13

Conservation of tropical forests: maintaining ecological integrity and resilience

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'When we try to pick out anything by itself, we find it hitched to everything else in the Universe.'
— John Muir, My First Summer in the Sierra (1911)

Introduction

Viewed from space, the earth's tropical forests form a narrow green belt around the equator, with three significant blocks: in northern South America, West and Central Africa, and the peninsula and islands of South East Asia and Australasia. The total area has been reduced by about 50% since the beginning of the 20th century (FAO 2001) but around 1200 million ha remains, or approximately 5% of the earth's land surface. This might sound a lot, but deforestation and degradation of tropical forests continue at a high rate worldwide (Curran et al. 2004; Laurance & Peres 2006) and has become a *cause célèbre* for conservationists.

There are several good reasons why we should worry about modifying tropical forests and reducing their area. These ecosystems are a key element of global cycles of water and carbon, and changes to them are likely to have repercussions on a global scale (Lewis 2006; Lewis et al. 2009). More locally, tropical forests provide a suite of ecosystem services for human populations living in and near them. These include harvestable resources of timber, firewood and bushmeat, as well as less immediately

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obvious services like erosion control and stabilization of water supplies (Gardner et al. 2009). From a biodiversity conservation perspective, concern about deforestation and modification is motivated by a strong desire to protect the extraordinary concentration of biological diversity within tropical forests. These habitats are the most species rich on earth and may contain up to 75% of all terrestrial species. Ultimately, the ecosystem services provided by tropical forests depend on the persistence of their component species.

Conservation of tropical forest biodiversity is a straightforward proposition, at least in theory. Sophisticated ecological models, intensive single-species conservation efforts and a nuanced understanding of the ecological processes structuring tropical forests seem unnecessary. We already know the most effective way to conserve this ecosystem: stop destroying and modifying it! In practice, things are not that simple. Only a small fraction of the world's tropical forests, about 10%, is currently protected within parks or reserves (Brooks et al. 2009), and the demands of growing populations in tropical countries make it unlikely that this area will increase much in future. Meanwhile, 'external' pressures on existing protected areas, many of which might be characterized as 'paper parks' (Brandon et al. 1998), will continue to increase.

In this chapter, while acknowledging the critical importance of maintaining large, core areas of tropical forests as free as possible from human interference (Gardner et al. 2009), we address the need for tropical forest conservation efforts in the wider tropical landscape, beyond the boundaries of strictly protected areas. We highlight the need to understand the resilience of tropical forests to anthropogenic perturbations, focusing on ecosystem-level processes, particularly food web changes, ecological cascades, and alterations to ecosystem functions. We review empirical evidence for the resilience of tropical forests to different anthropogenic drivers, consider what humans can do to maximize resilience at various scales, and suggest that it may be possible to maintain tropical forest biodiversity by working within the bounds of 'natural' disturbances. We suggest that conservation efforts in the wider tropical landscape may increasingly need to retain functioning and resilient ecosystems, rather than biodiversity *per se*.

Practical approaches for achieving resilience will vary. While much of the news about conservation in tropical forests is negative, there are 'good news' stories from around the tropics. What approaches for conservation of tropical forests are working, and might these be applied more widely? We focus on three situations where, for varying reasons, there is cause to be optimistic and where we believe that significant practical progress is being made towards establishing stable, resilient tropical forests both within and beyond the borders of protected areas.

Destruction versus degradation: ecosystem-level consequences

One common misconception is that human impacts on tropical forests are all-or-nothing: forest is either present or absent. A second common misconception is the romantic notion that only 'virgin' forests are of any value from a conservation perspective (Perfecto & Vandermeer 2008). If tropical forests are clear-felled for plantations or agriculture, the habitat that replaces the forest will indeed support very little of the original biodiversity. Habitat destruction of this sort is a major factor in some parts of the tropics, notably forests in South East Asia converted to oil palm plantations (Sodhi et al. 2004). However, it has been suggested that the total forested area in some tropical regions (particularly Latin America) may actually be increasing (Wright & Muller-Landau 2006), although these calculations are controversial (Laurance 2007). Most would agree that the trend globally is a shift from relatively unmodified forest to modified and 'secondary' forests of various sorts. These forest fragments are embedded in a matrix of agro-ecosystems, which themselves may have considerable conservation potential (Perfecto & Vandermeer 2008).

Given that it will only be possible to protect a small fraction of the earth's tropical forests entirely from human impacts, it might be argued that the real battleground for conservation lies in ensuring that the inevitable harm that people will cause to forests is minimized. Since widespread exploitation of tropical forests appears unavoidable, can we plan this exploitation and manage the habitats that replace natural forests in a way that minimizes the repercussions for biodiversity and associated ecosystem services? In the context of logging, for example, this might involve ensuring that disturbances are within the bounds of those that a forest might experience naturally: so-called 'ecological forestry' (Hunter 1999). Indeed, such disturbances will inevitably be beneficial for some disturbance-adapted species, including commercially important trees such as mahogany (*Swietenia macrophylla* King) (Brown et al. 2003).

While humans have modified tropical habitats significantly for thousands of years (Heckenberger et al. 2008), current human perturbations to tropical forests differ in terms of their scale and intensity. Furthermore, they coincide with other, escalating drivers of global change including climate change, fragmentation, invasive species and overexploitation (reviewed by Laurance & Peres 2006). The extent to which ecological communities are likely to recover to their natural state over the long term is therefore a matter of debate and ongoing research.

It is increasingly recognized that we need to take a more inclusive, ecosystem-level perspective on ecological responses to perturbations. There are at least three reasons why a wider community- and ecosystem-level perspective is important. First, the intricate interconnectedness of ecological networks means that species that are not affected directly by perturbations can, in the long run, suffer through trophic cascades and other indirect effects. Second, changes in richness or composition may not give a full picture of the *functional* consequences of losses or changes to biodiversity. Third, responses to human disturbance may be long delayed, such that tipping points or thresholds of resilience may be crossed, perhaps before the full impacts of human actions are recognized. Understanding these aspects of community and ecosystem responses to tropical forest modification may be key to understanding the extent to which tropical forests can be modified without jeopardizing their biodiversity.

Trophic cascades and food webs

Ecological communities are intimately connected through networks of interactions, both positive (e.g. those between mutualists such as plants and their pollinators) and negative (e.g. competitive interactions, and trophic interactions involving predators and prey, hosts and parasites). Where these interactions are specialized and obligate, local or global extinction of one partner can lead to 'co-extinction' of the other (Koh et al. 2004). More subtly, changes to one part of the network may have repercussions elsewhere. Pace et al. (1999) suggest that trophic cascades may be intrinsically less likely to occur in high-diversity systems like tropical forests. However, there are some clear examples, notably the top-down trophic cascades generated on tropical islands lacking top predators, where high herbivore densities severely restrict plant regeneration (Terborgh et al. 2001). Keystone species do not always occupy the tops of food chains, and it seems likely that similar cascades follow the extinction of species or groups of species at lower trophic levels. For example, local extinction or reduced abundance of a single species in a network of hosts and parasitoids can have widespread repercussions for the abundance of other species, even if these are not directly linked to the impacted species (Morris et al. 2004).

A further possibility is that the ecological processes that help to structure and maintain diversity will be disrupted (Lewis & Gripenberg 2008). 'Mobile links' such as pollinators and seed dispersers play key roles in the dynamics of plant populations. Their decline or loss therefore has the potential to reverberate through food webs (Gardner et al. 2009). Diversity-enhancing processes may also be disrupted by anthropogenic disturbance. For example, the Janzen–Connell mechanism (where specialized natural enemies such as seed predators inhibit regeneration near conspecifics, helping to maintain high plant diversity) may be weakened if seed predator populations are depleted by hunting or habitat modification (Dirzo & Miranda 1991; Bagchi et al. 2011). Since plants form the basis of all food webs, and plant diversity appears to be the main driver of diversity at higher trophic levels (Novotny et al. 2006), such effects can cascade up to affect the diversity of the wider ecological community and, ultimately, plant diversity will be reduced.

Functional changes

Ecosystem functions include the physical, chemical and biological processes or attributes that contribute to the persistence of an ecosystem (Loreau et al. 2002). Notable examples include decomposition, pollination and cycling of elements. Modification of tropical forests is a major concern to conservationists worldwide, because it could lead to shifts in ecosystem functions and ecosystem services (the subset of ecosystem functions that are directly useful for humans). Altered ecosystem functions and services could occur at the global level, for example by changing patterns of rainfall or geochemical cycles (Lewis 2006), but are more easily documented at a local scale. For example, pollination of agricultural crops can be highly dependent on the diversity of insect pollinators, which in turn is sensitive to the management of forested landscapes. In Central Sulawesi, Klein et al. (2003) found that fruit set of coffee was strongly and positively correlated with the diversity of pollinating bees visiting plantations. Bee diversity in turn decreased with distance from the nearest forest, providing a strong incentive for farmers to conserve natural forested habitats in proximity to plantations. Despite such examples, most of the evidence base on the relationship between biodiversity and ecosystem functioning still comes from highly manipulative studies carried out at relatively low levels of diversity, largely in temperate systems. There are very few tropical forest studies that quantify how variations in diversity affect ecosystem functions and services, and how varying levels of anthropogenic impact affect functionally important components of diversity (Lewis 2009).

Of particular interest here are measures of species' sensitivity to different forms and intensities of disturbance ('response traits') and their contributions to ecosystem function ('effect traits': Lavorel & Garnier 2002). Not all species contribute equally to ecosystem functions, and not all species respond similarly to perturbations. Larsen et al. (2005) studied dung beetle assemblages on forest fragments isolated on artificial islands in Lago Guri, a reservoir in eastern Venezuela created in 1986 by the flooding of 4300 km^2 of semi-deciduous tropical forest. Dung beetles use animal dung as a food source and often bury it to provision their offspring. Dung burial by beetles accelerates rates of nutrient cycling, increases plant productivity, and helps seed dispersal and germination. Using dung-baited pitfall traps on 29 of the islands and the adjacent mainland, the researchers found that the smaller islands supported fewer dung beetle species and fewer individuals. They also measured the ecosystem function of dung removal using artificial dung patches of a known mass and volume. Rates of dung removal were lower on islands with low dung beetle richness and abundance. In separate trials they found that large-bodied dung beetle species were particularly important in processing dung. However, these functionally important, large-bodied species were those most likely to go extinct following forest fragmentation: they were absent from the smaller islands. In this case, 'response traits' and 'effect traits' are positively correlated, potentially leading to an accelerating loss of function with loss of species. It should be a priority to determine if such correlations are widespread for other functionally important plant and animal taxa. A further source of uncertainty is that tropical forest modification often leads to the formation of novel species assemblages: interactions among existing sets of species unravel, and new interactions form (Gardner et al. 2009). Inevitably, such compositional changes will have functional consequences, but these are poorly studied.

Resilience

Resilience can be defined as the 'capacity of a system to recover to essentially the same state after a disturbance' (Scheffer 2009). From the perspective of tropical forest conservation, it is important to avoid exploiting tropical forests in a way that exceeds their capacity for resilience. The danger is that changes accumulate past a 'tipping point', beyond which the system enters an alternative stable state and from which recovery to the original state is difficult or impossible (Lenton et al. 2008). It is widely argued that Amazonian forests may be approaching a tipping point where deforestation and a warming climate interact to increase the frequency of severe droughts and forest fires, and reduce overall precipitation to a point where forest dieback cannot be reversed (Malhi et al. 2009).

Whether the high diversity of tropical forest systems makes them intrinsically more or less stable remains an area of considerable debate. In theory, if multiple species can deliver a particular contribution to ecosystem function (i.e. have similar 'effect traits'), and these species respond differently to environmental changes including human perturbations, then collectively the system will be better placed to weather these perturbations, i.e. it will be more resilient (Folke et al. 2004). However, Ehrlich & Pringle (2008) suggest that tropical forests may be less resilient to human impacts, compared with other tropical habitats such as savannas. For example, livestock farming in tropical savannas can largely co-exist with the maintenance of the savanna ecosystem, presumably because it closely mimics the 'natural system' of high densities of wild grazing ungulates.

Other authors point to evidence that tropical forests can be relatively resilient in the long term. For example, Wright & Muller-Landau (2006) suggest that Pleistocene-era fragmentation of forests (particularly in West Africa) and long-term clearance and hunting by indigenous peoples (particularly in Central America) will have acted as an 'extinction filter' (Balmford 1996), making the surviving species relatively resilient to future perturbations. However, it is hard to know for certain, because few tropical forests have escaped all anthropogenic impacts (Willis et al. 2004; Lewis 2006). The Upper Xingu region of the Brazilian Amazon is currently covered by a large swathe of intact forest, but archaeological evidence shows that large parts of this region were densely populated and heavily cultivated between approximately 1250 and 1600 AD (Heckenberger et al. 2008). Similarly, modern-day Belize probably has a smaller human population and a greater area of tropical forest than it did 1000 years ago at the peak of the Maya civilization (Wright et al. 1959). Few biologists would immediately recognize the forests of Belize and the Upper Xingu as secondary regrowth, and they clearly retain relatively high biological diversity.

There is a risk that our expectation of what a diverse, intact and functioning tropical forest ecosystem looks like will be distorted by 'shifting baselines'; past disturbance may have generated patterns in biodiversity that are already substantially altered from the natural state (reviewed by Gardner et al. 2009). For example, Hanski et al. (2007) investigated dung beetle diversity in Madagascar, where approximately 50% of the forested area has been destroyed in the past 50 years and about 10% of the original forest cover now remains. The dung beetle fauna of Madagascar is well known from extensive collecting in the late 19th and early 20th centuries, before major deforestation occurred, providing a rare opportunity to assess the extent to which current levels of biodiversity reflect the baseline situation. In extensive recent survey work, Hanski et al. re-found 29 of the 51 species that had been documented for

Madagascar. It seems likely that many or most of the 'missing' species have gone extinct, because they had small distributions that no longer contain suitable forested habitats. The key message here is that without either excellent historical data or baseline data from 'intact' forest sites to provide a reference, it can be difficult to know what we are missing.

Practical solutions

Having reviewed the rather depressing challenges facing tropical forest conservationists, we now consider some areas for optimism, where co-existence of human activities with tropical forest biodiversity has been demonstrated to work, or where best evidence suggests that it may be feasible for people to co-exist sustainably with forest biodiversity.

Maintaining and restoring biodiversity in secondary forests

While some species are unlikely to persist in human-modified tropical forests, many such forests support very diverse flora and fauna (Dent & Wright 2009). In a study comparing the diversity of 15 taxa among forests of different disturbance levels, Barlow et al. (2007) showed that between 5% and 57% of species are restricted to primary forest. The flipside to this, of course, is that 43–95% of species may persist in modified forests, hinting at their potential value for conservation despite their degraded status. The values presented by Barlow et al. (2007) are more comprehensive than many comparable studies, but probably typical. For example, logging concessions around protected areas in the Democratic Republic of Congo supported very high levels of mammal diversity, including a number of endangered species (Clark et al. 2009), although it is not clear what proportion of those species had formed viable populations or were simply transient individuals. Similarly, secondary forests in Gabon support many of the invertebrate species found in primary forests (Basset et al. 2008). The take-home message from studies such as these is that conservation, and even restoration, of tropical forests need not be an issue that is purely focused on primary forests.

There are, of course, caveats to such a general statement about the importance of degraded forests, and many of these caveats relate to the community and ecosystem perspectives that we emphasized in the preceding section. Most importantly, the presence of any particular species in a modified forest habitat does not necessarily mean that it can maintain a viable population (Gardner et al. 2009). For example, mammal community structure in central African logging concessions changes with distance to protected areas (Clark et al. 2009), strongly indicating that at least some of the individuals detected in logged forests belong to populations that persist in primary forest. Such a spillover effect can lead to misinterpretation of basic forest biodiversity data, as degraded forests may represent unrecognized population sinks for species which persist solely because they have source populations in nearby primary forest. This spatial dependence is important, but rarely quantified. Equally important is to know whether modified habitats and agroecosystems have the potential to act as breeding habitat rather than population sinks if we adjust the manner in which they are exploited. For example, while conversion of forest to oil palm invariably leads to extinction of most forest species, riparian strips (areas of forest left uncut bordering existing streams and rivers) are now routinely left within oil palm plantations to prevent erosion from the cleared land leading to heavy sediment loads in the river water, and are often a legal requirement. In Sabah, Malaysia, plantation owners must maintain a 30 m buffer of riparian forest either side of any river greater than 3 m in width. Such habitats may have considerable potential as reservoirs of forest biodiversity, but how their conservation value varies with their width and whether they are significant in themselves, or only because they act as corridors allowing dispersal between remaining forest fragments, remain important questions for guiding conservation policy and practice.

Understanding the spatial relationship between the diversity of degraded forests and their proximity to primary forest leads directly to an important tactic for maximizing biodiversity gains from conservation efforts in degraded forests: the gain per unit effort is likely to be higher if efforts are directed towards degraded forests which persist close to primary forests. The biodiversity value of degraded forests is something that can be improved through careful management and restoration, although the time frames involved are long. The simplest approach is to leave the forest to recover unaided. The likely success of this approach depends, however, on the initial extent of degradation, and while natural regeneration can eventually restore much of the original forest diversity, it is clear that many of these second-growth forests retain only a subset of the original forest species (Lamb et al. 2005; Bhagwat et al. 2008; Chazdon 2008). Moreover, natural recovery is not an option that will work in all cases, and further interventions may be required, of which the most common is to focus on the plant community and to plant or seed a forest with 'missing' tree species (Lamb et al. 2005). The expectation is that animals require the food, shelter and other ecological resources provided by the trees themselves and there is little point in introducing them to a site until after the tree community has been established.

Simply restoring species to a site is unlikely to be sufficient. The new arrivals need to form self-supporting populations, for which they may require the restoration of ecological processes, functions and disturbance regimes such as species interactions, nutrient cycling and hydrological processes (Chazdon 2008; Gardner et al. 2009). To some extent, these processes can be expected to restore themselves as the species come back and begin to form interacting networks. However, other processes, such as natural flooding regimes, might be affected by changes outside the area being restored, highlighting the importance of considering a forest restoration project as being embedded and integrated within a wider landscape context (Lamb et al. 2005; Gardner et al. 2009). Fire regimes represent an important case study. In the humid tropical forests of the Amazon, natural forest fires are a rare event that results in dramatic changes to the structure and composition of tree communities (Barlow & Peres 2008). Human modification to Amazonian landscapes, and human activities in those landscapes, have led to increased fire frequency in 42% of the Brazilian Amazon (Aragão & Shimabukuro 2010). Increased fire frequency compounds the impacts of fire on Amazonian forests, because initial fires alter environmental conditions in a way that increases the probability and intensity of further fires (Cochrane et al. 1999), and because changes in tree communities increase greatly in magnitude when forests are burned multiple times (Barlow & Peres 2008). Restoring degraded forests in this region will require the suppression of anthropogenically increased fire regimes (Aragão & Shimabukuro 2010) to reflect more closely the regimes observed in primary forest.

Local stakeholder engagement and action

The conventional stakeholders of tropical forests are usually large government and conservation NGOs, but the continuing decline of tropical forests globally requires wider engagement and action. Local stakeholders have successfully conserved forests in many tropical regions. For example, islanders in Western Samoa have substituted ecotourism for logging (Cox & Banack 1991; Lowman et al. 2006), and local priests of the Coptic Church have successfully conserved some of Ethiopia's last forest fragments (see below). Increased publicity describing their successes could inspire others and provide models for effective forest conservation solutions elsewhere.

In tropical agro-ecosystems, farmers represent an emerging group of forest conservation stakeholders. Eighteen countries in Africa are currently engaged in trials of fertilizer trees as part of a new agroforestry movement, 'evergreen agriculture' (Garrity et al. 2010). Canopy foliage provides shade and litterfall nutrients for crops grown below the trees. Tropical countries may learn from the experience of Australia, which suffered widespread social problems when deforestation threatened rural livelihoods (Heatwole & Lowman 1987). In this case, clearing for sheep and cattle grazing destroyed more than 90% of the original eucalypt forest cover across much of New South Wales. The resulting loss of insectivorous birds led to outbreaks of Christmas beetles that defoliated and ultimately killed the remaining forest fragments. This forest dieback was only reversed when local farmers instigated planting activities, using local seed sources. A programme called 'A Billion Trees by 2000' was initiated, one farm at a time (Heatwole & Lowman 1987).

In addition to farmers and graziers, religious leaders comprise another successful group of local stakeholders, and land use practices associated with religious observance can be valuable for tropical biodiversity (reviewed in Verschuuren et al. 2010). One notable case study is the Coptic or Christian Orthodox church in Ethiopia (Wassie-Eshete 2007; Jarzen et al. 2010; Lowman 2011). There are over 35,000 church buildings throughout the country, some dating back to 360 AD, each surrounded by a tract of native forest because biodiversity stewardship is fundamental to the church mission (Wassie-Eshete 2007). Loss of these last remaining patches of forest would represent extinction for many native trees, insects, birds, and mammals, since the remaining landscape matrix is arid farmland with little or no forested habitat (but see Wassie-Eshete et al. 2009). Many church forest tree species are listed as threatened on the IUCN Red List (Wassie-Eshete 2007). These forest patches not only preserve biodiversity, but also provide numerous ecosystem services: pollination, native seed stock, shade, spiritual sites, medicines from the plants, and fresh water conservation through sustaining rainfall patterns and underground springs (Jarzen et al. 2010; Lowman 2011). Pressure from subsistence agriculture and demand for firewood threaten these tiny forest fragments, which are embedded in an otherwise brown and arid landscape. Religious leaders are working with an international group of conservation biologists to educate local people about ecosystem services, focusing on insect pollinators as indicator species of forest health and utility (Lowman 2011). Ethiopia has lost more than 95% of its forest cover, but the partnership of religion and science has the capacity to save the remaining 5%, and perhaps ultimately lead to forest restoration (Bongers et al. 2006).

A final example of stakeholders facilitating conservation efforts is through local people engaging in ecotourism as a sustainable income stream. In many tropical regions, the payments derived from logging operations far exceed any economic benefits from conservation (Novotny 2010). In most cases, however, logging provides one-off, non-renewable profits that benefit local people in the short term only. Ecotourism revolving around, for example, canopy access walkways, bird watching, education-based nature tours, spas and holistic medicine (Weaver 2001) sustains 'green businesses' which provide long-term alternative income streams for villagers (Lowman 2009b). For example, over 20 canopy walkways now operate in tropical forests around the world, serving research, education and ecotourism (Lowman 2009a). Canopy walkways range in cost from US$100 to US$3000/m to establish, but then they can generate annual revenues for local stakeholders, as well as providing environmental education opportunities well into the future (Lowman & Bouricius 1995; Lowman 2004). Maintenance costs are minimal in these tropical ecotourism sites, usually because the locals have expertise (and pride) to undertake constant inspection and repair (Lowman 2009a).

For example, in the Sucasari tributary of the Rio Napo in Peru, the world's longest canopy walkway provides employment for over 100 local families as well as educating thousands of western visitors every year about rainforest ecology and conservation (Lowman 2009a). Costing some $250,000 to build, it generates revenue estimated at $1.2 million/year and, most importantly, it provides an economic incentive to conserve the primary forest. Revenues are significantly higher than those that could be achieved by felling the timber, because they have been sustained for over 15 years. In Western Samoa, a canopy access platform was similarly constructed, enabling local villagers on the island of Savaii'i to pay for their new school from ecotourism profits, instead of from selling logs. In a village where there is essentially no cash economy, the metrics are fuzzy but the conservation success is evident (Lowman 2009a).

Protecting a forest the size of a continent: good news from the Brazilian Amazon

One of the main challenges in tropical forest conservation is to work at a sufficiently large scale to ensure survival of viable populations, where individuals are able to move between protected areas through corridors of suitable habitat. This becomes particularly important in the context of global climate change, where some species may be unable to maintain viable populations in the face of warming temperatures or changing moisture supply. Their survival will then depend on their capacity to disperse to cooler or wetter locations.

Although there has been much media coverage of the possibility of climate change-induced 'dieback' of some tropical forest regions, such as in the Amazon basin, a more likely future scenario is one where forests persist under expected climate change, albeit with substantial changes in species composition in response to rising temperatures and changes in atmospheric CO_2 and rainfall regimes (Malhi et al. 2009; Zelazowski et al. 2011). Indeed, protection of sufficiently large areas of intact forest has long been seen as an important tool to help forest species adapt to global climate change, by maintaining the regional rainfall recycling and microclimate cooling services that forests provide (Malhi et al. 2009), as well as maintaining habitat to facilitate future range shifts.

Given the challenges of both adapting to and mitigating climate change, can forest area and a forest matrix be conserved on a sufficiently large scale? In recent years there has been an increased recognition of the role that tropical forest conservation can play in mitigating climate change, and expectation of a major increase in resources available for tropical forest conservation, in particular through the REDD+ (Reduced Emissions from Deforestation and Forest Degradation) mechanism. This has encouraged renewed hope that tropical deforestation can be slowed at global scales. In reality, finance is only part of the solution, and such aspirations run up against the economic and demographic pressures on landowners needing to earn income from food or cash crops such as cocoa, palm oil and beef, as well as the challenges of good governance and sustainable development in tropical forest frontier regions. The opportunities for tropical forest conservation have never been greater, but the challenges are also immense.

Amidst the conflicting reasons for optimism and despair for the future of tropical forests, a compelling and optimistic story emerges from the greatest tropical forest region, the Amazon rainforest of Brazil, which holds two-thirds of the overall Amazon forest. Since the 1980s the deforestation of the Brazilian Amazon has been one of the iconic images of the environmental degradation of the planet, as large areas of primary forest have been converted to cattle ranches, soya fields and small-holder farms, fuelled by government-supported road expansion and settlement schemes, and in many places accompanied by an atmosphere of lawlessness, corruption and poor governance. Over the decade 1996–2005, Brazilian Amazonia had a deforestation rate of 19,500 km^2 per year, about half of total global deforestation.

Then, from July 2005 to July 2010, something remarkable happened. Deforestation rates declined rapidly, dropping to 6450 km^2 per year by 2010, and with many indications that this decline will continue. This is a reduction by 67%, to the lowest levels of deforestation recorded since monitoring began in the 1980s. Such a reduction has led the Brazilian government to declare an intention to reduce deforestation rates by 80% below the 1995–2005 baseline by 2020, and some have suggested it is possible for net deforestation to come to a complete halt by the end of this decade (Nepstad et al. 2009). Such a turnaround is truly remarkable. If Brazil's ambitions can be achieved, it opens the prospect of Brazil achieving an advanced state of economic development with > 70% of its Amazon forest area still intact and supporting native ecosystems. This contrasts markedly with the heavily deforested and altered forest landscapes of North America, Europe and Asia, although the Atlantic Forest and *cerrado* savannas, the other major woody biomes of Brazil, have not fared so well, at least in part because government policy has shifted agricultural activity to these areas.

What are the factors that have driven this reduction in deforestation, and are they sustainable? What lessons do they hold for the future of other tropical forest regions? First, it is important to recognize the nature of deforestation in Brazil. Cattle ranching accounts for 80% of deforestation, with mechanized soya bean agriculture as a second major cause (Nepstad et al. 2009). Small-scale farming causes only a small fraction of deforestation. Hence, Brazilian deforestation is driven by (moderate to high) wealth, national economic integration and global market demand; it is not mainly driven by poverty, local demographic pressure or marginalization, as is the case in many other tropical regions. This level of organization and scale is the reason that deforestation rates are so high, but also means that these processes are more open to pressure for governance, certification and high environmental standards (Nepstad et al. 2009). Some of the initial causes of the reduction in deforestation rates have been economic, as the drop in the price of beef and soya over the period 2004–2006 reduced pressure for new land, but the subsequent rise in these prices coinciding with ongoing decline in deforestation rates suggests that the link between market demand and deforestation seems to have been broken. A number of features explain this dramatic decline.

Technical capacity and information

To manage deforestation, it is important to know where it is and why it is happening. Until recently, most tropical deforestation occurred under conditions of global ignorance. Brazil has led the world in open and sophisticated monitoring of its deforestation by satellite, both through its national space agency INPE and through environmental NGOs. INPE's PRODES system has been providing annual reports of forest loss. This has highlighted hot spots, enabled identification of illegal activities and also, importantly, raised the profile of deforestation. More recently, the DETER (detection of deforestation in real time) system, in parallel with similar initiatives driven by environmental NGOs such as Imazon in Brazil, has allowed monthly or shorter time scale reporting of deforestation activity (albeit at a lower resolution). This has become a powerful enforcement and governance tool, as new hot spots and drivers of deforestation can be acted on before they are a *fait accompli*.

Leadership and governance

Information is only useful if there is a will to use the information, and if it is used to make planning decisions that take forest conservation into account. In this regard, Brazil has shown environmental leadership. Some of this has been by a 'bottom-up' process of consulting stakeholders and planning resource use around new road expansion schemes, but much has also been by 'top-down' processes of enforcement of existing laws, and action on corruption and illegality. Such enforcement becomes much clearer when satellites provide deforestation information, and private land claims are clearly registered and demarcated. Recent examples of governance and enforcement include a federal campaign to identify and imprison illegal operators, including government employees. In 2008, the municipalities responsible for 50% of current deforestation were the focus of another federal campaign to register properties, publicize illegal holdings, cancel lines of credit for illegal landholders, and pressurize buyers of products from deforested lands (Soares et al. 2010).

Protected areas

Where sufficient governance exists, protected areas can be a powerful tool to assist in conservation of forest blocks in the context of regional development. Brazil has been extremely active in this regard, and has expanded its network of protected areas in Amazonia from 1.26 to 1.82 million km^2. This alone is estimated to have contributed 37% of the region's total reduction in deforestation between 2004 and 2006 (Soares et al. 2010).

Scientific institutions and civil society

The fact that controlling Amazonian deforestation has reached such a high profile within the Brazilian government (amidst intense political pressure for maintaining high deforestation from some lobbies) is a credit to the active engagement on these issues by informed Brazilian scientific institutions, and the active engagement by civil society groups, many well informed and with high technical capacity. This has led to Brazilian 'ownership' of the issue of Amazonian deforestation, encouraging dialogue and action. The technical capacity within Brazilian research institutes, governments and civil society is to some extent a product of decades of international scientific collaboration in Amazonia, through which many students and young scientists have been trained and have subsequently risen through the ranks of academia, government and civil society.

The remarkable decline in deforestation in Brazilian Amazonia has lessons for the wider tropics, despite the very different socio-ecological contexts and drivers of deforestation in different regions. Much is possible with leadership, technical capacity, open availability of information and good governance, and very little is possible without these factors. This has lessons for the surge of interest in financing forest conservation through REDD+. There is a need to build technical capacity and solve wider problems of governance and development at appropriate scale if REDD+ is to make a globally meaningful contribution to forest conservation. Even in Brazil, the challenges are ongoing, as a more crowded and wealthier world increases demand for food and biofuels at the expense of natural ecosystems.

Conclusions

Tropical forests are threatened by a suite of co-occurring human impacts. Chief among these threats are deforestation, overexploitation, climate change, habitat fragmentation and degradation, and invasive species. Often their effects will be synergistic; for example, both logging and climate change are likely to increase the frequency of damaging fires, which would be rare in unmodified forests (Barlow & Peres 2008). Ensuring the resilience of tropical forests and the persistence of their biodiversity in the face of this onslaught will require a pragmatic approach that extends well beyond the boundaries of protected areas. Reserves protecting core areas of undisturbed forest will remain the 'gold standard' for tropical forest conservation. We do not wish to downplay their importance: they are likely to be the only strategy able to guarantee the survival of a substantial proportion of tropical forest species. However, humans have already substantially modified much of the tropical landscape, and intense human pressures will inevitably shape its future. Part of the role for protected areas will be as a source of propagules able to colonize nearby human-modified habitats, allowing the natural restoration of species and ecosystem functions following perturbations. Informed by a landscape and ecosystem perspective, managers and scientists need to take advantage of any opportunity to maintain functioning, diverse ecological communities in human-modified tropical landscapes.

There is no single magic solution for tropical forest conservation and conservationists will need to be pragmatic, flexible and adaptable to promote the best solutions in the context of different economic pressures and varying ecological contexts. There is a wide range of possible solutions available in our armoury, some of which we have discussed in this chapter.

Destroying rain forest for economic gain is like burning a Renaissance painting to cook a meal.
— Edward O. Wilson

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