Chapter 29

Science Panel for the Amazon (SPA)

Working Group 10

Conservation, Restoration, and Forest Transitions (Opportunities, Strategies and Challenges in Terrestrial and Aquatic systems

Lead Authors: Jos Barlow & Plinio Sist

Restoration priorities and benefits within landscapes and catchments and across the Amazon basin

Jos Barlow, Plinio Sist, Rafael Almeida, Caroline C. Arantes, Erika Berenguer, Patrick Caron, Francisco Cuesta, Carolina R. C. Doria, Joice Ferreira, Alexander Flecker, Sebastian Heilpern, Michelle Kalamandeen, Alexander C. Lees, , Camille Piponiot, Paulo Santos Pompeu, Carlos Souza, Judson F. Valentim

Chapter 29

Restoration priorities and benefits within landscapes and catchments and across the Amazon basin

Jos Barlow[1], Plinio Sist[2], Rafael Almeida[3], Caroline C. Arantes[4], Erika Berenguer[1,5], Patrick Caron[6], Francisco Cuesta[7], Carolina R. C. Doria[8], Joice Ferreira[9], Alexander Flecker[10], Sebastian Heilpern[11], Michelle Kalamandeen[12], Alexander C. Lees[13], Camille Piponiot[14], Paulo Santos Pompeu[15], Carlos Souza[16], Judson F. Valentim[17]

[1] Lancaster Environment Centre, Lancaster University, Lancaster, UK, jos.barlow@lancaster.ac.uk
[2] CIRAD, Université de Montpellier, UR Forests & Societies, Montpellier 34398, France, sist@cirad.fr
[3] Department of Ecology and Evolutionary Biology, Cornell University, USA
[4] Center for Global Change and Earth Observations, Michigan State University, East Lansing, USA
[5] Environmental Change Institute, University of Oxford, Oxford, UK
[6] University of Montpellier, Cirad, Umr ART-DEV, Montpellier 34398, France, patrick.caron@cirad.fr
[7] Grupo de Investigación en Biodiversidad, Medio Ambiente y Salud - BIOMAS - Universidad de Las Américas (UDLA), Quito, Ecuador
[8] Laboratório de Ictiologia e Pesca, Departamento de Ciências Biológicas, Universidade Federal de Rondônia (UNIR), Porto Velho, Brazil
[9] Embrapa Amazonia Oriental, Trav. Eneas Pinheiro, Belém, Brazil
[10] Department of Natural Resources, Cornell University, USA
[11] Department of Natural Resources, Cornell University, 226 Mann Drive, Ithaca NY 14853, USA
[12] School of Geography, University of Leeds, Leeds, UK
[13] Department of Natural Sciences, Manchester Metropolitan University, UK
[14] Smithsonian Conservation Biology Institute & Smithsonian Tropical Research Institute
[15] Departamento de Ecologia e Conservação, Instituto de Ciências Naturais, Universidade Federal de Lavras, Lavras, MG, Brazil.
[16] Agroforestry Research Center of Acre, Embrapa Acre, Rio Branco, Acre, Brazil
[17] Agroforestry Research Center of Acre, Embrapa Acre, Rodovia BR-364, Km 14 (Rio Branco/Porto Velho), Rio Branco AC 69900-970, Brazil

Chapter 29

INDEX

Chapter 29

KEY MESSAGES

- Identifying priority locations for restoration across the Amazon basin depends mainly on targets (e.g., increasing carbon stocks or conserving threatened species). These priority regions must be identified through participatory approaches involving local peoples and governments, supported by up-to-date scientific evidence.
- Restoration strategies will be more effective if they consider complementary conservation measures, such as the protection of remaining primary forests (see Chapter 27).
- For long-term success, restoration policies and programs must generate socioeconomic benefits for local populations (e.g., food security, employment, and income opportunities) and raise awareness of the benefits that forests and other natural systems provide.
- Considering where and how to restore at the catchment or landscape scale can help return much higher benefits than simple site-based approaches.
- Implementing restoration at the landscape- and catchment-scale must consider a broad range of restoration options, from encouraging the natural regeneration of secondary forests to restoring economic activities in degraded lands. This will help ensure resto-ration delivers the greatest benefits to the broadest range of stakeholders.
- Restoring ecosystems in the context of climate change requires rebuilding more resilient ecosystems for the future, for example selecting tree species that are more adapted to drier climates or helping maintain the natural flow regimes in aquatic systems.

Chapter 29

ABSTRACT

Restoration can be applied in many different Amazonian contexts, but will be most effective at leveraging environmental and social benefits when it is prioritized across the Amazon basin and within landscapes and catchments. Here we outline the considerations that are most relevant for planning and scaling restoration across the Amazon.

Chapter 29

1. INTRODUCTION

When restoration has been identified as an important action to achieve a particular target (e.g. Chapter 28), the first tier of prioritization involves identifying which areas to restore. Across ecosystems, systematic conservation planning aims to support decision making regarding the allocation of resources (Margules and Pressey 2000). These approaches have been widely used to help identify priority areas for conservation or restoration across the world (e.g. Strassburg *et al.* 2020) and within catchments (e.g. Beechie *et al.* 2008; McIntosh *et al.* 2017). In this chapter, we go beyond the specific restoration options outlined in chapter 28 to examine benefits of planning conservation across the basin and in catchments and landscapes. We then outline how restoration can be used to speed up a forest transition in the Amazon, before outlining some of the important crucial societal benefits. Finally, we explore the resilience of restoration to climate change, and what measures could help encourage large-scale restoration across the Amazon.

2. PRIORITIZING RESTORATION ACTIONS ACROSS THE AMAZON BASIN

Despite a growing number of global and ecosystem level prioritization exercises (Crouzeilles *et al.* 2020; Strassburg *et al.* 2020) very few formal analyses exist prioritizing restoration actions across the Amazon basin (e.g. weplan-forests.org) or identifying optimal scenarios to realize multiple aims. Here we outline some of the key ecological and societal benefits that could be attained from a large-scale, basin-wide restoration program.

2.1. Conservation of Amazonia's threatened species and unique ecosystems

Habitat loss is the main cause of biodiversity loss globally and it is not surprising that the most threatened terrestrial species in the Amazon have distributions coinciding with the most deforested and degraded regions such as Andean slopes and the "Arc of Deforestation". In these regions, restoration could play a key role in supporting the conservation of some of the Amazon's most threatened forest-dependent species (Figure 1), including the recently rediscovered Belem Curassow *Crax [fasciolata] pinima* (Alteff *et al.* 2019), Black-winged Trumpeter *Psophia obscura*, and the Kaapori capuchin *Cebus kaapori*, which was only described in 1992, all of which are Critically Endangered on the

IUCN list. However, the first priority in these regions is avoiding further deforestation and degradation by protecting existing forests from logging and forest fires (Chapter 27, Silva Junior et al 2020), and to reduce hunting pressure by tackling commercial hunting and illegal trade, providing alternative livelihoods to communities dependent on bushmeat, changing cultural attitudes, encouraging community-based management with local benefits from ecotourism (Bragagnolo *et al.* 2019)or even incentivizing alternative hunting practices that are less likely to affect the rarest arboreal species (such as hunting with dogs (Constantino 2019).

Figure 1: (to be included) Amazonia's threatened species (photos of key taxa organised by their red list status). To include: Belem Curassow *Crax [fasciolata] pinima*, Black-winged Trumpeter *Psophia obscura*, the Kaapori capuchin *Cebus kaapori*, Harpy eagle, Jaguar and white-lipped peccary.

While proactive conservation needs to focus on the Critically Endangered and/or range-restricted Amazonian species, there are many widespread species that are of conservation concern that could be supported by large-scale restoration. These include large and charismatic vertebrates such as the Near-Threatened Harpy eagle and Jaguar and the Vulnerable White-lipped peccary (BirdLife International 2021, IUCN Red List for birds, IUCN Red list 2020). While these species also require alternative interventions across the basin to reduce hunting pressure and persecution (Section 27), their populations could also benefit from restoration actions that help reconnect remaining forests and important habitat such as flooded forests. Actions that allow degraded forests to recover will also be key, as they will improve keystone resources such as important fruiting trees that are vital for wide ranging species such as the White-lipped peccary, or a viable prey base for apex predators such as the Harpy eagle and Jaguar.

Species-based restoration actions in the Amazon needs to consider the different habitat types within the biome. Some of these hold distinct biota, most notably white sand forests (Guilherme et al. 2018), bamboo-dominated forests of SW Amazonia (Kratter 1997), varzea and igapó (Haugaasen and Peres 2007), and the savanna enclaves (De Carvalho and Mustin 2017). These ecosystems are both diverse and unique in their own right, and can hold high levels of endemicity. Some of these ecosystems are even yielding new species discoveries – the Near Threatened Campina Jay (*Cyanocorax hafferi*) was only discovered in 2002 and is endemic to campina enclaves in and around the Madeira-Purus interfluve. It is well known that afforestation of open habitats - including oil palm expansion in savannas - can have negative consequences for biodiversity (Fernandes et al. 2016) and it is vital that restoration efforts protects the integrity of Amazonian savannas and other unique habitat types (Lees *et al.* 2014).

2.2. Improved functional connectivity of river systems

One vital advantage of a basin-wide approach is that the integrity of river systems relies on a high degree of spatial connectivity that operates in multiple dimensions; that is, longitudinally (upstream-downstream), laterally (river channels-riparian zones-floodplains), and vertically (surface-subsurface-groundwater) (Ward 1989; Castello and Macedo 2016). Further, seasonal and interannual flows represent a temporal fourth dimension of connectivity. The river continuum concept (Vannote *et al.* 1980) and the flood pulse concept (Junk *et al.* 1989), two foundational paradigms describing riverine and floodplain structure and function, are premised on the importance of longitudinal and lateral connectivity as central organizing features of energy flow, food web structure, and nutrient dynamics of running water systems. Freshwater ecosystems display an acute dependency on subsidies of materials, nutrients, and organisms that originate from elsewhere in the riverscape and landscape, and restoration efforts need to ensure these material and organismal transfers are not disrupted by barriers (Freeman *et al.* 2003; Flecker *et al.* 2010). Likewise, maintenance of natural flow (Poff *et al.* 1997) and sediment regimes (Wohl *et al.* 2015) are fundamental for the functioning of rivers and floodplains. For example, sediments that build Amazon floodplains are transported long distances from their source of origin in the Andes (McClain and Naiman 2008). Thus, restoring aquatic ecosystems to more natural states involves supporting vital multi-dimensional linkages found throughout river basins, as well as sustaining the organisms embedded in these systems. Such restoration needs to focus on the full hydrological network, from headwaters though to the main channels.

2.3 Global and biome-wide climate benefits

According to the IPPC, adding up to 24 million ha of forest across the world every year until 2030 would store around one-quarter of the atmospheric carbon necessary to limit global warming to 1.5 °C above pre-industrial levels (IPPC 2018). Natural forest regrowth following complete or nearly complete removal of forest vegetation can therefore play a significant role in climate change mitigation (Chazdon *et al.* 2016a; Lewis *et al.* 2019; Cook-Patton *et al.* 2020). For example, the 2.4 Mha of secondary forests in tropical Latin America could accumulate a total aboveground carbon stock of 8.48 Pg C (petagrams of carbon) in 40 years (Chazdon *et al.* 2016b). This is equivalent to all the carbon emissions from fossil fuel use and industrial processes across all of Latin America and the Caribbean from 1993 to 2014 (Chazdon et al. 2016).

Where climate change mitigation is a priority, restoration will be most effective on a per hectare basis if it occurs where growth rates are fastest – which is generally in the aseasonal regions and in the western Amazon where soils are more productive (Heinrich *et al.* 2021), and where the previous land-use intensity was low. However, to date most deforestation has occurred in drier regions of the Amazon, and, as a result, most secondary forests (and also most opportunities for large-scale restoration) are in regions that are seasonally dry, have suffered higher land use intensities, and have low levels of remaining forests cover (Smith *et al.* 2020) For example, secondary forests in the Brazilian Amazon have a mean annual precipitation of 1,945 mm, compared to the regional average of 2,224 mm, while their average maximum climatic water deficit is −375.5 mm compared to a regional average of −259 mm. In the drier and most deforested regions, carbon accumulation rates of secondary forests are some of the lowest in the Amazon (Elias *et al.* 2020; Heinrich *et al.* 2021). However, this does not mean that these regions should not be a priority for restoration, as the slow growth is offset by the higher availability of land for restoration, and the lower opportunity costs of conducting restoration on degraded farmland that is often unprofitable (Garrett *et al.* 2017). Furthermore, forest restoration in highly deforested areas may be more important for biodiversity and climatic benefits: these new forest fragments may act as important habitat for species, and the increase in forest cover can potentially increase local rainfall (See section X). The importance of these opportunities for restoration are recognized within climate change targets – for example, the Brazilian state of Pará aims to restore 5 Million hectares of forest as part of its "Plano Estadual de Amazonia Agora", helping it achieve carbon neutrality by 2035 (Pará State Decree 941/2020).

Crucially, restoration may support the integrity of the biome itself, enhancing its resilience to climate change by reducing the influence of climatic extremes and avoiding dangerous tipping points resulting from climate and land-use change (Chapter 23). The climatic benefits from terrestrial restoration could also help the Amazon maintain its hydrological integrity, with evapotranspiration from restored forests contributing to the east-west transfer of moisture. This could help support aquatic ecosystems, ensuring the maintenance of river discharge dynamics across the basin, and even the nutrient transfer from freshwater to floodplains and beyond. Restoring the basins hydrological functioning would maintain a more humid climate, helping prevent the forest fires, which are one of the main determinants of any sudden tipping point (Nobre *et al.* 2016). However, care must be taken to ensure that restoration itself does not make landscapes more flammable; for example, secondary forest understories tend to be hotter and drier in the day than primary forests (Ray *et al.* 2005), and, depending on what systems they replace, have the potential to aid the spread of fire across landscapes.

2.4 Societal benefits

Restoration of forests and sustainable economic activities are a high priority for some of the most deforested regions of the Amazon, as these older deforestation frontiers include some of the municipalities with the lowest Human Development Index values (HDI), Rodrigues *et al.* 2009). The transformation of unproductive lands into productive and sustainable agricultural or agroforestry systems could yield many direct economic and social benefits (Chapter 30), but there are also many indirect effects of restoration that could provide benefits for society beyond the producers. For example, the climatic benefits of increasing forest cover (e.g. Alkama and Cescatti 2016) could mitigate some of the higher temperatures associated with climate change, thereby improving other economic activities across the landscape and supporting wellbeing. Some of these benefits could be of considerable economic importance, as maintaining or even reducing dry season length could enable the continuation of the 'double cropping' system that is vulnerable to climate change (e.g. Andrea *et al.* 2020). The restoration of aquatic systems will not only improve access to clean water, but could also support new fisheries. Landscape restoration can also be a very efficient tool for fire prevention and control, preventing the many negative social costs of fire (Chapter 19).

Restoration could also have important political consequences, although these remain understudied, especially in developing countries (Blignaut *et al.* 2013). Many Amazonian countries have included restoration as part of their NDC commitment to the Paris Agreement, and several Amazonian countries (Peru, Bolivia, Ecuador, Brazil) have made commitments for restoration through programmes such as Initiative 20x20. Ecological restoration, like all political initiatives, needs to be placed within the context of policies, negotiating over tradeoffs between competing objectives and different constituencies, and the distribution of scarce resources among diverse societal spheres (see Meadowcroft 2009). As a result, governance and institutional framework becomes significant. Viewed from such a perspective, negotiations can then develop around what types of restoration projects are to be implemented and where, and who manages the land afterwards (see Chazdon et al 2020). Restoration is likely to be important in this context as it influences many aspects of wellbeing targeted by political decision makers, from products that harvested from the restored ecosystems, health benefits such as water quality or changes in exposure to air pollution or high temperatures, the reduced exposure to natural disasters such as flooding, or the improvements in wellbeing from improved access to natural systems.

Recovering landscapes also generates additional value such as soil and water protection, microclimate regulation and goods. This change in political economic value of the landscape may generate new interests, which could potentially shift the balance of power, conflicts, and use of natural resources, as well as improving inequalities and land tenure rights (Ding *et al.* 2017). Expanding restoration beyond site or project level, to the landscape scale requires an increase in the number of stakeholders, thereby adding further complexity to governance that rarely corresponds to any single political unit. Such spatialization of governance is not about creating new layers for political and administrative structures, but about identifying new institutional domains for stakeholders to meet, negotiate and co-create the necessary conditions for restoration (van Oosten *et al.* 2021). It also ensures that governments uphold important constitutional responsibilities related to environmental protection and accessibility (see the Atrato River legal case in Colombia). Incorporating these benefits into political decision making could help garner support for the implementation of restoration across the basin.

3. LANDSCAPE AND CATCHMENT APPROACHES TO RESTORATION AND CONSERVATION

Once a region has been identified as a priority for restoration, landscape and catchment approaches can help ensure that the restoration actions are more effective and deliver the greatest benefits to the broadest range of stakeholders. The scale of these interventions varies, but is broadly as "*a contiguous area, intermediate in size between an 'ecoregion' and a 'site', with a specific set of ecological, cultural and socioeconomic characteristics, distinct from those of the neighbouring areas* (WWF 2004)".

Within the region of interest, landscape approaches aim to "*provide tools and concepts for allocating and managing land to achieve social, economic, and environmental objectives in areas where agriculture, mining, and other productive land uses compete with environmental and biodiversity goal*" (Sayer *et al.* 2013). They have become redefined as "*integrated landscape approaches*", reflecting the need to reconcile multiple and conflicting land-use claims and help establish multi-functional landscapes (Reed *et al.* 2016). The term now encompasses a wide-range of approaches (Reed *et al.* 2016), including aquatic approaches such as integrated watershed management (e.g. Shiferaw et al 2008). Restoration specific approaches include Forest Landscape Restoration (Ianni 2010) which is now promoted by many leading environmental NGOs, and developed by FAO into the Forest and Landscape Restoration Mechanism that aims to "*restore degraded landscapes by identifying and implementing practices that restore a balance of the ecological, social and economic benefits of forests and trees within a broader pattern of land uses*". The broad approach of FLR enables decision makers to consider all components of a landscape, from agriculture to restoration and forestry, and support long-term sustainability decisions through economic zoning (Celentano *et al.* 2017). They also call for a consideration of all ecosystems within a region, supporting restoration beyond *terra firme* forests, including savanna enclaves and flooded forests (Chazdon *et al.* 2020b; Ota *et al.* 2020; César *et al.* 2021).

Considering where and how to restore at the catchment or landscape scale can help return much higher benefits than simple site-based approaches. We outline some of the key benefits of planning restoration within landscapes and catchments.

3.1. Integrating aquatic and terrestrial systems

Terrestrial and aquatic systems are often considered separately but are inextricably linked. Moreover, considering them together can provide large benefits for aquatic biodiversity at no cost to terrestrial biodiversity (Leal *et al.* 2020a). It has long been established that riparian zones can act as buffers for sediment and nutrient retention (Peterjohn and Correll 1984; Allan 2004; Saad *et al.* 2018; Luke *et al.* 2019), can moderate extremes in stream water temperatures (Macedo *et al.* 2013), and are important for biodiversity in both streams and floodplains systems (Arantes *et al.* 2019; Dala-Corte *et al.* 2020). In southeast Brazil, modeling efforts using InVEST have explored different riparian restoration strategies that can reduce soil loss and river sediment export by filtering sediments before they reach streams (Saad *et al.* 2018). Even in highly modified agricultural landscapes, the condition of riparian zones can strongly influence stream water quality via nutrient retention. For example, research in the Brazilian Amazon-Cerrado frontier in the state of Mato Grosso highlights the capacity of functionally-diverse riparian vegetation to capture and sequester nutrients (Nóbrega *et al.* 2020). Concentrations of nutrients (organic carbon, total nitrogen, phosphorus, calcium, and potassium) in overland flow from croplands are substantially greater than from nearby riparian gallery forest. Moreover, soils from intact gallery forest, especially those with biodiverse plant assemblages with varied root systems, display properties that better enable nutrient uptake, as well the degradation of nutrients and pollutants as compounds travel through hyporheic zones. With respect to stream temperature, a study of 12 catchments in the upper Xingu watershed reported warmer water temperatures in streams from pasture and soya-dominated catchments, with daily maxima 3-4^{o}C higher than in forested catchments (Macedo *et al.* 2013). Collectively, these studies provide rationale for placing a premium on gallery forest and riparian zone restoration, for mitigating impacts on sediment export, water chemistry and thermal regimes.

Chapter 29

Source water protection involves a suite of management practices to protect water quality and quantity, especially in the context of water supplies for urban areas (Abell *et al.* 2019). When coupled with strategic land protection in targeted catchments, restoration can play an important role in source water protection, via activities such as revegetation, riparian restoration, livestock exclusion, and wetland restoration. Source water protection is an actively promoted restoration strategy in parts of the Amazonian Andes to improve water quality and preserve biodiversity (Bottazzi *et al.* 2018). In the Bolivian Andes, a program known as *Watershared* is a Payment for Ecosystem Services effort that pays farmers and cattle owners to prevent forest conversion and exclude livestock from riparian forest, all predicated on the notion that improving the condition of riparian zones translates into tangible outcomes for water quality and quantity. Contamination of drinking water by the bacterium *E. coli* is of particular concern where livestock graze freely in streams and fencing has been shown to be a successful strategy for reducing per capita human cases of diarrhea by preventing livestock intrusion (Abell *et al.* 2017). Similar practices of livestock removal coupled with riparian revegetation have been implemented elsewhere in the highlands of the tropical Andes to improve water quality and supply for urban areas (Goldman *et al.* 2010; Higgins and Zimmerling 2013).

In addition to water quality, land use modifies the magnitude and variability in river flows. Although studies have evaluated changes in river discharge with deforestation and the conversion of land to intensive agriculture in Amazon catchments (Hayhoe *et al.* 2011; Davidson *et al.* 2012; Dias *et al.* 2015; Farinosi *et al.* 2019), there have been few attempts to track stream flow responses to terrestrial restoration and afforestation. A systematic review of more than 300 study cases worldwide examining impacts of forest restoration on stream flows found nearly two-thirds of cases reported decreases in base flows, and an even greater proportion, nearly 80%, observed decreases in annual water yields (Filoso *et al.* 2017). Likewise, in an analysis of temporal trends in river flow responses following forest re-establishment, annual flows often displayed substantial reductions even after 25 years and were especially pronounced in catchments with high annual precipitation (Bentley and Coomes 2020). In an experimental study of hydrological response to land use and afforestation in the Ecuadorian páramo highlands, water balance and flow duration curves were compared among four small headwater catchments (Buytaert *et al.* 2007), including one afforested with pine (*Pinus patula*), a catchment with intensive livestock grazing and potato cultivation, and two catchments with intact páramo vegetation. Flow regimes were dramatically modified in the afforested catchment, with severe reductions in base and peak flows. Although the cultivated catchment also displayed altered flows, they were less drastic than observed in the catchment with planted pines. These results suggest that in the Andean highlands, afforestation by non-native tree species used to reduce hillside erosion could result in significant decreases in base flows and compromise water supply.

While the studies to date suggest forest restoration may reduce streamflow, this is just one of the many important response variables, and it is unlikely that results from temperate or arid zones are applicable in the humid tropics. For example, study in Madagascar shows how forest restoration can reduce erosion and flooding related to overland flow (van Meerveld *et al.* 2021). In a study in the Phillipines, increased infiltration in forest restoration more than offset the reductions in water balance from evapotranspiration, leading to a net positive trade-off benefit on streamflow (Zhang *et al.* 2019). Forest restoration could also support streamflow if it reduces landscape temperatures and increases rainfall (see Sections XX).

3.2. Improving landscape and catchment connectivity for biodiversity

Island biogeography theory has underpinned the discipline of landscape ecology, guiding much of the theoretical and empirical evidence on the outcomes of habitat fragmentation. There are long-running debates about the relative importance of habitat extent versus habitat fragmentation (changes in landscape configuration without changing habitat extent) (e.g. Fletcher *et al.* 2018; Fahrig *et al.* 2019), but a growing consensus recognizes that while habitat extent is the most important factor, the configuration of that habitat also matters for species across the world (Arroyo-Rodríguez *et al.* 2020). Crucially, tropical species are inherently more sensitive to fragmentation than temperate species (Betts et al. 2019). For example, many Neotropical understory birds have a limited capacity to fly more than a few tens of meters (Moore *et al.* 2008) and are reluctant to cross even small roads (Lees and Peres 2009), making them highly susceptible to human activities that fragment habitat into discrete patches (Ferraz *et al.* 2003; Lees and Peres 2006). Freshwater species are also susceptible to changes in connectivity (Hurd *et al.* 2016), and the Amazon's migratory catfish show the most spatially expansive metapopulations of all the world's freshwater fish (Hurd et al. 2016) The low dispersal ability is evident over evolutionary time scales, as rivers have been major barriers in the evolution of Amazonia's diversity (Chapter 3).

Given the high sensitivity of many Amazonian species to habitat fragmentation, restoration will be most effective if is deployed in a way that both increases habitat *and* maintains or enhances connectivity between remnant forest patches or in rivers to ensuring migration can take place and gene flow is permitted between populations. Mixed suites of restoration strategies can better improve connectivity between higher quality patches. Vegetation restoration efforts can create corridors that encourage movement between the last remaining habitat patches – such strategies have proven successful for increasing the population size and reducing the threat status for species such as the Black lion tamarin (*Leontopithecus chrysopygus*) in the Atlantic Forest. Similar approaches would support conservation efforts for some of the Critically Endangered species in the most deforested regions of the Amazon (Figure 1), including in the Maranhão-Pará border, Rondônia and in the Andean regions. However, enhancing connectivity in these regions will only be effective if carried out in conjunction with complementary conservation measures that protect the last remaining populations and habitats for these species (Chapter 27).

For some species, connectivity can be enhanced without physically connecting the disjunct patches. For example, high quality habitat will be functionally connected if species are able to cross the non-habitat "matrix" in between them (e.g. Lees and Peres, 2009). This permeability of the agricultural matrix could be influenced by restoration: cattle pastures and mechanized agriculture represent very hostile matrices, and even occasional trees (e.g. Rossi et al. 2016) or agroforestry could enable some forest species to disperse across the landscape, linking previously isolated faunal populations and facilitating seed dispersal and pollination for plants. Connectivity across the landscape – and benefits for aquatic systems – could also be enhanced by restoring a full network of riparian vegetation (Rossi, Jacques Garcia Alain Roques and Rousselet 2016; Kremen and Merenlender 2018).

3.3. Local climate benefits

Forest cover is known to influence local climate, reducing regional temperatures and maintaining rainfall (WG 3). Restoration in deforested regions could provide important benefits for local and regional climate (Mendes and Prevedello 2020). For example, studies across the world show that it can help reduce the effect of urban heat islands if conducted around cities (Bhagwat *et al.* 2008), and can reduce the occurrence of excessive stream temperatures (Hall *et al.* 2020). There is also some evidence that the configuration of forest cover in a landscape could influence climatic benefits of restoration, with more fragmented patterns actually increasing rainfall and maximising the reductions in land surface temperature (Mendes and Prevedello 2020). However, there is uncertainty about how this occurs at scale; one modelling study suggests that rainfall increases on agricultural land and decreases on the forests themselves (Garcia-Carreras and Parker 2011), which could increase forest flammability and enhance drought sensitivity. Furthermore, while a fragmented configuration may reduce the temperature of the deforested area, it is also likely to increase understory temperatures in the remaining forests, contributing to faster drying and increasing flammability. The local climatic benefits of restoring forests in a particular configuration require further research.

3.4. Reducing the risk of socio-environmental disasters

Landscape or catchment level restoration can be used to reduce the risk of events that are detrimental to Amazonia's people and nature. Forest fires and wildfires in open areas are a growing threat to Amazonia (WG8), and targeted restoration could be part of the solution. By influencing temperature and humidity (section 2.3), landscape-level restoration could make fuels on the forest floor less flammable by increasing humidity and reducing temperatures. Restoration could also be used to 'buffer' primary forest edges by creating green firebreaks; although we are not aware of any research into this, we believe such firebreaks could have two complementary roles. First, primary forest edges are drier and hotter than forest interiors, which contributes to them being frequently degraded due by fire incursion (Silva Junior *et al.* 2020); the restoration of closed canopy vegetation alongside primary forests would help buffer those forests edges from the hot microclimate of the agricultural matrix, making them less flammable, and could also help suppress pyrophytic grasses that help spread fires. Second, restoration alongside primary forests would help isolate those forests from the wider landscapes where ignition sources are most prevalent. While the use of these 'green firebreaks' remain untested in an Amazonian context, the 'Green Hug' project (Abraço Verde) in the Atlantic forest provides an interesting insight into the long-term viability of projects using agroforestry buffers to project forest edges (Chazdon *et al.* 2020a). Research is needed to evaluate the effectiveness of green firebreaks in the Amazon, including understanding the ideal widths and what active restoration measures (tree planting or enrichment) are required to maximise other benefits (e.g. economic returns). It will also be important to and minimise the risks that the restored areas such as sexcondary forests could themselves become 'wicks', helping conduct fire across the landscape (e.g. (Ray *et al.* 2005).

Catchment-scale restoration can also help mitigate the risk of flooding, which are exacerbated by deforestation (Bradshaw *et al.* 2007). Evidence from China suggest broadleaf trees are especially effective (Tembata et al 2020), casting doubt on the flood mitigation value of oil palm or other species that are planted at low densities. Models suggest that sub-catchment restoration of riparian forests is likely to be one of the most effective mechanisms to reduce flooding, with restoration across 10-15% of the catchment reducing the peak magnitude of flooding by 6% after 25 years (Dixon *et al.* 2016).

3.5. Meeting multiple aims and optimizing benefits

The landscape or catchment scale is often considered the most appropriate to consider different land use and ecosystem functions, meet different needs and consider trade-offs, and achieve multiple benefits (Reed *et al.* 2016).First, going beyond site-specific management and planning at the landscape or catchment level allows restoration to use optimization techniques to quantify the trade-offs or complementarity between various restoration targets. Such approaches are helping prioritize restoration across the world (Strassburg *et al.* 2020), and could allow restoration actions to achieve a broader range of benefits whilst minimizing the losses of each one (Stanturf *et al.* 2015). For example, within Amazonia, optimization has shown the complementarity between biodiversity and carbon storage targets, highlighting the large gains that can be made for biodiversity conservation with very minor reductions in carbon storage (Ferreira *et al.* 2018).

With so many potential co-benefits of restoration, it is vital that these are considered as part of an integrated planning process with full consideration of landscape and catchment processes (Reed *et al.* 2019). For example, peri-urban restoration aimed at providing climatic benefits for cities could also provide import social benefits if the species provide fruits or other products that are consumed locally. Similarly, restoration aimed at terrestrial conservation could also support aquatic biodiversity, without any cost to terrestrial conservation objectives (Leal *et al.* 2020a).

Planning beyond specific sites also allows restoration to consider and compare the relative benefits of a full suite of interventions, helping ensure efforts are invested in the most effective measures. For example, landscape-scale planning is essential to decide when and where to adopt active or passive restoration of secondary forests, or whether strategies should target reforestation or focus on alternative measures such as avoiding degradation of existing forests or economic recovery in degraded lands. Work on restoration options in the eastern Amazon is addressing this, and shows that avoiding degradation in existing forests can be a cost effective approach to conserving carbon and biodiversity.

4. ENCOURAGING A BROADER FOREST TRANSITION

Forest loss and gain across the Amazon can be seen in terms of a forest transition. The term forest transition, introduced by Mather (1992), refers to a change in forest cover (either shrinkage or expansion) over a given area (landscape, regional, national level) and time period. This process typically shows three main periods. First comes a phase of intensive deforestation due to forest conversion into agricultural lands and pastures, followed by a net gain of forest area through reforestation and restoration actions as well as passive natural regeneration. The third and last phase is a stabilization phase with a constant forest cover area. Europe and North America, and more recently some tropical countries, have already gone through their forest transition and have registered constant forest cover increase since (Mather 1992) (Meyfroidt and Lambin, 2010).

In most countries where a forest transition has occurred, the new forests are very different in structure, composition, and function. While generalist species can benefit, these new forests do not provide conservation benefits for the specialist species restricted to old-growth systems (Wilson *et al.* 2017; Lees *et al.* 2020)). Moreover, evaluations of forest transitions require an understanding of global trade and leakage. Improved environmental performance and expanded forest cover in more developed countries may have come at the cost of environmental destruction elsewhere, typically in the Global South (Lees et al. 2020). This can also occur within regions and ecosystems; within an Amazonian context, care needs to be taken to ensure conservation and restoration activities in one area do not simply push social and environmental pressures elsewhere, including from one region of the Amazon to another, or from the Amazon to other ecosystems (e.g. de Waroux *et al.* 2016).

While a forest transition may occur naturally in the Amazon, it is not yet the case as the most deforested regions of the basin have failed to see an increase in forest cover since 1997 (Smith *et al.* 2021). However, actions that avoid the loss and stimulate the gain are critical for the basin as a whole: the Amazon forest generates approximately one third of its own rainfall (Staal *et al.* 2018), and excessive deforestation could have huge environmental consequences, particularly on precipitation regimes and consequently on the capacity of the remaining forest to survive (Nobre *et al.* 1991; Oyama and Nobre 2003; Hutyra *et al.* 2005; Sampaio *et al.* 2007), with tipping point estimates ranging from 20-25% (Lovejoy and Nobre 2018) to 40% deforestation (Sampaio *et al.* 2007). Furthermore, if deforestation goes beyond these estimated thresholds, natural reforestation itself could also be hampered by unfavorable climatic conditions (e.g. Elias *et al.* 2020).

How can restoration mitigate the loss stage of the Amazon's forest transition? One way that it could help is if it was partly oriented towards timber production, which would release the pressure on natural forests which are still the main provider of timbers in the region. During the last 50 years of recent colonization of the Amazon, natural forests have been selectively logged and 108 Mha of forest (20% of the total forest area) are exploited for timber production (Food and Agriculture Organization of the United Nations and International Tropical Timber Organization 2011).

There are many reasons why it would be beneficial to replace timber production from natural forests with timber plantations on degraded land. First, although sustainable forest management practices are considered a potential tool for Amazonian forest conservation (Putz *et al.* 2008; Edwards *et al.* 2014) and provide income and employment (Putz *et al.* 2012), natural timber production itself is unsustainable under present-day conditions of logging intensities and rotation cycle duration (Sist et al. 2021). In the Amazon, selective logging regulations typically set a rotation cycle of 20 to 35 years with a logging intensity varying from 15 to 30m^3/ha (Sist et al. 2021). Several studies showed that under such extraction regimes, less than 50 % of the timber extracted can recover within this rotation duration (Schulze 2003; Sist and Ferreira 2007; Putz *et al.* 2012). A recent study simulating the timber recovery in all the region confirmed this result and showed that even under long rotation length of 65 years and a logging intensity of 20m^3/ha, the timber recovery would be only 70% (Piponiot *et al.* 2019a). This means that under the present logging regulations natural Amazonian forests alone will not be able to supply the timber market demand in the long term (i.e. as soon as in the second rotation, 30 years from now). Second, timber in natural forests generates low profits when carried out using best practices (Putz *et al.* 2008). Third, while it is much better than non-forest land uses for conservation and carbon storage, most logging practices in the Amazon continue to be illegal (Brancalion *et al.* 2018) and these generate high damage to the stand. Such practices open up the forests, making them more accessible to hunting (Peres 2001) and more vulnerable to forest fires (Holdsworth and Uhl 1997). Finally illegal logging continue to threaten the financial profitability of improved tropical forest management.

If reforestation was used to meet some of the demand for timber, it could decrease the pressure on natural production forests – allowing larger areas to be set aside for conservation and allowing lower-intensity management of production areas. It will also allow promotion to specific markets and uses of timbers coming from natural forests with higher prices than timbers from plantations. This new market for timber extracted from natural forests should take into account the specific wood property of old natural timber, the costs of sustainable forest management practices and finally the environmental services provided by well managed natural forests. Selective logging could be sustainable if it adopted much longer cutting cycles (65 years), and by reducing logging intensities ($10m^3$/ha instad of $20m^3$/ha presently) and incidental damage to the stand through reduced-impact techniques (Piponiot *et al.* 2019b; Sist *et al.* 2021). Although post-logging silvicultural interventions (e.g. liana cutting, future crop tree liberation, and enrichment planting) could be promoted to improve commercial yields from merchantable species, such practices might not be sufficient to meet rising demands for wood products on a long-term basis. Additional sources of timber such as plantations of exotic or native species, enriched secondary or degraded forests, integrated crop-livestock-forestry systems and other agroforestry systems could be implemented within forest restoration programs under the Bonn initiative (Lamb *et al.* 2005). The rising interest in tropical forest restoration, crystallized by the Bonn challenge in 2011, is a unique opportunity to initiate this forest transition by encouraging restoration of economically viable timber plantations in deforested areas of the Amazon and promoting the management of secondary regrowing forests in abandoned agriculture lands (Ngo Bieng *et al.* 2021). However, the success of any forest transition program depends primarily on forest law enforcement fighting against illegal logging and promoting sustainable silvicultural practices.

The theory of forest transitions focuses on the terrestrial part of a landscape, but what would an aquatic transition look like? Within the Amazon, avoiding the worst outcomes for aquatic systems will require preventing the most damaging new dams from being built, preventing land-use change, and regulating the use of harmful agrochemicals – all of which could be supported by alternative energy sources, novel bio-economies, and the encouragement of better agricultural practices. Within the aquatic zone itself, as discussed in chapters 20 and 30, overfishing might be mitigated by implementing, encouraging and strengthening co-management systems over large regions. Improving the status of fish populations would also benefit some key ecological processes in floodplain systems, as some of the species that have been declining with harvesting pressure such as tambaqui (Tregidgo *et al.* 2017) provide important ecosystem processes (Costa-Pereira *et al.* 2018).

Aquaculture also provides a complementary way to reduce pressure under fishing resources. Sustainable fishing and aquaculture practices could also reduce demand for protein such as beef, that requires orders of magnitude more land per kilo of protein, even when inputs are considered (Piva Da Silva 2017). Despite the potential of aquatic to contribute to curb overharvesting problems, many issues still require further analysis and investigation, as for example whether supplying farmed fish will relieve pressure on wild fish stocks, or if the many risks of Aquaculture (increased nutrient loads, risks on species introduction, increased demand on natural fish populations or crops as food sources for produced fish) can be managed.

5. ENSURING BROADER SOCIETAL BENEFITS FROM RESTORATION

Restoration exists within a social context, and therefore produces environmental conditions that must not only be ecologically sound but also economically feasible and socially acceptable. Whilst the primary aim of restoration is environmental, it is guided by cultural expectations and values which determine both the goals set for restoration and whether projects are judged to be successful (de Bell *et al.* 2020); and can be determined by people's perception of the landscape as places of traditions, memories, myths and stories. Restoration can, therefore, result in people feeling either alienated or unconnected to a once-familiar landscape. If applied to ecosystem restoration in Amazonia, it implies that there is some inherent worth in protecting and restoring nature.

A recent study showed that nearly 300 million people in the tropics live on lands suitable for forest restoration, and about a billion people live within 8 kms of such lands (Erbaugh *et al.* 2020). Many of these people live in poverty. Given the implicit challenges to restoring complex and stochastic systems, landscape restoration has considerable potential to include local populations and improve local livelihoods over the long term (Palmer *et al.* 2005; Reed 2008; Lee and Hancock 2011; Erbaugh *et al.* 2020). Through engaging a diverse range of stakeholders from public, private and civil society sectors, and building and sustaining such coalitions of support restoration is imperative. When done in this way, restoration can increase well-being through the sale of forest products, increases in food supplies, improved water security, and the promotion of the diverse cultural values people place on landscapes (Aronson and Alexander 2013; Sabogal *et al.* 2015; Brancalion and Chazdon 2017; Stanturf *et al.* 2019). In most cases this requires thinking beyond the individual site being restored and taking into account the broader benefits at the landscape scale: it is well documented that the success of FLR requires the empowerment and capacity building of local communities and their engagement in the decision-making process. For example, in Colombia restoration is helping to generate income in the long term, allowing communities and individuals to plan ahead in terms of future investments, and has been advocating messages such as 'trees planted today will generate income to pay for the education of their children'.

Along with the climate and biodiversity benefits from restored landscapes, restoration can influence tenure likelihood, feasibility and likely success of restoration efforts. In such scenarios, conflicting tenure regimes and property rights may complicate matters, especially if there are multiple landowners, public and/or private. While tenure security is not essential for restoration, tenure insecurity has been cited as a disincentive to invest in restoration (Fortmann and Bruce, 1991; Cotula and Mayers, 2009). Equally, landscape restoration may in turn affect tenure and land rights positively for many local and indigenous communities and landowners, where returning vegetation to the land may entitle them to legal tenure. It may also increase family incomes, employment opportunities, and community resilience (Adams *et al.* 2016; Erbaugh and Oldekop 2018). Though outside the scope of this study, reforested lands within the Brazilian Atlantic Forest have created over 200 jobs related to native seed collection, seedling production, planting, maintenance, and downstream manufacturing of timber and non-timber products (Calmon *et al.* 2011), as well as emerging opportunities from biodiversity-derived innovations. Using native species to restore a landscape is often considered a good value judgement. Therefore, restoration is not only an effort in technology, but also infused with value considerations, and similar restoration potential can be realised throughout Amazonia.

Regaining land title and authority over restored lands also has health benefits for many marginalized and indigenous peoples. Wellbeing encompasses much more than economic solvency and indicators of health can be categorized as material (food, water, shelter, security), social (identity, belonging and self-esteem) and spiritual/cultural benefits (related to sacred places, totemic animals and artefacts, beliefs, customs and languages (Hiemstra et al, 2014). Additionally, pollution often affects people's health, and remediation and restoration efforts need to consider a broad approach that includes physical and mental well-being. This is particularly relevant for oil and mining pollution, which have had direct effects on indigenous and marginalized communities in the Amazon (see Chapter 20). It is vital that the full social and ecological costs of mining are factored into decisions about where and when it takes place.

Restoring degraded landscapes can also offer a means to restore landscape connectivity, rebuild communities and decentralize governmental institutions. For instance, about 6000 indigenous people reside in the Xingu Indigenous Park in Brazil, with other communities inhabiting the heart of the basin in downstream extractive reserves of the Terra do Meio which have been negatively affected by changes in the quantity and quality of water that enters their lands (Schwartzman *et al.* 2013). Through restoration of the riparian forests around the springs and along all water courses draining into the Xingu Indigenous Park, run-off from crops and pastures were prevented from contaminating water bodies (Sanches et al., 2012; Schiesari et al., 2013), with a recovery of 50 km^2 of forests within the Xingu River Headwaters (Schmidt *et al.* 2019).

Box 1. the Xingu Seed Network as a social-ecological collaboration.

In the above example, to reduce restoration costs, collective action involving private landowners and local and indigenous communities was galvanised Futemma et al., 2002; Tucker, 2010; Urzedo et al., 2016; Schmidt et al., 2019). This is important as many governmental officials do not always appreciate the full extent of the importance of landscape to local and indigenous communities in terms of food security, income, nutrition, employment, energy sources and wellbeing. The principle of social involvement in restoration led to the creation of the Xingu Seed Network involving seed collection using traditional knowledge and promotes forest economy by generating income. This initiative encompasses 568 collectors from 16 municipalities of Mato Grosso state, distributed in 20 indigenous villages and 14 agrarian reform settlements, totaling 2564 ha under restoration with more than 300 landowners involved, generating US$380,000 (Durigan et al 2013, *Urzedo et al., 2016; Schmidt et al., 2019). By restoring degraded landscapes, it provides new opportunities to build relationships between private landowners and communities, and/or between communities and governments, based on collaboration rather than confrontation. While such progress is often slow at the landscape level, partly due to entrenched attitudes in bureaucracy, it offers significant potential shifts in attitudes and working relationships that can lead to evolution in socioecological policies (WG11)*

6. THE CLIMATE RESILIENCE OF RESTORATION OPTIONS

Restoring ecosystems in the context of climate change requires understanding when it is best to rebuild past ecosystems, and when it is better to attempt to build resilient ecosystems for the future(Harris *et al.* 2006). Determining where historical baseline targets are viable and where alternative targets must be considered is site-dependent and associated with projected changes(Jackson and Hobbs 2009). We consider these issues in terrestrial and aquatic systems.

6.1. Climate resilience of terrestrial restoration

Amazon's primary forests are affected by climate change and climatic extremes, resulting in increased mortality of individual trees (Phillips *et al.* 2009; McDowell *et al.* 2018), and changes in species composition(Esquivel-Muelbert *et al.* 2019) (see also Chapter 23), and several studies show strong associations between tree mortality and climate changes such as the increased intensity and the duration of dry season (Aleixo et al., 2019a; Adams et al. 2017) and warmer temperatures (Sullivan et al., 2020; Allen et al. 2010). But what about the sensitivity of secondary forests? Here we outline five lines of evidence suggesting they may be particularly sensitive to climatic change.

The first is spatial; secondary forests may be especially vulnerable to ongoing climate change as they are mostly situated in the drier and more seasonal parts of the Amazon where deforestation has predominated (Smith et al;. 2020). The second is physiological, secondary forests are dominated by fast-growing trees with low wood densities (Berenguer *et al.* 2018) or have large thin leaves that do not conserve water, and these may be especially vulnerable to drought by cavitation or carbon starvation (Phillips *et al.* 2009;

McDowell *et al.* 2018; Aleixo *et al.* 2019b). The third line of evidence is empirical; secondary forests monitored over time have been shown to have significantly lower rates of carbon accumulation during drier periods (Elias et al. 2020). This is in part driven by mortality: several studies in primary and secondary forests recorded higher tree mortality after global extreme climatic events associated with El Niño/la Niña Atlantic oscillation (NAO) in the Amazon in 2005 and 2016 (Chazdon et al., 2005; Philips et al. 2009; Leitold et al., 2018). However, in secondary forests it is also driven by reduced growth (Elias et al. 2020). The fourth reasons relates to their structure and microclimate; the low canopies and the high rates of stem turnover in secondary forests mean they have higher understory temperatures and lower humidity levels (Ray *et al.* 2005), making them more vulnerable to extreme climate conditions as well as to fire events (Uriarte *et al.* 2016). Finally, while many primary forest trees are known to have a very deep rooting depth (Nepstad), this seems less likely in secondary forests where average stem sizes are much lower. It is notable that seedlings are more vulnerable to drought in disturbed forests in Borneo, and that droughts also push the community composition back towards ruderal pioneers (Qie *et al.* 2019).

This heightened sensitivity to climate change could be offset if existing gradients in dry season intensity and rainfall drive adaptations to greater drought or heat sensitivity.. First, it may be that existing gradients in dry season intensity and rainfall have led to species adaptations that infer greater drought or heat sensitivity in the drier and hotter parts of the Amazon. Primary forests are responding to climate change through changes in species composition (Esquivel-Muelbert et al. 2018), and the fast turnover of secondary forests and the high dispersal capacity of pioneer species may facilitate these changes in secondary forests that are functionally connected to the rest of the forest in the region. As such, one possibility is that drought-resilient secondary forests emerge in the future, perhaps resembling the species composition and successional trajectories found in regenerating tropical dry forests where the initial stages of forest succession are dominated by species with drought tolerant traits (e.g. (Lohbeck *et al.* 2013). Where forests are unable to change naturally, or where a faster rate of change is desired, then enrichment planting could help encourage species with traits that are better adapted to heat stress of longer dry seasons. The cutting of climbers and liberation thinning could provide additional support (Philipson *et al.* 2020), although evidence from Borneo suggests that the benefits of liana cutting may be reduced during extreme droughts (O'Brien et al. 2019). Finally, restoration at the landscape scale could help maintain a cooler and more humid regional climate (see section 23)

Drought is not the only cause of mortality. Aleixo et al. (2019) showed that trees died more often during wet months than in drought years, and rain and storms that occur during the transition from dry to wet seasons in the Amazon might be the main cause of tree mortality during the wettest months (Negrón-Juárez *et al.* 2010). Forest restoration is also highly susceptible to fire, which can arrest successional processes in terra firme (e.g. (Berenguer *et al.* 2018; Heinrich *et al.* 2021) and flooded forests (Flores *et al.* 2017). As large-scale forest fires are strongly linked to extreme droughts (Chapter19), forest restoration activities need to be aligned with actions that reduce landscape flammability and support for farmers that enable them to control ignition sources.

6.2. Climate resilience of aquatic restoration

Hydrological effects of climate change are likely to more strongly impact Amazonia than other regions of South America (Brêda *et al.* 2020). Notably, uncertainties in restoration linked to climate change may be exacerbated by land use changes. This is especially true in the Amazon, where climate change interacts with deforestation in particularly strong ways. As an example, coupled climatic and hydrologic models forced under contrasting deforestation scenarios suggest that the consideration of climate responses to deforestation shift precipitation outcomes from mean positive to mean negative changes (Lima *et al.* 2014). In addition, deforestation can increase the duration of dry seasons and magnify seasonal amplitudes in discharge. Importantly, water balance changes are not confined to deforested sub-basins, as atmospheric circulation spreads the effects basinwide (Coe *et al.* 2009).

Changes in water balance associated with climate change and deforestation will likely affect floodplain and river ecosystems in many ways (see also Chapter 23). Decreased mean annual rainfall (Brêda *et al.* 2020) combined with increased frequency of extreme weather events in the Amazon (Marengo 2009) will change seasonal inundation patterns, impacting species composition and biogeochemical cycling in Amazonian freshwater landscapes.

Reduced inundation length can alter the selection for flood-tolerant species and ultimately the composition of floodplain forests; moreover, because floodplain trees generally lack traits linked to fire and drought resistance, the frequency, severity and ecological impacts of fires may exacerbate (Castello and Macedo 2016). In rivers, precipitation and discharge regimes regulate sediment transport and aquatic nutrient dynamics (Devol *et al.* 1995; Almeida *et al.* 2015, and flood extent governs the input and processing of vast quantities of organic matter produced in terrestrial and seasonally flooded ecosystems that is further outgassed as carbon gas (Abril *et al.* 2014; Almeida *et al.* 2017). In the biological realm, altered seasonality in flood regimes could affect plankton community interactions, with potentially cascading food web effects (Feitosa *et al.* 2019). Thus, in addition to understanding site-level conditions prior to disturbance, effective restoration of Amazon aquatic ecosystems should be attentive to watershed-scale hydrological, biological, and chemical alterations brought about by climate change.

7. ACHIEVING MEANINGFUL RESTORATION AT SCALE

Restoration science has developed rapidly over recent decades, and while some knowledge gaps remain in the tropics, it has reached a point where it can provide clear evidence-based guidance to support restoration in a wide range of contexts (Chapter 28) and across whole biomes and landscapes (this chapter). Restoration cannot happen in isolation – we have outlined here how it must be linked by a broader suite of conservation measures that avoid further loss (Section 26.1). Crucially, research has shown it needs to be integrated with society and political context, and can inform how to do it in a way that is as inclusive as possible of all people in a landscape (while recognizing that not all stakeholders will necessarily benefit) (Reed et al. 2018). But how can this knowledge be used effectively? Here we examine the policy levers and incentives that can support the large-scale restoration that is required to mitigate climate change, avoid dangerous tipping points, reduce pressure on primary forests, support local livelihoods, and develop a thriving and flourishing Amazonian bioeconomy.

7.1. Enforcement and Monitoring

Many have experimented with technological and organizational solutions to restore environmentally-sensitive and sustainable economic production[e.g.](Brondizio *et al.* 2021). Yet, they will not be taken to scale as long as the negative externalities of exploiting the forest's natural capital go unaccounted for. For example, the low market prices of illegal timber undermine the value of legal timber (Brancalion *et al.* 2018), making it much more challenging for companies that follow legal and certification scheme practices to fund the monitoring and enforcement required to ensure post-harvest forest integrity across expansive and remote concessions (see Chapters 14, 19 and 29). Countering this requires changes in policy and governance (laws, taxes, subsidies) to make illegal logging economically unattractive. Green financial institutions are key partners to invest in land and landscape restoration, which requires efficient tools to monitor and verify environmental performance at plot, farm, and landscape levels. Monitoring and enforcement is also key to avoiding perverse effects of economic restoration, where technologies and policies promoting greater agricultural or silvicultural productivity paradoxically lead to increased deforestation(Garrett *et al.* 2018), or where large-scale ecological restoration causes "leakage" of environmental harm [e.g.](Alix-Garcia and Gibbs 2017).

7.2. Incentive-based measures

Restoration can be incentivized by carbon and/or biodiversity offsetting, payments for ecosystem services (PES), and/or certification schemes. Yet, PES often fail in gaining scale(Coudel *et al.* 2015), and such market-based interventions can generate conflict and weaken social ties(Pokorny *et al.* 2012). Interestingly, less obvious policies may have important indirect effects on restoration dynamics, such as the Brazilian School Meal Program that has been fundamental in encouraging the consolidation of agroforestry systems and agrobiodiversity in some areas of the eastern Amazon(L. Resque *et al.* 2019).

7.3. Community-led restoration

Some site-level restoration actions can be implemented by liaising with a relatively small set of stakeholders, such as property owners or reserve managers. Yet, to achieve sustainable transformations across landscapes and catchments, it is vital that the restoration measures are viewed favorably by the people who live in or benefit from the economic activities. For example, implementing integrated farming systems on unproductive farmland requires the participation of all relevant stakeholders, both in the design and implementation of the research and extension programs to assure they meet the socioeconomic needs and the cultural values of the beneficiaries. This ensures that research and extension programs meet the socioeconomic needs and cultural values of intended beneficiaries (Leal *et al.* 2020b). Unsurprisingly, some of the most successful examples of active restoration involve strong community buy in and leadership. The Rede Sementes do Xingu & community-led fisheries restoration and management programs provide positive examples of community engagement and leadership. The success of restoration initiatives involving local people will be highly dependent on effective and long-term support for capacity building and technical assistance, and ongoing and wide-ranging social collaboration and participation (Chapter 30).

7.4. Policies

Restoration can also be supported at the national level through official commitments and legislation. For example, the Brazilian (NVPL, orforest code) sets forest-area limits for legal reserves, and requires vegetation to be preserved along watercourses and on other ecologically sensitive settings such as steep slopes (Brasil 2012). The NVPL allows landholders to compensate for past forest clearance by buying forests elsewhere; given issues around permanence, this has provided a mechanism to support restoration of illegal farmland on national parks (Giannichi *et al.* 2018). Yet national legislation varies greatly across Amazonian countries. Developing a common set of approaches could be encouraged by linking national policies to the many international declarations and incentives that promote restoration, including the New York and Amsterdam declarations, the Bonn Challenge and Initiative 20x20, Sustainable Development Goal 15 Life on Land, the Convention on Biological Diversity, the United Nations Framework Convention on Climate Change, zero deforestation commitments, and the fight against imported deforestation.

8. CONCUSIONS

To maximize its ecological and societal impact, restoration needs to be implemented in ways that consider its benefits across scales, including at the level of the biome, within landscapes and catchments, and across different groups of local actors and stakeholders. Applying the most appropriate restoration approaches to the right places will require novel prioritization exercises that consider multiple benefits and include the societal feasibility, ecological need, and the risks posed by climate change.

REFERENCES

Abell R, Asquith N, Boccaletti G, *et al.* 2017. Beyond the source: the environmental, economic and community benefits of source water protection. *Arlington, VA Nat Conserv*.

Abell R, Vigerstol K, Higgins J, *et al.* 2019. Freshwater biodiversity conservation through source water protection: quantifying the potential and addressing the challenges. *Aquat Conserv Mar Freshw Ecosyst* **29**: 1022–38.

Abril G, Martinez J-M, Artigas LF, *et al.* 2014. Amazon River carbon dioxide outgassing fuelled by wetlands. *Nature* **505**: 395–8.

Adams C, Rodrigues ST, Calmon M, and Kumar C. 2016. Impacts of large-scale forest restoration on socioeconomic status and local livelihoods: what we know and do not know. *Biotropica* **48**: 731–44.

Aleixo I, Norris D, Hemerik L, *et al.* 2019a. Amazonian rainforest tree mortality driven by climate and functional traits. *Nat Clim Chang* **9**: 384–8.

Aleixo I, Norris D, Hemerik L, *et al.* 2019b. Amazonian rainforest tree mortality driven by climate and functional traits. *Nat Clim Chang* **9**: 384–8.

Alix-Garcia J and Gibbs HK. 2017. Forest conservation effects of Brazil's zero deforestation cattle agreements undermined by leakage. *Glob Environ Chang* **47**: 201–17.

Alkama R and Cescatti A. 2016. Climate change: Biophysical climate impacts of recent changes in global forest cover. *Science (80-)* **351**: 600–4.

Allan JD. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annu Rev Ecol Evol Syst* **35**: 257–84.

Almeida CT, Oliveira-Júnior JF, Delgado RC, *et al.* 2017. Spatiotemporal rainfall and temperature trends throughout the Brazilian Legal Amazon, 1973–2013. *Int J Climatol* **37**: 2013–26.

Alteff EF, Gonsioroski G, Barreiros M, *et al.* 2019. The rarest of the rare: rediscovery and status of the critically endangered Belem Curassow, Crax fasciolata pinima (Pelzeln, 1870). *Pap Avulsos Zool* **59**: e20195946.

Andrea MC da S, Dallacort R, Tieppo RC, and Barbieri JD. 2020. Assessment of climate change impact on double-cropping systems. *SN Appl Sci* **2**: 1–13.

Arantes CC, Winemiller KO, Asher A, *et al.* 2019. Floodplain land cover affects biomass distribution of fish functional diversity in the Amazon River. *Sci Rep* **9**: 1–13.

Chapter 29

Aronson J and Alexander S. 2013. Ecosystem Restoration is Now a Global Priority: Time to Roll up our Sleeves. *Restor Ecol* **21**: 293–6.

Arroyo-Rodríguez V, Fahrig L, Tabarelli M, *et al.* 2020. Designing optimal human-modified landscapes for forest biodiversity conservation. *Ecol Lett* **23**: 1404–20.

Beechie T, Pess G, Roni P, and Giannico G. 2008. Setting River Restoration Priorities: A Review of Approaches and a General Protocol for Identifying and Prioritizing Actions. *North Am J Fish Manag* **28**: 891–905.

Bell S de, Graham H, and White PCL. 2020. Evaluating Dual Ecological and Well-Being Benefits from an Urban Restoration Project. *Sustainability* **12**: 695.

Bentley L and Coomes DA. 2020. Partial river flow recovery with forest age is rare in the decades following establishment. *Glob Chang Biol* **26**: 1458–73.

Berenguer E, Gardner TA, Ferreira J, *et al.* 2018. Seeing the woods through the saplings: Using wood density to assess the recovery of human-modified Amazonian forests. *J Ecol*.

Bhagwat SA, Willis KJ, Birks HJB, and Whittaker RJ. 2008. Agroforestry: a refuge for tropical biodiversity? *Trends Ecol Evol* **23**: 261–7.

Blignaut J, Esler KJ, Wit MP de, *et al.* 2013. Establishing the links between economic development and the restoration of natural capital. *Curr Opin Environ Sustain* **5**: 94–101.

Bottazzi P, Wiik E, Crespo D, and Jones JPG. 2018. Payment for environmental "self-service": Exploring the links between Farmers' motivation and additionality in a conservation incentive programme in the Bolivian Andes. *Ecol Econ* **150**: 11–23.

Bradshaw CJA, Sodhi NS, Peh KSH, and Brook BW. 2007. Global evidence that deforestation amplifies flood risk and severity in the developing world. *Glob Chang Biol* **13**: 2379–95.

Bragagnolo C, Gama GM, Vieira FAS, *et al.* 2019. Hunting in Brazil: What are the options? *Perspect Ecol Conserv* **17**: 71–9.

Brancalion PHS, Almeida DRA de, Vidal E, *et al.* 2018. Fake legal logging in the Brazilian Amazon. *Sci Adv* **4**: eaat1192.

Brancalion PHS and Chazdon RL. 2017. Beyond hectares: four principles to guide reforestation in the context of tropical forest and landscape restoration. *Restor Ecol* **25**: 491–6.

Brasil. 2012. Lei 12.641, de 25 de maio de 2012. Viewed

Brêda JPLF, Paiva RCD de, Collischon W, *et al.* 2020. Climate change impacts on South American water balance from a continental-scale hydrological model driven by CMIP5 projections. *Clim Change* **159**: 503–22.

Brondizio ES, Andersson K, Castro F de, *et al.* 2021. Making place-based sustainability initiatives visible in the Brazilian Amazon. *Curr Opin Environ Sustain* **49**: 66–78.

Buytaert W, Iniguez V, and Bievre B De. 2007. The effects of afforestation and cultivation on water yield in the Andean páramo. *For Ecol Manage* **251**: 22–30.

Calmon M, Brancalion PHS, Paese A, *et al.* 2011. Emerging Threats and Opportunities for Large-Scale Ecological Restoration in the Atlantic Forest of Brazil. *Restor Ecol* **19**: 154–8.

Carvalho WD De and Mustin K. 2017. The highly threatened and little known Amazonian savannahs. *Nat Ecol Evol* **1**: 1–3.

Castello L and Macedo MN. 2016. Large-scale degradation of Amazonian freshwater ecosystems. *Glob Chang Biol* **22**: 990–1007.

Chapter 29

Celentano D, Rousseau GX, Muniz FH, *et al.* 2017. Towards zero deforestation and forest restoration in the Amazon region of Maranhão state, Brazil. *Land use policy* **68**: 692–8.

César RG, Belei L, Badari CG, *et al.* 2021. Forest and Landscape Restoration: A Review Emphasizing Principles, Concepts, and Practices. *Land* **10**: 28.

Chazdon R, Brenes A, and Alvarado B. 2005. Effects of Climate and Stand Age on Annual Tree Dynamics in Tropical Second-Growth Rain Forests on JSTOR. *Ecology* **86**: 1808–15.

Chazdon RL, Broadbent EN, Rozendaal DMA, *et al.* 2016a. Carbon sequestration potential of second-growth forest regeneration in the Latin American tropics. *Sci Adv* **2**.

Chazdon RL, Broadbent EN, Rozendaal DMA, *et al.* 2016b. Carbon sequestration potential of second-growth forest regeneration in the Latin American tropics. *Sci Adv* **2**: e1501639.

Chazdon RL, Cullen L, Padua SM, and Padua CV. 2020a. People, primates and predators in the Pontal: from endangered species conservation to forest and landscape restoration in Brazil's Atlantic Forest. *R Soc Open Sci* **7**: 200939.

Chazdon RL, Gutierrez V, Brancalion PHS, *et al.* 2020b. Co-Creating Conceptual and Working Frameworks for Implementing Forest and Landscape Restoration Based on Core Principles. *Forests* **11**: 706.

Coe MT, Costa MH, and Soares-Filho BS. 2009. The influence of historical and potential future deforestation on the stream flow of the Amazon River – Land surface processes and atmospheric feedbacks. *J Hydrol* **369**: 165–74.

Constantino P de AL. 2019. Subsistence Hunting with Mixed-Breed Dogs Reduces Hunting Pressure on Sensitive Amazonian Game Species in Protected Areas. *Environ Conserv* **46**: 92–8.

Cook-Patton SC, Leavitt SM, Gibbs D, *et al.* 2020. Mapping carbon accumulation potential from global natural forest regrowth. *Nature* **585**: 545–50.

Costa-Pereira R, Lucas C, Crossa M, *et al.* 2018. Defaunation shadow on mutualistic interactions. *Proc Natl Acad Sci* **115**: E2673–5.

Coudel E, Ferreira J, Carvalho Amazonas M de, *et al.* 2015. The rise of PES in Brazil: from pilot projects to public policies. In: Handbook of Ecological Economics. Edward Elgar Publishing.

Crouzeilles R, Beyer HL, Monteiro LM, *et al.* 2020. Achieving cost-effective landscape-scale forest restoration through targeted natural regeneration. *Conserv Lett* **13**: e12709.

Dala-Corte RB, Melo AS, Siqueira T, *et al.* 2020. Thresholds of freshwater biodiversity in response to riparian vegetation loss in the Neotropical region. *J Appl Ecol* **57**: 1391–402.

Davidson EA, Araújo AC de, Artaxo P, *et al.* 2012. The Amazon basin in transition. *Nature* **481**: 321–8.

Dias LCP, Macedo MN, Costa MH, *et al.* 2015. Effects of land cover change on evapotranspiration and streamflow of small catchments in the Upper Xingu River Basin, Central Brazil. *J Hydrol Reg Stud* **4**: 108–22.

Ding H, Faruqi S, Wu A, *et al.* 2017. Roots of Prosperity: The Economics and Finance of Restoring Land The Economics and Finance of Restoring Land. Washington, D. C. .

Dixon SJ, Sear DA, Odoni NA, *et al.* 2016. The effects of river restoration on catchment scale flood risk and flood hydrology. *Earth Surf Process Landforms* **41**: 997–1008.

Edwards DP, Tobias JA, Sheil D, *et al.* 2014. Maintaining ecosystem function and services in logged tropical forests. *Trends Ecol \& Evol* **29**: 511–20.

Elias F, Ferreira J, Lennox GD, *et al.* 2020. Assessing the growth and climate sensitivity of secondary forests in highly deforested Amazonian landscapes. *Ecology* **101**: e02954.

Erbaugh JT and Oldekop JA. 2018. Forest landscape restoration for livelihoods and well-being. *Curr Opin Environ Sustain* **32**: 76–83.

Erbaugh JT, Pradhan N, Adams J, *et al.* 2020. Global forest restoration and the importance of prioritizing local communities. *Nat Ecol Evol* **4**: 1472–6.

Esquivel-Muelbert A, Baker TR, Dexter KG, *et al.* 2019. Compositional response of Amazon forests to climate change. *Glob Chang Biol* **25**: 39–56.

Fahrig L, Arroyo-Rodríguez V, Bennett JR, *et al.* 2019. Is habitat fragmentation bad for biodiversity? *Biol Conserv* **230**: 179–86.

Farinosi F, Arias ME, Lee E, *et al.* 2019. Future climate and land use change impacts on river flows in the Tapajós Basin in the Brazilian Amazon. *Earth's Futur* **7**: 993–1017.

Feitosa IB, Huszar VLM, Domingues CD, *et al.* 2019. Plankton community interactions in an Amazonian floodplain lake, from bacteria to zooplankton. *Hydrobiologia* **831**: 55–70.

Ferraz G, Russell GJ, Stouffer PC, *et al.* 2003. Rates of species loss from Amazonian forest fragments. *Proc Natl Acad Sci* **100**: 14069–73.

Ferreira J, Lennox GD, Gardner TA, *et al.* 2018. Carbon-focused conservation may fail to protect the most biodiverse tropical forests. *Nat Clim Chang* **8**: 744–9.

Filoso S, Bezerra MO, Weiss KCB, and Palmer MA. 2017. Impacts of forest restoration on water yield: A systematic review. *PLoS One* **12**: e0183210.

Flecker AS, McIntyre PB, Moore JW, *et al.* 2010. Migratory fishes as material and process subsidies in riverine ecosystems. In: American Fisheries Society Symposium.

Fletcher RJ, Didham RK, Banks-Leite C, *et al.* 2018. Is habitat fragmentation good for biodiversity? *Biol Conserv* **226**: 9–15.

Flores BM, Holmgren M, Xu C, *et al.* 2017. Floodplains as an Achilles' heel of Amazonian forest resilience. *Proc Natl Acad Sci U S A* **114**.

Food and Agriculture Organization of the United Nations and International Tropical Timber Organization. 2011. The State of Forests in the Amazon Basin, Congo Basin and Southeast Asia.

Fortmann L and Bruce J. 1991. You've got to know who controls the land and trees people use: gender, tenure and the environment. University of Zimbabw (UZ).

Freeman MC, Pringle CM, Greathouse EA, and Freeman BJ. 2003. Ecosystem-level consequences of migratory faunal depletion caused by dams. In: American Fisheries Society Symposium.

Garcia-Carreras L and Parker DJ. 2011. How does local tropical deforestation affect rainfall? *Geophys Res Lett* **38**: n/a-n/a.

Garrett RD, Gardner TA, Fonseca Morello T, *et al.* 2017. Explaining the persistence of low income and environmentally degrading land uses in the Brazilian Amazon. *Ecol Soc* **22**.

Garrett RD, Koh I, Lambin EF, *et al.* 2018. Intensification in agriculture-forest frontiers: Land use responses to development and conservation policies in Brazil. *Glob Environ Chang* **53**: 233–43.

Giannichi ML, Dallimer M, Baker TR, *et al.* 2018. Divergent Landowners' Expectations May Hinder the Uptake of a Forest Certificate Trading Scheme. *Conserv Lett* **11**: e12409.

Chapter 29

Goldman R, Ben\’\itez S, Calvache A, and Montambault J. 2010. Measuring the effectiveness of water funds: guidance document for development of impact measures. *TNC, Arlington, Virginia*.

Hall A, Chiu Y, and Selker JS. 2020. Coupling high-resolution monitoring and modelling to verify restoration-based temperature improvements. *River Res Appl* **36**: 1430–41.

Harris JA, Hobbs RJ, Higgs E, and Aronson J. 2006. Ecological restoration and global climate change.

Haugaasen T and Peres CA. 2007. Vertebrate responses to fruit production in Amazonian flooded and unflooded forests. *Biodivers Conserv* **16**: 4165–90.

Hayhoe SJ, Neill C, Porder S, *et al.* 2011. Conversion to soy on the Amazonian agricultural frontier increases streamflow without affecting stormflow dynamics. *Glob Chang Biol* **17**: 1821–33.

Heinrich VHA, Dalagnol R, Cassol HLG, *et al.* 2021. Large carbon sink potential of secondary forests in the Brazilian Amazon to mitigate climate change. *Nat Commun* **12**: 1–11.

Higgins J V and Zimmerling A. 2013. A Primer for Monitoring Water Funds. Arlington, VA: The Nature Conservancy. 2013.

Holdsworth AR and Uhl C. 1997. FIRE IN AMAZONIAN SELECTIVELY LOGGED RAIN FOREST AND THE POTENTIAL FOR FIRE REDUCTION. *Ecol Appl* **16**: 440–51.

Hurd LE, Sousa RGC, Siqueira-Souza FK, *et al.* 2016. Amazon floodplain fish communities: Habitat connectivity and conservation in a rapidly deteriorating environment. *Biol Conserv* **195**: 118–27.

Hutyra LR, Munger JW, Nobre CA, *et al.* 2005. Climatic variability and vegetation vulnerability in Amazônia. *Geophys Res Lett* **32**: L24712.

Jackson ST and Hobbs RJ. 2009. Ecological restoration in the light of ecological history. *Science (80-)* **325**: 567–9.

Junk WJ, Bayley PB, Sparks RE, and others. 1989. The flood pulse concept in river-floodplain systems. *Can Spec Publ Fish Aquat Sci* **106**: 110–27.

Kratter AW. 1997. Bamboo Specialization by Amazonian Birds1. *Biotropica* **29**: 100–10.

Kremen C and Merenlender AM. 2018. Landscapes that work for biodiversity and people. *Science (80-)* **362**.

L. Resque A, Coudel E, Piketty M-G, *et al.* 2019. Agrobiodiversity and Public Food Procurement Programs in Brazil: Influence of Local Stakeholders in Configuring Green Mediated Markets. *Sustainability* **11**: 1425.

Lamb D, Erskine PD, and Parrotta JA. 2005. Restoration of degraded tropical forest landscapes. *Science (80-)* **310**: 1628–32.

Leal CG, Lennox GD, Ferraz SFB, *et al.* 2020a. Integrated terrestrial-freshwater planning doubles conservation of tropical aquatic species. *Science (80-)* **370**: 117–21.

Leal CG, Lennox GD, Ferraz SFB, *et al.* 2020b. Integrated terrestrial-freshwater planning doubles conservation of tropical aquatic species. *Science (80-)* **370**: 117–21.

Lee M and Hancock P. 2011. Restoration and Stewardship Volunteerism. In: Human Dimensions of Ecological Restoration. Washington, DC: Island Press/Center for Resource Economics.

Lees AC, Attwood S, Barlow J, and Phalan B. 2020. Biodiversity scientists must fight the creeping rise of extinction denial. *Nat Ecol Evol 2020 411* **4**: 1440–3.

Lees AC, Moura NG, Almeida AS, and Vieira ICG. 2014. Noteworthy ornithological

records from the threatened campinas of the lower rio Tocantins, east Amazonian Brazil. *Bull Br Ornithol Club* **134**: 247–58.
Lees AC and Peres CA. 2006. Rapid avifaunal collapse along the Amazonian deforestation frontier. *Biol Conserv* **133**: 198–211.
Lees AC and Peres CA. 2009. Gap-crossing movements predict species occupancy in Amazonian forest fragments. *Oikos* **118**: 280–90.
Leitold V, Morton DC, Longo M, *et al.* 2018. El Niño drought increased canopy turnover in Amazon forests. *New Phytol* **219**: 959–71.
Lewis SL, Wheeler CE, Mitchard ETA, and Koch A. 2019. Restoring natural forests is the best way to remove atmospheric carbon. *Nature* **568**: 25–8.
Lima LS, Coe MT, Soares Filho BS, *et al.* 2014. Feedbacks between deforestation, climate, and hydrology in the Southwestern Amazon: implications for the provision of ecosystem services. *Landsc Ecol* **29**: 261–74.
Lohbeck M, Poorter L, Lebrija-Trejos E, *et al.* 2013. Successional changes in functional composition contrast for dry and wet tropical forest. *Ecology* **94**: 1211–6.
Lovejoy TE and Nobre C. 2018. Amazon tipping point.
Luke SH, Slade EM, Gray CL, *et al.* 2019. Riparian buffers in tropical agriculture: Scientific support, effectiveness and directions for policy. *J Appl Ecol* **56**: 85–92.
Macedo MN, Coe MT, DeFries R, *et al.* 2013. Land-use-driven stream warming in southeastern Amazonia. *Philos Trans R Soc B Biol Sci* **368**: 20120153.
Marengo JA. 2009. Long-term trends and cycles in the hydrometeorology of the Amazon basin since the late 1920s. *Hydrol Process* **23**: 3236–44.
Margules CR and Pressey RL. 2000. Systematic conservation planning. *Nature* **405**: 243–53.
Mather AS. 1992. The forest transition. *Area* **24**: 367–79.
McClain ME and Naiman RJ. 2008. Andean influences on the biogeochemistry and ecology of the Amazon River. *Bioscience* **58**: 325–38.
McDowell N, Allen CD, Anderson-Teixeira K, *et al.* 2018. Drivers and mechanisms of tree mortality in moist tropical forests. *New Phytol* **219**: 851–69.
McIntosh EJ, Pressey RL, Lloyd S, *et al.* 2017. The impact of systematic conservation planning. *Annu Rev Environ Resour* **42**: 677–97.
Meerveld HJ (Ilja) van, Jones JPG, Ghimire CP, *et al.* 2021. Forest regeneration can positively contribute to local hydrological ecosystem services: Implications for forest landscape restoration (L Flory, Ed). *J Appl Ecol* **58**: 755–65.
Mendes CB and Prevedello JA. 2020. Does habitat fragmentation affect landscape-level temperatures? A global analysis. *Landsc Ecol* **35**: 1743–56.
Moore RP, Robinson WD, Lovette IJ, and Robinson TR. 2008. Experimental evidence for extreme dispersal limitation in tropical forest birds. *Ecol Lett* **11**: 960–8.
Negrón-Juárez RI, Chambers JQ, Guimaraes G, *et al.* 2010. Widespread Amazon forest tree mortality from a single cross-basin squall line event. *Geophys Res Lett* **37**: n/a-n/a.
Ngo Bieng MA, Souza Oliveira M, Roda J-M, *et al.* 2021. Relevance of secondary tropical forest for landscape restoration. *For Ecol Manage* **493**: 119265.
Nobre CA, Sampaio G, Borma LS, *et al.* 2016. Land-use and climate change risks in the Amazon and the need of a novel sustainable development paradigm. *Proc Natl Acad Sci* **113**: 10759–68.
Nobre CA, Sellers PJ, and Shukla J. 1991. Amazonian Deforestation and Regional Climate Change. *J Clim* **4**: 957–88.

Chapter 29

Nóbrega RLB, Ziembowicz T, Torres GN, *et al.* 2020. Ecosystem services of a functionally diverse riparian zone in the Amazon--Cerrado agricultural frontier. *Glob Ecol Conserv* **21**: e00819.

Oosten C van, Runhaar H, and Arts B. 2021. Capable to govern landscape restoration? Exploring landscape governance capabilities, based on literature and stakeholder perceptions. *Land use policy* **104**: 104020.

Ota L, Chazdon RL, Herbohn J, *et al.* 2020. Achieving Quality Forest and Landscape Restoration in the Tropics. *Forests* **11**: 820.

Oyama MD and Nobre C. 2003. A new climate-vegetation equilibrium state for Tropical South America. *Geophys Res Lett* **30**: 10–3.

Palmer MA, Bernhardt ES, Allan JD, *et al.* 2005. Standards for ecologically successful river restoration. *J Appl Ecol* **42**: 208–17.

Peres CA. 2001. Synergistic effects of subsistence hunting and habitat fragmentation on Amazonian forest vertebrates. *Conserv Biol* **15**: 1490–505.

Peterjohn WT and Correll DL. 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. *Ecology* **65**: 1466–75.

Philipson CD, Cutler MEJ, Brodrick PG, *et al.* 2020. Active restoration accelerates the carbon recovery of human-modified tropical forests. *Science (80-)* **369**: 838–41.

Phillips OL, Aragão LEOC, Lewis SL, *et al.* 2009. Drought sensitivity of the amazon rainforest. *Science (80-)* **323**: 1344–7.

Piponiot C, Rödig E, Putz FE, *et al.* 2019a. Can timber provision from Amazonian production forests be sustainable? *Environ Res Lett* **14**: 064014.

Piponiot C, Rödig E, Putz FE, *et al.* 2019b. Can timber provision from Amazonian production forests be sustainable? *Environ Res Lett* **14**: 064014.

Piva Da Silva M. 2017. Livelihoods, Capabilities and Insurgent Citizenship in and around a rainforest metropolis: from violent urbanism to a new rurality?

Poff NL, Allan JD, Bain MB, *et al.* 1997. The natural flow regime. *Bioscience* **47**: 769–84.

Pokorny B, Johnson J, Medina G, and Hoch L. 2012. Market-based conservation of the Amazonian forests: Revisiting win–win expectations. *Geoforum* **43**: 387–401.

Putz FE, Sist P, Fredericksen T, and Dykstra D. 2008. Reduced-impact logging: challenges and opportunities. *For Ecol Manage* **256**: 1427–33.

Putz FE, Zuidema PA, Synnott T, *et al.* 2012. Sustaining conservation values in selectively logged tropical forests: the attained and the attainable. *Conserv Lett* **5**: 296–303.

Qie L, Telford EM, Massam MR, *et al.* 2019. Drought cuts back regeneration in logged tropical forests. *Environ Res Lett* **14**: 045012.

Ray D, Nepstad D, and Moutinho P. 2005. Micrometeorological and canopy controls of flammability in mature and disturbed forests in an east-central Amazon landscape. *Ecol Appl* **15**: 2.

Reed MS. 2008. Stakeholder participation for environmental management: A literature review. *Biol Conserv* **141**: 2417–31.

Reed J, Barlow J, Carmenta R, *et al.* 2019. Engaging multiple stakeholders to reconcile climate, conservation and development objectives in tropical landscapes. *Biol Conserv* **238**: 108229.

Reed J, Vianen J Van, Deakin EL, *et al.* 2016. Integrated landscape approaches to managing social and environmental issues in the tropics: learning from the past to guide the future. *Glob Chang Biol* **22**: 2540–54.

Rodrigues ASL, Ewers RM, Parry L, *et al.* 2009. Boom-and-Bust Development Patterns

Across the Amazon Deforestation Frontier. *Science (80-)* **324**: 1435–7.

Rossi, Jacques Garcia Alain Roques J and Rousselet J-P. 2016. Trees outside forests in agricultural landscapes: spatial distribution and impact on habitat connectivity for forest organisms. *Landsc Ecol* **31**: 243–54.

Saad SI, Silva J da, Silva MLN, *et al.* 2018. Analyzing ecological restoration strategies for water and soil conservation. *PLoS One* **13**: e0192325.

Sabogal C, Besacier C, and McGuire D. 2015. Forest and landscape restoration: Concepts, approaches and challenges for implementation. *Unasylva* **66**: 3.

Sampaio G, Nobre C, Costa MH, *et al.* 2007. Regional climate change over eastern Amazonia caused by pasture and soybean cropland expansion. *Geophys Res Lett* **34**: L17709.

Sayer J, Sunderland T, Ghazoul J, *et al.* 2013. Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. *Proc Natl Acad Sci U S A* **110**: 8349–56.

Schiesari L, Waichman A, Brock T, *et al.* 2013. Pesticide use and biodiversity conservation in the Amazonian agricultural frontier. *Philos Trans Biol Sci* **368**: 1–9.

Schmidt IB, Urzedo DI de, Piña-Rodrigues FCM, *et al.* 2019. Community-based native seed production for restoration in Brazil – the role of science and policy. *Plant Biol* **21**: 389–97.

Schulze MD. 2003. Ecology and behavior of nine timber tree species in Pará, Brazil: links between species life history and forest management and conservation.

Schwartzman S, Boas AV, Ono KY, *et al.* 2013. The natural and social history of the indigenous lands and protected areas corridor of the Xingu River basin. *Philos Trans R Soc B Biol Sci* **368**: 20120164.

Silva Junior CHL, Aragão LEOC, Anderson LO, *et al.* 2020. Persistent collapse of biomass in Amazonian forest edges following deforestation leads to unaccounted carbon losses. *Sci Adv* **6**: eaaz8360.

Sist P and Ferreira FN. 2007. Sustainability of reduced-impact logging in the Eastern Amazon. *For Ecol Manage* **243**: 199–209.

Sist P, Piponiot C, Kanashiro M, *et al.* 2021. Sustainability of Brazilian forest concessions. *For Ecol Manage* **496**: 119440.

Smith CC, Healey J, Berenguer E, *et al.* 2021. Old-growth forest loss and secondary forest recovery across Amazonian countries. *Environ Res Lett*.

Smith MN, Taylor TC, Haren J van, *et al.* 2020. Empirical evidence for resilience of tropical forest photosynthesis in a warmer world. *Nat Plants* **6**: 1225–30.

Staal A, Tuinenburg OA, Bosmans JHC, *et al.* 2018. Forest-rainfall cascades buffer against drought across the Amazon. *Nat Clim Chang* **8**: 539–43.

Stanturf JA, Kant P, Lillesø J-PB, *et al.* 2015. Forest landscape restoration as a key component of climate change mitigation and adaptation. International Union of Forest Research Organizations (IUFRO) Vienna, Austria.

Stanturf JA, Kleine M, Mansourian S, *et al.* 2019. Implementing forest landscape restoration under the Bonn Challenge: a systematic approach. *Ann For Sci* **76**: 1–21.

Strassburg BBN, Iribarrem A, Beyer HL, *et al.* 2020. Global priority areas for ecosystem restoration. *Nature* **586**: 724–9.

Sullivan MJP, Lewis SL, Affum-Baffoe K, *et al.* 2020. Long-term thermal sensitivity of earth's tropical forests. *Science (80-)* **368**: 869–74.

Tregidgo DJ, Barlow J, Pompeu PS, *et al.* 2017. Rainforest metropolis casts 1,000-km

defaunation shadow. *Proc Natl Acad Sci* **114**: 8655–9.
Uriarte M, Schwartz N, Powers JS, *et al.* 2016. Impacts of climate variability on tree demography in second growth tropical forests: the importance of regional context for predicting successional trajectories. *Biotropica* **48**: 780–97.
Vannote RL, Minshall GW, Cummins KW, *et al.* 1980. The river continuum concept. *Can J Fish Aquat Sci* **37**: 130–7.
Ward J V. 1989. The four-dimensional nature of lotic ecosystems. *J North Am Benthol Soc* **8**: 2–8.
Waroux Y le P de, Garrett RD, Heilmayr R, and Lambin EF. 2016. Land-use policies and corporate investments in agriculture in the Gran Chaco and Chiquitano. *Proc Natl Acad Sci* **113**: 4021–6.
Wilson SJ, Schelhas J, Grau R, *et al.* 2017. Forest ecosystem-service transitions: The ecological dimensions of the forest transition. *Ecol Soc* **22**.
Wohl E, Bledsoe BP, Jacobson RB, *et al.* 2015. The natural sediment regime in rivers: Broadening the foundation for ecosystem management. *Bioscience* **65**: 358–71.
Zhang J, Bruijnzeel LA, Tripoli R, and Meerveld HJI van. 2019. Water budget and run-off response of a tropical multispecies “reforest” and effects of typhoon disturbance. *Ecohydrology* **12**: e2055.