of these started flowering in the second year following the fire.

What Is Next?

- Outcomes of the initial experiments at the first two translocation sites were used to improve survival and growth of the subsequent conservation translocations, with herbivore exclusion and irrigation over the first summer applied across all plants.
- Detailed monitoring of plant survival, growth, reproduction, recruitment, as well as results from the prescribed fire will be used to develop a population viability model to inform future conservation actions, including the need for further conservation translocations, whether current conservation translocations are likely to be viable in the long term (>20 years) and what fire regime could be implemented to ensure population stability.
- Fire appears to be a major factor in ensuring recruitment and will need to be incorporated into future management of conservation translocations of this species.

Case Study 5: Conserving Stirling Range Endemic Species in Seed Orchards

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Rationale for the Conservation Translocation

The Stirling Range National Park, in the south-west of Western Australia, is a 1159 km^2 park containing the only major mountain range in southern Western Australia. An extraordinary 1500 plant species' population occurs within the national park, with many endemic species, some occurring on just a few of the mountain peaks. Many of the plant species grow slowly in the montane area of the park, with long juvenile periods, meaning that seed banks may take decades to build to adequate levels that can replenish populations following fire. Following the introduction of the soil pathogen *Phytophthora*, plant dieback has had a devasting impact on the park's flora, with many species experiencing declines in plant numbers and health and the extinction of entire populations (Barrett & Yates, 2015). Whilst aerial application of potassium phosphite has been used in high-priority areas (particularly those with large numbers of threatened plants species) with positive effect, it must be reapplied annually or biannually and is not a cure for *Phytophthora* dieback (Barrett & Rathbone, 2018). In addition, several fires in the past decades have impacted many of the threatened plant species in the range, occurring more frequently than plant species are able to replenish their seed bank. Post-fire herbivory of threatened plant species has also impacted negatively on post-fire recovery (Rathbone & Barrett, 2017). The combined impact of disease and increased fire, has led to many species requiring increased management intervention to avoid extinction.

The Conservation Translocation

While there has been an ongoing seed collection program for Stirling Range species since the early 1990s, with seed from many species collected and conserved in long-term storage before population extinction, collections are still considered inadequate for conservation translocations. In addition, the widespread impact of *Phytophthora* dieback throughout the national park, in combination with increased fire frequency, post-fire herbivory by native and introduced herbivores, and the inaccessibility of the montane habitats to conservation managers, meant that recovery actions such as augmentation and establishing new populations in the montane habitats were not considered feasible at this stage. As a result, we established translocation sites (seed orchards) in *Phytophthora*-free sites outside of the national park. These seed orchards were established for multiple threatened Stirling Range species, with the aim of maximizing seed production to support future recovery efforts, rather than establishing self-sustaining populations.

Major Outcomes

- Between 2003 and 2021, five seed orchards with 15 threatened Stirling Range species have been established, with most species growing in multiple sites.
- Seed has been collected and stored in conservation seed banks from six species, with the remaining species being currently too young to produce seed.
- Seeds collected from older seed orchards have been used as part of seed mix to establish recently planted seed orchards, with two sites being planted in 2021.

What Worked?

- Carefully tracking individual plants from seed collections through propagation and planting has allowed the source population to be taken into consideration in the planting design within seed orchards or in future conservation translocations. This is a consideration when deciding which source populations can be used in admixtures to maximize outcrossing; and which populations should not be used in admixture to reduce the potential for inbreeding.

- Irrigating plants boosted survival rates and maximized seed production.
- Exclusion fences around each site prevented grazing by vertebrate herbivores, which improved survival and seed production. In some specific cases, invertebrate control through targeted insecticide spraying was also required to ensure seed production.
- Locating a site where *Phytophthora* dieback did not occur was a crucial first step, followed by strict hygiene protocols to prevent infection being introduced into the sites. For more comments on this issue, see Case Study 1, the Disease impact and translocation of the Critically Endangered Feather-leaved banksia (*Banksia brownii*).

What Is Next?

- Seed collection from the older seed orchards will continue and will commence at more recently planted seed orchards once they reach maturity. Seed will be stored under conservation seed bank conditions.
- Small-scale trial introductions into montane habitats in the Stirling Range will commence where disease is shown to be absent or where active management with the fungicide Phosphite is occurring.
- Continued management of the seed orchards will be an important ongoing task. This will include the maintenance of fences, regular irrigation of juvenile plants, constant control of aggressive weeds, extermination of invertebrate pests when required, and fire exclusion (active suppression of wildfire where possible).
- Monitoring of plant survival, health, and levels of seed production will be ongoing to provide information to assist in managing these seed orchards.

Case Study 6: Recovering the Endemic Endangered Species Long's Braya (Braya Longii)

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Rationale for the Conservation Translocation

The Limestone Barrens area of northern Newfoundland (Canada) is home to over 100 rare plants ranked as critically imperiled or imperiled, which is an unusually high grouping for the Boreal Biome. This list of rare plants includes four species mentioned in Canada's Species at Risk Act (SARA) and the provincial Endangered Species Act, as endangered or threatened. These species are Long's northern rockcress (*Braya longii*), Fernald's northern rockcress (*B. fernaldii*), Barren's willow (*Salix jejuna*), and Griscom's arnica (*Arnica griscomii* subsp. *griscomii*).

The gravels that underlay the Limestone Barrens have high construction value and consequently have been quarried for more than 60 years. In addition, the coastal habitats of these rare species were bisected by a highway along the Great Northern Peninsula, resulting in widespread habitat loss and degradation, with significant loss of populations of these rare plants. As Long's braya is only found in a very restricted range along a narrow strip located in a 6 km stretch of coast near Flower's Cove, the

Braya Recovery and Action Plans (Environment Canada) called for restoration of an abandoned quarry and reintroduction of this endangered plant to an area that is adjacent to Sandy Cove Ecological Reserve (https://www.gov.nl.ca/ecc/natural-areas/wer/r-sce/). This was deemed necessary since natural (passive) recolonization of Long's braya was not seen to be occurring (Mason, 2014).

The Conservation Translocation

In 2016, with funding from both the provincial and federal governments and based on work by Copp (Copp, 2014), the original beach ridge contours, determined from pre-quarrying aerial photographs, were re-sculpted to reflect their former conformation. The overburden, which is topsoil that was piled into mounds during quarrying, was removed with the help of heavy equipment. Prior to the removal, local grade school students and Recovery Team members, together with community members, assisted in rescuing 18 different native species from the restoration area and replanting them in the following spring. Various loggers were installed to track soil temperatures and to record evidence of the natural freeze-thaw disturbance regime on which Long's braya depends to germinate. In the following year, once the soil had settled, Long's braya seed, sourced from the Memorial University Botanical Garden's ex situ collection harvested from the site, had their seed coats abraded to break dormancy, and were planted into 89 plots. In addition, seven other native species were seeded, along with the plants that had been removed prior to the rebuilding of the beach ridges (Fig. 8.7).

Major Outcomes

- *Was Braya longii successfully translocated*? Based on monitoring of the seeded plots since 2017, 10–15% of added seed was seen to have germinated and the emergent seedlings have persisted for five years. This rate of emergence is at the high end of expectations for direct seeding efforts based on earlier seeding experiments (Pelley, 2011). The plants flowered and set seed after 3–4 years.

Fig. 8.7 Quarry before and after physical restoration of beach ridges at Sandy Cove (Newfoundland, Canada). Insets showing *Braya longii* planting plot and a seedling emerging in the following year, with toothpick for scale

- *Are the plants healthy and persisting*? A small proportion of the translocated plants have been colonized by the same pests and pathogens present in natural populations. They include the non-native diamondback moth (*Plutella xylostella*), an agricultural pest that causes loss of biomass (Squire et al., 2009; Squires, 2010), and fungal pathogens that cause reduced seed output and low levels of mortality. However, the plants are successfully producing seed and persisting.
- *Has the nature's disturbance regime been reinstated*? Based on soil temperatures, the newly restored Limestone Barrens are experiencing the same freeze-thaw dynamic as natural barren habitat. In areas where the overburden was not completely removed (due to safety issues around garbage that had been dumped), both native and non-native invasive species have re-colonized.
- **What Is Next?**

We will continue to monitor reintroduction plots to track plant persistence and health, together with the freeze-thaw disturbance regime, and remove invasive species as funding permits. The goal is to incorporate the restored area into the Sandy Cove Ecological Reserve once the Limestone Barrens return close to their natural state.

Conclusions

As is evident from the foregoing discussions and Case Studies, technical, biological, and socio-political factors all contribute to the variable success of conservation translocations. We have emphasized here that following best practice guidelines can help improve chances for success related to technical and biological issues (Commander et al., 2018; CPC, 2019). Setting appropriate benchmarks to measure success and realizing that decades may be required to evaluate true success, since they are a key for managing expectations (Albrecht et al., 2011, 2019). We also strongly recommend that conducting conservation translocations in the manner of experiments and publishing results in accessible papers, is a systematic way to build our knowledge of the reintroduction of endangered species and translocation science. In addition, it will provide a solid foundation for the adaptive management practices of a translocation project (CPC, 2019).

The limited long-term success of translocations to date emphasizes the importance of a balance between translocation, ex situ conservation in seed banks, and the living collections of Botanic Gardens. In addition, there is a growing agreement that in situ conservation actions that include comprehensive surveys, targeted management, and studies on ecological processes and threats to natural populations are needed (Silcock et al., 2019).

As a final comment, we suggest that it is becoming clear that close collaboration with public organizations, Indigenous communities, concerned citizens, government agencies, and local officials will help us save biodiversity, ensure the long-term stability of translocated populations, and promote human well-being in a tangible and effective manner.

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Part II
Restoring Highly Human-Modified Systems

Chapter 9
Roadside Restoration with Native Plants: Partnering for Success in the Pacific Northwest of the USA

Lynda Moore, Kelly Evans, Helen Lau, Lee Riley, Vicky Erickson, and Robin Taylor-Davenport

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Introduction

The construction and maintenance of roads, in conjunction with concomitant roadside activities, have traditionally been viewed as having detrimental effects on the ecological functioning of the disturbed area and surrounding environment. These effects include significant habitat loss and fragmentation, ongoing wildlife disruption and mortality, increased erosion and water quality degradation, altered hydrologic cycles, and the creation of conditions conducive to the establishment and spread of undesirable plant species (Forman et al., 2003; Laurance et al., 2014). Recently, however, more thoughtfully designed road and roadside environments are increasingly valued and utilized for their ability to provide important conservation and restoration opportunities and outcomes. Indeed, there are examples of road modification and construction projects which have been purposefully designed to restore natural water flows to streams in order to protect endangered fish species or have been purposefully built to have the ability to filter pollutants from storm water runoff (Rammohan, 2006). Specifically constructed underpasses and overpasses have been integrated into major transportation projects to enhance wildlife habitat and connectivity while also improving visibility and safety for animals and motorists (Forman et al., 2003; McCleery et al., 2015). Revegetation and maintenance practices are also being adjusted to favor native plant species and provide refugia and dispersal corridors for at-risk plant populations and assemblages of bees, wasps, butterflies, and other important pollinator species (Forman & McDonald, 2007; Brown & Sawyer, 2012; Heneberg et al., 2017; Wigginton & Meyerson, 2018). In addition, roadside stops (parking bays, off-road parking) which are commonly used to provide social and community benefits for humans such as for the observation of notable scenic features, shady rest areas, and general comfort and respite from driving are also being developed with appropriate native flora (see Inset 9.1).

Inset 9.1: Thimmakka, Mother of Trees

Throughout history and across cultures, humans have established trees and vegetation along their travel ways to provide shade, comfort, and other amenities. Saalmarada Thimmakka, also known as Aalada Marada Thimmakka, is an inspiring example of how the perseverance and actions of even one or two people can elevate the quality of life and environmental conditions of their community for current as well as future generations (Wikipedia, 2021). Thimmakka's village gave her the honorary name "Saalumarada", which means 'row of trees' in the local Kannada language. For 80 years, Thimmakka and her husband grew and planted more than 8000 trees along a stretch of highway near their village in southern India. Each day they carried buckets of water and traveled over four kilometers to care for the young saplings as if they were their own children. When the 70-year Banyon trees she planted were recently threatened by a road widening project, Thimmakka appealed to high level officials to reconsider the project and seek other alternatives. The trees are now protected and managed by the local government. She also played an important role in constructing a water tank to store rainwater in her village. Thimmakka has received numerous prestigious awards for her tireless efforts, including the National Citizens Award of India in 1995 and the Padma Shri award – the fourth-highest civilian award in India – in 2019. In 2016, the British Broadcasting Corporation listed Saalumarada Thimmakka as one of the most influential and inspirational women of the world.

The road system footprint in the USA is especially large and widespread, directly covering over 6.9 million hectares, 1% of the national land surface (Steinfeld et al., 2007; Ament et al., 2014). However, recognizing that the ecological effects of roads can extend far beyond the edge of the pavement, it has been estimated that "roadside ecosystems" may comprise as much as 15–20% of the USA. This becomes an even larger area when unpaved roads are considered (Forman & Alexander, 1998; Forman, 2000). Given the growing recognition of the ecological impacts of the US road network, along with current indications of its persistence and expansion over time, the integration of positive ecological goals with transportation infrastructure objectives is becoming standard operating procedure as the country's future roads are constructed, updated, and modified. The new standards are often mandated by federal and state legislation and policy directives, as well as by regulatory agency requirements aimed at protecting both human health and environmental quality.

Several technical resources, developed over the last two decades, are available to support and advance a more ecological approach to road system design and management. The *Roadside Use of Native Plants* (Harper-Lore & Wilson, 2000) was an early foundational publication by the US Federal Highway Administration (FHWA). This group is the chief agency supporting state and local governments in the design, construction, and maintenance of the country's highway system, as well as roads on both federally and tribally owned lands. Other supporting publications soon followed, including *Road Ecology: Science and Solutions* (Forman et al., 2003) and the National Research Council of the National Academies of Science's *Assessing and Managing the Ecological Impacts of Paved Roads* (2005). Collectively, these resources have had a profound influence on the developmental activities of the transportation community in the USA and elsewhere. This is particularly true regarding native plant species and the recognition of their importance and broad utility in roadside revegetation and the maintenance of ecological health and function.

In addition to delivery of technical resources and guidance, FHWA began taking even greater leadership actions to move beyond regulation-driven mitigation approaches and into proactive environmental stewardship actions to promote healthy roadside ecosystems. Over 20 years ago, FHWA embarked on a novel partnership with the US Forest Service (USFS) to provide restoration services and native plant materials for federal and tribal public road projects in the western USA. This partnership was the genesis of the USFS Pacific Northwest Restoration Services Team (hereafter referred to as the USFS restoration specialists). This team is composed of practitioners representing various disciplines that support restoration of highly disturbed sites such as botany, genetics, horticulture, silviculture, and soil sciences. The partnerships between this USFS group and other agencies involved in the planning and implementing roadside projects have become a critical keystone for restoration success in many federal and state highway projects.

Establishing and managing roadside vegetation can be quite challenging in the Pacific Northwest due to steep erosive slopes, lack of topsoil, variable precipitation, and shallow friable soil depths. Recognizing that its unsuccessful revegetation practices were causing serious erosion issues and increased scrutiny from land management and regulatory agencies, FHWA looked to the USFS restoration specialists for assistance in all phases of the planning, implementation, and monitoring of revegetation activities on federal road projects. Because the revegetation contracts of such projects have been decoupled from the construction contracts, much duplication can be avoided through early and consistent involvement of the USFS restoration specialists. Beginning in the early phases of a road project, engineers, environmental specialists, and USFS restoration specialists began working together to communicate and coordinate revegetation activities within the larger context of the road project plan (Fig. 9.1).

Over the years, USFS restoration specialists have provided direct technical expertise to numerous complex and large-scale FHWA roadside revegetation projects while also developing significant collaborative partnerships with other federal, state, and local government agencies in the Pacific Northwest and beyond. These efforts have led to many advances and innovations, including new applications of native plant species, innovations in stock types and seeding/planting methods on harsh sites, novel erosion control/storm water management methods, utilization of biochar and wood waste materials, as well as new road design, maintenance, and monitoring techniques. The involvement of USFS restoration specialists results in more holistic planning, cost savings, and consistency throughout the project maturation. In addition, native vegetation concerns are more fully integrated into the

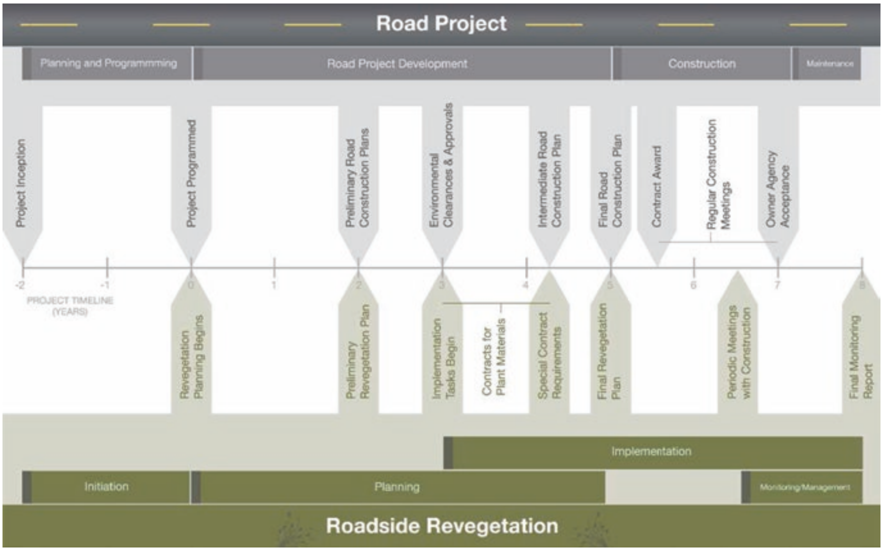

Fig. 9.1 Timeline for coordinating revegetation with the larger processes of road construction. (From Armstrong et al., 2017)

larger processes of road design and construction, resulting in reductions in stream sedimentation, increased plant diversity, more robust vegetation resilience, reduced spread of invasive plants, providing benefits to wildlife, and long-term ecological health and esthetic quality.

Methods and findings from the FHWA-USFS collaboration were synthesized by the two agencies in a comprehensive, state-of-the-art guide for practitioners and planners (Steinfeld et al., 2007) and a subsequent update entitled *Roadside Revegetation: An Integrated Approach to Establishing Native Plants and Pollinator Habitat* (Armstrong et al., 2017). The latter document was written to provide a record of current best practices for planning, designing, and implementing revegetation projects that would also create roadside habitat for pollinator species. Publications describing the partnership and projects from the perspective of USFS revegetation specialists include Landis et al. (2005) and Riley et al. (2015). Landis et al. (2005) provided a review of two challenging road projects in Oregon, plus trials which were designed to evaluate different native seed mixes and application techniques, as well as use of seeded mats on gabion walls. In addition, Riley et al. (2015) described the increasing success and expansion of the FHWA-USFS partnership over the ensuing 10-year period, with advancements in revegetation methods and improved strategies for seed sourcing and production, weed control, new stock types, and outplanting techniques for steep, rocky sites and riverbanks. Other innovations presented included bucket imprinting to reduce soil compaction and methods for the on-site production and application of wood fiber mulch.

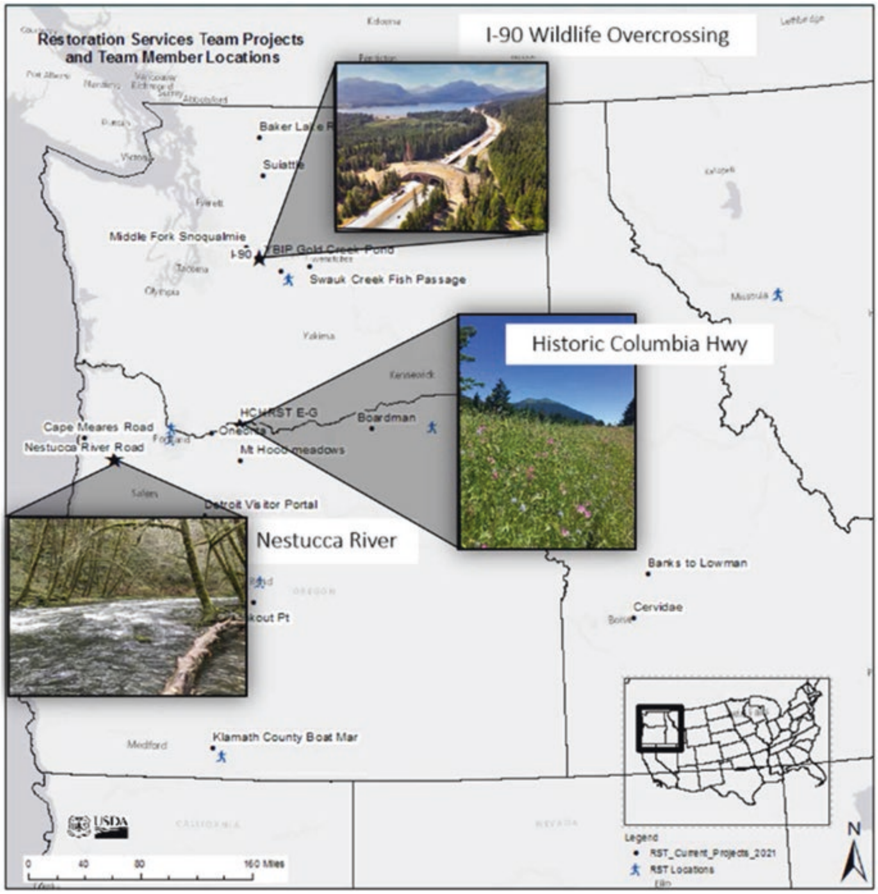

Fig. 9.2 Location map for case studies located in Washington and Oregon, USA. (Created by Helen Lau, Photo credit Anders Norman, Lynda Moore, Lee Riley)

This chapter provides details of the FHWA-USFS partnership work and innovations through presentation of three case studies of ongoing projects in the states of Oregon and Washington (Fig. 9.2). These projects were selected because they reflect diverse, high-profile efforts to accomplish collaborative, science-based roadside restoration in priority forest landscapes in the affected region. Several themes are held in common and woven throughout the three case studies. These themes are applicable regardless of the scope or scale of the roadside restoration project and are key to the overall project success. They include the following:

(i) Recognition of policies, laws, and regulations that promote use of native plant materials in landscape restoration, including roadsides and other highly disturbed sites.

(ii) The use of a seed zone framework for sourcing locally adapted native plant materials.
(iii) Building an interdisciplinary team of resource specialists with training and expertise in disturbed site restoration.
(iv) Obtaining committed funding from partners and collaborators, with an all-lands approach to stewardship and management. Partner engagement and communications with restoration specialists occur early and often throughout project development and implementation.
(v) Establishing an all-lands approach to support effective cross-boundary collaboration and shared stewardship of highly disturbed corridors.
(vi) Ensuring the stability of a strong nursery and agriculture infrastructure for providing diverse, high-quality plant materials.
(vii) Selecting private contractors on the basis of factors such as experience, knowledge, and past performance rather than simply as a function of low bid price. In addition, revegetation contracts are decoupled from construction contracts to allow plant biology to drive the timing of the work.

Overview of the Case Studies

These case studies have been chosen to illustrate how the FHWA-USFS partnership has led to many advances and innovations in restoration practice, including new applications of native plant species, innovations in stock types and seeding/planting methods on harsh sites, novel erosion control and storm water management methods, utilization of biochar and wood waste materials, and new road design, maintenance, and monitoring techniques. The involvement of USFS restoration specialists in these case studies has resulted in more holistic planning, cost savings, and consistency throughout the projects' maturation. In addition, native vegetation concerns are now more fully integrated into the larger processes of road design and construction, resulting in reductions in stream sedimentation, increased plant diversity, vegetation resilience, reduced spread of invasive plants, as well as providing benefits to wildlife and long-term ecological health and general esthetic qualities.

The first case study has a long history and involves the reconnection of past and present roadside practices. The Historic Columbia River Highway, which was fragmented during the construction of the current modern highway, provides a scenic corridor for both motorized and nonmotorized traffic. The project has high public visibility and interest by groups such as bicyclists and hikers for recreational purposes, as well by public agencies who have a pressing need to fulfill state legislative requirements. These legal requirements have been established for almost 30 years and have been formulated to assist in the reconnection of the fragmented historic highway. The fact that the project area lies within a USFS National Scenic Area that has its own unique management obligations and permitting requirements adds complexity to the project and required additional coordination among stakeholders. The result of this work is that a highly disturbed corridor, 4.3 km in length, was rejuvenated, reconstructed, and revegetated as an ecologically sound roadside project. The success of this enterprise was built upon more than 20 years of partnership between relevant agencies, and the trust, time, and enduring relationships developed during the work provided the foundation for a highly successful, innovative outcome.

The second case study details a novel partnership that was developed to utilize native plants to revegetate 24 km of disturbed roadside. Unlike collaborations involved in the Historic Columbia River Highway State Trail that were built on years of developed trust, the I-90 East Snoqualmie Pass Restoration Project (I-90 project) required the rapid formation of new partnerships and understandings. Building trust in such a situation included the joint drafting of details regarding contract structure and negotiations, through to finding commonality of purpose. It was important to establish a path forward which respected experiential and cultural differences, sometimes as fundamental as systematizing the definition of a "native plant." While the I-90 project did not inherently benefit from the long-term relationship between agencies that the previously highlighted Case Study had nurtured, through the diligence of the practitioners and integrity of the results a highly successful and innovative project, and a prized partnership has resulted.

The Nestucca River Road Access Project, the third case study detailed here, can be viewed as the culmination of most aspects of the previous two case studies. While initiation of the project partnership was built upon previous successes and relationships, personnel changes within the major partnering agencies throughout the planning stages of case study 3 resulted in loss of valuable communication channels. This necessitated the reestablishment of working partnerships that required development of trust and diligence on all sides. Once established, these relationships allowed flexibility with ever-changing timelines which consequently fostered innovative revegetation techniques to meet the project needs of harsh roadside habitats. It is worth noting that revegetating this 4.3 km Scenic Byway Corridor included camouflaging a mechanically stabilized earthen wall and installation of seeds and containerized plants utilizing novel practices such as hydromossing, as described in case study 3 below.

Several common themes were present throughout these case study presentations. These themes were found to be applicable regardless of the scope or scale of the restoration project and are key to the overall project success (see Inset 9.2). To help contextualize the key issues involved in the case studies, the following comments have been provided:

(i) *The importance of consistent supporting policies, laws, and regulations*: In 2008, the US Forest Service implemented the first national policy which provided agency direction on the development and use of locally adapted native plant materials as a first choice in revegetation activities carried out in national forests (USDA, 2008). This was followed by the National Seed Strategy in 2015 (USDI, 2015) that strengthened alliances and coordination of native seed management across public and private land ownerships, resulting in increased capacity, shared expertise, and an expanded knowledge of native seed production and use in restoration. Numerous laws are in place to support federal policy and national strategies, the most recent of which is the expansive and unprecedented 2021 Infrastructure Investments and Jobs Act (H.R. 3684). The Act provides extensive funding to implement the National Seed Strategy and prioritizes federal projects and grant programs that utilize native plants in revegetation activities – including pollinator friendly species and projects along road corridor rights-of-way.

(ii) *Identification of seed zones for sourcing locally adapted native plant materials*: Species-specific seed zones based on empirical genetic studies as well as climate-based provisional seed zones (Fig. 9.3) have been delineated in the USA to guide seed movement and minimize risk of maladaptation of plant materials at new planting site locations (e.g., Erickson et al., 2004; Johnson et al., 2013; St. Clair et al., 2013; Bower et al., 2014). Online tools such as Seed Zone Mapper (https://www.fs.usda.gov/wwetac/threat-map/TRMSeedZoneMapper.php) can be used to catalog available seed zone information and facilitate its use in seed collection and sourcing decisions. In addition to enhancing restoration outcomes through use of well-adapted plant material, seed zones create efficiencies and economy of scale in seed and plant production systems, as well as stability and predictability in the commercial market. Seed zones also provide a useful framework for seed use planning and create opportunities for the sharing

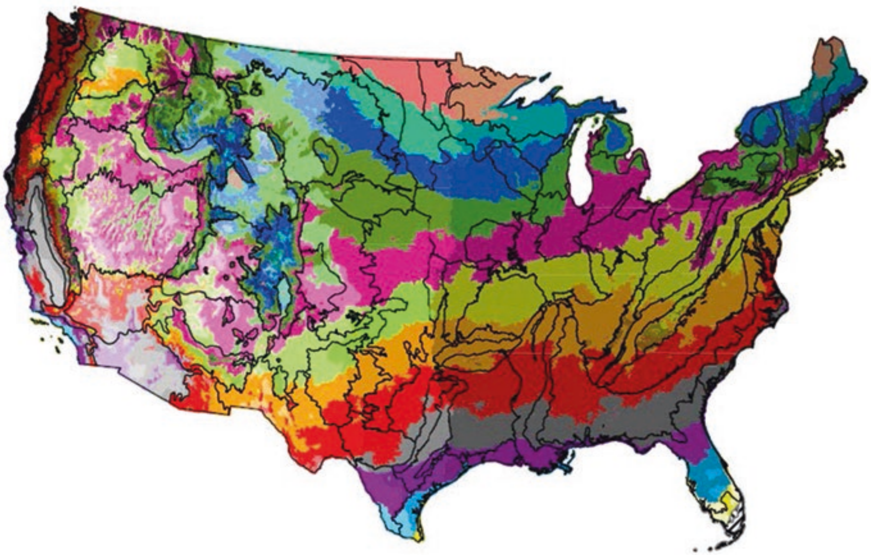

Fig. 9.3 Provisional US seed zones generalized provisional seed zones (Bower et al., 2014) have been developed using climate data (winter minimum temperature and aridity) along with ecoregional boundaries (Omernik, 1987) to delineate areas that have similar climates but differ ecologically

and exchange of plant material among landowners and seed banking programs and partners. Collectively these attributes help reduce plant material and overall restoration costs, leading to the increased availability and use of genetically appropriate plant materials in restoration.

(iii) *Involving a team of dedicated specialists and a trained workforce*: USFS restoration specialists consist of USFS employees who are highly specialized in the restoration of challenging sites including wetlands, riparian areas, roadsides, powerline corridors, and other anthropogenically disturbed sites. The team utilizes genetically appropriate and locally adapted native plant material in their restoration efforts and has expertise in botany, soils, horticulture, pathology, silviculture, and genetics. All phases of restoration, from project design and planning, seed and plant collection, plant establishment and implementation, and project monitoring, can thus be seen through to completion.

(iv) *Ensuring consistent funding commitments*: Successful projects require more than just technical information and need – they also require trust, time, money, commitment, communication, and use of systematic, comprehensive, and collaborative methods. Funding goes hand and hand with the long-term nature of most road-related restoration work. Time is required to plan and secure plant material, establish plants, monitor, and the opportunity to build on past successes, learn from failures, and test new methods. Consistent funding for research and development is needed to advance restoration knowledge and generate new technological innovations that reduce costs and improve project success and efficiency by instituting critical evaluation procedures.

(v) *Introduction of an "all-lands" approach and shared stewardship of highly disturbed corridors*: USFS restoration specialists often follow an approach where the project encompasses multiple ownerships and land stewardships, and this is termed an "all-lands" approach. Seed zones cover large areas, and, with shared stewardship, it is possible to use consistent native plant materials across the landscape regardless of ownership. This is an important consideration because the ecological effects of highly disturbed sites can extend far beyond the edge of pavement or the edge of disturbance. Impacts of these disturbances can include habitat fragmentation, wildlife mortality, water quality degradation, increased erosion, and an increase in noxious and invasive plant cover. Using a holistic approach throughout the disturbance can mitigate some of these impacts, and the ability to call upon added plant materials suitable for the site is often essential.

(vi) *Instituting supporting nursery and agriculture infrastructure*: USFS nurseries and seed extractories have contributed significantly to roadside restoration and other revegetation efforts by processing and storing seed, producing plant materials, and establishing studies to determine the effectiveness of different revegetation techniques and plant materials for disturbed sites. USFS nursery staff also provide training and consultations on seed use planning, collection

Fig. 9.4 A variety of containerized sedges, rushes, ferns, and wetland grasses growing at the Dorena Genetic Resources Center (Cottage Grove, OR) for outplanting on a roadside restoration project

methods, and seed handling, testing, and storage requirements. While the primary focus of most nurseries is on plant production, the USFS Dorena Genetic Resource Center (DGRC, Cottage Grove, Oregon, Fig. 9.4) has purposefully expanded their services to become the primary plant development center for the USFS partnership with FHWA and other federal, state, and tribal entities. These partnerships have been a driving force in extending DGRC capabilities to become a full-service restoration center for much of the western USA.

(vii) *Involvement of private contractors*: Regional contracts with prequalified pool of contractors perform a wide array of restoration-related activities from seed collection and production to plant installation. Because the revegetation contracts of such projects have been decoupled from the construction contracts, much duplication can be avoided through early and consistent involvement of the USFS restoration specialists. For many years, federal and state road projects were limited to accepting low bid revegetation options, but experience has proven that other criteria such as past performance, experience, and knowledge can be more valuable.

In essence, the presented case studies demonstrate that innovative use of native vegetation can meet regulatory requirements and achieve ambitious restoration priorities to enhance esthetic values and restore ecological function to highly disturbed road systems and surrounding environments. The best management practices used on the roadside restoration projects are applicable to any project with severe land disturbance such as powerline corridors, recreational trail reroutes, restoration of abandoned mine sites, and post-agricultural fields.

Inset 9.2: Keys to Successful Revegetation

Case Study 1: The Historic Columbia River Highway Project

Rationale and Strategy

The Historic Columbia River Highway (HCRH) was originally constructed between 1913 and 1922 (Fig. 9.5). The alignment took full advantage of the unparalleled beauty of the Columbia River Gorge. Located along the Oregon side of the Columbia River, the highway dazzled tourists and locals alike as it meandered near breathtaking waterfalls, panoramic vistas, and spell-binding geomorphology. Many of these scenic features include nearly 30 named waterfalls, including Multnomah Falls and Hole-in-the-Walls, together with the Crown Point Vista House that still delight

Fig. 9.5 Historic Columbia River Highway travelers in Mitchell Point Tunnel, circa 1915. (Photo credit Oregon Department of Transportation)

travelers today. The construction of Interstate Highway 84 (I-84) during the late 1940s and early 1950s disrupted the HCRH, leaving much of it fragmented and abandoned.

A directive was given to the State of Oregon in 1986 via the Columbia River Gorge National Scenic Area Act to reconnect the fragments of this historic highway. Further direction was provided by the Oregon Legislature in 1987 to the Oregon Department of Transportation (ODOT) to facilitate the development of the Historic Columbia River Highway State Trail (HCRHST) by preserving and enhancing existing HCRH segments. Since then, multiple partner agencies including the FHWA, ODOT, Oregon Parks and Recreation Department (OPRD), USFS, and the State Historic Preservation Office, together with a suite of private entities, have collaborated to reconnect the HCRH fragments. Of the original 117 km stretch of highway, 109 km are now currently open in the form of drivable motor vehicle roads and paved nonmotorized foot or bicycle paths. The approximately 8 km of the original HCRH which has not been treated has been surveyed, and plans are now in place to reconnect and incorporate this remaining fragment into the HCRHST (Fig. 9.6).

Starting in 2014, USFS restoration specialists began coordinating and implementing the native plant revegetation along the HCRHST as construction on each segment concluded. The team participated in planning meetings to understand the management objectives and needs of all partner agencies and to reduce redundancies in contracting. In-house collections of seeds and cuttings from native plants prior to the initiation of construction provided the material to federal nurseries for container plants or in private grower's fields for seed increase. These were then installed on the project site once construction disturbances had been completed. The overarching goals of revegetation included (i) to match the existing ecological environments as well as could be done, (ii) to assist partner agencies in meeting their regulatory compliance responsibilities, and (iii) to enhance habitat composition and function whenever possible. USFS restoration specialists implemented revegetation throughout the entire 4.8 km of new trail construction of the segment discussed here, including the management of several features of specific concern.

Fig. 9.6 Historic Columbia River Highway State Trail, segment completed in 2019. (Photo credit Oregon Department of Transportation)

Concerns and Barriers

Because the HCRHST lies within the USFS Columbia River Gorge National Scenic Area (CRGNSA), retaining and enhancing natural visual effects was a major concern. At the same time, care was taken to avoid or minimize disruptive effects on sensitive wildlife and plant habitat, pollution and sedimentation from entering streams and wetlands, and damage to heritage and artifact sites. Mitigation sites are often required by various regulatory agencies when impacts to resources cannot be fully avoided or minimized. In these cases, areas are identified during the planning of the project to receive intensive restoration as compensatory mitigation for effects elsewhere on the project site. For this reason, and to secure the required National Scenic Area Permit from CRGNSA, the project designers selected a 2 ha open area to serve as a mitigation site. The area was almost entirely encompassed with state listed noxious weeds such as Himalayan blackberry (*Rubus armeniacus*), Scotch broom (*Cytisus scoparius*), English ivy (*Hedera helix*), and many others. Such highly disturbed sites exhibiting a significant ecological imbalance are often selected for mitigation sites due to the extreme need to engage rehabilitation processes. In this respect, the restoration contractors repeatedly treated and removed the noxious weeds from the area prior to any construction activities. This work lessened the probability that noxious weed propagules would be distributed throughout the project site during construction. It also helped to reduce photosynthetic material of the plants, thereby diminishing the starch storage in any remaining root systems. This resulted in the lessening of intense post-construction treatments which would have been needed if the noxious weed populations remained intact initially.

Key Features

Restoration of the approximately 2 ha area not only served as mitigation for impacts incurred as a result of the project but also helped to address Section 3 of the 2014 Presidential Memorandum "Creating a Federal Strategy to Promote the Health of Honey Bees and Other Pollinators." This memorandum provides a general directive to increase and improve pollinator habitat on federally managed lands. Plant species that can act as host plants to provide feeding, nesting, resting, and rearing benefits to pollinating insects were utilized to support and encourage plant-pollinator interactions (Inset 9.3). To this end, attention was paid not only to nectar producing plants but also to plant architecture, the construction of microhabitats and climates through the use of large and small woody material, depressions to retain moisture, areas of vegetative refuge, patches of bare ground (which are necessary for some ground nesting pollinator species), as well as other pollinator sustaining habitat characteristics (Fig. 9.7).

Inset 9.3: Roadside Revegetation and Pollinators

The wide-ranging decline in insect and pollinator species, are occurring in response to stressors such as habitat loss, overuse of pesticides, invasive species, pathogens, and climate change. Reductions in numbers and species of insects are hugely

Fig. 9.7 Intentional divot (circled) in large woody debris to provide pollinator nesting site. (Photo credit Lynda Moore)

consequential. They are responsible for the reproduction of 85% of flowering plants and provide at least 35% of global food production. In short, insects are irreplaceable in the ecosphere. However, in searching for insect and pollinator habitats, it has been found that in some places, roadsides are home to intact and rare native plant communities that can no longer be found in surrounding lands. In the Midwestern Corn Belt of the US, for example, many remaining tallgrass prairie remnants, survive on roadsides and railroad rights-of-way, where now rare plant species that were once common across the region can be located. Iowa's landscape was once dominated by prairie, with a sea of grass and wildflowers covering more than 85% of the state. Now, with less than 0.1% of remnant prairie remaining, and more than 95% of Iowa's original wetlands destroyed, Iowa is the nation's most altered state.

In heavily altered landscapes such as the ones found throughout Iowa, roadsides are often the only natural or semi-natural habitat present. Pollinator diversity can be high on roadsides, with communities that include a significant portion of the species found in the region. Roadsides can be home to rare species of pollinators, such as the rusty patched bumble bee (*Bombus affinis*), as well as common species like painted lady butterflies (*Vanessa cardui*). Indeed, for some species of pollinators currently listed under the Endangered Species Act, or imperiled species that may become listed, roadsides are some of the last remaining patches of their habitat.

Roadsides supply pollinators with flowering plants that provide pollen and nectar, food for pollinators. Roadsides are also sites for pollinator breeding, nesting, and overwintering. Additionally, roadsides can increase habitat connectivity as corridors for pollinator movement. As my graduate school mentor described, several pollinator species have expanded their ranges using roadsides. State Departments of Transportation (DOTs) manage substantial amounts of land and associated natural resources across North America, and these acres hold the potential to create a network of habitats to support pollinators across urban and rural landscapes. DOTs can make a significant difference for pollinators by considering the needs of pollinators when maintaining roadside vegetation or revegetating after construction.

In addition to habitat benefits to pollinators, native roadside vegetation is also valuable in many other ways. It has been found that roadsides with healthy native plant communities can better resist invasion by noxious weeds, are more resilient to a changing climate, have improved erosion control and water infiltration, show higher carbon sequestration, and shelter more ground nesting birds and other small species of wildlife. The benefits of native plants and diverse plant communities along roadsides extend well beyond the roadside edges, since the effects of insects and other species can be observed in the surrounding areas. In addition, roadsides provide states with a way to showcase aspects of local natural heritage and beauty. While the primary role of roadsides is clearly to support transportation infrastructure and roadsides are not meant to be a substitute for natural habitat areas, they play an important role in the landscape, being an asset to pollinators, to DOTs, and to plant communities.

–Jennifer Hopwood
Senior Conservation Entomologist,
Xerces Society for Invertebrate Conservation

Pollinators on host plants of species used in Roadside Restoration Case Study sites. (Photo credits: Kelly Evans (USFS) and Kelsey Loeffler (Benson Farms, Inc))

Given the location of the project, two pollinator species of conservation concern were explicitly supported. First, habitat for the imperiled western bumble bee (*Bombus occidentalis*) was enhanced in this important part of its remaining range by providing a large and diverse resource of nectar producing plants. Second, and more specifically, breeding habitat to support the imperiled monarch butterfly (*Danaus plexippus*) was constructed. Milkweed is the only known plant genus to serve as host plant for the monarch butterfly larvae, and consequently two species of milkweed (*Asclepias fascicularis* and *A. speciosa*) were incorporated in the restoration planting scheme. Of critical importance in this regard was that there were nearby locations in the Columbia River Gorge which are known monarch butterfly migration and breeding areas, and this proximity was reinforced with observations of monarch butterflies close to the project area.

Early monitoring shows promising results for native pollinator populations at the mitigation site. Pollinator surveys were conducted prior to and post-construction using the Streamlined Bee Monitoring Protocol for Assessing Pollinator Habitat (Ward et al., 2014). While monitoring is ongoing and surveys have not been exhaustive, the frequency of native bee visits to the area 2 years following construction and habitat enhancement increased 3.5 times over observations prior to the construction efforts. In addition to native bees, observations of honeybees, flies, wasps, and damsel flies were made during post-construction monitoring. Future monitoring will also target monarch butterfly larvae and adult populations.

Esthetic augmentation included the creation of an earthen berm embankment that was later vegetated to screen a pump house, together with topographic sculpting created terrain variation to mimic existing conditions found elsewhere on the project site. Plants were installed in the pollinator meadow (mitigation site) in random groupings with some individual plants interspersed throughout. The area was also hydroseeded to create a natural appearing meadow-woodland complex.

There was a desire among the partners involved with this project to leave the mitigation site somewhat open, thus creating a meadow-like habitat to support the previously discussed pollinator initiatives. As such, the plant species which were installed at this site included increased numbers of forbs, low shrubs, and native grasses, rather than concentrating on trees alone. A total of 18 native forb and grass species were deployed in the hydroseed mixture. It is expected that without regular maintenance, the natural seed rain from the surrounding trees will eventually convert the open area into a covered area. Increased tree cover was actively avoided during restoration revegetation efforts, and dialog regarding future maintenance is ongoing with the involved land management agencies.

The topography of the project required that a substantial mechanically stabilized earth wall (MSE wall) be constructed directly adjacent to I-84. The height of the MSE wall varied from 3 to 9 m tall, and it was almost 610 m long. Each course, or layer, consisted of a welded wire frame that was 46 cm tall, leaving the front 23 cm exposed, to create a step with each tier. The unnatural appearance of this substantial engineered wall was a significant visual concern for the CRGNSA. To help blend the wall with the environs, it was decided to make the MSE wall a stepped, plantable wall. To do this, the exposed face of each tier was filled from top to bottom with topsoil to a minimum of 23 cm deep from front to back during construction. The soil was held in place with black geotextile fabric placed immediately inside of the welded wire frame.

The restoration contractor is imported and placed almost 1150 m^3 of a compost and topsoil mix on the face and at the base of the wall in nonuniform, undulating deposits. This material visually softened the edges of the wire basket tiers and provided micro-topography like the surrounding conditions. The face of the MSE wall was then sprayed with a hydroseed mixture of native forb, grass, and shrub seeds. The area at the base of the wall was planted with a variety of containerized forbs, shrubs, and trees. As the human eye is naturally drawn to lines, the seed mixture varied slightly when applied to long, linear features to break up the visual effect. After the containerized plant installation, the base of the MSE wall was then hydroseeded with a slightly different native mixture of forb and grass seeds from the mix that was applied to the wall face.

What Worked and What Did Not

One of the critical reasons this project was an overall success was the consistent, frequent, and open communication between the collaborators. The timeline of the restoration contract work revolved around the construction contractor's work with whom every effort was made to minimize conflicts or delays. To achieve this equitable state, communications through text messages, office meetings, on-site visits, phone calls, and emails occurred between all parties involved. FHWA published a weekly newsletter highlighting each week's construction accomplishments, providing timeline updates, and allowing coordination of schedules.

This frequent communication also allowed the team to provide native seed mixes and consultation to the project engineer in real time. Federal regulations require that if a contractor disturbs soil and then allows it to rest undisturbed for 14 days, they must install erosion control measures to mitigate soil migration, erosion, and dust. Rather than using annual, non-native grass seed as had been done in the years past, the project was provided with a mix of native perennial grass and forb species for this purpose. The FHWA engineer would project in 2–3 month increments how much land area would be disturbed but remain untouched and then was able to request enough seed to cover this area from restoration specialists who would mix and deliver the seed. The nativity and local adaptation of the seed that was provided greatly increased the efficiency of revegetation efforts to mitigate erosion through the native plant's extensive root systems, robust top growth, and persistence on the landscape.

Even though years of in-person meetings and conference calls go into the planning and design of these projects, not all communication is always conveyed throughout the hierarchy of personnel. Because multiple agencies were involved, each with their own management objectives and priorities, USFS restoration specialists had concerns regarding the potential of installed restoration plants being accidently mowed or otherwise damaged in some areas due to regular maintenance practices. Even though these concerns were discussed early and frequently in the planning process, this is precisely what happened in some areas. The agreement to provide vegetative screening along the HCRHST was not clearly communicated to the ground maintenance crews, and thus hundreds of native forbs, shrubs, and trees were killed through mowing operations.

There were also several areas designed to be pollinator habitat that received inappropriate mowing treatments. These areas were seeded with a mix that included 19 species of native grasses, shrubs, and forbs that, if allowed to follow their natural growth trajectory, would have created a meadow of balanced plant life forms with gaps between plants to support ground nesting pollinators. Because of repeated mowing, however, the aerial cover of the forbs species was reduced, allowing the grasses to grow very densely. The mowing also killed multiple shrub species and some flowering plant species that are not able to regenerate after the shoot is severed from the root. The problem has since been addressed, and management of the agency has made every effort to communicate the intention and need throughout their personnel levels. While mistakes are still occasionally made, the mowing of restoration plants has been drastically reduced since the problem was identified and thoroughly communicated.

Revegetation of highly disturbed sites can take years to become established. During approximately 2 years or so since the restoration described above was implemented, the project overall looks well-poised to develop into the types of habitats desired. The perennial forbs and bunch grasses have reliably germinated, and while some rebalancing of species density and diversity is naturally occurring, the overall plant population appears to be stable and sustainable. Ongoing weed control efforts are proving to be critical for long-term restoration success. The year 2021 represents the last year of contracted work on this project, and continuity of maintenance by the land management agency will be key to reducing competition from noxious weeds.

Major Outcomes

The extensive revegetation efforts on this project helped agencies meet their regulatory obligations and management objectives. The early involvement of the USFS restoration specialists in the planning process allowed the shared understanding of specific concerns and needs from the collaborating agencies. The team collected, procured, and increased genetically appropriate native plant seeds to satisfy the needs of the construction contractor's erosion control, constructed pollinator habitat to mitigate against all potential effects, and satisfied the National Scenic Area permit by visually blending the MSE wall with the surroundings. Noxious weed propagules were prevented from being transported throughout the project on construction equipment by pretreating them prior to the start of construction. Intentionally leaving the ground surface of the initial mitigation site roughened and decompacted created areas of water retention and microhabitats. These combined efforts have increased pollinator-plant interactions and have created so much of a natural looking recovery that the area is often unnoticed by the traveling public (Fig. 9.8). The fact that such a highly disturbed area could be passed by without notice is one of the greatest accomplishments for which a restorationist could hope.

Fig. 9.8 Pollinator habitat constructed at mitigation site of Columbia River Highway State Trail project in Oregon, USA. (Photo credit Lynda Moore)

Case Study 2: I-90 Snoqualmie Pass East Restoration Project

Rationale and Strategy

East of the Cascade Mountains of Washington State, the Washington State Department of Transportation (WSDOT) and several partners are working together to improve 24 km of Interstate Highway 90 running through USFS Nation Forest System Lands from Snoqualmie Pass to Easton (Fig. 9.9).

The I-90 Snoqualmie Pass East Restoration Project was initiated to upgrade this major thoroughfare over the Cascade Mountains to better accommodate the needs of a burgeoning population. Other major goals were to reconnect the habitat and associated flora and fauna to the north and south of the interstate highway, as well as alleviate concerns over increasing impacts of motorists on the native wildlife. This stretch of interstate highway is the surface transportation lifeline for US $500 billion per year of apples, cherries, wine, potatoes, and other agricultural products. Three reservoirs located along this roadway provide water for many agriculture practices in Washington State and strongly influence the hydrology within this watershed. The roadway also serves as a thoroughfare for masses of people to access both sides of Washington for various activities. On average in 2016, 31,000 vehicles traveled over Snoqualmie Pass every day with the traffic numbers doubling on weekends and holidays. Based on yearly annual report data, the traffic volumes are expected to increase 1% every year (WSDOT, 2016). Concerns regarding avalanche control causing around 65 h of road closures yearly and the freeway bisecting vital north-south wildlife populations added impetus to search for solutions (Fig. 9.10). Related to this concern of divided populations is the concept of environmental connectivity, not only for large roving carnivores but also for low mobility species such as fish, amphibians, reptiles, and mollusks. Consequently, the project discussed here took into consideration hydrologic features on the landscape, low and high flow water movement, and vegetation diversity, which is the largest component of connecting various habitats.

An important component to the restoration project was this consideration of the I-90 corridor as a major ecological barrier for many terrestrial species since it

Fig. 9.9 Image of I-90 Snoqualmie Pass East Restoration Project showing the stretch of interstate to be restored. (Image from WSDOT)

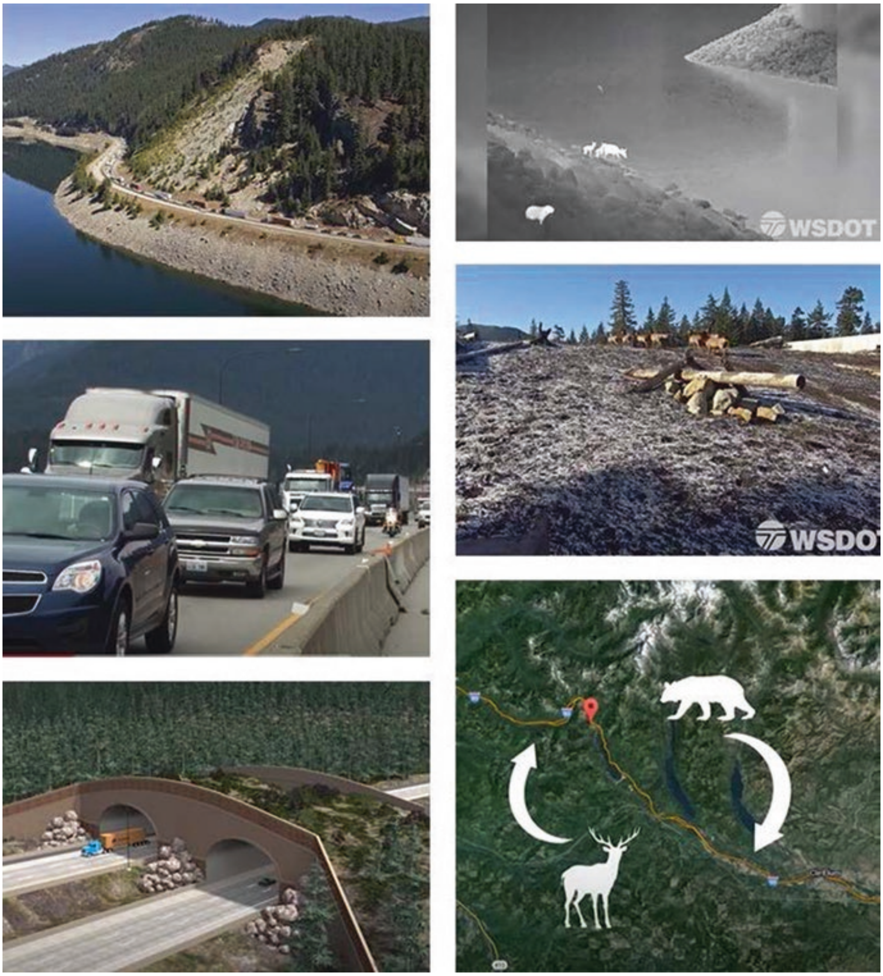

Fig. 9.10 Wildlife and human activity on Interstate Highway 90 in the Washington State Cascade Mountains. (Photo credits Washington Department of Transportation and Conservation Northwest)

inhibits unrestricted physical movement and genetic exchange across the corridor. In this respect, researchers at the USFS Wenatchee Forestry Sciences Lab in Wenatchee, WA, examined evidence of wildlife connectivity and genetic structure and found there were restricted wildlife movement and gene flow among populations throughout the North, Central, and Southern Cascades. This initial research indicated the major wildlife corridors north and south of the interstate were closely linked to site specificity due to the natural topography and human occupation impacts to the landscape. For more mobile species such as elk (*Cervus elaphus*), mule deer (*Odocoileus hemionus*), and large carnivores such as black bears (*Ursus americanus*) and cougars (*Puma concolor*), there is also the concern of public safety from vehicle collisions with these animals. To decrease the effects of I-90 as a barrier to ecological connectivity, wildlife crossing structures became an integral part of the I-90 expansion plan (Fig. 9.11).

In addition to the work being funded by both federal and state dollars, numerous key organizations were formed in 2004, such as the I-90 Wildlife Bridges Coalition, bringing diverse stakeholders together to advocate for and ultimately support the final design for the I-90 Snoqualmie Pass East Project to improve the roadway for both motorists and wildlife. Public support was a key component to provide necessary funding for a project of this scale and complexity. In addition to financial support, local universities, such as the Central Washington University located in Ellensburg, performed pre- and post-monitoring of several species and biological systems related to functional ecosystems on the overcrossings as well as the usage of high and low terrestrial and aquatic species mobility.

The heart of the project is primarily concerned with the state of USFS lands that require revegetation with genetically appropriate and locally adapted native plants to fulfill the needs of this project. These vegetation goals led to several years of reference site visits and planning, resulting in a joint USFS vegetation team consisting of two botanists and three WSDOT staff to oversee the planning for the revegetation component of the project. Each agency represented and justified its needs and expectations of a successful restoration for the roadside environs. WSDOT was focused on safety and low maintenance infrastructure, while the Forest Service focused on ecological sustainability. Finding common goals was a key to the success of the project, and this was supported by early and continuing communication. The designation of roles was integral to the overall success of the project, since the project was not without risk and vested interests, meaning that efforts had to be continually expended throughout all levels of the planning for this restoration project of such a considerable size and scale.

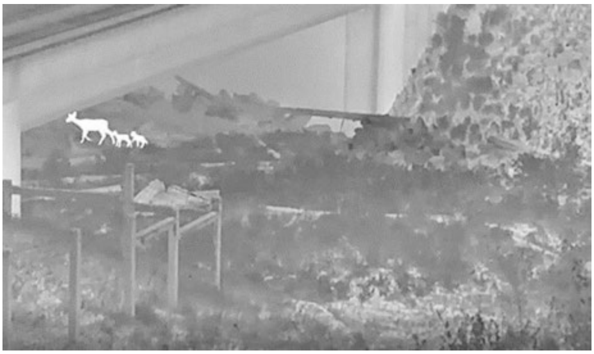

Fig. 9.11 Heat-sensing image of a doe and two fawns safely traversing through a wildlife undercrossing as traffic passes overhead. (Photo credit Washington Department of Transportation)

Concerns and Barriers

With the involvement of many different agencies, all with their individual intentions for this project, one initial hurdle to overcome was the establishment of trust – trust that each agency would deliver what was needed for the overall project to be successful. The revegetation team formed by representatives from both the FS and WSDOT worked together to first identify common goals for the project and identify major concerns. Alleviating these concerns began with identifying the division of work and designating individuals with specific skill sets to be responsible for elements of the project. Every aspect of the project needed to be considered, from the species selection, timing of the material collections, integration with the construction phases, plant propagation strategies and shortfalls, to the revegetation and monitoring phase. Each step was laid out in a way that allowed clear communication in joint meetings. This involved the preparation of large amounts of material which outlined risk and mitigation strategies, as well as precise detail about plant material quantities and timing of collection and propagation of materials. The revegetation team collaborated closely to incorporate both visual and design needs which led to site-specific revegetation plans and planting strategies to direct seed collection, site mapping, and the final collection of plant material. To aid the project, the common vision of supporting ecosystem services, wildlife-specific plant palettes, and native vegetation to support overall landscape resilience was kept in constant focus.

While identifying risks and mitigation strategies, partners realized the contractual advantages of working through the USFS restoration specialists and procurement services to support successful restoration practices. For many years, WSDOT was limited to accepting full roadside construction packages that included revegetation and the contract process by which low bid solicitations were selected. With the inclusion of the USFS, which developed several restoration services contracts that selected contractors based on restoration experience and skill rather than just price, WSDOT could see an immediate benefit to partnering and allowing greater control over restoration practices by revegetation experts. In the past, the transportation agency experienced very low plant survival and planned for a 50% replanting of their revegetated footprint. With the project site in full view of the public, there was immense pressure to maintain high visual standards and revegetation success as well as meeting the permitting requirements of regulatory agencies such as US Fish and Wildlife and Washington State Department of Fish and Wildlife. Meeting expectations of other key stakeholders, including the USFS and nongovernment agencies such as Conservation Northwest and Mountains to Sound Greenway and the public, is also paramount.

The USFS requirements for use of native revegetation on the project was at first considered unattainable based on previous WSDOT revegetation experiences and on outcomes of projects carried out with standard operating procedures for implementing roadside projects. Presenting both the USFS policy and then providing a full methodology for reaching the desired genetically appropriate and locally adapted native species criteria required extensive discussions and trust building. USFS restoration specialists have been successful in obtaining sufficient types and quantities of source plants for the project due to their extensive knowledge of the local plant species and their distribution, as well as an understanding of the diverse habitats present on the project and surrounding landscape. Initial efforts in the project focused on the mapping of source populations for wildland seed collection. This work generated long-term benefits in providing abundant source material for establishing grass and forb seed increase fields and diverse seed mixes for all 24 km of roadside needs over the lifespan of the project. There has also been the added benefit of building local seed and plant resources for use in other local restoration projects due to the highly productive nature of the seed increase fields. With the proper amount of planning, the barrier for seed collection was reduced because areas were preselected from reference sites and knowledge of plant propagation of various species increased success in seed availability. Planning for both early and late successional species, as well as a suite of species that can fill a variety of transitional habitats and tolerate highly disturbed sites as key species in the mix, maximizes the flexibility in developing successful revegetation plans (Fig. 9.12). Consideration for the full three-dimensional space that plants occupy, both above and below ground level, increases the restorative value of revegetation and increases the trophic levels affected. Although root structure is often overlooked, their value in soil stabilization cannot be overstated. Grass species tend to stabilize the upper most root zone, forbs and shrub roots stabilize below grass roots (down to 3 m or more), and trees stabilize soil layers at even deeper levels (Fig. 9.13).

Soil compaction and soil structure are a constant and consistent concern and barrier to vegetation growth and hydrological issues. Engineered design requires exactly defined guidelines regarding where the actual restoration is to return to

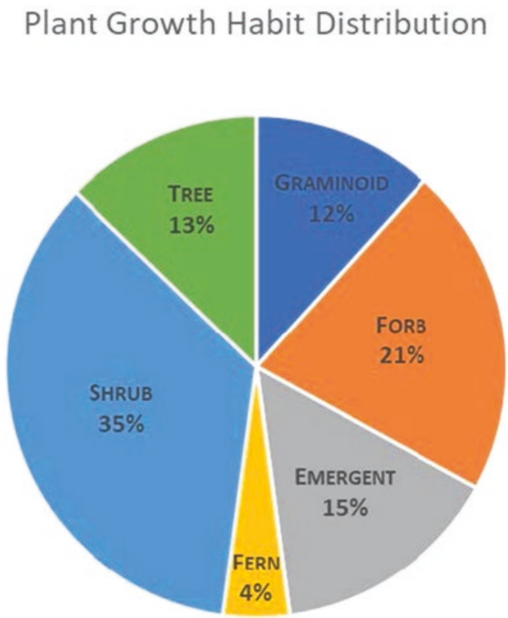

Fig. 9.12 Proportion of species life forms used in the I-90-Snoqualmie Pass East case study

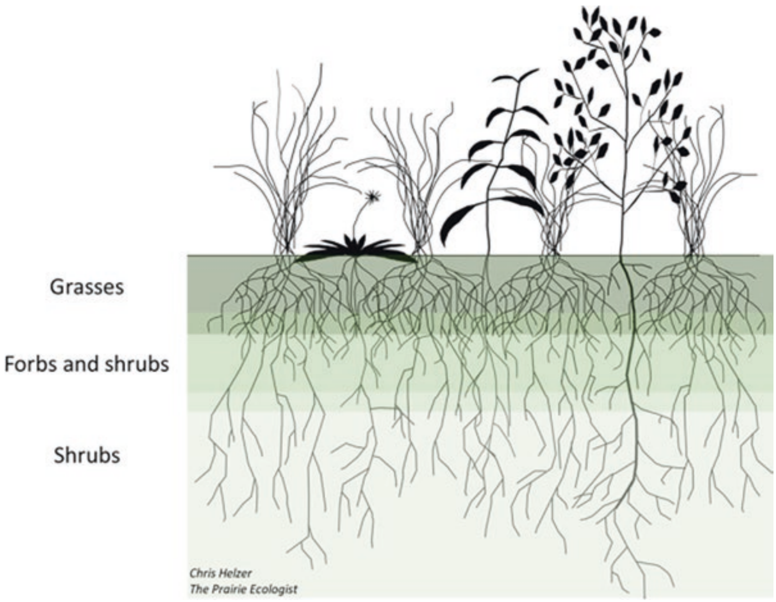

Fig. 9.13 Restoration efficacy is greatly increased when both above and below ground plant growth is considered in the planning process. (Figure credit Helzer, 2019)

natural conditions, but this is often messy and not often obvious at first glance. Planning is important in the restoration process, but having the flexibility to create or modify a site to be suitable for growing plants means that practice does not always adhere to the same strict directives as found in engineered road design. Most roadside settings require some level of compaction and are often completed with a final task termed “track-walking.” Track-walking is a common roadside construction technique that uses excavator treads to compact the ground post-construction. Reaching some balance of roadside compaction and the need to allow for spaces in the topsoil for root growth can be a major difference and practical barrier between the engineered road contract and the site preparation needed for revegetation. Opening communications regarding the finding of a common solution or a compromise led to leaving some areas compacted and some areas roughened with uneven surfaces.

Site preparation is one of the most important contributors to successful revegetation of native seed and containerized plants. In addition to soil compaction, one of the greatest difficulties facing roadside native plant seed mixes is simply the challenge of their placement on the roadside. Because of the mix of roadside maintenance operations and the need for snow management treatment such as salts and sands, plus the sheer number of vehicles acting as vectors for seeds of invasive species, the challenges to native plant revegetation can compound. To help mitigate these challenges, many native seed mixes are chosen because they are already adapted to growing in a harsh site or are present and adjacent to invasive plants, showing they are capable of resisting, at some level, invasive plant competition and external disruption. These additional challenges of site conditions and aggressive weed competition can prove to be problematic in some areas, and it may take some additional research and finessing of techniques to determine what species are best suited to an individual site. Mixing and matching grasses with different root structures, as well as using forbs to reach deeper into the soil profile or into rocky cracks, are some of the approaches used on the challenging site conditions of this project.

Allowing the strengths of each agency to assume design control at various stages through constant coordination and communication was an important element in the success of this project. Coordination throughout the planning, construction, and planting phases maximized the efficiency of work practices, reduced the inherent cost of each phase, and allowed for open consideration of new opportunities. The planning phase utilized agency direction, natural resource goals, financial opportunities, and established common goals. Various portions of the project were often at different stages, sometimes producing a large and valuable amount of material from rock, soil, and trees that was used for creating wildlife structures and producing shredded woody material as mulch for erosion protection on vulnerable slopes and for amending the soil in planting sites. Positioning such material on the landscape required staging, planning, and forecasting. In theory, a single contract to perform all these tasks would appear to be the easiest route, but in practice not every contractor will have the expertise to support each the complex resource goal involved in this project, and sometimes separating specific tasks can minimize risks. On this project, WSDOT managed all the major construction contracts and engineering for the larger habitat structures. A secondary contract managed by the USFS allowed for other types of structures to be built that required designs and labor on a smaller scale and are more focused on microhabitat creation and enhancement. Finding experienced native plant experts, to propagate and outplant native plant species suitable for a variety of habitats, is a critical component and key to project success. Every aspect of the project needed to be considered and integrated, from the construction phase and the adaptive engineering of habitat structures and microhabitat development, to the revegetation and monitoring phase. The USFS had both the botanical knowledge and the correct infrastructure to support the propagation of native plant materials through their specialized restoration contracts and experienced restoration practitioners.

Key Features

A multidisciplinary team assisted in designing the key features of the wildlife structures and habitat enhancement work and provided a restoration framework that helped achieve a more successful and holistic management structure. Significant involvement of a hydrologist who helped to develop a variety of hydrologic connectivity structures, and research, provided by the local university, established the need for a variety of microhabitats to suit both large roving carnivores and low mobility species. A vegetation team consisting of a horticulturist, landscape architect, and several botanists determined the mix of native plant species that would be best suited for the area. These plants needed to provide the best cover for both pollinators and herbivores while simultaneously competing against invasive plants. Key phases of this project incorporated many of the same concerns and barriers, and as solutions were identified and appropriate approaches were generated, these issues became informing elements of practice and contributed to a solid foundation of trust and mutual respect and a lasting partnership. All stakeholders were encouraged to provide feedback and had a voice in the process at all stages. These discussions, reflecting on what was successful and what was not so useful, contributed to better planning for future I-90 phases. The development of future projects will be facilitated by the increased collaboration, trust, and learned adaptive management resulting from this work.

The benefit, emerging from a project that spans many years over one location, incudes the ability for the construction team to monitor and adapt practices to meet new situations. During such an extended project, each planting and successful plant establishment, in both the nursery setting and field conditions, is carefully considered and adapted for future use. Native species prevalent over this landscape, referred to as workhorse species, became important components of the planting palette. Prior accounting for multiple successional periods aided in developing the planting mix. Using a diverse mix of locally adapted plants from both cuttings and seeds at a relatively heavy density (8640 plants/ha) allowed some natural selection to proceed on its own, increasing the resiliency of the site vegetation. Understanding the aridity differences across the project also aided in species selection as well as reference locations. The concept of assisted migration and the purposeful placement of plant species based on predictions from climate change models often arose during restoration efforts. Selecting versatile species known to have a broader range and which can thrive in a variety of environments allows them to meet short- and long-term project goals as well as uncertainties associated with changing climates and environmental conditions. Other important food and pollinator species were selected, albeit at lower rates, even if their survivability is unclear but were introduced to contribute to overall diversity. In addition, planning for monitoring and contingency planting, seeding, and several years of weed control not only allowed for adaptive actions but contributed to as well the understanding of site conditions not foreseen during the planning process.

What Worked and What Did Not

Part of the planting success was the strategy to select a wide range of plant species with differing ecosystem functions and root characteristics that would succeed in a variety of site conditions. Using topography, trees, rocks, and water, several habitats were created in the project to help support wildlife objectives as well as increase vegetation resiliency. For example, as revegetation proceeded throughout the footprint of the project area, declining grass was noted in areas that were particularly higher in rock content. This led to creation of a seed mix that contained a higher representation of forb species with better root systems for thin and rocky soil conditions. Successful seed mixes typically consisted of five to seven grass species and two to four forb species and incorporated one to two shrub species. In this respect, changing the ratio of grass-forb species increased the success rate in less organic-dominated soils. The flowering plant broadleaf lupine (*Lupinus latifolia*) successfully vegetated a steep, rocky slope that was recontoured, creating planting benches and pockets, and will help recovery by providing nitrogen to the nutrient-poor soils (Fig. 9.14). The area was also decompacted, and project-generated woody shreds were incorporated into the soil. The dense forb cover created a visually pleasing pollinator slope that was seen to provide shelter for a young fawn after the first year (Fig. 9.15). Other microhabitats, such as wetland areas, used primarily emergent species, both seed and plugs, as well as shrub species adapted to wetter conditions.

Container sizes were another important consideration in the planning process. Midsized container sizes (655 cm^3) of selected species such as baldhip rose (*Rosa gymnocarpa*) and low Oregon grape (*Mahonia nervosa*) experienced significant removal by wildlife, such as elk pulling the plantings out of the ground. Larger container sizes (2.8 l pots) remained in the ground and intact, although the plants were still heavily browsed. A useful species found to be resilient after browsing was red-osier dogwood (*Cornus stolonifera*), where surviving stock had aboveground vegetation browsed in the fall but resprouted from the base of the stem throughout the spring and summer. Several species did not propagate well in a nursery setting, such as Oregon boxwood (*Paxistima myrsinites*), or propagated inconsistently over

Figs. 9.14 and 9.15 Restored staging area after decompaction, incorporated shredded woody material, and seeded with native seed mix at a higher forb seed to grass seed ratio. (Photo credit Kelly Evans). Fawn hiding in lupine. (Photo credit Danielle Shurlow)

the years, such as Sitka mountain ash (*Sorbus sitchensis*). Some species struggled for the first few years in a project site, such as western sword fern (*Polystichum munitum*) and even vine maple (*Acer circinatum*), a species widely distributed across the forest.

Site preparation is a key feature to the success of any planting project, and this project was no exception. In every planting site, decompaction following the completion of the engineered structure proved to be a critical component for plant establishment. In an earlier phase, the seed mix struggled to establish in medians and roadsides. Although a layer of topsoil and mulch had been placed post-construction, the compaction below this layer was so great that a shovel could not penetrate through the top layers. This was a direct result of standard compaction from track-walking post-construction. In addition, outsourced topsoil, and mulch, increased the rate of non-native species not commonly documented in the area and suddenly appearing in project sites. As a result, local wood products became specified in the engineering contract to generate the shredded wood mulch used across the entire project. This did not become a costly operation, and the shredded material proved to be an economical benefit for use on the project site during high precipitation events to protect active sites from erosion. At the end of the construction season, incorporation of the material into the soil provided slope stabilization and barrier against invasive and non-native plants present in the project area.

An additional step not easily designed in a normal engineered contract was introducing site complexity. Increasing complexity can be achieved through slope-shaping undulations, berms, and depressions together with the addition of other natural features such as rocks and large woody debris to create microsites. These microsites can provide shelter for vegetation as well as for a variety of terrestrial and aerial species. Large debris can be buried on the surface or anchored, depending on the proximity to moving water. These microhabitats provide both habitat for a variety of species and an area for adjacent seeds to be cached as microbial hotspots (Figs. 9.16 and 9.17).

Major Outcomes

Bringing together a variety of stakeholders and interested parties, all having differing interests and sometimes competing needs, created a restoration project that met several goals that both the local community and biologists valued collectively. The building of trust through early and continual communication, combined with the obvious visual success of the work, created a long-lasting relationship that has continued not only on this project but with other projects nearby. In addition, other agencies recognized that the Forest Service has dedicated time and infrastructure to support skilled restoration specialists and the necessary tools to facilitate this type of specialized restoration work. This contrasts with work presented by Gibson-Roy et al. (2021) (Inset 9.4), which describes a successful large-scale roadside restoration project in Australia, but in this case, there was no recognized methodology for sustained support due to lack of infrastructure.

Figs. 9.16 and 9.17 Native seed recruitment and planted shrubs thrive within the microsite additions adjacent to a wildlife undercrossing. Wood and rock debris used to create wildlife habitat piles as well as shelter and microhabitats for vegetation. (Photo credit Helen Lau)

These types of native restoration projects have continued to encourage and justify their innate worth through added benefits to the pollinator community through increased species diversity. The increased interest and concern over pollinator decline (Kluser & Peduzzi, 2007; Potts et al., 2010), as well as the effects of climate change, have brought increased attention to the choice of seedings and plantings on the project site (Fig. 9.18).

Inset 9.4: The Glenelg Highway Restoration Project

Australia's temperate south-eastern lowland native grasslands are among its most threatened plant communities and records shown that, in Victoria for example, up to 99% of high-quality grasslands have been lost (Williams et al., 2015). Today, diverse grasslands are rare and are seldom found without non-indigenous components. A confounding restoration problem is that the natural return of native grasslands to pre-agricultural landscapes is constrained by a lack of source populations and increased soil nutrient levels (Dorrough & Scroggie, 2008). Under such settings, major reconstruction is one of the few viable ways of returning native grasslands to these landscapes.

The large arable eastern and southwestern road networks crossing Australia are dominated by deliberately planted or self-colonising exotic grass species. To restore two areas near the Glenelg Highway in Victoria, removal of two historic roadside tree plantations composed of non-endemic native trees and shrubs and their revegetation was recommended. This action was deemed necessary to connect several

Fig. 9.18 Interstate Highway 90 roadside restoration and wildlife overcrossing. (Photo credit Helen Lau)

high-quality remnant grassland patches which were found in the area. Because of the extent of disturbance to the highway environs, it was decided to employ direct seeding techniques following topsoil removal to treat weed banks. The sowing of high diversity grassland seed mixtures to this scalped land was considered a novel approach at the time. To ensure appropriate source selection and to reduce impacts on the already low availability of local populations, discrete amounts of seed were propagated at a seed production area (SPA) located on a farm in the project region. In the project, 100–150 mm of topsoil was removed from the treatment area, and then left untouched until the Spring, when it was sprayed with a follow-up herbicide. Subsequently, 50 species were surface broadcast as a ‘seed curtain’ and lightly press-rolled at a 75 kg per hectare sowing rate. In addition, salvage of several other native plants, such as *Eutaxia* and *Dianella*, supplemented the species mix.

Monitoring in the Spring of 2010 showed a greater than 95% survival rate of sown species and 85% survival of the salvaged species, giving a species composition that included a number of threatened listed species such as button wrinklewort, clover glycine, hoary sunray and yam daisy. These early outcomes, in combination with later results, gave project managers encouragement to support the combined use of soil removal and direct seeding. Both of the treated sites continued to require minimal intervention and outbreaks of non-native plants were managed by periodic spot spraying. In 2017, monitoring revealed both restorations also remained resilient to weed invasion with weed cover less than 2% at both sites. Sown species had expanded their range outside the sowing zones and interaction with fauna was increasingly common across both sites. This was seen in particular with species of reptiles and insects, showing that the restorations were acting as a functional habitat, an outcome seldom achieved in Australian restorations.

However, despite these many positive outcomes, similar approaches have not been taken up by Victorian or other Australian State Road authorities. This poor reaction is likely because Australian landscape regulators do not often support active restoration by road authorities or rural landholders, even though it is recognised that high biomass weedy roadsides represent significant fire risks to rural communities. Indeed, for this reason alone, restoring roadside vegetation to low biomass, stress tolerant, native vegetation, could save management agencies money. This lack of impetus has ensured that there is very poor sector capacity in the areas of seed production and restoration services to undertake works of this magnitude (Gibson-Roy et al., 2021).

–Dr. Paul Gibson-Roy (Manager Ecological Restoration)

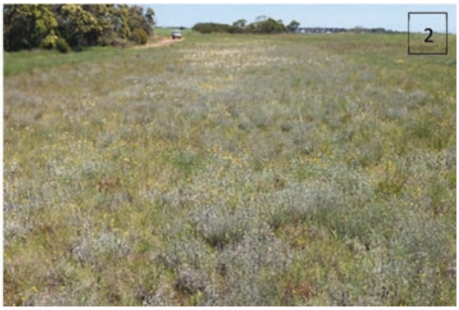

Photo panel caption: Monoculture of woody non-native species (1) and increased native species following successful roadside revegetation (2)

Mimicking natural landscapes with undulations created in the topography, microhabitats formed in the interstitial spaces between plantings, and creating habitat structures such as woody material and rock piles assisted in increasing pollinator habitat in the project area. Expanding the site complexity allows for passive restoration by other species not suited for plant propagation in nurseries. This emphasizes the need for good site preparation, since the desired future condition and vegetation of a site needs to attract and support fauna of all types. The long-term goal is to set up self-sustaining vegetation sites by (i) careful placement of appropriate components, (ii) being intentional with the species selections, (iii) including diversity to combat effects due to natural causes such as climate and hydrological changes, and (iv) allowing a natural restoration trajectory with plant succession. Returning to these sites in future years and observing the condition of the restored species together with species introduced by natural regeneration indicate the positive mechanisms of recovery of these sites and are indicative of restoration success. One major goal to restore this site was not only to introduce surviving native plant cover but to prime the site for natural succession and provide the necessary components for long-term success via self-sustaining natural regeneration.

Case Study 3: Nestucca River Road Access Project

Rationale and Strategy

The Nestucca River originates in the Northern Oregon Coast Range and is one of the state's major free-flowing coastal rivers, flowing for 80 km over an almost continuous bed of rock to create riffles, numerous white-water cascades, and occasional clear pools (Fig. 9.19). Due to its scenic value, it has long been popular with

Fig. 9.19 Nestucca River in western Oregon is a candidate for classification in the Wild and Scenic River System. (Photo credit Lee Riley)

recreation enthusiasts and sightseers. The Nestucca River is recognized as a high-quality anadromous fish stream and contributes significantly to wild fish production on the Northern Oregon Coast, with species including Oregon coastal coho salmon, steelhead, cutthroat trout, and the Pacific lamprey. The area also provides important foraging habitat for bald eagles and contains suitable habitat for the spotted owl and marbled murrelet, both of which are listed as threatened under the Federal Endangered Species Act. A 24.6 km segment of the Nestucca River was found suitable for designation as a component of the National Wild and Scenic River System and has tentatively been classified as a recreational river area. The National Wild and Scenic Rivers System was created by the Congress in 1968 and provides for protection of selected rivers of the USA that possess outstandingly remarkable values, including protecting and/or enhancing the free-flowing condition, water quality, and other aspects of suitable rivers.

During the past few decades, the area has seen an increase in visitation and recreation by the public. Access to the area is predominantly through the Nestucca River Access Road, which was designated to be a National Back Country Byway by the US Department of the Interior Bureau of Land Management (BLM) in 1989, specifically because of its high scenic value. The Nestucca Back Country Byway was originally designed and constructed in the 1950s and 1960s, almost exclusively to give access to, and the hauling of, harvested timber. The road was constructed prior to enactment of the Clean Water Act and the Endangered Species Act and therefore was not necessarily designed to account for high floodwater passage and/or fish passage in streams over which the road passes. Paving of this original gravel-surfaced road was done in a piecemeal fashion, progressing from both the East end and West end, with a section in the middle remaining as a graveled surface.

In 2014, the BLM, in cooperation with the FHWA, determined the need to upgrade the remaining 4.3 km of unpaved road. The upgrade would (i) provide visitors with a safe, scenic travel experience and greater access to public lands, (ii) minimize current and potential future adverse impacts to the environment by improving water quality and hydrologic function and reducing erosion, and (iii) assist in protecting, managing, and conserving federally listed threatened species and their habitats. In 2016, USFS restoration specialists began coordinating the native plant restoration on this project by attending meetings with cooperating agencies and contractors and developing site-specific restoration plans.

Concerns and Barriers

The 4.3 km gravel-surfaced section of the Nestucca Access Road was a safety concern for bicycles, motorcycles, and small passenger cars. The gravel surface limited its use as a commercial haul route during the wet season and produced dust and sediment during the dry season. Undersized and failing culverts became plugged with woody debris, threatening downstream water quality and fisheries resources. The entire project was situated in steep terrain with unstable soils and several wetlands, the latter causing numerous slump areas along the road. In addition, there were visual and safety concerns resulting from construction activity when viewed from the river as well as a small campground located within the project area.

Although this project was small, it involved a variety of complicated challenges, both planned and unplanned. The extreme steepness of the surrounding canyon walls required cut slopes to be compacted to avoid additional disturbance to the existing forest and prevent soil erosion into the river. Areas below the road required construction of several hundred meters of mechanically stabilized earthen (MSE) walls to maintain road stability, including the area above the small campground. Replacing culverts and installing new aquatic organism passages (AOPs) required much larger disturbance footprints than originally planned due to the instability of the soil and the lack of bedrock for placement.

The presence of designated wetlands along the road corridor presented an additional challenge to project completion. Under Oregon Department of State Lands (ORDSL) regulations, reestablishment of preexisting contours, function, and vegetation must occur in any wetland during and following construction.

The entire project was located on federal (BLM) lands, requiring the use of genetically appropriate, locally adapted native plant materials from seed and cuttings. These were required to (i) match the existing environment as much as possible based on reference sites, (ii) maintain or increase soil stability and reduce surface soil erosion, (iii) provide esthetic enhancement where appropriate, and (iv) assist cooperating agencies in meeting regulatory compliance. Plant material was needed to provide visual screening of vertical MSE walls, provide vegetative cover for fish habitat at aquatic organism passages, and restore the vegetation and functionality of all disturbed wetland and riparian areas.

As with all restoration projects, finding, mapping, and collecting appropriate seed and vegetative material had to occur prior to construction to allow time for plant production and to plan appropriate seed mixes to apply to disturbed areas. USFS restoration specialists collaborated with botanists and land managers from the BLM to develop and implement site-specific revegetation plans and species lists based on surrounding reference sites and knowledge of plant production.

Key Features

In the fall of 2020, USFS restoration specialists implemented many of the planned restoration activities. However, several unexpected modifications were necessary based on the final construction as well as the changing needs of the partner agencies.

Steep, compacted slopes in areas of high rainfall provide one of the biggest challenges to revegetation in any construction project. Hydroseeding, with the proper mix of perennial grasses and forbs at the proper time of year, is often the only tool available for applying plant material to stabilize these slopes. Hydroseeding requires large amounts of seed, often grasses and forbs, applied in a broad swath to the landscape. Establishing shrub and tree species can be a challenge because seed from these species is often in limited supply and planting can require special techniques, such as rappelling down slopes to maintain position while planting (Fig. 9.20).

Fig. 9.20 Rappelling to plant on steep slopes on the Nestucca River Byway project. (Photo credit Haley Smith)

A combination of these methods was used on this project. The slopes were sprayed with a hydroseed mixture of forb and grass seeds and then planted with a mix of container-grown ferns, flowering forbs, and shrubs to ensure slope stability and provide visual enhancement. In areas that proved too steep and rocky to warrant planting, a new method was attempted for applying valuable seeds to the slopes. Small balls of fiber and tackifier were filled with shrub and tree seed and hand-applied via wrist rockets, small handheld y-shaped devices with elastic strung between the prongs for shooting small projectiles, to slopes to control the placement of those species (Fig. 9.21). We note that drone technology has now advanced to the point where it is being considered for future seed ball applications.

Visual screening of MSE walls often presents a challenge depending on their construction. Previous screening efforts have been successful when USFS restoration specialists worked closely with the cooperating agencies and contractors to design stepped, plantable walls or walls containing seed-impregnated biodegradable fiber (Fig. 9.22). Neither of these methods was utilized on this project, but screening of these walls from the river, as well as the small campground, was required. Rapidly growing tree and shrub species were planted at the base of the walls. The BLM determined that further screening was necessary at the campground for the visual objective and a safety precaution to prevent climbing on the wall.

The MSE wall was constructed using only large rocks, and thus did not provide a plantable surface. As an experimental alternative, USFS restoration specialists treated the wall with locally collected moss and fern spores mixed with buttermilk,

Fig. 9.21 Wrist-rocket propelled seed balls. (Photo credit Erin Holiman)

wood fiber mulch, and tackifier. The mixture was applied using a small hydroseeder in the late fall, a process we have named "hydromossing" (Fig. 9.23).

Designated wetland areas throughout the project had been maintained by the construction of rock check dams and new culverts. Denuded of vegetation, the risk of slumping of both the road and surrounding areas remained. A variety of containerized sedges, rushes, and wetland grasses were grown from native seed collected from the area. These containerized plants were then installed both above waterline and within the wetland areas. The intention was to prevent erosion and return the areas to self-sustaining and functioning wetlands, as required under ORDSL permitting regulations.

Fig. 9.22 Stepped mechanically stabilized earth wall containing seed-impregnated biodegradable fiber and planted with native shrubs in 2016 and revegetated wall in 2021. (Photo credits Lee Riley)

Fig. 9.23 "Hydromoss" application to mechanically stabilized earth wall. (Photo credit Schuyler Hamilton)

What Worked and What Did Not

The Nestucca River Access Road project was a complicated project from the beginning. Although the project was scheduled to begin in 2016/2017, the project was delayed until construction was finally initiated in 2019. During that time, personnel changes occurred in all involved agencies, including BLM, FHWA, and USFS restoration specialists, as well as the construction contractor, which meant many lines of communication were confused or lost. Construction delays due to unforeseen complications also delayed completion of the project and all related restoration activities. Consequences of these delays included loss of project seed lots, re-propagation of restoration plants, and the unfortunate feelings of mistrust between agencies. No matter the size of the project, communication with all agencies and contractors from the beginning of the project through to the final restoration implementation and monitoring is the key to success. When construction of this project was nearing completion, there was a total breakdown in trust between the responsible agencies. However, the restoration team was able to act as mediator and facilitator in this situation, resulting in renewed communication, an understanding of the needs of the cooperating agencies, and completion of planned restoration activities.

Major Outcomes

The successes or failures of this project's restoration activities remain to be determined, but the hydromossing of the rock wall certainly looked promising 1 year after implementation (Fig. 9.24). The wetland areas were beginning to be filled with appropriate vegetation and appear to already be functioning in their natural state. These areas and the remaining restoration sites will be monitored for three additional years to determine which activities were successful and what, if any, additional work will be necessary.

Fig. 9.24 Mechanically stabilized earth wall 1 year following hydromossing. (Photo credit Lee Riley)

Conclusions and Implications for Practice

These high impact projects are producing measurable outcomes, demonstrating how the innovative use of native vegetation can meet regulatory requirements as well as achieve ambitious restoration priorities to enhance esthetic values and restore ecological function to highly disturbed road systems and their surrounding environments. We suggest that the lessons learned from the best management practices developed from these roadside restoration projects can also be applied to any project involving severe disturbance. These disturbances might be due to powerline installations, pipeline corridors, recreational trail reroutes, restoration of abandoned agricultural fields and abandoned mine sites, as well as natural phenomena such as landslides, floods, and wildfires.

It is important to note that the supporting policies and regulations, which were put in place in the USA, have set the framework for the use of locally adapted native plants for restoration. The development of seed zones, which helps to avoid maladaptation in plants transferred to new locations, was the direct result of such guidance and requirements. The beneficial downstream effects of coordinated policy include technology transfer, increased capacity, expanded knowledge, economies of scale in seed and plant production, cost savings, increased predictability and stability in seed inventory, economic benefit to private growers, and a vast improvement of project success.

Further, it is emphasized that the use of genetically appropriate and locally adapted plant propagules is paramount to the success of the case studies presented here. Selecting plant propagules from populations within the seed zone ensures that they will be well-adapted to the site's climatic conditions, thereby drastically increasing revegetation success. Understanding the native plant species composition prior to disturbance allows restoration specialists to identify plant species that will do well in the new environment developed by the project construction. Selecting plant species best suited for the future conditions can be as simple as finding plants currently growing near the project in similar conditions and then collecting seeds and cuttings from them.

These case studies are also a reminder that the restoration of significantly disturbed sites requires a high degree of planning and coordination. Planning efforts described in these case studies spanned years and involved a multitude of interest groups, agencies, and disciplines. Unfortunately, dedication of such time and focused attention on revegetation efforts is not a common practice. In fact, revegetation is often an afterthought rather than part of the foundation upon which restoration activities are built, and it needs to be recognized that native seed mixes and containerized native plants often take a minimum of 3 years to prepare for a project. However, with proper planning, restoration efforts can facilitate the mitigation of a construction project's environmental effects, visually improve the result of the disturbance, and assist with meeting regulatory agency obligations. In addition, by developing and leveraging multidisciplinary teams, the overall restoration effort will benefit from a more holistic strategy, allowing the team to look through the lens of soil composition and chemistry, climate, nutrients, plants, animals (including pollinators), societal expectations, recreation, the travelling public's safety, and other parallel considerations.

The ability to scale restoration work is another direct benefit of thorough planning. By learning more about the project needs, objectives, and timelines, restoration specialists gain better understanding of how best to implement their project. For example, if funding is limited or expected to come at different times, a phased approach might be best. In this way, discrete tasks, on which future work is dependent, can be implemented as funds are made available. In these situations, it is also beneficial to consider the successfully installed restoration plants as future seed sources. If the restoration specialist is unable to install as many individual plants as desired, it would be wise to select those that (i) have heavy seed set, (ii) expel their seeds ballistically, or (iii) have seeds that are attractive to small mammals. In this way, the seeds will be naturally distributed to cover more ground and produce additional plants without the need for further intervention. Similarly, seed mixes and containerized plant calculator spreadsheets (http://nativerevegetation.org/) allow restoration specialists to compose appropriate assemblages that can be used to cover up any sized area.

We conclude the chapter by noting some key implications that have become evident from these three case studies and which we believe provide useful guidance for practitioners dealing with other forms of severe landscape disturbance. These observations are as follows:

(i) Roadside corridors present significant yet often overlooked opportunities for achieving ambitious conservation and restoration objectives. Innovative use of native vegetation can meet both safety and regulatory requirements while also providing esthetic values and restoration of ecological function to highly disturbed road systems.
(ii) Successful restoration of highly disturbed roadsides requires extensive planning and coordination. Communication with all agencies and contractors from the beginning of the project through to the final restoration implementation and monitoring is a key to success, regardless of project size or complexity.
(iii) Early planning for seed supplies based on site evaluation and examination of reference areas enables procurement of adequate quantities of seed of adapted species and seed sources.
(iv) Use of a seed zone framework for collecting and sourcing native plant material can aid in planning efforts and improve project success through enhanced plant resiliency and adaptability to site conditions.
(v) Decoupling revegetation activities from construction contracts avoids duplication of effort and provides restoration specialists greater control over time-sensitive and biologically driven activities such as seeding and planting.
(vi) Selecting contractors based on criteria such as past performance, experience, and knowledge in addition to price will improve restoration success and planning for monitoring, contingency planting, and seeding. Several years of monitoring and weed control allows for adaptive actions, as well as gaining an understanding of site conditions not foreseen during the planning process.

(vii) Best management practices (BMPs) used on roadside restoration projects can be applied to other types of disturbed areas such as powerline corridors and recreational trail reroutes, as well as the restoration of mines and abandoned agricultural fields.

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Chapter 10
Issues Embedded in the Human Context of Urban Landscape Restoration

Adrian Marshall, Bruce Clarkson, and James Hitchmough

Contents

Introduction

History and Progress of Urban Landscaping

It is understood that the practice of urban restoration, which is the restoration of ecosystem functions that have been disturbed by anthropogenic actions, has its roots in two parallel concerns. The first came from the influential British landscape

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architectural movement of the mid-eighteenth century, which embraced and celebrated the picturesque; the second was the contemporaneous public urban health concerns which were brought on by the appalling effects of the Industrial Revolution.

The picturesque movement, championed by Capability Brown, favoured a naturalistic landscape, where an orchestrated presentation of nature, with wild, rough, or sublime elements was contrasted with framed views and a sense of cultivation (Ross, 1987). At first, this was very much the domain of the rural landed gentry, but as cities and urban planning developed, the picturesque was introduced into urban areas, and especially into the larger parks that already contained substantial natural elements, such as dense woodland. At the practical level, to achieve these changes, heavy engineering, involving significant alteration of hydrology and topography, was needed to address this desire to improve existing parklands (Finch & Woudstra, 2020).

At the same time, the industrial revolution was producing urban environments of astounding misery, where poverty, exploitation, poor sanitation, and intense pollution drove the spread of disease, death, and social injustice. The theory of the day had it that unhealthy miasmatic vapours carried diseases and it was thought that trees absorbed these vapours. It was here that the movement for creating large tree-filled parks was born, and this was considered to be a vital social reform as it created the 'lungs of the city' (Crompton, 2017).

However, improving sanitation, which was an urgent need for reclaiming the cities, required the framing of strict guiding laws. Without such legislation, which directed the development of sewerage works, the power of industrial greed would have continued to override compassion, and nothing would have changed (Fee & Brown, 2005). Such laws have been present in one form or another throughout urban civilisation, but the social changes brought on by the Industrial Revolution were in themselves revolutionary. As a consequence, the treatment of sewerage and the better management of stormwater drainage have led to significant improvements in aquatic systems across the globe.

In addition to the urban expansion driven by the Industrial Revolution, mass transport systems based on railways and private transportation systems involving cars and busses, led to the incorporation of villages and townships into the larger cities. In this consolidation process, significant larger landscapes, for instance Hampstead Heath, eventually became incorporated into the urban periphery of larger cities such as London. In the eighteenth century, this started a movement, in particular in the United States, to capture and preserve threatened urban scenic landscapes in the form of large regional parks (Carr, 1999). It is noted that this movement did not include the over-engineering and significant remodelling of the landscape that characterised the more ambitious picturesque approach.

It was sometime later that the idea that cities had their own ecologies arose, implying that they were not simply degraded and corrupted 'natural' ecologies. This idea followed from the massive destruction wrought by World Wars One and Two and gave birth to innovative studies examining the return of life, both human and plant-based, to devasted urban landscapes (Fitter, 1945; Sukopp, 2008). Further, increasing environmental awareness in the 1960s and 1970s led to key legislative changes and planning controls that continued on from the initial Industrial Revolution–inspired laws on sanitation and hygiene. In this respect, America's 1972 Clean Water Act has driven much of the restoration of ecological functions across that nation, although much remains to be done (Adler, 2010).

The almost tectonic shifts caused by the growth of industry led inevitably to the appearance of brownfields, which required large-scale post-industrial restoration (Hunter, 2014). The importance of urban ecology became firmly established, with ecological function increasingly becoming part of the narrative for urban greening projects, regardless of their size or nature. Now with climate change manifestly upon us, the arguments are rapidly shifting to how we can sensibly use ecology to build urban resilience (Egerer et al., 2021).

In this respect, cities are looking at unique and more nimble approaches to restoration. Whilst new land is expensive, the pressure for positive environmental change is significant. This has led to retrofitting public open spaces with new functionality, for example installing wetlands, birdboxes, bee hotels, and fish ladders. Changes to management regimes mean that fallen deadwood is retained in situ, lawns are mown less frequently, and irrigation used more sparingly. The exotic is being replaced with the native, showing that it is recognised that urban restoration must occur across all physical and temporal scales.

The magnitude of this change is nowhere better exemplified than by the focus on restoration of streets, facades, and roofs. These are uniquely urban possibilities, where the patches of green in urban road verges, albeit individually small, occur in such large numbers that they can constitute almost a third of a city's green space (Marshall et al., 2019). The City of Melbourne's streetscape biodiversity program is a significant example, where councils have piloted the use of a carefully curated and partly indigenous planting palette to restore native pollinator habitat and function in their urban streets (Tan et al., 2022), while focusing on species that can survive the harsh conditions of the urban streetscape. The increasing introduction of green roofs and facades, known as 'living architecture', is another exciting urban phenomenon. While these areas are often expensive, constrained, windswept, and exposed, as well as being isolated from natural terrestrial ecosystems, they nevertheless can significantly increase the area of vegetation present within the urban envelope and can work to turn public attention towards the (re-)greening of the environments in which we live (Besir & Cuce, 2018; Williams et al., 2014).

It is worth noting that some cities are now seen to be shrinking. Whilst Detroit, USA, is often quoted in this respect, this phenomenon is occurring in many countries, including China (Long & Wu, 2016). However, notwithstanding the notion that shrinking cities seem to be a blessing for their potential to facilitate urban restoration, it must be appreciated that this phenomenon is accompanied by profound social change, for example increased unemployment, which must also be factored into the overall ecological outcome (Sadler & Lafreniere, 2017).

Finally, urban restoration projects need to recognise and counter a number of persistent misconceptions about the potential for biodiversity conservation. It is important to recognise that: (i) small urban spaces can be as important as large areas, (ii) species can occupy unconventional habitats, (iii) creative design responses can produce significant conservation benefits, (iv) restoration will be most likely to succeed when it recognises future change in the urban context, (v) cities and towns are often biodiverse, and (vi) many threatened species can be found within cities (Soanes et al., 2019).

Recognising that the Urban Context is Social, Economic and Ecological

It is clearly important to appreciate that urban areas are generally characterised by (i) vastly altered water cycles, (ii) the massive clearing of the original vegetation, (iii) increased pollution of air and water, (iv) the interpolation of large areas of impermeable surface, and (v) raised average temperatures. In addition, existing vegetation is predominantly exotic, and there is continual human and transport movement across the entire cleared space (Parris, 2016).

Urban restoration projects therefore occur in the most human of contexts. Available open space is subject to multiple layers of ownership and governance, and restoration of landscape must compete with other land uses, such as sports fields and parking areas. Urban land areas are much more costly compared to equivalent areas of non-urban land, which not only makes sequestration of large portions of urban land for restoration prohibitively expensive but also means existing open space is constantly under intense pressure to be developed for economic gain. In many cases of applications to clear remnant plant matter, which would seem to be antithetical to the restoration movement, public and personal safety arguments are mounted, with emphasis for instance on ‘clear sight lines’ and lighting of areas at night.

Fortunately, there is a growing appreciation of the need for urban systems to work well ecologically. As in the times of the Industrial Revolution, people are again recognising that urban nature has essential human benefits, such as contributing to positive health and well-being, providing a sense of place, and meeting spiritual needs. It must be understood that, as a human construction, urban nature operates on a symbolic as well as functional level. The development of a project that changes people’s attitudes, or the instigation of a project that gives community members a sense of their independent agency in their capacity to affect change, is as important to the overall establishment of urban functionality as the project that restores ecological function. As a consequence, urban restoration projects are a hybrid organisation, navigating between nature and culture, and they increasingly have to meet demands for increased multifunctionality, to bring more healthy outcomes to more people.

Introduction to the Four Case Studies

An implication of the preceding discussion is that what constitutes a successful act of restoration in one urban setting is often subtly different from that obtained elsewhere. The four Case Studies given in this chapter give some sense of this complex issue of location specificity. The first Case Study, located in Waiwhakareke Natural Heritage Park on the outer fringes of Hamilton in New Zealand, began with a far-sighted understanding that its grazed farmland would one day become an integral part of the suburban residential fabric. Indeed, its transformation into kahikatea-pukatea forest, with its protected puna (springs) and the puna paru (the black iron-rich muds), would have been almost impossible if it had to begin from a traditional urban setting. However, clever and foresighted initial urban planning acted to preserve aspects of the original environment, and the project also benefitted from increased patronage due to its co-location with Hamilton Zoo. Ongoing community engagement has been the cornerstone of this restoration project's success. It was understood that whilst people can be seen as the problem, people are also the solution, and here they aggressively championed the values for which this Park stands.

Case Study 2 relates to urban development and infrastructure projects, which can offer opportunities for (relatively) large-scale restoration projects. These can be as varied as dockland redevelopments, the raising of rail lines, or the construction of new ring roads. In the case of the magnificent native meadow landscape of the Queen Elizabeth Olympic Park in England, this vast urban redevelopment project was an outcome of London hosting the 2012 Olympic Games. In the management of this project, the landscape was entirely reconstructed, the ground area was regraded, with new topsoil being brought in. Plantings were driven by an emphasis on intense visual display to brand the Olympic site, which led to a planting palette based on, but fundamentally altered from, the native meadows of the local area. The huge engagement that this project generated with the public offered an extraordinary opportunity to shift public attitudes towards a positive embrace of native wildflower meadows. The technical skill with which the plantings were created has meant weed presence has been kept to a minimum, ensuring the project's longevity in the inevitably tight economic conditions felt by all urban land managers.

The Case Studies of both Waiwhakareke Natural Heritage Park and Queen Elizabeth Olympic Park show the importance of fundamental research that can underpin and facilitate large-scale projects such as these. In both cases, the projects are strongly linked to long-term, stringently academic research programs, managed by universities.

Case Study 3, of the Australian Botanic Gardens Shepparton, suggests the importance of the botanic gardens as an institution for driving urban restoration, and highlights the fundamental social importance of community in achieving change. The role of good design is also fundamental to this project's success. Case Study 4, involving Zaryadye Park in central Moscow, is an even more radical departure from typical restoration projects. In this work, the plantings were designed to represent the four major biomes of Russia – steppe, tundra, wetland, and birch forest – in a limited space in the very centre of the capital city. It is fair to say that the power of this project is not necessarily what it achieves in terms of restoration per se, but rather how it has contributed to a change of mindset of the Russian people. It has brought a familiarity with Russian ecology to the people and has engendered a sense that such an intense focus on ecology can contribute to the fostering of a Russian local identity. It seems reasonable to assume that without this preparation of the metaphorical ground, the seeds of change through restoration will never prosper.

Case Study 1: Waiwhakareke Natural Heritage Park, New Zealand

Project Rationale(s) and Strategy(ies)

The Waiwhakareke Natural Heritage Park (WNHP), which is a Kirikiriroa/Hamilton flagship urban ecological restoration project, officially began in 2004 near the shore of the peat lake Waiwhakareke. It commenced with the planting of the first tree, a kahikatea (*Dacrycarpus dacrydioides*), which now stands almost 17 m tall. This represents about 1 m of height growth for every year of the project's life.

What sets this project apart from most, if not all, ecological restoration projects in New Zealand, is that this is a reconstruction from undamaged land, and its operation was underpinned by strong science. However, like any urban ecological restoration project, the journey has been bumpy and sometimes fraught with unforeseen difficulties. This account is focused on what has been learned during the reconstruction, and how this experience will help to achieve ongoing improvements. It is also to be hoped that the lessons learned may be useful for others contemplating similar projects in urban environments where there are similar conditions.

Key Project Features

The 65.5 ha WHNP is situated on the rapidly developing urban northern edge of Kirikiriroa/Hamilton, New Zealand's fourth largest city. In 2022, the Park is surrounded by a residential subdivision and the adjacent Hamilton Zoo, being set aside in 1975 by the farsighted council mayor of the time. It was then no more than degraded farmland with a eutrophied peat lake. In essence, it was an open space where, for some 35 years, grazing remained the predominant land use. Even now, small pockets of the Park are still used for grazing, which will cease when native plantings begin, an action which will initiate habitat restoration.

Foresight: Almost two decades after its inception, urban expansion has reached Waiwhakareke Natural Heritage Park. (CC0 1.0)

The WHNP was first mooted as a Living Museum Millennium Project and envisioned as being formed from a range of plant communities representing different regional flora of New Zealand. Despite external funding being initially rejected, a consortium of environmental groups, agencies, and individuals continued to lobby for the new Park. The concept evolved towards a public park which would restore and recreate the native plant and animal communities that once existed within Hamilton and, specifically, at the Park site.

Initial endorsement of the project came on 6 May 1998, and on 9 April 2003 the Hamilton City Council resolved to create a nature heritage park. By the Winter of 2004, concept plans had been approved and a small-scale lake margin planting began. The key partners in the project at this stage included the University of Waikato, Waikato Institute of Technology, the charity trust Nga Mana Toopu O Kirikiriroa (NaMTOK), Tui 2000 (a community group dedicated to bringing back the Tui, a native New Zealand bird, by the year 2000) and Hamilton City Council.

From the outset, the Park was conceived to involve a strong research focus, aiming to inform both the theory and practice of ecological restoration. In 2006, a rigorous and comprehensive management plan, charting the approach to restoration in the Park, was produced by the Parks and Gardens unit of Hamilton City Council, in conjunction with the University of Waikato Centre for Biodiversity and Ecology Research. This key document was founded on a body of scientific research on the ecology of the site, with cultural input from NaMTOK. Target ecosystems were identified and delineated (Clarkson et al., 2012; Clarkson & McQueen, 2004). Baseline monitoring was undertaken mostly under the aegis of the University of Waikato (2000–2005) and the Foundation for Research Science and Technology, of the Ministry of Science and Innovation, whilst the Ministry of Science, Innovation and Employment funded the research (2005–present). Monitoring expanded as the area of restoration plantings increased and now comprises 25 permanent plots (Farnworth et al., 2021). The project trialled a range of planting densities and compositions, with initial planting densities aimed at quickly gaining canopy control and suppressing weeds. Enrichment planting became important through developmental years 10–15, as pioneer shrubs and small trees lost their vigour and canopy gaps developed. These results are summarised in our review paper (Wallace & Clarkson, 2019).

It is relevant to note that a GIS-based planting selection tool, incorporating 235 plant species, has been developed to aid planning of new restoration plantings. For this work, an independent technical advisory group provides ongoing advice as required.

Community governance of the project has been vital to its success from its inception and continues today through the Waiwhakareke Advisory Group, which works with the Hamilton City Council. Restoration work is undertaken as part of normal Hamilton City Council operations, but for this project, there was major community input in terms of voluntary labour and fundraising for plants. Arbour Day, which has been celebrated every year since the first planting in 2004, is always a most significant annual event, contributing some $NZ 67,000 and, even more importantly, being the main driver for community buy-in and involvement. The value of building social cohesion, awareness, and community support which has been developed as part of the work, cannot be overstated. In 2019, 1800 people, mostly school students, attended and planted 28,000 plants over 3 ha in 3 hours, a feat only possible because of several days of military-style planning prior. Other critical elements of the community involvement have been (i) the Community Planting Officer role funded by Hamilton City Council, (ii) the Friends of Waiwhakareke volunteer planting and maintenance group, (iii) the Potters' Group, which raises eco-sourced plants to supplement those grown by the Hamilton City Council nurseries, (iv) commercial suppliers and (v) other community-oriented plant growers. The project is now at the stage where children and grandchildren of those who participated in earlier Arbour Days are attending the annual event.

Bruce Clarkson hard at work during the annual Arbour Day planting, a celebration and a product of intensive planning and enormous community engagement. (Photo © Kate Monahan)

16 November 2019 saw the official opening of the Park to the public. Until then, the focus had necessarily been on reconstructing the indigenous habitat needed to bring back the indigenous biodiversity. Because grazing was used to prevent weed invasion in areas where planting had still not occurred, entry needed to be restricted and managed to be compatible with the requirements of a working farm. Following the opening, Hamilton Council voted to commit to a multi-million-dollar development, in conjunction with Hamilton Zoo, to upgrade public facilities. These included a shared entrance way and parking area, a public café and toilets, and a viewing tower. Also planned for this work is a new education and function centre.

Major Project Outcomes

Today, some 40 ha of indigenous plantings have been established, the peat lake is rapidly improving in water quality, the catchment is being reclothed in native forest, and the facilities to enable the people of Kirikiriroa/Hamilton to experience indigenous nature in their own backyard have been installed. The initial vision for the project is clearly becoming realised. The remaining goal for this park is the construction of a predator-proof fence, the feasibility, costs, and benefits of which are currently being considered.

Restoration has brought rich biodiversity to the lake's margin. (CC0 1.0)

Achieving and retaining political and funding support for the project have been the two most significant hurdles confounding the Park's success. As is typical of many projects, the early stages were a time of enormous enthusiasm and good progress, and from 2004 to 2010 the political support for the project was strong. Most importantly, the council mayors of that time were responsive to community aspirations for the project, but, sadly, the growing city debt and reduced political support made progress increasingly difficult from 2010 to 2016. The low point in the project came in 2013 when the council voted to excise 5.1 ha of public land from the Park to sell it for subdivision. A protracted process followed, which saw the proposal overturned in 2016. This paved the way for the statutory protection of the full 65.5 ha areas of public land originally purchased by council in 1975 for open space.

The support of various Maori groups has been crucial, especially at pinch points such as the attempt to sell off a portion of the land. Of relevance is that the council recently added more direct Maori involvement to its functions in the form of Maori wards. Whilst Maori involvement is becoming increasingly important because a Treaty-based approach is now being incorporated into all operations, there is nevertheless more reorganisation still to be completed and more understandings about urban restoration to be shared within contributing groups.

Fortunately for the project, when local government and political support have been weak, the community has responded to fill the vacuum. The current mayor and councillors are fully supportive of the project but, in the local government system based on a council term of 3 years, the old adage ‘the price of democracy is eternal vigilance’ certainly applies. For the community, this vigilance regularly takes the form of formal submissions on annual plans, long-term plans, regional plans, structural plans, and the like.

The WHNP project gained its initial momentum because of a strong alignment of community aspirations and a strongly supportive council and councillors. This momentum, combined with a strong innate ecological research focus, has led to an outstanding example of urban restoration. The project has created a major increase in vegetation cover and native animal food sources which encourage native forest birds, and it is now the largest food source for the Tui in the city when the harakeke (New Zealand Flax, *Phormium tenax*) is flowering. Large numbers of birds are now seen feasting on land once devoid of native nectar feeders. In addition, the restoration of the lake function is well underway (Duggan, 2012). The project has inspired other North Island cities to do more in this respect, and its success has flowed into the related urban gully restoration programme, which has recently received significant funding.

Now abundant nectar-rich harakeke in flower, a welcome major food source for Tui, a native forest bird. (CC0 1.0)

Case Study 2: Native Wildflower Meadows at the Queen Elizabeth Olympic Park

Project Rationale(s) and Strategy(ies)

The extensive flowering meadow landscape that featured so prominently during the 2012 London Olympics was conceived by Professor John Hopkins, the lead client for the Queen Elizabeth Olympic Park landscape, who had read *The Dynamic Landscape* (Dunnett & Hitchmough, 2004). As a consequence, in 2007, with 5 years lead time, Professor Nigel Dunnett and I were appointed Principal Planting Design consultants on what would ultimately be the largest new park in London since the nineteenth century. This gave us unusual capacity to work with the landscape architectural master planners for the project (Hargreaves and LDA Design) to introduce ideas about the ecological landscape of which the meadows were a key element. This was quite unlike many projects that have ecological elements bolted on at the end, severely limiting the integration of ideas and outcomes.

We saw that this was an excellent opportunity to develop the native wildflower meadow concept. These are astonishingly floral areas, and were to capture the hearts and minds of the estimated four million people who would walk through the site during the games period in 2012. We were very aware of the current political context, which meant that, at that time, interest in wildflower meadows was largely restricted to people in the area of nature conservation and, to a lesser degree, in landscape architecture.

Fig 1. (**a**, **b**) The native wildflower meadows in full flower in mid July 2012. (CC0 1.0)

Our vision for hyper-floral meadows brought us into conflict with some people in conservation organisations because they believed that this would put more negative pressure on grass-dominated, 'real' wildflower meadows, of which they were often stewards. Their view was that the public needed to be educated to appreciate non-flowery vegetation, which they should consequently see as intrinsically good. Our contrary view was that it was important to give people the capacity to create a sense of value about wildflower meadows on their own terms. We knew from some of the preliminary environmental psychology research, and from our own experience making wildflower meadows in public projects, that the main factor that drove appreciation of native wildflower meadows was flower density per unit area (Southon et al., 2017). We hoped that our project would inspire people to see native wildflowers in situ, and for them to come to terms with less dramatic displays, thus moving their psychological position on meadows in a more positive direction.

Key Project Features

The overall Games landscape design strategy was to incorporate as little mown turf as possible, and to make everything that was not either turf, woodland, or hard surface, into a meadow. The Olympic Park is a contoured bowl with the River Lea running through its length, and it was designed to act as a stormwater retention site when the Lea floods. This meant there were many slopes within the park, and these were the primary areas to be carpeted in native wildflowers.

What we did not know was what the soil for the meadows was to be, until the year before sowing. We designed the meadow species communities in the belief that we would most likely be sowing into a layer of highly alkaline crushed soil laid on top of a permeable subsoil clay. The aim was to create highly nutrient-deficient, unproductive meadows that would only support a relatively low biomass in the longer term, because this would counter the meadows becoming dominated by grasses. We believed that in the very long term, perhaps 50 years or so, this strategy would retain forb floral richness and drama for the public. When it became clear that there was unlikely to be sufficient of this soil material, we collaborated with a specialist soil science company and found a local quarry with a deep sand deposit with a very low phosphorus level (14 ppm) and no weed seed bank. This was then used as the top 200–300 mm of the profile over a clay-based subsoil for the native meadows.

When designing the seed mixes, we used our knowledge of United Kingdom wildflower communities and our research and practice experience, rather than to slavishly follow the United Kingdom Native Vegetation Council (NVC) reference community. We developed two seed mixes: one for the ‘shadier’ community on the north and east-facing slopes, and the second for the ‘sunnier’ south and west-facing slopes. Some species with wide ecological amplitude were present in both communities, but in general, the sunnier communities included more species with lower biomasses, whilst the shadier community had larger biomasses. Species were deliberately selected to maximise flowering impact during the Olympic Games and closely thereafter. Flowering impact was maximised by using a target seedling density for each species in the mixes. The seed mix design used an approach developed by the author in which a target number of seedlings for each species is decided upon (‘the design’) and a formula is then used to calculate, for each species, how many grams of seed are needed to be added, given an estimate of typical field emergence and the number of seeds per gram. The seedling targets per square metre and the amount of seed required to achieve this target are shown for the two seed mixes in Tables 10.1 and 10.2. Estimates of seedling emergence are available for seed sown in a sand layer at field capacity from the author's previous work (for emergence values determined for more than 700 species, see Hitchmough, 2017).

This seed mix design technique was developed by the author for use in situations where the performance of the seed mix in the first two years is of critical importance to achieve stakeholder buy-in, as in prestige sites where public expectations are high. It is a way by which notions of design, as would be normal in planted vegetation, can be applied to vegetation established by sowing, to provide a degree of control and minimise uncertainty and risk. The fact that we had developed these ideas to minimise risk was a key factor in the Olympic Delivery Agency being prepared to agree to large areas of sown meadow at Olympic Park.

It is recognised that control over meadow composition cannot be exercised in the longer term, where the combination of the site and management techniques ultimately determine the meadow's composition. Conventional restoration ecology techniques are, in practice, often heavily optimistic and undertaken without any sense of what people might think of the appearance of the subsequent vegetation. Seed mixes are often designed on the basis of the NVC, with seed rate being determined by the abundance of a species in an established community. It is the author's opinion that this is not a sensible approach to take in politically contested urban sites.

Table 10.1 Target seedling densities and seed required for sunnier location seed mix

	Desired plants/m^2	Grams seed/m^2 required
Calamintha nepeta	10	0.01
Campanula glomerata	10	0.01
Centaurea scabiosa	10	0.44
Daucus carota	10	0.05
Echium vulgare	5	0.08
Festuca ovina	10	0.04
Galium verum	20	0.05
Leontodon hispidus	10	0.11
Leucanthemum vulgare	10	0.02
Linaria vulgaris	5	0.01
Lotus corniculatus	5	0.03
Malva moschata	5	0.13
Origanum vulgare	20	0.01
Primula veris	20	0.14
Prunella vulgaris	10	0.04
Salvia pratense	5	0.05
Scabiosa columbaria	20	0.29
Thymus polytrichus	20	0.04
TOTAL	*205*	*1.74*

Table 10.2 Target seedling densities and seed required for shadier location seed mix

	Desired plants/m^2	Grams seed/m^2 required
Achillea millefolium	5	0.01
Agrimonia eupatoria	1	0.20
Betonica officinalis	10	0.25
Centaurea scabiosa	3	0.13
Deschampsia cespitosa	5	0.01
Festuca ovina	20	0.08
Galium mollugo	5	0.02
Galium verum	15	0.04
Geranium pratense	5	0.30
Geranium sanguineum	3	0.20
Knautia arvense	5	0.08
Leucanthemum vulgare	10	0.02
Linaria vulgaris	10	0.01
Malva moschata	5	0.13
Origanum vulgare	15	0.10
Primula veris	15	0.08
Prunella vulgaris	10	0.04
Ranunculus acris	10	0.25
Sanguisorba officinalis	5	0.18
Succisa pratense	5	0.24
Trifolium pratense	1	0.01
TOTAL	160	2.41

Given that the Olympic Park was a project with a long lead time, it was possible to undertake trial sowings to build the confidence of the clients and contractors in the approach. In January 2010, we set up a large randomised block experiment at the contractors' compound. This involved a weed seed-free sowing mulch (100 mm deep) with three irrigation regimes. All plots were irrigated from March to June 2010 to ensure that the targets in the sowing mix design were achieved. The main purpose of the experiment was to look at how cutting and irrigation after June could be used to remove and then 'push' the meadow biomass to maximise the number of flowers present at the end of July, which in 2012 would correspond with the Games' opening day. In the London climate, most native wildflowers have finished flowering by the end of July, so we had to find a way to retard the flowering activity. The meadow experiments were photographed weekly in 2011 and the maximum flowering at the end of July was found to be when they were cut back to ground level and the biomass removed at the beginning of May (10–12 weeks before the end of July).

Fig 2. The meadow test plots sown in 2010, being used to assess the capacity of cutting (in 2011) on subsequent flowering time. (CC0 1.0)

Seed sowing of the Olympic Park's 10 ha of native meadows was scheduled for December 2010–March 2011 to ensure meeting the chilling requirements of some UK meadow species. Seed was procured mainly from the United Kingdom native meadow seed industry (which ensured United Kingdom ecotypes), but some species were only available in the volume needed from horticultural seed suppliers such as Jelitto Seeds. The two seed mixes for the dozens of sowing units across the park were made up by the author and his PhD student Zulhazmi Sayuti in the Department of Landscape at the University of Sheffield. We chilled seed of species that could not be sown in winter due to contractual issues in order to maximise their emergence in the field. The author trained the contractors' staff (Gavin Jones Ltd., and Frosts Landscape Ltd.) and the meadows were sown by hand using a sawdust carrier, with a combined seed and carrier mix sown in two passes at one handful per square metre in each pass. Seed was mixed in wheelbarrows, and then decanted to buckets for sowing; a quick efficient process when well organised. The seed was raked into the surface and irrigated from March to June to achieve the seedling emergence targets. Issues with the supply of water to the irrigation system led to initial problems in emergence for the first sowings, but eventually, however, these difficulties were resolved and excellent emergence, very close to target expectations, was achieved in nearly all sown areas.

A composted bark amendment had been mixed with the sand to increase water holding capacity, but as the seedlings started to grow, this proved problematic because nitrogen in the sand was utilised by bacteria to decompose the composted bark, leading to very slow seedling growth, which does not occur with untreated sand. By mid-summer 2011, the author was concerned that, at the observed growth rates, most species would not be large enough to reliably flower in 2012, especially since they would be cut off at ground level in May 2012 to push back flowering to the opening days of the Games. We decided that we would fertilise the meadow areas with a nitrogen-only fertiliser in order to temporarily boost growth but, at the same time, not to increase the longer-term phosphorous levels that we had been so keen to minimise. We recognised that higher phosphorous levels would promote unwanted grasses at the expense of the forbs. The author walked the site every 14 days with the contractors and instructed them on which areas were to have applications of an ammonium nitrate-based cereal fertiliser. In combination with short-term irrigation, this strategy proved to be very effective and by October 2011, we had achieved the meadow cover and plant size that we needed. Under normal circumstances the use of the nitrogen to accelerate growth would not have been needed, and indeed its use almost certainly led to higher mortality, through competition, of some of the slower-growing species. Nitrogen fertilisation ceased in late summer 2011.

There were virtually no weedy species in the sowings in 2011, due to the absence of a weed seed bank in the sand. The seed mix contained only one grass, *Festuca ovina*, with a very small amount of *Deschampsia* in the shadier mix, to prevent competitive dominance of grasses pre the Games window and maximise flower density and drama. Planning consents required the meadows to be over sown with a native grass mix after the Games. Rather than avoiding grass in the seed mix altogether, one species was included because it would become a more important part of the planting fabric post-Games, and to meet expectations that a meadow should include some grass. The meadows were cut, the biomass removed starting in early May 2012, and the flowering began on cue for the Games on 28 July 2012.

Major Project Outcomes

The meadows were extraordinarily dramatic, and many visitors were visibly fascinated, even moved, by the extraordinary experience. It would not be an exaggeration to say that experience of these meadows in flower shifted subsequent attitudes to meadow vegetation in British urban space.

The Olympic Park Management Authority organises an annual park walk every year with the designers of the park, and this has allowed ongoing involvement with the management and development of the meadows. The meadows became much grassier following the games as they were oversown, as required, with a grass mix. Given that much of the British landscape is dominated by grasses often to the exclusion of forbs, the author saw this as an undesirable decision, leading to a reduction in the forb diversity that is so rare in the UK.

The meadows are typically cut late in the year, in September or later, and this favours grasses and tall leafy stem forb species. In this area, the author was mystified by the ongoing decline in forb density and diversity in many of the meadows. On the 2017 walk, a chance conversation revealed that the maintenance contractors had not switched off the temporary irrigation system, and that the meadows had been irrigated throughout the summer since 2012, leading to grass dominance. Since the termination of the irrigation, grass biomass has decreased and forb biomass increased, and in most cases, the meadows are now looking very good, which is a testimony to the power of low productivity substrates when engineering ecological plant communities.

Fig 3. Wildflower density and diversity in 2021 is generally now very good

Case Study 3: The Australian Botanic Gardens Shepparton

Project Rationale(s) and Strategy(ies)

The Australian Botanic Gardens Shepparton is an example of a 20-year-long community-driven restoration project occurring within a rural township. It highlights the role of botanic gardens as an institutional network, in transformative public engagement and, increasingly, in the preservation of rare and endangered species. On Yorta Yorta country, the township of Shepparton, population 60,000, is the heart of the dryland and irrigation-driven food bowl of south-eastern Australia, located on the Goulburn River.

It took local community members a dozen years of advocacy before the Shepparton Council agreed to use a former municipal waste disposal site as the location for what would become the Australian Botanic Gardens Shepparton. Grossly disturbed sites such as waste facilities, quarries, and other brownfields are often the only sites available in the contemporary urban landscape for new public open space (Meyer, 2007), and these industries essential for urban development leave a legacy that makes their re-use as residential or industrial land use difficult. The 22.6 ha site consists of former grazing land, eight hectares of remnant woodland adjacent to the Goulburn River, and "Honeysuckle Rise" (named after a rare local sandhill species), which is the capped mound of urban waste and the site of main plantings to date.

Key Project Features

Botanic gardens are created landscapes, where detailed design processes underpin their layout and structure in order to increase their potential to provide public education and engagement outcomes. For this project, good early design decisions made the most of the site's potential. The initial extensive earthworks were undertaken to cap a towering pile of urban refuse. The height of the capped mound was further raised to enable views out over the adjacent woodland, while the whole mounded footprint was graded to provide easy access for visitors. The 'borrow pits', areas excavated to provide the layers of soil 'dressing' the waste pile during its time of operation, were also reworked to create a floodway to manage the frequent flooding from the nearby Goulburn and Broken Rivers, and which over time, as per planning, have developed into well-functioning wetlands. Expert advice from a soil specialist (Dr Peter May, University of Melbourne) led to the use of mature biosolids for soil improvement and to the use of early direct seeding of the mound surface with local native grass species. These works cost the local council little beyond what had to be spent on remediation. Over the years, the local community has organised for landscape architects to create a site master-plan, and to produce design-specific planting plans and include the inclusion of bespoke areas such as a Children's Garden. The planning committee also organised a flora survey of the site and used the four ecological vegetation classes identified in that survey as a basis for planting design (Mann, 2018). There has also been extensive use of recycled materials throughout the designed spaces, which proved to be a cost-effective means of emphasising the transformation of place and promoting core sustainability issues.

An early planting, designed to be viewed from the top of the mound, representing the grid of food-bowl dryland irrigation set against the sinuous presence of the Goulburn River, with remnant grassy woodland behind. (CC0 1.0)

Hardy species painstakingly established by community hand-watering are set against contemporary landscape architecture utilising recycled materials. (CC0 1.0)

One of the most important goals that the community-led committee of management codified in their vision for the Australian Botanic Gardens Shepparton was that it should promote and showcase native Australian and indigenous species with the aim of increasing their uptake within residential settings. In the Australian context, domestic yards are by far the largest category of green space within most cities (Marshall et al., 2019), and yet they are often dominated by exotic species. Shifting that compositional mix towards a greater use of native species can result in substantial changes to an area's ecology (White et al., 2005).

Engagement with the well-resourced Royal Botanic Gardens in nearby Cranbourne, which is the Australian native species-focused wing of Victoria's world-renowned Royal Botanic Gardens, as well as with other botanic gardens across Australia and New Zealand, has been important to the Shepparton Garden's development. As part of their initial planning process for the Australian Botanic Gardens Shepparton, community members undertook fact-finding tours to several botanic gardens, including Cranbourne. Early discussions with Cranbourne staff identified two areas for specific collections. Remarkably, the Australia Botanic

Gardens Shepparton is now the first botanic garden in Australia to focus on the common, widespread, and iconic genus *Acacia* – a fact that attests to the historic dominance of exotic species in the Australian psyche. The other specific collection is the *Thomasias*, a mostly Western Australian genus of small shrubs with papery sepals.

Recently, the Australia Botanic Gardens Shepparton joined Cranbourne's *Care for the Rare* program (Hird et al., 2012). Under this program, expert staff at Cranbourne propagate threatened species for ongoing custodianship by participating program members. Local landscape architect Melissa Stagg of Stagg Design has been working extensively with Cranbourne staff to identify species suitable for the Australian Botanic Gardens Shepparton and to incorporate the final 60 species selected into the indigenous beds that she has designed.

Gabion walls provide shade and shelter for species chosen under the Royal Botanic Gardens Cranbourne's *Care for the Rare* program. (CC0 1.0)

Beyond the *Care for the Rare* program, Cranbourne now also supplies specialist educational services to Australian Botanic Gardens Shepparton, for example running a biomimicry day for two local Shepparton primary schools.

Whilst low funding levels for the gardens have naturally caused some disappointment and frustration for those involved in their development, they have also acted to make this project richer by giving more people the opportunity to engage and feel ownership and investment. Funding from Shepparton Council has always been very tight and, as a consequence, the local community has been clever in seeking other sources of funding from a range of state and regional grants. Some of these sources have allowed various essential works in the bushland area of the Gardens, most notably weed removal and the removal of illegally dumped domestic and construction waste. Management of the remnant areas also includes the provision of walking tracks and signage. While monitoring of improvements to the flora has been, up until this time, limited to casual observation, the observed re-emergence of native orchid species suggests some success.

Recently, the Shepparton Council took over management of the gardens and this change of administration has led to major new infrastructure projects being funded. These projects include the planned installation of town water and solar-powered irrigation, while a bridge over the Broken River, together with a shared path, has finally made the direct and convenient connection to the nearby Victoria Lake precinct of Shepparton and the drawcard Shepparton Art Museum. As a consequence, visitation to the gardens is skyrocketing.

Major Project Outcomes

It is hard to overestimate the importance of the contribution of the local community and residents to this project. Not only were they fundamental to its inception and in providing the core vision for the project, but they have also provided the major workforce over the developmental and maintenance periods of the gardens. The plantings of Honeysuckle Rise have been propagated, planted and established by the Friends group, who have had to water this area by hand in the absence of electricity or town water supplies on site. The local birding group also conducts surveys every two months, while there are evenings devoted to spotting gliders. In addition to the involvement of these groups, the gardens have been regularly visited by school groups and the local ‘bush kindergarten’. The gardens also host a film night run by Shepparton Festival, and other community activities related to art and weaving classes, in conjunction with joggers and cyclists, are held at the Gardens. Interestingly, Covid 19 and the various restrictions to people’s activities have seen an increase in people visiting the Gardens, clearly seeking the respite from lockdowns provided by well-cared-for public open space. Many of the plants grown in the Gardens have come from a local nursery run by and for people with disabilities, and some of the Gardens’ management activities have been provided by a local Urban Landcare Group. The community-led RiverConnect project aims to return the Goulburn and Broken Rivers to their rightful stature across a region where their role as a resource for irrigation is critical. It would appear that the Australian Botanic Gardens Shepparton is a key to achieving that fundamental transformation. Of significant cultural importance is that the Australian Gardens section also now provide a place and opportunity to promote the traditional management practices of the Yorta Yorta Indigenous people of the area.

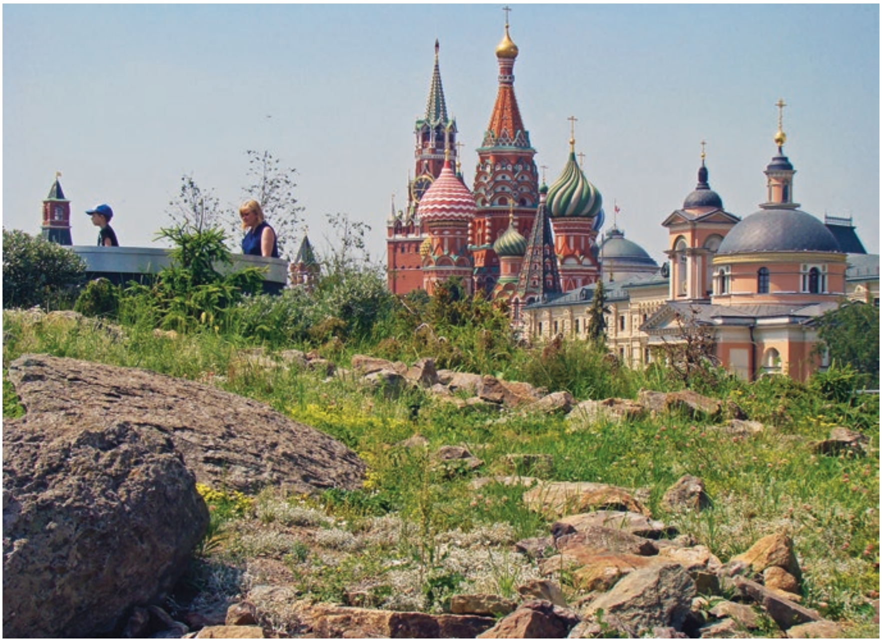
An old Dethridge wheel, an Australian invention vital for irrigation, repurposed into a planter carrying the rare Honeysuckle of the local sand hills. (CC0 1.0)

After many years of planning and action, the Australian Botanic Gardens Shepparton is a long-term project that has achieved the parallel and interdependent goals of undertaking native restoration, engagement with communities, and creating opportunities for education and transformation at a deep social level. This is a fundamental achievement and highlights the strength of the network created by and through this urban botanic gardens program.

Case Study 4: Zaryadye Park, Moscow

Project Rationale(s) and Strategy(ies)

In the very centre of Moscow, a mere stone's throw from the Kremlin and the domes of St Basil's, the 10 ha Zaryadye Park is an ambitious attempt to bring ecosystem-based planting to Russia, seeking to capture the essence of the country's steppes, tundra, forests, and wetlands (Walliss, 2020). Open since 2017, the Park was the result of an international design competition won by Diller Scofidio and Renfro (DSR), Hargreaves Associates, and Citymakers. It is a complex of spaces, and includes restaurants, a museum, an outdoor amphitheatre, a large underground car park, an educational centre, exhibition venues, and the Zaryadye Concert Hall, with two spaces for 1500 and 400 visitors. This Concert Hall includes a green roof, and having been visited by more than 12 million people, it has proved to be an extraordinary success in terms of its popularity (Evstratova et al., 2019).



Welcome to the tundra: In the heart of Moscow, Zaryadye Park brings a sense of wildness to its plantings. (CC0 1.0)

Key Project Features

The richness of the plantings is exceptional. Initially the Park included 150 species, but this has now increased to over 300. Many were grown to an advanced stage before planting, and all were sourced from local and regional nurseries from Russia. The Park has a staff of eight people in its administration area, and a landscape maintenance crew of 30–40 external contractors. This latter figure is seasonally less in winter when temperatures plunge, and the Park is covered in snow. To maintain the four biomes of tundra, forest, steppe and wetlands in a healthy condition, careful planning of soil depths and composition (Rappoport et al., 2019), irrigation, fertiliser application, and other management approaches are essential.

To find out more about this remarkable Park, and the nuances of its design, I recently interviewed (i) landscape specialist Evgeny Sapunov, in charge of the maintenance of Park's green spaces and who liaises directly with the government of Moscow, and (ii) Elena Voytsekhovskaya, head of public and educational programs of the Park.

Evgeny told me that Russia's parks are essentially a nineteenth-century design, and that restoration and ecosystem-based design is not practiced within urban settings. In this respect, our translator had difficulty with the appropriate phrase for 'restoration', and, as a consequence, Evgeny said:

> Zaryadye Park came as a bomb, so to speak, to the mentality of the Russian people. They were not ready to accept this concept yet. People from all over Russia came to this park and they were hoping to see roses or lilacs and instead they saw some sort of weed-type of plant. They've got something similar back at home, and they wanted to know what the point was of coming to this Park.

However, it is four years on now, and a significant social and learning change is underway. We were told that 'What was developing for decades overseas, is taking weeks in Russia. Even the tour guides are saying this part is related to tundra, this one is like steppe, as they slowly, gradually absorb [the concept]. This Park is like an educational train pushing through the mentality of Russians'.

Our informants were very reflective about this change. They said that:

> Zaryadye Park has been very innovative. We had this model of Japanese gardens where you just observe the beauty and people didn't realise that you can create a similar type of beauty using our own Russian plants like birch trees and pines. If before people were stepping on some the grass we have used, now a few years later they are noticing it, saying don't step there, because they know it is not just grass, it is a special grass.

Furthermore, they were able to explain their own position in this developing system, noting that:

> And when we, the Park managers, liaise with the local governments and departments of culture, we explain to them that this Park is not a frozen structure, it is a nature-related structure that is supposed to evolve and renew and move and the plants that are there are supposed to refresh, renew, and move around. It is a kind of exhibition of all parts of Russia.

As a testament to the insights involved in preparing this concept, to some extent, the Park has become a victim of its own success. There is increasing pressure now to include other ecosystems, and distinctive plants from many places across the vast area of Russia. Evgeny explained the magnitude of the effort needed to meet this pressure, telling us that:

> We have two species of birch tree in the Park but [there are] another 25 species of birch across Russia, so why don't we present all 27 of those so that people would be aware of it? Same for fir trees; why don't we show the other types?

Birch forest is one of the four Russian biomes represented at Zaryadye Park. (CC0 1.0)

Another complication which arises is that of Moscow's own ecological and climatic niche, since clearly not all the plants appropriate to the different biomes of Russia can be grown. The solution to this dilemma was to have an 'analogue landscape design', which did not purport to copy the exact species, but planted similar species that look alike but are not the exact ones. For example, sedum can grow in Moscow so, Evgeny says, we can use this in the place of moss.

Major Project Outcomes

The Park's success is now kickstarting similar projects elsewhere in Russia to the degree that some 15 are being planned, with two having already been constructed. The Park's success is also slowly encouraging the academic research community and, although the ecological expertise is currently nascent, there is interest from a new generation of professionals. Whilst some of these are trained overseas, there are some from Moscow University and from the Strelka Institute for Media, Architecture and Design. The latter is a newly formed urban design research institute. Our informants were hesitant to suggest that great progress has been made in this area, implying that there is still much to be done. It was also noted that the Parks' educational facilities include five well-equipped labs for young children (7–12 years of age) where they can study the micro-world of the Park, focusing on biotechnology and genetics.

Our informants were clearly very excited and proud of this urban development and summed up their feelings by saying: “This Park is a phenomenon in Russia, an educational project for all the Russians to understand what it is and how it works and how to look after it”.

Chapter Synthesis

These four Case Studies have made the claim that urban restoration is driven by community power, political will, and by available resources. What makes a fundamental understanding of urban restoration so complex is that these conditions vary enormously across the world. The development of technical advances, such as the methods to establish wildflower meadows in London’s Olympic precinct, or for the creation of distinct biomes as done at Zaryadye Park, Moscow, now allow various ecosystem types to be established in areas that, hitherto, could not have otherwise been contemplated. However, the resources and expertise required for such a restoration to be mounted are probably beyond the means of the majority of urban governments.

On the positive side, a richer understanding of the nuances of urban ecology is leading to less rigidly prescribed landscapes. This notion is developing in conjunction with the recently felt effects of climate change, which is leading to a renewed emphasis on urban greening and human ecosystem services. An example here is wetlands which are being (re)introduced as protective devices for restraining the impact of storm surges. These four Case Studies have suggested that we need to integrate urban environmental design and restoration projects with scientific and social research and monitoring to ensure we adapt natural ecological processes to a rapidly changing world. Large-scale urban planning will belatedly begin to build-in resilience by recognising movement corridors, retaining drainage lines, and retaining at least a robust patchwork of habitats across new urban areas.

Implications Drawn from these Case Studies

There are several important implications that can be taken from these Case Studies, notwithstanding their wide global scope, which involves engaging with different communities, different climates, and widely different plant species. These implications for urban restoration appear to be as follows:

(i) Understanding and establishing an agreed context is paramount for developing successful urban restoration activities.
(ii) The practical engagement with urban restoration projects is not a site for pure research activities. Urban conditions and anticipated outcomes indicate that

urban restoration is about presentation and changing attitudes as much as it is restoring ecological function.

(iii) Urban projects have the advantage of being experienced by many people on a day-to-day basis, giving them the experience and perspectives to change the way we think about nature and our environment.

(iv) Urban spaces can offer unique opportunities for ecological restoration. Particularly useful are rooftops and road verges, or indeed the 'cliff-face analogues' presented by the facades of tall buildings.

(v) Large-scale infrastructure projects can offer the best chance of large-scale restoration because they (i) have the capacity to reorganise urban spaces and land uses, (ii) can be politically sensitive, and (iii) they are often accompanied by aspirational goals for new urban regeneration.

(vi) To be effective, urban restoration projects need to use appropriate planning and urban design tools to ensure that they are suitably embedded in the urban fabric.

(vii) Opportunities need to be recognised and grasped when they arise.

(viii) Urban ecology is now becoming a way to define identity, and thus offers a practical avenue into establishing cultural adoption of natural processes.

(ix) Changing a law is often the best means of facilitating effective restoration.

(x) High-quality research programs are vital to creating opportunities for restoration, as well as for improving efficiencies and reducing long-term costs of environmental management.

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Chapter 11
A New Gold Standard in Mine Site Restoration to Drive Effective Restoration Outcomes

Kingsley W. Dixon and Tristan Campbell

Contents

Summary and Key Lessons
Mining operations often result in highly altered ecological systems due to the innate process of mining, where the original cover of plants and fauna are typically lost, there is large-scale removal of soil and alteration of geological profiles and often sites need to be managed in terms of the release of polluting or toxic materials such as salt, heavy metals or acid mine drainage. With Life of Mine typically lasting decades, there is usually a long delay from initial disturbance from mining operations to commencement of ecosystem recovery or restoration, which can create another layer of complexity.

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With the release of globally applicable Standards for the Ecological Restoration and Recovery of Mine Sites at the UN Biodiversity Conference in 2022 (COP15), companies for the first time have a standard of practice and outcomes for:

1. Effective pre-mining understanding of biodiversity contexts and ecosystem functional attributes expected after closure.
2. Establishment of agreed post-mine ecosystem outcomes and verified standards of practice to achieve this.
3. Use of mine waste materials as a resource for developing functional soils and substrates to assist restoration practice.
4. Early establishment of integrated monitoring systems to track progress against the planned restoration trajectory.
5. Identifying early stages of research interventions and adaptive management to address shortfalls in achieving desired ecosystem attributes.

Based on existing restoration standards for non-mined lands, these mining standards provide a globally robust framework to develop, implement and monitor mine site restoration to achieve best-practice outcomes. Adoption of the standards by the mining industry will enable a common language for understanding both community and ecological issues with rehabilitation after mining while allowing for project-specific restoration requirements that will create greater compliance capacity and sustain the Social License to Operate (SOL).

Examples and case studies are provided that illustrate the impacts of mining operations and their ability to achieve restoration outcomes within the Mining Standards.

Introduction

Mining continues to impact increasing areas of nature, wilderness and ecosystems to meet the global demand for minerals (Ferguson et al., 2021), especially for 'green metals' critical for the transition to alternative energy. This ever-increasing scale and need for effective and timely restoration of mining-altered ecosystems has never been more important. With active mines present in every recognised global biodiversity hotspot (see Fig. 11.2), and the global mining footprint exceeding 57,000 km^2 (Maus et al., 2020), sustainable and ecologically defendable restoration after mining remains both problematic and complex, with the industry unable to consistently deliver best practice outcomes (Cooke & Johnson, 2002; Lamb et al., 2015; Stevens & Dixon, 2017).

The Cumulative Environmental Impact (CEI) of mining, where mined land areas accumulate at a greater rate than company capacity to rehabilitate, is creating future economic, social and environmental liabilities and risks to the sustainability of surrounding ecosystems. These CEI areas now eclipse the technical and financial capacity of industry in many regions to effectively reinstate agreed land values, particularly as mine expansion gains pace (Vivanco et al., 2017). It is now widely agreed that the need for the mining industry to achieve long-term ecosystem reinstatement is paramount beyond the traditional safe, stable and non-polluting standards already in wide practice.

The Ranger Uranium mine operates within the World Heritage Area of Kakadu National Park in northern Australia. In 2015, rehabilitation and closure of operations by 2026 were estimated to cost \$526 million. This was revised to \$973 million by 2017 and is now expected to be \$1.6–2.2 billion (Tietzel & Sainsbury, 2022). Much of this cost escalation was due to a lack of understanding of the complexities associated with restoration and the need for early intervention in defining rehabilitation risks and deriving early solutions. Multiple stakeholders, including the Traditional Owners of the land and the local and global communities, are concerned that the restored Ranger mine must achieve a best practice outcome if closure and reincorporation into the World Heritage Area are to be achieved (Fig. 11.1).

The globally agreed aspirations of the mining industry to 'leave the land better than they found it' (Stevens & Dixon, 2017) have not been borne out by the many mines that have been abandoned or closed without achieving a satisfactory and agreed end land use (Cross et al., 2018). Indeed, the stated goals regarding

Fig. 11.1 Ranger Uranium mine has operated in the World Heritage Area of Kakadu National Park. It ceased production in 2012 and is now in closure phase. (Photo credit: Energy Resources of Australia Ltd., https://www.energyres.com.au/media/gallery/)

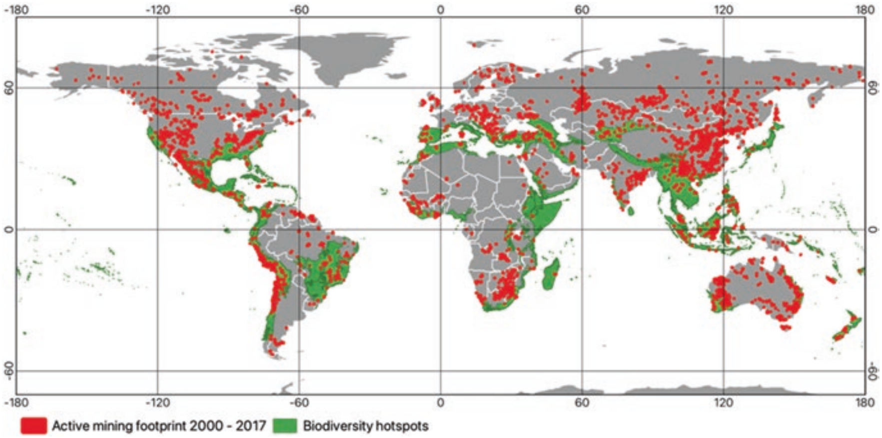


Fig. 11.2 Extent of active mines from 2000–2017 (Maus et al., 2020) overlayed on biodiversity hotspots (Hoffman et al., 2016) showing that all hotspots have mining often on a large scale. Many mines, though small in area, are in highly threatened ecosystems, such as those in the east coast of Madagascar and southwest Australia. Background map made with Natural Earth. WGS84 coordinates

biodiversity impacts by one of the world's largest mining companies, Rio Tinto, have been reduced from 'Net Positive Impact' in early 2004 (Rio Tinto, 2008) to a 2017 position of 'prevent, or otherwise minimise, mitigate and remediate the effects' (Rio Tinto, 2017). This lowering of environmental goals came the year after Rio Tinto had worked closely with the International Union for Conservation of Nature (IUCN) to develop internationally recognised protocols for net biodiversity gain (IUCN Business and Biodiversity Programme, 2017). Such a change in a global mining giant points to concerns that mining companies now realise their technical deficiencies and lack of expertise are impediments in mine closure.

In the globally significant mining province of Australia, the number of abandoned mines is estimated to be approximately 50,000 (Lamb et al., 2015), with less than a handful of the estimated 1400 active mines that operate in native ecosystems able to achieve even a modest restoration of the pre-mined vegetated state (Young et al., 2022).

Compared to other ecosystem restoration projects mine site restoration can be challenging due to the extreme ecological impacts of the extractive process and the extent of the resulting highly altered landscapes and substrates. This complex situation includes the loss of plants, animals and soil microbiota as well as the wholesale removal of soil from large areas (Muñoz-Rojas et al., 2016). Thus, restoration of mined lands for a pre-existing ecosystem often requires de novo reinstatement of biological and ecosystem functions, as a result of:

– Homogenisation and loss of topsoil materials, often extending into removal of parts of the underlying geological formations and growth medium (Cross et al., 2018).

– Altered surface and groundwater systems, with potential mobilisation of toxic materials (Licsko et al., 1999; Mendez & Maier, 2008).
– Loss of soil-based propagules including seed and bud banks and radical alteration to the soil microbiome (Golos et al., 2016; Valliere et al., 2022) compounded by seed supply that is often inadequate to deliver effective mine site restoration (Pedrini et al., 2020).

In addition to the increasing technical complexities associated with restoration of mined land, mining operations in many regions now operate under legal frameworks requiring the minimisation of environmental risk and some level of post-mining restoration to agreed standards. However, while these requirements can be well-intentioned, there is often a lack of consistent definition of the term 'restoration', which is often interchanged with terms such as rehabilitation, reclamation or revitalisation (Cross et al., 2018). As a consequence, the lack of regulatory frameworks to provide clear definitions for environmental reparation after mining, together with a high level of confusion within individual mining companies, both conceptually and technically, have resulted in ad hoc, and often poorly executed, rehabilitation of mine sites that falls well short of the pre-mined state.

Since the late 1900s, there has been an increasing need for mining companies to work within an informal societal framework, often referred to as the 'Social License to Operate' (SLO) (Prno & Slocombe, 2012). The community expectations of the SLO, and the costs to the mining companies for breaching this informal license, vary greatly around the world. This can lead to tens of millions of dollars in lost production for projects, or possibly billions of dollars for project costs and asset write-downs (Franks et al., 2014). Thus, post-mine ecosystem restoration is a major liability for the mining industry as legacy mines with large unrestored footprints accumulate at a faster pace than fully restored mines.

It is now recognised that, to be effective, mine site restoration must begin at the earliest point in the mining cycle. Such a strategy involves planning for closure and restoration prior to commencing mine construction. This is a key concept in the recently released International Principles and Standards for the Ecological Restoration and Recovery of Mine Sites (Young et al., 2022) and is summarised graphically in Fig. 11.3. Early and continuous stakeholder engagement is another of the key principles in mine site restoration approach set out by Young et al. (2022), and is one of the many reasons why effective restoration can assist with reducing overall mine costs and risks to the SLO. While voluntary reporting on environmental and social aspects of mining operations is increasing (Heenetigala et al., 2015), there is evidence that this can negatively impact on foreign investment, driving mining to countries or regions with lower standards and thereby creating environmental impacts where there is less financial, technical and/or legal enforcement to prevent or repair these impacts (Opoku et al., 2022).

In this chapter, we present case studies that provide practical examples of how various mining companies have approached post-mined land restoration. These studies include details on technical development for effective restoration planning, implementation and monitoring. All case studies cover issues with SLO

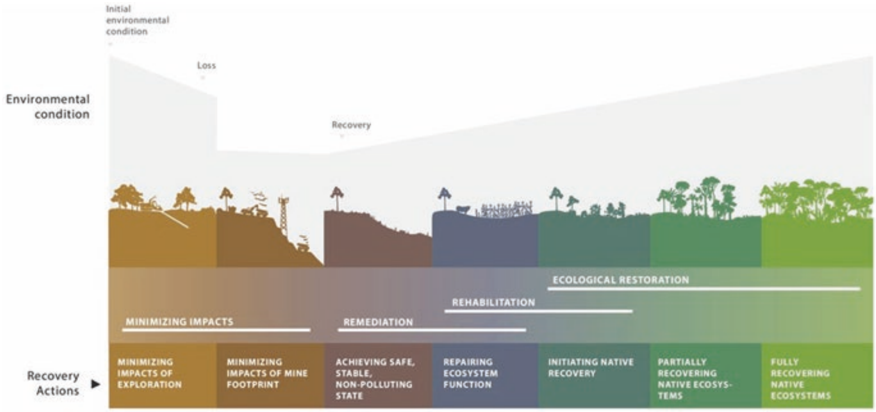


Fig. 11.3 The Recovery Trajectory for Mine Sites, showing how different activities are linked to different Life of Mine stages (from Young et al. 2022). The 'recovery monitoring' bar illustrates where monitoring informs new practice through an 'adaptive management' framework

expectations for ecological restoration at mine closure. A case study highlighting the integration of these aspects to achieve best practice outcomes concludes this practical example section.

Case Studies

Case Study 1: Topsoil Management for Effective Restoration

Case Study Background

Topsoil is the most important part of the soil profile for plant growth, as it contains the required nutritional, organic, regenerative and other compositional properties (Bradshaw, 1989). Thus, knowledge regarding the quality of topsoil is crucial for the effective planning of restoration activities. This need for topsoil is often further enhanced by the presence of propagules of plants indigenous to the area (Golos & Dixon, 2014), depending on the land use of the mining project area prior to mining development (e.g. native seed banks are dramatically reduced or not present in agricultural land or if topsoil stripping and storage has been poorly undertaken). It is well known that both the compositional properties to support plant growth and the presence of propagules are not characteristics of subsoils below the topsoil layer (Cooke & Johnson, 2002).

In modern mining restoration projects, the process of the mining operation typically means that the topsoil is removed, stockpiled and then utilised at a later time for restoration purposes (Abdul-Kareem & McRae, 1984; Golos & Dixon, 2014). However, it is apparent that the storage time of the topsoil has a variable impact on its biological components (Birnbaum et al., 2017; Muñoz-Rojas et al., 2016; Rokich et al., 2000), and whilst this may adversely impact restoration outcomes (Gann et al., 2019), it is not always the case (Birnbaum et al., 2017).

While physical and chemical properties have long been the focus of indicators of soil quality (Rabot et al., 2018), and soil organic carbon is often seen as a key parameter amongst these properties (Hueso-González et al., 2018), soil biological quality is now being seen as an increasingly important issue (Bastida et al., 2008). This has been found to be particularly important in disturbed ecosystems (Schloter et al., 2018) where disturbance can result in disequilibrium in the soil microbiome. Therefore, soil quality indicators need to be considered in relation to the combined function of the physical, chemical and biological properties of the soil (Muñoz-Rojas, 2018). A summary of commonly used bulk measures of each of these property types is provided in Table 11.1.

When considering soil as a functioning microhabitat, over 80% of the internal processes are seen to be mediated by microbes (Nannipieri & Badalucco, 2003). From the extraordinary diversity of microbes in healthy soils, many microorganisms are functionally redundant, where the loss of one or more groups of organisms can be substituted by another that provides an equivalent functional role (Kennedy & Stubbs, 2006). Therefore, while the advent of fast and cost-effective genomic sequencing provides a wealth of detail on the soil microbial composition, this may not relate directly to soil functionality.

Table 11.1 Commonly used bulk measures of physical, chemical and biological properties of soil

Soil quality property type	Common bulk indicators
Physical	Bulk density
	Soil texture and structure
	Aggregate stability
	Porosity
	Plant available water
	Hydraulic conductivity and infiltration
Chemical	Organic and total C
	Organic and total N
	Available nutrients (P, K, pH)
	Electrical conductivity
	Cation exchange capacity
	Carbonates
Biological	Microbial diversity and biomass
	Microbial respiration
	Microbial community
	Enzymatic activity
	Earthworms, nematodes

Adapted from Muñoz-Rojas (2018)

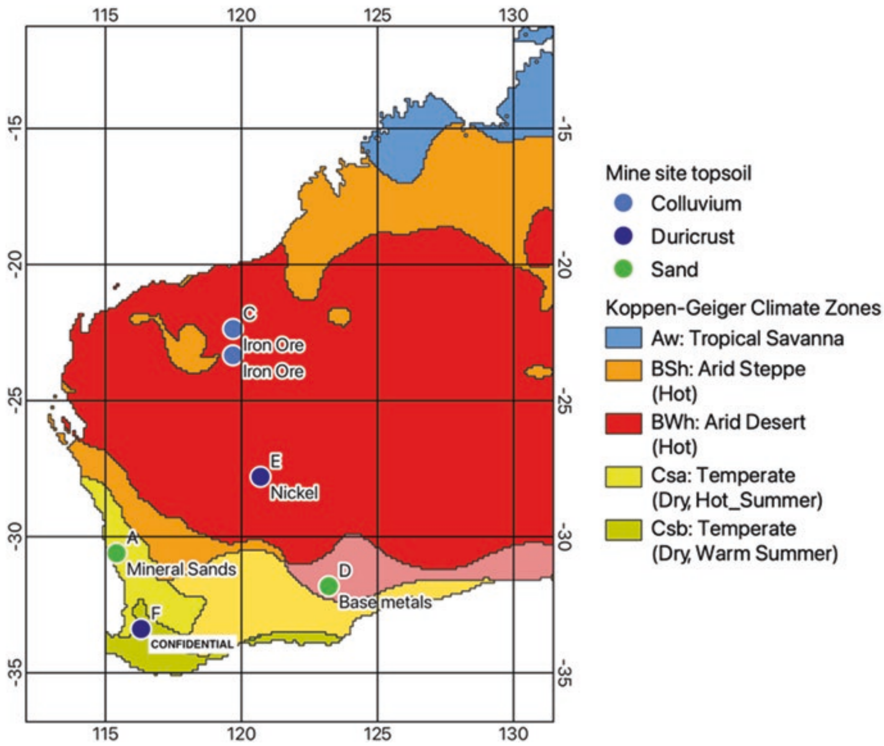


Fig. 11.4 Location of mine sites used for soil restoration functionality assessment. Sites are labelled with their site ID referred to in this case study and the commodity type of the mine. (Adapted from D'Agui et al. 2022)

This case study, which is one of the most comprehensive in the mining industry, draws extensively from work by D'Agui et al. (2022) and Valliere et al. (2022) on the relationship between soil function and potential for restoration in the form of plant biomass assays. Their work used microbial soil assessments from undisturbed reference sites and stockpiles of different ages across six mine sites in Western Australia and represents one of the most comprehensive studies of its type for mines globally. These sites represent a wide range of mining resource types, soil types and climatic regions (Fig. 11.4). The sample collection and analysis workflow are summarised in Fig. 11.5 and described more fully in D'Agui et al. (2022). With characteristics of particular seedling growth patterns from this dataset as described by Valliere et al. (2022) together with statistical analysis of soil microbial communities by D'Agui et al. (2022), this Case Study focuses on topsoil handling across the different mine sites and soil types as one of the key 'lessons learnt' in the particular challenges faced in mine site restoration projects.

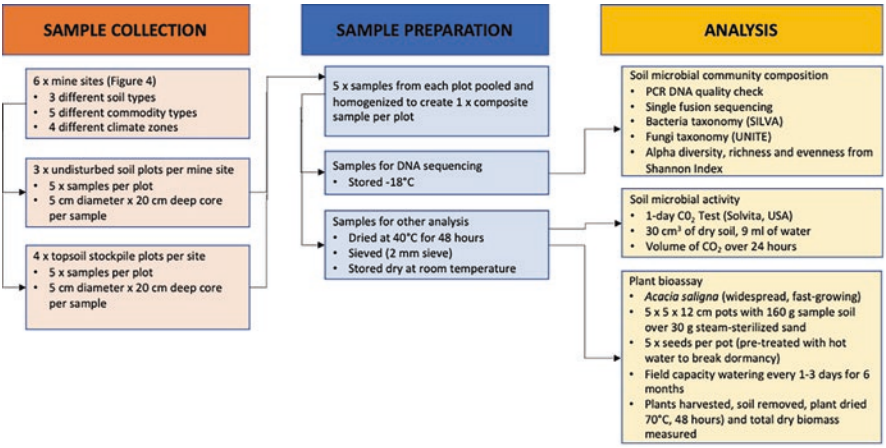


Fig. 11.5 Sample collection and analysis workflow. (Bacterial database from Quast et al., 2013, fungal database from Dietrich et al., 2013, Shannon Index as per Wagner et al., 2018 and plant bioassay as per Valliere et al., 2022)

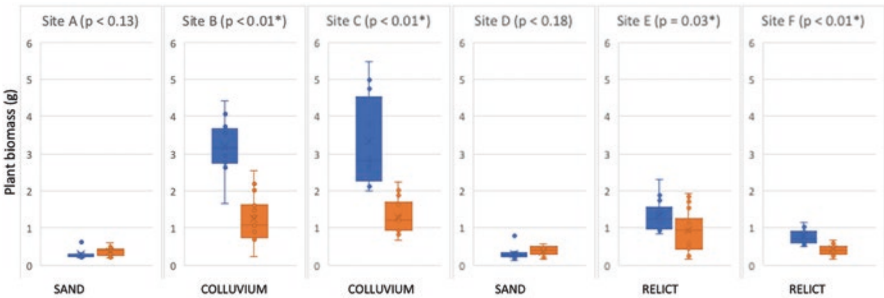


Fig. 11.6 Plant assay biomass by site with soil type labelled. Blue bars are samples from undisturbed reference soils and orange from topsoil stockpiles. ρ-values are from a single-factor ANOVA test between reference and stockpile samples for each site. (Adapted from D'Agui et al. 2022)

Case Study Results

Plant bioassay results for the six different mine sites, in the form of dry weight of *Acacia saligna* seedlings grown in soil samples, presented in Fig. 11.6, are separated into the undisturbed reference soil samples and topsoil stockpile samples for each site. It is clearly apparent that there is a significant range in biomass values, with median biomass for each site ranging from 0.3–3.2 g for the reference samples and 0.4–1.2 g for the stockpile samples. There is also a wide range of relationships between the reference biomass versus the stockpile samples for each site. The stockpile sample biomass was statistically significantly less than the reference sample biomass for four sites (albeit only just significant for one of these) and was slightly more for the remaining two sites, although not statistically so.

From these data, a pattern for the wide range of biomass values emerges when the typical topsoil type of each mine site is considered. Colluvium soils typically have significantly higher biomass for both sample types than the relict soils, whilst the relict soils have significantly higher biomass than the sand soils. This pattern is still valid when relative biomass per site is used (stockpile biomass being on average 38% less than the reference biomass for colluvium, 64% less for relict soils and 30% more for sand).

While this consistent pattern is not surprising for reported sites B and C, since both sites are from mines extracting the same commodity in the same region, each pair of sand and relict soil types are from different commodity mines and markedly different climatic regions (see Fig. 11.4). As a consequence, these data allow a degree of predictive capability for restoration establishment for the potential of stockpiled topsoils based on the relatively simple parameter of soil type regardless of a mine's commodity type and climate. In this respect, it is to be expected that in-field restoration will be more challenging for sites of the same soil type but having a more extreme climate, such as sites E and F, which are both relict soil types but are found in arid (hot) and temperate (warm) climates, respectively.

Figure 11.7 shows the relationship between residence time in the stockpile and plant biomass, grouped by soil type. As with the reference versus stockpile sample biomass results, the relationship with time in the stockpile shows consistency between soil types.

Biomass from stockpiles in the sand sites over time is initially significantly higher than the reference sites (at 6 months storage time), then similar biomass to the reference sites from 1 to 3 years and then revert back to significantly higher biomass than the reference samples after three years. Studies in similar sandy environments have shown the arbuscular mycorrhizal fungi (AMF) portion of the soil microbiome to be dramatically reduced after one year of stockpiling, but then recovering after 5–10 years of stockpiling (Birnbaum et al., 2017). However, the plant

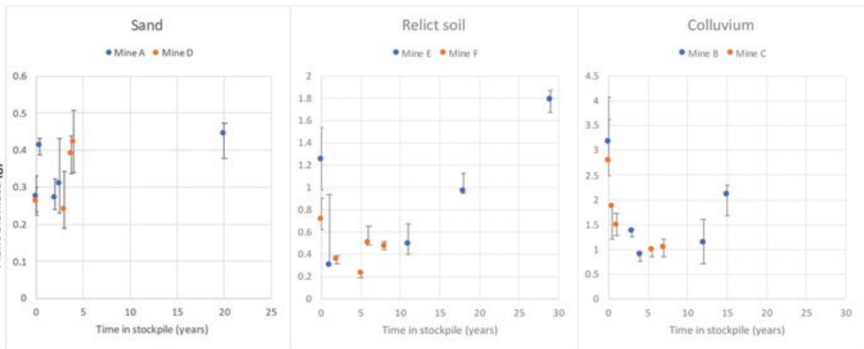


Fig. 11.7 Time in stockpile versus bioassay results grouped by soil type. Reference samples are at '0' years. Points are median values, whilst error bars show the 25th and 75th quartiles of the samples. (Graphs created from data associated with Valliere et al., 2022)

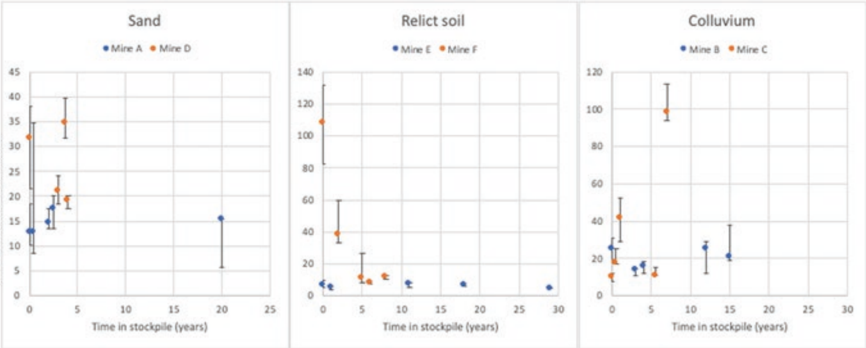


Fig. 11.8 Stockpile time versus soil microbiome activity (CO_2 test). (Graphs created from data associated with Valliere et al. 2022)

assay biomass in Birnbaum's study had an inverse relationship to both time and AMF presence, and it was concluded that "there is likely an unaccounted-for effect of other biotic variables on observed results" (Birnbaum et al., 2017, p. 243) such as soil pathogens. This confirms the work by Jasper et al. (1987), who found that, in similar soils to sites A and D, AMF spore density in the soil was a poor predictor of mycorrhizal formation in roots. For the data presented in this case study, the biomass for the sandy sites was either similar or greater than the reference sites regardless of time, since stockpiling and the soil microbiome activity was either similar or less than reference sites (see Fig. 11.8). Therefore, it seems that bulk soil microbiome assessments were again poor predictors of plant growth. Similarly, there were no clear relationships between any of the soil microbial community measurements and biomass for the sandy sites.

The relict soil and colluvium soil type sites had a more consistent pattern of markedly lower biomass of stockpile samples compared with reference soils within six months of stockpiling, with biomass remaining low but starting to increase after 10–15 years of stockpiling (Fig. 11.7). As with the sandy sites, there was no relationship between biomass and soil microbiome activity (Fig. 11.8). According to the soil microbial community measurements, however, there appears to be a positive relationship between biomass and fungal taxa diversity for each of the relict sites (Fig. 11.9), particularly as the biomass of the stockpile samples approaches (or exceeds) the reference biomass (Site E). Nonetheless, this is not true for the colluvium sites, with fungal diversity for the stockpiles similar to, or higher, than the reference sites, despite having lower biomass. Indeed, there is an inverse relationship between fungal diversity and biomass for stockpiles at Site B (which had longer stockpile times than site C), though this pattern does not include the reference site. Further examination of the other microbiome activity data did not reveal any relationships between these data and biomass for any soil type, site or stockpile age.

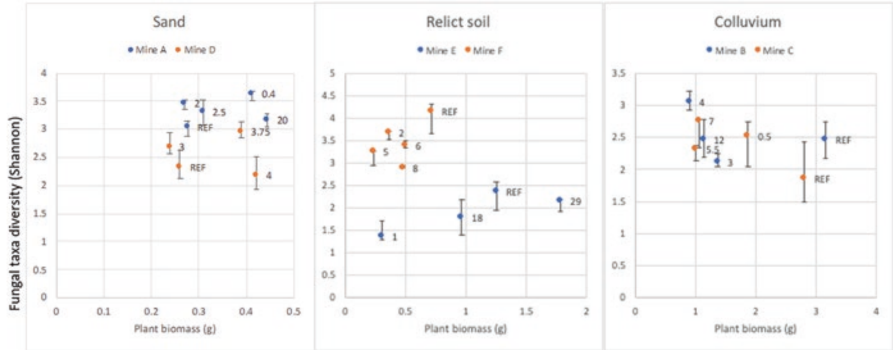


Fig. 11.9 Plant biomass versus fungal taxa diversity. Data points labelled as REF for reference sites or age of stockpile in years. (Graphs created from data associated with Valliere et al. 2022)

Case Study Lessons Learnt

As discussed in the Case Study rationale, topsoil quality for restoration is a combination of the physical, chemical and biological properties of the soil. This is more complex in mining restoration projects, as the topsoil is often removed and stockpiled for many years, or, in some cases, blended or lost completely. This often results in topsoil being respread as thinner layers post-mining than the pre-development ecosystem soil structure with subsequent significant impact on plant growth or not at all.

Although being time- and resource-intensive, the use of plant bioassays is one of the few simple and easily deployed empirical measures to guide restoration of stockpiles at any given time in their lifespan. As demonstrated in this case study, combining plant bioassays with other datasets can lead to relationships between restoration potential and other, more straightforward, soil measurements. However, it is clear that topsoils from native ecosystems are highly variable in their microbiome and plant growth responses to soil stripping, storage and replacement. Thus, site-specific topsoil use will need to be nuanced with empirical studies undertaken to determine what, if any, supplements (e.g. organic amendments, nutrients) or specialised handling approaches will be required.

Case Study 2: Going Beyond Plant Indications – Fauna Monitoring for Assessment of Functional Ecosystem Restoration

Case Study Background

Ecological restoration programs often use landform and floral characteristics as the key success criteria (Ruiz-Jaen & Aide, 2005), as it is generally assumed that restoration of vegetation, and habitat structure will result in recovery of fauna (Toth et al., 1995; Young, 2000). This is neatly summarised as the “Field of Dreams” hypothesis by Palmer et al. (1997), where “if you build it, they will come” (Palmer et al., 1997, p. 295).

The extent of this assumption is apparent from the review of over 300 publications by Wortley et al. (2013), with less than one-third of publications on restoration considering fauna, and only 11% considering vertebrate fauna. A review specifically of mine site rehabilitation articles in Australia by Cristescu et al. (2012) found only 20 publications on fauna monitoring despite there being over 328 operating and 1113 historic mines in Australia at the time of the article. While this Field of Dreams hypothesis can be correct (Pearson et al., 2022), it is far from universal and the use of habitat proxies for assumed faunal recolonisation may be deeply flawed (Cristescu et al., 2013).

A global review of over 100 articles assessing fauna in mine-site restoration by S. L. Cross et al. (2019) found that even when fauna are considered, the assessments were primarily focused on overall species diversity and richness (invertebrates in particular) rather than the functionality of the restored ecosystem with respect to

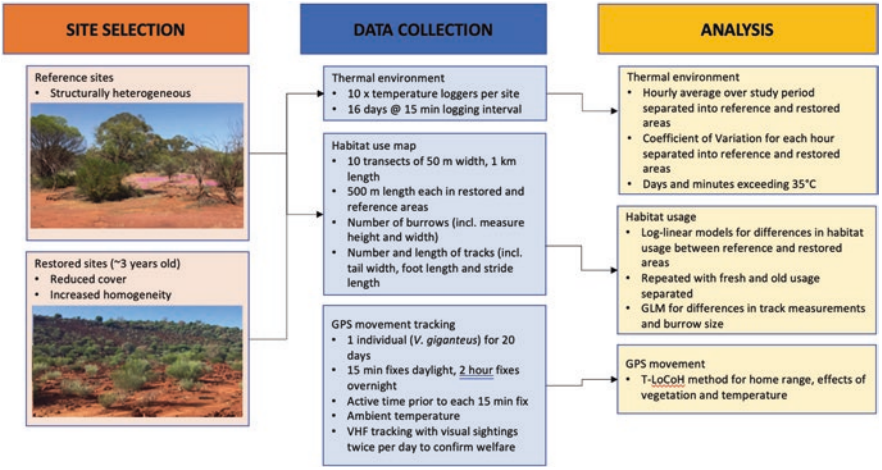


Fig. 11.10 Data collection workflow from S. Cross, Craig, et al. (2020b) and S. L. Cross, Tomlinson, et al. (2020c) used to develop an approach to understanding faunal occupancy on restored vs native reference sites. (Photo credit: S. L. Cross)

fauna. As the role of fauna in the environment is key to ecosystem functionality (Cross et al., 2020a, b, c; Gagic et al., 2015), consideration of the activity of fauna in restoration is key to assessing restoration success and the potential for improved restoration outcomes through ecosystem processes driven by fauna (Catterall, 2018).

For this case study, we summarise the findings of S. Cross, Craig, et al. (2020b), who assessed habitat use and movement of monitor lizards (*Varanidae*) within unmined reference locations and within early-stage restored sites, at a mine site in the biodiverse semi-arid Mid-West region of Western Australia, approximately 415 km northeast of Perth. The study assessed a variety of indicators of monitor lizard activity (such as tracks, diggings, and burrows) as well as thermal differences in habitats to assess habitat occupancy of reference and restored sites. The data collection workflow is summarised in Fig. 11.10. More detailed descriptions of the data collection and analysis for this case are presented in S. Cross, Craig, et al. (2020b, c), noting that this case study has mainly focused on the outcomes and lessons learnt aspects.

Case Study Results

As the habitat usage data in Fig. 11.11 shows, S. L. Cross, Craig, et al. (2020b) found that there was significantly more activity detected in the reference areas compared to the restored areas, both in total and for each of the usage types of burrowing, diggings and tracks. Overall, there was approximately three times the level of habitat usage in the reference areas compared to the restored sites.

The average size of monitor lizards, as inferred by average burrow sizes presented in Fig. 11.12, shows that, overall, S. L. Cross, Craig, et al. (2020b)

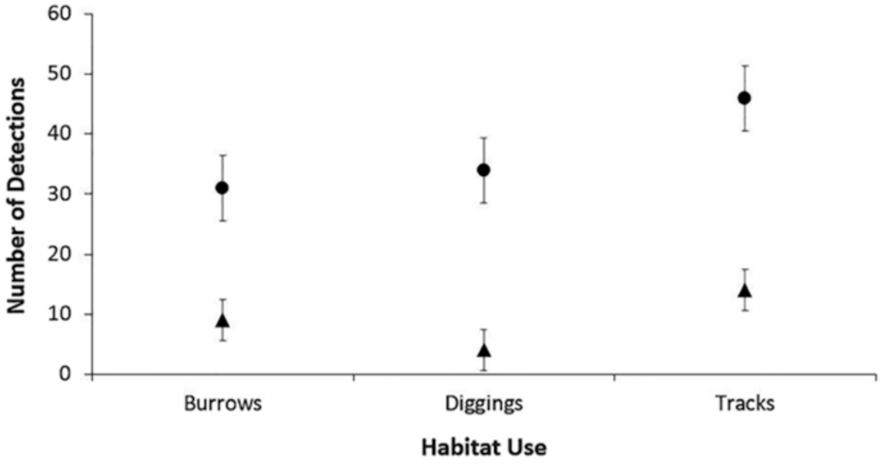


Fig. 11.11 Total recorded habitat use with Standard Error bars. Circles are from reference areas and triangles from restoration areas. (Image from S. L. Cross, Craig, et al. 2020b)

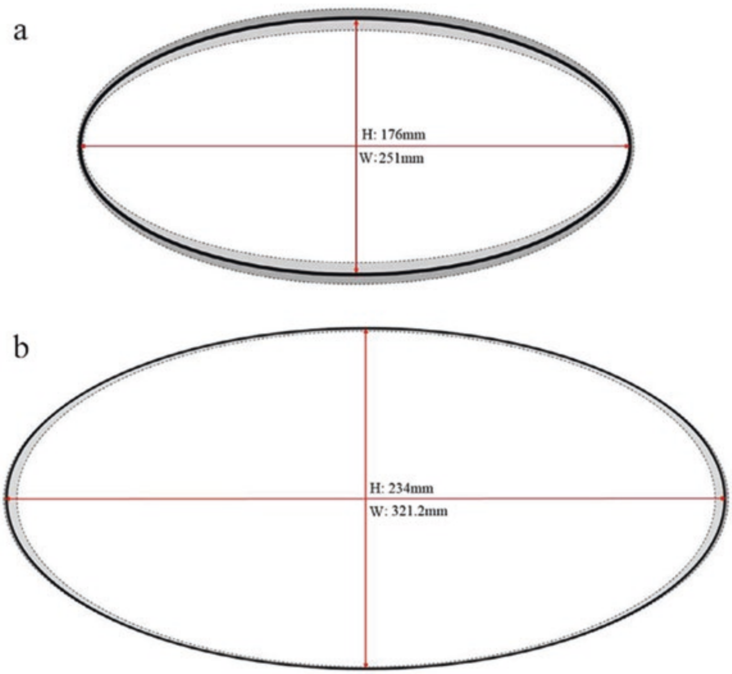


Fig. 11.12 Variability in burrow width and height (mm) between (a) reference vegetation and (b) restoration vegetation, drawn to scale. The middle oval in each figure represents the average burrow size, and dashed lines and shaded areas show the average plus/minus one standard error. (Image from S. L. Cross, Craig, et al. 2020b)

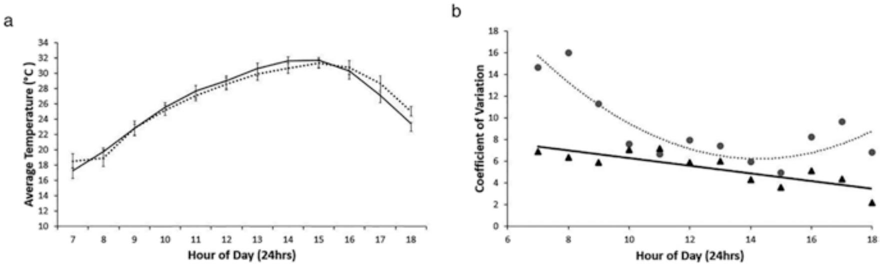


Fig. 11.13 Thermal environment of reference and restoration sites during daylight hours (07.00–18.00): (a) average hourly temperatures in reference (...) and restoration (−) vegetation, +/− Standard Error (SE) and (b) coefficient of variability in hourly temperatures in reference (•) and restoration (▲) sites. Trendlines in reference (...) and restoration (−) vegetation do not relate to a statistical function but emphasise differences in thermal patterns between reference and restoration areas. (Image from S. L. Cross, Craig, et al. 2020b)

detected that significantly larger monitors were burrowing in the restored area compared to the reference area. However, there were no significant differences between reference and restoration areas in total track length, tail width, and foot and stride length. S. L. Cross, Craig, et al. (2020b) inferred that larger-bodied monitors are better suited to utilise restored landscapes, likely due to their increased tolerance of fluctuating temperatures, whereas the smaller-bodied monitors can only use these areas opportunistically (i.e. crossing restored landscapes without burrowing).

Track lengths measured by S. L. Cross, Craig, et al. (2020b) resulted in approximately half of all tracks through both areas indicating that the monitors were crossing the areas with minimal variation in direction (travel proportion of 1, effectively a straight line). However, the reference areas had a maximum travel proportion of 4.27 versus 1.26 for the restoration area, indicating that there was a greater variability of usage of the reference areas by the monitors than the restored areas. This supports the habitat usage data in Fig. 11.11, with burrows, diggings and tracks all significantly less common in the restored areas.

The difference in habitat usage and travel proportions between the restored and reference areas reported by S. L. Cross, Craig, et al. (2020b) indicates that while the restored areas are indeed being used by monitors, the usage is more infrequent and opportunistic than in the reference areas. It was inferred that this may arise from a lack of key resources in restoration areas (such as food), the need to minimise the time in high-risk areas which have less vegetation cover, or possibly from the higher degree of thermal variability in the restored areas (Fig. 11.13), since the metabolic cost of thermoregulating in the more thermally variable restored areas is high (particularly for smaller individuals).

Case Study Lesson Learnt

S. L. Cross, Craig, et al. (2020b) concluded that while monitor lizards were detected in both restored and reference areas, the difference in usage of the areas and burrow sizes indicates that these areas are far from equivalent in terms of an ecosystem for monitor lizards. As a result, this case study highlights how observations of presence may well not indicate the persistence of residence or be a reliable proxy for ecosystem functional support for sustained fauna return.

With restoration activities typically focused on returning plant diversity to areas, the 'build it and they will come' hypothesis assumption that fauna will naturally return may lead to an understatement of the faunal habitat requirements required for effective ecosystem restoration. S. L. Cross, Craig, et al. (2020b) suggested that providing fauna refuges (such as hollow logs and debris piles for this case study) may be needed to assist with earlier return-to-site of large fauna that rely on such structural complexity, thus catalysing earlier return of ecosystem functions.

Case Study 3: Restoration of Abandoned Mine Sites – Summitville, USA

Case Study Background

In response to the impacts of industrial contamination in the 1970s, the United States enacted the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA or Superfund) (Rahm, 1998). The Act effectively 'put site owners and operators on notice that contaminating sites with hazardous substances can have severe consequences' (Danielson et al., 1994). Four decades later, there are over 1800 active Superfund sites, although less than a quarter (448) of these sites have been 'deleted' where the Environmental Protection Agency has determined that no further action is required (https://www.epa.gov/superfund/superfund-national-priorities-list-npl). Approximately 21 million people, or 6% of the US population, live within 1 mile of a Superfund site and the presence of a Superfund site is still strongly linked to reduced life expectancy (Kiaghadi et al., 2021).

Case Study Description

The Summitville site in Colorado commenced as an underground operation, changing to an open-pit gold mining operation in 1984 (Gray et al., 1994). Due to the geological setting, open-pit operations exposed significant volumes of sulfide-bearing rocks to water- and metal-laden, acidic surface and groundwater contamination became a serious issue (Warhurst & Mitchell, 2000). This was followed by failure of the closed-system cyanide processing facility. In 1992, the mining company declared bankruptcy and spring floods the following year caused significant overflow from the mine site to nearby rivers, resulting in over 50 km of 'dead' river (Laitos & Ainscough, 2017). The site was declared a Superfund site in 1994, only 10 years after open-cast mining operations commenced.

Restoration of the Summitville site represents some of the most challenging situations associated with mine site restoration, with contamination still being produced on site long after cessation of mining operations and pollutants capable of being transported off site into nearby groundwater and surface water systems. Contaminants of concern that are present in the soil and waste rock and mobilised by water include arsenic, cadmium, chromium (VI), cyanide, lead, manganese, mercury and zinc.

As discussed by Rieder et al. (2013), ecosystem restoration of the Summitville site was heavily constrained by the natural conditions (extreme temperature range, winter snow packs, short growing season) in addition to the challenging chemical conditions of the topsoil and waste rock material (with even the stockpiled topsoil being extremely acidic). Restoration commenced with an exhaustive literature review in 1995 to quantify the range of constraints to establishment of a self-sustaining plant community and identification of potential remedies to these constraints. In parallel with this research, works commenced on-site to detoxify areas of the site and significantly reduce surface and groundwater off-site contamination.

To test the efficacy of the potential treatments identified in the desktop review, Rieder et al. (2013) conducted a greenhouse bioassay program (similar to Case Study 1 in this chapter) from 1995 to 1996. This program tested the effects of 36 combinations of different source materials, organic matter treatments and acidity neutralisers on plant growth characteristics. In addition to the bioassay data to compare plant growth potential, aboveground plant tissue samples were screened for heavy metal contamination.

The greenhouse trials found the most effective amendments were to add organic material in the form of manure or mushroom compost, include some topsoil and neutralise the acid with agricultural-grade lime. Seven of the most promising findings from the greenhouse program were then tested in field trials from 1997 to 1999. The optimum treatment options from the greenhouse were validated in the field trials, which showed that while the addition of topsoil in the field trials did not improve plant growth it did improve seedling survival and therefore overall biomass production.

Site-wide restoration commenced in 1999, seven years after the cessation of mining operations and after four years of applied restoration research. 200 hectares of waste rock were treated with agricultural-grade lime and mushroom compost, 15 cm of topsoil added (and treated with agricultural-grade lime and fertiliser then direct seeded with a combination of six grass and forb species.

Monitoring of vegetation cover and species richness over the following decade (Fig. 11.14), as presented by Rieder et al. (2013), showed a rapid increase in vegetation cover over the first five years to 2004, indicating successful establishment of plants from the seeding program, before a slight decline from 2004 to 2007 and a slight increase in average and maximum cover after 2007. The species-specific

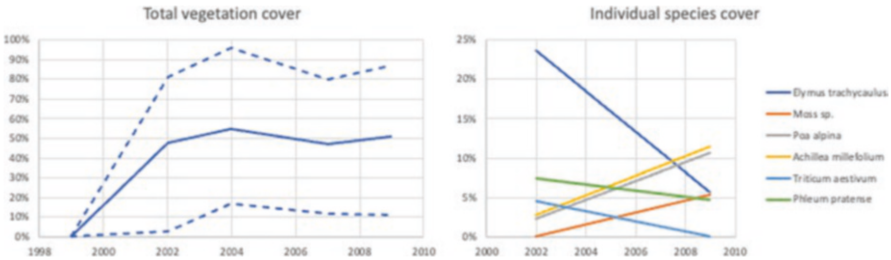


Fig. 11.14 Vegetation cover over time since seeding (1999). Left: Solid line is average vegetation cover, dashed lines are minimum and maximum cover ($n = 69$). Note that reference plots surveyed in 2009 had average cover of 88% for subalpine meadows and 59% for subalpine forest understories Right: vegetation cover for species that formed a significant proportion of total cover in 2002 or 2009. (Created from data presented in Rieder et al., 2013)

comparisons between 2002 and 2009 showed that the majority of the key species (both those in the seeding mix and those germinated from the topsoil seed bank) had increased in cover over this time, including some comparatively long-lived native species. Notably, slender wheatgrass (*Elymus trachycaulus*) decreased from an average of 24% cover in 2002 to 6% in 2009. This is expected as slender wheatgrass is short-lived (generally from 3–5 years) and does not establish seedlings well in older restoration communities. The other two species that declined in cover are *Phleum pratense* (an introduced weed) and *Triticum aestivum* (common wheat -another weed). The decline in total vegetation cover between 2004 and 2007 due to the decline in cover of these three species was likely an expression of the natural succession stages of the restoration program and indicated the site was on a desirable ecological recovery trajectory. This result highlights the need for site-specific monitoring metrics that reflect the reference ecosystem, as while the floral diversity has increased since 2004 a simple metric of vegetation cover would indicate potential beginnings of failure of the restoration program between 2004 and 2007.

Case Study Lesson Learnt

The Summitville case study is an example of regulatory failure during the design, approval and operating stages of mine life. The substantial environmental impacts of the project were avoidable if planning from the outset of the mining operation considered the likely mobility of toxic materials given the nature of the minerals being mined that are well known to create major environmental risks.

This study does show that well-planned and science-led approaches towards restoration for even the most toxic and challenging of sites can result in restoration to a high standard, with the restoration program at Summitville now achieving a 4.0 star level of restoration when assessed using SER's standards for mine site restoration (Young et al., 2022) as shown in Fig. 11.15.

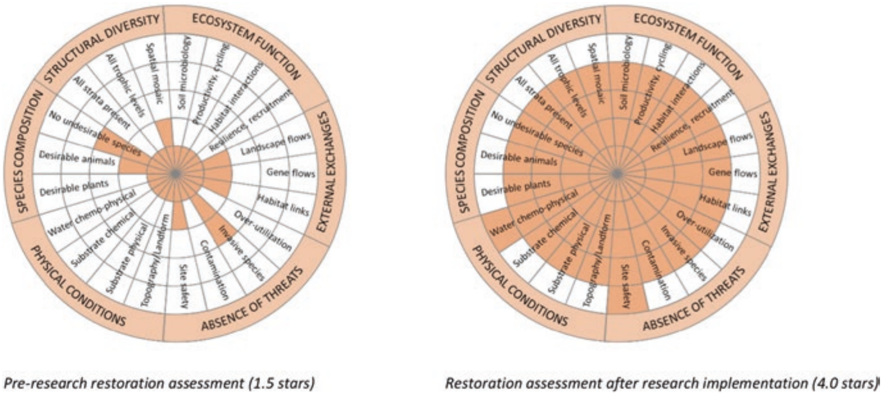


Fig. 11.15 Pre- and post-restoration ecological assessment of the restoration program at Summitville Gold Mine as per SER's standards for mine site restoration. (From Young et al. 2022, Figure S1iii)

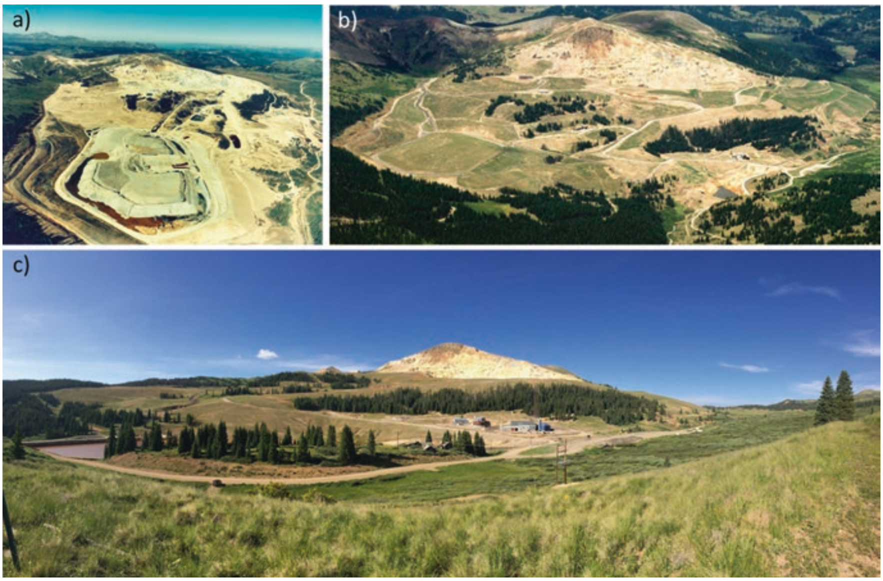


Fig. 11.16 Summitville restoration site. (**a**) site prior to reclamation in 1993, (**b**) 2 years after revegetation in 2002 and (**c**) the site in 2014. (From R. Young et al. 2022, Figure S1iv)

Importantly, on-ground restoration commenced after five years of theoretical and applied scientific research to maximise the likelihood of success. Without this level of due diligence, the restoration outcomes may well have been significantly poorer, resulting in additional time and costs to achieve the outcomes (see the Ranger Uranium Mine example in the Introduction to this chapter).

While the restoration program at Summitville has achieved remarkable successes (Fig. 11.16), the continued acidification of the water in the mine-void means that the water treatment plant operating in 2011, which processes up to 8000 litres of water per minute and extracts over 1 ton of heavy metals per day, will need to operate 'forever' (Boardman & Aviles, 2019).

Case Study 4: Restoration and Social License to Operate in Bauxite Operations in a Biodiversity Hotspot

Case Study Background

Alcoa generates 7% of global aluminium metal production, primarily from ancient native jarrah forests in the southwest of Australia, which is recognised as a global biodiversity hotspot. This large-scale activity impacts potentially up to 800 native plant species in one of the world's most species-rich temperate forest ecosystems (Davey & Gossage, 2014) with 900 ha of primary forest removed per annum. The Alcoa operation now has a cumulative footprint of 320 km^2.

Alcoa's operations are the largest recurrent land clearing activity in the southwest Australian hotspot and operate entirely in native forest with a complex, relict soil structure and containing a significant density of old-growth trees (over 200 years old) and other habitat features vital for critically endangered species such as Forest Red Tail and Baudin's Cockatoos. This is one of the largest single commodity mining and largest strip-mining operations in any of the 36 global biodiversity hotspots (Maus et al., 2020).

For mining operations, native vegetation is fully cleared followed by stripping of topsoil then removal of 5m or more of bauxite followed by rehabilitation. Stripped topsoil is returned and if correctly managed can retain a viable soil seed bank (Koch & Hobbs, 2007). To further assist restoration, supplemental seeding and nursery stock planting are then undertaken in the recontoured pits that remain after removal of the bauxitic duricrust. Alcoa self-assess restoration outcomes with less legally binding obligations on the quality of the revegetated mines than comtemporary legislation requires.

The legal framework for these mining operations stems from local State legislation developed in the 1960s and was designed to minimise environmental obligations on mining companies to encourage investment in what was then an under-invested mining industry in Western Australia. As a result of the legal framework, there were obligations on the company to reinstate a forest cover (originally exotic species) but was silent on the requirement to match a native reference ecosystem. Alcoa exceeded these loose regulatory controls and now aims to return 70% of the pre-mined diversity and richness through its rehabilitation activities (Davey & Gossage, 2014). However, unlike other mining operations in Australia, Alcoa decides which species and the ecosystem composition that comprise the 70% goal. This is within the context that most plant species in the southwest hotspot do no migrate from the margins or other forest areas due to the peculiarities of the southwest Australian flora (Hopper et al., 2016). However, with increasing demand

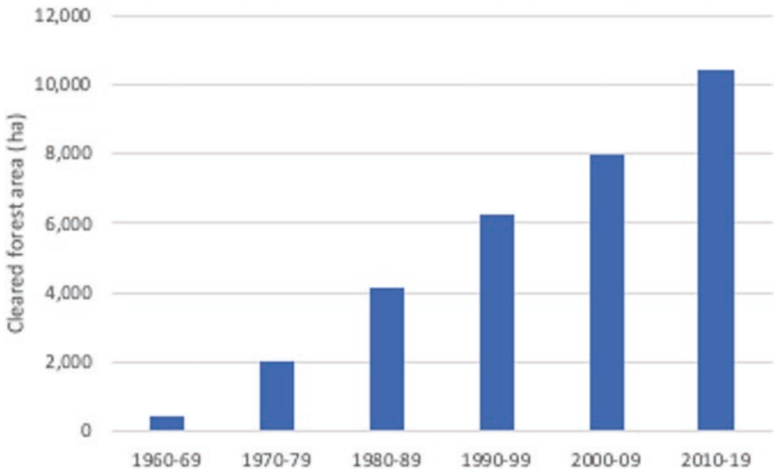


Fig. 11.17 Area of forest cleared each decade by bauxite mining in Western Australia. Note this is per decade, not cumulative. (Data from Hon A. Sanderson 2022)

for aluminium, Alcoa has, in the past 30 years rapidly increased the scale of mining from hectares to square kilometers (Fig. 11.17). Importantly, key framework plant groups, which include major families such as Ericaceae, Dillenianceae, Rutaceae and many dryland grass-like groups (Poaceae, Cyperaceae, Restionaceae) and until recently, trees such as *Persoonia* spp. are often absent or at substantially lower frequency than expected in the native reference (Chia et al., 2016; Willyams, 2012).

Case Study Description

While direct return of topsoil is an effective means for ensuring a high diversity of native species (Daws et al. (2022), Alcoa have found that logistical constraints mean that the company has ability to directly return seed-rich native topsoil (i.e. topsoil that is not stockpiled prior to use) to 35% of rehabilitation sites (Davey & Gossage, 2014). These deficiencies in direct return of topsoil despite being shown by Alcoa's research to be the most effective approach for return of the jarrah forest ecosystem result in species deficits in the understorey component even though collecting and broadcasting of up to 2700 kg of seeds (from a combination of wild-collected and nursery sources) and propagation of up to 230,000 seedlings of recalcitrant species is occurring annually (Grant & Koch, 2007; Willyams, 2021).

The effects of the topsoil deficit are compounded by the pace of Alcoa's proposed expansion of mining activities, and this has led to growing community concern over the ability of Alcoa to reinstate plant species and protect old-growth environmental values such as large trees and tree hollows for denning, roost and forage (key aspects of the Standards recovery wheel). Although Alcoa presently does not have plans to mine the 69% of the remaining Northern Jarrah Forest biome,

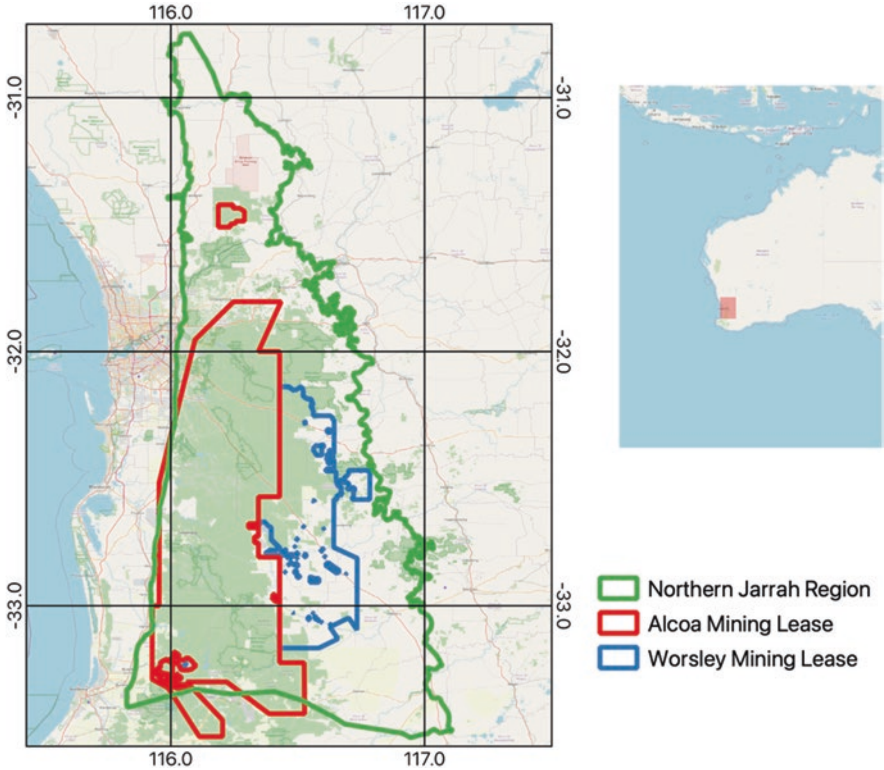


Fig. 11.18 Extent of Alcoa mining lease versus Northern Jarrah Forest biogeographic region. Northern Jarrah Forest region from Department of Agriculture Water and the Environment (2020), Mining Lease information from https://geoview.dmp.wa.gov.au/geoview/ (accessed 15/04/2022). (Basemap information from OpenStreetMap contributors 2021)

a further 26% of the region has other active bauxite mining leases in addition to exploration plans for other minerals such as nickel. Thus, mineral leases can potentially impact more than half of the biome (particularly when habitat fragmentation and life cycle disruption is considered as well as vegetation clearing), which has the potential to trigger IUCN listing of the biome as an Endangered Ecosystem (Bland et al., 2017) (Fig. 11.18).

There is also a cumulative legacy of underperforming rehabilitated sites, with an estimate of 60,000 ha where some common plant groups found in the native forest, such as Ericaceae, are at low abundance while other sites exhibit an overdominance of reseeder species compared with resprouter species such as legumes (Koch & Hobbs, 2007). In terms of restoration of the floral aspects of the restored ecosystems, nett results have shown that, despite over 90% of the plant species being present in restored sites, and with the overall vegetation structure approaching that of unmined areas after several decades, the level of plant species similarity as measured by the Sorensen index (i.e. proportions of species present) is generally 60%.

In addition, there is no evidence that replanted tree seedlings will attain old-growth status via a stable and climate-resilient trajectory with the Northern Jarrah Forest highlighted by the IPCC as a biome most at risk of collapse due to drying, warming climate (Lawrence et al., 2022).

Faunal recolonisation metrics indicate a high return of highly mobile species such as birds with 95% of species inhabiting the rehabilitated areas; however, occupancy does not mean that these areas are fulfilling sustainable habitat requirements for all species such as nest hollows (Nichols & Grant, 2007). The ratio of occupancy compared to residency for birds is lower than those found for reptiles by S. L. Cross, Craig, et al. (2020b), as presented earlier in this chapter. Indeed, the inability to 'restore' old-growth elements does not allow restoration to provide suitable habitat for hollow-bearing dependent forest species (Johnstone et al., 2013). Such species include the nationally threatened Baudins and, at-risk, Forest Red Tail Cockatoos (Department of Environment and Conservation, 2008), which has led to community concerns that mining cannot achieve like-for-like forest ecosystems restoration with the consequent permanent loss of biodiverse elements.

The issues of a full restoration of a functional ecosystem for such an environment, are well illustrated with regard to the return of invertebrate species to mined areas. For example, ant species have been seen to begin returning to pre-mining levels after initial disturbance, and it has often been assumed that this trend towards full restoration continues (Nichols & Nichols, 2003). However, longer-term studies have shown that other species' composition has not returned after 37 years of restoration efforts, and therefore convergence may require even longer timeframes, or maybe will not occur at all (Majer et al., 2013) (Fig. 11.19).

Whilst Alcoa have achieved advances in their ability to reinstate species into a highly diverse and complex forest ecosystem, their operation of large-scale strip-mining within a biodiversity hotspot has created significant challenges (see below). Though outcomes to date have been impressive, the company faces substantial challenges in meeting the requirements of new guidance documents such as the International Mining Standards (Young et al., 2022). In this regard, these Standards recognise that restoration cannot restore old-growth forest ecosystem values, since such values may take centuries to achieve, and are, in addition, problematic within an environment beset by climate change.

Case Study Lessons Learnt

Alcoa operates at a scale and pace that is challenging when dealing with high diversity, old-growth ecosystems. Mining activities result in the total loss of the forest ecosystem and result in juvenilisation of large areas of rehabiltated ecosystems located on a significantly altered, and reduced, regolithic profile with no evolutionary equivalent (Hopper et al., 2016). After five decades of operation, and in the light of major and impending impacts of climate change on the Jarrah forest regions, the company will need to address their multiple biodiversity impacts on this ecologically sensitive area.

Fig. 11.19 Snottygobble, *Persoonia longifolia* is a key tree species that is widespread in Alcoa's mined jarrah forest but is often missing from restored sites. Propagation methods have only recently been resolved that enable the species to be hand planted into rehabilitated sites. However, it is unclear how the number of plants can be generated to replant Alcoa's four-decade-old legacy sites where the species is lacking

As the southwest forests are the largest native remnant contributor to the 30% of the southwest Australian biodiversity hotspot that remains intact, the ecosystem is considered to be one of the top 10 Australian areas of those facing impending ecological collapse and is fast approaching critical tipping points (Laurance et al., 2011). How Alcoa will address, both scientifically and practically, the growing community anxiety about the loss of such an important ecosystem that provides many other important human services and economic returns such as recreation, tourism and water supplies, is an emerging issue that is unlikely to diminish compounded by the company's emerging need to address full and informed consent from Traditional Custodians. This growing concern is despite Alcoa's significant commitment to rehabilitation, with current recreational users of the forest rating even the long-restored areas as significantly 'low quality' for bushwalking and mountain biking compared to unmined areas (Rosa et al., 2020). The rehabiltated areas are widely perceived as 'looking like a plantation' due to the lack of understorey diversity old and large trees and natural landforms (Fig. 11.20).

The International Principles and Standards for the Ecological Restoration and Recovery of Mine Sites, provide key guiding principles that define where and when restoration should be applied, and provide the measures by which restoration and

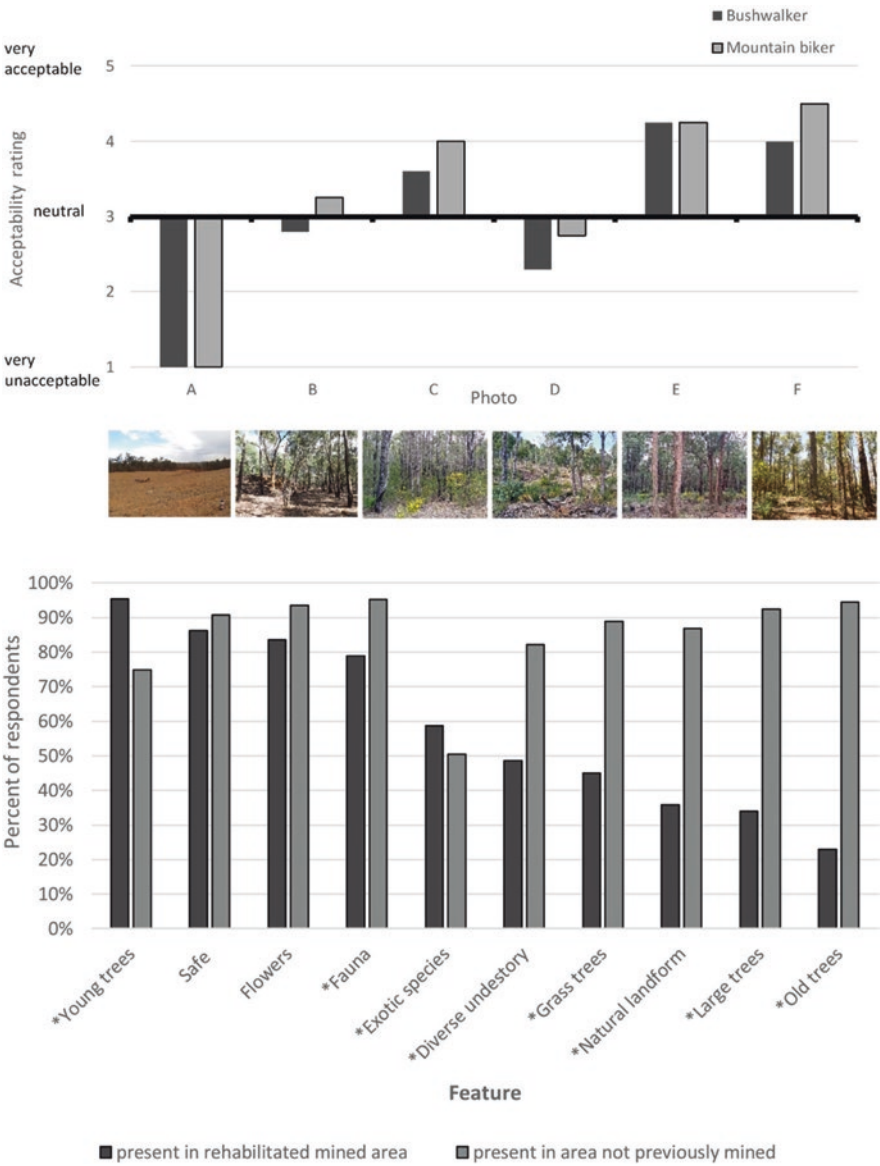


Fig. 11.20 Community perceptions of mined, restored and old-growth Jarrah forests (from Rosa et al. 2020). In the top panel, A is a recently mined area, B an older rehabilitated area using exotic species, C current restoration practices, D current rehabilitation practices with ecological thinning, and E and F unmined areas (historic disturbance from logging and fire)

rehabilitation outcomes can be assessed and globally benchmarked. How Alcoa will deploy these Standards and institute measures of success such as the Recovery Wheel (Young et al., 2022) will require the company to consider how they operate in native ecosystems and how their practices are commensurate with current and future community expectations. Importantly, as a large multinational mining company, Alcoa has obligations to the global community to demonstrate that it can operate within the frameworks of international best practice, such as the mining restoration standards (Young et al., 2022).

Case Study 5: Can Best Practice Be Achieved?

Case Study Background

Given the inter-related complexities of ecological restoration of mine sites, the question inevitably arises of how realistic is the expectation of restoration of ecosystems to their pre-mining status. Hanson Construction Materials (Hanson) is a sand extraction company that operates in Banksia woodlands north of Perth. These woodlands have been recognised as a Threatened Ecological Community (TEC) in Australia by the national government, and, as such, represent a highly biodiverse ecosystem within a broader globally recognised biodiversity hotspot (Myers et al., 2000). Banksia woodland is a finely balanced ecosystem and has limited capacity to survive the effects of long-term urbanisation, threats from degradation such as weeds, disease and climate change, or direct human impacts (Jason C. Stevens et al., 2016).

Case Study Description

Hanson, early in their mine development, adopted a restoration focus in order to achieve the highest possible restoration outcome following mining activities. This was a key issue which was introduced to allow operation of their mines in such a threatened ecological community. For almost 30 years, the company has invested in cross-disciplinary research, including seed science, plant ecophysiology, horticultural research and restoration technology development, in order to solve what were previously major technical impediments to reinstating a Banksia woodland community. As a result, Hanson now routinely achieves one of the highest post-mining biodiversity restoration outcomes in Australia. The restored areas routinely achieve greater than 100 plants per 5 m^2 with a diversity of more than 150 different native species (Fig. 11.21). Hanson's continuous improvement ethos has resulted in restoration programs approaching five-star outcomes based on the Society for Ecological Restoration standards (R. Young et al., 2022).

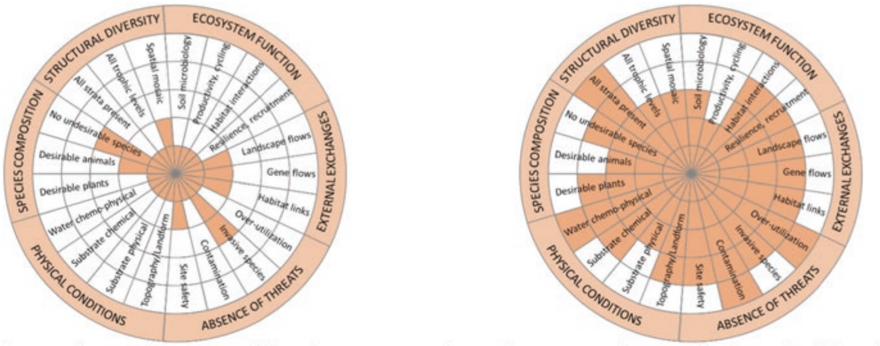
Prior to the commencement of the collaborative research program, the diversity and sustainability of post-mined Banksia woodland had low diversity and low

Fig. 11.21 Banksia restoration of sand mining operation, (**a**) post-seeding and (**b**) 15 years after commencement of restoration. (From Young et al. 2022)

abundance levels. Although some plant cover goals were achieved, the diversity of species at Hanson sites was well below that of natural recruitment in undisturbed woodland. This was attributable to a lack of key knowledge elements, including an understanding of seed viability, seed dormancy, growth medium requirements, soil profile reconstruction and invasive species control. The company was also hampered by an inability to scale up their restoration output through more effective mechanisation. However, through continuous research partnerships for more than 25 years, Hanson has achieved 'beyond compliance' outcomes in their restoration programs.

One of the challenges in Banksia woodland restoration identified early in the mine development, involved habitat-matching to ensure locally-sourced seed corresponded with the species in the local native reference ecosystem. This challenge was further complicated by the ecosystem being an 'old, climatically buffered infertile' landscape (Hopper et al., 2016). Banksia woodlands are not succession-based ecosystems, meaning that restoration requires a 'single pass' operation to reinstate species composition. For example, it is critical in developing restoration protocols to achieve the highest possible seedling recruitment levels in the first year of restoration. In addition, Banksia woodlands are highly susceptible to weed invasion. The Hanson research partnership developed a weed management strategy that is now routinely applied to achieve minimal weed impacts across restoration sites.

Once the reference ecosystem was established and a single-pass restoration approach was adopted, the restoration sites were able to be rebuilt from the ground up. Adaptive research was used throughout the process, and this continues to be undertaken. For example, after initial earthworks, surprising levels of previously undetectable compaction, termed 'cryptic compaction', were discovered in replaced topsoil. The observation of this problem resulted in more careful land forming through ripping and replacement of topsoil to optimise plant root development into the soil profile. This practice has been found to avoid compaction and irregular root development patterns. Ensuring adequate root development resulted in a five- to eight-fold increase in seedling abundance. Measuring soil impedance is now a routine approach used by the company. Early learnings also guided improved outcomes such as the finding that topsoil must be stripped with precision to less than 10 mm



Pre-research restoration assessment (2.0 stars)

Restoration assessment after research implementation (3.5 stars)

Fig. 11.22 Pre- and post-restoration ecological assessment of one of Hanson's sand quarries as per SER's standards for mine site restoration. (From Young et al., 2022)

in order to optimise native propagule abundance. This practice had a bonus in that the topsoil could be respread at a shallower-than-expected depths (less than 5 cm), thus extending topsoil availability for restoration. To ensure topsoil continues to be established to the correct parameters, operator-training systems on seed burial effects and topsoil stripping and replacement depths have been established in the company.

As discussed in detail in Chapter 12, the reliable seed supply chains that provide this crucial resource in the necessary quantity and quality (i.e. with appropriate collection or growing protocols, processing techniques, viability testing and germination protocols) is a key limitation in many ecosystem restoration projects. Prior to Hanson's two decades of research support, there was little known about how to germinate the seeds of many species and how to optimise seed-based restoration. Although research continues on problematic species such as those with seed dormancy, substantial improvement in seed use efficiency has reduced the dependence on wild-collected seed and optimised seed use, including use of wind-break fencing to prevent seed wastage as a result of wind erosion.

The nett effect of the integrated research program and operational improvements has improved Hanson's restoration outcomes from 2.0 stars to 3.5 stars (Fig. 11.22), with some sites achieving near 5-star status. Research and operations continue to strive to improve outcomes from the current benchmark.

Conclusion

A simple 'one-size-fits-all' solution to global mining restoration programs is unlikely to achieve environment outcomes that are fit for purpose and meet society's expectations. Mines and the ecosystems in which they occur vary at the biome, national and regional scales, with individual mining projects needing to draw upon location-specific solutions. Each mine is likely to have unique knowledge needs associated with (i) their particular climate and geology, (ii) the composition of waste material generated, (iii) the diversity, composition and complexity of pre-mined ecosystem and (iv) the regulatory and socio-economic environment of the region.

The timely restoration of functional, biodiverse, representative and self-sustaining native ecosystems on post-mining landscapes, now represents a major challenge to the global mining industry. Through wider adoption of standards to guide mining restoration, such as SER's Mining Standards document (Young et al., 2022), mine site ecological restoration can be achieved more effectively and efficiently. As demonstrated in the case studies presented in this chapter, key aspects of the Standards include:

1. **The need for effective pre-mining understanding of biodiversity context and ecosystem function.**

This enables restoration components to be integrated within an operational framework and provides data for key components of the restoration program. This includes the early identification of key knowledge or resource gaps, for which solutions can be derived during the project life.

2. **Using mine waste materials as a resource for developing functional soils and substrates.**

From improved pre-mining understanding of the site, more effective use of topsoil stockpiles and mine waste material can be achieved with the potential for substantial cost savings. For projects with toxic or plant-hostile waste materials and substrates, this improved understanding early in the project can assist with the development of appropriate/realistic mine closure time frames or enable effective intervention well ahead of mine closure.

3. **Early establishment of an integrated monitoring system for the restoration trajectory linked to adaptive management.**

Monitoring must be adequately planned for and executed from the mine planning stage through to the point of full ecological recovery, which may be decades after cessation of mining operations in some cases. This needs to extend beyond plot-based measurements of floral diversity and growth to consider spatial mosaics of the landscape, ecosystem function and as a result, the resilience of the restored ecosystem. Monitoring then informs future practice through development of adaptive management loops.

Overall, a more strategic, science-practice interface and strategic investment are required for the project proponents to aggregate and apply appropriate and effective knowledge and learning that ensures that environmental decision-making is relevant to the site, biome and land impact created well ahead of closure deadlines. Early adoption of mining restoration standards (Young et al., 2022) will enable proponents to develop a restoration program that (i) is both efficient and effective, (ii) embeds social values that assist in protecting the Social Licence to Operate, (iii) delivers the best ecological outcomes, (iv) minimises the risk of restoration failure and (v) leads to more timely mine relinquishment, which will minimise commercial risks and future liabilities.

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Part III
Other Important Aspects of Restoration

Chapter 12
Strengthening the Global Native Seed Supply Chain for Ecological Restoration

Simone Pedrini, Danilo Urzedo, Nancy Shaw, Jack Zinnen, Giles Laverack, and Paul Gibson-Roy

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Summary and Key Lessons
A global push to restore degraded terrestrial ecosystems requires an enormous and ever-increasing quantity of native seeds. Unfortunately, in most cases, such precious resources are not available in the quantity, quality and diversity required to support such restoration efforts. Limitations to seed supply have been identified in many countries, and numerous public and private initiatives of varied scope and magnitude have embarked on the journey of addressing the issues currently crippling the native seed supply chain. This chapter presents Case Studies of five prominent native seed supply systems that have developed independently in different parts of the World (Brazil, Western United States, Midwest United States, Europe and Australia) in the hope of providing useful guidance and inspiration for the improvement of existing and emerging native seed markets.

Although it is difficult to generalise from such diverse ecological and geopolitical scenarios, we suggest that a few key lessons can be drawn from these Case Studies. These include:

(i) where feasible, large-scale farm production of native seed is an excellent approach to increase seed availability, improve quality, decrease costs and safeguard natural populations from risks associated with overharvesting;
(ii) a pragmatic approach from regulators and some academics, for the development of effective regulatory frameworks should:

- meet the need to preserve natural ecosystems in terms of regulating wild collections, mandating the use of appropriate genetic material, and providing a definition for 'appropriate genetic material', which might vary among regions and countries;
- provide guarantees to seed users through quality assurance and certification schemes;

- be economically viable for suppliers who need practical seed transfer zones and realistic seed procurement plans; and
- clearly differentiate between native seeds and similar competing material that is not appropriate for ecological restoration, including varieties or cultivars and exotic species;

(iii) the promotion of community-led native seed supply chains, which highlight the role of local and Indigenous groups in co-creating regional arrangements to strengthen seed sources whilst promoting local engagement with political processes to transform social injustices.

Overall, a closer collaboration and constructive dialogue between seed suppliers, users, policymakers and academics, at local, national, and international levels, will hopefully lead to more interconnected and reliable native seed supply chains, capable of supporting the increasing demand for native plant material needed to meet global restoration targets.

Introduction

Native seeds are a crucial resource in supporting global restoration efforts. The UN Decade on Ecosystem Restoration has stimulated and re-energised ambitious pledges at national and international levels to restore degraded ecosystems over vast stretches of landscape (UN decade on Ecosystem Restoration, 2021). However, such ambitious initiatives will most likely struggle to deliver meaningful outcomes unless major knowledge gaps are filled, logistical issues solved and capacity development improved. In this respect, the supply of native seeds and plant material is a major bottleneck to achieving effective ecological restoration of diverse and complex ecosystems over large scales (Pedrini & Dixon, 2020).

Notwithstanding the increased global commitment for restoration, many initiatives across the world, which have been advertised as environmentally sound and ecologically appropriate, are falling short of their promises and expectations (Jones et al., 2018). One example is shown by carbon farming projects, whereby considerable areas have been planted to capture atmospheric CO_2 by means of plant biomass carbon sequestration. These projects have attempted to introduce restorative principles such as the use of natives and species diversity for the provision of plant matter (Renwick et al., 2014). However, the selection of species is often limited to a handful of trees and a few shrubs, whilst understory grasses and forbs, (both annual and perennial), are usually neglected. Even grassland restoration projects typically rely on a limited species pool compared to the diversity of the reference community (Gann et al., 2019). While such shortcomings are sometimes due to neglect, it is undeniable that in most initiatives, limited species diversity is not through lack of knowledge or aspiration, it is because of a pressing lack of seeds. Practical shortcomings are also not just limited to species diversity, since even for species commonly used in restoration, the quantity of available seed is not adequate to satisfy increasing demands. In addition to restricted supply, at times the origin of seeds might not be environmentally appropriate; available seeds might be coming from completely different ecological environments, carrying with them the potential risk of maladaptation and failures in plant establishment. Furthermore, if seeds of the right species and origin are available in sufficient quantity, the seed quality could be low. It has been found that many factors, such as poor seed set due to unfavourable weather or unhealthy populations, inappropriate late or early collection timing, in conjunction with poor handling and storage practices, can markedly affect seed quality (De Vitis et al., 2020; Pedrini et al., 2020).

Problems related to native seed diversity, quantity, origin, and quality are well-known by seed suppliers, researchers, local communities and restoration practitioners. As a result, many public initiatives, grassroots organizations, and private enterprises across the world have been working on a variety of solutions to overcome these hurdles (Pedrini & Dixon, 2020). It is clear that research on seed biology and ecology can improve our understanding of phenomena such as seed maturation and dispersal, germination requirements, storage behaviour, and dormancy alleviation, whilst genetic studies can provide good information on appropriate provenancing strategies. However, without the support of government institutions, well-regulated markets and committed commercial native seed suppliers, including private, public or NGOs, native seed supplies won't be economically viable.

This chapter provides Case Studies from different parts of the World (Fig. 12.1) where native seed supplies have, in the main, developed independently from one another and under differing circumstances, resulting in a wide variety of divergent approaches and convergent solutions.

The aim of this discussion is to provide examples and potential templates for the development of reliable seed supply systems in areas where these currently do not exist and, in doing so, possibly provide inspiration for countries and regions attempting to improve their own local native seed supply. In developing countries, the implementation of native seed supply systems or regional networks goes beyond business models for restoration, and represents an opportunity for community participation to promote environmental justice in Indigenous and community territories.

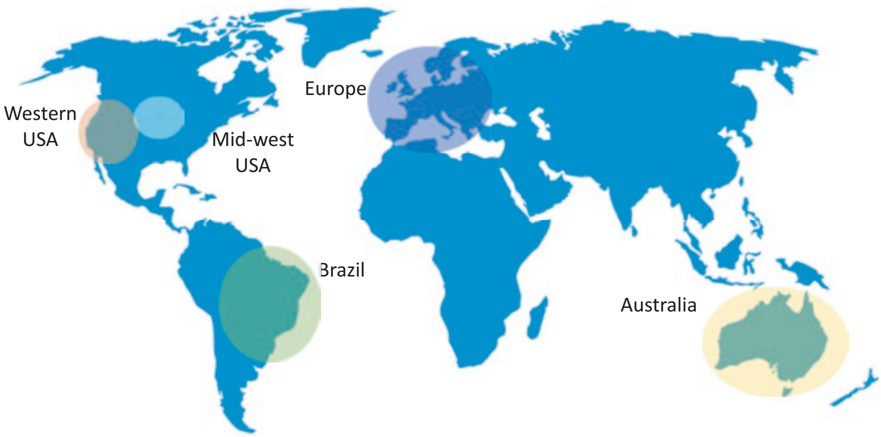

Fig. 12.1 Geographical location of the case studies presented in this chapter

The Case Studies

In our first Case Study (1), we present a picture of how local engagements have had significant ecological, social, and political outcomes in Brazil, noting how community-led networks collect and supply seed for numerous restoration initiatives across the country. The second Case Study (2) focuses on the world's largest native seed supply system, which has been established in the Western United States since the 1940s. Revegetation of vast areas of publicly managed land have created a situation whereby a number of very large seed buyers can provide the stimulus (and market) to sustain a well-developed native seed supply that relies heavily on seed production farms and supports numerous native seed businesses. Still in North America, Case Study 3 is located in the highly disturbed, agricultural Midwest region of the Unites States, where the demand for native seeds for publicly and privately funded restoration initiatives, mostly focused on native tallgrass prairie, is supplied by highly advanced and dedicated native plant nurseries. As with the agricultural Midwest in USA, in Europe (Case Study 4), the primary demand for native seeds comes from grassland restoration projects and many European countries have independently developed native seed supply systems (e.g. VWW in Germany and Vegetal Local in France). Recently, private native seed producers from across the continent have formed a network of knowledge sharing and mutual assistance to face common challenges, such as scaling up production, customisation of equipment, establishing seed transfer zones, and promoting the implementation of appropriate regulations. In contrast to North America and Europe where most seeds are cultivated on farms, Australian seed-based restoration relies primarily on native seeds collected from natural populations (Case Study 5). The industry in Australia is currently highly fragmented and relatively small, but recent initiatives aim to develop native seed farms and introduce seed quality testing measures.

Case Study 1: Brazil's Seed Networks for Landscape Restoration

Indigenous and community participation in seed supply systems offers significant opportunities to increase regional capacities for scaling up restoration efforts while achieving sustainable livelihood opportunities at the local level (de Urzedo et al., 2021). In Brazil, more than 12 million hectares of degraded lands on private property must be restored to meet the legal requirements of the Forest Code (Law 12,651/2012) that obliges landowners to protect specific portions of the native vegetation. The most notable environmental liability is concentrated on the Amazon agricultural frontier, representing 4.8 million hectares of degraded ecosystems (Soares-Filho, 2013). This domestic restoration pledge is also part of Brazil's contribution to its international environmental commitments to be achieved by 2030, as exemplified by the Paris Agreement (UNFCCC, 2014). To implement this ambitious target, the Federal Government has established a set of political instruments for law enforcement, including a national facilitation plan and a geospatial information system (Ministério do Meio Ambiente, 2017).

Co-creating Native Seed Networks in Brazil

Achieving Brazil's restoration goal would demand up to 15.6 thousand tonnes of native seeds over the next decade requiring the engagement of more than 57,000 collectors per year (de Urzedo et al., 2020). While well-designed seed supply chains may foster restoration actions and regional development opportunities, the current Brazilian native seed sector is considerably limited by the lack of seed availability, both in quantity and quality, to meet large-scale restoration needs (Freire et al., 2017). One of the interventions to innovate seed supply systems has emphasized the inclusion of multiple stakeholders through regional seed networks (Schmidt et al., 2019). The first national action to build a native seed sector occurred in the early 2000s, when the Ministry of the Environment funded the creation of eight seed networks in different biomes throughout the country (Piña-Rodrigues et al., 2020). Seed networks were conceived as decentralized initiatives with the engagement of varied organizations and stakeholders, primarily researchers and practitioners, to elaborate seed sourcing strategies and to facilitate regional interventions (de Urzedo et al., 2019). In each Brazilian biome, these seed networks have mobilized incentives based on seed technical expertise, capacity building and technological developments. As a result of this domestic incentive, other seed networks spontaneously emerged elsewhere in Brazil, through place-specific operations, local partnerships and regional restoration market demands (Piña-Rodrigues et al., 2020).

Community-Led Seed Supply Systems

Most emerging seed networks arise from grassroots action. These have the advantage of including local knowledge, practices and participation in seed collection as a driver to boost supply systemsand generate socioeconomic gains for local communities (Schmidt et al., 2019). The Xingu Seed Network in the south-eastern Amazon demonstrates how these community-led production chains can integrate large-scale seed collections with equitable participation from culturally diverse groups in large restoration actions. After 15 years of experience, the Xingu network reached an annual supply capacity of more than 25 tonnes of native seeds from 220 native species through the participation of 600 seed collectors (de Urzedo et al., 2021). The majority of the collectors are women from Indigenous and rural communities. They undertake seed supply operations and build management processes needed to coordinate multiple activities inside and outside their local groups (de Urzedo et al. 2016). The Xingu Seed Network initiate commercial practices in response to restoration demands, based on identifying and mobilizing regional restoration markets. A business management office links groups of collectors to agreed restoration projects by establishing contracts and ensuring the continuity of the demands for subsequent years (Fig. 12.2). These primary restoration markets are projects that apply direct seeding techniques using mixes of seeds from native and green manure species in order to accomplish mandatory restoration as required by national regulations (Campos-Filho et al., 2013), including the Forest Code and the National Environmental Policy (Law 6938/1981). These seed supply operations confirmed the feasibility of developing decentralized and participatory systems that allow local communities to coordinate their activities and establish local negotiations and agreements, respecting place-specific realities and needs (de Urzedo et al., 2021).

The Xingu Seed Network has become nationally recognized as a successful seed supply arrangement (de Urzedo et al., 2020). Since its inception, other grassroots initiatives elsewhere in Brazil have adopted this model to build bottom-up actions to mobilize resources and incentives to instigate participation in restoration activities (Schmidt et al., 2019). Over the last decade, dozens of seed networks have been co-created to connect local groups with restoration market demands in different biomes (Piña-Rodrigues et al., 2020). In central Brazil, a group of 15 seed collectors was initially mobilized in 2012 to supply plant material of diverse native grasses, forbs, shrubs and trees for small-scale savanna restoration projects in the Chapada dos Veadeiros National Park (Schmidt et al., 2019). Remarkably, after a few years of profitable activities, this group expanded and reached 60 seed collectors in eight different communities, leading to the formation of the Cerrado de Pé Association (Sampaio et al., 2020). Their business activities are facilitated by the Cerrado Seed Network, which has supported the commercialization of more than 36 tonnes of native seeds from 70 native species over the last 6 years for savanna and grassland restoration projects (Sampaio et al., 2020). Globally, these different Brazilian seed networks reveal an innovative mechanism to include local capabilities and knowledge in restoration practice leading to improved community engagement and the creation of conditions that lead to multiple environmental, social and economic benefits (de Urzedo et al., 2021).

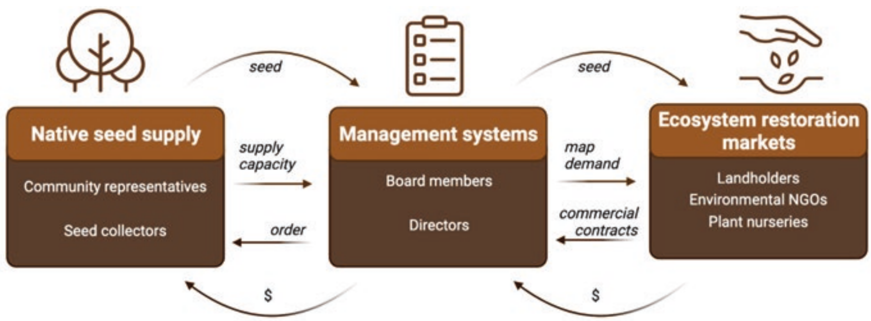

Fig. 12.2 Native seed supply operations of the Xingu Seed Network that link Indigenous and rural communities that collect and process native seed with regional restoration markets in the south-eastern Amazon. (Figure by Danilo de Urzedo)

Challenges for Scaling Up Community-Led Systems

Although there are promising local organizational systems and expertise available to upscale native seed availability in Brazil, restrictive regulations and lack of law enforcement on restoration outcomes illustrate the existence of clear barriers to transforming restoration pledges into actual outcomes at larger scales (Freire et al., 2017). The vast networks performing informal native seed collection and production are largely 'invisible' to the regulatory authorities in Brazil (de Urzedo et al., 2019). According to the National Seed and Seedling System (Law 10,711/2003), plant material suppliers must comply with various technical procedures and complete required documentation to report operations from selecting seed collection areas to support their trade operations. Unfortunately, many of these legal obligations, including quality control procedures, do not consider the specific conditions of native plant species and the socioeconomic realities of collectors and producers in regional and remote regions (Schmidt et al., 2019). For example, while Brazil's seed law compels producers to test the seed quality in a certified laboratory, there are only 16 laboratories officially registered to validate the seed lots in the country (de Urzedo et al., 2021). As a result, seed suppliers face significant limitations in meeting administrative requirements and other technical tasks to participate in restoration markets (Piña-Rodrigues et al., 2020). These restrictive technical regulations for ensuring procedures and quality standards can change local practices into informal protocols, leading to structural limitations in improving and expanding the emerging supply chain.

Addressing the seed supply shortage is not only a matter of enhancing technical knowledge and expertise, but also of collective work practices to develop partnerships and collaborations in specific projects (Sampaio et al., 2020). Establishing inclusive restoration markets fostered by domestic and international policies should consider new platforms to include local communities by integrating local knowledge with seed technology to improve seed supply operations (de Urzedo et al., 2021). The interconnections of diverse networks of stakeholders to enforce incentives and promote bottom-up approaches can provide enduring capacity to restore degraded lands on considerable scales (de Urzedo et al., 2019). In this respect, a transformative seed supply chain must be the trigger to enhance an equitable distribution of socioeconomic restoration benefits and diverse forms of participation in order to remake policies and incentives as tangible instruments for implementing successful restorations across geographical spaces (Schmidt et al., 2019).

Case Study 2: Western USA, the Largest Native Seed Supply in the World
About 47% of lands in the 11 Western states of the USA are publicly owned and managed by Federal agencies, with the US Department of the Interior (USDI) Bureau of Land Management (BLM) and the US Department of Agriculture (USDA) Forest Service (USFS) responsible for the largest areas (Fig. 12.2). Additional public lands are managed by State agencies and local governments. Federal lands are highly varied topographically and include forested mountain ranges and large expanses of the country's hot and cold deserts. The need for restoration and

rehabilitation results from a legacy of uncontrolled livestock grazing and increasing threats from exotic plant invasions, altered fire regimes, energy development, provision of transportation and energy corridors, recreation areas, burgeoning urbanization and climate change. To illustrate the scale of the problem, wildfires burned an average of 1,000,000 ha on BLM lands and 725,000 ha on USFS lands nationally from 2016 through to 2020 (National Interagency Coordination Center, 2020). In addition, noxious and other exotic weeds currently impact about 32 million ha of BLM lands (USDI Bureau of Land Management, 2021) and 1.4 million ha of USFS lands (USDA Forest Service, 2004) (Fig. 12.3).

Seed production for wildland seedings on public lands in the Western United States began in the mid-1900s, providing useful seed supplies, primarily of exotic grasses, for stabilizing watersheds degraded by excessive livestock grazing. These grasses were also seeded widely to improve forage on degraded rangelands and to combat the invasion of exotic weeds such as halogeton (*Halogeton glomeratus*). Over time, increasing numbers of native grasses, shrubs and some forbs came into use for wildlife habitat improvement, mined land reclamation, forage production and repair of degraded riparian areas. Grasses and forbs, primarily formal cultivar releases developed by the USDA Soil Conservation Service (later the Natural Resources Conservation Service) and other Federal agencies, were produced in

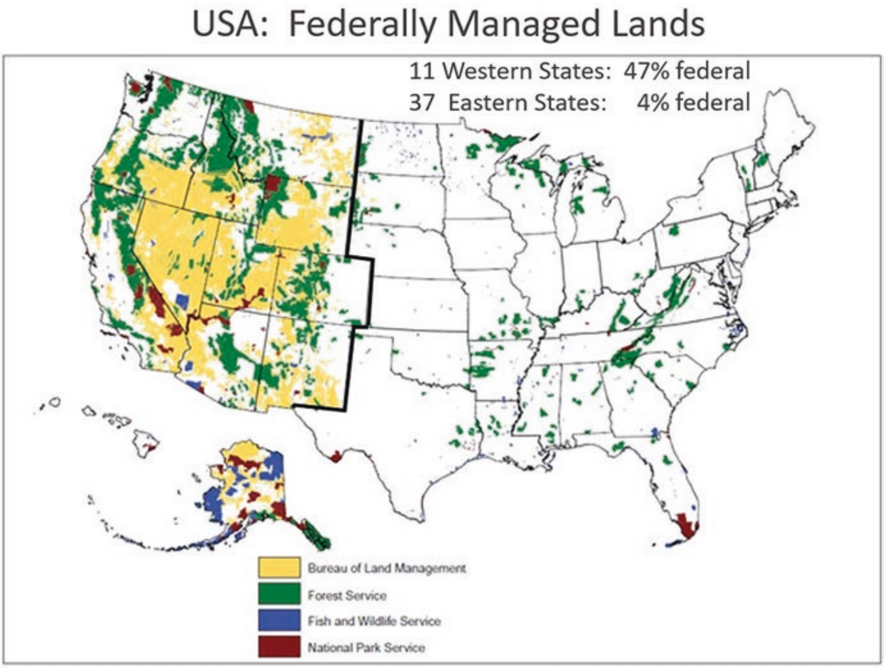

Fig. 12.3 Federally managed United States lands in the 11 Western states showing the land management agencies. (Source: GAO analysis of U.S. Geological Survey's National Atlas Web site data)

agricultural seed fields, while most shrubs and minor amounts of grass and forb seed were collected from wildlands. In the 1980s, the Association of Seed Certifying Agencies introduced the Pre-variety Germplasm Certification system (PVG) (AOSCA, 2003), which provides third-party verification of seed origin and genetic identity for wildland collected seed and its certification as Source-identified (SI). The PVG program also offered a Select class, which allowed simplification of releases of native plant materials exhibiting specific traits or those adapted to specific geographic areas in need of restoration. Both the Source-ID and Select certification categories are now widely used. Critical shortages of native seed for rehabilitation following extensive wildfires in 1999 and 2000 led Congress to direct the USDI and USDA to develop plans to increase native seed supplies and provide for their efficient management and use (USDI & USDA, 2002). In response, the Native Plant Materials Development Program was established by BLM and the Native Plant Restoration Program by the USFS. The National Seed Strategy of 2015, drafted by representatives of 12 agencies of the Plant Conservation Alliance's federal committee, built on the progress of these programs by outlining a framework for public/private coordination to assess Federal seed needs and private sector capacity to meet these increasing needs (Plant Conservation Alliance, 2015). The Strategy outlined steps to strengthen Federal restoration programs and the native seed industry and promote research, develop of decision-support tools, and stimulate communication to further these goals and better implement national initiatives with regard to climate change, invasive species, pollinators and the effects of rangeland fire. The National Seed Strategy is currently being updated to reflect recent progress, ongoing challenges, and the heightened need to better incorporate climate change into restoration planning.

Restoration in the Western United States

Executive orders, laws, agency policies and guidelines provide direction for seeding or planting on Federal lands, but directives and recommendations applicable to the use of native plant materials vary among agencies. Use of native seed is generally prescribed, but for the BLM and FS, exceptions are permitted when needed species, sources, or quantities are not available.

Procurement strategies are generally forced to differ between unplanned and planned seedings. Unplanned seedings, such as those following natural disasters, often utilize large quantities of commercially available native and exotic seeds. There is usually a short window to provide site stabilization or to preclude weed invasion, and consequently, readily available plant materials are procured and substitutions for desired species or sources are often accepted. Other problems arise when large-scale seedings are needed to resist exotic weed reinvasion following management control, for improving degraded rangeland where diversity has been reduced, or for generally enhancing severely disturbed wildlife habitat. Such projects may not be funded adequately or may have insufficient lead time to permit procurement of preferred native plant materials. These exigencies arising from short project timeframes and fluctuating demands encourage seed collectors and producers to continue providing widely-used native and exotic species that can be grown on large scales and can be reliably marketed at the expense of native species of the required diversity and appropriate origin.

Planned seeding and planting projects on public lands often include reclaiming closed roads, attending to campground rehabilitation, upgrading habitat improvement for rare species, together with many other smaller-scale efforts. Appropriate timelines and budgets for such projects often allow for more careful determination of relevant plant materials in terms of species selection, geographically appropriate sources, available quantities and plant material type, and allow procurement using one or more of the following options, which can sometimes require several years:

- Ordering and accessing material available from private seed and plant suppliers;
- Organising wildland seed collection from selected source areas by agency personnel or contracted collectors;
- Arranging seed farms or seedling production at private sector facilities.

The framework provided by agency native plant development programs, the National Seed Strategy (https://www.blm.gov/programs/natural-resources/native-plant-communities/national-seed-strategy) and recent national initiatives provide direction to restore resilient, biodiverse native communities using genetically appropriate seeds. These have been crucial steps in stimulating efforts to strengthen all links in the native seed supply chain from the planning phase to application in the field to improve the restorative value of both unplanned and planned seedings. Challenges to the use of genetically appropriate native seed include (i) wide and unpredictable annual fluctuations in federal funding, (ii) difficulty in predicting the occurrence, location and extent of wildfires and other natural disasters, and (iii) for many native species, inadequate knowledge of their seed biology and their appropriate use in restoration.

Seed Needs Assessments, Species Selection and Seed Sourcing

Tools such as seeding records and fire histories for specific areas or seed zones, fire risk models and long-term weather predictions, can be used to improve estimates of annual seed needs for unplanned emergency situations. Species selection for restoration plantings is guided by documents including SER's International Principles and Standards (Gann et al., 2019), available taxonomic and ecological literature and databases, site and reference area inspections, and local restoration experiences (Erickson & Halford, 2020). Increasing research and practical knowledge of the biology and use of common restoration species as well as species that are coming into use are expanding the choices available. Selection of seed sources that are potentially adapted to the specific planting site conditions of topographically diverse regions, is aided by the development of empirical seed zones for individual species based on recognised adaptive characteristics (Erickson & Halford, 2020). Provisional seed transfer zones, based on biogeographic regions with similar climatic and environmental characteristics, can be used for the many species lacking empirical seed zones (Bower et al., 2014). The use of seed zones for planning can improve restoration outcomes, enabling the sharing of plant materials within zones, to provide clear and defined markets for seed producers, allow for economies of scale and aid in stabilizing the native seed market (Erickson & Halford, 2020). Empirical and provisional seed zone maps are available on the Western Wildland Environmental Threat Assessment Centre's (WWETAC) website (USDA Forest Service, 2021a), and the Climate Smart Restoration Tool aids for the selection seed lots adapted to current or predicted future conditions at the planting site (USDA Forest Service, 2021b).

Wildland Seed Collection

Speculative commercial collection on public lands requires the purchase of permits from agency offices. Permits specify the areas for collection and the maximum quantity of seed that can be collected to prevent population depletion. Agency contracts for collection specify the amount required, collection protocols and required site data. Landowner permission and possibly payment is required for seed collection on private lands. Wildland collection totals range from a few pounds for specific forbs, to thousands of pounds for dominant shrubs such as big sagebrush (*Artemisia* spp.).

Collectors are responsible for training their crews. Protocols are available that prescribe (i) collection and seed cleaning procedures as well as equipment maintenance for individual species, (ii) appropriate guidance for collection in order to maximize the genetic diversity of the seed lot, and (iii) limitations on harvesting to ensure sustainability of the seed source (USDI Bureau of Land Management, 2018). At a practical level, mobile mapping applications and weather prediction tools aid in planning collection trips and recording site locations.

Shrub seed is manually collected from native stands as efforts to produce shrubs in agricultural settings have not proven economical, though there are some exceptions. Some extensive shrub populations, such as big sagebrush (*Artemisia tridentata*) and antelope bitterbrush (*Purshia tridentata*), which are in demand for local seedings, are frequently collected. Limited research has been conducted on managing such stands for seed production; their use to date has been controlled primarily through the process of issuing permits. In this respect, controlled wildland shrub seed collection provides seeds lots of known origin with high genetic diversity. The BLM, for example, now purchases big sagebrush seed on a seed zone basis, thus quickly obtaining large quantities of seed for immediate use in appropriate regions. Sagebrush procurement from 2015 to 2020, for example, averaged about 16,800 kg of pure live seed (P. Olwell, USDI BLM, personal communication, 2021).

Some easily harvested grasses and forbs that occur as extensive monocultures on wildland sites are harvested manually for direct use in seedings. On level terrain, mechanical harvesting is sometimes possible if weeds are not problematic. Most grasses and forb seed, however, is produced in seed farms, and only small wildland collections are required to establish seed production fields. These collections are made in-house by agency personnel, contractors or trained volunteers. There is a need to increase collaborative collection and curating of seed collections of adequate size to provide stock seed when needed to shorten the seed production timeline.

Concerns with wildland seed collecting, in addition to annual variations in production, are several: (i) costs of collecting from remote areas are high, particularly as the timing of seed maturity or crop condition may be difficult to predict; (ii) many species mature over an extended time period, but multiple collection trips may be cost-prohibitive. On varied terrain, however, plants at different stages of seed maturation are commonly available; (iii) populations of many species are being lost or reduced due to wildfires or weed invasions, introducing genetic limitations and the area from which seed can be safely collected; (iv) exotic species invasions increase the risk of collecting weed seed that may be difficult to remove from seed lots during cleaning; seed lots containing prohibited noxious weed seeds cannot be sold; (v) in many areas, there are few or no professional seed collectors; (vi) inconsistencies in permitting procedures among offices can create confusion for collectors; and (vii) training for seed collectors is limited.

The BLM's Seeds of Success program (Barga et al. 2020) is a national public-private collaboration among federal agencies and non-federal partners that provides seed collection training for recent college graduates. This program has the aim of conserving the nation's plant resources and providing seed for restoration (Haidet & Olwell, 2015). To date, more than 278 teams have made more than 26,000 seed collections of 5800 species from 43 states to support this program (Plant Conservation Alliance, 2021). Seeds are placed in seed banks at the USDA Agricultural Research Service storage facilities for long-term conservation and to provide a working collection that is made available to researchers, educators and plant breeders. Excess seed beyond that required for storage (10,000 pure live seeds) can be used for seed farming or immediate planting by the agency field office where the collection was made (Haidet & Olwell, 2015). Most SOS seed is cleaned by the USFS Bend Seed Extractory, which has had experience cleaning 3400 different native species (K. Harriman, personal communication).

Seed Production in Farm Settings

Many native grasses are grown economically in seed fields using protocols modified from production practices developed for related pasture and forage species, involving readily available planting, harvesting and seed cleaning equipment. Widespread native grass species that are easily grown and commonly used in seedings are produced in large quantities. Source-identified material for specific seed zones and species that are either infrequently requested, more difficult to grow, or low yielding and therefore more expensive, is generally grown on contract. From 2016 to 2020, BLM grass purchases averaged 195,900 kg (432,000 lbs) for cultivars, 165,000 kg (364,000 lbs) for select germplasms, and 45,000 kg (99,000 lbs) for source-identified material (P. Olwell, USDI BLM, personal communication, 2021). Native forb use by federal agencies remains limited, but has been increasing over the last two decades in response to growing emphasis on conserving biodiversity and protecting pollinators. The forbs which are currently seeded are primarily exotic legumes planted for forage and soil stabilization (Fig. 12.4).

Increasing native forb seed production and use is problematic. Forb species are numerous; they represent many diverse plant families and often include subspecific taxa or multiple ploidy levels. They are characterized by a variety of fruit and seed types, dormancy syndromes, growth habits, and pollinator, seeding and establishment requirements. Many perennial forbs are less competitive than grass species, and some require two or more years to begin flowering. Growers have developed protocols for producing some species but until recently, there has been little research directed towards the technology required for the use of individual species in revegetation.

Some progress is being made with selected species being grown in large quantities and at reasonable prices. Most native forb seed is grown and certified as Source-Identified using wildland collected seed, but an initial multiplication at a federal nursery or by a private producer may be necessary to produce enough seed to plant a seed field (Fig. 12.5). Research has identified cultural practices, including planting methods, irrigation requirements, pollinator identification and management, and harvesting techniques for several high priority species. Results have been published in a wide variety of journals and are being synthesized in online databases. There is

Fig. 12.4 Seed farming of native forbs: Forb seed is collected from wildland stands and therefore source identified. The initial multiplication may be produced at a federal or private facility for the production of private sector seed farms. (Pictures by Nancy Shaw, label provided by the Utah Crop Improvement Association)

a critical need for additional research on the addition of native forbs to seedings to improve establishment success and to increase users' willingness to add them to seed mixes.

Seed Procurement and Management

Improving the availability of genetically appropriate seed when needed for future planned or unplanned seedings, or in order to reduce market fluctuations, requires improved procurement tools, facilities and reliable funding. Better prediction of seed demand, in addition to procurement tools that permit forward contracting and expanded storage capacity to maintain seed quality, are aiding this process. Increased planning within or among public agencies or public/private cooperatives that operate in similar landscapes is needed to scale up seed orders as growers are often reluctant to produce small acreages or plots. Cooperative planning also helps to reduce costs and ensure that all seeds will be used effectively.

One approach introduced by the USFS and BLM to increase the availability of species, sources and quantities of native seed available when needed, is the development of procurement tools using forward contracting. Use of new purchasing instruments such as the BLM's Indefinite Quantity/Indefinite Delivery (ID/IQ) contracts and the Forest Service's Blanket Purchase Agreements, can ensure production, particularly of species and seed sources that are not regularly available on the open market. Instruments of the two agencies differ in detail, but both facilitate contracting with a pool of pre-qualified experienced growers, thus streamlining the contracting process. Stock seed, composed of seed from a single, or preferably multiple populations, of the species from a specific seed zone and in some cases a specified elevation band, is certified as Source-Identified, cleaned, and tested for purity, viability and weed content. The seed is then provided to the successful bidder, or in some cases, the seed might be collected by the grower as part of the contract. The grower is guaranteed payment for the production of a specified amount of seed, and

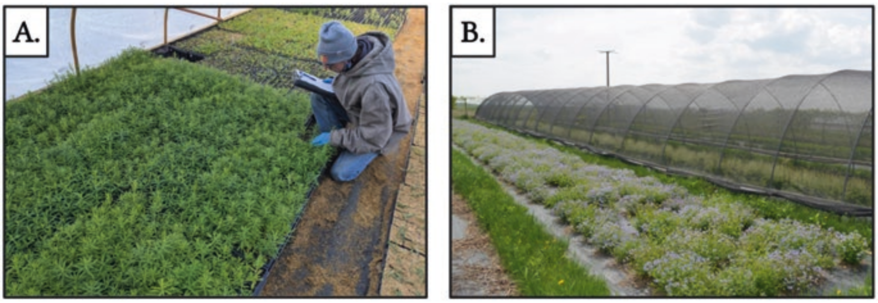

Fig. 12.5 Private native plant nurseries provision many of the materials used in restoration. (**a**) Plugs of *Asclepias tuberosa* are inspected by a worker for field production. (**b**) SPAs of *Polemonium reptans* (outside, foreground) and *Mitella diphylla* (within hoop house)

the agency retains first rights of refusal for any seed produced beyond that amount. Any excess seed can be sold on the open market. The seed thus procured is placed on the agency's seed inventory for purchase by field offices when needed. Payments are returned to the fund for further contracting. The BLM's ID/IQ contract was instituted in 2019 to accommodate production for seed zones in eight ecoregions. In summer 2021, there were 94 taxa from 40 seed zones under production (Plant Conservation Alliance, 2021). The first seed produced via the ID/IQ delivery orders arrived at the BLM seed warehouse in the Fall of 2021, providing readily available seed supplies adapted to areas experiencing frequent fires or other disturbances.

Increasing short- and long-term seed storage capacity can also aid in providing a greater diversity of species and seed sources when needed. BLM seed storage facilities now provide 1.18 million kg of storage under ambient warehouse conditions and 61,000 kg of cold storage (USDI BLM National Seed Warehouse System, unpublished data). Greater capacity is required, however, to provide adequate long-term storage. Capacity for improving native seed supplies and stabilizing the native seed market is expected to improve with the recent passage of the H.R. 3684 Infrastructure Investment and Jobs Act, which makes available \$200 million "to establish and implement a national revegetation effort on Federal and non-Federal land, including to implement the National Seed Strategy for Rehabilitation and Restoration." The Act also makes available \$200 million for invasive species control (House – Transportation and Infrastructure, 2021).

Case Study 3: Native Seed Supply and Ecological Restoration in the Midwest of the USA

The Midwestern United States lies in the inner continent of North America, which historically included vast stretches of woodlands, wetlands and tallgrass prairies. Prior to the historical movements to conserve natural areas, much of the Midwest was rapidly converted to agriculture, urban centers, or to other human land-uses in the nineteenth Century (Schwartz, 1997). The Midwest is now a hub of industrialized agriculture with intense human impacts felt across the landscape. The near-complete destruction of tallgrass prairies, a figure that exceeds 99%, occurred in most Midwestern states (Samson & Knopf, 1994). Loss of wetland and savanna habitats have been similarly severe, and degradation of remaining habitats through the loss of biotic integrity exacerbated by invasive species, is now a serious a conservation concern.

Because of the intensity of human impacts on natural areas, the Midwest is a restoration hotspot and has become a laboratory for restoration praxis. In fact, the Midwest could be considered the birthplace of ecological restoration, starting with the Curtis Prairie Restoration in Wisconsin (Allison, 2002). Nearly 100 years after this first restoration, the emphasis on tallgrass prairie restoration continues and has been harmonized with an advanced native plant industry. Wetland, savanna and woodland restoration are also practised in the region, but prairie restorations are a supreme example of landscape coverage and financial impact and are the primary focus of this Case Study.

Restoration Drivers and Practitioners

The restoration industry across the Midwest is financially driven by governmental programs and private interests. The former sources range from Federal, State and municipal programs, whereas the latter encompasses private organizational or individual restoration efforts that are not government-incentivized or mandated. Non-profit organizations, such as The Nature Conservancy, together with private businesses, often have excess land holdings that they want to improve for wildlife habitat or for public relations. Many private individuals conduct ecological restorations for environmental, pragmatic, or hobbyist reasons. In contrast, some restoration efforts are the direct result of government incentives or mandates. The most pertinent example of government-supported restoration is the Federal Conservation Reserve Program (CRP). CRP is a conservation easement program that contracts landowners to convert croplands to a more natural ecosystem using native plant species, by retiring and restoring the ex-agricultural land. CRP was developed under the Food Security Act in 1985; the primary goals of the legislation were to reduce soil erosion in highly erodible land and reduce surplus cropland. CRP landowners receive annual payments for a 10-year contract and are also partially reimbursed for the cost of land conversion. Practices within CRP address different habitat types and conservation targets. For example, the permanent native grass practice (CP2) targets the permanent establishment of native, warm-season grasses to limit soil erosion and reduce nutrient runoff, whereas the Pollinator Habitat Initiative (CP42) targets pollinator conservation through forb-rich plantings. Prairie or prairie-like habitats are commonly emphasized, but other habitats can also be targeted such as wetlands in the CP23 practice. CRP plantings cover millions of acres in the Midwest and are therefore critical drivers of the restoration and native seed markets (Gibson-Roy, 2018).

In addition, there is direct government support for restorations beyond the CRP. For example, State Departments of Transportation conduct roadside native seedings for habitat creation and beautification. Furthermore, because wetlands are protected under the Clean Water Act, compensation is required when organisations impact wetlands, often through off-site wetland restorations. These government programs collectively provide a foundation for consistent demand for native plant materials.

Practitioners using native seed in the Midwest are a mix of individuals across the private sector, non-governmental organizations and governmental institutions. Government agencies conducting restorations include Federal agencies such as the U.S. Fish and Wildlife Service, State-level governments, including natural resource departments, and local governmental organizations like city park districts. Compared to the Western United States, however, private landowners appear to be much larger buyers of seeds than Federal or State governments.

Midwest Native Seed Supply

These private and government-related restoration efforts in the Midwest have led to the development of some of the most advanced native plant industries in the world. For this work, native plant nurseries are key suppliers of native seeds. These are almost exclusively private companies that explicitly specialize in cultivating and promoting native plant biodiversity (Fig. 12.5). They typically sell non-cultivar native plant materials and large businesses are especially, tied to ecological restoration. Many of the seed mixes sold by native nurseries are marketed toward CRP programs. Native plant nurseries are scattered throughout the Midwest and comprise only a small fraction of total plant vendors. Nonetheless, these native plant nurseries are reservoirs of native plant diversity. Over 40% of the species native to the region can be purchased (Zinnen & Matthews, 2022), including nearly ¾ of species found in tallgrass prairies (White et al., 2018).

Native plant nurseries are essential for providing the materials for restorations in the region. Collection of seeds and plant materials from natural populations is often not feasible for businesses or restoration practitioners, because remnant habitats are limited in size and number and are often too badly degraded. Although practitioners and nursery workers can occasionally target natural or restored sites for seed collection, much of the seed is produced in seed production areas (SPAs). Native seed production is a valuable source of income to the native plant trade because it upscales the availability of native plant materials (Gibson-Roy, 2018; Zinnen et al., 2021). Most SPAs cultivate herbaceous, perennial species owing to the predominance of prairie, emergent marshland and savanna restorations (Fig. 12.5b). Major native nursery SPAs commonly span multiple hectares and provide seed to a large regional, rather than a restricted, local market (Gibson-Roy, 2018; Zinnen et al., 2021).

Challenges

Although the midwestern native industry is large, diverse and stable compared to other regions, it still has challenges with supplying native plant biodiversity. For example, the cultivation of wild plants in native nurseries can lead to unfavourable ecological restoration outcomes. Specifically, SPA cultivation can directly cause or intensify genetic problems in plant materials. The main concerns include the loss of genetic diversity, particularly in regards to genetic erosion and inbreeding depression, as well as shifting of plant traits in cultivation due to artificial selection. Such genetic changes can reduce the quality of native seeds by reducing their ability to survive and persist when planted in particular areas (Espeland et al., 2017; Zinnen et al., 2021).

An additional challenge linked to the previous genetic concerns is the availability of local ecotypes. Whilst some of the larger native plant nurseries use a network of growers that can provide more localized seed sources to practitioners, the use of regional or distant seed material is common in the region (Gustafson et al., 2001) due to pragmatic or financial constraints for both seed producers and users. Some extreme cases of non-local ecotypes include the occasional planting of cultivars, often varieties of dominant grass species, in restorations. Non-local ecotypes are likely to have poorer survival or performance in restoration settings or have disparate functional characteristics compared to local ecotypes.

Another major challenge is the insufficient supply of some plant taxa and functional groups. Many species are rare in cultivation, being found in only a handful of native nurseries. Moreover, these and similar species are nominally, but not functionally, available to practitioners. In other words, many species are available only as small volumes of seed, destined to be curiosities or specimens rather than to provide appreciable components of large-scale ecological restorations. Likewise, several functional and phylogenetic groups are underrepresented or entirely absent in seed mixes, as shown by sedges, hemiparasites and short-lived herbaceous species (Sivicek & Taft, 2011; Barak et al., 2017). Some restoration projects also trade characteristic species of one community type for another. For example, in compensatory wetland restorations, hard to obtain (or completely unavailable) shade-tolerant wetland species were replaced by widely available and showy prairie species (Tillman, 2021).

Similarly, financial incentives in restorations can exacerbate an overemphasis on "workhorse" species (Zinnen et al., 2021). Workhorse species are those that are frequently used in large volumes in ecological restorations, since they tend to have inexpensive and readily available seeds based on their agronomic suitability in SPAs. Workhorses also reliably establish successfully in restorations. In prairie restorations, examples are C_4 grasses such as big bluestem (*Andropogon gerardii*) and switchgrass (*Panicum virgatum*), as well as the showy wildflowers like purple coneflower (*Echinacea purpurea*) and black-eyed Susan (*Rudbeckia hirta*). These species can be naturally common or dominant in reference communities and are therefore important to ecosystem function. Nonetheless, workhorse overemphasis might come at the expense of ecologically valuable species that are difficult or impractical to source leaving them as underutilized species. This can be observed in seed mixes tailored to CRP programs, where typically, CRP mixes consist of the same suites of 15–25 charismatic prairie species, with few mixes (~10% available) reaching or exceeding 30 species (Fig. 12.6) in contrast to the many small prairie remnants (<1 ha) where more than 100 species can occur.

Together, these challenges lead to the broader issue of compositional mismatches between ecological restorations and midwestern habitat remnants. Such disparity could be further explained by the emphasis of most restoration projects on achieving partial recovery instead of striving for the complete recovery of a reference ecosystem, as defined by the International Standards for Ecological Restoration (Gann et al., 2019). In this respect, financial concerns are often a limiting factor, with diverse, species-rich or functionally unique seed mixes being commonly more expensive. This is of concern since practitioners must inherently limit the costs and establishment challenges of their restoration projects, an issue which is particularly

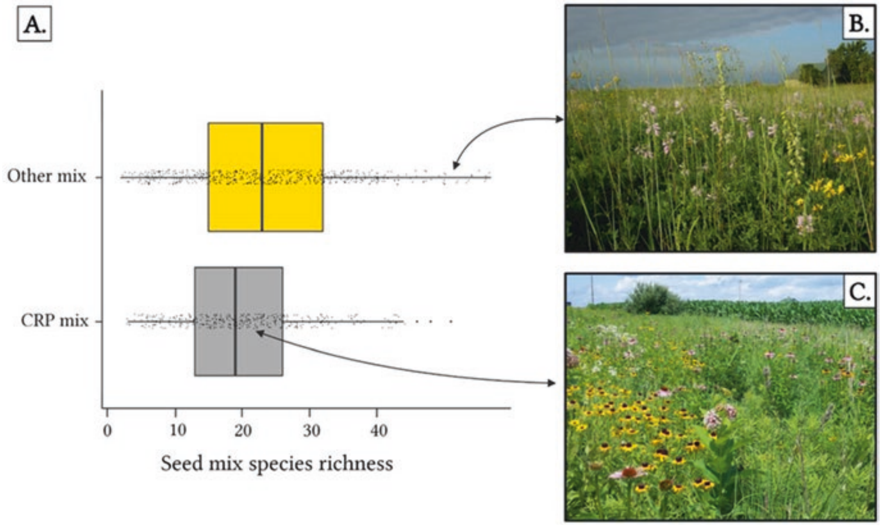

Fig. 12.6 CRP mixes are readily available from native plant nurseries but have poor to modest species diversity. (**a**) Boxplots comparing CRP and non-CRP seed mix native species diversity. Shown are the interquartile range (box), median (black line), individual seed mix richness (small black points), and richness outliers (large black points). These seed mixes are from database of seed mixes available in the Midwest, with 407 CRP and 624 non-CRP mixes represented. CRP mixes are significantly less diverse than non-CRP mixes. (**b**) A few restorations on the landscape utilize diverse seed mixes, such as this private restoration in north-eastern Illinois. It is marked by high species and functional diversity, as well as greater floristic quality due to the presence of late-successional, ecologically sensitive species. (**c**) Although showy, CRP plantings usually consist of <30 species, often of early-successional species and those that are cheap and easy establishers. Note the greater structural complexity in the diverse prairie (**b**) compared to the CRP planting (**c**). (Images by Pizzo Native Plant Nursery, Don Gardner, and Pheasants Forever Habitat Store)

relevant to government-incentivized restorations, where budgets are discrete and limited. The result is that many midwestern seed mixes and restorations simply lack the diversity and high-quality indicator species found in remnants. Often, late-successional and ecologically sensitive species are missing in ecological restorations, though there are some exceptionally diverse restorations in the region (Fig. 12.6b, c).

Lessons Learned and Synthesis

The Midwest experience highlights the importance of combined and continuous private, public and government support for restoration. Although the scale of restoration in the Midwest is smaller than in the Western states, the native seed industry is diverse and highly advanced, being able to support specialized native plant nurseries and provide a reliable seed supply. Its formation represents an intersection of government policy and private enterprises. It is also a region where non-incentivized and private restorations are common and form a substantial component of the market demand.

Case Study 4: The European Native Seed Supply Chain
In contrast to the other Case Studies presented in this chapter, Europe consists of many independent states, with 27 members of the European Union (EU) plus other countries including Switzerland, the United Kingdom, Iceland, Norway, Albania, Belarus, Ukraine and parts of Russia and Turkey. Being geographically and climatically very diverse, the European continent's environmental conditions range from the hot and dry Mediterranean basin to the frigid arctic and alpine tundra, and from the wet Atlantic coast to drier continental interiors. Consequently, the conditions and challenges for restoration and native seed production vary widely, with densely populated countries like the Netherlands having highly altered landscapes compared to other areas of sparse human population where the impact of people has been limited. Clearly, intensive management of most areas for grazing animals, arable agriculture and forestry over hundreds or thousands of years, has profoundly transformed natural ecosystems across the continent.

In addition to the range of restoration challenges across the continent, the supply of native seeds varies dramatically from country to country, with little or no activity in some countries to a mature and well-developed industry in others such as such as Germany, Austria and Switzerland. Development of restoration and native seed supply systems has generally taken place within individual countries with international developments in research and regulatory practices governed by EU seed regulation and environmental protection. The situation for native tree seeds is different from that of grasses, forbs and most shrubs. In the EU, many species of tree seeds are covered by the Plant and Forest Reproductive Materials (PRM) regulations (https://ec.europa.eu/food/plants/plant-reproductive-material/legislation/future-eu-rules--plant-and-forest-reproductive-material_en). These cover both native seeds for restoration planting in the zones of their collection and seeds for timber production or amenity use which may not be native to the zone where they are being used (Jansen et al., 2019). The trade in tree seeds has existed for longer than for other native species, and consequently is more regulated and has closer links between producers (collectors), the intermediate users of seeds (nurseries) and the end-users (timber producers and reforestation projects). In this Case Study, we will cover the development of seed supply systems of non-tree species, especially for the restoration of grassland ecosystems.

History

In the 1980s, research on restoration methodology established an approach to using native seeds which demonstrated the potential for species-rich grassland creation. By the late 1990s and early 2000s, ambitious programs with the goal of

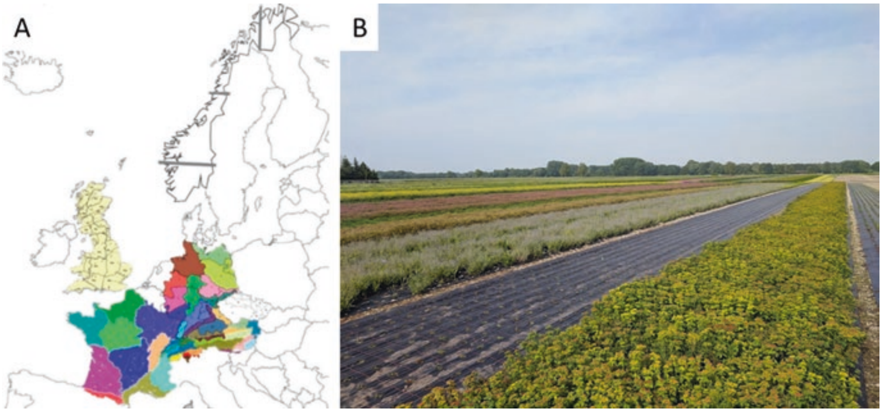

Fig. 12.7 (**a**) Native seed transfer zone across countries in Europe from DeVitis et al., 2017 (the zones for Great Britain are for native tree species). (**b**) Large scale production of native seeds in southern Germany. (Source: Simone Pedrini)

securing the germplasm of most of the world's wild species (as potential insurance against their extinction in the wild) resulted in the creation of an extensive network of native seed banks across Europe and the World. This endeavour was led mainly by research and conservation institutions such as the Millennium Seed Bank (UK, https://www.kew.org/wakehurst/whats-at-wakehurst/millennium-seed-bank) and resulted in the publication of guidelines and principles underpinning appropriate seed collection and curation that are still applied and are relevant to this date (ENSCONET, 2009).

Various initiatives by local government, NGOs and small private enterprises started collecting and propagating seeds of native flora across different European countries from the 1980s. While conservation seed banks were ramping up native seed collection for ex-situ conservation, private companies and NGOs started increasing native seed supply by adopting mechanical harvesting of semi-natural plant communities and, in conjunction with arable seed crop production. In countries such as Austria, Germany, Switzerland and the UK, native seed suppliers and users started to produce seeds of local ecotypes. Later, seed transfer zones, primarily based on biogeographical regions, began to emerge (Fig. 12.7). Similar zoning systems were more recently proposed for other countries such as France (vegetal local, https://www.vegetal-local.fr/), Norway, and Hungary (Jørgensen et al., 2016; Clément & Malaval, 2019; Cevallos et al., 2020).

Most of these initiatives have emerged and have been developed independently in various countries to address local legislative requirements and market needs including restoration, conservation, infrastructure development and urban landscaping. This resulted in companies and organisations independently developing similar approaches, strategies, and values. This state of affairs meant that, until recently, companies from different countries had limited opportunities to interact and collaborate on a systematic basis. However, the local nature of production and markets have limited direct competition, making increased interaction and knowledge exchange among native seed producers greatly beneficial. The European Union funded project NASSTEC (Native Seed Science, Technology and Conservation), provided the first opportunity for native seed suppliers across the continent to come together and ultimately led to the foundation of the European Native Seed Producers Association (ENSPA) in 2020. As of early 2023, 24 companies from 17 countries have joined the Association (https://native-seed.eu/).

Native Seed Market(s) in Europe

A recently published survey reported that native seed producers in Europe are mainly small, privately-owned companies usually formed as a result of the owners' genuine interest in conservation and restoration of natural environments and the desire to diversify farm production with alternative crops (DeVitis et al., 2017). There are some larger co-operatives (Austria) and non-profit organisations, but most enterprises are small owner-run farming businesses. Environmental NGOs have often been involved in harvesting from native plant communities, but are rarely involved in growing seed crops (e.g. hay time project https://www.yorkshiredales.org.uk/about/wildlife/projects/hay-time-project/). There has been little engagement of the larger agricultural or horticultural seed producers or plant breeders. The native seed sector is considered to be a small niche market beset with numerous production complexities, such as high species number and complicated origin requirements, that traditional agricultural seed companies are not equipped to handle. This may change as the native seed market expands and there is some involvement in production from larger producers of seeds of animal fodder crops.

Although national governments are large users of native seeds, for example in infrastructure restoration projects, there has been little intervention in the free market in the form of buying stocks to place in storage or to smooth supply and demand to stimulate production, as has been seen in the Western United States. Some larger projects have placed contracts in advance, allowing private producers to plan collections and arable multiplication of stocks. Examples of this approach are the UK Channel Tunnel project or the upgrade of the main road linking the northern highlands with the central lowlands in Scotland, but in most cases fulfilling demand has been left to the market. This has created some problems in matching supply and demand, since the production process is long, while many users continue to expect to be able to buy the seeds they need directly from available stock in a store. However, information regarding the required large stock holdings do not yet exist, and this situation is exacerbated by the nature of the producers, who are small organizations with little access to capital, lacking the available resources to speculatively produce specific stocks.

Regulation

There is relatively little regulation of native seed production and marketing in Europe. All countries have their own legislation which protects natural areas or individual species, or which generally limits seed collection, often requiring licensing or permission for particular sites. The degree of restriction and administration of regulation varies widely and, in some cases, can make it difficult even to collect small amounts of common species. The first EU regulation applicable to natives emerged from the agricultural seed sector and covered a list of 24 species and 4 genera (grasses and Fabaceae), used for fodder production and, more recently, for grassland restoration (Abbandonato et al., 2018). The marketing of such species is governed by an old european directive from 1966 (66/401/EEC) implemented to protect seed users (farmers) and guarantee a high-quality supply of seeds of the correct cultivar (Abbandonato et al., 2018). The requirements of the legislation for cultivar registration includes tests for Distinctness, Uniformity and Stability (DUS) that are not applicable to genetically diverse and locally representative lots of native seeds. According to such legislation, native seeds of those fodder species could not be legally sold.

However, over time, local, national, and European legislation for the conservation and restoration of ecosystems, including the keystone EU directive for the conservation of natural habitats (92/43/EEC), started recommending or mandating the use of appropriate vegetal material of local origin, thus creating a contradiction within the regulatory framework. The first attempt to solve such inconsistency was an EU directive (2010/60/EU) that provided certain derogation to the old directive and partially legalized the commercialisation of native seeds. Unfortunately, such regulation was promoted by plant breeders, designed with a traditional agricultural mindset, with limited input from the Directorate-General environment (in charge of the natural habitat protection directive) and little to no consultation with native seed producers. Such regulation introduced an arbitrary sale restriction, limiting the sale of native seeds at 5% of the total amount of seeds sold each year. It also mandated quality requirements that are not applicable to natives, such as limitations in the number of seeds of *Rumex* species which are allowed. We note that some species of that genus are weeds in arable agriculture but in grasslands, they are important components of the plant community (Tischew et al., 2011). Following the release of the EU directive in 2010, each member state (even the ones without a native seed market) had to formally adopt the directive by transposing it into a national law. However, due to its limitations, in most European countries where such legislation was adopted, it was either deemed not applicable to native seeds for restoration or not enforced. Within the UK, England and Wales this legislation is largely unenforced and appears to be ineffective, while Scotland took the approach that while native seeds were not being used for crop production, they did not fall into the existing legislation. However, in countries with a well-developed native seed market, such as Germany, this legislation is currently enforced and will create serious problems as the effort for restoring degraded ecosystems increases. For example, the

German Federal Act for the Protection of Nature, mandates the exclusive use of native seeds in the natural environment from 2020 (Mainz & Wieden, 2019). Native seed producers are scaling up production and soon will reach the legal limit of 5% for those 24 species and 4 genera. However, questions still arise; What will happen then? Will restoration stop? Will it be performed with selected varieties?

Self-regulation from producers and users as an alternative to State regulation has emerged in some countries. The Flora Locale initiative in the UK provided a Code of Conduct relating to seed origin for some 20 years, but ran out of funding and no longer exists despite being widely used. The Code of Conduct created by ENSPA provides a framework that can be adopted in any country and applies to origin, labelling and seed quality (https://native-seed.eu/index.php/about-us/code-of-conduct).

Native Seed Quality Issues

Lack of an appropriate regulatory approach to native seed trading may have led to problems in the quality of native seeds currently sold in Europe. Ryan et al. (2008) and Marin et al. (2018) aquired seeds from commercial native seed suppliers across the continent and tested their quality (purity and viability), revealing a considerable number of substandard seed lots when compared with other lots of the same species. The use of ISTA (International Seed Testing Association) methods and seed testing staff trained and experienced in the ISTA approach has sometimes caused difficulties in finding standardized methods as many laboratories are accustomed to testing cultivars that have less variation in dormancy and germination behaviour than native material. European countries have followed a 'minimum standards' approach to regulating seed quality which does not suit native seeds which have more variability than agricultural or horticultural species. A pure live seed approach, where seeds of varying quality can be traded provided the quality is known, allows an appropriate price and sowing rate to be worked out and is therefore more suitable. This would require Europe to adopt a policy similar to that used for tree seed or more like the one used in the USA. There is a danger for native seed production that the EU will seek to extend minimum standards to a much wider range of species than at present, causing problems for native seed production. Where there is no specific legislation, the general trading requirements of 'suitability for purpose' applies at a basic level, where a significant proportion of seeds in a seed lot should be alive.

Challenges and Opportunities

Native seed production systems in each European country present different challenges. These are usually dependent on the level of development of their native seed markets, competing products and local or national legislation. In countries with sophisticated native seed production and supply systems, such as Germany, the current major issues are related to upscaling production to match the rising demand. In countries like France, where producers and retailers of selected varieties of native species, are very relevant, the challenge is to differentiate the two products, ensuring that users are aware of the differences and that the "native seed label" is not misused. In the UK, where seed production can be difficult in the humid, cool climate, seed imports from further east in Europe are often cheaper and more readily available but may be of populations adapted to quite different environmental conditions. The challenge may be to significantly increase supply without compromising quality and ensure access to capital to develop production without accepting the limitations of multinational seed companies. In other countries where demand is limited, such as Italy, the introduction and enforcement of legislation requiring the use of native seeds in protected areas, the development of seed transfer zones, and certification scheme would encourage the development of a viable native seed market.

Regardless of country and level of development, there are some common and ubiquitous issues for all European native seed suppliers. The first of these is the fluctuation in demand. Unlike the Western US, where public agencies such as the BLM are a major buyer of native seeds annually, thus guaranteeing a minimum demand, Europe does not have similar agencies, and demand is usually fragmented and unpredictable. Moreover, many seed users have a limited understanding of production times for native seeds and expect large quantities of seed in unreasonably short time frames, maybe weeks or months, for production which actually might require years.

Further needs for this emerging industry are to (i) advance the understating of native species biology, ecology and distribution, (ii) limit the impact of collections from natural populations, and (iii) optimise processing and production systems to minimise risks of genetic bottlenecks whilst increasing production efficiency and quality.

Case Study 5: Seed Resourcing in Australia

As in other countries, native biodiversity in Australia remains under threat due to a range of anthropogenic factors, and the most recent Federal Government 'State of the Environment Report' concluded that the overall condition of Australia's natural environment was poor and deteriorating (Jackson et al., 2017). There are many environmental pressures affecting biodiversity, and these include habitat clearing, fragmentation, overgrazing, invasive species and climate change, together with the inevitable interactions among these pressures. Added to these are the increasingly obvious effects of climate change such as those witnessed in the widescale impacts to flora and fauna from uncontrolled bushfires experienced in eastern Australia during the summers of 2019/2020 (Godfree et al., 2021, Levin et al., 2021). Together, these pressure significantly out-weigh current investment in biodiversity conservation (Cresswell & Murphy, 2017; Metcalfe & Bui, 2017).

Given the pressure arising from the scale of habitat loss over recent decades, there has been some increase in government and public focus on the need to actively restore degraded landscapes (Mortlock, 2000; Gann et al., 2019). However, as discussed in earlier Case Studies, this action requires a consistent market for restoration and an effective native seed supply chain. To date, restoration in Australian agricultural landscapes focuses almost exclusively on tree and shrub reinstatement, making them the prime focus of the native seed sector (Hancock et al., 2020). Nevertheless, more recently, there has been a growing recognition of the importance and feasibility of restoring higher levels of species and functional diversity (Gibson-Roy & Delpratt, 2015). The implication of this recognition is that there is an increasing need to ensure the availability of native seeds from a much broader range of species and functional types for restoration.

The sector supplying native seed and restoration services in Australia, is composed of individuals, commercial businesses, non-government organisations, universities, community and Indigenous groups, together with governments at Local, State, and Federal levels (Hancock et al., 2020). It has developed in a largely ad hoc rather than structured and coordinated fashion over many decades, and, as a result, these various groups tend not to work together in the most effective and efficient manner. Government programs provide the main source of funding for restoration (Salt, 2016) and underwrite most seed related activities (Broadhurst et al., 2017). Restoration activities related to the mining are another large native seed user (Mattiske, 2016), as is plantation forestry (Bush et al., 2018). These emerging native seed markets include (i) those providing trees for carbon sequestration schemes (Jackson et al., 2017), (ii) initiatives that install native plants in urban settings (Marshall, 2015), (iii) agricultural production of farm fodder (https://www.stipa.com.au/), (iv) efforts made for salinity control, (v) production of bush foods, essential oils and pharmaceuticals (Environment and Natural Resources Committee, 2000; Gott et al., 2015) and (vi) development offset schemes (e.g., https://www.environment.nsw.gov.au/biodiversity/assessmentmethod.htm).

Major Concerns and Barriers

A national survey of the Australian seed sector provided a broad characterisation of its structure and practices (Gibson-Roy et al., 2021a, b). Worryingly, it highlighted many structural and capacity constraints representing critical barriers to undertaking landscape-scale restoration. Among these concerns were that the seed related workforce is typically small in relation to sustainable business size, and is extremely limited in terms of infrastructure, capacity and training levels. Particularly worrying was that voluntary standards for seed quality and testing were seldom adhered to or enforced, meaning that native seed supplies are seldom tested prior to use. It also highlighted that whilst most practitioners believe that wild seed sources cannot meet future demand, the likelihood that seed production could be used to supplement wild supplies was also found to be equally low due to poor incentivization and support for those approaches. This leaves what little seed production is currently in place, at small-scale and poorly developed levels.

The seed supply chain begins with seed collection, and for protected native species or communities, this requires a government permit in order to protect species from indiscriminate harvest. The survey found this process causes practitioners great concern, with many suggesting the process was ineffective in regulating the collection, consequently creating many problems for practitioners wanting to source seed (Gibson-Roy et al., 2021a). Negative views, which were commonly related to current procedures, perceived them as being overly complicated, with difficulties in interpreting decision-making processes, and with long turnaround times. This combination would lead to missed seasonal collection opportunities, and for this reason, many practitioners choose to collect seed without a licence.

Survey findings also confirmed that the diversity of species available for restoration is low and is mainly restricted to woody strata. Further, the annual quantum of native seed sourced or purchased by respondents was characterised as extremely low, usually less than 100 kg, indicating that the market for native seed and restoration services in Australia is poorly developed. Also, due to supply limitations and poor market development, native seed prices are magnitudes higher than those for comparable non-native species used for pastures, crops and trees, which also severely inhibits their use at large scale (Pedrini et al., 2022).

Key Features of This Work

During the 1990s and 2000s, seedbanks (often run by NGOs, land conservation agencies or community groups) were funded in some parts of Australia to hold and disseminate native seed. These seed banks primarily received and held tree and shrub seed on consignment for collectors, then sold them on to planting projects. However, due to the tree- and shrub-centric nature of the seed base, together with the defunding of seedbanks (M Driver, personal communication 2021) and a rise in interest for grassy community restoration, some potential users began to highlight the need for a wider range of species in collections (Hitchmough et al., 1989). However, obstacles created by limited wild supply, including difficulties associated with sourcing and collecting seed under what are often difficult and arduous conditions, has prompted some groups to explore seed production approaches in more detail, primarily to create more effective ways to produce seed in the quantities, diversity, and quality required for restoration. A prime example of this was the Grassy Groundcover Research Project (GGRP) that conducted efforts over a decade and a half, which eventually led to the development of a set of production approaches for ground layer crops that are now used more broadly by practitioners in Australia.

The GGRP, which was initiated in the early 2000s, aimed to test field-scale methods for restoring grassy communities. The rarity of such efforts meant that seed supply was a major barrier. Also, because at that time, restoring diverse communities was viewed as unfeasible, there was effectively no commercial market for seed of those species, and there were limited options to harvest from remnant areas.

Hence, the GGRP project was compelled to test and develop seed production methods to generate the seed required for testing restoration methods.

Early seed production areas (SPAs) were linked to geographical regions where restoration sites were to be located. This meant several SPAs were established, each using similar approaches. To improve the likelihood of capturing a wide range of genetic traits representative of the source populations, seed was collected from large numbers of non-related individuals from single/multiple populations on multiple occasions over each ripening season.

Initially, crops were grown in small polystyrene boxes of the type used to transport vegetables. These were light, insulative, cheap and easy to source, and so were an ideal option to test if a broad range of species could be cultivated, maintained and eventually used to produce seed (Fig. 12.8). Plants were grown in commercial-grade potting mixes and irrigated. New genetic material was brought into production after, or sometimes before, two harvest seasons were completed. Boxes were arranged to create block planting areas for crops and to facilitate ease of harvest.

For most species, the polystyrene box systems proved effective in supplying much larger amounts of seed than would have been available from wild populations. Over time, larger and longer-lasting materials were used to build container systems, but as with the foam boxes, these tended to suit small crop footprints. Thus, in order

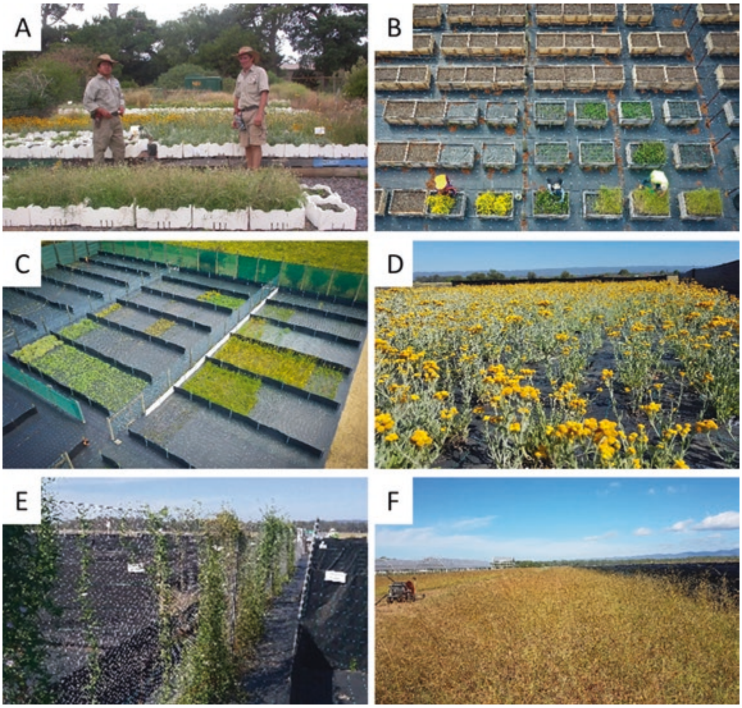

Fig. 12.8 Seed Production Images showing different growing systems (**a**) polystyrene boxes; (**b**) wooden boxes; (**c**, **d**) weed mat; (**e**) trellis, and (**f**) field. Source: Paul Gibson-Roy

to create larger crop beds, the GGRP, together with other producers, began to test growing crops by seeding and/or planting directly into woven weed mat-covered ground at various densities (Fig. 12.8). This approach proved very successful for a number of reasons. Much larger crop beds could be established, and as a consequence, yields increased. These beds were typically divided into separate species, and plants were irrigated by overhead spray or using surface or subsurface drip irrigation. Growing plants directly into soil and with weed competition greatly restricted, resulted in strong plant growth and improved seed production. Beds were set up to facilitate seed harvest by removing seed from the plants by hand or with brush harvesters, or by using brushes and vacuums to harvest dispersed seed from the covered ground surface.

Weed mat growing systems are now the main approach used in Australia. Growers establish crops either in flatbeds, raised beds, or vertically on a trellis. SPA footprints using these approaches range from hundreds of square metres to several hectares in size. These systems have proved extremely effective for growing and maintaining a wide range of ground-layer species and functional groups, including grasses, wildflowers, shrubs, perennials and annuals. Interestingly, some growers have also tested growing species, almost always grasses, as field crops, in a similar manner to wheat or oat crops. And while this can achieve much larger scale beds at a reduced cost, the difficulty of developing effective and economical exotic weed control methods, particularly to treat weeds from the same plant group as the native crop, has meant field SPAs have often been abandoned, with growers reverting to smaller scale but higher yield weed mat systems.

Major Outcomes

An 'Australian model' of seed production, centred on container and weed mat systems, has developed over the past two decades. Run as typically small-scale enterprises and often developed by private operators, land management agencies or NGOs, these have created the seed resources necessary to properly test the restoration of native grassland and grassy woodlands. These areas are both among Australia's most threatened plant communities. A key outcome of the endeavours of these seed producers has been the development of techniques and approaches to growing native seed from a wide range of species and functional groups at the quantities and quality required for field-scale restoration, particularly for direct seeding application. With this key resource available, it has been shown that high diversity restoration is indeed feasible, with numerous examples of resilient and trophically diverse restored communities scattered across several states.

What Worked

As discussed, seed is a fundamental requirement for ecological restoration, but in Australia, markets for restoration are poorly developed, and seed supply chains are limited in effectiveness (Gibson-Roy, 2022). Further, continuing landscape degradation, including those from the effects of climate change impacts, means wild seed supply is unable to meet any future goals for landscape-scale restoration without supplementation from anthropomorphic seed production.

The outcomes achieved by those who have developed these production approaches have paved the way for a much larger and effective seed production capacity. However, seed production enterprises come at an economical cost that cannot be underestimated, since their establishment and operations require considerable time, inputs and labour. Whilst small and simple SPAs, of say 100 s of square metres can be relatively inexpensive to develop, they still require labour to maintain. Not surprisingly, complex multi-species and multi-system SPAs, requiring larger infrastructural needs such as processing sheds, nurseries and storage sheds together with higher staffing requirements, can be very expensive to develop and operate, often having cost structures in the vicinity of millions of dollars.

It is critical, therefore, that regulators, with advised input and support from the sector, devise ways to create clear market incentives for investment in seed production, such as through direct funding support or tax relief. However, the need for the widespread development of seed production capacity in Australia will only eventuate if strong, long-term markets for restoration are supported to create the need for an effective and resilient native seed supply chain.

Final Synthesis

These five Case Studies have been provided to give a broad overview of the diversity of scenarios and complexities involved in the establishment and growth of native seed supply networks across the world. The variety of regional arrangements demonstrate how local engagement, technical procedures, technological development and political processes can shape the structure of a seed supply chain, which then directly impacts the outcome of restoration projects. However, it is clear that there is not a one-size-fits-all model, nor a simple solution that can be instituted to fix all of the supply reliability issues. When structuring a native seed supply system from scratch or improving an existing situation, it is important to keep in mind some fundamental questions such as:

- Who is paying for the restoration?
- Is there enough reliable and consistent funding to sustain a viable native seed market?
- Are there supplies of exotic species or selected varieties that can be misrepresented as natives?

- What are the incentives, regulations or policy settings that facilitate high-diversity restoration?
- What are the incentives, regulations or policy settings that facilitate appropriate genetic provenancing?
- Have seed transfer zones been developed? If not, can this be done to create effective and appropriate delineations for seed collection and seed transfer?
- Are seed quality standards available, required or enforced?
- Are seed buyers informed and aware of the complexities and timelines required for the procurement of native seeds?
- Are there people available with the motivation, skills, experience, and access to funding to organise and develop the seed supply? Is there support for the development of specific technologies matching the needs of seed and restoration sectors?

The answers to such questions could provide cues or guidance on what is working well, what is not working, and where knowledge gaps may still exist. Ideally, by involving all supply chain stakeholders in collaborative assessments and reviews of this nature, strategies to move a sector forward can be developed. It is worth remembering that the final outcome of a robust native seed supply chain is to support cost-effective, locally engaged and scalable seed-based native restoration. This means that both suppliers and users would greatly benefit from closer collaboration and co-development of ad hoc solutions for specific projects/ecosystems. Early negotiation and the development of realistic procurement timelines and plans between suppliers and users are fundamental to ensure seed is available when needed. Moreover, the development and evaluation of novel and effective seed cleaning technologies, dormancy alleviation treatments and the application of seed coating technologies could emerge from constructive collaboration between seed suppliers, practitioners, local communities, mechanical engineers and research institutions.

Such collaborations should go beyond the local or regional scales. There is a growing global community of seed producers, restoration practitioners, community leaders and scientists working on native seed-based restoration, often focusing on similar problems. Solutions that work in Brazil might also be applicable in Africa or South Asia, and the SER thematic section International Network for Seed Based Restoration (INSR) is a good example of a platform that can support such multidisciplinary and transcontinental collaborations and knowledge sharing.

As a final comment, we suggest that the following implications may be useful in guiding work in this area:

- Community-led systems enhance native seed supply and trigger local participation in political processes to improve access to restoration co-benefits;
- Native seed production at the farm level increases seed availability and quality at lower costs compared to wild seed collection;
- Participation of multiple stakeholders in the formulation of regulations and incentives to facilitate the supply, trade and use of native seeds can lead to more adequate technical and political instruments to scale up restoration;

- Collaboration between seed suppliers, users, regulators and scientists in decision-making processes can strengthen the social and political organization systems of the native seed supply chain.

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Chapter 13
Restoration Genetics – A Consideration of Lessons and Opportunities

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Contents

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Summary and Key Lessons

Genetics has provided key insights for improving ecological restoration outcomes over several decades. It is now well established that low genetic diversity and high inbreeding can impact seed set and seedling vigour for many plant species limiting seed and species availability for restoration activities. This can also explain poor restoration outcomes if seedlings fail to thrive or if an inbreeding population is established. The world of genetics has changed rapidly in the last 10 years, with new technologies now producing orders of magnitude more data than anything that has been generated previously. In addition, computational advances are allowing us to stitch together disparate datasets such as those collected from soils, climate, and genomics to better understand the observed outcomes for species and ecosystems. This bodes well for the future of restoration as it will allow us to develop more accurate and sensitive predictive models regarding species choice, restoration location, and the ability of species to cope with change over time. It is also important to recognise that it is now possible to screen the genomes of hundreds of plants and species rapidly and cost-effectively. If this was routinely undertaken, it could significantly improve restoration success as well as create a long-term legacy for future generations who will be charged with managing the ecosystems of our planet.

Our discussions in this chapter highlight just a few of the genetic issues faced by restoration practitioners and agencies when selecting and planting germplasm:

- The translocation Case Study which shows how genetics can guide the identification and use of germplasm to maximise long-term population persistence. While this is an extremely valuable technique for rare and threatened species, we suggest that the same principles can be applied to commonly used restoration species to improve long-term restoration outcomes. Clearly, it is to our detriment that genetic analysis is not a routine part of restoration activities.
- We reflect upon the understanding of climatic and environmental scales over which seed can be moved without serious detrimental effects which have challenged practitioners for many years. While the early 'local is best' approach to seed collection and use has been appropriate in the past, rapidly changing climates are challenging this paradigm. We consequently suggest that a shift to more sophisticated seed collection and management strategies is necessary. Whilst considerably more research in this area is needed, early results indicate that some species do have the capacity to move to alternative landscapes without detrimental outcomes, but for some species, this may require a staged approach.
- We consider areas where traditional land use has dramatically changed conditions and where novel ecosystems have developed. In such circumstances, local genotypes may not be able to survive. It is thought that increasing genetic variation using multi-source germplasm as well as developing ecologically appropriate germplasm are both potential solutions to this issue.
- We note that polyploidy in restoration species is a poorly known condition and that there may be inadvertent mixing of different ploidy levels at restoration sites. Whilst this potentially compromises long-term population persistence, new technologies have improved the rate at which we can screen plants for ploidy differences which can be routinely used for screening restoration species.
- We recognise that although restoration practices have been occurring for decades, early seed collection approaches did not necessarily appreciate the importance of planting a broad genetic base. Consequently, we suggest that these early restoration sites may require the planting of additional genetic variation to reduce the impacts of inbreeding.
- Experience with restoration projects has suggested that long-term monitoring is required to allow us to capture the temporal dynamics of restoration and introduce adaptations as required. Genetics can play a key role in this regard.
- We consider it important that, to meet the increasing future needs of restoration, multidisciplinary teams with secure long-term funding will be required to underpin a 'whole-of-system' approach. Including genetics as a routine component of the restoration toolbox can provide important information to improve restoration outcomes.

- Practitioners routinely need easy access to the latest information and materials to accurately guide their restoration activities. In this respect, we have provided a list of some of the key resources currently available.

We suggest that if restoration activities are to rise to challenges such as changing land use and climate, it is critical that the field remains innovative. While genomics can provide critically important information, we recognise that it is just one part in a complex system. Consequently, we must continually fight for long-term funding of multidisciplinary research to improve and extend essential on-ground understandings and outcomes.

Introduction

Serious concerns about biodiversity loss emerged in the 1980s, and by the 1990s the global imperative to arrest and reverse this loss was recognised by several key international initiatives (Cardinale et al., 2012). Indeed, in 2001, the Millennium Ecosystem Assessment undertook steps to assess the status and trends of the planet's ecosystems and found that humans had changed ecosystems more rapidly in the previous 50 years than any other time in human history (Millennium Ecosystem Assessment, 2005). This assessment found that while net gains in human well-being, societal and economic development had certainly occurred, these had been to the detriment of ecosystems. It was also concluded that it is likely that ecosystem degradation would worsen over time. More recently, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) determined that human-mediated land degradation is negatively impacting on 3.2 billion people, driving a possible sixth mass species extinction, and costing more than 10% of annual global growth product in lost biodiversity and ecosystem services (IPBES, 2018). This report also indicates that investments to avoid land degradation and restore degraded land are economically sound and essential for reaching many Sustainable Development Goals (IPBES, 2018).

Genetic, species, and ecosystem diversity are fundamental drivers of biodiversity, and consequently are central to the long-term conservation and restoration of organisms. Conservation genetics had its genesis in the 1980s when a series of papers outlining the theoretical importance of genetics and evolution for plant viability were published (Oostermeijer et al., 2003). Concerns about native seed collection and use soon arose, particularly in regions where high vegetation fragmentation was anticipated to reduce genetic diversity and elevate inbreeding (Young et al., 1996). As a consequence of this early work, the field of restoration genetics emerged. Since then, researchers have striven to understand the extent of genetic diversity and the nature of population genetic structure in many species, and to understand how this information can be used to improve restoration practices and policy. Notwithstanding this clarity of purpose, there is still a significant challenge for practitioners. The translation of complex scientific findings into field-based actions that improve practice and ultimately the success of outcomes of ecological restoration is a difficult and demanding task. In this respect, restoration genetics is a relatively large field of endeavour, ranging from the exploration of species genomes to measuring quantitative traits of restored populations. Genetics can also be applied to a raft of restoration questions, including (i) what is the most efficient way to source and use plant material? (ii) Have self-sustaining populations been established by previous restoration activities? (iii) Do all plants have the capacity to adapt to climate and location changes? This chapter details eight key lessons that we have learned during our various journeys in conservation and restoration genetics.

Lesson 1: Genetic Tools Are Important for Optimising Plant Translocations

Background

Plant translocations consist of bringing new genetic variants (such as seeds, spores, cuttings, rhizomes, tubers, or plug plants) into nonviable populations which are genetically depauperate and inbred and into extirpated populations, in an effort to create new genetically diverse populations (Weeks et al., 2011). This method is recommended when species are critically endangered and when no other solutions, such as reconnecting populations through biological corridors, soil seed bank expression, and recolonisation from neighbouring populations, have been found to be satisfactory. What is required for restoration is the fostering of enough genetically and demographically viable populations, and this must be carried out in a complementary way to management practices implemented to restore habitat quality and to favour pollinator service and recruitment (Colas et al., 2008; Menges, 2008; Zimmer et al., 2019).

Genetic tools can significantly contribute to optimise translocation success by identifying target populations for rescue and in selecting appropriate populations to use as source material for translocation (Sgrò et al., 2011; Ottewell et al., 2016; Maschinski & Albrecht, 2017; Commander et al., 2018). In addition, genetics can support the evaluation of the effectiveness of plant translocations aimed at restoring or (re)creating demographically viable and genetically resilient populations (Schwartz et al., 2007; Menges, 2008). Indeed, genetic methods, especially when putatively neutral markers (i.e. that have no effect on fitness) and adaptive markers (i.e. that are under natural selection) are combined, allow us to infer many key determinants of translocation success (or failure) which cannot be assessed demographically. In particular, determining mating processes, assessing contemporary gene flow, estimating effective population size and clonal extent, understanding the genetic quality of seed sources (e.g. number of contributing parental genotypes, level of offspring relatedness), assessing inbreeding and outbreeding depression, determining the contribution of sexual reproduction to recruitment, and the degree of admixture (crosses between seed sources) in the post-translocation established generations can be readily assessed using genetics (Zavodna et al., 2015; Bragg et al., 2020; Van Rossum & Hardy, 2022). A genetic approach can be especially important when the remaining potential germplasm sources consist of small census-sized populations or when there may be a suspicion of clonality, small effective population sizes, or disrupted pollination (Maschinski & Albrecht, 2017; Barmentlo et al., 2018; Van Rossum et al., 2021).

What Are the Major Concerns and Benefits for Using Genetic Tools to Optimise Plant Translocation Success?

Assessments of the genetic status, including the levels of genetic diversity and the extent of inbreeding of both source and target populations for plant translocations, have been reported in the literature to have delivered many relevant outcomes. While there is a need to promote a highly diverse genetic pool in the source material used for translocation (Menges, 2008; Sgrò et al., 2011; Commander et al., 2018), attention should be paid to the possibility of co-occurring highly differentiated genetic lineages related to past events or to local adaptation (Gentili et al., 2018; Vera et al., 2020; Bobo-Pinilla et al., 2021). It is thought that these lineages might be reproductively isolated and consequently result in outbreeding depression in any intraspecific hybrid progeny (Edmands, 2007; Martin et al., 2017) or in unsuccessful seed production (e.g. in the case of phenological differences (Patterson et al., 2004; Jones et al., 2011)). Mixing such reproductively isolated genetic lineages together for population reinforcement or for (re)introduction purposes is therefore not recommended, and it is suggested that separate conservation plans should be implemented. Moreover, many species remain as small census- or effective-sized populations, which can consequently be genetically depauperate and inbred or contain only a fraction of the total species genetic diversity (Angeloni et al., 2011; Van Geert et al., 2015; Ottewell et al., 2016). At the same time, supposedly large populations may in fact be highly clonal with a reduced number of genotypes despite their apparently large census sizes (Jones et al., 2005; Van Rossum & Raspé, 2018; Tierney et al., 2020; Van Rossum et al., 2021). In addition, another problem arises when an adult generation and their seeds to be used for founding translocated populations show different genetic patterns through disrupted contemporary gene flow resulting in genetic loss, inbreeding and inbreeding depression in the seed founders, which is not yet expressed in adults (Van Geert et al., 2008; Thomas et al., 2021; Van Rossum & Le Pajolec, 2021). For self-incompatible species, having a high number of compatible mates in the transplants is particularly important for optimising reproductive success (Colas et al., 2008). A low number of compatible genotypes and high genetic relatedness, where transplants mainly consist of full sibs (siblings) or half sibs making them incompatible mates, have been found to restrict seed production despite otherwise good pollinator services (Berjano et al., 2013; Wiberg et al., 2016). In such cases, using several mixed or nonlocal sources appears to be a suitable restoration option, suggesting that relying on self-reinforcement through increasing population size by using plants from the same population should be strongly discouraged (Zavodna et al., 2015; Van Rossum & Raspé, 2018; Tierney et al., 2020).

Research investigations using practical genetic monitoring of translocated populations have highlighted several key factors of translocation success or failure. The most important result is that mixing several, nonlocal or genetically differentiated sources, provided these are not reproductively or ecologically isolated, is preferable to the use of a unique, local seed source for plant translocations. First, it maximises genetic diversity (neutral and adaptive variation) in the transplants and provides a sufficient number of compatible mates for self-incompatible plant species (Reckinger et al., 2010; Ritchie & Krauss, 2012; Fant et al., 2013; Bowles et al., 2015; St. Clair et al., 2020; Monks et al., 2021). Second, it can increase translocation success in the early stages of establishment (Schäfer et al., 2020) and stimulate population resilience to both extreme and changing environmental conditions (Maschinski et al., 2013; Prati et al., 2016). Finally, it can reduce the risk of inbreeding issues and increase plant fitness through heterosis, due to greater fitness of the heterozygotes which arise from outcrossing, in the newly established generations (Willi et al., 2007; Costa e Silva et al., 2014; Zavodna et al., 2015; Barmentlo et al., 2018; Van Rossum & Le Pajolec, 2021). It is considered that the fostering of such benefits might outweigh the risk of maladaptation or outbreeding depression that might arise by using genetically differentiated or nonlocal sources (Bowles et al., 2015; Zavodna et al., 2015; Barmentlo et al., 2018; Ralls et al., 2018).

Plant translocation success depends on the establishment of new generations produced by sexual reproduction, which requires extensive pollen or spore flow to avoid or counterbalance inbreeding issues (Menges, 2008; Ritchie & Krauss, 2012; Fant et al., 2013; Monks et al., 2021; Van Rossum & Hardy, 2022). Contemporary pollen dispersal across a translocated population which encourages admixtures through interbreeding between seed sources can be manipulated and promoted by planting design. For example, establishing large founding population sizes may buffer transplant mortality and optimise population pollinator attractiveness for allogamous plant species. At the same time, a randomised spatial arrangement of the sources may favour outcrossing, whilst site ecological management may promote flowering (Colas et al., 2008; Albrecht & Long, 2019; Silcock et al., 2019; Van Rossum et al., 2020; Van Rossum & Le Pajolec, 2021). It has also been found that site management actions such as mowing, grazing, fire, or soil scraping may need to be employed to provide suitable conditions for germination and recruitment, possibly by preventing founder and genetic drift effects in the newly established generations (Betz et al., 2013; Reynolds et al., 2013).

The main aim of plant translocations has been to prevent short- and mid-term population extirpation and consequent extinction of critically endangered species (Maschinski & Albrecht, 2017; Silcock et al., 2019; Gargiulo et al., 2021). In this regard, a major challenge that restoration will have to face in the coming years is to integrate the need for guaranteeing adaptive resilience of populations in the context of strong changing environmental conditions. This environmental modification may not only be with climate conditions, but also with increasing alteration of wild habitats and of biotic interactions, with these coming from intensive land use, eutrophication, and resource exploitation (Sgrò et al., 2011; Breed et al., 2019; Phillips et al., 2020; Bell, 2021; Dalrymple et al., 2021; Diallo et al., 2021; Pazzaglia et al., 2021). We are confident that newly available genomic and modelling tools (Braidwood et al., 2018; Borrell et al., 2020; Fremout et al., 2020; Seaborn et al., 2021) will certainly contribute to our ability to sharpen plant translocation programmes in these changing contexts.

Case Study 1: Using Genetic Tools to Evaluate Source Population Genetic Status and Translocation Success of Three Critically Endangered Plant Species in Belgium

Many nutrient-poor grassland habitats have undergone severe decline and eutrophication in Western Europe. This decline has persisted for several decades, leading inexorably to the decline of many specialist species. As Belgium is a highly populated and urbanised country, remaining natural areas often consist of small and isolated fragments, which are consequently under significant reproductive stress. The technique of restoring large, continuous areas and creating biological corridors that reconnect these fragments is not always possible, making assisted gene flow an important complementary measure to supplement traditional restoration practices. The European Union LIFE project 'Herbages' (https://www.life-herbages.eu/) was initiated by Natagora, the Wallonia Region and Meise Botanic Garden in southern Belgium. From 2013 to 2020, 629 hectares of wild habitats consisting of heathlands, *Nardus* grasslands, marshes, fens, dry calcareous grasslands, and sandy grasslands were restored from a collection of reforested and degraded areas using traditional ecological management. The management included deforestation and scraping of the topsoil where necessary, which was usually followed by grazing by goats, sheep, or horses or mowing with litter removal. In addition, hay transfer and seed mixture sowing from already restored sites were introduced to provide new biological material. Plant translocations were also implemented in sites which had already been restored or were under restoration, introducing several critically endangered, insect-pollinated herb species which were found to remain only as a few, mostly small, isolated populations in Belgium. These endangered species included the self-incompatible species *Arnica montana* (Asteraceae) and *Campanula glomerata* (Campanulaceae), and the self-compatible species *Dianthus deltoides* (Caryophyllaceae) (Fig. 13.1). Genetic tools based on molecular markers (plastid DNA and nuclear microsatellite markers), together with fitness-related quantitative traits, were used to assess population genetic status and to monitor translocation success for these three species.

Fig. 13.1 Plant translocation of *Campanula glomerata* in 2015 (left) and flowering transplants in 2018 (right) (Photo credit: DJ Parmentier)

For *Arnica montana*, occurring in the *Nardus* grasslands, small wild remnant populations maintained low genetic diversity and were highly clonal. To augment two remnant populations and to create one new population, the two large genetically diverse but highly differentiated wild populations remaining in Belgium were used to translocate 700 transplants into each of the three sites (Van Rossum & Raspé, 2018). For *Campanula glomerata*, only small populations of less than 30 flowering individuals remained in small fragments of calcareous grasslands. These were highly genetically diverse groupings but were differentiated from each other despite the short geographic separation distances. This indicates that effective pollen dispersal within populations occurred, but that there was restricted gene flow between the separated populations, possibly due to barriers to pollen and seed dispersal related to habitat fragmentation (Van Rossum et al., 2022). These small populations were used as mixed seed sources for translocating a population of 500 transplants in each of the four sites where calcareous grasslands were being restored. For *Dianthus deltoides*, six populations were created in sandy grasslands restored from reforested areas or in formerly exploited sand quarries. Each of the 6 sites were populated with 500 transplants. However, genetic analyses revealed that two of the four large census-sized populations used as seed sources were highly clonal, having low genetic variation in the adults together with those offspring used for transplant propagation. This situation led to reduced fitness performance consistent with inbreeding depression (Van Rossum et al., 2021; Van Rossum & Le Pajolec, 2021).

The strategy to use several seed sources in a mixed planting design and to translocate a high number of founders in ecologically managed areas has been successful in founding highly genetically diverse populations. It has facilitated effective contemporary pollen flow, leading to admixed recruits which have resulted from crosses between transplants from different seed sources in the newly established generation (Fig. 13.2; (Van Rossum et al., 2020; Van Rossum & Le Pajolec, 2021)). In this regard, higher plant fitness due to heterosis was found for outcrossed progeny of *Dianthus*. It is thought that the high phenotypic plasticity also observed for plant growth in the first newly produced generation might contribute to adaptive response to new translocation environments in the short term and to resilience to changing environmental conditions over the long term. However, some problems resulting in

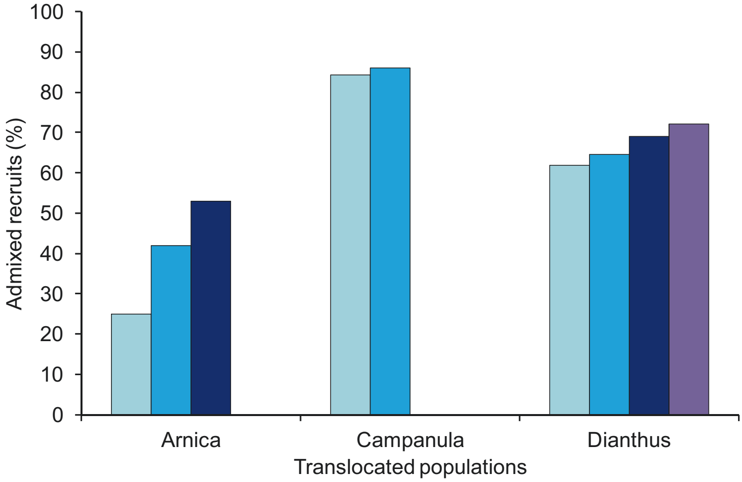

Fig. 13.2 Percentage of admixed recruits resulting from crosses between seed source origins, in the first established generation in 2–4 translocated populations of (i) *Arnica montana* (two seed sources), (ii) *Campanula glomerata* (five seed sources), and (iii) *Dianthus deltoides* (four seed sources). Different colours indicate different translocated populations

high transplant mortality have arisen in some translocation sites, likely associated with insufficient site preparation of the recipient translocation area. In particular, (i) existing vegetation was too recently cleared, this being followed by seed bank germination of many ruderal species that were able to grow quickly and densely, providing high competition; (ii) the retention of trees across the restored grasslands has resulted in shading and heavy cover by leaf litter, leading to adult mortality and low seed germination; (iii) there was excessive trampling by the introduced grazing herd; (iv) habitat conditions were unsuitable, in particular the soil depth was too shallow in formerly exploited sand quarries for *Dianthus* propagation; (v) there was concentrated herbivory on the transplants from molluscs such as snails and slugs and rodents for *Campanula*; and, (vi) Spring drought occurred for several consecutive years.

Lesson 2: We Need to Shift to More Sophisticated Seed Sourcing Approaches

Background – Mixing Seed Sources for Climate Resilience: Limitations and Considerations

Climate is a major agent of selection and is an important factor defining the geographic boundaries of a plant species' distribution. In addition, climate helps to shape the appearance of a plant, from tall forest trees in wet environments to short, robust shrubs in alpine environments. It has been observed that variation in climate across a species distribution can drive populations to have different forms, which are termed 'phenotypes'. Phenotypic variation among populations is often the outcome of evolution selecting different traits that maximise survival and reproduction in a particular climate. Such local adaptation is common among plant species (Hereford, 2009) and underpins current 'local-is-best' conservation and ecological restoration practices.

As climates are seen to be changing across the world, the previous tight associations between phenotype and climate are becoming increasingly decoupled. This mismatch between locally adapted phenotypes and the selective forces that initially drove their evolution is resulting in species maladaptation, which is manifested as a decrease in population fitness. Already, the expression of maladaptation has been evidenced in range-wide dieback from climate stresses such as drought and heat (Allen et al., 2010; Brodribb et al., 2020). In these situations, plants with long generation times and poor dispersal capabilities are predicted to be most at risk to subtle changes in the local home-site climate (Aitken & Bemmels, 2016), and this presents a major challenge to traditional local-is-best practices.

The last decade has seen numerous alternative seed sourcing strategies emerge that aim to build population resilience in conjunction with tolerance to future climates, which have been done by purposely augmenting the genetics of the local population. Depending on the objectives of the seed sourcing strategy, this often involves introducing seed from non-local populations that are within gene-flow distances that may be more genetically diverse known as 'composite provenancing' (Broadhurst et al., 2008) which broadens the gene pool for selective filtering. Seed introducing from non-local populations that possess traits beneficial to withstanding future climates is termed 'climate-adjusted provenancing', and this acts to enhance resilience to future climate changes (Prober et al., 2015). A useful review of the various seed sourcing strategies and how to best implement them are provided by Harrison et al. (2021a).

Understanding the distance over which seed can be moved along climate and environmental gradients before maladaptation manifests is critical to the success of these alternative seed sourcing strategies. One approach informing these distances is the calculation of transfer functions estimated from multi-population common garden trials (Mátyás, 1994; Rehfeldt et al., 1999). These functions model how a performance trait, be it growth, survival, or reproduction, varies in response to the transfer distance, where this transfer distance is the difference between the planting site climate (or environment) and the home site climate (or environment) of each tested population (Fig. 13.3). Transfer functions provide a wealth of information for ecological restoration. These can highlight the optimal phenotype for the planting site, which is represented by the peak of the response curve denoted by Y_v in Fig. 13.3, plus important climate variables related to performance, and the null transfer distance over which performance is not significantly different from the optimal phenotype. This latter function is represented by the blue shaded area under the response curve and is denoted by X_{null} in Fig. 13.3. The null transfer distance is estimated as the intersection between the lower confidence interval of the peak,

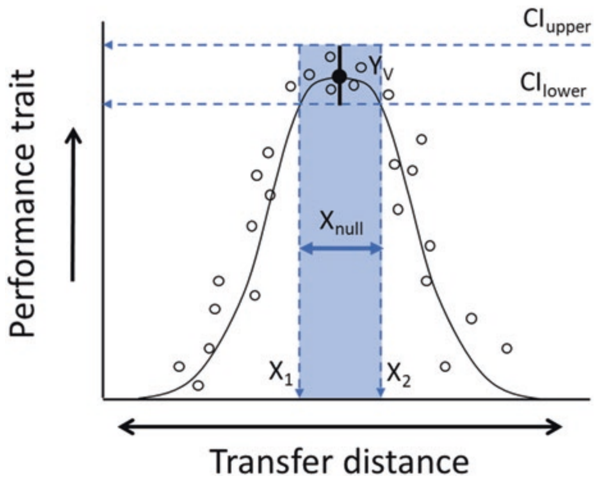

Fig. 13.3 Transfer function (black curve) for a performance trait (y-axis) in relation to the transfer distance (x-axis). Open circles represent each population planted at the common environment site. The optimal phenotype for the site represented by the peak of the response curve (Y_v) is shown by the filled circle with its 95% confidence interval (CI). The intersect of the lower CI with the response curve (X_1 and X_2) represents the null transfer distance (X_{null}) over which there is no significant loss in performance

denoted by CI_{lower} in Fig. 13.3, and the response curve. The transfer distance corresponding to these intersections, denoted by X_1 and X_2 on the x-axis, represents how far seed can be transferred along a climate or environmental gradient without a loss of performance.

Case Study 2: What Have We Learned from Transfer Functions?

Most of our knowledge on the extent of population transfers comes from long-term common garden trials established for economically important forestry tree species. An example of this is the Illingworth Trials discussed by Rehfeldt et al. (1999). However, the last decade has seen a resurgence in common garden trials, particularly to inform transfer distances for ecologically important species, by embedding experiments within restoration plantings (Bailey et al., 2021). These trials have revealed several general lessons on the movement of seed for ecological restoration in the face of climate change.

The most common pattern of species range shifts, as they track their changing climate envelope, has led to the observation that populations can be successfully moved poleward and upslope. However, as many forest trees often have poor dispersal capacities, there have been increasing recommendations to assist the migration of these species across the landscape to help them keep pace with their climate envelope. In this regard, the extent to which populations can be actively moved upslope and towards the poles remains a key research priority. To investigate the limitations of latitudinal and elevation translocations, Bailey et al. (2021) established a common garden field trial comprising 52 range-wide populations (sourced from 15 mainland Australia populations and 37 Tasmanian populations) of the forest tree, *Eucalyptus pauciflora*, at a mid-elevation restoration site on the island state of Tasmania, Australia. They found that the mainland populations of *E. pauciflora*, which have been separated from the Tasmanian gene pool for nearly 15,000 years following the flooding of the Bass Strait at the end of the last glacial maxima, were able to successfully establish following a 7° southward shift in latitude. However, the mainland populations showed significantly poorer survival and growth compared to the Tasmanian populations. Focusing on the 37 Tasmanian populations, Bailey et al. (2021) showed *E. pauciflora* had a broad elevation transfer distance, with populations being able to move, on average, 200–300 m upslope and downslope before a significant decrease in growth and survival relative to the optima.

A second observation is that populations can be successfully moved along climate gradients but there are limitations. Research has shown that the climatic or environmental distance over which populations can be transferred can be broad, and while it varies considerably among species, general patterns are beginning to emerge from studies in North America (Table 13.1). First, the direction of the transfer does

Table 13.1 Summary of transfer distances for forest trees in North America

Country	Species	Performance trait	Mean annual temperature		Mean annual precipitation		Reference
			X1	X2	X1	X2	
Canada	*Pinus contorta* ssp. *latifolia*	Survival	−4.8 °C	3.1 °C	−769 mm	426 mm	Rehfeldt et al. (1999)
		Growth	−1.6 °C	1.6 °C	−249 mm	839 mm	
	Pinus contorta ssp. *contorta*	Survival	−4.0 °C	2.4 °C	−1080 mm	−29.1 mm	
		Growth	−2.9 °C	3.2 °C			
USA	*Picea mariana*	Growth	−4.9 °C	2.7 °C	−350 mm	372 mm	Pedlar et al. (2021)
	Picea glauca	Growth	−4.9 °C	1.9 °C	−391 mm	214 mm	
	Pinus banksiana	Growth	−5.5 °C	2.9 °C	−291 mm	450 mm	
	Pinus strobus	Growth	−7.7 °C	2.5 °C	−1117 mm	130 mm	
	Betula alleghaniensis	Growth	−6.7 °C	2.2 °C			
	Fraxinus americana	Survival	−3.5 °C				Steiner et al. (2021)
	Fraxinus pennsylvanica	Survival	−4.1 °C				

Shown are the intersects of the 95% confidence interval with the response curve (X1 and X2, see Fig. 13.3) for mean annual temperature and mean annual precipitation. Negative values indicate population transfers from cooler/drier environments to warmer/wetter environments and the positive values the opposite not always result in a consistent change in survival and growth performance. For example, the breadth over which dry populations can be moved onto wet sites is much broader than the breadth of moving wet populations onto dry sites. Second, there is a broad range over which cooler climate populations can be transferred onto warmer sites before significant decreases in growth and survival are detected. However, there is a much narrower transfer distance of warm populations onto cooler sites. From a climate mitigation perspective, these results indicate there might be limitations to how far populations, originating from warmer environments, can be transferred onto a currently cooler site that is predicted to become warmer in the future. Indeed, both Grady et al. (2015) and Camarretta et al. (2020) found cold temperatures constrained the distances over which populations could be transferred, suggesting that transfers may need to be staggered through time in line with the site becoming warmer.

A further indication is that population translocations of a species are stable when planted in different plant communities, but transfer distances are context dependent. In this regard, most common garden field trials are established as monocultures, which is not realistic in ecological restoration. To determine if population transfer distances were stable, meaning whether constant population performance is observed when planted in a monoculture compared with a diverse plant community, Camarretta et al. (2020) used a common garden field trial experiment. This work established two focal *Eucalyptus* species, *E. pauciflora* and *E. tenuiramis*, as (i) a monoculture, (ii) a mixed eucalypt planting of the two focal eucalypts, and (iii) a community treatment where each focal eucalypt was planted with another tree species or a shrub species. While mortality of some non-eucalypt species in the community treatment resulted in the effect of the treatment diminishing with time, the population transfers were stable irrespective of which co-planted community was assessed. However, despite a lack of association between a focal species population performance and the diversity of the plant community, Camarretta et al. (2020) found the population transfer functions of the focal eucalypts to be species specific and context dependent. That is, although populations were moved along similar a climate gradient, the transfer distances were not the same for the two eucalypts, which could be partly attributed to the location of the trial site relative to the species distribution. The trial site is within the central core of the widespread *E. pauciflora* distribution but at the upper elevation limit, or 'leading edge' for the regional distribution of *E. tenuiramis*. The poor performance of the lowland *E. tenuiramis* populations translocated upslope to this site was attributed to increased cold and insect damage compared to the local population of this species. This finding is not surprising given populations at the leading edge of a species distribution have limited gene flow and are often locally adapted to the much harsher environments (Hampe & Petit, 2005), thus highlighting the importance of considering population translocations relative to their population structure.

Considerations When Introducing New Genotypes to a Foreign Environment

The decision to move a species and its attendant population needs careful consideration of how introduced genotypes may interact and shape the broader biological communities they are intended to support. This is particularly important when moving foundation species such as trees, which are often ecosystem engineers. Community and ecosystem genetics provides an elegant framework to study how heritable traits of a foundation species extend beyond the individual to influence community composition and processes of an ecosystem (Whitham et al., 2006). Indeed, heritable population differences in tree leaf traits and foliar chemicals have been shown to shape contrasting canopy arthropod and fungal community compositions, highlighting the role of foundation species as drivers that shape the geographic mosaic of biodiversity in forest ecosystems (Barbour et al., 2009; Gosney et al., 2021). The decision to move a plant species or population should not be undertaken in isolation, but must carefully consider the potential of co-evolved plant-plant interactions that may confer fitness benefits in stressful environments. For example, Grady et al. (2017) found *Popular fremontii* grew significantly better when co-planted with a local neighbour from the same original home-site compared to a foreign neighbour species from a different home-site, and indeed survival was markedly greater in the case of planting *P. fremontii* with local rather than foreign populations of *Salix exigua*.

When moving populations, it will be increasingly important to consider protecting the investment from indirect wildcards of climate change, such as invasive mammal species. For example, using a replicated experiment planted within and outside a deer proof fence, Bailey et al. (2021) showed that exotic fallow deer (*Dama dama*) selectively browsed populations of *E. ovata* and *E. pauciflora*, significantly altering the species and population composition in the planting. This highlights the potential impact of pest and pathogens on individuals from non-local populations that may not have co-evolved with the local suite of pests, compromising the strategies to build adaptability and resilience to future climates.

Lesson 3: It Is an Increasingly New World that We Are Restoring

Background

Novel ecosystems are self-perpetuating ecosystems with altered ecosystem structure and/or functioning resulting from intentional or inadvertent human activities. This may involve the introduction of invasive species, altered fire regimes, modified soils, and changes in land use (Hobbs et al., 2006). As these activities lack natural analogues and there are major constraints precluding restoration to their historical state, such novel ecosystems both present a challenge and offer a focus for the research and practice of ecological restoration (Hobbs et al., 2006). Under such circumstances, local native plant materials may not be the ones best adapted for novel ecosystem functioning, while ecologically appropriate plant materials, intentionally developed to tolerate altered prevailing site conditions, may offer more viable alternatives (Jones, 2013).

What Are the Major Concerns and Benefits for Using Plant Breeding to Improve Restoration Outcomes?

Plant breeding methodology has the potential to deliver more efficacious novel plant materials in terms of restoring novel ecosystems by means of (i) increased genetic variation or (ii) enhancement of the ecosystem with desirable functional traits (Jones et al., 2015). Increasing genetic variation, sometimes termed 'assisted gene flow', at restoration sites may stimulate natural selection, thereby enhancing long-term restoration success in novel ecosystems. For example, genetic variation, and thus potential adaptability, may be increased by either using multiple-origin sourced material from a broader geographic range (Larson et al., 2000; Jones, 2003; Rice & Emery, 2003; Breed et al., 2013) or from geographic regions towards which the local climate is changing (Prober et al., 2015). Likewise, using multiple germplasm sources may increase genetic variation at a site relative to a single source (Jordan et al., 2019). This simple, inexpensive, and flexible 'prime-the-pump' strategy can deliver raw genetic material to a restoration site, upon which contemporary natural selection processes can then operate (Broadhurst et al., 2016). It has been argued that such an approach may result in outbreeding depression (Hufford & Mazer, 2003), but it is thought that this potential is likely only for predominately cross-pollinated species (Templeton et al., 1986) and only when hybridisation occurs across taxonomic boundaries or highly distinct environments (Frankham et al., 2011). Moreover, it has been shown that natural selection may correct outbreeding depression when it does occur (Carney et al., 2000; Erickson & Fenster, 2006).

In addition, human-assisted evolution, accomplished through the development of ecologically appropriate plant materials with desirable traits, offers another hope for repairing altered ecosystem structures and functions of novel ecosystems (Jones & Monaco, 2009). Such novel plant materials may reduce the risk of restoration failure, especially in highly altered or stressful environments, which are those in greatest need of restoration (Jones et al., 2015). Hybridisation can be used to enhance genetic diversity (Larson et al., 2003) and encourage natural selection for local adaptation (Jones & Monaco, 2009). Artificial selection may confer tolerance to environmental stresses (Chivers et al., 2016), remediate damaged environments (Jones & Monaco, 2009; Jones et al., 2015), provide ecosystem services (Brummer et al., 2011) and improve adaptation to changing climates (Prober et al., 2015). Such plant materials may be useful when local populations have been extirpated or when site conditions have been dramatically altered in such a way that local populations are no longer sufficiently adapted to the site.

Ecology, physiology, and genetics may illuminate critical traits for breeding of restoration material (Jones et al., 2015). Such traits might include those related to seedling establishment, competitive ability against weeds, persistence, prolificacy, herbicide tolerance, and seed harvest (Chivers et al., 2016). Artificial selection could be directed at any heritable trait expressed at any point in the plant's life cycle. While artificial selection can enhance desired traits, careful attention must be paid to effective population size (N_e) during selection to preclude potential inbreeding and concomitant losses of genetic diversity ([Chivers et al., 2016]; see discussion below). In addition, upon completion of plant material development, testing should be implemented to verify utility for restoration as mentioned in the long-term monitoring lesson, as well as to ensure that sufficient genetic variation has been retained.

Case Study 3: Improving Seed Production in Salina Wildrye – A Key Restoration Species for Altered Habitats That Are Difficult to Source

Salina wildrye (*Leymus salinus* [M.E. Jones] À. Löve) is a native, perennial cool-season grass, that is potentially useful for restoration of damaged rangelands. However, its poor seed production, a result of having few flowering spikes (Jones & Larson, 2018), has precluded its adoption by the native seed industry in the western USA (Jones, 2019). Nevertheless, considerable demand exists for this species for use on the Colorado Plateau, a region drained by the Colorado River in Utah, Colorado, New Mexico, and Arizona. Thus, Salina wildrye plant material, with greater seed-production potential, could make seed of this species available to practitioners for the first time.

To increase seed production, a programme was launched to select for increased spike number in the naturally occurring 9043501 population of Salina wildrye (C0). Two cycles of selection for salinity tolerance (C1, C2) were followed by two cycles for increased spike number in the first seed-production year (C3, C4). Later, a replicated experiment assessed breeding progress for increased spike number and seed yield measured across 2013–2015 (Jones & Larson, 2018). In 2013, the first year of seed production, selection increased ($p < 0.05$) spike number by 4.3 spikes per plant (19.8%) per cycle of selection, but no change ($p > 0.10$) was seen in 2014 or 2015 (Fig. 13.4a). Seed yield also increased ($p < 0.05$) by 0.32 g per plant per cycle (36.8%) in 2013, while again no increase ($p > 0.10$) was seen in 2014 or 2015 (Fig. 13.4b). Thus, selection conducted in the first year of seed production was found to be effective only for that year of seed production (2013), not for the two subsequent years (2014–2015). This suggests that the second and third years of seed

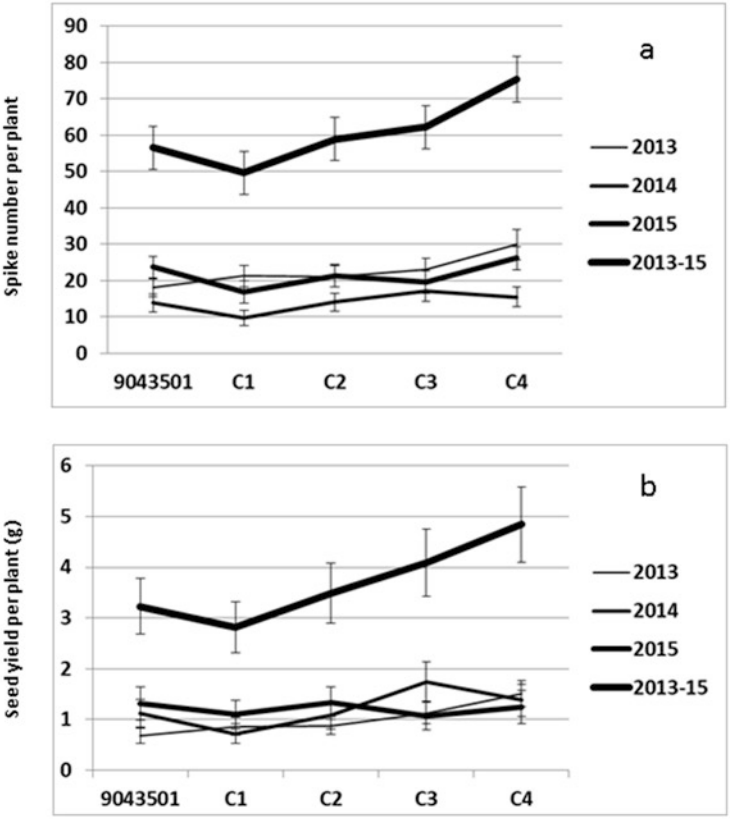

Fig. 13.4 (**a**) Spike number per plant and (**b**) seed yield in grams per plant over 3 years (2013–2015) of 9043501 Salina wildrye (C0) following one cycle of phenotypic recurrent selection for salinity tolerance (C1), following a second cycle for salinity tolerance (C2), following a third cycle for spike number per plant (C3), and following a fourth cycle for spike number per plant (C4). Replicated plots were established at Millville Farm (Millville, Utah) from greenhouse transplants on 16 May 2012. Permission to reprint from Jones and Larson (2018)

production are under separate genetic control from the first year. If so, additional selection in the second and third years would be desirable to enhance seed production in those years.

Using amplified fragment length polymorphism (AFLP) DNA markers, we measured genetic similarity to determine whether it had increased across the four cycles of selection. In theory, whenever a less-than-infinite number of selected individuals is used to generate a cycle of selection, an undesirable increase in genetic similarity can be expected. However, the increase may be negligible if the effective population size (N_e) is large, e.g. > 100, while fewer individuals result in (exponentially) greater losses of variation (Basey et al., 2015). Thus, it is desirable to keep the number of individuals as high as possible in each cycle, though the law of diminishing returns applies as the number increases. In addition, it is important to remember that N_e is

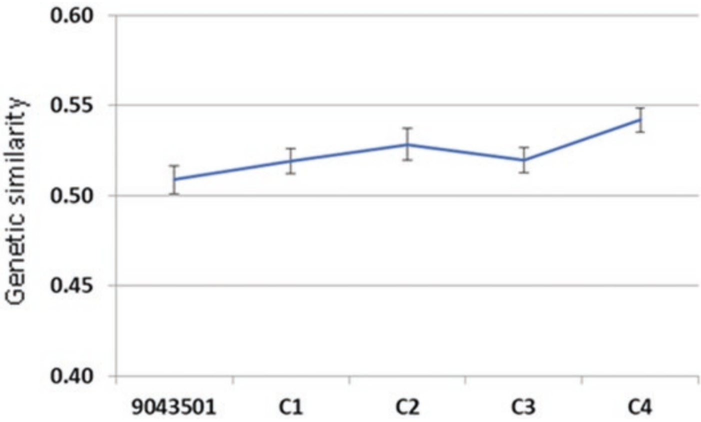

Fig. 13.5 Genetic similarity as determined by AFLP markers among individuals of 9043501 Salina wildrye (C0) and four subsequent cycles of selection (C1-C4). A gain in genetic similarity is equivalent to a loss of genetic variation, i.e. genetic variation = 1-genetic similarity. Permission to reprint from Jones and Larson (2018)

necessarily reduced below the number of intermating individuals when they contribute unequal numbers of gametes to that cycle of selection, which virtually always occurs in practice (Johnson et al., 2002).

Genetic monitoring detected a 6.7% loss[1] in genetic variation, which is an increase in genetic similarity between individuals, from C0 to C4 (Fig. 13.5). Presumably, this was due to genetic drift resulting from a finite number of individuals being used in each cycle. Genetic drift is the resultant loss of genetic variation due to the inevitable intermating of relatives in small populations, due to an insufficient number of parents being used in a selection cycle. When a loss in genetic variation is detected, as found here, genetic variation may be reintroduced to the population by inserting germplasm from individuals outside of the genetically compromised population (Swindell & Bouzat, 2006). This practice is termed 'detect and correct'. Thus, once the loss was detected, seven C0 (unselected) individuals with high spike number were intermated with the C8 selections to correct for the lost genetic variation, although the impact of this intermating on genetic variation has yet to be verified. The C9 cycle, termed Prolific Germplasm, displays prolific spike production and concomitant higher seed-production potential (Fig. 13.6).

[1] Genetic variation = 1-genetic similarity. Thus, according to Fig. 13.4, genetic variation (C0) = 1–0.5088, and genetic variation (C4) = 1–0.5419. The percentage loss of genetic variation over four cycles = [(1–0.5088) - (1–0.5419)]/[(1–0.5088) x 100] = 6.7%.

Fig. 13.6 A seed-increase block of Prolific Germplasm Salina wildrye after two cycles of selection for salinity tolerance and seven cycles of selection for spike number per plant (C9). Block established at Richmond Farm (Richmond, Utah) from greenhouse-grown transplants in early May 2020. Photo taken on 29 June 2021, the first seed-production year.

Lesson 4: Too Little Is Known About Polyploidy and Cytotype Distribution for Almost All Restoration Species

Background

The number of chromosomes in the nucleus of a cell, termed the 'chromosome compliment', is often characteristic of plant taxa (Bennett, 2004). Cell nuclei with a single set of chromosomes are termed monoploid, whilst those with two sets are called diploid. Many organisms, including humans, inherit one set of chromosomes from each parent, and are consequently diploid. Organisms with three (triploid) or more sets of chromosomes are referred to as polyploids (Darlington, 1937). Polyploidy is a heritable state, and well-known polyploid plants include cotton, sugar cane, and coffee. Although somewhat debated, polyploids are often characterised as either *autopolyploid* or *allopolyploid* depending on how these are formed. Plants are termed *autopolyploid* when individuals have more than two sets of chromosomes inherited from the same parental species and are termed *allopolyploid* when individuals have more than two sets of chromosomes which come from different species. There are often strong reproductive barriers between newly derived allopolyploids and their diploid progenitors (Otto & Whitton, 2000; Rieseberg & Willis, 2007), whereas in autopolyploidy barriers between diploid and polyploid may be less severe but are nonetheless important (Ramsey & Schemske, 1998; Soltis et al., 2007).

It is estimated that 35% of plant species are polyploid (Wood et al., 2009). Some 32% of monocots are thought to be polyploid and it is especially prevalent in grasses; 18% of dicots are estimated to be polyploid and this may be more prevalent in perennial herbs (Stebbins, 1971; Otto & Whitton, 2000; Hilu, 2004; Kolář et al., 2017). Polyploid species are found in all floras (Soltis et al., 2015), with the incidence of polyploidy as one moves further away from the equator (Rice et al., 2019). There is also a prevalence for polyploids to occur in environments with climatic and edaphic fluctuations (Parisod et al., 2010). While ploidy level within many species is stable, different ploidy levels, called cytotypes, can exist in some species. A review of more than 60 studies by Kolář et al. (2017) found a high frequency of cytotype diversity within populations, with diploids and tetraploids being the most commonly observed combination. This intraspecific cytotype variation has been shown to follow broad geographic/environmental boundaries in many species. For example, investigation of cytotype diversity in *Solidao altissima* (Asteraceae) across Minnesota (USA) found that hexaploids dominated in forests, while tetraploids were dominant in prairies (Etterson et al., 2016). These authors concluded that this likely represents a level of environmental adaptation. Similarly, diploid cytotypes of *Themeda australis* (Poaceae) from south-eastern Australia are primarily found in cooler and wetter regions, whereas tetraploids are found elsewhere (Hayman, 1960; Godfree et al., 2017). Switchgrass (*Panicum virgatum*, Poaceae) is diploid in the lowland tallgrass prairies of the USA and polyploid in uplands, with an interaction zone between these cytotypes (Casler et al., 2015). A north-south cline with an east-west transition zone was observed between diploid and tetraploid plants of Argentinian *Paspalum intermedium* (Poaceae), with diploids having a narrower range of ecological settings than the tetraploids (Karunarathne et al., 2018). Although limited data are available, there is growing research examining the role and importance of polyploidy on microbial communities as well as pollinator, herbivore, and pathogen interactions (Vamosi et al., 2007; Segraves & Anneberg, 2016; Segraves, 2017).

What Are the Major Concerns and Benefits of Polyploidy Associated with Restoration?

Whilst polyploidy is thought to confer a fitness advantage, disadvantages such as the masking of deleterious mutations can nonetheless occur (Comai, 2005; Otto, 2007). Increases in the size of plant genomes via polyploidy can result in cells becoming larger to accommodate this change. For some species this can be evident through larger leaves, flowers, and fruits and is known as the 'gigas effect' (Stebbins, 1950). However, this effect is not a uniform response across all polyploid plants, with many being difficult to differentiate from their diploid progenitors (Otto, 2007; Soltis et al., 2007; Vamosi et al., 2007). We note that fitness advantages associated with polyploidy have certainly been observed in some key restoration species. For example, Godfree et al. (2017) qualified relationships between ploidy, population fitness, and climatic stress in *Themeda triandra* (Poaceae) and observed that drought- and heat-stressed tetraploids produced four times the quantity of viable seeds than diploids growing under similar conditions. These seeds were also consistently heavier with longer awns, which was interpreted as a fitness benefit for inland tetraploid populations. In bunchgrass, *Pseudoroegneria spicata* (Poaceae), which is commonly used for ecological restoration and to stabilise roadsides in parts of the USA, tetraploid plants grew larger than diploids under common garden conditions, but there were no differences in seed weight, germination rates, or mortality (Gibson et al., 2017). In contrast, Butterfield and Wood (2015) found that ploidy did not influence functional traits in *Bouteloua gracilis*, a dominant C_4 grass frequently used for restoration on the Colorado Plateau (USA). Gargiulo et al. (2019) determined that polyploidy, longevity, and a clonal reproductive system have made *Pulsatilla vulgaris* more resilient to demographic decline.

The prevalence of polyploidy in plants has several important implications for restoration with intraspecific ploidy variation being found in one-third of 115 commonly restored species in the USA (Kramer et al., 2018). Depending on the species being used for restoration, there is the potential to inadvertently mix ploidy levels and impact population viability by producing sterile and/or less fit interploidy offspring (Burton & Husband, 2000; Baack, 2005; Otto, 2007). There is also the possibility of hybridisation if species that have been geographically separated are brought together – the opportunity for this function increases if seed for target species are not available for restoration projects and species substitution occurs. This practice can be exacerbated if taxonomic boundaries between species are poorly defined and unclassified taxa are brought together. The prevalence of polyploidy in grasses (Hilu, 2004) and perennial herbs (Stebbins, 1971) suggests that these plant groups are most at risk of negative consequences of seed mixing (Delaney & Baack, 2012). While there is the potential for negative adverse effects to occur through the mixing of species and ploidy levels, some authors suggest that judicious collection and use of polyploid germplasm may be important to future-proof species as climates change (e.g. Godfree et al., 2017). The complexity of polyploidy and cytotypes variation has led some authors such as Casler et al. (2015) to propose the development of gene pools and zones of deployment for some species.

While polyploidy and cytotype distribution are important impacts on how we collect, produce, and use seed for restoration, the lack of available knowledge on which decisions for restoration species can be based is staggering. For example, an assessment of chromosomal knowledge for rare plant species across the continental US (416 listed species, 236 genera, 73 families) found that less than half of these species had an available chromosome count, and for those that did, the sampling intensity was often too small to allow confidence in the data (Severns & Liston, 2008). Flow cytometry is now an increasingly accessible method to determine ploidy levels (Dirihan et al., 2013) making it possible to rapidly test all restoration species to provide practitioners with information required to make informed decisions. One resource that might be useful in this respect is the Chromosome Counts Database (CCDB), which is a community database (Rice et al., 2015) and is available at: http://ccdb.tau.ac.il/#:~:text=The%20Chromosome%20Counts%20Database%20%28CCDB%2C%20version%201.58%29%20is,that%20will%20be%20updated%20regularly%20by%20the%20community.

Case Study 4: Using Ploidy Knowledge to Improve Restoration Outcomes with a Rare Australian Grassland Species with Cytotype Variation

Temperate native grassy landscapes in Australia have undergone severe loss and degradation since European settlement in Australia in the late 1770s, leaving them as one of Australia's most threatened plant communities (Morgan, 2001). The Grassy Groundcover Restoration Project (GGRP) was initiated to recover these communities in Victoria through a collaboration between the University of Melbourne and Greening Australia, an environmental non-governmental organisation. The project began in 2004 and continued until 2019, leaving a legacy of grassy restoration sites across south-eastern Australia. There were many early barriers to this project, including apathy, disbelief, and disinterest from agencies and conservationists despite the dire state of these grasslands. A key barrier was the lack of seed being available from grassland species to assist restoration actions, which was further exacerbated by the rarity of some species that the project was seeking to restore. Consequently, a highly successful programme to grow seed in seed production areas was initiated, and this resulted in more than 200 grassland species being available for restoration.

The button wrinklewort (*Rutidosis leptorrhynchoides*, Asteraceae, Fig. 13.7) is a grassland daisy endemic to south-eastern Australia that is listed as endangered by Federal, Victorian, and New South Wales (NSW) governments, and as vulnerable in the Australian Capital Territory (ACT). This species occurs in two regions – one in NSW and the ACT, where populations are diploid with 22 chromosomes, and the other in Victoria, where diploid, tetraploid (44 chromosomes), and mixed populations exist (Brown & Young, 2000). Low numbers of triploid (33 chromosomes) and aneuploid (chromosomes ranging 21–46) plants have been observed co-occurring with apparently stable chromosome numbers of $2n = 26$ and $2n = 52$ (Young & Murray, 2000). Knowing of this cytological complexity in advance has resulted in the strategic collection of seed from all plants in a nearby small tetraploid population (within 3 km of the restoration site), as well as from a much larger tetraploid population further away (100 km) to establish this species as part of the larger GGRP seed production (SPA) programme (Gibson Roy, 2010). Planting of SPA seed as well as 150 plants was undertaken in 2009. Surveys in 2010 indicated that 90% of the plants had survived and that widespread seed germination had occurred, establishing a population of >1000 plants.

Fig. 13.7 Image of *Rutidosis leptorrhynchoides* plant (left) and restored grassland (right). (Photo credit: Paul Gibson-Roy)

This project highlights the importance of understanding ploidy levels across a species range to help select material for restoration. Had the cytological information for the button wrinklewort been unavailable, it is possible that diploid and tetraploid populations could have been placed into production, resulting in infertile triploid seed being produced. Since some triploid plants do occur in the wild, it is likely that SPA seed would have germinated, but using this seed would have placed the fate of the population at risk if these plants were infertile and failed to produce successive generations. It is also likely that this would have limited seed production and long-term population persistence.

Lesson 5: Older Restoration Projects Probably Need Additional Plantings to Increase Genetic Diversity and Ensure Long-Term Persistence

Background

As the urgency to conserve and restore biodiversity grew in the 1980s and 1990s, a major challenge for restoration practitioners became how to gain access to the information required to make critical decisions about how and where to collect, store, and use native seed. Concerns about the negative effects of diminishing animal and plant populations began in earnest in the 1980s, and access to increasingly sophisticated molecular techniques has substantially increased our knowledge of rare and threatened species (see Oostermeijer et al. (2003) and refs therein). In the mid-1990s, the genetic consequences of population fragmentation began to emerge (Young et al., 1996) with Mortlock (2001) expressing concerns about the genetic base of revegetation seed in Australia shortly thereafter. A major driver of this concern was that commercial collectors had insufficient time to consider genetic issues and needed to maximise their seed return for minimum effort (Mortlock, 2001). Anecdotally, early seed collections in Australia often favoured seed from one or a few neighbouring plants, especially if these were known to consistently produce abundant seed crops. It is unclear how widespread these early practices were, but it is possible that a significant number of early restoration projects across Australia were based on seed with a low genetic base. Other issues that may be present, but are not yet well known, include the possibility that commercially produced seed mixes may have high levels of inbreeding or may be genetically differentiated. Consequently, it is possible that many of these older plantings have established inbreeding populations that will fail to persist over longer time frames. A meta-analysis of 48 studies comparing genetic diversity in restored and natural populations found that in 46% of the studies there was higher genetic diversity in the restored populations, while in 55% of the studies genetic diversity was lower in restored plantings (see (Jordan et al., 2019) and refs therein). Unfortunately, as there are too few studies comparing restored and natural populations to partition these data either temporally or spatially, it is difficult to determine if restored populations require additional plantings to improve levels of genetic diversity.

What Are the Major Concerns and Benefits of Low Genetic Diversity in Restoration?

Evidence linking genetic diversity and fitness continues to mount (DeWoody et al., 2021), and this understanding is important to restoration practitioners for several reasons including: (i) many restoration plantings occur in novel and/or hostile environments, such as degraded or abandoned farmlands, road verges, and mine sites, where higher genetic diversity increases the chances of restoring genotypes that can cope with the unfamiliar conditions (Gamfeldt & Källström, 2007); (ii) using high genetic diversity can increase survival rates especially early in the restoration process (Schäfer et al., 2020); and (iii) genetic diversity can increase the probability of adapting to change (Weeks et al., 2011). Perhaps the most compelling argument of all, however, is that if we are going to the expense and effort of restoring plant species and communities, why would we not seek to maximise the genetic diversity of founder populations?

Restoring areas with low genetic base germplasm does not necessarily mean that a restored site requires reinforcement with new genetic sources since this may be countered by long-distance pollen flow between remnant and restored populations (Millar et al., 2008; Millar et al., 2012; Broadhurst, 2013), although Aavik et al. (2013) found that gene flow between natural and restored populations was restricted unless populations were within dispersal distance or were in numerically large proportions. However, we once again have a very limited understanding of spatial scale and importance of gene flow between restored and natural populations, and lack information on whether this can counteract any negative consequences of plantings with low genetic diversity.

Case Study 5: Low Genetic Diversity Characterises Older Yellow Box Restoration Plantings

Yellow box (*Eucalyptus melliodora*, Myrtaceae) is highly valued restoration species across south-eastern Australia that has been severely impacted by land clearing and land use change. It has important biodiversity benefits providing food and shelter for many vertebrate and invertebrate species and also benefits agriculture by providing shelter and shade for livestock, as well as tolerating a broad range of soils, waterlogging, and salinity (see (Broadhurst, 2013) for references). Many mature yellow box trees in agricultural landscapes now exist as scattered paddock (field) trees which, like scattered trees across the planet, are declining primarily due to age and stress. Consequently, yellow box restoration in this region (Fig. 13.8) has been occurring for more than 20 years with many of these populations now being reproductively mature. It is possible that seed collection practices for some of these early restoration plantings were not optimal, and seed with a low genetic base was used. Since using a low genetic base can impact the long-term persistence of populations, a genetic assessment of five restored sites planted between 1989 and 1995 using three generations of plants was conducted. Mature scattered trees close to the restoration sites were the oldest generation sampled and provided an indication of the genetic diversity that existed in the site prior to the restoration plantings. Seed from restored trees represented the youngest generation and were assessed separately since these are the future of the restoration plantings – any genetic issues in this cohort could impact on the long-term persistence of the site. These two cohorts were then compared to genetic diversity in the restored trees. While there were some

Fig. 13.8 Yellow box planting site in New South Wales Australia. Photo – Linda Broadhurst

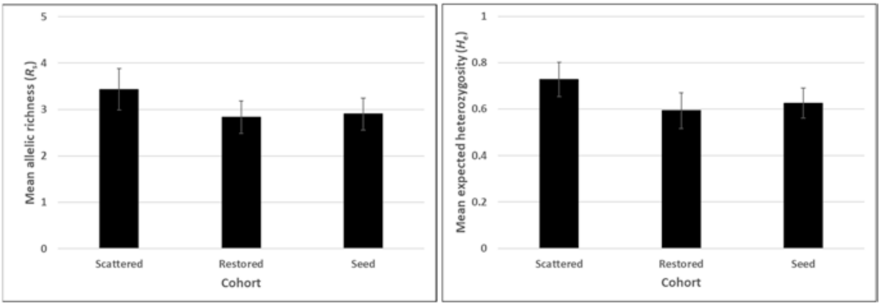

Fig. 13.9 Levels of allelic richness (left) and expected heterozygosity (right) in three yellow box cohorts showing that overall scattered trees had higher levels of diversity than the restored trees or seed from restored trees. See Broadhurst (2013) for more detail

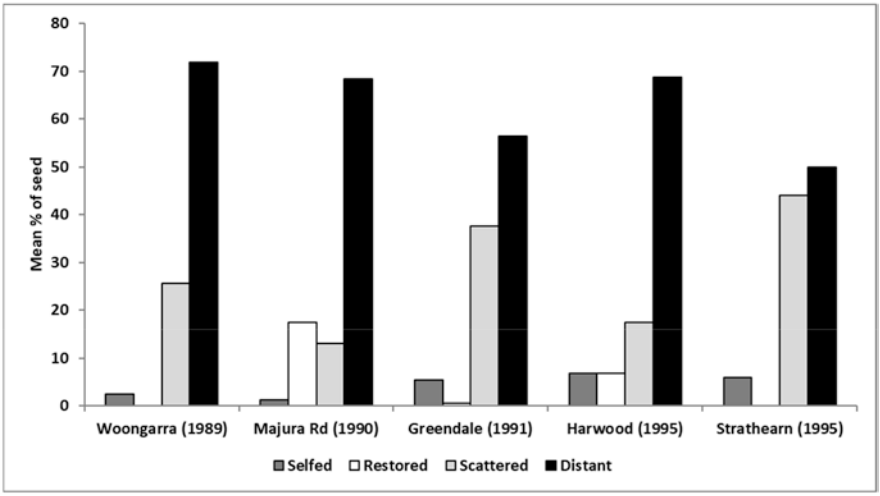

Fig. 13.10 Mean percentage of seed collected from restored trees that were self-pollinated (Selfed), pollinated by trees from within the restoration site (Restored), pollinated by nearby scattered trees (Scattered), or pollinated by trees more than 1 km away (Distant). See Broadhurst (2013) for more detail

differences among the different cohorts of plants overall, the scattered trees had significantly more genetic diversity than the restored trees and their seed (Fig. 13.9).

The sampling design of this study also made it possible to determine the location from which pollen travelled into the restored sites (Fig. 13.10). Four possible sources of pollen were assessed: (i) selfing, where trees were fertilising themselves, (ii) restored trees, (iii) scattered trees, and (iv) distant sites. As with many other eucalypt studies, very little self-fertilisation was detected. While scattered trees did contribute 20–40% of pollen fertilising the restored trees at most sites, the largest proportion of seed (>50%) were fertilised by pollen from trees at distant sites, which were defined as being more than 1 km away.

Key findings from this study are that: (i) early restoration plantings may not have sufficient genetic diversity to replace that being lost as scattered trees decline and die, (ii) these historic plantings may need genetic reinforcement to ensure long-term population persistence, and (iii) landscape context is important – the loss of distant patches of remnant vegetation can have unintended and unseen consequences for pollen movement into restored sites.

Lesson 6: The Importance of Long-Term Monitoring

Background

Restoration aims to implement long-term, self-sustaining plant populations, and to maximise the success of this endeavour, plant materials are required that are best adapted to the restoration site, both now and into the future (Jones, 2013). Surrogate measures such as (i) seed transfer zones and (ii) matching environmental parameters at the collection and restoration sites, are both common ways to match plant materials to a restoration site. However, a direct approach, involving performance testing using fitness-related quantitative genetic traits, may be a more pertinent way to accomplish this goal. A complicating factor in this regard is that under changing environmental conditions, a plant material best adapted to past conditions may not be the most fit in the future (Shackelford et al., 2021). Consequently, selection of plant material needs to consider performance under both short- and long-term site conditions. It is important to remember that restoration genetics is not a discipline based just on molecular markers and the data derived, but rather it employs a range of approaches including quantitative genetics to improve restoration outcomes.

What Are the Major Concerns and Benefits of Long-Term Monitoring of Plant Materials?

Long-term monitoring is necessary to capture the temporal dynamics of restoration. It provides essential information about performance over time, including the influence of species interactions and changing environments. When seeded with long-lived species, short-lived species need to be able to compete and reproduce in the presence of longer-lived material, both in current and in future climates. Networks of trials planted across an array of sites with varying combinations of key environmental variables, such as soil classifications, disturbance histories, and climatic patterns (e.g. Bailey et al., 2021), can provide invaluable data to answer the practical questions of what to plant and where to plant it for long-term success. Data should be collected over a biologically relevant time frame, measuring both short-term establishment and long-term persistence, to yield a robust meta-data set that can inform choice of species and plant materials for the most effective seed mixes.

Long-term monitoring, good-quality meta-data, and long-term funding for ongoing assessments are all essential for understanding performance of plant material over restoration-relevant timeframes. Plant materials of the same species are not the same; they represent different genotypic arrays and respond differently to a site's environmental parameters (Jones et al., 2021). Plant materials vary in fitness across a variety of environments, and they therefore vary in fitness over time with environmental change. Species, as well as populations within species, can differ in persistence, leading to changes in the composition of the restoration plant community over time. It has been noted that some populations tend to do better across a variety of sites than do others, and these 'generally adapted' populations are particularly useful for restoration purposes (Jones, 2013).

Case Study 6: Performance Testing Plant Materials to Improve Restoration Outcomes

Every fall, the USDA-ARS Forage & Range Research Laboratory (Logan, UT, USA) establishes replicated dormant-seeded trials, which are intended for spring germination, to compare released and experimental plant materials of a variety of species. Plots consist of six monoculture rows replicated eight times. Over time, these trials are established at a variety of sites maintained over several years with frequency data collected annually. Assessments made at Year 1 ('establishment') and after some time ('persistence') may contribute to the detailing of relative performance of species and plant materials (within species) at various sites which represent different environments. These data are also used to justify or deny proposals to publicly release experimental plant materials for restoration use.

Asay et al. (2001) demonstrated variation in performance between native and introduced species between environments that received higher and lower precipitation. Yakima, Washington, is normally a very dry site (average rainfall 211 mm/year). However, when precipitation was 67% and 77% higher than normal in the first and second year of a plantation, native plants, especially 'Secar' Snake River wheatgrass, did well relative to introduced species. On the other hand, at Curlew National Grassland, Idaho (average rainfall 307 mm/year), when precipitation was 38% and 20% lower than normal for the first 2 years, respectively, introduced species performed conspicuously better than natives. Precipitation for the first 2 years of this investigation was 40% lower at Curlew than at Yakima, and it was consequently observed that native species differed from introduced species in their response to biomass harvest, with natives showing conspicuous mortality while the introduced species did not.

Rigby et al. (2018) highlighted the importance of long-term data to reveal changes in growth performance over time, as well as differences between plant materials within species. While good establishment of eight native grass species was found at four sites, only three species, Snake River wheatgrass (*Elymus wawawaiensis*), thickspike wheatgrass (*E. lanceolatus*), and western wheatgrass (*Pascopyrum smithii*), displayed reasonable five-year persistence. In a meta-analysis over all studied sites, varying patterns between initial establishment and persistence were identified. Western wheatgrass displayed the poorest establishment, which was likely the result of seed dormancy, yet it evidenced the greatest increase over time, likely due to rhizomatous spreading (Table 13.2). Three long-lived perennials, thickspike, Snake River, and bluebunch (*Pseudoroegneria spicata*) wheatgrass, showed modest increases over time, while three short-lived perennials, Squirreltail (*E. elymoides*), Indian ricegrass (*Achnatherum hymenoides*), and slender wheatgrass (*E. trachycaulus*), along with the long-lived perennial, basin wildrye (*Leymus cinereus*), showed declines (Table 13.2). Rigby et al. (2018) also reported differences among plant materials within species at individual sites.

Robins et al. (2013) performed a meta-analysis of 34 studies and found poor establishment at sites receiving an average of less than 310 mm of annual precipitation. Of the native grasses, thickspike wheatgrass showed the best establishment at low-precipitation sites. As in Rigby et al. (2018), western, bluebunch, thickspike, and Snake River wheatgrasses were able to maintain their establishment-year stand frequency into the third year, while basin wildrye, slender wheatgrass, squirreltail, and Indian ricegrass all declined. Some superior plant materials were 'Discovery' Snake River wheatgrass and 'Whitmar' bluebunch wheatgrass for establishment and White River Germplasm Indian ricegrass for establishment and three-year persistence.

In summary, meta-analyses of fitness-related quantitative traits can be useful for identifying the best species and plant materials for establishment and persistence

Table 13.2 Establishment (year 1) and persistence (increase from year 1 to 5) based on multiple plant materials of eight cool-season native grass species across four sites

Species[a]	Establishment (%)		Species	Persistence (change in plants/m^2)	
SWG	28.5	a	WWG	+4.2	a
BWR	28.2	a	TSWG	+0.8	ab
BBWG	28.0	a	SRWG	+0.5	b
SRWG	27.5	a	BBWG	+0.2	b
SQT	27.2	a	SQT	−1.9	bc
TSWG	25.5	a	IRG	−2.5	bc
IRG	24.4	ab	BWR	−2.7	bc
WWG	16.0	b	SWG	−3.2	c

Means are not significantly different if followed by the same letter based on a least significant difference of $p = 0.05$

[a]*SWG* Slender wheatgrass, *BWR* Basin wildrye, *BBWG* Bluebunch wheatgrass, *SRWG* Snake River wheatgrass, *SQT* Squirreltail, *TSWG* Thickspike wheatgrass, *IRG* Indian ricegrass, *WWG* Western wheatgrass

across sites and for individual sites. Because sets of plant materials are tested across a range of diverse sites, inferences may also be made regarding genotype-by-environment interactions between plant materials and sites. Performance rankings of plant materials can be expected to vary across sites with varying types and degrees of stress, particularly for persistence (Table 13.2).

Lesson 7: Long-Term Funding of Multidisciplinary Teams Is Critical for Developing a Whole of System Approach to Restoration

The many problematic issues currently limiting restoration success and the pressing issue of climate change have meant that restoration must continually search for innovative solutions. Ecological restoration is an applied field, dependent on several botanical disciplines for supporting theory and context. As seen in the Case Studies presented in this chapter, whilst genetics has an important role in restoration, there are, however, still barriers to it being routinely employed in restoration activities. Cost and time are often cited as obstacles to including genetics in restoration projects, especially when restoration funding is rapidly rolled out, leaving little time to collect and process material. Nevertheless, it is increasingly clear that there are immediate benefits to be gained by having genetic information prior to planting. These benefits include guiding germplasm selection, underpinning the development of self-sustaining populations, together with avoiding poor outcomes and project failures. In addition, new genomic technologies are now rapidly producing thorough and comprehensive scans of plant genomes at relatively low cost, opening up opportunities to significantly increase knowledge available for restoration activities.

Consequently, it is time for a more mature approach to restoration, where problems are now routinely attacked by multi-disciplinary teams with connections to practice, agriculture, and industry (e.g., Harrison et al., 2021b). Collaborative actions such as this, under the auspices of a dedicated Institute, research centre, or hub, are the best way to prepare for and meet the future needs of ecological restoration. These agencies would be charged with a public mission to address both the science and practice of ecological restoration, and be committed to a three-pronged mission of research, outreach, and education. Outreach or extension staff should be interspersed among various teams in order to (i) feed research needs and observational information from the field back to the Institute staff and (ii) move research results out to their clientele in the restoration sector. Educational opportunities should also be provided locally, at distant field locations, and online. To foster daily interdisciplinary interaction and, ultimately, research collaboration, the Institute should be located on a university campus in a single building that houses both Institute employees and university faculty. The Institute should provide co-mentorship for university graduate students who can be trained to become leaders for the next generation. Affiliation with a local botanic garden is also desirable since these can play a role in wildland seed collection and plant material evaluation as well as contributing taxonomic and propagation knowledge and providing additional experiential opportunities for graduate students.

Evaluating and monitoring restoration success, together with determination of the reasons for failure, is critical for identifying best practices and increasing the probability of successful implementation of restoration projects (Wortley et al., 2013). A battery of nine key attributes, encompassing vegetation structure, species diversity, and abundance, plus ecological processes, can be utilised to assess success (Ruiz-Jaen & Aide, 2005). Empirically based research on restoration outcomes is expanding, although investigation of socioeconomic aspects still lags behind that of biological aspects of restoration.

The disciplines of ecology, phenology, genetics, evolutionary biology, physiology, and social science all relate to restoration success, and all are being, or should be, incorporated into restoration practice. Among these, the link between community ecology and restoration is probably the most advanced. Community assembly activities have been integrated into restoration to the greatest degree, with lesser emphasis on succession theory and potential future emphasis on functional traits (Wainwright et al., 2018). Currently, phenological data are used primarily to assess biotic resources rather than to improve restoration decision-making, but there is great potential for phenology to better inform restoration strategies (Buisson et al., 2017). We have seen that genetic diversity may impact individual fitness, population persistence, and ecosystem processes (Kettenring et al., 2014; Mijangos et al., 2015). Though genetics is generally underappreciated in terms of its potential contributions to restoration relative to ecology, genetic applications are rapidly becoming more frequent, both for restoration decision-making and evaluating restoration success (Mijangos et al., 2015; Breed et al., 2019). Also, whilst phylogeny, the evolutionary history of organisms, can be used to maintain biodiversity and preserve evolutionary potential, thus supporting ecosystem functioning and stability (Hipp et al., 2015), in reality it is only just beginning to be applied to restoration (Van Rossum et al., 2022). As of 2015, this discipline had yet to be applied routinely as a restoration tool (Hipp et al., 2015). Physiology has also been a mostly neglected potential ally of restoration practice, but field instruments are now available for measuring plant-stress response, thus enabling appropriate choices of species and plant materials for restoration seedings (Cooke & Suski, 2008).

For the most part, restoration is dependent on a consistent supply of diverse seeds. Because large-scale wildland seed collection is limited by the extent of the natural resource, seeds for broadscale restoration will increasingly need to come from ex situ seed production areas, which fall predominately within the private-sector native seed industry (De Vitis et al., 2017; Jones, 2019; Zinnen et al., 2021). This industry's health can be dramatically impacted by public policy (Zinnen et al., 2021). Seed production (Jones, 2019) and seed processing (Pedrini et al., 2019) are highly specialised ventures that would greatly benefit from research. Seeds may be collected directly from the wild, propagated from wild collections, or developed through hybridisation and/or selection (Jones, 2019). These two plant breeding tools can be used to enhance genetic diversity (Larson et al., 2003), encourage natural selection for local adaptation (Jones & Monaco, 2009), confer tolerance to environmental stresses (Chivers et al., 2016), remediate damaged environments (Jones & Monaco, 2009; Jones et al., 2015), provide ecosystem services (Brummer et al., 2011), and improve adaptation to changing climates (Prober et al., 2015). Such plant materials may be useful when local populations have been extirpated or when site conditions have been dramatically altered such that local populations are no longer sufficiently adapted to the site.

Establishing and maintaining an Institute to accomplish restoration goals will require substantial start-up funding, as well as annually recurring expenditures. An endowment could be built through philanthropic donations from those interested in restoring Nature for future generations. Additional financial support could be derived from plant-material royalties and seed sales, as well as consulting fees and training sessions.

Case Study 7: What Should a Collaborative Institute Look Like?

We have said that multidisciplinary and long-term research requires significant investment and commitment. In Fig. 13.11, we outline potential objectives, staffing, and programmes that could underpin a new Institute dedicated to the long-term improvement of restoration outcomes. It should be noted that not all

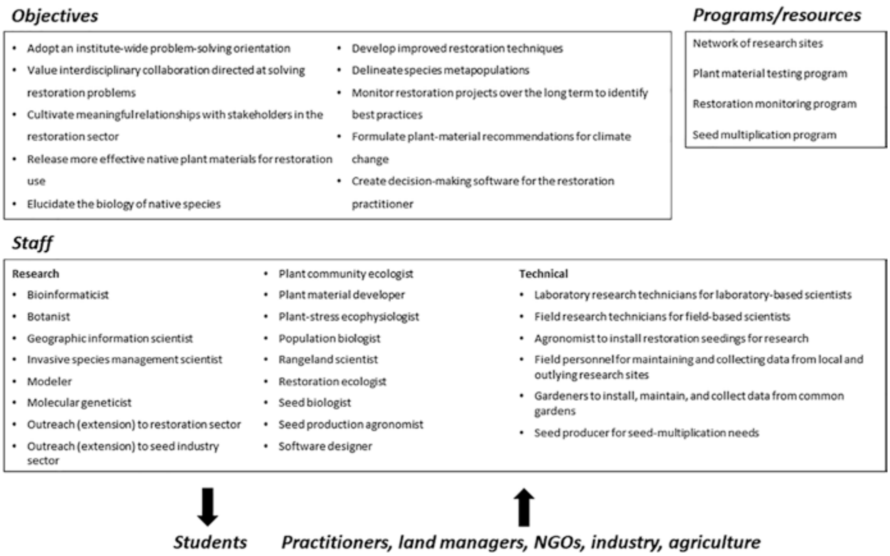

Fig. 13.11 Proposed objectives, programmes, and staffing for an Institute dedicated to improving restoration outcomes

resources need to be new or from the same institute, but rather be formed from a long-term commitment from a range of agencies or organisation of existing staff. One of the greatest impediments to research at this time is the plethora of short-cycle funding cycles that cannot support long-term data gathering. Consequently, we challenge funding agencies to take a risk and make a long-term (10–20 years) investment in restoration and the necessary associated research. We also advocate that this should have dedicated input from practitioners and land managers as well as input from representatives of non-governmental organisations (NGOs), agriculture, and industry, possibly through a strategic board or oversight committee.

Lesson 8: Sourcing Restoration Information and Guidance That Includes Genetics Can Be Challenging

Linda Broadhurst and Francisco Encinas-Viso

Generalisations linking life history traits and genetic diversity have been available since late 1970s (Hamrick et al., 1979) with updates and expansions in subsequent decades (Hamrick & Godt, 1996; Gitzendanner & Soltis, 2000; Broadhurst et al., 2017). There are also an increasing number of tools, decision trees (Byrne et al., 2011; Harrison et al., 2021a), and online resources to help seed collectors to maximise genetic diversity during collection and to decide where this seed might be more confidently used, especially under climate change conditions. In Table 13.3, we highlight some of these resources that may be helpful when trying to maximise the genetic basis of native seed for restoration and for determining where this seed can be most profitably sown or propagated for planting.

Chapter Synthesis

Restoration genetics has developed over the past decade to become an important provider of information to improve the success of our actions. As climates change and the global imperative to restore biodiversity grows, this field can play an even greater role. However, to do this requires a shift in how ecological restoration is funded and undertaken. The next two decades represent a watershed for restoration, where crucial decisions about which species need to be restored and where this should happen must be made. Getting this right is critical to provide the scaffold from which global biodiversity can recover. Case Study 1 demonstrated how genetic information can be instrumental in translocation success as well as providing key learnings for adaptive management and that broadscale restoration would benefit from using a similar approach. Choosing and using seed for restoration is one of the

Table 13.3 List of resources that provide some guidance on genetics in restoration. Resources are ordered according to date where known

Organisation	Date	Title	Link
Society for Ecological Restoration Australia	2021	National standards for the practice of ecological restoration in Australia. Edition 2.2	http://seraustralasia.com/standards/National%20Restoration%20Standards%202nd%20Edition.pdf
Florabank Consortium	2021	Florabank guidelines	http://www.florabank.org.au https://www.anpc.asn.au/florabank/
Australian Network for Plant Conservation	2021	Plant germplasm conservation in Australia	https://www.anpc.asn.au/plant-germplasm/
Reforest Now	2019	Genetics and the endangered. A guide for practitioners	https://61259172-56d2-41c7-881c-aae993555b5e.filesusr.com/ugd/6567fa_8c0f7eacf96544dfa82c7d547f2b2ab5.pdf
Interreg Europe	2019	SUSTREE: Conservation and sustainable utilisation of forest tree diversity in climate change	https://www.interreg-central.eu/Content.Node/SUSTREE.html
Botanic Gardens Conservation International and (BCGI) and International Association of Botanic Gardens (IABG)	2018	Species recovery manual	file:///C:/Users/Linda/AppData/Local/Temp/MicrosoftEdgeDownloads/22611535-11c0-49ca-b29a-545b99f7d8d1/Species_Recovery_Manual.pdf
Australian Network for Plant Conservation	2018	Guidelines for the translocation of threatened plants in Australia. Third edition	https://www.anpc.asn.au/translocation/
Hancock, N., Harris, R., Broadhurst, L. & Hughes, L.	2018	Climate-ready revegetation. A guide for natural resource managers. Version 2	https://www.anpc.asn.au/wp-content/uploads/2019/08/Climate-Reveg-Guide-v2-2018-DOWNLOADABLE-unsecured1.pdf
Harrison, PA	2017	Provenancing Using Climate Analogues (PUCA) Harrison et al. (2017)	https://github.com/peteraharrison/PUCA
EUFORGEN (European Forest Genetic Resource Programme)	2015	Use and transfer of forest reproductive material in Europe in the context of climate change	http://www.euforgen.org/fileadmin/templates/euforgen.org/upload/Publications/Thematic_publications/EUFORGEN_FRM_use_transfer.pdf
Plant conservation Alliance (PCA)	2015	National Seed Strategy for rehabilitation and restoration	https://www.blm.gov/programs/natural-resources/native-plant-communities/national-seed-strategy

(continued)

Table 13.3 (continued)

Organisation	Date	Title	Link
Australian Association of Bush Regenerators (NSW) Inc.	2013	AABR's guiding principles for ecological restoration and rehabilitation (draft). 2013	https://www.aabr.org.au/_upload/learn/WhatIsBR/ER_Statement_AABR_2013.pdf
Bureau of Land Management (BLM), all BLM land, country wide	2008	Integrated Vegetation Management Handbook 1740–2. United States Department of the Interior Bureau of Land Management	https://www.blm.gov/sites/blm.gov/files/uploads/Media_Library_BLM_Policy_Handbook_H-1740-2.pdf
Forestry Commission - UK Government	1999	Using local stock for planting native trees and shrubs	https://www.forestresearch.gov.uk/tools-and-resources/provenance-trials-of-native-tree-species/
Federal Highway Administration (FHWA)			https://tallgrassprairiecenter.org/irvm/transportation-alternatives-seed
The Royal Botanic Garden Sydney		Restore and renew	https://www.restore-and-renew.org.au/
Commonwealth of Australia		Climate change in Australia	https://www.climatechangeinaustralia.gov.au/en/projections-tools/climate-analogues/
Forestry Commission Scotland		Seed sources for planting native trees and shrubs in Scotland	https://forestry.gov.scot/publications/18-seed-sources-for-planting-native-trees-and-shrubs-in-scotland/viewdocument/18
(USDA) Natural Resource Conservation Service (NRCS), country wide		Conservation practice standards are used to determine which species should be used and which seed sources are most appropriate, but that can be highly variable. The standards do not determine seed transfer zones. The relevant standards are: 340, 342, 512, 327, 512, 420, 643 & 550. United States Department of Agriculture. Natural resource conservation service. 2015–2020 (conservation practice standards are updated every 5 years, therefore the majority are 2015–2020)	https://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/technical/cp/ncps/?cid=nrcs143_026849
U.S. Forest Service (USFS), All USFS administration units		FSM 2000 – National Forest Resource Management (Chapter 2070 – Vegetation Ecology) Amendment No. 2000–2008-1. 2008. U.S. Forest Service. Washington, D.C.: U.S. Forest Service National Headquarters	https://www.fs.fed.us/wildflowers/Native_Plant_Materials/documents/FSM_2070.pdf

German Federal Government		Development and practical implementation of minimal requirements for the verification of origin of native seeds of herbaceous plants. Prasse et al. (2010) development and practical implementation of minimal requirements for the verification of origin of native seeds of herbaceous plants (in German). In cooperation with Verband Deutscher Wildsamen- und Wildpflanzenproduzenten. DBU, reference no. 23931.	https://www.dbu.de/OPAC/ab/DBU-Abschlussbericht-AZ-23931.pdf (interview to Anna Bucharova)
Ministry of Agriculture and Food – France		La politique nationale de conservation des ressources génétiques forestières (national policy for the conservation of genetic resources in forestry)	The research has been done through multiple papers on individual species. Some listed here: https://agriculture.gouv.fr/la-politique-nationale-de-conservation-des-ressources-genetiques-forestieres

most challenging decisions facing practitioners, especially as climates change and species may no longer thrive in their home environments. Case Study 2 indicated that there is some adaptive capacity in species to cope with being moved to a new environment, but more research is required to enable confident decisions to be made in this regard. In those places where the restoration site is so different from historical conditions, it may be necessary to select different genotypes that are more likely to survive. Case Study 3 highlighted that even if we want to use these genotypes, seed availability may be a limiting factor and approaches such as plant breeding may be required to ensure that these species are available for restoration. It is well established that low genetic diversity and high inbreeding can impact on seed set and seedling vigour for many species. A lesser known issue is that of polyploidy and cytotype variation, and in this regard. Case Study 4 documented how knowing cytotype variation prior to commencing a restoration project can avoid unexpected restoration outcomes including the failure of populations to persist. Another challenge to population persistence may exist in older restoration plantings if seed collections did not have a broad genetic base. In Case Study 5, scattered yellow box trees had higher genetic diversity than restored trees and their seed, suggesting that additional genetic diversity may be required to maximise long-term persistence. Experience has shown that understanding how species are tracking over biological time is critical to improving and adapting our restoration practices, and Case Study 6 highlighted the potential infield implications if this is not done. Elevation of the importance of restoration by the UN has been a watershed moment for the planet and emphasises that we must come together collaboratively to succeed in halting ecosystem degradation. Case Study 7 proposes the bringing together of multidisciplinary teams and stakeholders with long-term funding (>10 years) to tackle some of our most pressing restoration issues. A major challenge for many practitioners is having access to genetic information on which to base the seed collection decisions, and Lesson 8 provides a list of some of these resources that might be useful.

Acknowledgements Thanks are due to Nola Hancock for her help compiling Table 13.3 which is a subset of information provided in Hancock, N. and Encinas-Viso, F. (2021). *Native seed transfer zones in Australia — How far can seed go? Project Phoenix, Greening Australia. Melbourne* (https://www.greeningaustralia.org.au/wp-content/uploads/2021/07/2.10.pdf).

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Part IV
Socio-Economic Aspects of Restoration

Chapter 14
Ecological Restoration: A Critical Social and Political Practice

Lilian M. Pearce, Ella Furness, Daniel T. Spencer, and Mark Bachmann

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Summary

Ecological restoration is active in framing how the past is understood and how the future is imagined. It is a practice that can be shaped in many ways and by many actors, resulting in complex and often contradictory cultural and ecological outcomes. In this context, ecological restoration is a social practice filled with the meaning-making and the messiness of real life. However, with appropriate good will and determination, these aspects can be celebrated and harnessed.

Community-based conservation projects that aim for both social and environmental outcomes are a cornerstone of community development programs and therefore tend to feature strongly in international development policies (Berkes, 2007; Hunt, 2010). The social outcomes and experiences in projects that are intended to be community-based are crucial to the ongoing success of restoration projects and for the development of the community itself. Critical developments in the recognition and inclusion of social aspects of restoration have occurred in recent years, as reflected in the latest versions of both the *International principles and standards for the practice of ecological restoration* and the *National Standards for the Practice of Ecological Restoration in Australia* (Gann et al., 2019; Standards Reference Group SERA, 2021). However, the social and political dimensions of ecological restoration still require further investigation and attention. As Elias and others articulate, 'urgent attention is needed to the power and politics that shape the values, meanings, and science driving restoration; and to the uneven experiences of these processes' (2021, p. 3). Active work is required at all project stages to ensure that ecological restoration projects integrate into local communities responsibly and equitably.

This chapter focuses on the key role of people in this enterprise. It shares experiences and outlines tools to encourage more inclusive, nurturing, and enduring restoration communities of practice. It describes the ways that certain people have understood, challenged, and participated in ecological restoration. Through three Case Studies, drawn from diverse contexts, it demonstrates the integral role that social and cultural aspects play at all stages in the planning, initiation, and long-term management of restoration projects. It is the contention of this work that the field of ecological restoration has the strength, opportunity, and responsibility to enhance the ethical and political dimensions of its activities. It asserts that both the success of ecological restoration projects and the cultivation of more equitable and thriving communities can be nurtured through careful location-specific practice.

Key Lessons
A review of the history of ecological restoration helps to reiterate the importance of the social and political aspects of our practice and the potential and capacity for restoration work to contribute to positive societal change. In this respect, the ethical implications of ecological restoration will be determined, in a significant way, by understanding, and appropriately responding to local historical, cultural, and ecological contexts.

We suggest that sensitivity to local history and culture is necessary to facilitate inclusive participatory practices that challenge existing inequalities and power imbalances. In this respect, without taking seriously the histories, cultures, and knowledges of First Nations peoples, as well as diversity in local knowledges, restoration runs the risk of continuing the inequitable pattern of erasure that underpins settler colonialism.

Inviting non-professionals and wider community groups into the planning and practice of ecological restoration projects can be vital for achieving local support, ensuring good governance and longevity for schemes, and building connected, caring, and committed communities.

Such hands-on involvement in ecological restoration can play an important role in improving public understanding of ecosystems and of the practice of ecological restoration itself, as well as fostering care and reflection about human relationships with lands and waters.

It is suggested that documenting and sharing project history, together with a description of the focussed 'corporate knowledge' of ecological restoration projects, preserves precious ecological knowledge, expands restoration possibilities, and provides a pathway for long-term success in preserving the World's precious ecological and cultural diversity.

Understanding the Social Aspects of Ecological Restoration

It is important to appreciate that ecologists study organisms *within* their environments, and thus ecology is a relational science that intimately includes people. Social, historical, and political factors necessarily participate and interact in ecological science, practice, and thinking (Mitman, 2006). At the same time, ecology is a very practical science and involves working in the field, and cooperating intellectually with forestry, agriculture, and fisheries. It is also important to appreciate that, in the case of ecological restoration, the work often directly responds to anthropogenic damage, which implies that an essential consideration is the halting of such processes before further exacerbating problem areas. Furthermore, in all settler-colonial nations, as in and much of the rest of the world, ecological restoration is being conducted on often unceded and deeply storied Country, steeped in deep-time cultures. Ecological restoration projects operate within this framework, and must respectfully negotiate layered value systems, governance, and funding mechanisms, requiring considerable thought regarding often-contested narratives of place.

The role of people, values, and ethics in ecological restoration has been debated since the inception of this profession. In its early days, ecological restoration was clearly positioned as a new environmental paradigm that would reintegrate humans into nature and heal environmental damage. Restoration practitioners were caught off-guard by powerful and negative philosophical responses to restoration (Spencer, 2015). The criticisms spurred vocal discourse about the abilities, qualities, and ethics of the practice, where in 1982, the Australian philosopher Robert Elliot charged restorationists with deception by 'faking nature'. Elliot's assertion is that restored ecosystems are technological productions that purport to be, but are far from, equal value (Elliot, 1982). Philosopher Eric Katz added *The Big Lie: Human Restoration of Nature* (1992) and later *Further Adventures in the Case Against Ecological Restoration* (2012). Katz's main thesis is that restoration promotes human domination and control, produces human artefacts not 'nature', and subverts environmental protection. Elliot's and Katz's further concern is for the capacity for restoration to 'green-light' development and promote environmental offsetting by suggesting that damaged environments can be replicated or produced elsewhere (Elliot, 1982; Katz, 1992, 2012). Today, ideas about the fundamental separation of nature and culture have evolved, and the roles and applications of ecological restoration have increased in both scope and scale. Restoration practices are increasingly applied in a variety of 'working landscapes' to repair after past damage. The practice of restoration ecology can be informed by the unique needs of individual places and is encouraged to be defined by clear targets, goals, and objectives, all of which are clearly informed by people and values.

While it has changed over time, the role of history has been central to shaping restoration goals. An early goal of 'historical fidelity', understood as returning to a pre-disturbance, 'native', or 'original' ecosystem, has worked to legitimise ecological restoration within the environmental sciences and defend it from philosophical critiques. Through good intentions, and staying true to local historical-associations, restoration practitioners have been able to deflect charges of 'human control' or of an 'anything goes' approach that has been mounted to counter preservationist thinking. Philosopher Andrew Light contributed his 'pragmatic perspective', differentiating between 'benevolent' and 'malicious' restoration based on the claims and intention behind restoration work (Light, 2000). Light further backed the importance of ecological restoration, arguing for its central role in the cultivation of 'ecological citizenship', which assumes that maintaining moral responsibilities for nature are part of being a good local citizen (Light, 2002, 2005). These key philosophical critiques of restoration discourse remain within the confines of Western frameworks that reinforce nature-culture dualisms and in many cases are centred on settler-relationships with land. Possibilities, rather than ethics, are the focus of this debate. J. Baird Callicott explains that the 'simple and easy understanding of the appropriate norm for ecological restoration is premised on two myths that were prevalent in its early days. These were the wilderness myth and the ecological-equilibrium myth' (Callicott, 2002, p. 418). However, it has been argued that the histories of First Nations peoples are denied and actively erased through the acceptance of the 'wilderness myth'. With the added rebuttal of the ecological-equilibrium notion, both myths are out of date, resulting in ecological restoration narratives being seriously complicated. As considered further below, it is increasingly important to reflect on the origins of ecological restoration and to collectively shape the future of the practice in an inclusive way.

In 2003, Canadian philosopher, anthropologist, and restoration ecologist Eric Higgs published *Nature by Design.* His contributions actively confronted some of the problematic ethical implications of restoration projects, and worked to enhance their capacity to be 'morally good' through these four qualities: (i) restoring for ecological integrity; (ii) being informed by history; (iii) including an element of 'wild design' that provides 'openings for nature and culture, as one being, to go wild', and (iv) practising participatory and community-based, or what he coins 'focal restoration', wherein one can 'rebuild our concern with things that matter' (Higgs, 2003, pp. 226, 285). Restoration, then, is understood to be a socially significant practice that can alter social relationships with human and non-human communities.

How the Use of History in Ecological Restoration is Changing

Debates about how history ought to be employed to inform ecological restoration have existed since its inception and remain at the heart of the discipline. Today, the role of history in the theory and practice of restoration is changing, and its consideration is more relevant than ever. Global environmental change only heightened the challenge of defining a fixed concept of ecological restoration that had existed since the 1990s (Higgs, 2003; Higgs et al., 2018). Today, it is clear that dramatic ecological range shifts are rendering ecosystems that were local to one place in the past, unsustainable in present and future conditions (McCarty, 2001; Walther et al., 2002). Accordingly, ecological restoration practitioners are grappling with how to navigate orientation to the past during a fast-changing present. Amidst accelerated change, ecologists disagree with one another about the spatial and temporal scales and baselines for shaping restoration goals. Many now believe that for future landscapes to be sustainable, ecological, and functional, new species-assemblages might have a place even if historically they have not (Choi, 2004, 2007; Higgs et al., 2014). 'Ecological restoration is rooted in ecological history', assert ecologists Stephen Jackson and Richard Hobbs. Yet, they continue, 'the environment has drifted, and so too have the targets' (Jackson & Hobbs, 2009, p. 567). Seeking historical states demands an increasing body of resources, time, and labour (Hobbs et al., 2009, p. 603). Amidst global change, scientists working practically in this area contest the plausibility of returning to fixed ecological baselines and propose alternative uses of history in ecological restoration (Jackson & Hobbs, 2009; Alagona et al., 2012). Indeed, the Second Edition of the *International Principles and Standards for the Practice of Ecological Restoration* (Gann et al., 2019, pp. 26–31) includes 'Principle 3: Ecological restoration practice is informed by native reference ecosystems, while considering environmental change'. Further, the *Australian National Restoration*

Standards (Edition 2.2) explains that 'Reinstating local indigenous ecosystems in cases where irreversible environmental change has occurred requires anticipation and, if necessary, mimicry of natural adaptive processes' (Standards Reference Group SERA, 2021, p. 5). As Hobbs and others explain, future decisions about investments will be shaped by changing cultural values, environments, and livelihoods. They ask: 'will we be capable of understanding what is best in a rapidly changing world?' and emphasise that '[R]estoration will involve a complicated set of decisions rooted in historical understanding and be open to many potential trajectories' (Hobbs et al., 2009, p. 604).

In 2016, a path-breaking paper in the journal *Ecological Restoration* by Higgs and others presented a 'version 2' of the role of history: this was as a *guide* for restoration, rather than as an exact *template* (Higgs et al., 2014). Higgs and his colleagues make three justifications for this shift: (i) that First Nations peoples' influence on pre-settler ecologies destabilises colonially-informed ideas of 'pristine' nature; (ii) that ecological systems are dynamic, with some reaching irreversible 'novel' states; and (iii) that the dramatic social and ecological change of the Anthropocene has destroyed hopes of complete ecological return. Elsewhere, Hobbs has written on the importance of not creating false expectation of the abilities of ecological restoration that can further collapse hope (Hobbs, 2004). History in their 'version 2′ acts as a tool to interrogate and construct narratives about such issues as ecological cycles, species mobility, ecosystem contingencies, mythologies, and moral dilemmas that arise through restoration practice. In this form, history as a tool expands the interpretation of science, identifies key ecological legacies, and influences the choices available to restoration practitioners.

Increasingly, the concepts of 'thresholds' and 'feedbacks' have been included in restoration ecology theory to explain scenarios where reversal of ecological conditions to previous baseline is out of reach (Suding et al., 2004). The concept of 'novel ecosystems' offered a way to focus on improving the ecological function of parts of the landscape that many ecologists assert cannot be wholly restored (Hobbs et al., 2009). It was developed by restoration ecologists to cater for those ecosystems for which a historic state is seen as beyond being attainable; places which demanded active debates about social values (Hobbs et al., 2009, p. 599). Though not without heavy critique (Murcia et al., 2014; Simberloff et al., 2015), novel ecosystem theory offered a way to integrate those places that may be considered too far degraded or damaged into a caring ethic. Regardless of one's perspectives on novel ecosystem theory, the debates about novel ecosystems pushed ecological restoration practitioners to confront the complications of working within complex hybrid places, shaped by both human agency and climate change. In doing so, it insisted on clear articulation and debate about the possibility for recovery to any 'pre-disturbance' state, and the scope of ecological restoration practice.

From here, social constructions of 'nature' and particularities of the ethics of ecological restoration were also more openly debated. Following philosopher Alan McQuillan, ethicist Gretel Van Wieren outlines how poststructuralist understandings of nature can be adopted in *defending* the ethics of ecological restoration, for if there is no singular 'real' 'knowable' or 'authentic' nature, then it cannot be measured by Western-scientific ontological methodologies perspectives (Van Wieren, 2013a, p. 62). In 2004 restoration ecologists Mark David and Lawrence Slobodkin published a paper in *Restoration Ecology*, arguing from an ecological deconstructionist perspective that the definition of restoration goals and objectives, as well as ideas of ecological health, are characterised by values, not science (Davis & Slobodkin, 2004).[1] Indeed, in 2000 ecologist Jill Lancaster described the concept of ecosystem 'health' as a 'ridiculous notion in a scientific context because there can be no objective definition of 'health' or method for defining degrees of health' (Lancaster, 2000, p. 213). Responses to such views highlighted the willingness of the Society for Ecological Restoration (SER) to accept the importance of social values and roles, thus extending the overtly interdisciplinary nature of the practice.

The Role of Values in Ecological Restoration

It is undeniable that social aspects shape restoration goals, project acceptability, and success. Many factors drive the setting of restoration goals, including a wide range of ideas about nature, social norms, individual experience, and ecological science. Values underpin a conservation ethic and inspire restoration work. The desire to retain diversity and halt species extinction is hardly contentious; ecological science is integral to understanding change as well as tenability of restoration projects (Winterhalder et al., 2004). As ecologist Young D Choi asserts: '[W]e, not nature (although we make a significant reference to it), set the goals and scopes of restoration based on our own judgement' (Choi, 2007, p. 352). Of course, once *in the world*, material realities and non-human agency participate in restoration projects in unpredictable ways, calling on restoration practitioners to constantly innovate and respond accordingly. Through practice, restorationists are unable to deny that 'ecosystems and social systems that depend on them are inextricably linked'.[2]

The historical and cultural origins of one's own value frameworks and ideas of environmental belonging shape an understanding of ecological restoration work. Academic and novelist Raymond Williams describes nature as one of the most complex terms in the English language, experienced via many environmental notions and human relationships over time (Williams, 1980, p. 64). Normative underpinnings of the 'right' kind of landscape have been explored by scholars in a broad range of fields including landscape sociology, environmental psychology,

[1] Note: a response to Davis and Slobodkin rejected many of their premises and conclusions. See (Winterhalder et al., 2004).

[2] Resilience ecologist Carl Folke and others remind us that the lack of recognising this link is the cause of 'many of the serious, recurring problems in natural resource use and environmental management' (Folke et al., 2011, p. 722).

philosophy, place literature, human geography, and history, framed by ontological beliefs about nature and human relationships with it.[3]

The separation of nature from culture is a product of Western intellectual thought that represents just one of many ways to explain and order the material world. Furthermore, it is incompatible with many First Nations peoples' cosmological and ontological frameworks. It is not insignificant that the emergence of ecological restoration was situated in colonised ecologies marked by Judeo-Christian value systems that gave 'man' dominion over nature.[4] Indeed, Van Wieran describes how ecological restoration is carried out as a public spiritual practice by Benedictine nuns, directly cultivating a sense of the sacred in relationships with the land (Van Wieren, 2008, 2013b). In practice, the notion of 'man's dominion' is complicated in ecological restoration, where humans (and indeed particular humans) are presented as the cause of degradation and disturbers of 'climax' scenarios on the one hand, and an integral part of socio-ecological systems necessary for regaining ecological integrity on the other hand. Within these complications lie important implications that play out in restoration work. What is important is that 'the science of ecology does not become biased toward one particular political or economic slant' (Temperton, 2007, p. 346) and that the ethical aspects of the practice are brought to light. As philosopher Gretel Van Wieren explains of restoration:

> [I]nsofar as it is characterized by a healthy measure of critical reflection regarding its assumptions about a wounded and healing creation, it may be able to enter the public sphere with a distinctive vision of land's and people's regeneration (Van Wieren, 2013a, pp. 64–65).

Increasingly, the ecological restoration community is evolving to include intentional consideration of values and social outcomes of projects. The second edition of the SER's *International Principles and Standards for the Practice of Ecological Restoration* includes the introduction of specific social principles, and a *Social Benefits Wheel* to track the social development targets and goals via *community wellbeing, stakeholder engagement, benefits distribution, knowledge enrichment, restoring natural capital, and sustainable communities* (Gann et al., 2019), while Principle 6 in the Australian National Standards (Edition 2.2.) is that 'Social aspects are critical to successful ecological restoration' (2021, p.19). Such inclusions are critical work in the development of the field. We encourage the ongoing attention to the integration of social and ecological aims and outcomes for restoration projects.

[3] For examples see (Cronon, 1995a, b; Head et al., 2005; Reid & Beilin, 2015; van Holstein, 2016); human-landscape interactions have been explored as a sense of belonging (Head and Muir (2007), *Backyard*; Lien and Davison (2010), 'Roots, Rupture and Remembrance: The Tasmanian Lives of Monterey Pine'), a sense of place (Relph, *Place and Placelessness*), for redemptive actions (Jordan, *The Sunflower Forest: Ecological Restoration and the New Communion with Nature*; Van Wieren, 'Ecological Restoration as Public Spiritual Practice'; Van Wieren, 'Restored to Earth: Christianity, Environmental Ethics, and Ecological Restoration'), and to maintain an imagined wilderness in contrast to urban life (Cronon, *Uncommon Ground: Toward Reinventing Nature*; Cronon, 'The Trouble with Wilderness: Or, Getting Back to the Wrong Nature'), to name just a few.

[4] For a full account of theological thought and its influence on restoration thinking see (Jordan & Lubick, 2011).

The genuine and practical integration of social, cultural, and ecological outcomes remains an ongoing challenge; one that is played out in each project according to the unique histories and values relevant in the particular place. In many cases, social aspects remain measured only after the fact, as a way of judging success, while socio-political drivers and outcomes of restoration work remain poorly understood. The more these aspects became central to project design, goal-setting, and long-term measurements, the better.

Pluralising Cultural Perspectives in Ecological Restoration

Opportunities exist to welcome the plural voices within local places and expand the social and ethical benefits of the practice beyond its Western science and colonial origins. As ecological restoration practice flourishes around the globe, complex cultural and ecological scenarios bring both new challenges and new opportunities. Within the restoration community, few publications consider the interconnectedness of ecologies and the presence of plural cultural perspectives and practices. For example, the founder of the Indigenous Peoples' Restoration Network Dennis Martinez promotes 'ecocultural restoration', and ecologists Priscilla Wehi and Janice Lord argue for including 'cultural practices' in ecological restoration (see Higgs, 2003; Kimmerer, 2011, 2013; Wehi & Lord, 2017). Such work makes an important contribution to restoration discourse, but remains marginal to much restoration praxis, and, where incorporated, does so primarily for projects that involve First Nations communities and/or affect livelihoods. For example, Temperton highlights that local communities should be seen as 'stakeholders' in order to reduce the risk of restoration projects failing (Temperton, 2007, p. 346).[5] As our case studies demonstrate, there are many groups that make up a community, and it is important to take the time to get to know the diversity of voices, perspectives, and interests in the places that we work, so as to direct projects in a way that supports multiple stakeholders and encourages project success and longevity while supporting local cultures.

There is also great strength in drawing on many types of knowledge in ecological restoration; something that is being increasingly recognised within the field (see Principle 2 in Gann et al., 2019, p. 6). Social and ecological scientist Yadav Uprety and others have reviewed the inclusion of 'traditional knowledge' in ecological restoration. They concluded that its main contribution thus far is 'in construction of reference ecosystems, particularly when historical information is not available; species selection for restoration plantations; site selection for restoration; knowledge about historical land management practices; management of invasive species; and post-restoration monitoring' (Uprety et al., 2012, p. 225). They argue the further role of traditional knowledge in ecological restoration projects, which they see as

[5] See also (Cairns Jr., 1995).

complementary to science, and as a powerful tool to enhance the 'social acceptability', economic feasibility, and 'ecological viability' of projects (Uprety et al., 2012, p. 225). Other non-Indigenous local knowledges are also central to understanding change over time, reference ecosystems, values, and commitment to projects.

Restoration ecology is a complex social, political, and ecological practice. It was born of colonial societies and Western science, imbued with certain assumptions about nature and culture and knowledge. By expanding restoration practice to being informed by multiple types of knowledge as encouraged in the recent International Standards (Gann et al., 2019), the traditional hierarchies of power and knowledge are softened. Critical and generous attention to these aspects of restoration will enable the practice to continue to move beyond unintentionally perpetuating uneven social dynamics. A committed integration of social and cultural aspects of the practice can support the ongoing learning and healing potential of collaborative, inclusive, and cross-cultural restoration work.

Case Study 1 Preamble

There is great potential to be gained from accounting for First Nations land practices and expanding restoration activities to consider what anthropologist Michelle Cocks calls 'bio-cultural diversity'. Bio-cultural perspectives represent the multiple dimensions of culture that enable cultural resilience in the face of change. For bio-cultural diversity, cultural values extend beyond 'natural areas', to resources and livelihoods generated in peoples' relationships with them (Cocks, 2006, p. 190). In settler-colonial contexts in particular, ecological restoration and other environmental management practices can provide a formal avenue for First Nations groups regaining access to land and sovereignty. Case study 1 reflects on the lessons learned over more than 20 years of a significant First Nations-led restoration project.

Case Study 1: Jocko River Restoration on the Flathead Reservation, Montana, USA

Daniel T. SpencerGermaine White, and Rusty Sydnor

In the beginning, when I saw the land, it was beautiful. This land was good…All our waters, our creeks were flowing along good…It is there in the water—that is where there were many animals—fish and other things. And by that, we are wealthy from the water.

Mitch Smallsalmon, Pend d'Oreille elder, 1977 (Smith, 2010, p. 4).

Montana's Flathead Reservation is home to three Indigenous tribes: the Bitterroot Salish, Upper Pend d'Oreille or Q'lispé, and the Kootenai, who make up the

Fig. 14.1 Logo of the Confederated Salish and Kootenai Tribes (CSKT). (Photo courtesy of Confederated Salish and Kootenai Tribes)

Confederated Salish and Kootenai Tribes (CSKT) (Confederated Salish & Kootenai Tribes, 2021a) (Fig. 14.1).[6] Historically, the territories of the three Tribes covered all of what is now western Montana, extending into parts of Idaho, British Columbia, and Wyoming (Confederated Salish & Kootenai Tribes, 2021a). Under the Hellgate Treaty of 1855, the three Tribes ceded to the United States much of their Indigenous territories, while reserving to themselves the land that now makes up the Flathead Indian Reservation (FIR) (US Senate, 2016). The Tribes also reserved fishing, gathering, and hunting rights on and adjacent to Reservation lands, including the 'exclusive right of taking fish in all streams running through and bordering' the Reservation, and 'the right of taking fish at all usual and accustomed places' (Kappler, 1904, p. 724).

The entire Reservation, and many of the lands reserved for fishing rights, lie within the watershed of the Clark Fork River. Straddling the headwaters of the Clark Fork are the cities of Butte and Anaconda, once home to one of the world's largest and most productive copper mining complexes. One hundred and fifty years of mining and smelting severely contaminated Silver Bow Creek, one of the main headwater tributaries to the Clark Fork, with toxic levels of arsenic and heavy metals, killing all aquatic life in the creek and severely impacting the Clark Fork River (Chavez & Mullen, 2011). A '500-year flood' in 1908 spread mine and smelter tailings 120 miles down the length of the Clark Fork River, eventually impounding behind the recently constructed Milltown Dam, 5 miles upstream from the growing city of Missoula (Bonner Milltown History Center, 2021). Since the early 1980s, the US Environmental Protection Agency (EPA) has designated the entire 120 miles of the

[6] Note: the Salish comprise two different tribes, the Bitterroot Salish and the Pend d'Oreille or Q'lispe. While historically distinct, they share a common language and culture.

Clark Fork River from Butte and Anaconda to Milltown, a Superfund complex[7] (the largest such complex in the United States), and initiated a multi-year process to remediate and restore the Upper Clark Fork watershed.

While the Upper Clark Fork watershed is outside of the exterior boundaries of the FIR, the CSKT retain fishing, gathering, and hunting rights in the watershed from the Hellgate Treaty. When in 1983 the State of Montana filed a lawsuit against the Atlantic Richfield Company (ARCO), the 'responsible party' for the damages to the Clark Fork watershed, the CSKT joined the lawsuit, together 'contending that decades of mining and smelting in the Butte and Anaconda areas had greatly harmed natural resources in the basin and deprived Montanans of their use' (Montana Department of Justice, n.d., p. 1). Eventually, in 1998 the Tribes finalised a 'Consent Decree' with ARCO who agreed to pay $USD18.3 million in damages to the Tribes 'to pay for the restoration, replacement, and/or acquisition of injured natural resources in the Upper Clark Fork River Basin (USFRB), as compensation for natural resource damages basin-wide' (Confederated Salish and Kootenai Tribes, p. 1, 2000; Brooks, 2012). So began more than 20 years of continual and ongoing restoration of the Jocko River.[8]

One of the species devastated by the mining and smelting damages to the Clark Fork River is the bull trout (*Salvelinus confluentus*). A char of the family Salmonidae native to northwestern North America and sacred to the Salish and Pend d'Oreille, the bull trout helped the Salish peoples to sustain their way of life for millennia prior to the arrival of white settlers.

> [O]ne of the keys to the long-term success of the Salish and Pend d'Oreille way of life, as Pend d'Oreille elder Mitch Smallsalmon said, was the water – the clear, cold, abundant waters of the tribes' territories, and the fish that teemed in almost every creek, river, and lake. *Kʷem̓t šey̓ še nk̓ʷúlexʷ qe sqʷyúlexʷ łiʔe l sewłkʷ*, Mr. Smallsalmon told us. 'By that, we were wealthy from the water.' And of all the 'wealth' that swam through those sparkling waters, none was more important to tribal people, to their survival and their wellbeing, than the greatest of all the native fish – *aay*, the bull trout (Smith, 2010, pp. 4–5).

With the settlement award, the Tribes decided to start a holistic restoration project, integrating ecological, cultural, and spiritual dimensions of the Salish and Pend d'Oreille's relationship to bull trout. Restoration of the bull trout is an example of Indigenous restoration of a 'cultural-keystone species,' selected 'because of their vital roles in both material and nonmaterial aspects of a culture' (Kimmerer, 2011, p. 261). As Robin Kimmerer, noted Potawatomi scholar, writer, and ecologist,

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[7] In 1980 the U.S. Congress passed the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), commonly referred to as 'Superfund' for the fund it created to remediate hazardous waste and toxic contamination sites. CERCLA 'provided broad Federal authority to respond directly to releases or threatened releases of hazardous substances that may endanger public health or the environment.' Superfund: CERCLA Overview, https://www.epa.gov/superfund/superfund-cercla-overview

[8] The State of Montana reached a separate settlement with ARCO for $230 million in 1999. 'Summary of 2008 Settlement of Clark Fork River Remediation and Natural Resource Damages Claims and Related Restoration Plans', https://semspub.epa.gov/work/01/554358.pdf. The CSKT has sole authority to administer and spend the $18.3 million settlement awarded to the Tribes.

explains: 'The renewal of these animals and other cultural keystone species inspires, and is inextricably connected to, the revitalization of indigenous communities' (Kimmerer, 2011, p. 261).

The CSKT chose the Jocko River watershed that flows through the southern stretches of the Flathead Reservation to implement the Bull Trout Restoration Project. Spearheaded by the CSKT Natural Resources Department, especially the Fisheries Program, and working in conjunction with several other CSKT departments and private restoration consultants, the CSKT listed four reasons for choosing to restore the Jocko River: (i) its similarities in size, hydrology, and species composition to Silver Bow Creek, the primary site of injury to the fisheries in the UCFRB; (ii) its listing as a 'core area' for the threatened bull trout; (iii) its support of a healthy population of native Westslope Cutthroat trout; and (iv) impending threats to the Jocko watershed from high rates of development (Confederated Salish and Kootenai Tribes, n.d.). An interdisciplinary team developed the Jocko River Master Plan to guide restoration of 22 miles of the lower Jocko River, focusing especially on several rivers reaches that had experienced channel straightening, vegetation removal, and levee construction for flood control in the 1950s, which severely damaged the health of the river and fishery (Daniels et al., n.d.).

While the ecological restoration of the Jocko River has largely succeeded in returning the river to its historic channels and flow as well as restoring the fishery,[9] perhaps the most distinctive and innovative part of the project has been the development of a highly experiential curriculum, *Explore the River: Bull Trout, Tribal People, and the Jocko River* (Confederated Salish & Kootenai Tribes, 2021b). It was developed by Germaine White, a member of the Pend d'Oreille Tribe and Information and Education Specialist for the CSKT Natural Resources Department, together with non-Tribal member David Rockwell, DVD/website design and development and science researcher and writer. *Explore the River* is 'an integrated multimedia curriculum framed by the cultural values of the Salish and Pend d'Oreille People' that seeks to reconnect the children of the Flathead River, both Indigenous and non-Indigenous alike, with its rivers and streams, by integrating 'tribal and scientific knowledge about water, fish and wildlife, and the relationship that people have had with the Jocko River and other streams, both past and present' (Confederated Salish & Kootenai Tribes, 2021b).[10]

Grounded especially in the core Salish value of reciprocity, students learn from both scientists and tribal elders, instilling Salish values of respect, honesty, humility,

[9] Through 2020, the Jocko River restoration has drawn on natural resources damages settlement funds to acquire land and transfer grazing rights in order to protect 4330 total acres in the watershed, 26 miles of streams, 50% of the ecological floodplain, and 17 miles of the lower main-stem river channel through passive and active restoration. Native trout are being restored through construction of fish passage structures, irrigation diversion management, and managing aquatic and terrestrial invasive species. (Sydnor, 2021).

[10] For the role of restoration in also addressing the climate crisis, see, 'Addressing the Climate Crisis: Infusing Tribal Culture into Climate Science Education,' https://tribalcollegejournal.org/addressing-the-climate-crisis-infusing-tribal-culture-into-climate-science-education/

generosity, courage, kindness and compassion, patience, humour, good cheer, warmth, endurance, strength, fortitude, cooperation and helpfulness, selflessness, quiet and calm, thoughtfulness, level-headedness, self-restraint, self-discipline, responsibility, self-respect, observation, listening, and relatedness into a restoration process that integrates the ecological, cultural, and spiritual dimensions of working to make a watershed whole again (Confederated Salish & Kootenai Tribes, 2021b).

As Germaine White writes about this effort,

> In this tribal view, the natural and spiritual worlds are valued equally. Animals and plants are respected because they were here before us and have nurtured us from the beginning of time. We honor them by never taking more than we need, never failing to leave something for others, and never wasting. In short, we care for them, and they take care of us. Similarly, we value, honor, and respect our elders and ancestors, and we love our children. For them we want to ensure the continuation of our languages and cultures, of which water (water that is cold, clean, complex, and connected) is a central part. We can leave no greater gift to future generations (Confederated Salish & Kootenai Tribes, 2021b).

White notes that education was integral to the restoration planning from the beginning. However, 'there was no roadmap – we got to learn as we went along. No one said here's how to do education for a watershed-scale restoration. Because I had worked for the Cultural Committee for so many years, I wanted the restoration to be framed by our cultural values. And that meant beginning by listening to the elders'. Part of the concern is that younger people are losing their connection to their traditional lands, and restoration was seen as one way to begin to remedy that. 'Children now are like buffalo born behind a fence,' she noted. As White listened to the elders talk about their ancestral lands, she was astonished at how many place names referred to bull trout. The abundance of bull trout was central to what White calls their food sovereignty survival strategy: when other foods were unavailable due to drought or severe winters, bull trout provided year-round sustenance. Hence it became clear that restoring the Jocko River meant restoring the sacred and threatened bull trout. 'So, culture really informed this project from the beginning, and education was central,' White noted (White, 2021).

The restoration of the bull trout to the Jocko River watershed is an example of Indigenous-inspired restoration that combines the Traditional Ecological Knowledge (TEK) of the CSKT with the Scientific Ecological Knowledge (SEK) in the western discipline of restoration ecology. White notes that the restoration project began by first listening to the Tribal Elders for their TEK on bull trout, before using the SEK of restoration ecology to develop and implement the Jocko River Master Plan (White, 2021). And TEK has continued to influence the adaptive management practices guiding the project, in particular, learning to listen to the river in making management decisions (Sydnor, 2021).

In reflecting back on the project, White concludes, 'It was a very thoughtful project. We took a year or more to plan the project. We carefully examined what we were going to do and how to do it. We wanted the restoration to be informed by our history, our culture, and our geography' (White, 2021). Rusty Sydnor, a restoration botanist with CSKT who has worked on the Jocko River restoration since 2004, agrees. While the restoration work in the Jocko River utilises the tools of western science, ‘Our work is really driven by the CSKT’s and especially the Tribal elders’ deep reverence for the Bull Trout, the river, and the land,’ he notes. Among the many lessons learned from two decades of restoration work on the Jocko, is learning to listen to the river itself. After the initial mechanical work to reconnect the river to its floodplain, the restoration team planted tens of thousands of seedlings in the riparian zone, only to see a 25-year flood event the next spring damage significant portions of the planting areas. The flood, however, created the necessary conditions for cottonwood and willow regeneration, and the river began to heal itself. Restoration then shifted from an emphasis on revegetation with native plants to re-establishing natural processes and letting the river take the lead. ‘Once you let rivers do what they do naturally, they do miraculous things, and we just get out of the way’ (Sydnor, 2021).

Another shift the project has made is moving to using tribal workers and staff to do the actual restoration work, as they gradually gained experience and knowledge. One result is an increasingly experienced and highly trained tribal team now capable of combining western and Indigenous knowledge and practices in other restoration projects on the Reservation. Sydnor also observes an important shift in perspective in working with Tribal elders: ‘In the work I’ve done with the Tribes, it’s the long-range vision the Tribal elders have that has impressed me. Having occupied this land for millennia, a long-range vision is an important part of their outlook. While I can focus sometimes on the short-term problems with the project, when the elders visit the sites, they are so excited about what they see in the river returning to health.’ Sydnor believes that developing this long-term perspective and vision is critical to partnering with First Nations people – looking back to their history and forward to the future.

What is the future of the Jocko River restoration? Sydnor notes that the Jocko restoration efforts began with funding from the Atlantic Richfield (ARCO) Superfund settlement, and are now supported by three separate funding sources. The majority of the original ARCO funding was expended between 2000 and 2014 for land acquisition and habitat restoration and improvement efforts, primarily for the benefit of bull trout. In 2014 the CSKT withheld a portion of the ARCO funding to create an endowment to support, in perpetuity, annual operations and maintenance costs (O&M) – such as boundary fence maintenance, noxious weed control, and trespass monitoring – on those lands acquired with ARCO funding. Starting in 2006, the CSKT began using funding from Bonneville Power Administration (BPA) to mitigate impacts on native fish due to the construction and operation of Hungry Horse Dam on the South Fork of the Flathead River, a hydroelectric facility near Columbia Falls, Montana. The BPA funding, like ARCO, can be used for land acquisition as well as other habitat restoration efforts to benefit Bull and Westslope Cutthroat trout. To date, the CSKT have used these funds to acquire and restore approximately 2133 acres of riparian and wetland habitat to benefit bull trout, mostly from land previously used in grazing cattle (Geum Environmental Consulting, 2019).[11] In 2021, the CSKT and BPA reached an agreement whereby BPA created an endowment for O&M funding on lands acquired through the BPA Hungry Horse Dam settlement. Lastly, the CSKT have also been able to use funding to mitigate impacts on fish and wildlife Reservation-wide, due to the operation of the Seli'š Ksanka Qlispe', known as the SKQ, Dam on Flathead Lake for land acquisition and habitat improvement projects in the Jocko River watershed. The SKQ Dam funding for the Jocko River is much smaller in scale compared to the ARCO and BPA funding, but nonetheless has been used to complement these larger funding sources (Sydnor, 2021).

Restoration efforts in the Jocko River watershed have been successful, in part, due to the fact that most of the CSKT staff that were involved in the litigation efforts in the 1990s regarding ARCO, BPA, and SKQ (Kerr) Dam and the implementation of these three settlements throughout the 2000s still work for the CSKT. Many, if not all, of these staff will likely be retiring within the next decade. Unfortunately, the CSKT do not (as of yet) have a specific successional plan in place for continued staffing and leadership of habitat restoration and protection efforts in the Jocko River watershed. The continuation of these efforts in perpetuity, however, are memorialised in various conservation easements, inter-governmental agreements, and CSKT Tribal Council resolutions. Coupling these legal requirements with the strong cultural reverence for bull trout, the CSKT are deeply committed to continuing habitat restoration and protection efforts in the Jocko drainage regardless of the number of CSKT staff involved (Sydnor, 2021).

There have also been moments of conflict that have had to be carefully negotiated. For example, the removal of livestock from approximately 21,000 acres in the upper watershed of the Jocko River in 2010 initially created conflict between the CSKT Tribal Lands Department and the Jocko Valley Indian Stockmen's Association. A survey of over 100 Tribal members found that nearly three-quarters favoured eliminating cattle grazing from the upper Jocko drainage. The conflict was resolved through the CSKT Fisheries Program hiring a 'range rider' to keep livestock grazing compliant with the plan, and by moving habituated cattle to a recently acquired ranch. Within 5–6 seasons, once cattle were removed, the CSKT observed dramatic improvement in riparian and wetland habitat quality (Sydnor, 2021).

Has the Jocko River Restoration been successful? White says, 'Yes, I would say it's been very successful. Our children are learning about caring for our rivers and the bull trout are coming back. There is still a lot to do, but the restoration of the Jocko is an important beginning' (White, 2021). Sydnor agrees. 'Yes, I would also say it's been successful. But there are certain elements where it's too early to tell' (Sydnor, 2021). One of these is the long-term recovery of the bull trout itself. Factors beyond the scope of the Jocko River restoration, such as warming waters due to climate change and downriver dams that impede the historic migration of the bull trout, may yet doom the future of the bull trout in the Jocko watershed, even as other

[11] For more see: 'A Case History of Stream and Riparian Recovery on the Flathead Indian Reservation after Cessation of Cattle Grazing in the Upper Jocko River Drainage,' Craig Barfoot, Fisheries Biologist, CSKT, 2018.

Fig. 14.2 Students from Ms. Rhonda Howlett's fourth grade class at Arlee Elementary on a Jocko River restoration field trip. (Photo courtesy of Confederated Salish and Kootenai Tribes)

native fish such as the Westslope Cutthroat trout are rebounding and thriving. In that sense, the Jocko River restoration exemplifies both the successes and the limits inherent in many restoration projects: there are limits to the amount that change restoration can effect, due to pre-existing infrastructure such as roads, railroads, and dwellings, threats from ongoing development, and changing environmental conditions such as climate change. But the vital re-connection of human communities to the land continues and deepens. As Robin Wall Kimmerer observes, 'We need acts of restoration, not only for polluted waters and degraded lands, but also for our relationship to the world' (Kimmerer, 2013, p. 195) (Figs. 14.2, 14.3 and 14.4).

Interim Reflection: Expanding the Moral Community: The Cultivation of Care

Ecological restoration participation can be one way to build an environmental ethic of care (Keulartz, 2012). Ecological restoration is an educational and performative world-making practice, and it has been suggested that restoration performs 'the maintenance of the mental, psychological, moral, and spiritual structures' (Jordan & Lubick, 2011, p. 14). One of the important qualities of restoration is the

Fig. 14.3 Confederated Salish and Kootenai Tribal Fisheries crew electrofishing on a restored reach of the Jocko River. (Photo courtesy of Confederated Salish and Kootenai Tribes)

Fig. 14.4 CSKT river restoration specialist, Calvin Tanner, talking with students and Germaine White at a Jocko River restoration site. (Photo courtesy of Confederated Salish and Kootenai Tribes)

'questions it raises, and the ambiguities it dramatizes' (Jordan & Lubick, 2011, p. 5), and how it provides such meaningful educational opportunities. Asking and answering questions is central to restoration practice. In this way, it is a catalyst for conversation between people, their environments, their communities, and their histories. To philosopher Mikhail Bakhtin, self and culture are *created* through dialogue. Monological relationships mean that outcomes are fixed, whereas in dialogical engagements with place, outcomes remain open-ended (Holloway & Kneale, 2000, p. 76).

Restorationists are continually required to adapt their goals and their view of the moral imperative of why they participate in restoration. Historians of restoration William Jordan and George Lubick ask: 'how does a society come to…recognize or confer value not only on the members of the community made valuable by their familiarity but also on the unfamiliar other?' It is this quality, of an 'enlarged sense of moral enfranchisement', that they argue underpins what they call 'ecocentric restoration', and what becomes increasingly important as places undergo rapid change (Jordan & Lubick, 2011, pp. 18, 19). 'The importance of commitment in the present, amidst an uncertain future' is well understood by some restoration participants as being good in and of themselves; aware that 'this whole project in a way is looking after a landform which in all likelihood will totally disappear' (Pearce, 2018, p. 167). Discussing the ethics of ecological restoration, environmental philosopher Daniel Spencer states: 'I know for me it makes a difference about how I conceive restoration if I start with an expansive moral community with many moral interests that are more than human' (Spencer, 2015). As Cairns highlights, growing fears that restoration may encourage destruction 'may be justified unless ecology is accompanied by a changed environmental ethic' (Cairns Jr., 1995, p. 9).

Restoration is positioned to champion an ethic of reciprocity and a culture of sensitivity, inclusion, and negotiation. Personal affective encounters are central to ecological restoration work. As Leopold wrote, 'we can only be ethical in relation to something we can see, feel, understand, love, or otherwise have faith in' (Leopold, 1966, p. 251). Reciprocal relationships with nature can be nurtured through the giving acts of restoration practice (see Jordan, 2003). From botanist, professor, and member of the Potawatomi Nation Robin Kimmerer's perspective, '[E]cological restoration can be viewed as an act of reciprocity, where humans exercise their care-giving responsibility for ecosystems' (Kimmerer, 2011, p. 457).

Case Study 2: Preamble

Within the restoration community, there is a long tradition of acknowledging the potentially 'transformative' impact that involvement in restoration can have upon participants. The idea that participation in restoration can foster a connection with nature can be traced throughout the development of ecological restoration, both as a practice and as a field of research.[12] As far back as 1934, Aldo Leopold explicitly declared restoration as a social as well as ecological project. Here we look at a case of ecological restoration which demonstrates how participatory restoration can be managed in such a way that participants are given the opportunity to connect to nature.

Case study 2 reflects on the impacts of participating in a grounded, practical and physical experience of ecological restoration.

Case Study 2: Trees for Life in the Scottish Highlands: Ecological Restoration as a Way of Connecting with Nature

Ella Furness

The restoration which is the basis of this case study was organised by an organisation called Trees for Life (TfL). Trees for Life have been working since 1993 to restore the Caledonian Forest and all its constituent species of flora and fauna within the Scottish Highlands. They have been particularly successful at developing stability and support though public participation. Within their work they are committed to enabling participants to feel a deeper sense of connection with nature. This commitment to developing relationships with nature is central to deep ecology, which contends that 'people exploit what they have merely concluded to be of value, but they defend what they love' (Berry, 2000, p. 40).

To further these ends, members of the public can attend voluntary work weeks with TfL, which run throughout Spring and Autumn of every year. These weeks take place in the sparsely populated mountainous area that makes up TfL's core area (Fig. 14.5). The weeks are facilitated by the group's leaders (known as Focalisers) who live with up to ten volunteers for the duration of the week.

At TfL participants carry out a variety of work tasks, predominantly tree planting and removing non-native species. The tasks and the social experience of working together are both important in enabling participants to feel connected to nature; together they offer an opportunity to create a feeling of belonging within the ecosystem (see Furness, 2021a, b). The motivation that this sense of connection brings can be seen in the words of this participant, who has developed a relationship with the land:

I think [the sense of connection to nature is] because we're interacting…when you come here you see the broken ecosystem and landscape. You're coming here planting trees and you're playing your role in building it and you're just… it's like you're fitting into the ecosystem in a way. You're planting a tree, you're sowing the seed of life. You're generating it.

[12] For thorough explanation of this topic, see Furness (2018)

Fig. 14.5 Trees for Life's core area for restoration. (Source: Trees for Life)

...we talk about all these things like ecosystems services, carbon sequestration, restoration… but to me…going to help prevent flooding…would not be enough to motivate me to get out of bed on a rainy day and come and do it [the work]. To me, the motivation comes from, purely from personal love of the forest.

Below are some practical methods for enabling connection to nature which were learned through researching participants' experiences at TfL.

The Fundamentals

There are a range of tools that can be used to welcome people into a project. Some are instinctive, and many project leaders may use them as a matter of course. Other tools take much thought and planning. A series of observations over the course of a number of voluntary work weeks have revealed that some of the things that TfL does to welcome participants:

(I) They give people an induction to the site, its history, the vision of the project, the tasks to be carried out and their contribution to the overall vision. This induction is an opportunity to start people on a journey to feeling that they are part of the unfolding story of the ecosystem in which they are working.

(II) They provide sets of clothing that signal group belonging. For example, TfL ensured every participant had a matching high-visibility top and quality safety equipment that fitted them appropriately.

(III) They ensure access to professional standard tools and equipment which are in sufficient supply to allow participants to carry out allotted tasks.
(IV) They have routines for sharing food and cooking together, since it is well-accepted that sharing food encourages conversation and making social connections. As far as possible, TfL choose food appropriate to the vision of the project. In this respect, it is worth asking how the food being provided could have the least ecological impact. Consistency between the stated vision of the project and the group's everyday actions are important for fostering participants' confidence in the project.
(V) They give people responsibility. Professional grade tools need cleaning and maintenance. Food preparation and washing up are important tasks in themselves. Encouraging all people to take turns sharing the tasks that are necessary for the group's subsistence is in itself a key bonding activity. In addition, the sharing of essential tasks created variety in the work schedule and has enabled people to learn and/or use a wider range of skills than those that are directly required for practical restoration.

Group Cohesion and the Role of the Leader

To people who live in urban environments or who have little experience of the outdoors, nature itself may be unfamiliar, and the notion that it may be possible to feel a sense of belonging in the natural world may be intimidating or strange. Enabling participants to feel safe in this context is important when asking them to grapple with novel ideas. The role of a leader is not only to be responsible for the completion of practical work but to be a guide and facilitator of the social experience. A leader needs to use their own emotional skills to bring the group together and to ensure everyone is included. This is sometimes described as 'affective labour'. The product of affective labour is 'a feeling of ease, well-being, satisfaction, excitement, passion – even a sense of connectedness or community' (Hardt, 1999, p. 96). Here a leader describes this work:

> [It's] trying to … read… people's psyche, what might people think of a situation? How might they react? Try to gauge the group the whole time, there's a lot of psychology involved, you need to do that to make sure that you assimilate everybody into the group and that we assimilate into the group…

It is up to the leader to ensure that people feel like they are doing important work. In participatory ecological restoration the social process of involvement must be seen as being as important as the ecological outcome.

Bonding Tools

Trees for Life use 'sharings' to help the group cohere. At these 'sharings', all participants are present, and they are invited to reflect upon their experiences or share something of themselves. The group is facilitated to ensure that no interruptions or commentaries are made on personal contributions and that participants feel comfortable and safe in the session. Group building starts with an introductory sharing when everyone is required to introduce themselves and share their reasons for attending the session. Another opportunity can arise at the end of a session when participants are asked to reflect upon their experiences up to this point. Such activities can be presented as an informal 'check-in' activity in order to prevent participants from being daunted by the format. TfL emphasise the fact that for a sharing to work, it is 'important to let your group know that it is completely fine if they don't want to speak … it is important to ensure that people feel safe; your job is to hold the space open and invite people in' (Trees for Life, 2015, p. 33). At TfL sharings often helped create a collective group identity with jokes and observations that became part of a shared culture. Sharings serve to reinforce the social connection that develops within groups by establishing the group as a safe and respectful environment, breaking down barriers, and reducing the dominance of loud people or the risk of cliques forming.

As well as sharings, some leaders use games and activities to break down inhibition in groups. They can be simple things, like doing stretching exercises together before a physically demanding task or reflecting upon some favourite things about the day or afternoon over chocolate after work.

The Work of Restoration

There were several elements of the work of ecological restoration that fostered feelings of connection to nature. To varying degrees, these elements can be incorporated into the workplan of any restoration project. There is for example:

Physical immersion into the natural environment, which can make people feel more intimate with their surroundings. The act of handling the soil and plants, and being exposed to the weather, gave people an opportunity to become familiar with elements of non-human nature through their senses. This is seen as an opportunity to be immersed in the world of embodied sensation. In parallel with immersion, the engagement with hard physical tasks often gave participants a feeling of a sense of elation and achievement that made them feel positive about being in nature and made them feel as though they had earned their place within nature.

Engaging in a clear narrative of the practice. Restoration is explicitly about repairing past degradation of sensitive land areas, and leaders went to some lengths to explain to participants the history of the land in question. A discussion of how the land they were working on had been damaged, together with a vision of what it might look like in years to come through their contribution, was a powerful part of the narrative. Engaging with this narrative enabled people to feel as though they were part of a story of changing landscape. In addition, being able to feel as though they are 'doing something', enabled participants to feel that their actions were a meaningful contribution towards ameliorating environmental degradation. It was this *doing something*, the simple practical act of planting a tree or the act of cutting down invasive plants (a task which participants became experts in a few hours) that participants themselves often regarded as the most important aspect of their experience with restoration. There is a clear personal payoff obtained from carrying out work which is practical and can be witnessed at the end of the day.

Along with the narrative, the underlying symbolism of the tasks becomes central to the practice. Some of the tasks involved in restoration are more effective than others in enabling people to cultivate feelings of connection to the land. At TfL, tree planting is particularly effective because it is easy to see the outcome of participants' contributions, and it has been commented that it feels like planting a gift. It is common in Western culture to plant trees as dedications to people who have died, and this overlap means that restoration work becomes even more significant and symbolic, enmeshing the participants' personal understandings and beliefs with the ecosystem. Other tasks such as removal of invasive species were less effective in creating symbolic connections and fitted less easily into a simple gift-giving or redemptive narrative. However, whatever the work entails, it is important to celebrate the achievements together, to sit back at the end of the session and see the progress that has been made.

Bringing a certain mind-set to the practice. Notwithstanding the very practical nature of restoration, the nurturing of an appropriate mental state is of paramount importance. This will start with suitable education and knowledge sharing, which is a powerful tool for creating connection. The role of leaders in this first developmental step was important, and the detail of how they delivered information and stimulated people's curiosity in the natural world, was of critical importance. Leaders often explained the ecosystem in terms of relationships, and they invited people to see themselves as part of a bigger whole. Having said this, the leaders did not portray themselves as teachers, although many have considerable knowledge of their area. Rather, they were proactive in inviting participants to share their own expertise and knowledge with the group. Many participants could contribute knowledge of botany, ornithology, history, or geology, or contribute a technique for doing a particular task.

Mindfulness, reflection, and meditation were practices to cultivate a certain mindset where people paid quiet attention to their work and their surroundings. The introduction of mindfulness helped people to pay close attention to world around them, enabling them to be present in the moment, and to be free of their everyday concerns. When interviewed eight weeks after their volunteering experience, participants often said that this quality of observation and presence was what particularly made the experience of carrying out restoration significant.

Leaders guided observational group meditations where people stood together and paid attention to their bodily sensations such as listening to the wind or feeling their feet on the ground or meditations where people watched leaves fall from trees or trees bend in the wind. At other times, quiet meditations alone allowed people to observe their surroundings and think about the reasons why they were doing the work. Often after these meditations people shared their experiences and reflections together as a group. Leaders also used visualisation to create connection. For example, during tree dedications people were asked to imagine what the ecosystem would look like in five, ten, fifty or two hundred years.

Approach Cautiously: Some Limitations

Much of the foregoing discussion of the social effects of engagement with restoration arose from the particular circumstances of the experience. TfL specifically run weeklong camps where people are required to live together in remote locations. These provide an opportunity for groups of strangers to quickly become acquainted, and they usually feel a sense of growing allegiance to their group. This kind of rapid social cohesion would take significantly longer if circumstances meant that a group could only meet weekly or even less frequently to work together for short periods. Effects may also be limited if people have inconsistent attendance. It is also clear that some groups need more guidance or encouragement to bond than others. In this respect, a willingness among participants to discount their everyday status is usually sufficient for the group to begin to integrate, but some groups do not bond. At the individual level, leaders at TfL have worked with participants whose integration and accommodation in the group have been hindered by the existence of serious mental health problems, emotional distress, or active addictions. In addition, English is the common language for these weeklong camps, but, on occasion, limited understanding of English by one or more participants has posed a challenge for communication. Personality clashes can also raise challenges.

Also, it is apparent that not every participant wants to be involved in group or individual meditation. The early work of gauging the sensitivities of the group that the leaders do is most important in this regard, as it has been found that these more mindful or meditative practices may be alienating to some participants. Not everyone feels safe or receptive to the idea of quiet contemplation, and it is therefore important not to push people to do anything with which they are uncomfortable. Checking in with groups and getting anonymous feedback after various sessions is a routine part of TfL's work which has been put in place to meet individual needs in a non-threatening way. This is important to note.

Some restoration projects will not lend themselves to the methods and approaches described above, and indeed some project managers may not have the desire or resources to maintain focus on the social potential of the work. However, where there is the capacity for this kind of approach, tailored to the unique ecological and social circumstances of each project, there is the possibility of both ecological and social transformation. In giving people visceral experiences of landscapes that have been degraded and giving them the opportunity to contribute something tangible towards remediating that degradation, Trees for Life have been able to create a

Fig. 14.6 A volunteer planting trees by hand. (Photo: Ella Furness)

Fig. 14.7 Participants walking to a planting site with tools, the walk was over an hour each way. (Photo: Ella Furness)

different future in the minds of their volunteers. This is the power of encouraging participation in restoration: the possibility of restoring ecological hope and belonging, at a time when so many are desperate for both. These methods have enabled Trees for Life to gain significant public support for ecological restoration and rewilding practices. They were hosting 430 volunteers annually at the time of this research (Furness, 2018) (Figs. 14.6, 14.7 and 14.8).

Fig. 14.8 Looking at a wood ant (*Formica rufa*) nest underneath a Scots pine (*Pinus sylvestris*). A leader explains how the ants make their homes from the pine needles under the trees, and in turn eat the pests that harm the pine: A symbiotic relationship. (Photo: Ella Furness)

Interim Reflection on Building Communities of Practice

Through restoration practice, sensitive and detailed attention to surroundings can contribute to the grounding of an individual, both mentally and physically. Through focussed work-practises, carried out with like-minded individuals, communities are forged and strengthened. Restoration practice can therefore be described as one of phenomenologist Edward Relph's examples of 'communal experiences', where shared perceptions of place are negotiated, and experiences and social norms are reinforced. These experiences, Relph explains, cultivate a feeling of 'existential insideness', of 'belonging', as opposed 'existential outsiderness' defined by feelings of alienation (Relph, 1976, p. 55)[13] and through environmental practices such as ecological restoration, '[T]he landscape becomes the home-territory' (Reid & Beilin, 2015, p. 101).

One of the consequences of this work is the way in which it performs and reinforces certain ideas about belonging. The assertion that a person is at 'home' can be presented in multiple ways, and one of these, which is particularly relevant in this context, is through labour. Writer Gavin Van Horn draws on anthropologist Laura DeLind's notion of 'sweaty sacrifices' to describe how through restoration labour, '[M]eaningful relationships are invited, forged, and restored' (Van Horn, 2016). Historian Lorenzo Veracini's writing is useful here, reminding us that in

[13] Of particular relevance here is that it is suggested that the use of paid labour to professionalise and streamline the practice risks the precluding of the benefits of other participant's ethical encounter.

settler-colonial discourses, history is displaced while 'settler groups emphasise suffering as a strategy for legitimising their claim to country' (Veracini, 2007, p. 274). That is, labour and suffering became surrogate tools for legitimacy in a settler-colonial context. We are mindful that ecological restoration must be wary of becoming a practice that, in satisfying an individual's need for security and belonging, inadvertently silences other histories and other voices. It is essential that participation in ecological restoration work is not limited to a privileged few and must not exclude minorities and marginalised peoples within local places. Without taking seriously the histories, cultures, and knowledges of First Nations peoples as well as more recent migrant groups in particular places, restoration runs the risk of continuing the pattern of erasure that underpins settler colonialism.

Case Study 3: Preamble

Learning from Local Knowledge and History

Through engaging the diversity of existing relationships, interests and strengths found in local communities, restoration projects and restored environments are more likely to thrive. By inviting wider participation, diverse value systems can be simultaneously supported and integrated through the common shared desire for establishing flourishing ecological and social systems. This has been demonstrated in the chapters' previous Case Studies. Conservation programs, policy documents and the wider restoration community have an opportunity to respect local peoples and local knowledges, by bring people from diverse backgrounds together and to bridge traditional chasms between those in productive or preservationist camps.

Evidently, the knowledges and cultures that are unveiled with a more detailed understanding of place, offer important lessons to restoration ecology science and to cultural change. Today, community-based knowledge and conservation are presented as necessary to counter the abstraction and lack of engagement of top-down Western science (see Berkes & Turner, 2006; Berkes, 2007, 2009; Araujo et al., 2012). Recent directives of the Convention on Biological Diversity Aichi Targets and Intergovernmental panel on Biodiversity and Ecosystem Services call for greater public engagement in science, acknowledging that enhanced inclusivity is likely to produce longer-lasting and more holistic, place-based approaches to environmental conservation.[14] While a great deal of important knowledge and experience can be carried by just a few people, it is vital that this experience and know-how, the 'corporate knowledge' of projects, is not lost.

[14] Indeed, Target 14 highlights the role of restoration in meeting the targets. It read: '[B]y 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and wellbeing, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable' (Convention for Biological Diversity 2011, p. 2).

Placed-based environmental, cultural, and oral histories can provide rich stories and knowledges to inspire and inform restoration practice (Pearce, 2017, 2019, 2021). Effort put into such research can enrich an understanding of the 'layered landscapes' within which we all operate (Hourdequin & Havlick, 2011, 2016). This underlying work is critical to identifying and responding to the cultural and material causes of damage and degradation thus meaningfully widening the scope of understanding and accountability, and drawing together stakeholders and funding sources for restoration work (such as the focus on environmental contamination in case study 1).

New inclusive dialogues with place are also central to expanding recuperative potential. As Plumwood advocates: '[W]e need a cultural paradigm shift in many linked areas to adopt a partnership or dialogical model of relationships with nature in place of currently disabling centrist control' (Plumwood, 2002, p. 238). It is useful here to remind ourselves of Higgs' important notion of 'focal restoration' for its potential for participation, positive political change, and the cultivation of ecological citizenship (Higgs, 2003).

The following case study documents the importance of community collaboration, adaptation and engagement for restoration success.

Case Study 3: Wetland Restoration Case Study: Reflections on Community Participation in Nature Glenelg Trust (NGT) Projects

Mark Bachmann, Nature Glenelg Trust

Nature Glenelg Trust (NGT) has been delivering ecological restoration projects in south-eastern Australia since 2012 and has shown particular success in restoring the hydrological regime, which is the natural depth and duration of inundation, at dozens of previously modified wetlands. The sites are situated on the traditional lands of a wide range of Aboriginal communities, from the Wurundjeri in the east near Melbourne (Victoria) to the Kaurna in the west near Adelaide (South Australia), and many other important groups situated between them. Our primary objective in undertaking this work has always been to initiate the recovery of degraded environmental values, including the return and recovery of threatened species. However, we have increasingly come to understand that, although this was not our initial focus, these projects also have a major social dimension, whose importance to every step in the process (initiation, delivery, and monitoring) cannot be understated. This is underpinned by the collaborative, open, and flexible way in which we have developed and delivered projects, which we suggest has driven our track record of success.

It is important for us to state here, that if you are reading this case study in order to find a simple process to be followed, which is often how the scientific method trains us to think, then you will be disappointed. This is an intuitive but logical way of approaching communications and involving people in restoration work – a combination of actively shaping, but also reacting and adapting to, the social dynamics that are intertwined in every environmental issue. This is a philosophy more than a method, because no two social or ecological landscapes are the same.

As a result of this inherent diversity and complexity, the nature of community involvement in NGT projects has varied considerably.

Restoration Project Concept Development and Initiation

We see that ecological restoration projects can commence in a wide range of ways, often broadly aligned with land tenure, but also influenced by the activities of various organisations involved in land management. In our work for example, in relation to wetlands:

(i). *Projects on public land* are often driven by community appetite for action and/or government agency policies, strategies, or plans. This has often included public land currently managed as nature reserves, where historic changes that pre-date reservation status are causing detrimental ongoing impacts to environmental values.

(ii). *Projects on private land* are often initiated by private landholders seeking support and/or prompted by direct approaches made by grant-funded programs. This usually involves land that is being managed for production, such as agriculture or forestry where existing values are being compromised by drainage and subsequent land use changes.

(iii). *Projects on land where a change of tenure is required to facilitate action* are negotiated and led by either government or non-government organisations, as the future site owner, often with strong community backing. This land may be in strategic locations for biodiversity restoration but can have a long history of agricultural production. This category includes land now owned by NGT.

For interested community members (who are not landholders), the level of motivation towards, and type of involvement in, each of these project development pathways varies, but they are particularly important for public land or private land which is specifically managed for community benefit.

Overcoming Barriers to Restoration Action

Now that NGT has become involved in progressing wetland restoration projects under each of these project development pathways, we have some observations to share about overcoming barriers to restoration action. With the exception of sites entirely contained within private land where the agreement of the landholder has been reached, most other wetland restoration scenarios involve a more complex set of discussions with a very diverse group of people. The list is varied but may include: neighbours, numbers of community group members, academics, government agency staff, public land managers, and other interested people.

Sometimes the community strongly support action, but there is government agency hesitation or concern, while in other cases it is the reverse. In many other cases, there is no uniform collective 'position' by any group of people or agency, and restoration efforts can stall. This is a common cause of escalation of community frustration.

Resolving these issues requires adopting a method of flexible, effective communication, tailored to suit the unique circumstances surrounding each site. Methods that we have used that can help to create an atmosphere conducive to finding a way forward have included:

(i). Talking honestly and openly to all parties and listening to their concerns, in a place where they are comfortable.
(ii). Doing the homework necessary to identify if the perceived obstacles to action are real, and sharing this information in a transparent way. For example, providing updates as the work unfolds shows the community and others a level of respect that is then usually reciprocated. It also flags in advance where a project is heading before anything is locked in, and triggers the necessary, key conversations.
(iii). Removing or addressing active sources of tension, in particular pulling the issue out of the political arena.
(iv). Providing a fresh, independent, scientific, and well-informed perspective on the issue.

Throughout this process, NGT works as both an informed intermediary and active participant, adjusting our approach and accommodating new information that comes to light. Building the necessary trust with all parties is something that is often easier to do when not a historical player in the issue and is also greatly assisted by being independent of government (especially for building trust with the community).

Reaching the point where everyone is engaged, listening, and open to receiving new information is a critical step because this type of trust is the pre-requisite for agreeing on solutions. Confidence levels can then further grow, as NGT has then been able to:

(i). Investigate and address real information gaps.
(ii). Devise and communicate a way forward that everyone can support.
(iii). Propose trial measures to test ideas and assumptions when still in doubt.
(iv). Help resolve the difference between real and perceived risks.
(v). Work towards the lasting options that emerge which everyone can agree to.

Some readers may have heard of the mantra 'decide, announce and defend' as a public engagement strategy, which is sometimes adopted by government. It refers to a method where the technical work that informs an approach to an issue or decision occurs without genuine public input, scrutiny or discussion before a decision is made. NGT's philosophy of engagement with the community is the opposite. Our approach is to actively share, inform, discuss and negotiate with all interested parties (especially the community) while we investigate a site, prior to any decisions being made. The importance of making complex science and technical information easy to understand and share is vital to this process. This is greatly assisted is taking the time to source, review and integrate other forms of historic information, to sufficiently guide and calibrate our understanding of a site and its post-colonial past. Despite this type of multi-faceted approach being initially more time-consuming and energy intensive, it can result in a deeper shared understanding of a site and its history of change being achieved across disparate groups with an interest in a site. In turn, this ultimately leads to faster agreement and more effective, lasting action.

Implementing Restoration Action

Fortunately, wetland restoration is also particularly suited to trial solutions that provide a demonstration of potential outcomes, building confidence in all parties. This tactic has been necessary at a wide range of sites, but often it is instituted for very different reasons. Some examples that show the value of trial structures (such as designed geo-fabric sandbag structures) are their ability to:

(i). Physically demonstrate inundation impacts to satisfy neighbours.
(ii). Determine whether ecological and hydrological changes are desirable.
(iii). Test gaps in technical knowledge, such as testing the final water level in densely vegetated wetlands where remotely sensed elevation data can be quite inaccurate.
(iv). Provide a viable initial alternative to high-cost 'hard' infrastructure, such as concrete weirs.
(v). Sensitively manage restoration at culturally significant sites.
(vi). Be adjusted in real time, and even reversed, if necessary.
(vii). Be converted to a permanent solution in the future where appropriate, by covering with earth and revegetating, as has now occurred at multiple sites.

Perhaps most importantly, trial structures also provide an ideal opportunity for hands-on community participation (Fig. 14.9). Our community sandbagging events have attracted a wide range of people, groups and organisations. This type of practical activity, directly contributing towards a lasting solution is also very therapeutic for community members who may have felt frustrated by a long period of inaction.

Fig. 14.9 Community involvement in trials (left to right, top to bottom): Gooseneck Swamp, Brady Swamp, Long Swamp Phase 1, Walker Swamp, Mt. Burr Swamp, Long Swamp Phase 3, and Glenshera Swamp. (Photo: Mark Bachmann)

Lasting Project Outcomes

Having often acted on their concerns, worked to build the consensus for action, and then initiated restoration works, NGT often finds that those community members who have contributed significantly to the initiation work of the project, are often interested and motivated to participate in the steps that follow. On public land sites, these steps might involve various forms of ecological and hydrological monitoring, and on NGT Reserves it might also involve a wider array of land management activities such as seed collecting, revegetation and weed control. In many cases, it can extend to assisting with infrastructure projects like maintaining and repairing fences, sheds, and equipment.

An example of this selfless work is given by NGT volunteer Gordon Page, who first assisted with sandbagging for the wetland restoration trial at Long Swamp simply 'because it sounded a bit different'. He has gone on to assist NGT with a wide range of other projects as a volunteer, including the sourcing and installation of donated second-hand windows in the shearing shed overlooking the restored Mount Burr Swamp. The shed has now been repurposed into an education facility to accommodate schools and other visitors to this NGT Reserve where they are able to learn about environmental science and ecological restoration (Fig. 14.10).

Fig. 14.10 Gordon Page's volunteering with NGT began after involvement with a sandbagging project. He is pictured here overlooking Mount Burr Swamp through the windows he installed. (Photo: Mark Bachmann)

Interim Reflection on the Engagement of Volunteers

There have been many instances of support for NGT managed projects by volunteers in the past few years. We feel that by bringing them with us on the restoration journey, then involving them in implementing solutions and the ongoing maintenance of a better state and trajectory of recovery, the community in general now feel more invested in these wetland restoration projects and their future care. From both the long-term social and ecological perspective, this is an incredibly important outcome.

Concluding Remarks on the Critical Social and Political Practice of Ecological Restoration

Emotions in Ecological Restoration

The affective elements of restoration expressed in this chapter underpin, in a powerful way, the essential component of emotions to the successful engagement of people with practice. Emotions, particularly shame and grief, have previously been considered in ecological restoration (see Jordan, 2003; Hobbs, 2013), but as this chapter demonstrates, the emotions experienced through the practice itself can be central to its transformative capacity. The accessing of notions of 'loss, nostalgia, disappointment, frustration, hope and love all form part of the restoration praxis, by cultivating new knowledge, reflection and conversation' (Pearce, 2018, p. 170). Emotions also engage people with history through a parallel kind of healing potential. Dwelling on the act of restoration, understanding the multiple causes of degradation, and plotting out alternatives for ways of being in relationship with place, are all part and parcel of a wider social understanding.

Those who forge strong bonds with a place and the land also open themselves up to the need to confront the overwhelming realities of ecological decline, extinction, and the wider impacts of climate change. As Leopold powerfully describes:

> [O]ne of the penalties of an ecological education is that one lives alone in a world of wounds. Much of the damage inflicted on land is quite invisible to laymen. An ecologist must either harden his shell and make believe that the consequences of science are none of his business, or he must be the doctor who sees the marks of death in a community that believes itself well and does not want to be told otherwise (Leopold, 1966, p. 167).

In 2013, Hobbs wrote a paper that applied Kubler-Ross's psychological 'stages of grief' to relationships with environmental change. He suggested that clashes in restoration approaches and ethics could be attributed to people sitting at different points along a grieving journey through the phases of denial, anger, bargaining, depression, and acceptance (Hobbs, 2013). Unlike with human death, Hobbs suggests that conservation losses are 'diffuse, chronic, and uncertain' (Hobbs, 2013, p. 147). While Hobbs primarily writes about the loss of species and ecosystems, the communal experience of loss shared in ecological communities can resurrect a more ethical, relational, collective grief.

In a colonial context, the origins and cultural significance of recent and accelerated extinction and loss cannot be overlooked. Australian philosopher Deborah Rose explains that extinction can be considered a settler-colonial legacy of 'both genocide and ecocide' (Rose, 2004, p. 35); a perspective which spurs a different kind of collective grief. For many First Nations, 'loss of species is loss of kin, of language, of culture, of relationship, and of ways of life' (Pearce, 2019, p. 263). Writing on extinction, Australian philosopher Thom van Dooren describes how the experience of mourning can make us more conscious of our relationships with other species and instil a caring responsibility. He advocates that 'taking it seriously, not rushing to overcome it – might be the more important political and ethical work of our time'. 'The reality', he writes, 'is that there *is* no avoiding the necessity of the difficult cultural work of reflection and mourning.' 'This work', he continues, 'is not opposed to practical action, rather it is the foundation of any sustainable and informed response' (Van Dooren, 2014, p. 4). The outcomes we desire may not always be possible; the ecological and cultural work of recuperation is never remotely done, but by staying with the discomfort of that fact, we might afford different possibilities.

From a position of ontological plurality that takes seriously inter-species relationships, conservation and restoration practitioners are encouraged to ask: What other elements of the place have been lost? What other elements of place are worthy of mourning? What other elements of the place ask for healing? 'One way to live and die well as mortal critters', explains Haraway, 'is to join forces to reconstitute refuges, to make possible partial and robust biological-cultural-political-technological recuperation and recomposition, which must include mourning irreversible losses' (Haraway, 2015, p. 160). Hope comes in repeatedly turning up, (and never giving up), seeing value in action, and caring amidst uncertainty.

Clear, creative language that welcomes uncertainty and emotion, with 'vision and poetry' (Collins & Brown, 2008, p. 217), can better deal with the complexities faced in practice while also enhancing resonance with policy-makers and the general public (see Jørgensen et al., 2014). Similarly, there is a place for all forms of creative projects that convey intimate stories of situated experiences of restoration.[15] These stories are bioregional, personal, and performative and hold the potential for strong cultural change.[16]

Enlivening a Politics of Practice

In 2017, David Greenwood published a paper that reconsidered Aldo Leopold's 1934 University of Wisconsin Arboretum speech, often cited as the first public articulation of the rationale for ecological restoration (Callicott, 1999; Meine, 2010). Leopold, writes Greenwood, 'would have viewed it as quintessentially pragmatic to call out—publicly and explicitly—the forces that make it [ecological restoration] necessary' (Greenwood, 2017, p. 684). As sociologist Hannah Holleman powerfully critiques in a historical study of the North American Dust Bowl, '[m]ainstream environmentalism … offered the illusion of resolution, while the social drivers of the crisis remained intact' (Holleman, 2018, p. 10). A danger of ecological restoration work is that it can appease the concerns of the people creating satisfied 'do-gooder' communities, making them (and society at large) feel they have done enough to look away from on-going industrialisation, land degradation and cultural violence on neighbouring (or indeed the same) land. Ecological restoration needs to be partnered with the mutual endeavour of political action and policy change to protect environmentally and culturally significant places. Beyond this, as the case studies have demonstrated, it can participate in overdue social change that works towards more just and inclusive communities.

Ecological restoration efforts are increasingly woven into relationships with a mesh of financial markets and programs that aim to contribute to many of the critical environmental, economic and social challenges of our times. This will increase as ecological restoration efforts scale up and increasingly professionalise through

[15] For example, The University of New South Wales and Landcare Australia have partnered in a storytelling project called *Rescue,* a platform to document and share restoration stories with an aim to create a podcast. It is a powerful collection of the emotional, personal, community meaning behind restoration work, premised on the idea that '[I]n rescuing we too receive something in return' (Miller 2018, p. 1).

[16] For a powerful example of restoration practice involving a literal performance to create 'living myths of regeneration' at the *Mallee Fowl Festival* in Central Victoria, see (Mathews 2011, p. 281).

the UN Decade on Ecosystem Restoration (2021–2030). It is crucial that neoliberal economics and associated inequitable frameworks are not uncritically accepted. Neoliberalism seeps into restoration practice in problematic ways: capitalist markets controlled by extractive industries drive investment and decision-making; ecological systems are written into tradable markets and environmental offsets accessible to a select few. There is a risk of ecological restoration perpetuating patterns of colonialism and destructive neoliberalism.

The social equity aspects and political implications of market-based programs in restoration work are only recently being revealed (see Elias et al., 2021; Kandel et al., 2021; Kariuki & Birner, 2021). Market-based programs like Payment for Ecosystem Services (PES), Farmer Managed Natural Regeneration (FMNR), and Natural Capital Accounting (NCA) come with their own cultural assumptions and exclusions of certain peoples, knowledges and values. For example, a study on gender equity in market-based ecological restoration programs in Kenya found that power imbalances condition socio-economic outcomes of PES schemes and that 'governance structures exclude women from decision-making processes and from receiving direct PES benefits despite their labor contributions to restoration activities' (Kariuki & Birner, 2021, p. 77). For NCA, First Nations communities are widely underrepresented in program development and 'Indigenous knowledge is rarely considered in the design of natural capital measurement systems' (ClimateWorks Australia, 2019, p. 12). It is the responsibility of the ecological restoration community to reflect on the social and political implications of restoration work, especially as it expands into new regions and new scales. By understanding and appropriately responding to the local historical and cultural contexts within which restoration programs occur, existing inequalities and power imbalances can be addressed head-on. This forms a crucial part of wider and intricately associated restorative practices.

The ethic that underpinned contemporary restoration discourse attempted to break down dominant Western dualisms and extractive mentalities. As Higgs states: '[I]f ecological restoration exists only to perpetuate the separate estates of nature and culture, it will not break the pattern' (Higgs, 2003, p. 240). In order to respond to the systems that create degradation, we still need substitutes for 'oppressive rationalist and dualist structures' that 'help us acknowledge our ecological embeddedness' (Plumwood, 2002, pp. 36, 3). For example, Aldo Leopold's important 'land ethic' has been described as 'ecofeminist friendly', because of its focus on perceptivity and receptivity (Norlock, 2012, p. 507). It is the breaking down of dualisms and the nurturing of a relational and responsible ethic that matters here, a built-in relationship with local communities and a commitment to cultivating care and respect. The role of people is always unfolding, responding, weighing-up and reacting. With this in mind, ecological restoration can help to both envisage and practice a new relationship with human and more-than-human worlds.

The Importance of Flexibility and Innovation

Ecological restoration is strengthened by expanding its elements of surprise, wonder, and potential to act. In 2018, a detailed critique of the 'standards' approach was offered by Higgs and others. Their central argument is that a standards-led rather than principles-led approach can (i) hinder adaptability, flexibility and innovation, (ii) discourage restoration efforts in heavily degraded areas, (iii) exclude important, yet unquantifiable social and cultural categories of concern, (iv) avoid facing real complexity in diverse contents, and (v) impede the continual development of a relatively young practice (Higgs et al., 2018). These authors reiterate the strength of a principles-based approach, first articulated by Keenleyside and others to allow restoration to retain more flexibility and inclusivity so necessary to respond to the messiness of complex and changing social and ecological systems (Keenleyside et al., 2012).

Recent focus on technical standards can propel ecological restoration into becoming reduced to a modernist technological fix while overshadowing the significant social, cultural, and political work of ecological restoration. We need, as philosopher Marion Hourdequin and geographer David Havlick advise: '[A] richer, more empirically informed analysis of the social, ecological, historical, political, and institutional contexts in which such restorations take place' in order to identify 'important questions that might be missed by traditional ethical analyses of restoration' (Hourdequin & Havlick, 2011, p. 86). Principles-driven participatory restoration, conscious of local history and culture, can 'provide alternatives to dominant contemporary narratives of crisis, fear and commodification, and nurture relationships between people and place through change' (Pearce & Furness, 2016, p. 12).

Starting with Place

The democratic and political potential of ecological restoration resides in its capacity to react in a participatory and reflective way, within, and in response to, localised and particular histories, ecologies, cultures, and communities. As much as leaning towards strong science (necessary to understand ecological decline and guide strong, well-informed, and appropriate restoration projects), contemporary ecological restoration also needs to lean into social and political cultures. The case studies in this chapter present powerful examples and tools for expanding the recuperative potential of ecological restoration.

As this chapter demonstrates, the individual and communal social practices within restoration can cultivate generous relationships with place. Questions of responsibility and justice are uncovered. Moral boundaries are pushed, education flourishes, and people weave themselves into committed local relationships through affective experiences. These subversive characteristics of ecological restoration that run counter to fast, extractive and harmful relationships with place can influence its larger contribution to much-needed societal change. They can help to identify and resist the systemic nature of environmental injustices and drivers of environmental harm, rather than responding only to their outcomes.

Through probing questions, the complex specificities of place, with which ecological restoration must converse, can be better understood. This in turn expands pathways to response and capacities for action.

Questions to Guide Site-Specific Restoration Practice

What is the history of this place?
What cultures and cultural practices coevolved with the ecosystem(s) found here?
What cultures and cultural practices led to ecological degradation here?
What other violence or wounds need attention?
Who has a voice, who is left out and where are the ongoing injustices?
What kinds of futures are desirable to the expansive local community?
What unique ecological values exist here now?
What are the unspoken questions with difficult answers?

The urgency that climate change presents only sharpens the need for a turn towards complexity and uncertainty and to clearly articulate values. The present challenges call for attuning the senses to local stories, local lives, and local opportunities. The resounding lesson for the field of ecological restoration is that projects that overlook or dismiss the capacity for ecological restoration to be a critical social and political practice will fail both practically and culturally. Responsible restoration looks both backward and forward. Conducted this way, ecological restoration can be experienced as a powerful practice of conversation, commitment, and care.

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Chapter 15
The Economics of Ecological Restoration

Neil Perry

Contents

Summary and Key Lessons
To date, there have been limited applications of cost-effectiveness analysis and cost-benefit analysis by those involved in ecological restoration, yet these can be powerful tools to improve the effectiveness and efficiency of management and research efforts. First, cost-effectiveness analysis allows restoration practitioners to develop procedures that achieve the greatest ecological value for the limited money which has been made available for restoration. Second, cost-benefit analysis is needed to justify the importance of restoration projects, giving these projects a better chance of competing for government and other funds in a field of worthy causes. At the same time, it is recognised that the use of economic tools can unintentionally commodify and therefore undermine ecological restoration. Nevertheless, using the language and practices of economics, while consciously maintaining an environmentally friendly ethical position, will markedly strengthen restoration proposals.

In this regard, estimating the size of the restoration economy in terms of tangible economic outputs and number of jobs created provides a powerful narrative for supporting restoration work. With a clear narrative outlining their estimates of output generation and employment creation, restoration practitioners can respond to the

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jobs-versus-environment rhetoric attached to development proposals. For this purpose, economic tools such as input-output analysis and economic impact analysis can be adjusted to estimate the size of the restoration economy. This will assist individual restoration projects to be accompanied by positive employment and output impact analysis.

Researchers have already begun analysing the economics of on-farm restoration. From a theoretical perspective, hedgerows and pockets of biodiversity have been shown to increase mean yields and reduce the variance of yields, which is valuable for risk-averse farmers. In particular, hedgerows have been shown to provide a return on investment in 7–16 years through increased pollination and pest control. Using similar arguments, economics research has the potential to reduce information deficiencies that cause a 'market failure' – a situation where the private market is not efficient and government regulation is needed – and can increase the probability of obtaining further support for restoration work. Return-on-investment research can also challenge long-held institutions or norms in farming which may sometimes serve to obstruct restoration strategies. In this regard, however, research needs to be carefully replicated and contextualised because yield and variance are geographically and crop specific.

With regard to policy, large-scale restoration projects aiming to meet international agreements will be clearly enhanced by incentives for landowners, but top-down policy is often too blunt and does not address the root cause of environmental degradation, which is a lack of appreciation for the role of restored ecosystems. This can be because the value framework that drives neoliberal, capitalist societies is focussed on growth, jobs, and profits which mitigate against arguments for effective restoration approaches. Cooperative approaches to large-scale restoration may be able to challenge this value framework by empowering the position of local communities, building trust amongst different governance layers, and using restoration solutions that are culturally and geographically specific.

More specifically, biodiversity markets which are crafted as policy solutions can create incentives to restore land, but these come at the cost of more universally valuable biodiversity actions. Traditionally, markets are seen as appropriate in standard economic theory, but the ontology underlying such theory is one of atomistic and independent agents. That is, humans are assumed to act independently, or in an unconnected way, to maximise their utility, rather than responding to a connected network of human values. This theoretical structure does not map well with the interdependence of elements of nature and society, and it is suggested that cooperative solutions may be needed to address the multiple market failures and institutional norms that currently exist in society.

Management Implications

This discussion presents and explores five issues that, it is suggested, will lead to more effective and efficient restoration programs. Each of these five issues has an economic focus which will give greater traction to funding applications and relevant arguments regarding ecological research. These issues are as follows:

- The need for economic-advised research on the cost-benefit analysis and cost-effectiveness analysis of restoration management, which is essential for ecological practitioners to engage in the language of economics, while also arguing for restoration from an ecologically sound underlying ethic.
- The importance of a research agenda to estimate the size of the ecological restoration economy, which should be enhanced so that ready-made tools are available to calculate the economic impact of restoration projects in order to counter the unhelpful jobs-versus-environment rhetoric.
- The recognition that niche research on the economic benefits of restoration for farm productivity is valuable, but it is context-dependent. More funds need to be allocated to contextualising this work to comprehensibly bridge information deficiencies in order to change institutional norms in farming communities.
- On the policy front, large-scale restoration to meet the objectives of international programs requires buy-in from locals. Top-down policies that encourage restoration are needed, but to change values in society, cooperative solutions amongst local stakeholders are also essential.
- At a more micro-level scale, landowners need to be compensated for restoration work in a way which includes the opportunity cost of their land. However, biodiversity markets will not change underlying values, thus cooperative solutions may be preferable.

Introduction

The discipline of economics is concerned with the allocation of resources such as land, labour, and capital. These resources are scarce in our community, and they need to be allocated to the most valuable uses to ensure economic efficiency. Whilst it is becoming more widely recognised that ecological restoration is one of these uses, it must compete with other, equally important areas. For example, restoration of native landscapes uses land that is increasingly valuable for agricultural, residential, and industrial purposes. It is therefore essential, from an orthodox economics perspective, that researchers and land managers can persuasively demonstrate that restoration is the most valuable use of the contested land in the context of a particular time and place.

The pressing need for this work is indicated by the fact that, globally, many countries have signed international agreements, such as Aichi Target 15, Milestone A1 of the Post 2020 Global Biodiversity Framework, the Bonn Challenge and the UN Decade on Ecosystem Restoration, all of which require significant ecological restoration to be carried out. In addition to these signposts, in many countries, members of society value restoration activities for aesthetic, recreational, cultural, and bequest reasons. These are in conjunction with the ecosystem services they provide, such as pollination services, flood and soil protection, heat amelioration, and carbon sequestration. Yet, notwithstanding these contributions to the well-being of society, restoration must still compete for funding and support with alternative concerns such as health and education. In this struggle, the promotion of ecological restoration is inhibited by those institutions and norms or value frameworks that favour industries producing immediate income, output, growth, profits, and jobs.

Conflicting goals and pressing needs suggest that ecological restoration as a discipline needs to embrace the principles and language of economics. However, restoration practitioners must be aware that the theoretical and policy prescriptions of economics are grounded in a particular type of philosophical ethic which may be unfamiliar to many. Utilitarianism, or more generally consequentialism, focusses on the human welfare and aggregate outcomes of actions rather than the duties and obligations inherent to human actions (Wilber, 1999; Bromley, 2004) and entails a particular ontological structure that may not be fit for the purposes of ecological restoration (Perry & Primrose, 2015). This creates a conundrum because, on the one hand, economic tools and arguments must be used to compete equally for funds and resources, and to counter the rhetoric of jobs-versus-environment. On the other hand, in embracing economics and therefore commodifying nature, restoration practitioners may undermine the real foundations of natural systems over the long term.

It is becoming clear that this philosophical conundrum is evident in many policies which support ecological restoration. Ecologists themselves often openly support the introduction of biodiversity markets, considering that this is the best way to at least get something done for nature in the political environment of the day. However, it has been pointed out that such environmental pragmatism (Spash, 2013) is detrimental to the foundations of restoration thought over the long term, because it supports the very values that are the cause of environment degradation to begin with. Ecologists therefore need to understand and grasp the underlying ethical, ontological, and epistemological structure of economics, allowing them to cast a critical eye over restoration policy prescriptions that derive too simplistically from the unrealistic theory of orthodox economics (Spash, 2013: 354). Heterodox economics, which involves analysis of the institutions or norms in society, starts from a different ethical position, broadening the value framework and involving a more interconnected ontological structure which may be a better basis upon which to approach the economics of restoration (Perry & Primrose, 2015). Nevertheless, restoration still needs to at least understand and speak the language of orthodox economics to compete on a level footing in the global economy; hence, the philosophical conundrum arises.

In this chapter, I have attempted to provide a background to the orthodox economics of ecological restoration while also casting a critical eye over the outcomes and recommending ways to avert the conundrum using five quasi-case studies. First, I look at the work that has been undertaken in cost-benefit and cost-effectiveness analysis. As these tools are increasingly being required to justify all projects funded by governments around the world, they are critical to justifying restoration projects in debates held in neoliberal economies. They also help to achieve the greatest value for money from restoration funds, but in doing so they are bound to commodify nature. Thus, ecologists need to more critically understand the tools whilst maintaining their own ethical justifications for promoting restoration work.

Second, I consider the economic tool known as economic impact analysis, the use of which would allow ecological restorers to speak the language of economics and counter the jobs-versus-environment rhetoric that currently occurs around the world and which inhibits restoration work. We note that impact analysis has already been applied to the restoration economy, and this case study indicates how it can be enhanced to provide ready-made tools for ecological restorers to promote the economic impact of their work.

Third, I turn to the understanding of niche work on the economic benefits of on-farm ecological restoration. This case study shows how and why economic benefits arise from restoration, but it is so location and crop specific that it does not totally assist in filling information gaps for farmers in other locations, nor does it break the long-held institutions and norms that favour land clearing.

Ultimately, I suggest that it is these institutions, norms, and values that need to change and top-down policy that derives from orthodox economic theory is unlikely to work for these aims. Thus, the fourth and fifth case studies focus on an alternative large-scale restoration project that favour a cooperative solution, and the fundamental issues around biodiversity markets as policy prescriptions.

To an economist familiar with the shortcomings of orthodox economics, the history of research in the economics of conservation and valuing nature, and with experience in working with governments in the area of cost-benefit analysis and conservation economics, I propose that these are the fundamental elements that all ecological restoration experts need to understand. Such knowledge would allow ecologists to better articulate the language of economics, counter-arguments from industry lobby groups and propose alternative solutions to neoliberal governments. However, the elements I described here also warn against the blind adherence to the policy prescriptions and commodification that arises from orthodox economics, of which all ecologists must have some understanding.

Case Study 1: Cost-Benefit and Cost-Effectiveness Analysis

The literature on ecological restoration could benefit from a detailed economic analysis of costs and benefits. With limited past applications, several authors (Birch et al., 2010; Newton et al., 2012; De Groot et al., 2013; Kimball et al., 2015; Iftekhar et al., 2017; Wainaina et al., 2020) have highlighted the need for, and approach of, cost-benefit and cost-effectiveness analysis. They have also provided key lessons and directions for the future. In this case study, I discuss the two related methods of cost-benefit analysis and cost-effectiveness analysis with two particular goals in mind. First, in conservation and restoration, it makes sense to derive the greatest value for money from the limited funds available. In aggregate, restoration outcomes can be improved if the funds are allocated to the projects with the greatest benefit per dollar of funds spent, which is the principle of cost-effectiveness. Second, cost-benefit analysis and related economic justifications are becoming increasingly common in all aspects of social and environmental decision-making around the globe. Neoliberal governments, as a part of their political stance, require evidence-based policy and decision-making based on economic returns. If restoration practitioners cannot justify projects within these parameters, then they risk being sidelined by other environmental and social projects which can be justified on the basis of economic costs and benefits. Of course, analysing restoration, or indeed any environmental or social project in economic terms, risks the commodification of nature, therefore fundamentally undermining the philosophy of maintaining natural areas (Polanyi, 2001[1944]: 75). However, understanding and speaking the language of economics allows restoration practitioners to anticipate economic arguments while, at the same time, they can continue to ground proposals in their own strong philosophical ethic.

Cost-benefit analysis (CBA) can be thought of as the practical application of the underlying ethic of economics. Economics is grounded in consequentialism, and more specifically utilitarianism, which focuses on the aggregate outcomes of an action, questioning whether the consequences of an action are in general positive (Wilber, 1999; Bromley, 2004). In this regard, the Pareto efficiency criterion in economics holds that an action is good or right if someone is made better off without anybody else being harmed. For example, if an ecological restoration project improves the welfare, the utility or the happiness of at least one person without harming any other, it is justifiable. However, given limited funds or scarcity of resources, together with the trade-offs that scarcity implies, actual Pareto improvements are rare. For example, a restoration project might require the cessation of farming on someone's land or the use of funds that could otherwise go to new roads or medical research, all of which could reduce the welfare of others. Alternatively, the 'Kaldor-Hicks' compensation test suggests that a potential Pareto improvement can justify the support of projects when the winners of a project or policy can (theoretically at least) compensate the losers (Kaldor, 1939; Hicks, 1939). Cost-benefit analysis is the practical application of the potential Pareto improvement rule. If applied consistently, it is argued that everyone wins, regardless of whether compensation actually takes place for any given project (Boardman et al., 2018).

Of course, there are numerous objections to the use of CBA for project justification which are based on (i) the differential political and economic power of winners and losers, (ii) the inability to monetise certain environmental and cultural benefits and costs, and (iii) the appropriate discount rate to use, where the discount rate converts future costs and benefits to present values. For example, in the case of the latter, a large discount rate reduces the present value of future environmental benefits from restoration projects. However, the importance of the logic of cost-benefit analysis cannot be denied. It is a key strategy in assuring the best value for money from restoration projects and is therefore influential in convincing neoliberal governments that ecological restoration should proceed.

A related but alternative approach is cost-effectiveness analysis (CEA) which might be used when monetising the environment is undesirable or impossible. For example, rather than monetise human life, health economists use a measure of the quality of someone's health – the quality-adjusted life year (QALY) where a score of 1 is perfect health for 1 year – and seek to fund projects that achieve the best

QALY values per dollar of expenditure (Weinstein et al., 2009). CEA does need a common metric of success, and in the current discussion, this could be the amount of native vegetation cover, the number of species per area, or the value of ecosystem service provision. Cost-effectiveness analysis then proceeds by prioritising the ecological restoration projects with the greatest success potential per dollar of costs.

Both CBA and CEA research have been scarce in ecological restoration, but there are notable exceptions which highlight the key features of the work, including a study by Newton et al. (2012) who applied cost-benefit analysis to landscape-scale restoration in Dorset, England. CBA requires a counterfactual situation (which represents a control or *status quo* condition) and one or more alternative scenarios which represent the project being evaluated in comparison to the *status quo*. This is often referred to as a comparison of 'with the project' to 'without the project' (Boardman et al., 2018). In the CBA approach, the 'willingness to pay' for a proposed restoration scenario represents the benefit and the cost is the 'opportunity cost', which is an estimate of the economic value of the 'resources' invested in the project in their alternative use, which can be the land, the required labour, and the invested capital. For example, Newton et al. (2012) considered three restoration projects in their Dorset study, which had varying degrees of restoration coverage across the landscape and a pre-project baseline condition as their counterfactual situation.

It must be said that, in this type of analysis, valuing costs is often easier than quantifying benefits. The opportunity cost of labour, materials, and land, for example, are readily available from market prices, although technically speaking these prices should be adjusted to remove market distortions in order to derive the true social opportunity cost of the resources. The quantity of labour, materials, and land required for restoration projects is also often known. Benefits are more difficult to derive because these often involve 'public goods'; that is, goods that are not traded in markets because they are 'non-rival' and 'non-excludable'. For example, people may be willing to pay for the aesthetic beauty of a restored landscape, but they do not express that willingness to pay in a normal market situation. This is because people cannot be excluded from appreciating aesthetic beauty (non-excludability), and one person's appreciation does not reduce the benefit for others (non-rivalry). To address this issue, economists have derived non-market valuation principles (see Iftekhar et al., 2017), but in the project described by Newton et al. (2012), the authors focussed on benefits that can be readily estimated. For example, whilst they valued carbon sequestration, timber, crops, and livestock production, which all have market prices, these prices do not reflect the underlying social cost of carbon, or timber, due to market distortions. Newton et al. (2012) supplemented these monetised benefits with non-monetised cultural, aesthetic, and recreational values, estimates which were elicited from stakeholder surveys.

For a carbon-focussed project, such as the one described in Newton et al. (2012), clearly the larger the restoration, the greater is the amount of carbon that is sequestered. At the same time, there should also be more cultural, aesthetic, and recreational benefits as the scale of the restoration project increases. However, only the loss of carbon sequestration functions is valued for CBA purposes, and the restoration projects considered in the study reduce timber availability, together with crop and livestock production on the land involved in the project. Seen in this light, there is little additional benefit arising from the restoration projects in the study compared to the status quo, even though it is known that restoration projects are very costly. This led Newton et al. (2012) to conclude that 'ecological restoration is unlikely to deliver net economic benefits in landscapes dominated by agricultural land use' (p. 571), which highlights a fundamental and common problem for restoration activities. The authors have not estimated all the benefits arising from their restoration project and thus a general CBA conclusion cannot be drawn. In most cases, however, it will be more difficult to justify restoration projects that occur on agricultural land because the opportunity cost of the land is higher, which highlights the need to estimate all the benefits of restoration.

A further issue is that estimating non-monetised aesthetic and recreational benefits for a project alongside quantified market values provide little practical insight. Although cost-benefit analysis guidelines around the world certainly highlight the need to include qualitative or non-tangible benefits (The Treasury, 2017: 17; HM Treasury, 2020: 24), these are rarely taken into account when political decisions are made. The focus for most decision-making is on the bottom line, which contains the monetised net benefits. As such, for preparing contested proposals, it is imperative to calculate or estimate all benefits, unless there are clear net benefits from the easily accessible monetised benefits of restoration. Of course, estimating recreational and aesthetic values or biodiversity value is not easy, but there are standard estimates available in the literature. However, these are all fundamentally place-specific and suggested methods that translate benefits from one location to another, which is known as the benefit transfer method (Iftekhar et al., 2017). However, this approach can be criticised when political power is at play.

Moving on from Newton et al.'s (2012) work, a second application of economic enquiry in this case study focusses on cost-effectiveness analysis. Kimball et al. (2015) used an experimental approach to assess the cost-effectiveness of different restoration methods over a 25-hectare site in Southern California which involved 87 individual plots, with restoration methods varying on the extent and type of site preparation, seeding, planting, and maintenance. The authors calculated the specific costs involved, controlled for land aspect, slope, and initial conditions, and determined the methods that produce the greatest percentage of native plant cover per dollar of cost. It should be noted that the most ecologically effective method in terms of native plant cover can also be the most costly approach. Therefore, the authors concluded that 'the most successful method for restoring high native plant cover is often different from the method that results in the largest area restored per dollar expended, given fixed mitigation budgets' (Kimball et al., 2015: 1). Thus, by applying economic decision-making criteria when confronted with a limited budget, it was shown that greater native plant cover can be achieved, in aggregate, if the budget is allocated to the most cost-effective method.

Kimball et al. (2015) stressed that their specific results in terms of the most cost-effective method are dependent on a number of factors. For example, environmental variation strongly influences the results of restoration. While the authors could control for site characteristics within their experiment, they were unable to recommend specific strategies for other locations. Restoration success is clearly very context-dependent, but the researchers did generalise by stating that the money spent on seeding, planting, and maintenance was cost-effective in comparison with site preparation, although some caveats here are also site-specific.

One of the key challenges that arises from the research on CBA and CEA is that there are no standard measures of success from which comparable benefits can be calculated. Newton et al. (2012) used a narrow range of ecosystem services whilst Kimball et al. (2015) used the percent of native plant cover, which is easy to measure but may not account for the diversity of the native plant assemblages. In other work, there are a number of success measures (Wortley et al., 2013). A broad range of ecosystem services have been calculated by Birch et al. (2010), whilst other authors use the willingness to pay for restoration (Iftekhar et al., 2017), and some have used the market price for biodiversity offsets or other measures of replacement cost as a measure of restoration benefit (BBOP, 2009). It is evident, from this variety of measures, that there is a clear need for a standardised measure of restoration benefits that can be priced or valued.

One possibility for a standardised measure of restoration benefits is the one developed by the Australian state of New South Wales (NSW) in their Biodiversity Offset Scheme (Department of Planning and Environment, 2021), with the metric referred to as the Biodiversity Assessment Method (BAM) (Department of Planning, Industry and Environment, 2020). While controversial, the BAM metric of ecological condition and improvement under alternative management regimes could be used to compare benefits across sites and broader locations. Indeed, the BAM is used in this way to make the biodiversity offset market work. Of course, the biodiversity offset price should not be used to directly monetise the benefits of restoration. Although this occurs in NSW when assessing the cost-benefit analysis of new restoration projects, the price has no real relation to the marginal social benefit from restoration. Instead, the offset price is determined by government requirements to buy offsets and the opportunity cost at offset sites, which is based on the value of land at those sites. However, the ecological metric (the BAM) underlying the creation of biodiversity credits could be used as a standard for many types of ecosystem restoration projects in different jurisdictions. From this position, the cost-effectiveness of different restoration projects could be estimated and funds could be allocated according to the restoration credits created per dollar of costs.

A second challenge is to include the opportunity costs in each situation. Many studies include only the costs of labour and materials to seed, plant, maintain, and prepare sites (Wainaina et al., 2020). However, from an economic perspective, the opportunity cost of the land upon which the project is based is conceptually the most important cost, because the restoration project removes the option of doing something else on the land. This creates a land-use conflict and stimulates political objections to the restoration projects, so it is best to address the opportunity costs upfront.

Proving that the economic benefits of a restoration project outweigh the opportunity costs is both difficult and expensive. There is a continual need to value all types of ecosystem services created, and assess biodiversity values, recreational values, and aesthetic values. As mentioned earlier, this process acts to commodify nature by bringing nature into the economic realm, which is a questionable strategy for ecologists (Perry & Primrose, 2015). An alternative tactic is to initially recognise that this calculative approach is based on a particular, non-universal ethic that is embedded in economics, allowing us to alternatively highlight our stewardship role over the environment, the rights of nature, and the precautionary principle required to maintain diversity (Convention on biological diversity, 1992). In essence, it is recognised that trying to achieve value for money using cost-effectiveness analysis speaks sufficiently to the language of economics without monetising nature, while stewardship and the precautionary principle apply a more deontological ethic, which is more appropriate for the protection of nature (Perry & Primrose, 2015).

Case Study 2: Economic Impact Analysis

When an infrastructure project such as a new coalmine or industrial development is proposed, the proponent will argue that it creates jobs and has a positive economic impact, such as its contribution to Gross Regional Product. This approach provides a powerful narrative, even when the development is liable to destroy the natural environment. This jobs-versus-environment tension is played out in all countries around the globe and is commonly used to justify environmentally damaging policy and projects. This rhetoric is so embedded in developmental arguments that jobs are often mistaken for benefits in a CBA, even though employment is a cost of the project (Boardman et al., 2018: 19). While this is an error, there is no denying that arguments around job creation are powerful and ecologists proposing restoration projects could create their own narrative by explaining the employment and output impact of the restoration economy. Following BenDor et al. (2015a, b) and my own work in analysing the transition from carbon-intensive production (Perry & Hewitson, 2019), this case study of research into the economics of ecological restoration explains how to confront the jobs-versus-environment rhetoric, and also suggests how to estimate the economic impact of restoration projects.

As explained in BenDor et al. (2015a), there is a difference between the economic value of a restoration project and its economic impact. In a cost-benefit analysis, the analyst assesses the willingness to pay an amount for a project when determining the benefits, then deducts opportunity costs in order to calculate the net benefit, which is the value of the project. In contrast, the economic impact is usually measured solely by the jobs and output created. Mining proposals and other development proposals routinely measure the economic impact and include it as an argument for the proposal. In some Australian jurisdictions (for example, NSW Government, 2015), an economic impact statement is a regulatory requirement in addition to a cost-benefit analysis. A restoration project provides a similar economic stimulus with an upfront or short-term increase in employment and long-term permanent employment. This impact can be analysed using various economic techniques but the simplest and most readily available technique is input-output analysis.

The statistical agencies of many countries around the world publish input-output tables, which map the interdependencies of industries (Miller & Blair, 2009). For example, the agricultural services sector provides inputs for agriculture which in turn provides goods for the retail and wholesale sector, food manufacturing, and for final consumption. The input-output tables can be used to model the direct effect of an investment or sales increase in one sector on the interdependent sectors and the changes in employment, wages, and salaries. In turn, the increases in wages and salaries are spent on goods in the economy (as well as imports), which creates further output and supply chain dependencies.

Using the job creation figures associated with ecological restoration and the spending within specific industries, an economic analyst can determine the total output and employment gains from an ecological restoration project or the ecological restoration economy as a whole. For example, BenDor et al. (2015b) surveyed businesses in the United States engaged in restoration activities and has estimated that 126,000 permanent jobs were directly created. The direct economic impact of this was estimated to be a US$ 9.5 billion in output using input-output tables, whilst an impact analysis also revealed that an additional 95,000 jobs and US$ 15 billion in output is created indirectly due to the supply-chain effects and induced expenditure of those workers.

While this is an aggregate analysis for the economy as a whole, the economic impact of individual restoration projects or restoration in regions can also be determined. This simply requires an estimate of either the quantity of paid labour or the total expenditure needed for restoration in a region, which is usually known by ecological restoration practitioners together with the input-output tables for the region. Input-output tables for an economy can be modified for individual regions using various techniques (Miller & Blaire, 2009, Chap. 3) and suppliers of statistical packages often do this work for the analyst at a relatively small cost. For example, armed with the total required expenditure on rehabilitation after mining cessation, Perry and Hewitson (2019) estimated the direct economic impact of the rehabilitation on jobs, wages and salaries and output in a small, carbon-intensive region. Along with other known expenditures on green industries, these jobs partly offset the loss of jobs that will arise if nations around the world meet their carbon emission reduction targets and coal prices fall.

These examples demonstrate the viability of estimating the positive economic impact of restoration projects, which goes some way towards arresting the jobs-versus-environment rhetoric. However, there are always difficulties with this work and standardisations are needed. First, to perform an economic impact analysis, the analyst needs either total expenditures on restoration or the number of jobs created. While this should be relatively easily obtained for individual projects, it becomes difficult when looking at aggregate effects across regions, countries and the world. In particular, restoration work arises from multiple different government agencies, the private sector, and from non-profit organisations (BenDor et al., 2015a). Second, the economic impact of the restoration work will vary depending on the industrial classification for which this work is applied. Countries use different and broad industrial classification schemes, meaning that restoration work, like the green economy more generally, does not fall neatly into these classification boundaries. In Perry and Hewitson (2019) for example, Australian Industrial Classifications and percentages were used for mining remediation work as shown in Table 15.1. To provide a comparison, Table 15.2 shows the US industrial classifications used by BenDor et al. (2015a).

The differences in total output can be significant when different industries are assessed to be central to restoration work and standardisation is required. However, there is considerable potential for input-output analysis or other economic impact

Table 15.1 Remediation investment (Australia)

Industry sector	% of rehabilitation investment
Heavy and civil engineering construction	50
Road transport	5
Professional, scientific and technical services	10
Auxiliary finance and insurance services	10
Exploration and mining support services	25

Source: Perry and Hewitson (2019)

Table 15.2 The top 15 industries in the restoration economy (United States)

Industry sector	% of total
Architectural, engineering, and related services	36.4
Support activities for crop production	23.3
Other heavy and civil engineering construction	10.1
Administration of Environmental Quality Programs	7.6
Public administration	4.6
Professional, scientific, and technical services	3.0
Management, scientific, and technical consulting services	2.4
Highway, street, and bridge construction	2.2
Support activities for road transportation	1.5
Fishing	1.3
Other professional, scientific, and technical services	1.2
Other specialty trade contractors	1.1
Real estate	0.5
Construction	0.5
Support activities for forestry	0.2
Other industries	3.9

Source: BenDor et al. (2015a)

methods, such as computable general equilibrium analysis, to be developed for justifying restoration projects. Tools can be developed to allow restorers to enter the restoration investment or number of jobs and hence uncover the economic impact of their work. This creates a powerful narrative to confront the jobs-versus-environment rhetoric.

Case Study 3: Economic Benefits of On-farm Ecological Restoration

Along with forestry, industrial development, and residential expansion, agricultural production has also been responsible for degrading natural systems. Consequently, there are multiple opportunities for restoration. In this respect, in agriculture, taking an economic perspective can assist in justifying substantial restoration work and for articulating the value of restoration for agricultural production at both local and regional scales. This case study of research into the economics of ecological restoration theorises restoration in agricultural settings, and examines experimental work in Southern California that provides evidence of a positive return on investment from such activities. I also discuss lessons from this work, reflecting on the inhibitors that must be overcome in order to see farming communities embrace ecological restoration for the health of their agricultural landscapes.

As is typical of an orthodox economics perspective, the justification for ecological restoration on farmland depends solely on the costs and benefits. The private benefits of restoring biodiversity could include pest resistance and pollination, shade, and reduction in temperatures, together with protection against erosion, drought and floods. Social benefits that extend beyond the landowner's property could include species preservation, flood protection, erosion control, drought protection and evapotranspiration from vegetation. The costs are clearly opportunity costs that result from the reduction in agricultural production on restored land and the costs of creating and maintaining the restored ecosystems.

At a more detailed level, however, restoration can be modelled as a type of self-insurance that substitutes for crop insurance. Biodiversity hedgerows and patches, as well as investments that restore soil biodiversity and other agro-ecological techniques, can reduce the variance of crop returns and increase yields. Baumgärtner and Quaas (2010) investigate the theoretical implications of an insurance value for biodiversity. However, as biodiversity on a property provides benefits to neighbouring properties, the variance and mean returns on a farm depends on both biodiversity on the farm and biodiversity in the region. For example, a more vegetated region will experience more evapotranspiration, given existing conditions, suggesting that it will have less overall erosion and more drought and flood protection.

A confounding issue arises, however, since biodiversity produces both private and social benefits. Therefore, an individual's decision to restore biodiversity on an agricultural property will be less than what is optimal from a social perspective because the decision will be made on the basis of the private returns from biodiversity (increased yield and reduced variance) and neglect the social benefits for other properties and other members of society (including people in cities who value biodiversity and species preservation). From a theoretical perspective, the solution to this problem is to create a Pigouvian subsidy for landowners to restore biodiversity to the socially optimal level (Baumol & Oates, 1988). Examples of Pigouvian subsidies include the Conservation Reserve Program in the United States and newly created environmental markets, which aim to protect biodiversity and sequester carbon by providing farmers with a payment for restoration action. However, it is often the case that such subsidies do not exist, are hard to access, or are subject to significant information deficiencies such as the impact of restored land on yields and land values. In addition, many existing farm subsidies actually encourage land clearing.

At the level of private benefits and costs, this theoretical model relies on there being a relationship between either the agricultural yield or the variance of yield and biodiversity, noting that such evidence does exist. The literature on biodiversity and ecosystem functioning supports the positive relationship between biodiversity and mean yield (see Hooper et al., 2005). More specifically, evidence from California suggests that hedgerows of biodiversity increase the numbers of beneficial insects which assist in pest control and pollination services and directly increase profits by reducing the cost of insecticides as well as increasing yields in adjacent crops (Long et al., 2017; Morandin et al., 2016). These studies were carried out with an experimental design incorporating control paddocks on the same farm and with the same crop, which allowed the impact of hedgerows to be monitored over several years. The return on investment for hedgerows was estimated to be 16 years for farms not requiring additional pollination services and 7 years for farms where pollination services were needed. In addition, overall food security was not negatively affected by the addition of hedgerows.

While the theory and evidence for the positive effects of this system are encouraging, information problems and other market failures persist. For private individuals to make rational decisions regarding the amount of biodiversity on their properties, they require a considerable amount of information about the impact of biodiversity on the mean and variance of yields. Unfortunately, this information is not readily available in most locations since the costs and benefits are location specific and depend on the type of agricultural production. It should also be remembered that there are many different types of restoration activities. Whilst some farmers might plant hedgerows and maintain or protect large trees or pockets of remnant vegetation or grasslands on their properties, others may recreate wetlands or practice some form of permaculture or biodynamic farming. Each of these types of biodiversity would have different consequences for the mean and variance of yield. Farmers also require knowledge of the amount of biodiversity in their region as the return on restoration on their own properties may diminish as the restored areas in their region grows. All this suggests the need for a large-scale, government-funded research program and publicly available information services. Governments need to correct the market failure caused by information deficiencies by funding experiments to inform farmers of the benefits of restoration in different types of locations and for different farming activities.

Nevertheless, even with the availability of complete information, the overriding problem is that the social benefits from biodiversity restoration are large, and this means that private biodiversity restoration levels are inefficiently low. Farmers have an incentive to wait for others to create biodiversity, which is an example of a 'free rider' problem. High discount rates are also an implicit impediment to biodiversity creation, since the return on investment for farmers reduces as private discount rates increase. Whilst Pigouvian subsidies partly correct this additional market failure, systemic debt issues in agriculture increase private discount rates and mitigate against optimal biodiversity at the farm level, which again highlights the need for government policy to correct the market failure.

Case Study 4: Large-Scale Restoration Projects

At the macroeconomic scale, large-scale restoration projects are needed to both restore natural capital to its optimal level (Figueroa, 2012; Farley & Gaddis, 2012) and to meet international obligations under ratified agreements (Murcia et al., 2016). Natural capital has become the limiting factor in society's push for improved well-being and is therefore more important than human-made capital (Costanza et al., 1997). However, restoring ecological systems in developing and developed countries creates a land-use conflict, and thus the socio-economic and political economy issues that surround restoration projects will inhibit success unless confronted and addressed. For example, a top-down restoration policy can be too blunt and not context specific or geographically relevant. An outcome of this non-specificity is that locals charged with enacting the policy, and whose lives are directly affected by the changed circumstances of land use can react negatively to such top-down requirements. It is suggested that cooperative approaches are needed because these will lead to changes in the value framework used by society, where a value framework refers to the basis upon which decisions are evaluated and justified. For example, standard value frameworks in neoliberal, capitalist economies include growth, jobs, profits, economic efficiency, and cost-benefit analysis. However, alternative value frameworks can be introduced, based on sustainable development goals or planetary health, which may be needed to justify large-scale restoration projects. Thus, a transition process is required to engage locals and define the appropriate value framework for that region and to drive a cooperative approach to ecological restoration. This case study of research analyses a cooperative approach for large-scale restoration projects.

As indicated in the introduction, international agreements such as Aichi Target 15, milestone A1 of the post-2020 Global Biodiversity Framework, the Bonn Challenge, and the UN Decade on Ecosystem Restoration require increased restoration of natural ecosystems. Countries can try to achieve these restoration targets in a variety of ways but many have been using top-down, state-led approaches with complex legal and regulatory instruments (Pinto et al., 2014: 2213; Murcia et al., 2016: 215). Indeed, Pinto et al. (2014) argue these top-down approaches have been ineffective because of excessive bureaucracy, a focus on regulatory punishments, and a lack of enforcement and incentives (Pinto et al., 2014: 2213). In particular, top-down approaches lack participation due to poor governance approaches and a lack of shared values.

The problem here is the classic 'free-rider' issue. As mentioned earlier, restoration is a public good, the benefits of which are non-excludable and non-rival, and, as such, there is a disincentive to act alone. In this regard, while creating incentives can work in theory, multiple examples of market failures mean that incentives will not always be successful. In particular, restoration can involve market failures such as a lack of information, a lack of enforcement opportunities, as well as additional external costs and benefits beyond the specific restoration goal itself. In addition, there are institutional issues related to the cultural values and norms of the societies where restoration is to occur. These market failures create practical impediments to the theoretical solution of creating markets and incentives for restoration.

As an alternative, Pinto et al. (2014) report on the Atlantic Forest Restoration Pact (AFRP) in Brazil which pursued a different agenda based on cooperation. The cooperative solution resonates with the work of Nobel Prize–winning economist Elenor Ostrom who investigated when Commons, traditionally maligned in economics, can be successful. Examples of Commons, or common-pool resources, include irrigation systems and pastoral resources owned by nobody but maintained by everyone for the benefit of all. The eight design principles relevant to this system are as follows (Ostrom, 1990):

- Commons need well-defined boundaries which are fundamentally related to property rights. In the context of restoration, this rule relates to the right to harvest ecosystem services from restored ecosystems, particularly for provisioning ecosystem services such as food, water, and wood for fuel.
- Proportional equivalence between costs and benefits is required. Those who can harvest provisioning services and those who enjoy the benefit of other ecosystem services are also the people who provide the inputs to restoration. As such, rules relating to the benefits and costs must be context dependent and geographically relevant.
- Collective choice determines the rules, and the individuals affected by the harvesting and restoration rules are the ones who make the rules.
- Commons must be monitored, and again the people doing the monitoring should be the people affected by the rules.
- Graduated sanctions. The rules of successful commons suggest that initial punishments can simply be notices of infraction without penalty. This tends to ensure trust because it highlights to the individuals that others will also be noticed if they break the rules. Continued infractions can lead to higher-order punishments.
- Conflict resolution mechanisms. Users have access to rapid, low-cost, local conflict arenas to ensure that grievances can easily be heard and resolved.
- The right to organise and define their institutions, where rules will not be challenged by external governments and long-term tenure rights will be observed.

- Nested enterprises are needed for large-scale commons. Where commons are large, such as the AFRP, hierarchically nested organisations are needed. This again allows local differences in rules, whilst recognising the constraints imposed by the larger organising body.

The AFRP is based on the principle that the probability of success of a project is increased through cooperation. Cooperation will improve public policy, provide effective financial incentives for restoration, discourage degradation, lead to the development of appropriate legal instruments for restoration programs, and establish good governance for restoration (Pinto et al., 2014: 2217). As such, groups in the AFRP came together to form a coalition with shared values that were determined by the stakeholders. Sequential targets were created as a result of this coalition in order to restore 15 million hectares of degraded landscape and to prioritise the recovery of pastureland that was relatively poor in agricultural productivity. The targets were determined under the principle of cost-effectiveness, and the objectives of the AFRP include the generation of socio-economic benefits, jobs, and income, all of which attempt to reduce land-use conflict (Pinto et al., 2014: 2219–20). As a result of the success of these actions, the AFRP has effectively become a countervailing power (Galbraith, 1952) against agribusiness lobby groups which is needed to ensure a balance amongst competing political aims (Perry, 2013).

To do all this, the AFRP adopted seven governance structures and instruments:

- An easy-to-follow membership process that ensures members agree with the objectives of the AFRP and specific aspects of the restoration technologies and monitoring protocols used.
- A coordination council to develop the strategic plan and vision of the coalition, and to define goals, standards, rules, and principles. Importantly, the coordinating council is comprised of member institutions.
- A paid executive secretary who oversees the work of the council as well as training and communications.
- Decentralised regional units which have autonomy to strategically coordinate and manage their activities.
- Working groups that ensure cooperation and participation amongst the various institutions and ensure an alignment with the common goal. As such, working groups cover technical-scientific methods, socio-economic outcomes, and communication with others.
- Training and capacity building to disseminate knowledge and align goals.
- A monitoring protocol using a participatory approach of partner institutions.

While not a classic Common like those studied by Ostrom (1990), the AFRP is a demonstration of bottom-up cooperative management. Shared values created the goals of the Pact, members have an interest in the outcome, and members create the rules that they also oversee and monitor. The shared values and rules are location- and context-dependent and the institution is community recognised and therefore not undermined by external governments. The organisations are nested as they need to be for such a large-scale Common, and those who benefit from the Common are aligned with the work they do. The AFRP demonstrates that there are viable alternatives to the traditional economic policy approaches, which are created from a theory that assumes a different ontological arrangement of society to those that currently exist.

Case Study 5: Ecological Restoration Policy

The majority of recent movement on the policy front at a more micro level of analysis has been through the creation of markets for biodiversity. Like a Pigovian subsidy, an offset market can create an incentive to restore biodiversity through payments to landowners (and at the same time, disincentives to clear land). While this is theoretically a perfect solution, in reality the ontology of economics does not match the ontology of society or ecological systems and markets are far from a panacea. This means that other mechanisms need to be explored that address the root cause of the problem, which is generally seen to be the value framework facing actors in modern neoliberal economies.

Biodiversity markets have been increasing internationally over the last 20 years. Madsen et al. (2010) found 39 existing mitigation compensatory schemes around the world and another 25 in development with a global market value of US$ 1.8–2.9 billion. To inform policy in other countries, this case study focusses on the experience of the Australian state of New South Wales (NSW) which has one of the most sophisticated biodiversity markets in the world. The biodiversity offset scheme (Department of Planning and Environment, 2021) has a buyer and seller of last resort, exchange ratios between different plant community types, an online biodiversity credit calculator that factors in past trades to predict prices, and outsourced ecological consultants who apply the biodiversity assessment method (Department of Planning, Industry and Environment, 2020) to determine biodiversity credit liabilities for developers and credit assets for landowners.

The idea of a biodiversity market is firmly grounded in economic theory. When biodiversity is destroyed due to land clearing, new mining projects, or residential and commercial development, there is a marginal external cost (MEC) to members of society who value biodiversity. In this respect, the MEC is taken to be equal to society's willingness to pay to avoid biodiversity loss. Whilst a Pigovian tax (Baumol & Oates, 1988) would be one way to correct this market failure, with the tax being set equal to the MEC, economists in government favour market solutions due to the political difficulties of imposing new taxes, and the theoretical market price of biodiversity credits is equivalent to the Pigovian tax and the MEC.

In the NSW Biodiversity Offset Market, the target is 'no net loss' to biodiversity. Every loss must be matched by an equal gain through ecological restoration that produces a biodiversity or ecosystem offset. By creating the market, the developers pay a tax and landowners receive payments equal to the marginal external benefit of biodiversity created. In addition, the market works to ensure that the least productive land comes online as offsets first, holding everything else equal. Thus, the market clearly applies the principle of cost-effectiveness.

Whilst this is the theoretical position, it is very different in reality. As mentioned previously, the ontology of economics is one of the atomistic and independent agents. Thus, the neat picture presented of a patch of biodiversity destroyed and a nearby patch created is accurate within this theory. However, in reality, nature is interdependent and a destroyed area can never be perfectly replaced. Every patch of biodiversity is unique, and this reality of uniqueness creates difficulties for a market because there can only ever be one buyer or one seller of a patch of biodiversity. A credit in one area can never be precisely matched elsewhere. But a market does not ever work with one buyer or one seller, and thus to make the market work, patches of biodiversity must be made to be tradeable with each other. This can be achieved, for example, through the Biodiversity Assessment Methodology in NSW and the creation of exchange ratios between different plant community types. In addition, a created market intermediary, the Biodiversity Conservation Trust, is used to fill the gaps in the market and to ensure that those wishing to purchase credits are able to do so.

Other market failures also mitigate against the creation of a well-functioning market. Landowners often need to pay the upfront cost of creating credits with little knowledge of when they will be sold or the price of those sales. Landowners also have lack of information about what they are giving up by engaging in restoration. In particular, they do not understand the long-term impact on their land values, suggesting that landowners cannot make informed, rational decisions. However, such a market failure can be corrected through government policy and research. For example, the upfront cost of creating credits could be payable only when offsets are sold, with the government insuring the upfront cost beforehand.

An important ecological issue is also created in the NSW Biodiversity Offset Market by a political decision that makes it possible to create offsets simply by protecting existing biodiversity, rather than creating new biodiversity. Credits always have some level of restoration, such as through the enhancement of conserved sites (through planting or seeding) or by invasive species control, but the fact that credits can be earned for biodiversity protection embeds biodiversity loss into the system (Maron et al., 2012) in contrast to the 'no net loss' requirement. There are also losses to biodiversity when development occurs, and biodiversity is only restored at offset sites some years later. This time delay is built into the system when developers can offset biodiversity destruction by paying into the Biodiversity Conservation Trust with the Trust, then seeking options for offset sites.

As a restoration policy, biodiversity markets are currently ineffective. They embed biodiversity destruction into the system and while landowners do earn income for their restoration work, they are not paid the marginal external benefit of their work. The supply of credits that simply reflect protection rather than restoration drives the price of credits down and below the marginal external benefit of restoration work. The credit price is also driven by land prices or the opportunity cost of the land, rather than any willingness to pay for restoration. As such, the incentives in the system are for low-valued agricultural land to be the source of offsets which may or may not have any relevance to where biodiversity restoration should occur for the aims of representativeness, persistence (Margules & Pressey, 2000), area connectivity, or any other ecological requirement.

Alternatives to biodiversity markets exist. Restoration priorities could be strategically identified in regions using ecological and conservation planning knowledge with landowners compensated for the restoration work on their properties at a price that is equal to the marginal external benefit and marginal opportunity cost of their work. In this respect, the Conservation Reserve Program in the United States may be a better model because the government effectively rents land for a set period of time, which creates more certainty for landowners, more demand for restoration, and a more fully developed restoration supply market which reduces costs (see Gibson-Roy, 2018; Gibson-Roy et al., 2021a, b). However, such a policy still commodifies nature which is a fundamental institutional issue that mitigates against restoration. Conventions and norms in farming communities and the mining and construction sectors reflect the value framework used in society and in the economy. This common value framework in a modern, neoliberal economy relates to growth, profits, and efficiency rather than a respect for nature and a knowledge of the human health and ecological benefits of restoration. Change will come with cooperative approaches more akin with the large-scale restoration project in Brazil mentioned in the previous case study, which involves a way to change values and norms in society. However, this shift in values is not easy to achieve, and it will not occur while biodiversity is simply commodified within biodiversity markets.

Chapter Synthesis

From the perspective of ecological economics, natural capital and human-made capital are complementary (Costanza et al., 1997). The degradation of natural capital directly reduces human wellbeing and it also reduces the degree to which other forms of capital can operate effectively. There are also critical limits or thresholds for natural capital which makes the relationship between degradation and wellbeing non-linear. As described by Steffen et al. (2018), ecological systems can flip to a new state when critical limits are reached, affecting the wellbeing of humans and animals in more general ways. Under these insights, and given the extent of global biodiversity loss, ecological restoration is required as an investment in natural capital for human wellbeing.

In contrast, from an orthodox economics perspective, natural and human-made capital are substitutes. Theoretically, this means that human wellbeing is not dependent on natural capital because new human-made capital (such as new technology) can substitute for the loss in natural capital. However, even within this orthodox economics framework, it can be noted that humans value the environment and ecological restoration can be justified although government involvement. First, the public good element of ecological restoration means that social benefits are normally greater than private benefits. Thus, left to the market for direction, restoration will be suboptimal and there is a role for government to step in and provide, in theory, Pigovian subsidies or other forms of compensation for undertaking restoration. Alternatively, the government may decide to directly provide restoration. Second, complicating this area is that information on the private and public benefits of restoration is generally lacking and restoration can be difficult to justify for both private actors and the public sector. Thus, government-funded research is needed on the public and private benefits of restoration work.

In addition, institutional economics emphasises issues related to norms and conventions in society which mitigate against restoration. Here, commodifying nature is a norm in itself and follows the value system in neoliberal economies, which is focussed on growth, markets, and profits. Following this approach, action to restore natural systems requires a change in these norms or to this value system. Values, norms, and conventions can change with policy, or due to environmental crisis, but values will not change with policy that reinforces the commodification of nature, such as environmental markets. As such, direct government provision of restoration and cooperative solutions to restoring landscapes at catchment scales are needed, along with secure and certain compensation for landowners for their restoration work as part of the cooperative solution.

Implications

- Research on cost-benefit analysis and cost-effectiveness analysis is needed so that ecological restoration practitioners can speak the language of economics whilst also arguing for restoration from a different underlying ethic.
- The research agenda to estimate the size of the ecological restoration economy can be enhanced so that ready-made tools are available to calculate the economic impact of restoration projects and counter the jobs-versus-environment rhetoric.
- Niche research on the economic benefits of restoration for farm productivity is valuable but it is context-dependent. More funds need to be allocated to this work to bridge information deficiencies and change institutional norms in farming communities.
- On the macro-policy front, a top-down policy will not change underlying values and cooperative solutions may be preferable.
- On a more micro scale, landowners need to be compensated for restoration work including the opportunity cost of their land but cooperative solutions are needed to change values.

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Part V
Final Synthesis

Chapter 16
A Final Word from the Editors

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We began this publication with the belief that there are many valuable lessons to be learnt about ecological restoration from the work of academic and practitioner communities who are deeply involved with restoration activities, and it was our desire to draw together some of these approaches and related Case Studies. In this regard, the editors have been greatly encouraged and pleased by the responses of the many contributors to this project who have come forward to share their experiences for the benefit of others. As you the reader will be now aware, this book has drawn together many high-profile researchers and restoration practitioners from across the world, all sharing a deep commitment to restoring specific aspects of nature, and demonstrating a willingness to communicate their experiences of so doing.

The rationale for developing this collection of lessons was to systematically document some of the many journeys that have been taken toward long-term restoration action to begin repairing the significant amount of human-mediated damage to our natural world. We believe that by compiling these accounts, in the words of the best specialists available in these areas, we can assist in creating tangible, multi-focused pathways toward conserving, enhancing, or reconstructing plant communities and the habitats that these create. In this way, we can assist in the systematic return of the many forms of nature that are so critical and deeply rooted to the psychological and physical well-being of humans, societies, and the ecosphere.

Each chapter has been structured to present foundational information as well as insights and learnings that focus on specific aspects of ecological restoration. These perspectives are critical in developing (i) our essential theoretical understanding of the key elements of restoration programs, (ii) the ways in which these understandings can be used to underpin and inform practical restoration undertakings, and (iii) balanced strategies which deploy available resources to their optimum extent and which are designed to achieving long-term solutions. For each of these chapters, the authors provide underpinning information for the reader about key issues related to their specific topics, describing and discussing the principles and approaches that have been taken to address a suite of interdependent restoration issues. Many chapters have, as their backbone, Case Studies selected by the authors to illustrate concrete action(s) that have been undertaken to repair damaged areas of nature in recent times. The structure and content of the chapters have been developed to provide the reader with clear insights into the thinking of those involved with each case study, shedding light on their motivations, their rationales for action, the approaches taken, and finally, their setbacks and successes. Critically, because the Case Studies shed light on experiences from different parts of the world, they highlight the universal importance of assuring strong levels of commonality of action within a project, while, at the same time, describing the differences in issues that require immediate attention, which may be mediated by geographical location, physical circumstance, and community requirements in specialized localities. These Case Studies have been included to provide the reader with clear descriptions of the actions and approaches which were necessarily undertaken to meet specific restoration goals, scenarios, and requirements. Importantly, the authors also discuss those things that worked and those that did not, and they have examined the reasons why such outcomes may have eventuated. In providing this focus, the authors have afforded the basis for synthesizing comments on "where we are at" and "how now to go forward." These comments are not only relevant to individual chapters, but can be used to allow the reader to determine their own response to their own specific restoration projects. The editors believe that the real value of these insights is that they go beyond the often-clinical descriptions of primarily successful restoration activities portrayed in scientific papers or indeed those which are used to promote grand goals of governments and their agencies, to instead providing clues and suggestions to novel approaches which might be taken in emerging contexts.

A Time for Reflection

As the editors now collect their final thoughts on this publication, we are forced to reflect that, across the planet, there are increasing numbers of seriously degraded ecosystems in urgent need of repair. This consideration leads many to ask the difficult question, "*Why do we want to bother restoring specific areas natural landscapes when the global restoration task is so daunting?*". Others may further ask, "*Can we even repair the damage that has already been done in a restricted area?*" Encouragingly, an increasing number of people are now committing to this task of restoration. It is now widely recognized that humankind has, for a wide range of reasons, had negative impacts on the welfare of other species and ecosystems, often as a result of fundamental errors of mismanagement and misunderstanding. As a consequence, we need to avoid ignoring our moral obligation to the planet, by recognizing ourselves as part of the global ecosystem. Indeed, the current human-created crisis not only jeopardizes the fate of countless species, but we now see that it threatens our own existence. That said, regardless of what motivations actually drive ecological restoration, what really matters is that ecosystem repair takes place.

Perhaps our greatest challenge is to fully understand and reconcile the paradox that the current qualities of nature, because of human activity, have been irrevocably altered. Indeed, for as long as humans remain the dominating global force, nature will always display a character that reflects a human-affected form. And so, while efforts to restore and preserve nature are well intentioned, it is also important to realize that, in most cases, restoration can only address some proportion of the problems of disturbed natural elements. This consequent diminution of the original complexity and diversity of delicate ecospheres will inevitably affect their overall function and trajectory. This dilemma becomes especially obvious in the context of ecological restoration when viewed in the light of questions such as: How do we compensate for the loss of predators or ecosystem engineers? For restored systems to function, it is clear that it falls upon humans to do as best they can, to fill those roles. These responsibilities, concepts, and questions obviously create a great deal of disquiet among restoration ecologists. For example, mimicking the role of extinct apex predators implies that humans must cull over-large populations of previously predated species, a role that will inevitably clash with concepts of ethics and animal welfare. Likewise, mimicking the effects of extinct large herbivores will necessitate the management of significant amounts of plant biomass through fire or mechanical methods, which both have important secondary issues regarding effectiveness and undesirable outcomes. Humans must face the reality of this altered state of nature in their quest to repair and maintain ecosystem function and trajectory. Thus, the true cost to manage restored, and indeed conserved, landscapes, while yet to be fully determined, is certain to be large and ongoing.

The Value of Ecological Restoration

Ecological restoration encompasses the philosophy and practice of repairing degraded, damaged, or altered ecosystems. The need for this discipline has arisen because, for millennia, nature has been directly and indirectly negatively affected by humans. Early humans knowingly or unknowingly impacted global species diversity through overhunting or by their use of fire to alter landscape compositions. In more recent times, mega-human populations and technologies have exacerbated these problems. Across many areas of the globe, landscapes have been totally cleared for agriculture, forestry, cities, and towns. Humans have accelerated the spread of some plants and animals beyond their historic ranges, with subsequent negative impacts on endemic species, while the mass release of manufactured products and compounds to all parts of the globe has had dire effects on the earth's atmosphere, soils, oceans, rivers, and wetlands.

Encouragingly, humans now better recognize responsibility for these impacts, understanding also that other species and communities deserve, and require, protection and sequestered place. Critically, and thankfully, this concept has now been accepted by national governments and global bodies such as the United Nations. Many countries have clearly stated goals and commitments toward environmental protection through conservation. However, while conservation of remaining natural lands is fundamental to these efforts, the enormous scale of nature's loss is such that the need to reconstruct and repair natural lands through ecological restoration is now recognized as an imperative for returning elements of balance between nature and human-modified lands. For this to be achieved across the globe at the scale and integrity required, we believe that restoration must be strategically and collectively planned and resourced, so that it can be undertaken at local, regional, national, and global levels. Only by using these layered, structured, and coordinated processes can we hope to ensure that restoration programs will be appropriately devised, funded, encouraged, and supported. Furthermore, it is important that restoration actions be undertaken which utilize the most suitable training, approaches, and techniques. In this way, it could be more confidently predicted that ecological restoration will achieve significant, sustainable, repeatable, and long-term outcomes.

We Have Progressed

In recent times, there has been a growing acceptance that nature can be better protected if it is more accepted and integrated within, rather than excluded from, human landscapes. Examples of how this has been achieved to date can be seen in various Case Study examples that demonstrate how restored lands can provide biological and other benefits to the community. And while it must be conceded all are in some form of diminished anthropogenic state, they should nevertheless be prized and valued for the "*gain*" in sustaining natural processes that each represents.

It is imperative that ecological restoration becomes successful because this shows us that, with effort, humans can effect positive change. People must not, as the many contributors to this book have shown, be resigned to our negative effects on nature. Instead, we should be buoyed by the knowledge that with will and effort, ecological restoration can create a pathway for people and communities to rectify impacts on nature at local to global scales. Such belief underpins this book, asserting that the restoration of nature to a stated or reference condition is feasible if we accept the realities of human influence, and provide the efforts and inputs required to reach those goals. In this way, the Case Studies presented here provide important windows for readers, offering a view of what has been achieved to date. By following this path, we hope that these lessons will provide the inspiration and knowledge required for others to expand the reach of their ecological restoration efforts in the future.

People and Practice

As this book demonstrates, ecological restoration is a function of people. If it is to succeed, people and communities must cooperate. It is, therefore, fundamentally, a social practice that requires collective will and effort. Its motivations are many and varied, and for many practitioners it creates meaning and a sense of positive contribution toward a greater universal good that counterbalances the often harshness of individual and short-term practices. Chapter 14 shows how the very underpinnings of ecological restoration are based on an interactive human-centric framing of how the past is understood and how the future is imagined. This framing can be influenced by a person's culture, ethnicity, gender, social position, economic status, politics, and personality. As a result, the practice of ecological restoration is necessarily variously molded and undertaken, creating paths not only to different ecological outcomes, but to those which can display complex, contradictory, and sometimes dissonant social and cultural outcomes. For these reasons, goals and strategies for repairing nature must acknowledge and be sensitive to local human histories and cultures and be cognizant of social inequalities and power imbalances. If both equity and inclusiveness underpin the practice of ecological restoration, it will benefit from the participation, input, and knowledge of all who undertake it, meaning that restored nature will then be something celebrated and valued by all.

For people to engage meaningfully with ecological restoration, they must have some understanding of the value of nature, together with an appreciation of the benefits that it brings to their lives and the lives of others. They must also have some understanding of the essences of ecological restoration itself – what is it, what can be achieved, and by what means? Therefore, champions from all fields of restoration practice are needed to build public understanding and support and to inspire our youth and indeed all of the wider community to participate in ecological restoration. This will be vital not only for achieving biodiversity outcomes but also for building more connected, caring, and committed communities.

Chapter 15 reveals that a great challenge for repairing nature at global scales continues to be the unbalanced economic values placed on natural capital and human-made capital. The continued loss and degradation of nature demonstrates how economic theory fails to support and protect natural capital. Intrinsically, while societies may benefit from the outcomes of ecological restoration, there may be little financial return for some individuals. This means there is scant incentive for private investment in such an undertaking and emphasizes the continuing need for public investment (as subsidies or other forms of compensation). Therefore, while setting grand targets for global restoration, such as those under the UN Decade of Restoration, are indeed clearly laudable, those societies with differing economic capacities must still find ways to pay what is required to meet those goals. Additionally, we all must continue to find better ways to ensure that ecological restoration actions are economically feasible.

For example, in recent years, Biodiversity Markets aiming to "Offset" the loss of nature from development have increased around the world. Most aim for "no net loss" of biodiversity, whereby any form of loss through development must be matched and exceeded through conservation and/or ecological restoration to provide a "gain." In economic terms, these markets create incentives for landholders to preserve or restore nature and, conversely, as a disincentive for others to destroy it. There, does, of course, remain flaws in these approaches, leading to concerns regarding whether destroyed areas can be replaced, and if markets can fully support the true cost of restoring and maintaining an Offset site into perpetuity. Further, it is asked, how can these markets assist repairing nature in areas where little development occurs, and how can we avoid the high cost of administration that can create inequities for those who become involved in these markets.

Other approaches are also used to meet biodiversity goals. In countries throughout the EU and in the United States, subsidy schemes have been used with great success to persuade farmers to conserve or restore native vegetation on parts of their lands. However, as with many things, their goals are often multifaceted and so these also aim to suppress the production of surplus agricultural products. Nevertheless, through direct payments for landholders, such programs create financial incentives for restoring nature. They also have significant flow-on benefits by helping to build capacity in the restoration sector, such as through the development of seed production farms to supply seed for restoration, or from investment in the building of equipment and machinery for undertaking restoration. The net effect is that restoration, as an integrated part of farming businesses, has created restored habitat at the scale of millions of hectares over recent decades.

Governments can create other positive restoration pathways, such as through policy directives that support ecological restoration. For example, Chap. 9 describes how, in the USA, Federal Government directives stipulate that State Transport Authorities must allocate certain percentages of funding toward restoring native vegetation on roadsides. As Case Studies in that chapter show, this has also created large markets for native seed and restoration services to meet those requirements. In many areas, this has led to US roadsides becoming wonderful repositories for native species. These outcomes not only provide biodiversity benefits, but they also provide secure career pathways and incomes for those in the sector. In addition, there are huge economic benefits for adjoining communities when these plantings become splendid floral attractions for tourists.

Many chapters and Case Studies highlight the importance of coordinated and strategic planning in the underpinning and achievements of restoration goals which, in the best-case scenarios, are undertaken in a way that is respectful, inclusive, and representative of all stakeholders. Ideally, restoration planning and resourcing is also done from regional, catchment, state, national, and even global levels. This degree of forward thinking can lead to the development of clear time and location-based restoration goals for all vegetation community types at all landscape levels, thus ensuring that they are underpinned by a realistic understanding of all inputs, resourcing needs, and costs. Together, these approaches increase the chance that restoration will be strategic, is adequately funded and resourced, and uses the most appropriate knowledge and techniques. This will increase the probability that projects will succeed, that restoration targets are reached, that biodiversity is preserved, and that vibrant, thriving, environmentally focused industries evolve to create meaningful careers and business opportunities for individuals and communities.

Nowhere is the need for forward planning more evidence than for native seed resourcing. Native seed is perhaps the core ingredient required for seed and plant-based ecological restoration. Several chapters describe how this key component is most often highly limited in quantity, quality, and diversity in the landscapes where restoration is most needed. While Chap. 12 discusses the situation where underdeveloped native seed supply chains are a feature in many countries, it certainly shows, through Case Studies, examples where governments and private initiatives have attempted to address and overcome this crippling issue. Among the areas of focus and success were (i) in the use of farmed seed production approaches to increase seed supply and species diversity, which have improved seed quality and lower seed costs; (ii) in the setting and use of standards for practice including in seed sourcing, handling, and testing; (iii) in the development of seed transfer zones to provide economic security for growers and to improve the genetic health of restorations; and (iv) in the development of community-led native seed supply chains to create opportunities for local cultural groups to derive social and economic benefits.

A significant milestone in the practice of ecological restoration was reached in recent years through the development of International Principles and Standards for the Practice of Ecological Restoration (Gann et al., 2019). This detailed and thoughtful guide for all those involved in the field of ecological restoration was compiled by the Society for Ecological Restoration (SER), an international non-profit organization with a global member base. The document itself provides a common structure and framework for planning, designing, implementing, documenting, and assessing ecological restoration. Importantly, to guide practitioners, it also presents a set of core principles that underpin the various components of ecological restoration. International Standards have also been developed by SER for Native Seeds in Ecological Restoration covering all facets of seed collection, production, handling, and use. Furthermore, SER Standards covering mine rehabilitation and restoration represent another large advance in coordinating and synchronizing our global efforts. Together these initiatives all represent critical sets of resources that are freely available to all, which provide internationally relevant guidance and instruction for all involved in ecological restoration.

As shown throughout various chapters, the task of practitioners is often to use ecological restoration to repair particular vegetations and the various species and communities that compose them in highly degraded landscapes. Thus, the focus of several chapters falls on community repair, including restoration of grasslands, savannahs, forests, and wetlands. It is here that perhaps the most important feature of this book comes to light – across the myriad of scenarios and vegetations highlighted in these pages are numerous examples of successful ecological restoration through community action. Of course, they also highlight the many obstacles and barriers that are faced, but they reveal insights into how these were addressed or overcome.

The pages of this book reveal that there is a broad continuum of interventions used by practitioners to achieve various restoration goals. These range from those that assist natural regenerative processes to those that actively reintroduce species as plants or seed, together with combinations of both approaches. Cases Studies highlight work that has pushed the prevailing boundaries of knowledge and technique to overcome issues such as over-nutrification of soils, altered hydrology, seed procurement, woody regrowth, dispersal limitations, micro-site limitations, herbivore impacts, and seed and plant establishment, showing what can currently be achieved in this field.

A Final Note

Our aim was that this book should provide the reader with a broad range of information on various aspects of ecological restoration, by identifying a range of situations and ecosystems relevant to its application, accompanied by concrete examples illustrating its practice. By taking this path, we hope that not only will it increase readers' understanding of the field of ecological restoration and why it is needed, but also that it will shed a clearer light on its practice and where it has been successfully applied. We hope that the basic knowledge provided in each chapter, amplified by insights from each of the Case Studies, will inspire and enable readers to do their part in continuing the mission to repair some of the areas of damaged nature and, by doing so, create a more equitable and healthy world, where humans and other species can inhabit our planet more harmoniously.

Reference

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