Chapter 9

RESTORATION OF TROPICAL FORESTS

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Restoration Ecology: The New Frontier, Second Edition. Edited by Jelte van Andel, James Aronson.

9.1 INTRODUCTION

Tropical forests are the most diverse ecosystems on Earth. These forests are found within 23.5° N or S of the equator in Asia, Oceania, Africa and Central and South America, in areas with a mean monthly minimum temperature of >18°C and a difference of <5°C between the warmest and coldest months. There are many tropical forest types that range along both moisture and elevation gradients (Holdridge *et al*. 1971). Rain forests are typically categorized as areas receiving >1500 mm of rain with >100 mm of rain in each month, whereas seasonal tropical forests generally have 3- to 6-month dry seasons when rainfall is <50 mm per month, and dry forests receive <1500 mm rainfall annually (Whitmore 1998). Tropical forests are found from sea level to beyond 3000 m elevation, and are categorized as lowland (<800–1000 m), premontane (~1000−−1500 m) and montane (>1500–3000 m), although the transitions between these altitudinal belts vary regionally. Tropical forests are found on well-drained soils, as well as on seasonally or permanently flooded soils (riparian forest or freshwater swamps) and along coasts (mangroves). This chapter focuses on nonwetland tropical forests.

Over half the tropical moist forest biome has been cleared over the last several decades (Asner *et al*. 2009), and much of the remaining forest is affected by fragmentation, selective logging and hunting. While the rate of tropical deforestation has slowed in some countries, it continues at a rapid pace in others (Asner *et al*. 2009). The causes of deforestation are complex and interrelated, and their relative importance varies by region (Geist & Lambin 2002). In Latin America, much of the forest has been cleared for pasture land and commercial agriculture, such as soybeans, sugar cane and coffee. In Africa, forest land is often cleared or degraded to supply firewood and/or arable land for subsistence agriculture; recently there has been increasing concern about the extensive bushmeat hunting and how the paucity of large mammals within some forests will alter plant–animal interactions, such as herbivory and seed dispersal, and in turn forest dynamics (Wright 2003). In Asia, much of the forest has been cleared for timber and/or agriculture with the land area in industrial-scale agriculture increasing in recent years. Smaller areas of tropical forests have been highly impacted by mining and oil drilling. As the majority of forest clearing has been carried out for a mix of logging and different types of agricultural uses, this chapter focuses on the restoration of such lands.

Despite ongoing pressures to clear tropical forests, there is substantial interest in restoring these forests, and tropical forest cover is increasing in certain regions (Asner *et al*. 2009). The motivation for restoring tropical forests comes from an interest in enhancing or restoring the delivery of **ecosystem services** (e.g. sequestering carbon, minimizing erosion and improving water quality, maintaining hydrological cycling and harbouring biodiversity) and maintenance of **natural capital** (Aronson *et al*. 2007a). In particular, there is increasing awareness that tropical forest clearing is the cause of approximately 15% of carbon emissions globally (van der Werf *et al*. 2009), so any effort to reduce carbon emissions must slow deforestation and forest degradation, and increase carbon stocks in degraded and cleared forests, a set of measures known collectively as *REDD+*. Moreover, there is a pressing need to restore forests to reduce erosion into nearby waterways and to conserve the high diversity of tropical forest organisms. In certain areas of the tropics, agricultural land is being abandoned and undergoing secondary succession due to changes in commodity prices and rural–urban migration (Marín-Spiotta *et al*. 2008; Letcher & Chazdon 2009). Together these changes offer opportunities to restore tropical forests, which necessarily must be balanced with the need to sustain human **livelihoods** in these regions.

Given the limited funds available for restoring tropical forest, it is critical to assess what type of **intervention** is needed in order to select the most effective forest restoration strategy (Lamb *et al*. 2005; Holl & Aide 2011). In this chapter, I first discuss factors limiting tropical forest *recovery*. I then briefly outline an approach for selecting an appropriate methodology from the different strategies used to facilitate tropical forest recovery and describe these methodologies in detail. This framework is broadly applicable to a range of tropical forest types, and I draw on examples from many of them. I conclude the chapter with thoughts about how to pay for tropical forest restoration and important considerations for tropical forest restoration in the future.

A common assumption of many restoration efforts is that if the full complement of plant species is restored this will result in the colonization of other groups of species (e.g. fauna and microbes), as well as the recovery of desired ecosystem functions (e.g. nutrient cycling and erosion control). Given that our knowledge on these topics for tropical forests undergoing restoration is limited, in this chapter I focus on summarizing results to date on restoring tropical forest plant communities, note when relevant information is available on other organisms and ecosystem functions and highlight the need for research on specific topics.

9.2 FACTORS AFFECTING THE RATE OF NATURAL RECOVERY

The rate of autogenic tropical forest recovery from human disturbance is notoriously variable. In some cases, tropical forest biomass and species composition may recover within a couple of decades (Marín-Spiotta *et al*. 2008; Letcher & Chazdon 2009), whereas in other cases the areas 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). In order to most effectively design tropical forest restoration strategies, it is critical to understand factors influencing the rate of natural tropical forest recovery generally, and to identify which factors are most important in the specific system being restored.

9.2.1 Ecological processes affecting tropical forest regeneration

Tropical forest plants commonly colonize degraded lands from seeds primarily dispersed from sources outside the site. Alternatively, plants may establish from three sources within the site – the *in situ* seed bank, seedlings already present and established at the time of land abandonment and/or resprouts from stumps, roots or stems – if these modes of regeneration remain after human disturbance. Many studies demonstrate that *dispersal* of animal-dispersed seeds – the primary form of dispersal in tropical rain forests – is often extremely low in former agricultural lands (reviewed in Holl 2002; Meli 2003). Moreover, most seeds falling in abandoned agricultural land are from species already present in the site or from a few small-seeded pioneer species, and seed rain is usually concentrated under remnant trees. Therefore, tropical forest recovery is commonly limited by a lack of **propagules** of forest species. In some sites, particularly those that were logged or used for short-term or low-intensity agriculture, there may be regeneration from resprouting and seed banks within the site; but, the extent of these forms of regeneration is highly variable (discussed below).

If forest plants either arrive at or are present in the site, then several factors may limit their establishment, survival and growth. These include competition from aggressive ruderal vegetation, stressful microclimatic conditions, limited soil nutrients, high seed predation/seedling herbivory and plant pathogens (Holl 2002; Meli 2003), and their relative importance varies from site to site.

A factor that has been shown to limit survival and growth of forest species in many sites, particularly in agricultural land, is aggressive existing vegetation that was either planted or rapidly colonized after the land was abandoned. **Exotic** pasture grasses (e.g. *Imperata cyclindrica*, *Melinus minutiflora, Pennisetum* spp. and *Saccharum spontaneum*) often form a monoculture in previously grazed areas (e.g. Hooper *et al*. 2002; Kettle 2010). In other cases, dense ferns (e.g. *Dicranopteris* spp. and *Pteridium* spp.) or other ruderal vegetation compete with forest seedlings (Cohen *et al*. 1995; Zimmerman *et al*. 2007). This residual vegetation may slow recovery by providing shelter for seed and seedling predators; competing for soil moisture, nutrients and light; increasing the probability of fire; reducing seed germination and emitting allelopathic chemicals which limit seedling growth (reviewed in Holl 2002).

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 are much lower in agricultural lands compared to forests. These microclimatic conditions may reduce survival and growth, particularly of later successional seedlings which are adapted to regenerate in the forest understorey and light gaps. Moreover, drier conditions in pastures, along with high grass biomass, make them particularly susceptible to the spread of fire, which kills the seeds and seedlings of most forest species (Janzen 2002; Nepstad *et al*. 2008). In contrast, pasture grasses are generally well adapted to resprout after fire, so repeated fires can lead to a transition to savanna vegetation (i.e. grasslands with trees spaced at a sufficient distance that the canopy does not close) in drier areas.

Next, in some sites, seedling growth is limited by soil conditions. Much of the tropics are covered by oxisols and ultisols, which have low nutrient levels and high acidity, whereas some areas have more fertile, volcanic soils, such as andisols and inceptisols. After human use, soils may be highly compacted, which impedes root growth and water holding capacity. Many tropical soils are phosphorus deficient (Vitousek & Sanford 1986), but nitrogen and other nutrients may also be limiting in some sites. Moreover, in certain types of soil, aluminium or iron may be present in sufficiently high levels as to be toxic to plants and therefore to slow or inhibit tree growth (Davies 1997). Finally, many tropical trees form mycorrhizal associations, which facilitate phosphorus uptake, but agricultural land uses may substantially alter microbial communities (Carpenter *et al*. 2001; M.F. Allen *et al*. 2005), which in turn affects *nutrient cycling*.

Seed predation and seedling herbivory can also be major obstacles to recovering agricultural lands. Common seed predators in the tropics include small mammals, bruchid beetles and leaf-cutter ants, which can consume more than 80% of seeds of some species within a few weeks (Nepstad *et al*. 1990; Jones *et al*. 2003). Likewise, herbivores – primarily insects but also mammals – can cause extensive damage to certain plants (Nepstad *et al*. 1990; Holl & Quiros-Nietzen 1999).

Finally, in some cases, plant pathogens cause extensive mortality of certain trees planted in restoration projects. For example, M.F. Allen *et al*. (2005) recorded >50% mortality of one tree species (*Cochlospermum vitifolium*) planted to restore tropical dry forest in Mexico, apparently caused by *Fusarium* sp., a widespread pathogenic fungus. Seed predation, seedling herbivory and plant pathogens are notoriously variable among species and over time, and thus can have a strong impact in some cases, but are less commonly the crucial factors limiting forest recovery.

9.2.2 Overarching factors affecting the rate of colonization and establishment of forest plants

The relative importance of the above-mentioned processes in influencing dispersal and establishment limitation, as well as the rate of natural recovery within specific sites, is influenced by the adaption of specific systems to local abiotic conditions, the past land use type and intensity and the surrounding landscape matrix (Holl 2007; Figure 9.1). In the tropics, the rate of natural forest *recovery* is largely related to the abiotic conditions with which these systems have evolved, including rainfall, temperature and soil type (Holl 2007). Natural **resilience** is often higher in drier than wetter tropical forest, in part because dry tropical forests have a much greater percentage of trees, shrubs and vines with wind-dispersed seeds (Vieira & Scariot 2006), which more readily disperse into open areas. Moreover, resprouting is more common in tropical dry forests (Vieira & Scariot 2006). Recovery also tends to be faster in relatively warmer and lower elevation areas, which generally favour more rapid growth (Zarin *et al*. 2001). Finally, differences in soil fertility across sites can strongly influence plant growth rates and forest recovery (Moran *et al*. 2000; Zarin *et al*. 2001).

The relative importance of different factors limiting forest recovery is also influenced by past land use

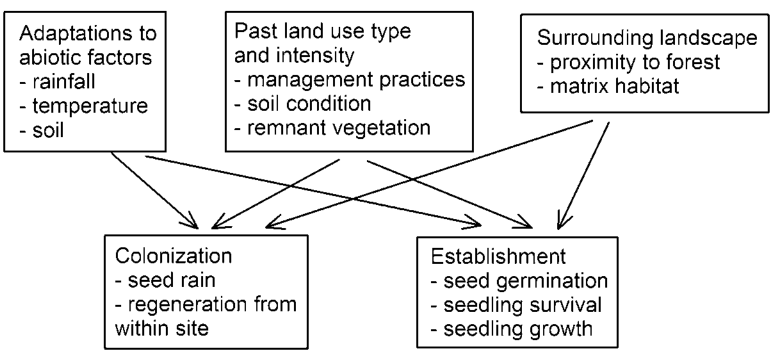

Figure 9.1 Factors affecting colonization and establishment of forest plants in cleared lands in the tropics. (Modified from Holl 2007.)

Figure 9.2 Former site of shifting agriculture in semi-evergreen seasonal forest (annual rainfall 1000–1200 mm) on the Yucatan Peninsula in Mexico. The site was used for 2 years for subsistence agriculture and had been abandoned for only 2 years when this photograph was taken. Substantial regeneration is taking place due in part to extensive resprouting. (Photograph by Karen D. Holl.)

history (Figure 9.1). The intensity of past land uses, which can range from selective logging to grazing to small-scale or industrial agriculture, affects many site-specific factors (reviewed in Chazdon 2003; Holl 2007). For example, both the intensity and duration of past land use affect the availability of propagules within a site (i.e. seed bank, resprouts and existing seedlings); lands used for selective logging or shifting agriculture are more likely to retain a seed bank of forest species than those used for extended grazing or monoculture crop production (Meli 2003; Holl 2007), which results in more rapid forest recovery (Figure 9.2). Past land use practices also influence the remnant vegetation in a particular site, which in turn may affect the rate of recovery. Aggressive pasture grasses are generally present in former pasture lands, but are much less likely to be an obstacle to recovery in lands that have been logged for timber or used for other agricultural crops. In contrast, if remnant trees are intentionally retained within agricultural lands, such as shade trees for coffee, cacao, tea or grazing animals in pastures, these trees can play an important role in facilitating natural recovery by attracting seed-dispersing animals, ameliorating stressful microclimate conditions and increasing soil nutrients (Loik & Holl 1999; Harvey *et al.* 2004). Even within a given land use type, the frequency and intensity of disturbance can vary; for example, the biomass accumulation rate in land used for shifting agriculture decreases with the number of times that land has been cleared following a fallow period (Lawrence *et al.* 2010). A host of management practices, including fire, herbicides and the use of heavy machinery, strongly affect the rate of recovery.

In addition to the natural resilience of the system, as expressed by rates of natural reseeding and reprouting, and the type and intensity of past land use, the composition of the surrounding landscape matrix also affects tropical forest recovery (Figure 9.1). Many studies have demonstrated the importance of proximity to intact tropical forest as a source of forest seeds and seed dispersers (Rodrigues *et al.* 2009). In areas used for large-scale industrial agriculture, such as sugar cane in Brazil, and elsewhere, nearby seed sources are often lacking (reviewed in Holl 2002), which can greatly limit the potential for natural recovery. The surrounding landscape matrix also affects recovery in other ways. Small patches of forest, riparian vegetation, living fences and remnant trees within the agricultural land use matrix that serve as sources of seeds, enhance the mobility of seed-dispersing fauna (Harvey *et al*. 2004; Chazdon *et al*. 2009a), and in turn facilitate the recovery process. In contrast, surrounding land uses can impede survival and growth of forest species within adjacent regenerating forest through the spread of agricultural chemicals, invasive species, or fire. Given the importance of the surrounding landscape matrix in tropical forest recovery, promoting surrounding agricultural practices that facilitate faunal movement and minimize negative impacts on recovering areas should certainly be part of efforts to facilitate forest recovery, although the remainder of this chapter focuses on restoration strategies within specific restoration sites.

9.3 SELECTING A RESTORATION STRATEGY

As a first stage in any restoration project, one needs to clearly identify the goals (Figure 9.3). These goals and specific objectives will necessarily need to be developed along with a consideration of the resources (e.g. financial, labour, sources of seeds or seedlings) available to achieve these goals and the natural **resilience** of the target ecosystem (Holl & Aide 2011). A goal of most tropical forest restoration projects will be to restore the species composition and processes of the pre-disturbance forest. However, given the competing needs of providing for human livelihoods and maximizing certain ecosystem services, there will be trade-offs concerning which goals will be prioritized (Holl & Aide 2011). For example, is it more important to restore the full suite of plant species or to maximize erosion control or sequester carbon? Large areas of cleared tropical lands are planted with trees that will be used for wood and fibre products, or that serve to diversify agricultural cropping systems. Such reforestation efforts do not aim to 'restore' the full complement of forest species and functions, but rather serve to increase tree cover and provide some ecosystem services. In addition, they may be critical for meeting the needs of people living in the region.

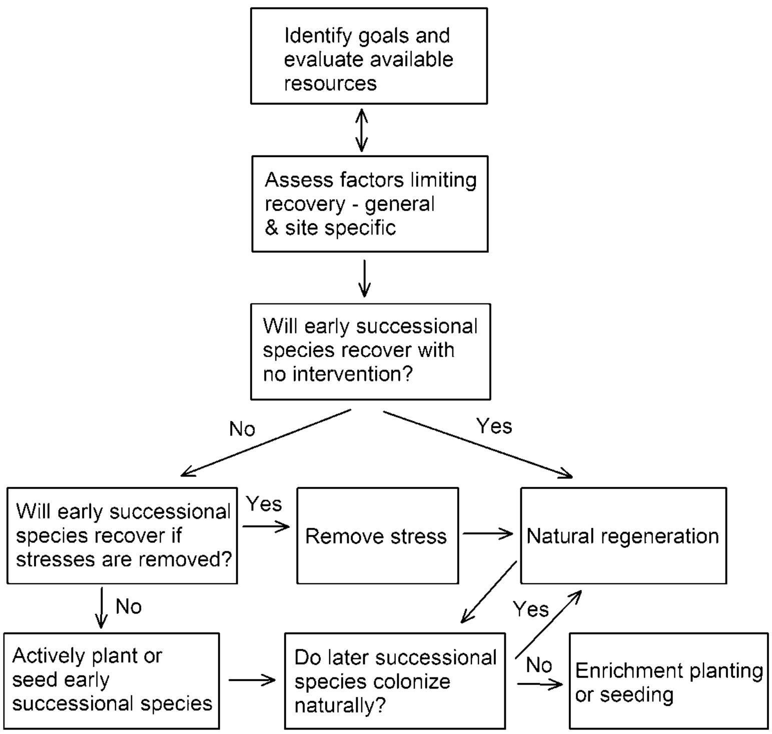

Figure 9.3 Process for selecting a tropical forest restoration strategy.

Next, it is important to assess which of the factors discussed in the previous section are likely to slow the initial phase of recovery and document the rate of natural recovery, in order to most efficiently use the limited resources available for the large expanses of degraded tropical lands that should be restored. The likely rate of natural recovery rate can be assessed by understanding the ecology of the system, observing natural recovery in the area to be restored and neighbouring sites, and reviewing similar cases from elsewhere, as reported in the literature.

A first question to ask is whether woody species, in particular those typical of early succession in the target area, will either resprout in or colonize the site naturally. There are a growing number of examples of tropical secondary forests that have recovered similar biomass, and in some cases similar species composition – as compared to nearby **primary forests** – within a few decades (e.g. Dent & Wright 2009; Letcher & Chazdon 2009). For example, in Puerto Rico, forest cover has increased from <10% in the 1930s to >40% by the year 2000, and forest biomass and species richness in these regenerating forests are similar to those of mature forests (Aide *et al.* 2000). If regeneration is likely to be high, then lands should be left to recover naturally. Not only is this approach less resource intensive, but also, in cases where there is extensive natural regeneration, planting trees can either slow recovery (Sampaio *et al.* 2007) or strongly influence the **successional trajectory** (Murcia 1997; Celentano *et al.* 2011).

If recovery is slow, the next question to ask is whether there is a residual factor affecting seedling establishment that, if removed, would accelerate the rate of natural recovery (Figure 9.3). If there is sufficient colonization through seed dispersal and from sources within the site, then it may not be necessary to actively introduce seeds or seedlings, which is usually time consuming and expensive. For example, large-scale tropical dry forest restoration efforts in north-western Costa Rica have focused on controlling fire during the dry season, which has been largely successful in allowing for natural recovery (Janzen 2002). This approach of removing barriers to establishment is sometimes referred to as an **assisted natural regeneration** (Dugan *et al.* 2003; Shono *et al.* 2007). Shono *et al.* (2007) discuss an approach for restoring *Imperata cylindrica*–dominated grasslands in Asia, where they mark all woody regeneration and then clear surrounding grasses to reduce competition and fire risk. They also clear fire breaks around the sites and do follow-up with enrichment planting of specific target species. These efforts to remove obstacles to natural regeneration without actively planting or seeding are usually much cheaper and leave less of a human imprint on the long-term species composition than planting and maintaining tree seedlings from the outset (Lamb *et al.* 2005), which can strongly affect the successional trajectory.

If colonization of woody species is limited by lack of **propagules**, due to low seed dispersal and sparse regeneration within the site, then it is necessary to intervene to introduce woody species (Figure 9.3). This is done most commonly by planting tree species and, in some cases, by direct seeding or creating more favourable habitat for seed-dispersing animals. Different approaches for introducing forest plant species are discussed in more detail in the following section.

As the naturally regenerating or actively restored forest matures, one must consider whether the full complement of later successional plant species recolonize over time (Figure 9.3). Many studies show that tropical **secondary forests** ranging in age from 30 to 60 years tend to lack large-seeded, mature forest species (Martínez-Garza & Howe 2003; Zimmerman *et al.* 2007); this is likely due to dispersal limitation, which is exacerbated in locations where large seed dispersers have been overhunted (Wright 2003). Therefore, numerous authors have suggested that for former agricultural lands where pioneer species rapidly colonize, the most effective restoration strategy is to plant selected mature forest species that do not readily recolonize (Martínez-Garza & Howe 2003; Lamb *et al.* 2005). In some cases, even if forest species are introduced in the early stages of restoration, it may be necessary to plant later successional species once site conditions are more favourable (discussed in more detail below; Lamb *et al.* 2005; Kettle 2010). Some later successional species have shown high survival in recently abandoned pasture (Loik & Holl 1999; Hooper *et al.* 2002), but more research is needed on when in the successional process these species should be introduced to maximize survival and growth.

9.4 STRATEGIES FOR ACTIVELY REINTRODUCING PLANT SPECIES

As noted earlier, if some or all native forest species do not colonize the site within a time frame that is suitable for the project, then it will be necessary to take actions to actively increase colonization. In this section, I discuss considerations for choosing methods to introduce and manage these species.

9.4.1 Tree planting

The most commonly used strategy is to **reintroduce** woody species, as seedlings, cuttings or seeds, which can help to overcome several of the obstacles to recovery. Specifically, trees provide the canopy architecture to encourage seed dispersal by birds, shade out light-demanding pasture grasses, ameliorate stressful microclimate conditions, and often improve soil structure and soil nutrient availability (Holl 2002; Cusack & Montagnini 2004; Figure 9.4); these effects in turn serve to enhance seedling establishment of forest species. There are several choices to be made when designing a planting strategy, including species selection, planting density, propagation method and seedling maintenance protocol.

Native species should be used preferentially for tropical forest restoration. Over the past two decades, there has been considerable research to increase knowledge of the natural history and nursery requirements of many tropical tree species (e.g. timing of fruiting, germination requirements and light and soil requirements), and to screen which of these species establish and grow rapidly in plantations (e.g. Rodrigues *et al*. 2009; Kettle 2010). Some studies have shown that planting fast-growing **exotic species** is a cost-effective approach to accelerating recovery, as these species may facilitate establishment of native species in the understorey (Ashton *et al*. 1998; Janzen 2002). These exotic trees can be logged after roughly a decade, allowing the native species to grow while also providing a source of income to landowners if there are nearby markets. Concerns, however, have been raised about potential long-term negative effects of these species on soil chemistry (Boley *et al*. 2009; Wei *et al*. 2009).

A few different native species–planting approaches are used (Lamb *et al*. 2005). Most commonly, a small subset of rapidly growing species are planted in hopes that they will create a closed canopy within 1–3 years and thereby facilitate forest recovery by attracting dispersers and creating microhabitat conditions favourable for forest seedling establishment (Lamb *et al*. 2005). The specific species selected, particularly when only a few are used, can strongly influence the composition of species that recruit in the understorey (Cusack &

Figure 9.4 Former pasture lands in Costa Rica 3 years after abandonment. The two sites looked identical at the start of the study. Control treatment (left) and plantation of four tree species to shade out grasses and facilitate seedling establishment (right). (Photographs by Karen D. Holl.)

Montagnini 2004; Wydhayagarn *et al.* 2009). Others have planted a mix of 20–30 slow- and fast-growing species to facilitate recovery, which provides more initial diversity (Tucker & Murphy 1997). A less common approach is to plant a large number (>50) of species with different growth rates and of different successional stages and growth forms at the outset (Lamb *et al.* 2005; Rodrigues *et al.* 2009); this requires more resources and knowledge for seed collection and propagation, and later successional species often have lower survival rates when planted into recently abandoned agricultural lands (Florentine & Westbrooke 2004). A final strategy is to plant larger seeded species that do not colonize naturally after an initial canopy has become established, either by planting fast-growing species or by natural establishment (enrichment planting; Lamb *et al.* 2005).

Tree seedlings and cuttings

The most common propagation method is to collect seed and then grow seedlings in nurseries for 2 months to a year prior to planting. This minimizes losses due to failed seed germination, seed predation and mortality of very young seedlings, which can be considerable (Holl 2002). Moreover, since tropical forest trees fruit throughout the year, seeds can be collected and germinated in the nursery at different times, and then the seedlings planted out simultaneously. Many tropical species have survival rates greater than 80% and grow 1–2 m per year, but survival and growth rates are notoriously site specific (Butterfield 1995; Holl *et al.* 2011).

Tree seedlings are usually planted in rows separated by 2–4 m, depending on the resources available and the desired rate of canopy closure; clearly, denser plantings require more seedlings but are more likely to close canopy sooner. Another approach is to plant patches of one or several tree species, which mimics the natural colonization process whereby shrubs and trees establish patchily (Zahawi & Augspurger 2006). This patch-planting approach is less resource intensive, given the smaller number of seedlings to grow, plant and maintain (Holl *et al.* 2011). It may, however, be more difficult from a practical standpoint, due to the irregular planting design, and will likely result in slower canopy closure (Holl *et al.* 2011), which in turn exposes young trees to more competition from aggressive grasses and the risk of fire during the dry seasons.

Some species of tropical trees are commonly propagated from cuttings of branches (often called ‘stakes’) to provide shade in cropping systems and serve as living fence posts. These stakes (typically 1–2 m tall) provide another alternative for reforestation and may develop more rapidly, in terms of above- and below-ground biomass, than those grown from seed (Zahawi & Holl 2009; Kettle 2010). Stakes do not require nursery facilities for propagation and are less costly to propagate, plant and maintain than seedlings. However, only certain species can be propagated vegetatively, large stakes can be difficult to transport, and stakes may result in lower genetic diversity when planted out than new stands produced from seedlings.

Direct seeding

Several past studies have demonstrated highly variable success with direct seeding, even amongst different species at the same site (Nepstad *et al.* 1990; Hooper *et al.* 2002). This variability is likely due to differences in seed germination rates, as well as losses due to seed predation, seedling herbivory, competition with pasture grasses and desiccation during seasonal dry spells. Direct seeding seems to be most effective for larger seeded species that have more resources stored within the seeds than smaller ones, and when seeds are introduced as part of enrichment planting after the canopy has closed (Nepstad *et al.* 1990; Hooper *et al.* 2002; Cole *et al.* 2011). Direct seeding is far less costly than planting seedlings or stakes (Lamb *et al.* 2005; Cole *et al.* 2011).

Regardless of whether seeds are directly introduced in the field or germinated under greenhouse conditions, collecting tropical seed from a wide variety of species can be difficult, as many tropical forest trees do not set seed every year. Moreover, many tropical forest tree seeds rapidly lose viability when dried, making storage impossible. In most cases, introducing a mixture of seedlings, cuttings and seeds will be most effective, depending on the biology of the individual species.

Seedling maintenance

After forest species are introduced, it is often necessary to maintain them for one to a few years following introduction. In areas where there are dense pasture grasses or other ruderal vegetation, this vegetation is commonly cleared with a machete, or less commonly a grass-specific herbicide (e.g. poast and fusilade), every few months, for a year or two (Craven *et al.* 2009; Holl *et al.* 2011). If soil nutrients are particularly low, then fertilizing seedlings may increase growth rates, although the effect of fertilizer on seedling growth is mixed (Carpenter *et al*. 2004; Zanini & Ganade 2005). If herbivory is high, then it may be necessary to take actions, such as fencing, to protect seedlings from mammalian herbivory (Holl & Quiros-Nietzen 1999) or use fungicides on leaf cutter ant colonies to control ant herbivory (Nepstad *et al*. 1990). Such maintenance efforts necessarily increase costs and can also have negative side effects, so they should be carefully tailored to site conditions. For example, widespread use of fertilizers may inhibit succession by encouraging the establishment and growth of weedy species (Harcombe 1977), and herbicides can reduce natural regeneration by forest species (Griscom *et al*. 2009).

9.4.2 Encouraging seed dispersal

Given that tropical forest recovery is often dispersal limited, another potential means for facilitating recovery, particularly in pastures lacking remnant trees, is the use of artificial structures, such as perches for birds, to enhance seed dispersal by frugivores. If such structures are successful in attracting animals into abandoned agricultural lands, then this approach would seem promising. Not surprisingly, however, such efforts have met with mixed results, since tropical forest trees can be both dispersal and establishment limited, and perches only serve to overcome dispersal limitation.

Several studies have investigated the use of bird-perching structures, which generally consist of branches or posts 2–5 m tall with a perching structure on the top (Holl 1998; Shiels & Walker 2003). In all these studies, a variety of fruit-eating birds were observed on perches, and seed rain under perches was higher than in areas without perches, but well below the number in forest and under remnant trees. Most seeds falling below perches were from ruderal species that are widespread in active agricultural land or pioneer species. The few studies that have measured seedling establishment below perches, however, have only found higher establishment of seedlings where pasture grasses were cleared or competition with other vegetation was minimal (Miriti 1998; Shiels & Walker 2003), suggesting that perching structures will only serve to facilitate recovery if barriers to establishment are low. Kelm *et al*. (2008) have suggested that bat boxes may serve as a strategy to increase seed dispersal by bats, but their efficacy in increasing seed dispersal in open pastures and their effect on seedling establishment have yet to be demonstrated.

A number of studies have also demonstrated that piles of branches and logs serve as perching structures and shelter for a number of bird species, while reducing light levels and temperatures at the soil surface, and providing safe sites for woody seedling establishment (Peterson & Haines 2000). In one study in Venezuela, artificially created piles of logs resulted in higher woody seedling establishment during the first year following pasture abandonment (Uhl *et al*. 1982). This promising technique for facilitating recovery should be tested more widely.

9.5 PAYING FOR TROPICAL RESTORATION

Given the large areas of land that have been deforested in the tropics and the competing need to provide for human livelihoods, a major question is how to pay for restoring tropical forests, particularly given that most are located in countries with relatively low per capita income. In addition to the costs of active restoration strategies, removing lands from agriculture or other uses for tropical forest restoration necessarily means a loss of income to people in these regions, particularly in areas that are particularly good for farming or have high value for resource extraction. Therefore, for forest conservation and restoration to succeed over the long term, they must be balanced with production systems carried out in other areas (e.g. for agriculture and timber). Additionally, landowners and other relevant **stakeholders** must be compensated for foregone income on lands where direct use is restricted. Fortunately, there are some promising approaches for paying for these costs.

First, given the increasing recognition of the need to restore **natural capital** to improve the flows of **ecosystem goods and services** that forests provide, both locally and globally (Aronson *et al*. 2007a), there are increasing numbers of programmes worldwide, and specifically in tropical forests, that financially compensate land owners for maintaining and restoring watersheds that provide these services (Wunder 2007). The efficacy of these programmes has been variable and, particularly with respect to tropical forest restoration, has generally not been well evaluated to date (Wunder

2007). However, there is some evidence that payments to small farmers and indigenous groups in Costa Rica have helped to cover the initial costs of tree planting and increase the number and diversity of trees planted (Cole 2010).

Secondly, as discussed previously, as part of regional, national, and global effort to reduce carbon and other greenhouse gas emissions, there has been an enormous amount of discussion about programmes to pay landowners in the tropics to reduce net carbon emissions from tropical deforestation, and enhancing long-term carbon storage in degraded lands. Certainly, such efforts must focus first on preserving intact forests. Moreover, there are complicated issues related to respecting indigenous peoples' rights, establishing baseline rates of deforestation and ensuring that payments are effective in reducing the drivers of deforestation (Blom *et al.* 2010; Brown 2010). Nonetheless, funding from such programmes may soon dramatically increase the amount of funding available for tropical forest restoration. Regenerating forests have the potential to sequester a considerable amount of carbon, particularly over the first few decades (Marín-Spiotta *et al.* 2008). However, there are certain to be trade-offs to consider between maximizing **carbon sequestration** or other ecosystem services, on the one hand, and maximizing **biodiversity** conservation in restored tropical forests, on the other (Lamb *et al.* 2005).

Finally, agro-successional restoration (i.e. incorporating a range of agroecology and agroforestry techniques as a transitional phase early in forest restoration) could be used as a means to defray restoration costs and provide for human livelihoods (Lamb *et al.* 2005; Vieira *et al.* 2009). Many of the management techniques used by farmers for decades to reduce weed competition and enhance soil fertility when cultivating tropical crops and trees are similar to those used for restoration. For example, interplanting some forest tree species with shade-tolerant agricultural crops, such as cacao or coffee, may help to reduce the initial costs of planting and maintaining tree seedlings. Others have suggested managing cows to both disperse seeds of some hard-seeded tree species (Miceli-Méndez *et al.* 2008), as well as to reduce grass biomass in areas where forest tree seedlings have been established. However, using grazing animals as a restoration tool requires careful management to minimize damage to forest seedlings.

9.6 PERSPECTIVES

Tropical forest restoration is critical to both conserve biodiversity and maintain ecosystem services. It is not, however, a substitute for preventing deforestation of existing forests. It is heartening that some tropical forests are able to quickly recover the structure and species composition of intact forests (Chazdon *et al.* 2009b). But rates of recovery are extremely variable depending on factors illustrated in Figure 9.3, and there are many other cases where it appears that less common and mature forests species have not recovered after a number of decades.

There is a strong mismatch between the temporal scale of human decision making and that of ecosystem recovery. In other words, humans want to see change quickly, or even immediately, but most ecosystems take years to decades to recover from disturbance without active interventions aimed at restoration. In most cases, a bit more patience to allow systems to follow the natural recovery process is needed. By waiting for at least a few years, it is possible to assess whether intervention is necessary and, if so, how to best allocate efforts. This caution is necessary not only because of the lack of resources available for restoration, but also because extensive human intervention can actually redirect recovery to a state quite different from the previous forest.

This chapter has focused on restoring plant communities, but certainly it is also necessary to consider whether faunal species colonize and utilize sites targeting for restoration (Bowen *et al.* 2007, Lindell 2008). This will depend on both nearby source populations and habitat quality. Two recent reviews of faunal utilization of naturally regenerated secondary tropical forests (Bowen *et al.* 2007; Chazdon *et al.* 2009b) reported that the number of faunal species generally increases with time from abandonment, but that the degree to which the number and composition of faunal communities recover is highly variable and that, not surprisingly, it is influenced by past land use and the surrounding landscape. There have been fewer studies of fauna in actively restored sites, and additional research is needed. Studies suggest that restoration efforts that succeed in enhancing vegetation structural complexity increase the number of both insects and birds (Grimbacher & Catterall 2007; Morrison *et al.* 2010), and that proximity to intact forest is important (Anderson 1993; Grimbacher & Catterall 2007).

Although ecosystem services are a motivation for restoring tropical forests, there has been relatively little study of the effect of forest restoration efforts on the recovery of ecosystem processes, besides carbon accumulation. Some studies provide promising results for the prospect of restoring nutrient cycling in restored forests (Macedo *et al*. 2008; Celentano *et al*. 2011) but long-term studies are important to determine how successful restoration efforts are in promoting the recovery not only of tropical forest structure, but also of the processes and functions that translate into ecosystem services to society.

Finally, throughout this chapter I have discussed tropical forest restoration efforts to date, which generally aim towards a species composition similar to that present prior to disturbance. Tropical forests, like all ecosystems globally, will be strongly impacted by climate change (J. Wright *et al*. 2009). Increasing carbon dioxide levels will not only directly affect plants, but also result in increases in temperature, decreases in cloud cover at high elevations, and changes in the timing and amount of precipitation. This means that in the tropics, as elsewhere, restorationists will have to make difficult decisions about selecting sources of seeds locally versus from more extreme conditions. They will also need to consider both the physiological tolerances of plants and animals to a changing climate and how these tolerances, along with limited capacity for dispersal, will affect ecosystem processes and biotic interactions.

ACKNOWLEDGEMENTS

I appreciate helpful comments from the editors, B. Ferguson, and D. Lamb on a previous version of this manuscript.