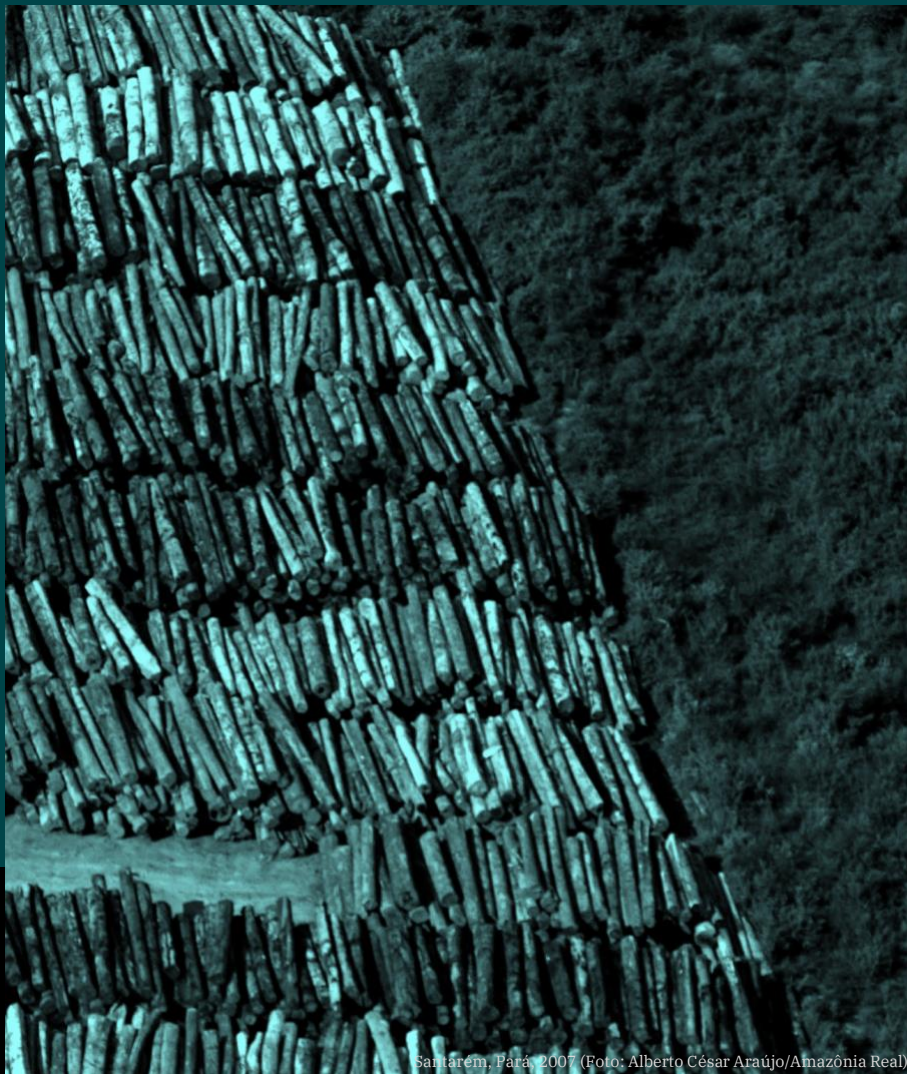


Chapter 19

Drivers and ecological impacts of deforestation and forest degradation



Santarém, Pará, 2007 (Foto: Alberto César Araújo/Amazônia Real)



Science Panel for the Amazon



About the Science Panel for the Amazon (SPA)

The Science Panel for the Amazon is an unprecedented initiative convened under the auspices of the United Nations Sustainable Development Solutions Network (SDSN). The SPA is composed of over 200 preeminent scientists and researchers from the eight Amazonian countries, French Guiana, and global partners. These experts came together to debate, analyze, and assemble the accumulated knowledge of the scientific community, Indigenous peoples, and other stakeholders that live and work in the Amazon.

The Panel is inspired by the Leticia Pact for the Amazon. This is a first-of-its-kind Report which provides a comprehensive, objective, open, transparent, systematic, and rigorous scientific assessment of the state of the Amazon's ecosystems, current trends, and their implications for the long-term well-being of the region, as well as opportunities and policy relevant options for conservation and sustainable development.

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INDEX

GRAPHICAL ABSTRACT	2
KEY MESSAGES	3
ABSTRACT	3
19.1 INTRODUCTION	4
19.2 DEFORESTATION: AN OVERVIEW OF DIRECT DRIVERS AND IMPACTS	5
9.3 MAIN DRIVERS OF DEFORESTATION AND THEIR ASSOCIATED IMPACTS	10
19.3.1 AGRICULTURAL EXPANSION	10
19.3.2. INFRASTRUCTURE	11
19.3.2.1. <i>Roads</i>	11
19.3.2.2 <i>Hydropower dams</i>	16
19.3.2.3 <i>Urbanization</i>	16
19.3.2.4 <i>Railways and waterways</i>	19
19.3.3. MINING	19
19.3.3.1 <i>Minerals</i>	19
19.3.3.2 <i>Oil and gas</i>	20
19.4 DEGRADATION: AN OVERVIEW OF DIRECT DRIVERS AND IMPACTS	23
19.4.1 UNDERSTORY FIRES	25
19.4.2 EDGE EFFECTS	27
19.4.3 LOGGING	28
19.4.4 HUNTING	30
19.5 CONCLUSIONS	31
19.6 RECOMMENDATIONS	31
19.7 REFERENCES	31

Graphical Abstract

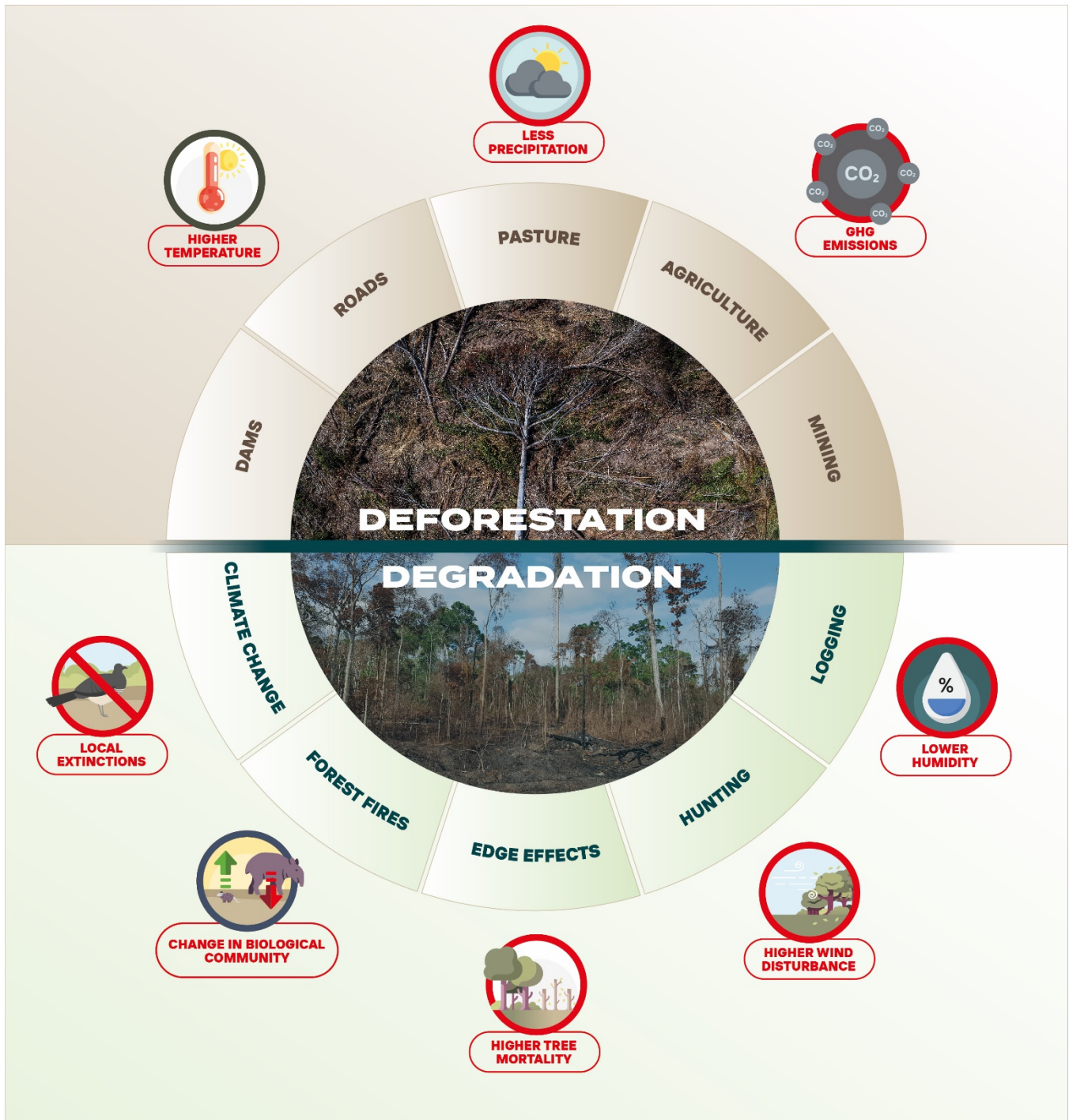


Figure 19.A Graphical Abstract

Drivers and Ecological Impacts of Deforestation and Forest Degradation

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Key Messages

- By 2018, the Amazon lost approximately 870,000 km² of primary forests.
- There are at least 1,036,080 km² of degraded Amazonian forests.
- Agricultural expansion, mainly cattle ranching, is the greatest driver of deforestation in the Amazon.
- Deforestation leads to local, regional, and global impacts.
- Forest degradation encompasses significant changes in forest structure, microclimate, and biodiversity.
- Deforestation and forest degradation are responsible for enormous quantities of CO₂ emissions.

Abstract

Deforestation, the complete removal of an area's forest cover; and forest degradation, the significant loss of forest structure, functions, and processes; are the result of the interaction between various direct drivers, often operating in tandem. By 2018, the Amazon biome had lost approximately 870,000 km² of its original forest cover, mainly due to agricultural expansion (pasture and croplands). Other direct drivers of forest loss include the opening of new roads, construction of hydroelectric dams, exploitation of minerals and oil, and urbanization. Impacts of deforestation range from local to global, including local changes in landscape configuration, climate, and biodiversity; regional impacts on hydrological cycles; and global increase of greenhouse gas emissions. Of the remaining Amazonian forests, 17% are degraded, corresponding to approximately 1,036,080 km². Various anthropogenic drivers, including forest fires, edge effects, selective logging, hunting, and climate change can cause forest degradation. Degraded forests have significantly different structure, microclimate, and biodiversity as compared to undisturbed ones. These

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forests tend to have higher tree mortality, lower carbon stocks, more canopy gaps, higher temperatures, lower humidity, higher wind exposure, and exhibit compositional and functional shifts in both fauna and flora. Degraded forests can come to resemble their undisturbed counterparts, but this depends on the type, duration, intensity, and frequency of the disturbance event. In some cases, this may prohibit the return to a historic baseline. Avoiding further loss and degradation of Amazonian forests is crucial to ensure they continue to provide valuable and life-supporting ecosystem services.

Keywords: Deforestation, forest degradation, cattle ranching, agriculture, mining, wildfires, edge effects, selective logging, hunting, biodiversity loss, CO₂ emissions

19.1 Introduction

Across the Amazon, deforestation and forest degradation are the result of the interplay between various underlying and direct drivers acting at global, regional, and local scales (Armenteras *et al.* 2017; Barona *et al.* 2010; Clerici *et al.* 2020; Rudel *et al.* 2009). Underlying drivers are factors that affect human actions (IPBES 2019), such as lack of governance and variation in both the price of commodities and the price of land (Brandão *et al.* 2020; Garrett *et al.* 2013; Nepstad *et al.* 2014). Conversely, direct drivers represent the human actions that impact nature (IPBES 2019), including the expansion of pastures and croplands, opening of new roads, construction of hydroelectric dams, or exploitation of minerals and oil (Fearnside 2016; Ometto *et al.* 2011; Sonter *et al.* 2017). Drivers often act simultaneously, making it very difficult to quantify their individual impacts. For example, road construction and paving leads to the creation of new urban centers and the advance of the agricultural frontier (Fernández-Llamazares *et al.* 2018; Nascimento *et al.* 2021). Although each of these drivers (road building, urbanization, and agricultural expansion) will increase deforestation rates, it is very difficult to estimate their isolated impacts on ecosystems processes and functions.

The impacts of deforestation and forest degradation can be direct or indirect and have local, regional, or global consequences (Davidson *et al.* 2012; Magalhães *et al.* 2019; Spracklen and Garcia-Carreras 2015). The most obvious direct impact of deforestation is biodiversity loss—species-rich forested areas are converted to species-poor agricultural lands. However, there are more cryptic

impacts resulting from deforestation and forest degradation, such as changes in local temperatures and regional precipitation regimes or increased global greenhouse gas emissions (Longo *et al.* 2020; Mollinari *et al.* 2019). These impacts can interact with others, amplifying their individual effects. For instance, changes in precipitation patterns can increase plant mortality, leading to more greenhouse gas emissions, which in turn contribute to further changes in climate (Esquivel-Muelbert *et al.* 2020; Nepstad *et al.* 2007).

Although both the direct drivers and the impacts of deforestation and forest degradation do not necessarily occur in isolation, we will discuss them separately in this chapter, trying to acknowledge the role of different drivers across the Amazon, as well as their varied impacts. We start by presenting a general discussion of deforestation, followed by a detailed presentation of its main drivers, namely agricultural expansion (including both pasture and croplands), infrastructure, and mining. Whenever possible, we also try to quantify the direct and indirect impacts of each individual driver. We then present a general framework of degradation of Amazonian forests, discussing in more detail its main drivers, including understory fires, edge effects, selective logging, and hunting. The quantifiable impacts of each of these drivers are discussed in their individual sections. Despite the tight links between underlying and direct drivers of deforestation and forest degradation, the former is not dealt within this chapter, but rather in Chapters 14 to 18. Finally, although the direct drivers of deforestation and forest degradation also impact aquatic ecosystems and human well-being, these are discussed elsewhere (Chapters 20 and 21, respectively).

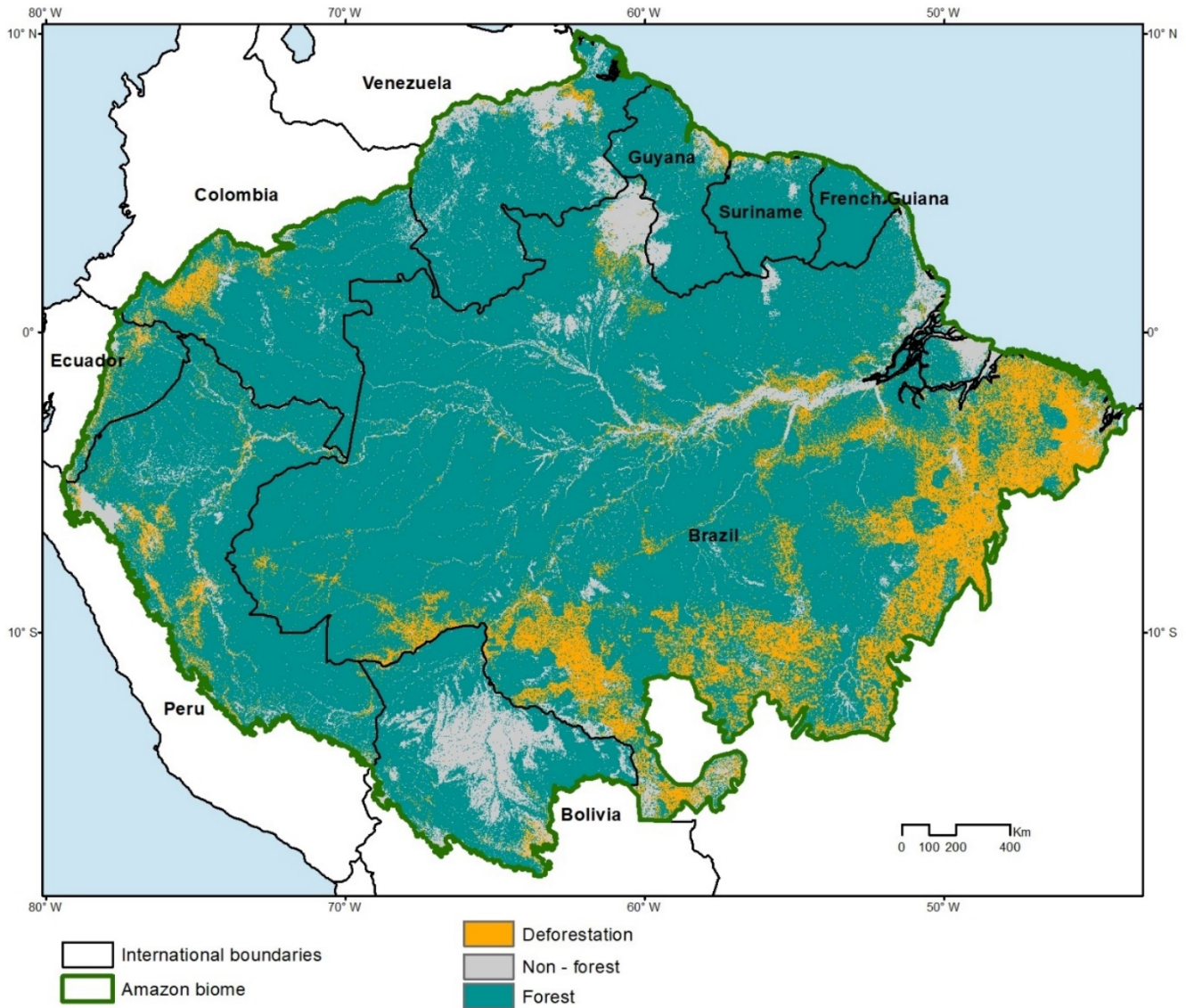


Figure 19.1 Current land occupied by either natural vegetation or pasture and agriculture across the Amazon biome. Cumulative deforestation data is shown until 2018 (MapBiomias 2020) and analyzed according to (Smith et al. 2021).

In this chapter, we focus only on the Amazon biome (Figure 19.1), therefore using a different geographical limit than those used in previous chapters; however, most maps will present both limits for the reader’s reference.

19.2 Deforestation: An overview of direct drivers and impacts

Deforestation is defined as the complete removal of

an area’s forest cover (Putz and Redford 2010). In the Amazon, 867,675 km² had been deforested by 2018 (MapBiomias 2020), equivalent to 14% of its original forested area (Fig. 19.1). Most deforestation has been concentrated in Brazil, which lost approximately 741,759 km² of forests (MapBiomias 2020; Smith et al. 2021) – an area 15 times greater than that lost by Peru, the country with the second largest deforested area (Fig. 19.2a). In relative terms, the country that lost most of its Amazon

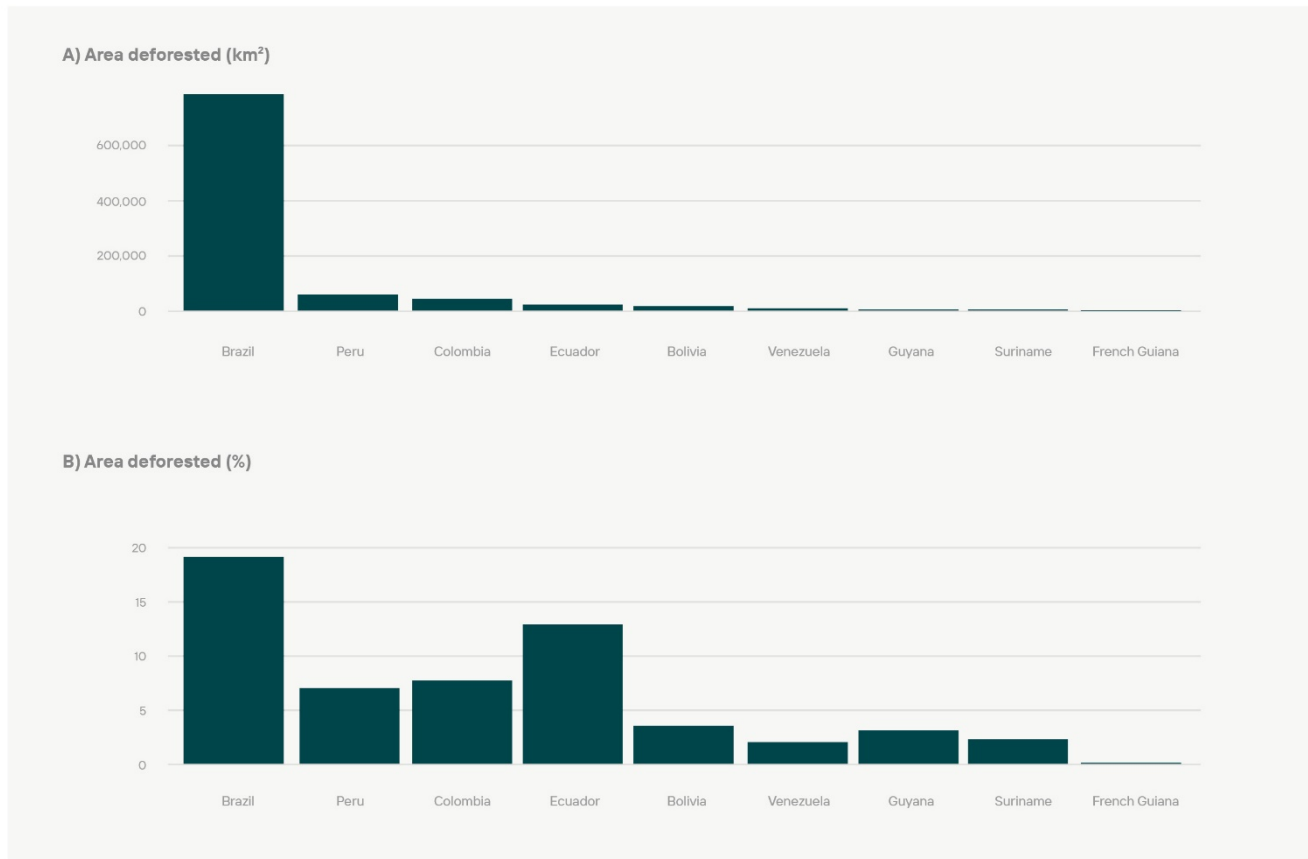


Figure 19.2 Country-level deforestation in the Amazon biome. A) Cumulative deforestation until 2018. B) Percentage of the biome deforested in each Amazonian country or territory. Data obtained from MapBiomias 2020 and analyzed according to Smith et al. 2021.

biome was Brazil (19%), followed by Ecuador (13%). To date, French Guiana, Suriname, and Venezuela have the greatest proportion of original vegetation cover, 99.85%, 97.92%, and 97.89%, respectively (Fig. 19.2b).

Deforestation varies not only across space, but also across time. Between 1991 and 2006, annual deforestation was consistently above 20,000 km², peaking in 2003 when 31,828 km² of forests were lost (MapBiomias 2020). From 2007 until 2018, annual deforestation in the region was much lower, ranging between 9,918 km² and 17,695 km² (Fig. 19.3). By 1990, only 5% of the forests in the basin had been lost. However, this figured reached 9% in 2000 and 12% in 2010 (MapBiomias 2020; Smith *et*

al. 2021). See Annex I for a time series of forest loss in each Amazonian country.

Amazonian deforestation has been mainly driven by agricultural expansion (including both pastures and croplands), although other drivers also contribute, such as mining and infrastructure development, including urbanization and the building of roads, railways, waterways, and large-scale hydropower dams (Fig. 19.4).

These drivers often act in tandem, creating positive feedbacks. For instance, after the building of large roads crossing the Brazilian Amazon, there was an influx of migrants to the region, creating new cities and expanding existing ones. In rural

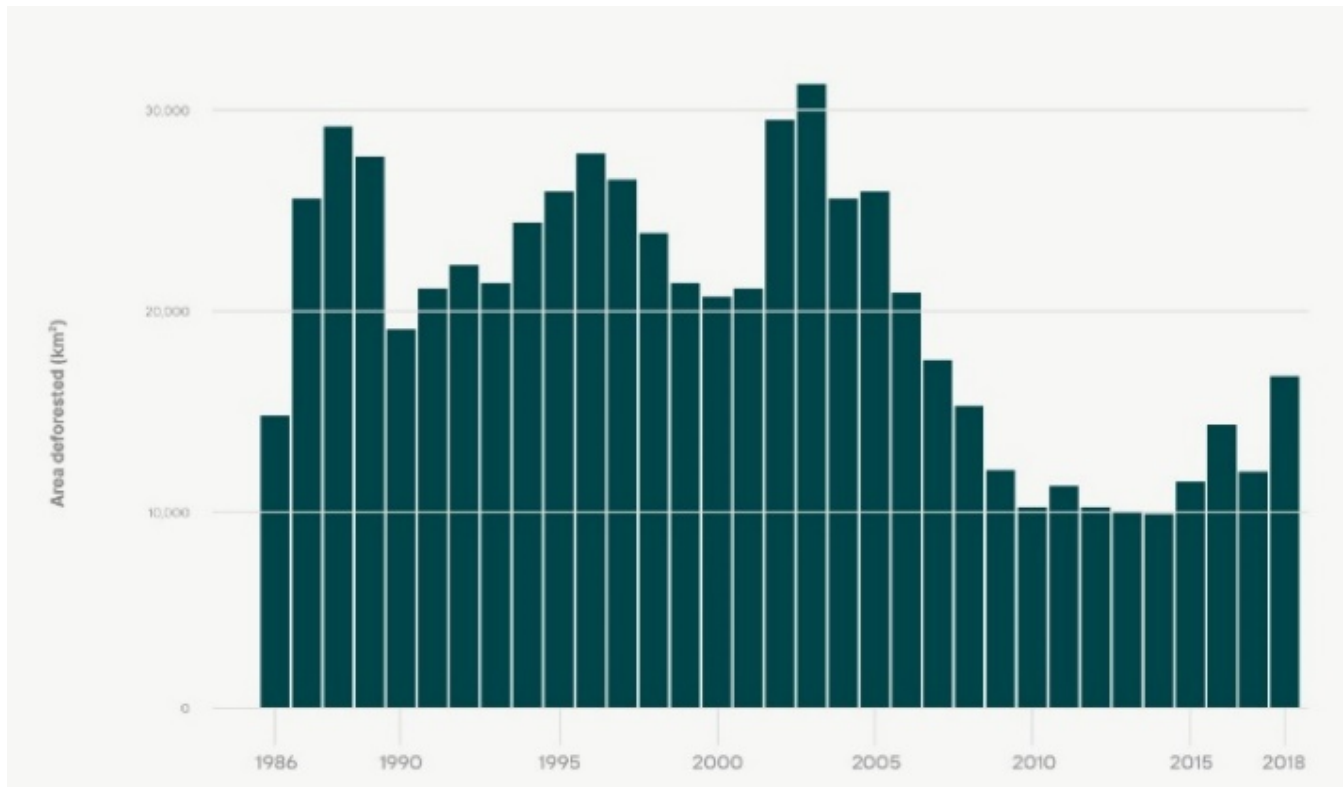


Figure 19.3 Annual deforestation across the Amazon biome. Deforestation data comprises the period of 1986 until 2018 (MapBiomias 2020).

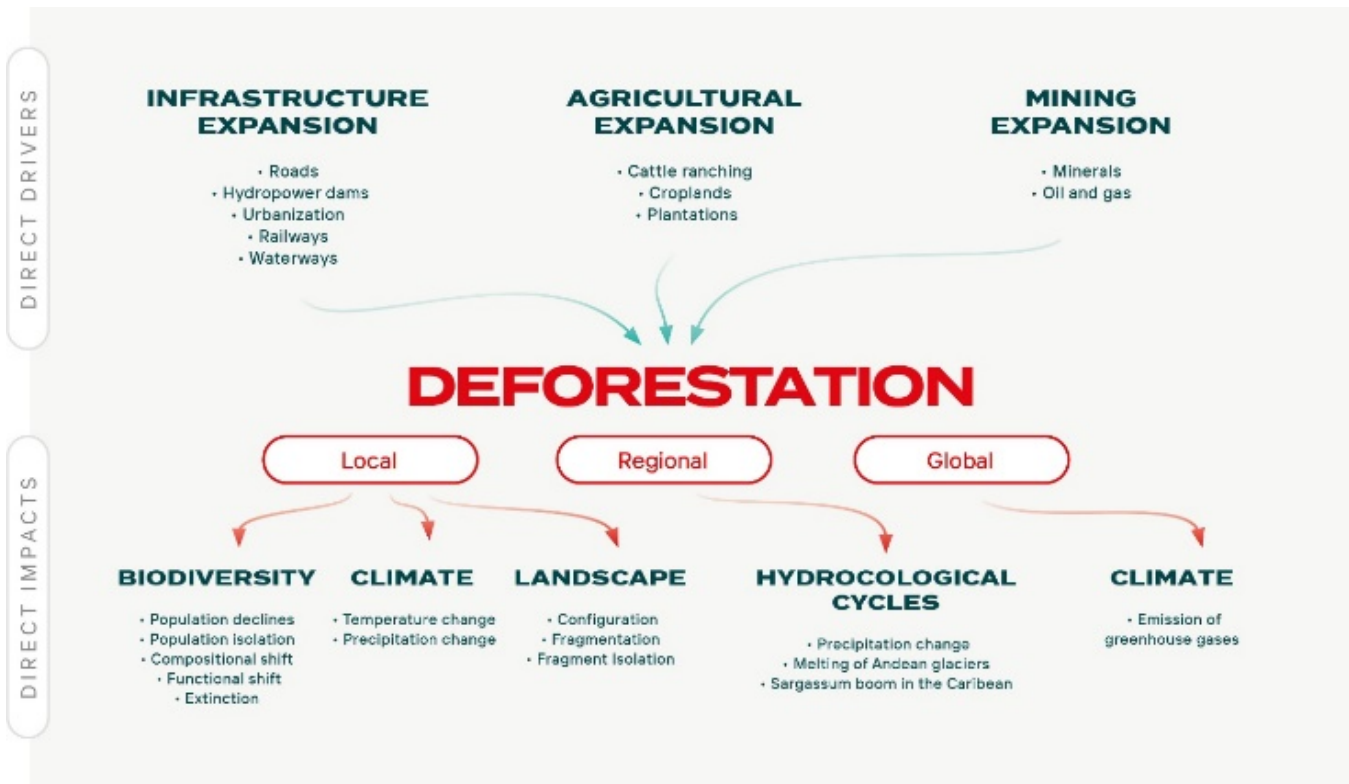


Figure 19.4 The direct drivers of deforestation and its direct impacts at local, regional, and global scales

BR 163 and Transamazônica

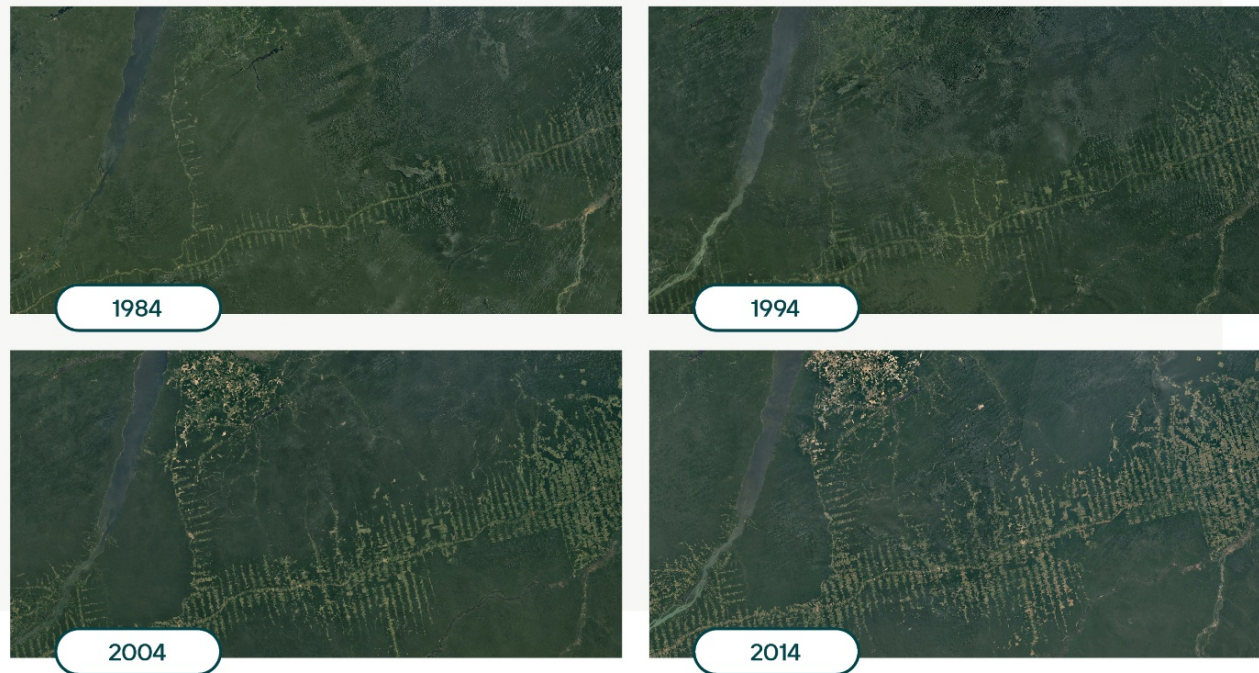


Figure 19.5 Deforestation driven by road building, urbanization, and agricultural expansion, resulting in a fishbone pattern of deforestation. Images from the BR-163 Highway and the Transamazon Highway in the Brazilian Amazon.

areas, numerous secondary roads branching off the main highway were constructed by agricultural settlers, leading to the well-known pattern of fishbone deforestation (Fig. 19.5). In the sections below, we discuss each direct driver of deforestation individually, highlighting, whenever possible, how their relative importance differs across Amazonian countries.

Deforestation can lead to a wide range of direct ecological impacts, which are locally, regionally, and globally relevant. Of the local impacts, biodiversity loss is extremely concerning, with several species of trees, mammals, birds, reptiles, amphibians, and terrestrial invertebrates classified as globally threatened (IUCN 2021). The number of Amazonian threatened species is highly conservative, as the majority of Amazonian species have not even had their status assessed (Box 19.1). Although to date there is no record of a regional extinction, some may have already occurred, especially in

plants and invertebrates, given the large number of species yet to be described in these taxa (Lees and Pimm 2015; Stork 2018; ter Steege *et al.* 2013). Fine-scale endemism may also contribute to undetected extinctions, as many species may only have very restricted geographic distributions (Fernandes 2013), occurring in very small areas (Box 19.2).

Forest fragmentation, or the subdivision of remaining forest cover into variable-sized forest patches, is another local impact of deforestation which reshapes landscape configuration. An increase in forest fragmentation is caused by continued deforestation (Armenteras Barreto *et al.* 2017; Broadbent *et al.* 2008; Laurance *et al.* 2018; Numata *et al.* 2017). Between 1999 and 2002, approximately 5,000 new fragments were created annually due to deforestation in the Brazilian Amazon (Broadbent *et al.* 2008). Although most Amazonian forests remain in large, contiguous blocks, there are over

Box 19.1 Why current tallies of threatened species in the Amazon are gross underestimates

To understand how many Amazonian species are threatened we first need to know how many species there are in the biome. It is estimated that 86% of existing species on Earth and 91% of species in the ocean still await formal scientific description; just 1.7 million species have been catalogued to date (Mora *et al.* 2011). The bulk of this undiscovered diversity is expected to be found in tropical forests like the Amazon. Undertaking the first step and putting names to life on Earth is the greatest impediment to understanding how much of that life is threatened with extinction. Global estimates of over one million threatened species (e.g. IPBES 2019) are derived from estimates of the total number of species that may exist combined with ratios of how many described species are threatened. For example, around 10% of described insects are known to be threatened with extinction.

The number of species officially listed as threatened in the Amazon is thus low for a variety of reasons. Firstly, we are unlikely to have described more than 10% of all the species in the biome. Secondly, even for those species that have been named, the Red Listing process disproportionately covers vertebrate species and not other species on the evolutionary tree of life. Even many vertebrate species which have been officially assessed have been classified as ‘Data Deficient,’ meaning there is insufficient information available to apply the criteria and evaluate their conservation status. The vast majority of described species have not been assessed, either because of a lack of information about their geographic distribution, responses to global change, or population trends, compounded by a lack of human resources to carry out the task of assessment and verification (IPBES 2019). Thirdly, taxonomy is an iterative process and genetic data increasingly point towards a mismeasure of Amazonian taxonomic diversity by uncovering multiple lineages within described species which have not shared genes for very long period of time (as much as millions of years), and which might be better treated at the species level. This taxonomic inflation (Isaac *et al.* 2004) tends to produce more ‘new’ restricted range species, which are thus more likely to meet Red List criteria if their ranges have suffered intensive habitat loss.

The current low level of ‘officially’ threatened species is thus primarily a product of a dearth of knowledge about how many species inhabit the Amazon biome and what proportion of this ‘unknown’ biodiversity is therefore threatened. Secondly it also reflects shortcomings in our knowledge of the response of ‘known’ species to habitat loss, fragmentation, and disturbance, and how their geographic ranges overlap with regions exposed to stressors. In summary, we currently do not yet know how many Amazonian species are threatened.

50,000 fragments between 1-100 ha (Haddad *et al.* 2015).

The distribution of small forest fragments across the Amazon is not even; rather, fragmentation is concentrated along the southern and eastern edges of the biome, along major roads and rivers, and around urban centers (Montibeller *et al.* 2020; Vedovato *et al.* 2016). Deforestation also promotes fragment isolation, with forest patches becoming more distant from one another as well as from large contiguous forested areas (de Almeida *et al.* 2020). While fragment size affects the mainten-

ance of viable populations of both animals and plants, fragment isolation disrupts dispersion and movement. The smaller the fragment, the smaller its chances of sustaining the original pool of forest species (Laurance *et al.* 2011; Michalski *et al.* 2007; Michalski and Peres, 2005), with large-bodied animals and those that are highly dependent on forest habitat being particularly affected (Lees and Peres 2008; Michalski and Peres 2007). Fragment isolation is more harmful to species with low vagility, which are unable to cross open, non-forest matrices (Lees and Peres 2009; Palmeirim *et al.* 2020). To date, negative impacts of fragment size and/or

isolation have been detected throughout the Amazon, affecting leaf bryophytes, trees, palms, birds, carnivores, and primates (Laurance *et al.* 2011; Michalski and Peres 2007). Forest fragments also experience a whole range of edge effects, which lead to their degradation (see Section 19.4.2).

Local temperature and precipitation are also affected by deforestation. Land surface temperature is 1.05-3.06°C higher in pastures and croplands than in nearby forests, with this difference becoming more pronounced during the dry season (Maeda *et al.* 2021). Furthermore, as forest cover decreases at landscape scales, the hotter the landscape becomes, such that landscapes with a lower number of remaining forest patches can be up to 2.5°C hotter than those with greater forest cover (Silvério *et al.* 2015). Forest loss also leads to reduced precipitation (Spracklen *et al.* 2012; Werth 2002), as 25-50% of Amazonian rainfall is recycled from forests (Eltahir and Bras 1994). Therefore, forest loss accrues a decrease in rainfall, increasing the risk of large-scale forest dieback (see Chapter 22 to 24). It is estimated that deforestation has already decreased precipitation by 1.8% across the Amazon (Spracklen and Garcia-Carreras 2015), although changes in rainfall patterns vary across the basin and between the wet and dry seasons (Bagley *et al.* 2014; Costa and Pires 2010). Additionally, widespread deforestation negatively influences precipitation outside the Amazon Basin, influencing regional hydrological cycles. A modeling study suggests that 70% of precipitation in the La Plata Basin; located in Argentina, Bolivia, Brazil, Paraguay, and Uruguay; depends on moisture recycled over the Amazon (Van Der Ent *et al.* 2010).

Regionally, Amazonian deforestation has surprising and very diverse impacts, such as accelerating glacier melting in the Andes and contributing to sargassum blooms in the Caribbean. The burning of recently felled forests as part of the deforestation process (Box 19.3) releases black carbon to the atmosphere. Smoke plumes then transport black carbon to the Andes, where it can be deposited over glaciers, speeding up glacier melt. This process is highly seasonal, peaking during high-fire months

(Magalhães *et al.* 2019). Thousands of kilometers away, in the Caribbean Sea, recent sargassum blooms are likely influenced by anomalous nutrient inputs into the Atlantic resulting from Amazonian deforestation (Wang *et al.* 2019). Sargassum blooms negatively impact tourism and fisheries, and cause community shifts in seagrass meadows and increased coral mortality (Tussenbroek *et al.* 2017).

At a global scale, greenhouse gas emissions are the most-pronounced impact of forest loss in the Amazon. Between 1980 and 2010, the Amazon lost an estimated 283.4 Tg C annually due to deforestation, resulting in yearly emissions of 1040.8 Tg CO₂ (Phillips *et al.* 2017). Deforestation-related emissions are not homogeneous in space or time; for example, Brazil’s annual emissions from Amazonian deforestation are eight times greater than those of Bolivia, the second largest emitter in the basin between 1980 and 2010 (Table 19.1). Overall, emissions have decreased in the region, being higher in the 1980s than the 2000s (Phillips *et al.* 2017).

Table 19.1 Estimated annual carbon loss due to deforestation in the Amazon between 1980-2010 (Phillips *et al.*, 2017).

Country	Carbon loss (Tg C year ⁻¹)
Bolivia	28.6
Brazil	223.9
Colombia	6.5
Ecuador	2.5
French Guiana	1
Guyana	1
Peru	17.9
Suriname	1
Venezuela	1

9.3 Main drivers of deforestation and their associated impacts

19.3.1 Agricultural expansion

Across the Amazon, deforestation has been driven mainly by agricultural expansion, particularly

cattle ranching (Nepstad *et al.* 2009), because of several public policies (See Chapter 14 and 15). In the Brazilian Amazon alone, it is estimated that 80% of deforested areas are occupied by pastures (Ministério do Meio Ambiente 2018). In the early 2000s, large-scale cropland expansion, principally soy, became increasingly important as a driver of deforestation. This pattern reversed (Macedo *et al.* 2012) due to extensive conservation policies, including the soy moratorium, and the creation of a number of protected areas in the regions of Brazil where most soy-related deforestation was taking place (Nepstad *et al.* 2014; Soares-Filho *et al.* 2010). Currently, soy expansion in the Brazilian Amazon occurs mostly on areas that were previously pastures, instead of directly replacing forests (Song *et al.* 2021). In Bolivia, however, soy is still expanding; the region of Santa Cruz has been identified as the largest deforestation hotspot in the Amazon, mainly due to forest conversion to soy fields (Kalamandeen *et al.* 2018; Redo *et al.* 2011). Since the mid-2000s, palm oil has become a growing threat to Amazonian forests, especially in Colombia, Ecuador, Peru, and the eastern part of the Brazilian Amazon (Furumo and Aide, 2017). Although palm oil plantations often replace other agricultural land uses, especially cattle ranching, it has been documented directly replacing primary forests (Castiblanco *et al.* 2013; de Almeida *et al.* 2020; Gutiérrez-Vélez and DeFries 2013). For example, between 2007 and 2013, 11% of deforestation in the Peruvian Amazon was driven by oil palm plantations (Vijay *et al.* 2018). Illicit crops, more specifically coca, is also a driver of deforestation, particularly in Colombia, but also in Bolivia, Ecuador, and Peru (Armenteras *et al.* 2006; Dávalos *et al.* 2016). However, its impact on forest loss is much smaller than that caused by licit commodities (Armenteras Rodríguez *et al.* 2013). Since 2016, following the peace agreement between the Colombian government and the Revolutionary Armed Forces of Colombia (FARC), the role of coca-driven deforestation has decreased, with areas previously in conflict being deforested for pasture, including inside protected areas (Clerici *et al.* 2020; Prem *et al.* 2020).

Direct impacts

Although croplands and pastures hold some animal species, the ecological communities in these areas are dramatically different from those of forests, both in terms of taxonomic and functional composition (Barlow *et al.* 2007; Bregman *et al.* 2016); with almost all forest-dependent species being lost. Among agricultural land uses, pastures hold significantly more taxonomic diversity than areas of mechanized agriculture (e.g. soy fields) for various taxa (Solar *et al.* 2015). Tree plantations also harbor an impoverished subset of forest species. For example, in an oil palm plantation in Peru, <5% of bird species were also found in forests (Srinivas and Koh 2016). In summary, the contribution of agricultural lands to Amazonian biodiversity conservation is negligible (Moura *et al.* 2013), highlighting the irreplaceable value of forests (Barlow *et al.* 2007).

Indirect impacts

In addition to GHG emissions during the deforestation process, pastures further contribute to emissions due to regular burning (Box 19.3) and bovine enteric fermentation (Bustamante *et al.* 2012). Significant changes in the physical and chemical properties of the soil, such as soil compaction and changes in nutrient concentration (Souza Braz *et al.* 2013; Fujisaki *et al.* 2015; Melo *et al.* 2017), are also a result of forest conversion to pastures and croplands in the Amazon. Pesticide and herbicide use in agricultural systems is often excessive in the region (Bogaerts *et al.* 2017; Schiesari *et al.* 2013), but the impacts of this in terrestrial ecosystems have neither been described nor quantified.

19.3.2. Infrastructure

19.3.2.1. Roads

Major official roads and highways, i.e. those built by the government, extend deep into the Amazon; only the western part of the basin is largely road free (Figure 19.6). Official roads, even if unpaved,

Box 19.2 Fine-scale endemism in Amazonian birds reveals threats of deforestation

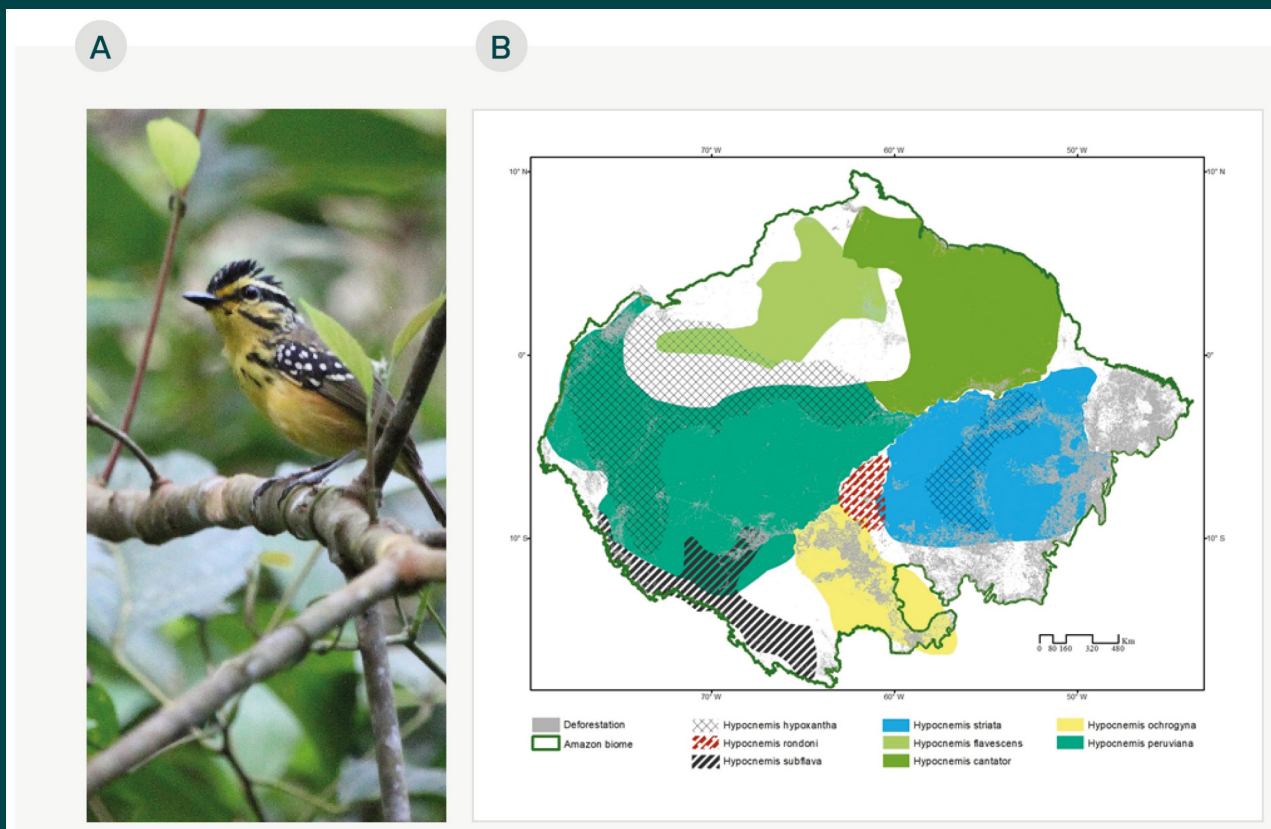


Figure 19.B2 There are two subspecies of Yellow-browed Antbird (*Hypocnemis hypoxantha*) which have disjunct Amazonian distributions. This is the eastern *ochraceiventris* subspecies and it is likely that this species will be subject to taxonomic revision in future. Photo taken in Belterra, in the Brazilian Amazon, by Alexander Lees.

Amazonian biodiversity is non-randomly distributed across the basin, with geographic discontinuities like large wide rivers conspiring alongside topographic heterogeneity, climatic variation and biological interactions to delimit species ranges. Many species of vertebrate have long been recognised as being restricted to Amazonian ‘areas of endemism’ delimited by major rivers; with different ‘replacement species’ present on either side of these fluvial barriers. These areas of endemism are often viewed as planning units for conservation, including protected area designation (da Silva et al. 2005). Understanding patterns of endemism is however dependent on both how complete our biodiversity inventories are, and how refined our taxonomy of different groups is. For example, a revolution in avian taxonomy driven by the usage of molecular toolkits coupled with vocal characters has revealed previously unrecognised fine-scale cryptic diversity. This pointed towards indicated a mismeasure of Amazonian avian diversity because of a reliance on morphological characters to define species limits, characters which may be highly conserved in some lineages of rainforest birds (Fernandes 2013, Pulido-Santacruz et al. 2018). The impact of the usage of new quantitative criteria for species diagnosis has been an increase in the overall number of bird species in Amazonia and an increase in the number of threatened species – as ‘splits’ affecting formerly wide-ranging ‘parent’ species create multiple ‘daughter’ species with smaller range sizes. For example, a taxonomic revision of the ‘Warbling Antbird’ *Hypocnemis cantator* (Thamnophilidae) species complex by Iser et al. (2007) elevated six populations (two of which even occur in sympatry)

Box 19.2 continued

- then regarded as subspecies - to species status based on vocal differences. This taxonomic decision was subsequently reinforced by molecular data (Tobias *et al.* 2008) and later a further member of this species complex - *Hypocnemis rondoni* was later described with a tiny range in the Aripuanã-Machado interfluvium within the Rondonia area of endemism (Whitney *et al.* 2013). These discoveries and taxonomic rearrangements mean that several species in this complex have restricted ranges which overlap the Amazonian Arc of Deforestation and are thus threatened with global extinction – e.g. the Vulnerable *Hypocnemis ochrogyna*. Such fine-scale endemism is likely to be a common Amazonian biogeographic phenomenon which merits urgent consideration in systematic conservation planning efforts (Fernandes 2013).

often spawn networks of unofficial roads, i.e. those built by local actors, providing further access to previously inaccessible forests, resulting in the classic ‘fishbone deforestation’ pattern (Figure 19.5). In terms of total length, the network of unofficial roads is so extensive that it surpasses official ones (Nascimento *et al.* 2021).

Direct impacts

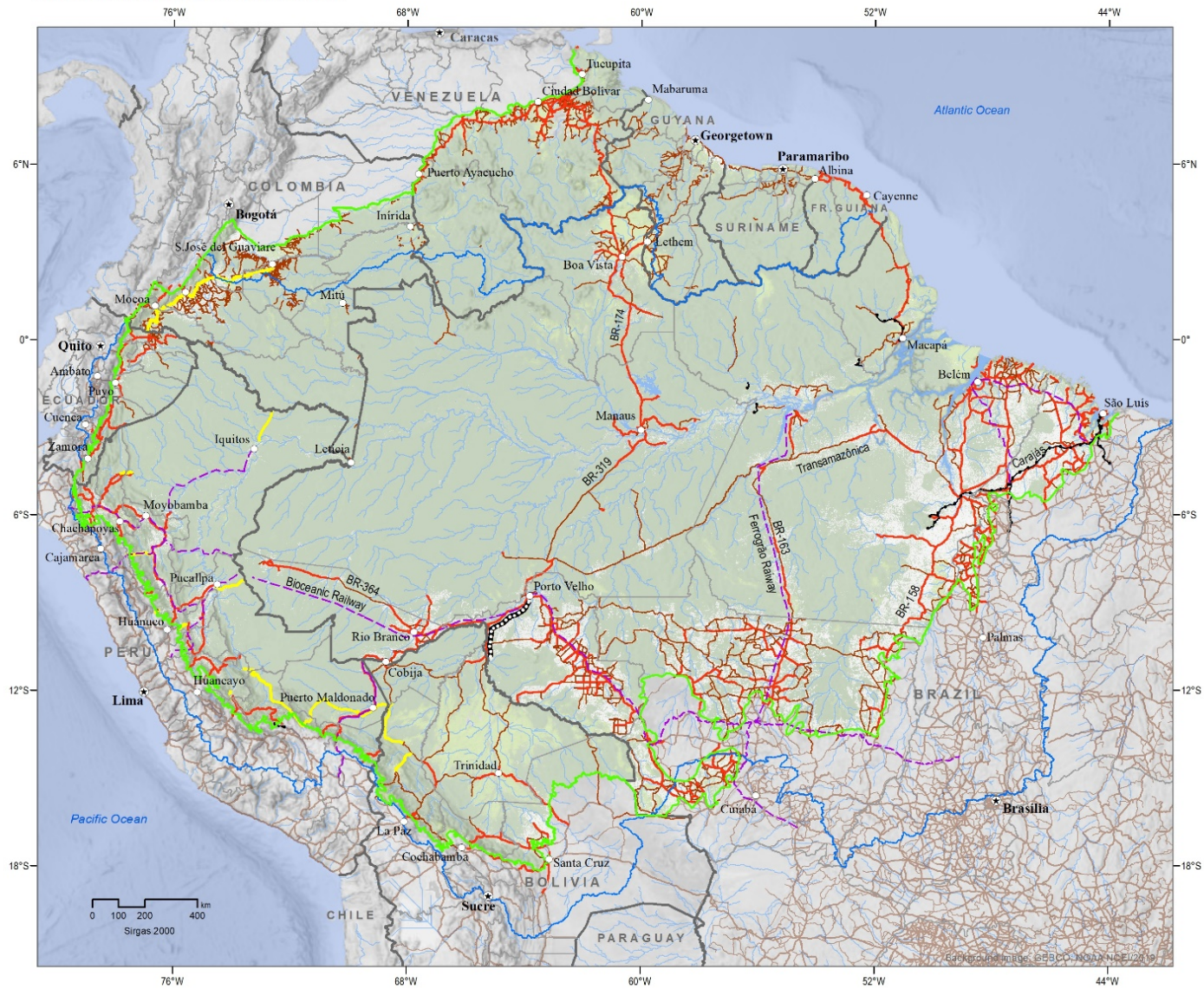
The impacts of roads on terrestrial wildlife in the Amazon are diverse and multi-faceted (Laurance *et al.* 2009). Their direct effects are dwarfed by their indirect impacts, but nonetheless remain important. First, roads lead to high levels of roadkill mortality. For example, over the course of 50 days of monitoring a 15.9 km stretch of road in Napo (in the western Amazon), 593 animals were killed, including reptiles, amphibians, birds, and mammals (Filius *et al.* 2020). Occasionally, roadkill includes threatened species, such as Harpy Eagles, Giant Anteaters, Giant Armadillos, Giant Otters, Red-faced Spider Monkeys, Lowland Tapirs, and Red-billed Toucans (de Freitas *et al.* 2017; Medeiros 2019). Given the approximately 40,000 km of official roads across the Amazon, roadkill is highly underreported and understudied. Second, roads can act as direct drivers of habitat fragmentation, isolating populations on either side (Lees and Peres 2009). Widths of just 12-25 m can restrict the movements of bird species adapted to the forest understory (Laurance *et al.* 2004; Laurance *et al.* 2009).

Indirect impacts

The greatest impacts of roads are indirect. The construction of official and, subsequently, unofficial roads increases land values, as it makes agriculture and ranching more profitable, since products can be transported to urban centers and ports (Perz *et al.* 2008). In turn, higher land prices lead to land speculation that motivates deforestation to secure land possession (Fearnside 2005). Roads also induce migration, leading to invasions and settlements (Mäki *et al.* 2001; Perz *et al.* 2007). As a result, the presence of roads is strongly associated with deforestation in the Brazilian (Laurance *et al.* 2002; Pfaff *et al.* 2007), Peruvian (Bax *et al.* 2016; Chávez Michaelsen *et al.* 2013; Naughton-Treves 2004), and Ecuadorian Amazon. However, in the case of Ecuador road construction is linked to oil concessions (Mena *et al.* 2006; Sierra 2000). The paving of official roads provokes direct deforestation along highways (Fearnside 2007; Asner *et al.* 2010) and induces displaced deforestation; pastures are often sold to be converted into more profitable croplands, such as soy, and ranchers who have sold their land move into rainforest areas to establish new ranches (Arima *et al.* 2011; Richards *et al.* 2014).

Roads also stimulate forest degradation, including selective logging (Amachar *et al.* 2009; Merry *et al.* 2009; Asner *et al.* 2006), as they provide machinery access (e.g. logging trucks, skidders) to areas that still contain valuable timber. The opposite can also be true; often loggers open small roads to extract

OFFICIAL ROADS AND RAILWAYS



SPA, 2021

- Amazon biome
- Amazon basin
- State border
- National border
- State/national capital
- Forest
- Non-forest areas or without vegetation
- Areas of agriculture and ranching
- Railways**
- Built
- Planned
- Route of the former railroad Madeira-Mamoré
- Roads**
- Planned
- Paved
- Unpaved (or no information)

Sources: RAISG (Roads & railways in 2020; reference boundaries; cities); MapBiomos Amazonia (Land use in 2018); WCS (new classification Amazon basin)

Figure 19.6 Planned (yellow), paved (red), and unpaved (brown) roads across the Amazon, as well as existing (black) and planned (purple) railways. The Amazon biome is outlined in green, while the Amazon Basin (the limit used in other chapters of this report) is outlined in blue.

Box 19.3 Fires, deforestation, and drought lead to forest degradation

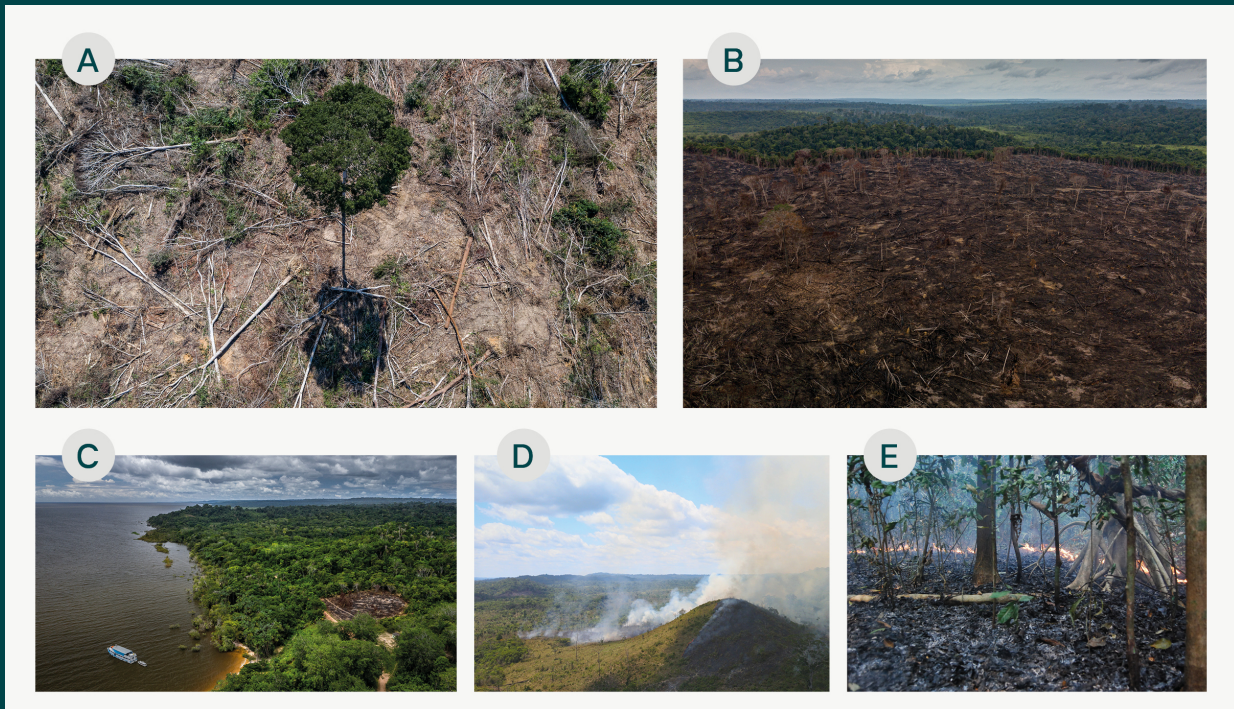


Figure 19.B3 A) An area recently deforested (Photo by Marizilda Cruppe/Rede Amazônia Sustentável); B) A large deforested area that has been recently burned (Photo by Flávio Forner/Rede Amazônia Sustentável); C) A small area deforested and burned for subsistence agriculture (Photo by Marizilda Cruppe/Rede Amazônia Sustentável); D) Fire in pastures (Photo by André Muggiati); E) Understory (Photo by Erika Berenguer).

Fires are an intrinsic part of the deforestation process in the Amazon (Barlow *et al.* 2020). First the land is cleared, and trees can be felled using a variety of methods, from chainsaws to bulldozers. Then, felled vegetation is left to dry for a period of a few weeks to a few months into the dry season. When the felled vegetation is dry, it is set on fire, transforming most of the biomass to ash. The land is then ready to be planted. Fires are also used in subsistence agriculture, which is often called slash-and-burn. Traditionally used by Indigenous Peoples and small landowners, fires are used to burn a small patch of land which has been recently deforested. After a few years of agricultural use, this area will be abandoned, and left as fallow, as the farmer rotates agricultural production to another fallow. Finally, fires are also used as a common management tool in pastures, to remove weeds and small trees and increase productivity. However, fires from deforestation, subsistence agriculture, or pastures can escape into surrounding agricultural areas, leading to economic losses as crops, fences, and buildings are burned (Cammelli *et al.* 2019). They can also escape to surrounding forests if it is a dry year, as leaf litter with <23% moisture can sustain a fire (Ray *et al.* 2005). Fires in Amazonian forests, or understory fires, tend to be of low intensity, with flame heights ranging between 10-50 cm, and slow moving, burning 300 m per day (Cochrane *et al.* 1999; Ray *et al.* 2005). Understory fires can be blocked by the canopy and hard to detect by remote sensing approaches (Pessoa *et al.* 2020). However, recent technological developments, such as the Visible Infrared Imaging Radiometer Suite (VIIRS) and the Continuous Degradation Detection (CODED) have been fundamental in mapping understory fires across the Amazon, thus helping to reveal the true extent of fires and overall forest degradation (Bullock *et al.* 2020; Oliva and Schroeder 2015; Schroeder *et al.* 2014).

target trees (Gutierrez-Velez and MacDicken 2008; Johns *et al.* 1996; Uhl and Vieira 1989), which can then drive additional degradation. Proximity to roads is also highly correlated with forest fires, even in non-drought years (Alencar *et al.* 2004). This is due to the influx of migrants and agricultural expansion surrounding roads (Figure 19.5), thus resulting in more deforestation and pasture-related fires, which can escape into forested areas (Box 19.3).

19.3.2.2 Hydropower dams

Substantial energy resources exist in the Amazon, some actively exploited and others as potential reserves (Ferreira *et al.* 2014). There are currently 307 hydropower dams either in operation or under construction, with proposals for at least 239 more (Figure 19.7). Of these, some are considered megadams, of >1 GW capacity. Hydroelectric dams not only disrupt aquatic ecosystems (Chapter 20), they also have severe consequences for terrestrial ones.

Direct impacts

Most hydropower dams require an area to be flooded, acting as a reservoir. Both floodplain (*várzea*) and upland (*terra firme*) forests are killed by reservoir flooding (Lees *et al.* 2016), resulting in high levels of CO₂ and CH₄ emissions due to the decomposition of submerged trees (Figure 19.8; see Chapter 20). Although seasonally flooded forests can survive several months under water, they die if flooded year-round. Forests bordering the reservoir also suffer stress, including reductions in the rates of photosynthesis of trees (dos Santos Junior *et al.* 2015). Depending on local topography, islands containing upland forests can be formed after flooding. Newly-formed islands suffer from edge effects and fragmentation, as they have been cut off from the rest of the previously contiguous forest. Reservoir islands have significantly different species composition of both fauna and flora than adjacent mainland areas (Tourinho 2020, Benchimol and Peres 2015), a pattern particularly pronounced on small islands, where large-bodied fauna be-

come extinct (Benchimol and Peres 2020). Invertebrates are also negatively impacted by flooding; one study found that thirty years after the reservoir was filled, several islands completely lacked dung beetle species (Storck-Tonon *et al.* 2020). Dams also affect forests downstream; altered flood regimes can even impact forests 125 km away from the reservoir (Schongart *et al.* 2021), resulting in large-scale tree mortality (Assahira *et al.* 2017), leading to the loss of crucial habitat for a variety of organisms (e.g. arboreal mammals, birds, and plants) which can become locally extinct (Lees *et al.* 2016). Finally, dams can also affect the status of protected areas; for example, the planned São Luiz do Tapajós Dam resulted in part of Amazonia National Park being degazetted in Brazil (Fearnside 2015a).

Indirect impacts

The construction of hydroelectric dams also leads to indirect impacts; for example, the population attracted to the region boosts deforestation in the area surrounding the dam (Jiang *et al.* 2018; Velastegui-Montoya *et al.* 2020). Furthermore, dam construction often results in socio-economic problems, such as increases in violence and lawlessness, and the displacement and destruction of the livelihoods of both Indigenous and non-Indigenous communities (Athayde *et al.* 2019; Castro-Diaz *et al.* 2018; Moran 2020; Randell 2017).

19.3.2.3 Urbanization

Approximately 70% of Amazonians live in urban centers (Padoch, C. *et al.* 2008; Parry *et al.* 2014), with the largest city, Manaus, hosting >2.2 million inhabitants (IBGE 2021). Urban expansion is currently concentrated in small and medium cities (Richards and VanWey 2015; Tritsch and Le Tourneau 2016) and results from various processes, from rural-urban and urban-urban migration to displacement due to armed conflict and intrinsic population growth (Camargo *et al.* 2020; Perz *et al.* 2010; Randell and VanWey 2014; Rudel *et al.* 2002). See Chapter 14 for more details on historical migration to Amazonian cities.



Figure 19.7 Planned and active hydropower dams and waterways across the Amazon biome. The Amazon biome is outlined in green, while the Amazonian Basin (used in other chapters in this report) is outlined in blue. Sources: WCS Venticinque 2016; RAISG 2020.

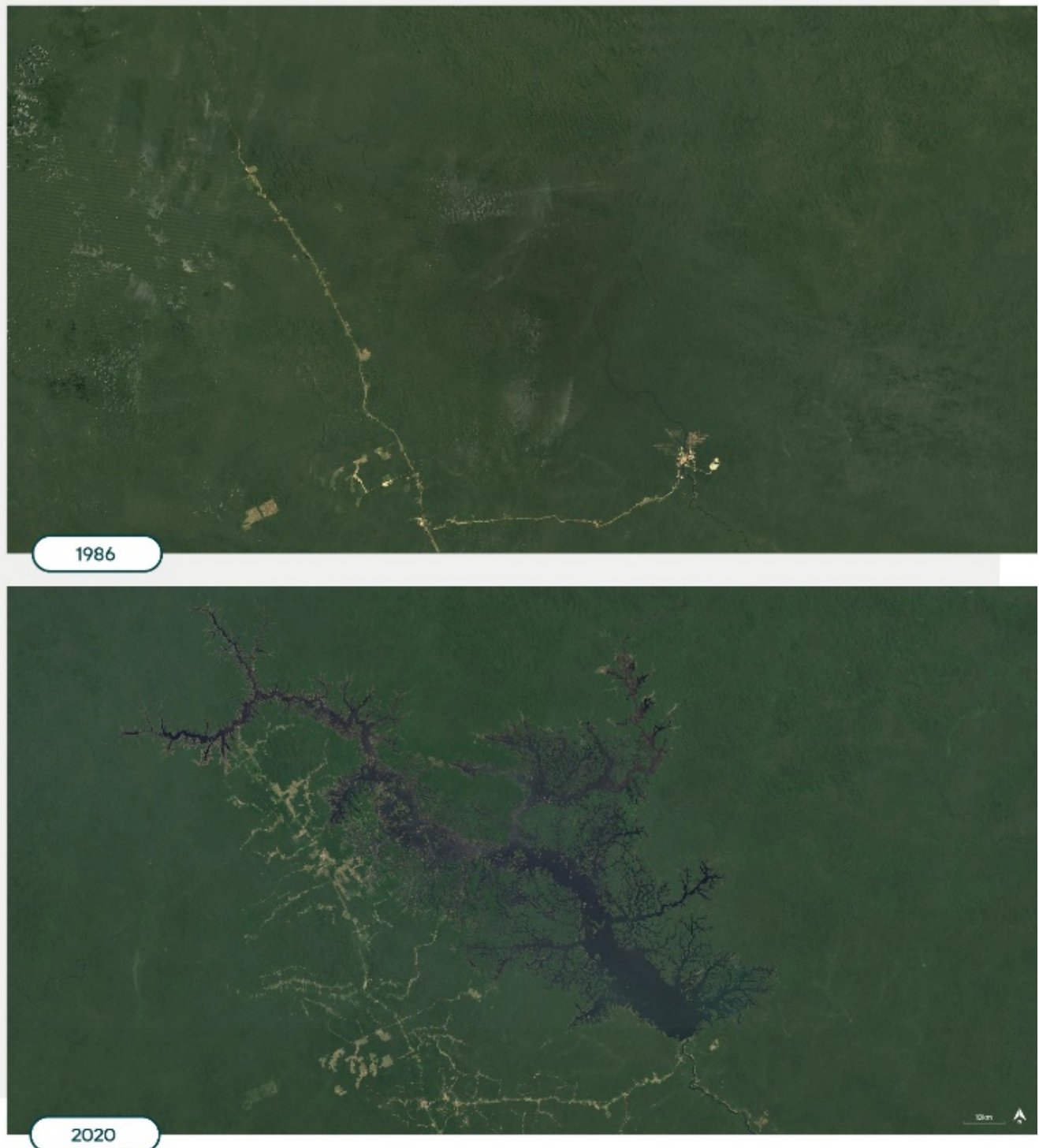


Figure 19.8 Flooding of the reservoir of the Balbina dam in Brazil. a) Before (1986) and b) after (2020) the flooding. Source Google Earth.

Direct impacts

Urban and suburban sprawl increase deforestation (Jorge *et al.* 2020), especially in frontier settlements. Amazonian urban biodiversity is poorly studied, but is generally taxonomically depauperate and typically dominated by a small subset of common species found in secondary habitats (Lees and Moura 2017; Rico-Silva *et al.* 2021). As observed elsewhere, urbanization also influences the local climate, which becomes hotter (de Oliveira *et al.* 2020; Souza *et al.* 2016).

Indirect impacts

Many rural-urban migrants continue to consume forest resources, therefore playing a role in forest-use decisions (Chaves *et al.* 2021; Padoch, C. *et al.* 2008). For example, surveys of two Amazonian cities on the Madeira River showed that 79% of urban households consumed bushmeat, including terrestrial mammals and birds (Parry *et al.* 2014).

Animals hunted for urban consumption can be sourced from forests located up to 800 kilometers away and frequently include threatened species, such as Black Curassow, Giant Armadillo, Gray Tinamou, Red-faced Spider Monkey, Lowland Tapir, Red-billed Toucan, and White-lipped Peccary (Bodmer and Lozano 2001; Bizri *et al.* 2020; IUCN 2021; Parry *et al.* 2010, 2014).

19.3.2.4 Railways and waterways

Across the Amazon, the density of railways and waterways is much lower than that of roads (Figures 19.6 and 19.7). As a result, there are few studies on the impacts of these forms of infrastructure on terrestrial ecosystems (See Chapter 20 for impacts of waterways on aquatic ecosystems).

Direct impacts

Opening railways in the Amazon results in deforestation and fragmentation of the forest that is cut by the rail line, impacting the movement of animals that cannot cross even narrow clearings

(Laurance *et al.* 2009). There is currently no published investigation into the direct impacts of waterways on Amazonian forests.

Indirect impacts

The limited movement of passengers along railways mean that levels of adjacent deforestation are far lower relative to roads. However, railways can still indirectly induce deforestation. For example, between 1984 and 2014, approximately 30,000 km² of forests were lost in the area of influence of the Carajás Railway in the Brazilian Amazon (Santos *et al.* 2020). However, some of these impacts are hard to disentangle from that of roads built near some of the railway stations.

Railways present important risks for the future of the Amazon. The “Ferro Grão” Railway, also located in the Brazilian Amazon, would link soy areas in Mato Grosso (the southern Amazon) to the port in Miritituba on the lower Tapajós River, with access to the Amazon River (Figure 19.6). The lower freight costs of Mato Grosso’s soy transported by the Ferro Grão Railway can be expected to contribute to the conversion of pasture to soybeans, possible leading to displaced deforestation, as seen elsewhere when roads were paved (Fearnside and Figureido 2016). Another proposed railway would connect Mato Grosso to the port of Bayóvar in the Peruvian state of Piura (Dourojeanni 2015). This railway, known as the “Railway to the Pacific” in Peru, could also contribute to soy expansion and displaced deforestation in Brazil. The same pattern of displaced deforestation is expected as a result of the proposed Tapajós and Tocantins waterways, which would stimulate pasture conversion to large croplands (Fearnside 2001).

19.3.3. Mining

19.3.3.1 Minerals

Mining is a major source of environmental impacts in the Amazon, with 45,065 mining concessions either under operation or waiting for approval, of which 21,536 overlap with protected areas and

Indigenous lands (Figure 19.9). While some minerals; such as bauxite, copper, and iron ore (Souza-Filho *et al.* 2021); are extracted through legal operations conducted by large corporations (Sonter *et al.* 2017), gold mining is largely illegal (Asner and Tupayachi 2017; Sousa *et al.* 2011). Despite its illegality, gold mining has become far from artisanal, and is now a semi-mechanized activity, employing large and expensive machinery such as prospecting drills and hydraulic excavators (Massaro and de Theije 2018; Springer *et al.* 2020; Tedesco 2013).

Direct impacts

Overall, the extent of mining-driven deforestation is far smaller than that caused by agricultural expansion (see Section 19.3.1). However, it still represents the main driver of forest loss in French Guiana, Guyana, Suriname and parts of Peru (Dezécache *et al.* 2017; Caballero-Espejo *et al.* 2018). For example, in Guyana, mining led to the loss of c. 89,000 ha of forests between 1990 and 2019, an area 18 times larger than that lost to agricultural expansion in the same period (Guyana Forestry Commission 2020). In Suriname, 71% of deforestation is attributed to mining (The Republic of Suriname 2019). In the southeastern Peruvian Amazon, approximately 96,000 ha were deforested due to mining between 1985 and 2017 (Caballero-Espejo *et al.* 2018), including areas inside the Tambopata National Reserve and its buffer zone (Asner and Tupayachi 2017). In a single year, deforestation due to gold mining in the Madre de Dios region resulted in the direct loss of c. 1.12 Tg C (Csillik and Asner 2020).

Another direct impact of mining is the potential biodiversity loss in one of the Amazon's smallest ecosystems, the cangas. This is a ferruginous savanna-like ecosystem associated with ironstone outcrops in the eastern Amazon (Skirycz *et al.* 2014). It originally occupied an area of 144 km², but 20% of this area has been lost to mining of iron ore (Souza-Filho *et al.* 2019). Despite the small area occupied, the Amazonian cangas has 38 endemic vascular plants, 24 of which are considered rare (Giulietti *et al.* 2019). The cangas is also rich in endemic cave-

dwelling fauna (Giupponi and Miranda 2016; Jaffé *et al.* 2018). Little is known about the impacts of mining in this unique ecosystem. The direct and indirect impacts of mining on aquatic ecosystems and human wellbeing are addressed in Chapters 20 and 21, respectively.

Indirect impacts

Indirect impacts of mining activities are often greater than direct ones. In Brazil, for instance, mining was responsible for the loss of 11,670 km² of Amazonian forests between 2000 and 2015, corresponding to 9% of all deforestation in that period (Sonter *et al.* 2017), with effects extending 70 km beyond the boundaries of mining concessions. Mining also stimulates forest loss by motivating the construction of roads and other transportation infrastructure that leads to high levels of human migration and consequent deforestation (Fearnside 2019; Sonter *et al.* 2017). The Carajás Railway, in the Brazilian Amazon, is an example of this (see Section 19.3.2.4). Finally, mining can lead to increased logging and deforestation for charcoal production, especially to be used in pig iron production (Sonter *et al.* 2015).

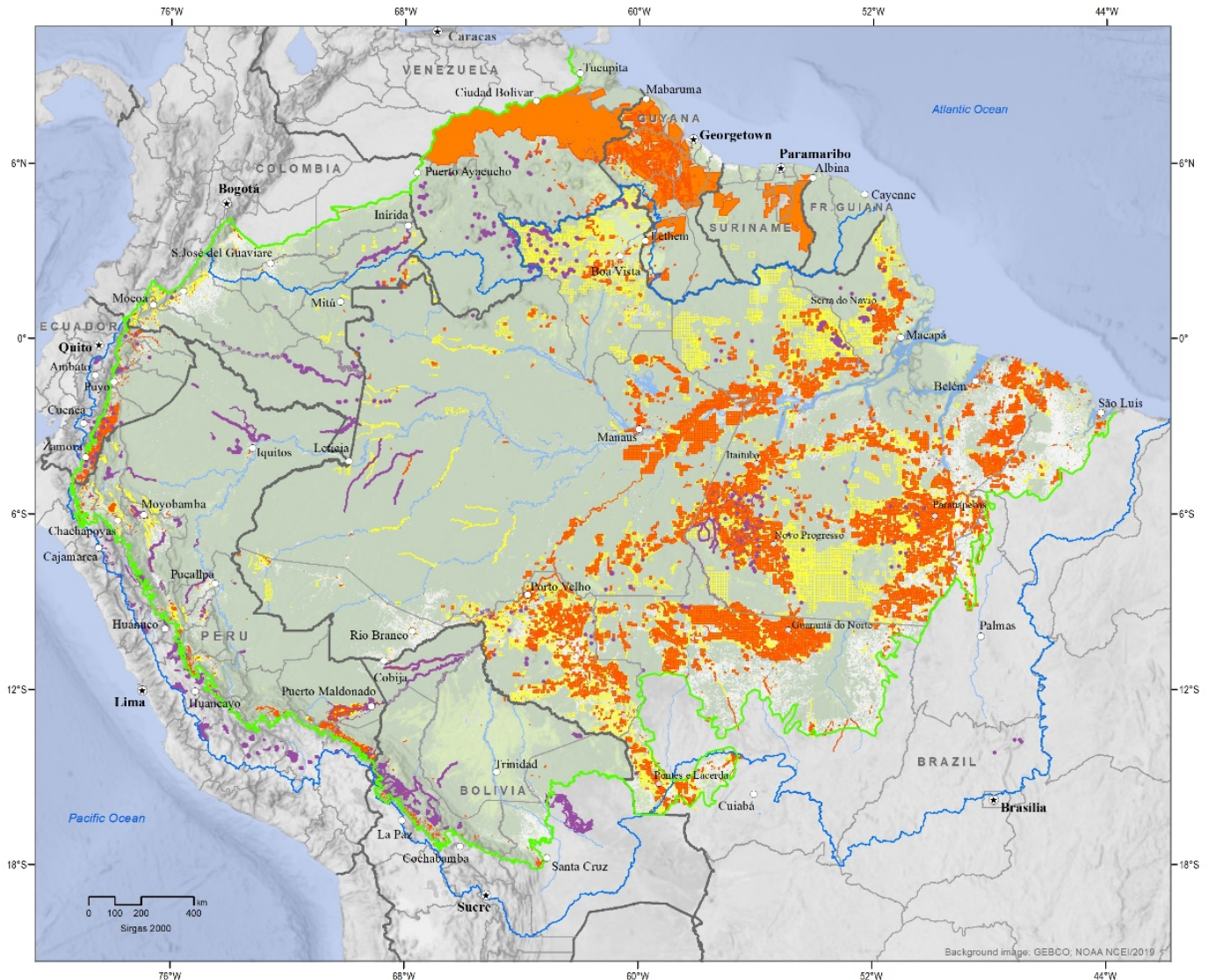
19.3.3.2 Oil and gas

Oil and gas exploitation occur mainly in the western Amazon, where exploitation of crude oil started in the 1940s, and grew substantially from the 1970s onwards (Finer *et al.* 2009; San Sebastián and Hurtig 2004). Currently, 192 oil and gas leases are under production and 33 are being prospected; some of these overlap with protected areas and Indigenous lands (Figure 19.10).

Direct impacts

Major threats from hydrocarbon development include deforestation and oil spills, as has occurred on numerous occasions in Colombia, Ecuador, and Peru (Cardona 2020; San Sebastian and Hurtig 2004; Vargas-Cuentas and Gonzalez 2019). For example, in the northeastern Ecuadorian Amazon, 464 oil spills occurred between 2001 and 2011,

MINING: OFFICIAL CONCESSIONS AND ILLEGAL ACTIVITIES



SPA, 2021

Sources: RAISG (Official mining concessions and illegal mining activities in 2020; reference boundaries; cities); MapBiomas Amazonia Land use in 2018); WCS (new classification Amazon basin)

- Amazon biome
 - Amazon basin
 - State border
 - National border
 - ⊙ National capital
 - State capital
 - Main city
- Land use
 - Forest
 - Non-forest areas or without vegetation
 - Areas of agriculture and ranching
- Illegal mining
 - Locations where illegal mining is occurring
 - Rivers with ongoing illegal mining activities
- Official mining concession areas
 - Potential or applied for
 - In operation or under exploration

Figure 19.9 Illegal (purple) and legal mining that is either planned (yellow) or under production (orange) across the Amazon. The Amazon biome is outlined in green, while the Amazon Basin (used in other chapters) is outlined in blue. Sources: WCS-Venticinque 2016; RAISG 2020.

OIL AND GAS LEASES ACROSS AMAZON

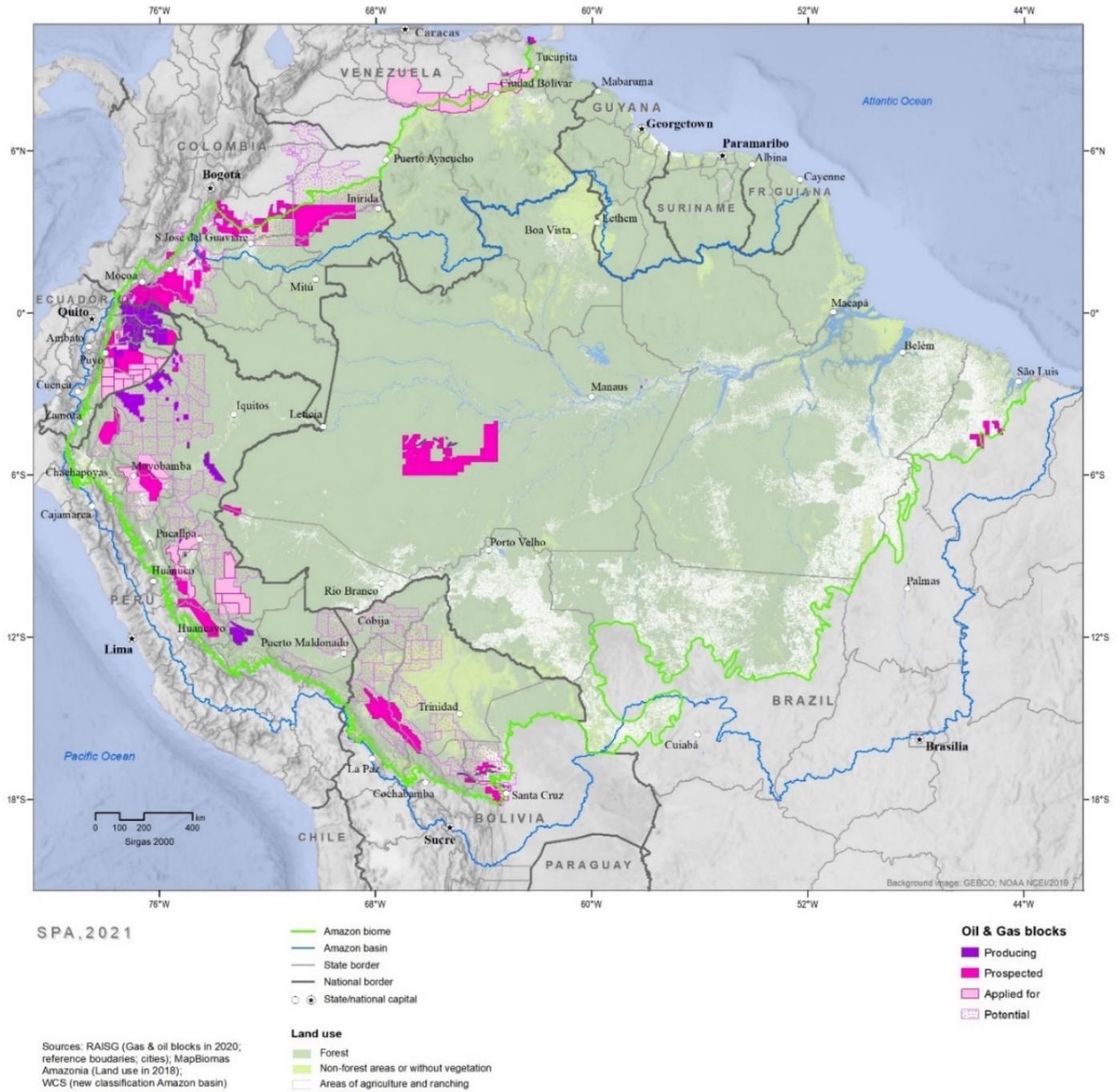


Figure 19.10 Oil and gas leases across the Amazon. The Amazon biome is outlined in green, while the Amazonian Basin (used in other chapters in this report) is outlined in blue. Sources: WCS-Venticinque 2016; RAISG 2020.

totaling 10,000 metric tons of crude oil released into the environment (Durango-Cordero *et al.* 2018). This corresponds to approximately 25% of the amount leaked in the Exxon Valdez oil spill. However, the number of oil spills across the Amazon is largely underestimated (Orta-Martínez *et al.* 2007). The impacts of oil spills on terrestrial ecosystems remain poorly understood. Nevertheless, it has been reported that Lowland Tapirs, Pacas, Collared Peccaries, and Red-brocket Deer consume soil and water contaminated by oil spilled from oil tanks and abandoned wells (Orta-Martínez *et al.* 2018). It is unclear how this consumption may affect animal populations.

Indirect impacts

As is the case of mineral exploitation, indirect effects of oil and gas exploitation on terrestrial ecosystems dwarf direct ones. The construction of a large road network to access oil fields has led to colonization of previously remote areas, especially in Ecuador, resulting in increased deforestation (Bilborrow *et al.* 2004). Animal populations around these roads are negatively affected (Zapata-Ríos *et al.* 2006), with large and medium-sized mammals and game birds declining by 80% (Suárez *et al.* 2013). Some of these roads penetrate protected areas and Indigenous lands, where they have led to deforestation, habitat fragmentation, logging, overhunting, vehicle-wildlife collision, and soil erosion (Finer *et al.* 2009).

19.4 Degradation: An overview of direct drivers and impacts

Forest degradation is defined as the reduction of the overall capacity of a forest to supply goods and services (Parrotta *et al.* 2012), representing a loss in ecological value of the area affected (Putz and Redford 2010). While deforestation is binary (i.e. either the forest is present or absent), forest degradation is characterized by an impact gradient, ranging from forests with little, although significant, loss of ecological value, to those suffering with severe disruption to ecosystem functions and processes (Barreto *et al.* 2021; Berenguer *et al.* 2014; Longo *et*

al. 2020). In total, c. 1 million km² of Amazonian forests were degraded by 2017 (Figure 19.11), equivalent to 17% of the biome, mostly in Brazil (Bullock, Woodcock, Souza, *et al.* 2020). These degraded forests are a persistent part of the landscape, as only 14% of them were later deforested (Bullock, Woodcock, Souza, *et al.* 2020).

Several anthropogenic disturbances act as direct drivers of forest degradation in the Amazon (Figure 19.12), such as understory fires, selective logging, edge effects, hunting, and climate change (Andrade *et al.* 2017; Barlow *et al.* 2016; Bustamante *et al.* 2016; Phillips *et al.* 2017). A forest can be degraded by the occurrence of a single or multiple disturbances (Michalski and Peres 2017; Nepstad *et al.* 1999). For example, a forest fragment experiencing edge effects may also be logged and/or burned (Figure 19.13). Between 1995 and 2017, 29% of degraded forests across the biome experienced multiple disturbances (Bullock, Woodcock, Souza, *et al.* 2020). Furthermore, climate change is an omnipresent driver of degradation, affecting all Amazonian forests, whether already degraded or not (see Chapter 24).

A disturbed Amazonian forest can be characterized as degraded due to significant changes in its structure, microclimate, and biodiversity, all of which impact ecosystem functions and processes. For example, understory fires, selective logging, and edge effects can lead to elevated tree mortality, increased liana dominance, greater presence of canopy gaps, decrease in forest basal area and carbon stocks, changes in stem density, and a decrease in the presence of large trees, accompanied by an increase in the occurrence of small-diameter individuals (Alencar *et al.* 2015; Balch *et al.* 2011; Barlow and Peres 2008; Berenguer *et al.* 2014; Brando *et al.* 2014; Laurance *et al.* 2006, 2011; Pereira *et al.* 2002; Schulze and Zweede 2006; Silva *et al.* 2018; Uhl and Vieira 1989). These structural changes can result in significantly higher light intensity, temperature, wind exposure, and vapor pressure deficit, as well as lower air and soil humidity (Balch *et al.* 2008; Kapos 1989; Laurance *et al.* 2011; Mollinari *et al.* 2019). These abiotic and bi-



Figure 19.11 Forests degraded (red), and deforested (White) across the Amazon Basin. The Amazon biome is outlined in green, while the Amazonian limits used in other chapters in this report is outlined in blue. Sources: Bullock, Woodcock, Souza, et al., 2020; Mapbiomas 2020.

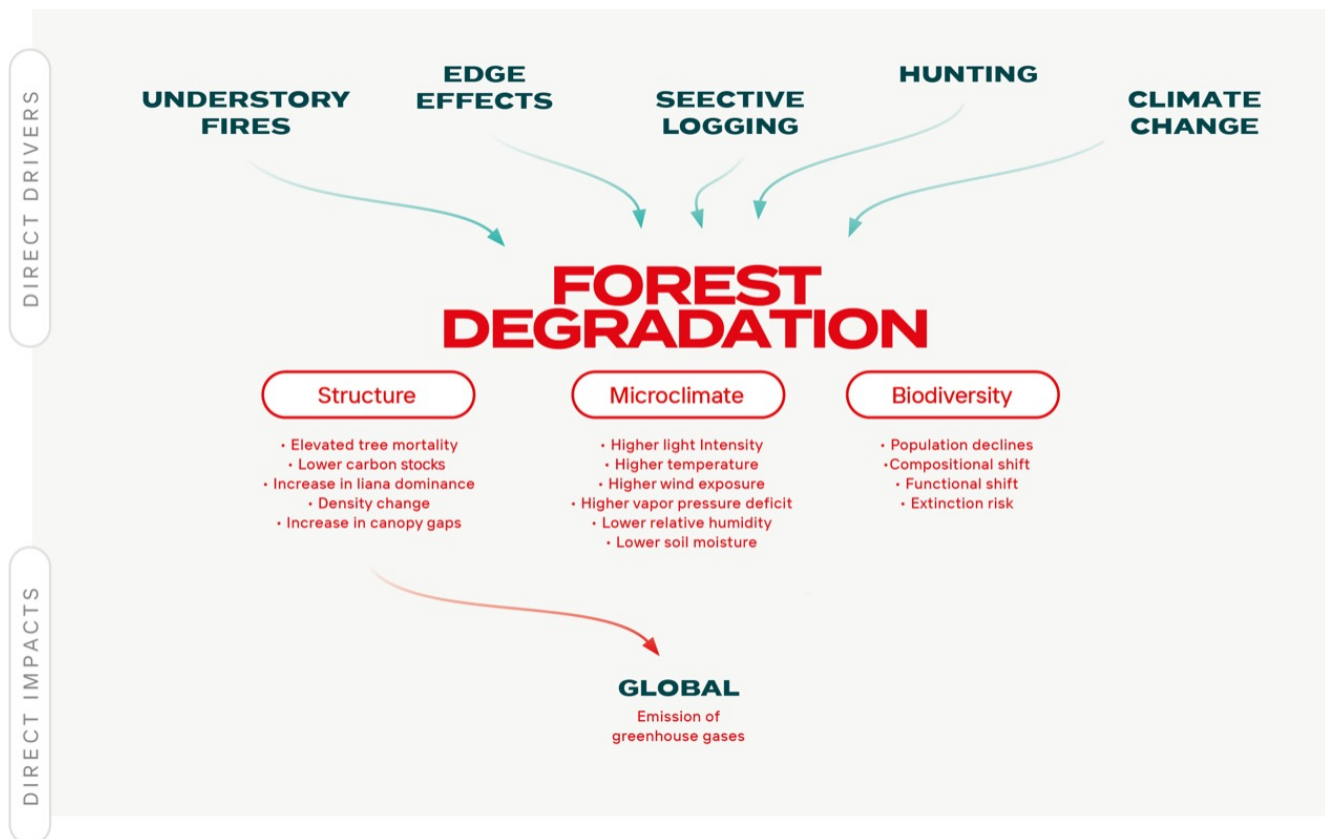


Figure 19.12 Direct drivers of forest degradation in Amazonia as well as their direct impacts at the local and global scales.

otic changes affect biodiversity, which is further impacted by hunting. Communities of both fauna and flora will experience compositional and functional shifts, with some species declining severely, leading to local extinctions (Barlow *et al.* 2016; de Andrade *et al.* 2014; Miranda *et al.* 2020; Paolucci *et al.* 2016; Zapata-Ríos *et al.* 2009). The duration of the impacts of anthropogenic disturbances on Amazonian forests varies depending on the nature, frequency, and intensity of the disturbance; while logged forests may return to baseline carbon stocks within a few decades (Rutishauser *et al.* 2015), burned forests may never recover their original stocks (Silva *et al.* 2018). Recovery of degraded forests is also dependent on their landscape context, i.e. whether there are forests nearby that can act as sources of seeds and animals, thus speeding up recovery.

There is a large gap in our understanding of the re-

gional impacts of forest degradation; a knowledge gap with an urgent need to be filled. Globally, the main impact of forest degradation is an increase in greenhouse gas emissions due to carbon loss (Aguiar *et al.* 2016). It is estimated that CO₂ emissions resulting from forest degradation already surpasses those from deforestation (Baccini *et al.* 2017; Qin *et al.* 2021).

19.4.1 Understory fires

In most years, and in most undisturbed forests, the high moisture load in the understory of Amazonian primary forests keeps flammability levels close to zero (Nepstad *et al.* 2004, Ray *et al.* 2005, 2010). However, thousands of hectares of forests burn across the basin every year (Aragão *et al.* 2018; Withey *et al.* 2018). These understory fires, also called forest fires or wildfires, spread slowly, have flame heights of 10-50 cm, and release little energy



Figure 19.13 A small forest fragment, surrounded by soy fields, which has been selectively logged and then burned during the 2015 El Niño, in Belterra, Brazil. Photo: Marizilda Cruppe/Rede Amazônia Sustentável.

(≤ 250 kW/m) (Brando *et al.* 2014, Cochrane 2003). However, their impacts can be enormous as Amazonian forests have not co-evolved with fires.

Direct impacts

Understory fires cause important long-term ecological impacts. They cause high levels of stem mortality, negatively affecting carbon stocks (Barlow *et al.* 2003; Berenguer *et al.* 2014; Brando *et al.* 2019), and forests take many years to recover. One study conducted across the Amazon estimated that burned forests have carbon stocks that are 25% lower than expected 30 years after fires, with growth and mortality dynamics suggesting recovery had plateaued (Silva *et al.* 2018). Fire impacts also vary regionally. Mortality rates tend to be lower in forests in the drier regions of the Amazon, potentially reflecting regional variation in bark thickness (Staver *et al.* 2020). Impacts are much higher in flooded forests than in *terra firme* (Box 19.4). In the south of the basin, in the ecotone between the Amazon and the Cerrado, native and exotic grass species have been observed to invade burned forests (Silvério 2013); a pattern not recorded elsewhere in the region. In the southwest of the basin, burned forests have experienced an increase in dominance by native bamboo species (Silva *et al.* 2021). Both grass and bamboo invasion significantly increase the flammability of these

already burned forests (Dalagnol *et al.* 2018; Silvério *et al.* 2013).

High tree mortality caused by understory fires leads to significant taxonomic and functional changes in the plant community, which loses high-wood density climax species and sees a dominance of light-wood pioneer ones (Barlow *et al.* 2012; Berenguer *et al.* 2018). It is currently unknown whether burned forests will eventually return to their original plant community composition. Due to changes in forest structure and in the abundance of fruiting trees, fauna is also impacted by understory fires. For example, fires extirpate many forest specialist birds and mammals, while favoring species that occur in forest edges and secondary forests (Barlow and Peres 2004, 2006). Additionally, understory fires negatively affect the abundance of several orders of leaf-litter invertebrates, such as Coleoptera, Collembola, Dermaptera, Diptera, Formicidae, Isoptera, Hemiptera, and Orthoptera (França *et al.* 2020; Silveira *et al.* 2010). These changes are long-lasting even in continuous forests where there should be no barriers to recolonization (Mestre *et al.* 2013). All these direct impacts of a young secondary forest, with an open canopy and few large trees (Barlow and Peres 2008).

Future of fires and their impacts

Interactions between climate and land-use change across the Amazon can create the conditions needed for more widespread and intense fires (Malhi *et al.* 2008, de Faria *et al.* 2017, Brando *et al.* 2019). As the climate changes, we expect to observe increased frequency of extreme weather events and warmer climatic conditions (Le Page *et al.* 2017, de Faria *et al.* 2017, Fonseca *et al.* 2019). At the same time, deforestation continues to promote forest fragmentation and associated edge effects are much greater in forests that have burned multiple times, in which structure resembles more that (Alencar *et al.* 2006, Armenteras *et al.* 2017). In some regions of the Amazon, we can already observe how interactions among such factors have contributed to larger and more frequent

Box 19.4 Wildfire impacts on floodplain forests

Although Amazonian floodplain forests are inundated for several months every year, they are remarkably flammable when compared to *terra firme* forests, particularly in black-water rivers (Flores *et al.* 2014, 2017; Resende *et al.* 2014; Nogueira *et al.* 2019). Because of flooding, the forest litter takes longer to decompose and accumulates, forming a root mat (fine roots and humus) on the topsoil that can spread smoldering fires during extreme drought events (dos Santos and Nelson 2013, Flores *et al.* 2014). Compared to *terra firme* forests, the understory of floodplain forests is also slightly more open, allowing fuel to dry faster (Almeida *et al.* 2016). As a result, when wildfires spread, they can be intense, killing up to 90% of all trees by their root systems (Flores *et al.* 2014; Resende *et al.* 2014). After a single fire, forests can still recover slowly, but remain vulnerable to recurrent fires for decades. Along the middle Rio Negro, for instance, half of all burned forests were affected by another fire, which caused them to become trapped in an open vegetation state (Flores *et al.* 2016). Recent evidence reveals that after a first fire, the topsoil of floodplain forests begins to lose nutrients and fine sediments and gain sand. At the same time, tree composition shifts, with species typical of white-sand savannas becoming dominant, together with native herbaceous plants. In only 40 years, forests on clay soil are replaced by white-sand savannas due to repeated wildfires (Flores *et al.* 2021). Floodplain forests are therefore fragile and flammable ecosystems, and because they are widespread throughout the Amazon, they may potentially spread fires across remote regions (Flores *et al.* 2017) that could accelerate large scale tipping points (see Chapter 24). Plans to manage fire in the Amazon must take into account the existence of these flammable floodplain ecosystems, to prevent fires from spreading when the next major drought occurs.

understory fires that have burned close to 85,000 km² of primary forests in the southern Amazon during the 2000s (Morton *et al.* 2013, Aragão *et al.* 2018). As changes in climate and land use continue in the near future, they may trigger fires burning even larger areas (Le Page *et al.* 2017, Brando *et al.* 2020). Consequently, fires could become the main source of carbon emissions in the Amazon, surpassing those associated with deforestation (Aragão *et al.* 2018, Brando *et al.* 2020).

A major cause for concern is that the current transformations in forests caused by climate and land-use change will not only burn large areas, but also kill more trees than they currently do. In the south-east Amazon, for an increase of 100 kW/m in fire line intensity, tree mortality increased by 10% (Brando *et al.* 2014). With more edges and drier climatic conditions, we expect fire line intensity to greatly increase, potentially causing the mortality of many more trees, and subsequently resulting in even more CO₂ emissions. In addition, some projections point to a potential expansion of fire geography to historically wetter areas, a likely effect of

the combination of climate and land-use change.

19.4.2 Edge effects

Between 2001 and 2015, around 180,000 km² of forest edges were created in the Amazon (Silva Junior *et al.* 2020). The resulting proliferation in edge habitat, often with no habitat ‘core’, is ubiquitous in farm-frontier landscapes in the Brazilian (Broadbent *et al.* 2008; Fearnside 2005; Numata *et al.* 2017; C. H. L. Silva *et al.* 2018), Bolivian (Paneque-Gálvez *et al.* 2013), Colombian, Ecuadorian, and Peruvian Amazon (Armenteras and Barreto *et al.* 2017).

Direct impacts

At local scales, increases in light intensity, air temperature, vapor pressure deficit, and wind exposure, accompanied by decreases in air humidity and soil moisture, result in desiccation around edges (Broadbent *et al.* 2008; Kapos 1989; Laurance *et al.* 2018), which may extend hundreds of meters into adjacent forests (Briant *et al.* 2010). This

change in microclimate contributes to elevated tree mortality, which in turn lead to biomass collapses, especially within the first 100 m of a forest edge (Laurance *et al.* 1997; Numata *et al.* 2011). Across the Amazon, 947 Tg C were lost between 2001 and 2015 due to edge effects, representing a third of the losses from deforestation in the same period (Silva Junior *et al.* 2020). Carbon losses are not offset by tree growth or recruitment; forest edges suffer a drastic change in species composition, becoming dominated by lianas and trees of smaller size and with lower wood density, which store less carbon (Laurance *et al.* 2006; Michalski *et al.* 2007). Ultimately, the proliferation of pioneer trees causes forests close to edges to present higher tree densities than those further away (Laurance *et al.* 2011).

It is not only the flora that is directly impacted by edge effects; both vertebrate and invertebrate fauna also experience considerable compositional and functional shifts, with some species thriving while others decline (Bitencourt *et al.* 2020; Santos-Filho *et al.* 2012). Overall, generalist species are favored by edge habitats, while specialists become restricted to the forest core. This may lead to local extinctions of specialist species unable to adapt to new disturbed conditions, favoring edge and gap specialist species or even facilitating colonization and range expansion for non-forest species (Palmeirim *et al.* 2020; Mahood, Lees and Peres 2012; Rutt *et al.* 2019). For example, ungulates avoid forest edges, while rodents have similar abundances in forest edges or cores (Norris *et al.* 2008). Among invertebrates, a striking example is that of leaf-cutting ants; within the first 50 m of a forest edge, the density of colonies increases almost 20-fold when compared to the interior of the forest (Dohm *et al.* 2011).

Indirect impacts

Forest edges are more susceptible to other types of disturbance (Brando *et al.* 2019), especially understory fires (Armenteras, González, *et al.* 2013; Devisscher *et al.* 2016; C. H. L. Silva *et al.* 2018). This is mediated by changes in the structure and

composition of the vegetation, in addition to the microclimatic alterations that occur when an edge is created (Cochrane 2003), which are exacerbated by climate change (Cochrane and Laurance 2008; Cochrane and Barber 2009). Fragmented forest regions in the basin experience a higher frequency of forest fires, including Bolivia (Maillard *et al.* 2020), Brazil (Silva *et al.* 2018; S. S. da Silva *et al.* 2018; Silvério *et al.* 2018), and Colombia (Armenteras, Barreto, *et al.* 2017; Armenteras, González, *et al.* 2013)

19.4.3 Logging

Timber production through selective logging is one of the most important activities in tropical forest areas (Edwards *et al.* 2014). The Pan-Amazonian countries represent 13% of the tropical sawnwood production, where Brazil alone is responsible for more than half (52%) of the production followed by Ecuador (11%), Peru (10%), and Bolivia (10%). Venezuela, Colombia, Suriname, and Guyana represent the remaining 17% (ITTO 2020) (Figure 19.14). The extent of logging activities in Amazonian countries is also large. In the Brazilian Amazon, selective logging affects an area as large as that deforested annually (Asner *et al.* 2005, 2009; Matricardi *et al.* 2020), concentrated mostly along the deforestation frontier and surrounding major logging centers (SFB and IMAZON 2010). Selective logging is the second most common driver of forest degradation in the Brazilian Amazon, behind only edge effects (Matricardi *et al.* 2020).

Direct impacts

The illegality of logging in the countries of the Amazon Basin is commonly associated with conventional logging practices, which differ from reduced-impact logging (RIL). Conventional logging extracts a higher amount of timber per hectare (e.g. volume and number of species) and does not follow a coherent infrastructure extraction plan which would allow less impact for future harvest (i.e. less roads and logging decks) (Lima *et al.* 2020; Sist and Ferreira 2007). Conventional logging practices increase soil compaction from unplanned skid trails (DeArmond *et al.* 2019), and have a larger impact on

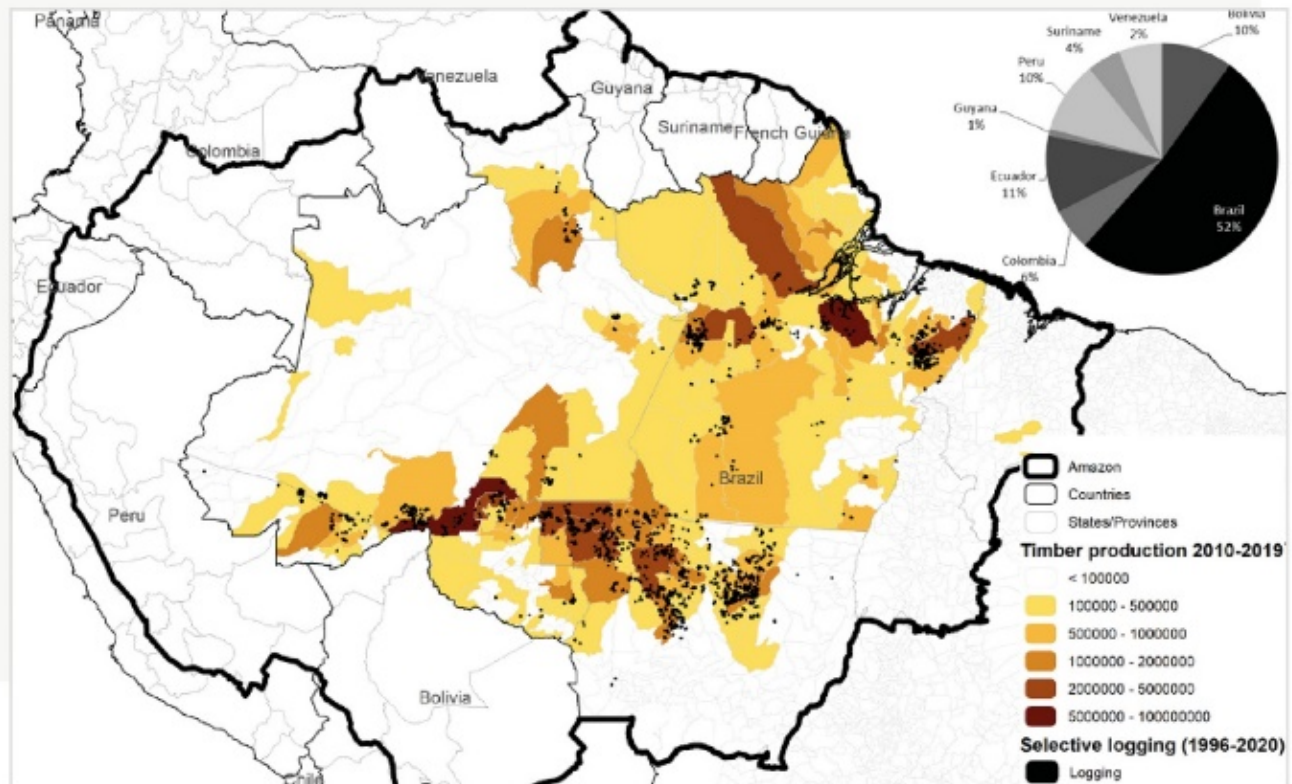


Figure 19.14 Selective logging across Amazonia. Pie chart – distribution of timber production in Amazonian countries (ITTO 2021). Map - legal timber production by Brazilian municipality from 2010 to 2019 (IBGE 2020).

reducing carbon stocks (Sasaki *et al.* 2016), increasing necromass and tree fall (Palace *et al.* 2007; Schulze and Zweede 2006), and enhancing CO₂ emissions (up to 30%) when compared with unlogged forest (Blanc *et al.* 2009; Pearson *et al.* 2014). In addition, conventional logging practices have greater impacts on biodiversity when compared to RIL, including reducing species abundance, richness, and phylogenetic and function diversity, mainly during the first years after logging (Azevedo-Ramos *et al.* 2006; Jacob *et al.* 2021; Mestre *et al.* 2020; Montejo-Kovacevich *et al.* 2018). Changes in species richness and abundance may in part be explained by post-logging increases in individuals' physiological stress (França *et al.* 2016). Ultimately, these lead to subsequent impacts on ecosystem processes; for example, in the Brazilian Amazon, selective logging led to the decline of dung beetle richness and significantly

changed their community composition, which in turn decreased rates of soil bioturbation, a function performed by these animals (França *et al.* 2017). Distinct logging practices also impact ecosystem dynamics and services in logged forests in the Amazon. Logging affects energy and water fluxes due to changes in albedo and surface roughness caused by high levels of canopy openness, mainly in the short-term (1-3 years) (Huang *et al.* 2020). These practices also promote warmer temperatures inside the forest (Mollinari *et al.* 2019), and depending on the intensity of extraction, biomass recovery for further cutting cycles is compromised.

Indirect impacts

The road network created by selective logging provides access to new hunting grounds (Robinson *et*

al. 1999), which can lead to declines in animal populations. Logging also facilitates the occurrence of understory fires; the intense canopy damage caused by logging activities leads to microclimate changes in the first two years following the logging operations (Mollinari *et al.* 2019). The hotter and drier forest is therefore more likely to sustain understory fires (Uhl and Vieira, 1989).

19.4.4 Hunting

Currently, there are ongoing population declines in many mammal, reptile, and bird species associated with over-harvesting, which are biased towards large-bodied species. The results of this defaunation can have profound consequences for species composition, population biomass, ecosystem processes, and human well-being in over-hunted Amazonian landscapes.

Commercial exploitation of animal hides in the 20th century was intense; between 1904 and 1969, it is estimated that 23.3 million wild mammals and reptiles of at least 20 species were commercially hunted for their hides (Antunes *et al.* 2016). This commercial exploitation is now much reduced, although approximately 41,000 peccary skins (mostly Collared Peccary, *Pecari tajacu*) are exported for the fashion industry annually (Sinovas *et al.* 2017). Exploitation is now predominantly for food, with Peres *et al.* (2016) estimating that hunting affects 32% of remaining forests in the Brazilian Amazon (~1M km²), with a strong depletion of large vertebrate populations in the vicinity of settlements, roads, and rivers (Peres and Lake 2003).

Direct impacts

Impacts vary across species depending on their life-history characteristics; taxa that are typically long-lived, with low birth rates, and long generation times are more vulnerable to local extinction (Bodmer *et al.* 1997). For example, in southeastern Peru, hunting resulted in the local extirpation of large primate species and reduced populations of medium-sized primates by 80% (Nuñez-Iturri and Howe 2007). Vulnerability to hunting may also be

exacerbated by biogeographic quirks, with hunting representing a major threat to micro-endemic species like the Black-winged Trumpeter (*Psophia obscura*) or terrestrial species restricted to specific habitats which are more accessible like the Wattled Curassow (*Crax globulosa*), which is found only along more accessible river-edge forests. Habitat loss, fragmentation, and human-driven disturbances such as logging and forest fires interact synergistically with hunting in reducing and isolating populations that do not use the non-forest habitat matrix, inhibiting 'rescue effects' from neighboring forests and hence source-sink dynamics (Peres 2001). Additionally, there is evidence of sublethal impacts from hunting on Amazonian vertebrates, with lead being found in the livers of Amazonian game species (Cartró-Sabaté *et al.* 2019).

Although hunting represents the major driver of direct defaunation, there are other drivers of loss including human-wildlife conflicts arising from livestock depredations by Jaguar (*Panthera onca*) (Michalski *et al.* 2006) and Harpy Eagles (*Harpia harpyja*) (Trinca *et al.* 2008). The wildlife trade also impacts a diverse set of taxa; for example, live parrot exports average 12,000 birds annually, mostly wild-caught individuals from Guyana, Peru, and Suriname (Sinovas *et al.* 2017) and ~4,000 Night monkeys (*Aotus* sp.) were estimated to have been sold to a biomedical laboratory on the Colombian side of the tri-border region of the north-west Amazon (Maldonado *et al.* 2009). Direct depletion for the pet trade has a long history and likely drove regional extinction of species such as the Golden Parakeet (*Guaruba guarouba*) from as long ago as the mid-19th century (Moura *et al.* 2014). Although trade has been reduced by effective command-and-control strategies, it remains the main threat to regionally Critically Endangered species like the Great-billed Seed Finch (*Sporophila maximiliani*) (Ubaid *et al.* 2018).

Indirect impacts

Overhunting may have pervasive impacts on Amazonian forests by disrupting or entirely removing 'top-down' control on ecosystems that are

mediated by large-bodied predators and herbivores, leading to widespread and potentially irreversible ecosystem alteration and to loss of resilience and function (Ripple *et al.* 2016). Overhunting disrupts the ecological interactions between plants and their seed dispersers, as some large mammals perform non-redundant seed dispersal services (Ripple *et al.* 2016). As a consequence, there is a shift in recruiting patterns of saplings in heavily hunted areas (Bagchi *et al.* 2018), with an increase in wind-dispersed and small-seeded species (Terborgh *et al.* 2008). This, in turn, could lead to a decrease in forests' future carbon stocks, as the species favored in hunted forests tend to have lower carbon storage capacity (Peres *et al.* 2016).

19.5 Conclusions

As of 2018, approximately 14% of the Amazon biome had been deforested, mainly due to the replacement of forests by pastures. Forest loss affects local temperature and precipitation, with increases in land surface temperatures and reductions in precipitation of up to 1.8% across the Amazon. Local extinctions are also a direct result of deforestation. The fact that there is no official record of a regional or global species extinction in the Amazon should bring no comfort, as a vast number of species remain to be described by science; it is possible, and even likely, that species are disappearing before they become known. Forest fires, selective logging, edge effects, and hunting put additional pressure on biodiversity, contributing to severe compositional shifts in remaining forests. The interactions between the multiple drivers of deforestation and forest degradation amplify their individual effects. An immediate halt to the drivers of deforestation and forest degradation is necessary to avoid further greenhouse gas emissions and biodiversity loss.

19.6 Recommendations

- Governments, the private sector, and civil society need to take urgent action to avoid further deforestation in the Amazon, particularly of primary forests. Avoiding loss of primary forest is

by far the highest priority to avoid carbon emissions, biodiversity loss, and regional hydrological changes.

- Governments must close down markets for illegal products (e.g. timber, gold, and bush meat).
- Implement an integrated monitoring system for deforestation and forest degradation across the basin with comparable, transparent, and accessible datasets. Datasets can be generated through partnerships between governments and the scientific community. It is no longer acceptable for deforestation to be the sole focus of forest monitoring.
- Develop basin-wide environmental impact assessments for infrastructure, such as roads, waterways, and dams, as their impacts are not only local. Planning must account for the indirect impacts of infrastructure on surrounding ecosystems, as these can outweigh direct impacts.
- Licensing, concessions and permits for forest conversion and infrastructure development must be accessible across the Amazon Basin to support integration with ground and satellite-based monitoring systems, enabling supply-chain traceability and risk assessment of investments.
- Urbanization needs planning to replace the current, organic encroachment mode.
- Develop a fire risk monitoring system and an early warning system to prevent and combat forest fires, especially in years of extreme drought when fires are more likely to escape from non-forest land uses. These should be accompanied by programs stimulating alternative land-management techniques that do not use fire.
- Restrict logging concessions to companies employing reduced-impact logging techniques, in order to decrease forest flammability and promote a sustainable forest-based economy. It is crucial that logging concessions spare part of their territory to act as sources for recolonization of logged areas.

19.7 References

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