Chapter 24

Science Panel for the Amazon (SPA)

Working Group 8

CLIMATE CHANGE IN THE AMAZON: TENDENCIES, IMPACTS AND ECOLOGICAL CONSEQUENCES

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THE RESILIENCE OF THE AMAZON FOREST TO GLOBAL CHANGES: ASSESSING THE RISK OF TIPPING POINTS

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1   **ACRONYMS AND ABBREVIATIONS**

2   AFEX   Amazon Fertilization Experiment

3   DGVMs   Dynamic Global Vegetation Models

4   eCO_2 CO_2 fertilization effect

5   ESMs   Earth System Models

6   FACE   Free-air concentration enrichment

7   GHG   Greenhouse gas

8   HadCM3   Hadley Center Coupled Model, version 3

9   IPCC   Intergovernmental Panel on Climate Change

10   LGM   Last Glacial Maximum

11   LSMs   Land Surface Models

12   MHDE   Mid-Holocene Dry Event

13   PFTs   Plant functional types

14   SDTF   Seasonally dry tropical forest

15   SST   Sea surface temperature

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1 **KEY MESSAGES**

2   • Five tipping points described in the literature comprise disturbances triggered by
3   changes in climatic conditions and human activities, and associated large-scale
4   feedback mechanisms. Nevertheless, the heterogeneity in forest responses
5   throughout the Amazon basin (i.e., how resistant and recoverable different forests
6   are) seem to be key in determining the systemic resilience of the entire Amazon
7   system, and should be a research priority;

8   • Based on empirical evidence, four different ecosystem configurations have been
9   proposed for Amazonian forests: (i) a closed-canopy seasonally dry tropical forest
10   state; (ii) a native savanna state; (iii) an open-canopy degraded state; and (iv) a
11   closed-canopy secondary forest state. Due to the existence of novel feedbacks
12   associated with invasive plants and human-modified landscapes, we consider the
13   open degraded state and the closed-canopy secondary forest state as more likely to
14   occur over broad areas, particularly across the 'arc of deforestation';

15   • Further studies focusing on how past underlying conditions (e.g., soil fertility and
16   rainfall regimes) selected species that cope differently with the same amount of
17   disturbance will be key to unveil how such response heterogeneity may either
18   increase or dampen the systemic resilience of the Amazon ecosystem;

19   • Amazonian tipping point has been best studied so far with the use of models.
20   Despite continuous model improvements that reduce the uncertainties, there is a
21   lack of observational (field and remote sensing) and experimental evidence to
22   improve these models and evaluate their results. As such, there is no
23   reasonable/strong scientific agreement, from a modeling perspective, on the
24   likelihood of crossing such an Amazonian tipping point in the future. Priority areas
25   for model-data integration are CO_2 fertilization effect, soil nutrient limitation,
26   recruitment/mortality dynamics, plant functional diversity, as well as reducing
27   uncertainty in Amazon rainfall projections.

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ABSTRACT

Here we review and discuss existing evidence of ongoing changes in the Amazon forest system that may lead to resilience loss and tipping points in which the ecosystem may shift persistently to an alternative state. Grounded on the theory of complex dynamical systems, we analyze the state of the Amazon forest and its potential trajectories in the 21st-century, aiming to provide support for a science-based management scheme for enhancing its systemic resilience. This review is based on five systemic tipping points for which there is evidence, being four climate-related: (1) annual rainfall value below the range between 1,000 and 1,500 mm/yr, (2) dry season length above seven months, (3) for Amazon lowlands, maximum cumulative water deficit above 200 mm/yr, (4) a global increase of 2°C on the equilibrium temperature of the Earth; and one associated with human-induced changes: (5) the 20-25% accumulated deforestation of the whole basin. Evidence suggests that depending on varying combinations of stressing conditions, disturbances and feedback mechanisms, forests could be replaced at local scales by: (i) a closed-canopy seasonally dry tropical forest; (ii) a savanna state; (iii) an open-canopy degraded state; and (iv) a closed-canopy secondary forest. Local-scale forest collapses could trigger cascading effects on rainfall recycling, intensifying drought seasons, wildfires and leading to massive forest loss at continental scales, particularly at the southwest of the basin. The probability of crossing such tipping points depends largely on heterogeneities across the system including geological, physical, chemical and cultural processes that influence connectivity and the likelihood of contagious disturbances. Biodiversity patterns were historically shaped over the past 60 million years by these processes and still today influence forest adaptive capacity and resilience. Thus, maintaining biodiversity is critical for enhancing resilience and reducing the risk of systemic forest collapse in the near future.

Keywords: tipping points, resilience, biodiversity, heterogeneity, connectivity, climate change, land use change.

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1 **GRAPHICAL ABSTRACT**

2 TBD

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4

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1. **INTRODUCTION**

The Amazon is a complex dynamical system that has been constantly changing for at least 60 million years ago (Ma), with geological, hydrological and evolutionary processes shaping the system that we know today (see Chapters 1-7 and Figure 24.1). While the Amazon River was formed around 7 Ma, forests expanded over non-forest habitats, and during the same time, massive wetlands retreated at the western parts of the basin. This process altered the courses of most rivers, causing new geographical barriers to emerge, altering the distribution of species, and creating conditions for diversification and speciation (Hoorn *et al.* 2010, see also Chapters 1 and 2). More recently, during the Holocene, around 13 thousand years ago (ka), humans arrived in the Amazon (Roosevelt 2013) and began to contribute to further changes in the landscapes and alter plant species distributions (Levis *et al.* 2017, see also Chapters 8 and 10).

As a result of the interplay between these processes (both natural and anthropogenic) operating at different spatial and temporal scales, the Amazon is currently an extremely heterogeneous and biodiverse system (see Chapters 3 and 4, and Figure 24.1b). Forest tree communities across the basin are formed by different sets of species with contrasting functional traits selected by continental to local environmental conditions, with main drivers of this heterogeneity including soil (Quesada *et al.* 2012), climate (Davidson *et al.* 2012; ter Steege *et al.* 2013; Esquivel-Muelbert *et al.* 2017), topography (Oliveira *et al.* 2019) and microclimate (Barros *et al.* 2019). Savannas also occur along the fringes of the Amazon basin and as "islands" within the dominant forest habitat (Prance 1996). The varying types of forest and non-forest habitats that exist are connected through a rich web of ecological interactions, which have contributed to maintaining the whole system resilient for the past 45 ka. Such resilience has been observed even under the extremely dry conditions of the Last Glacial Maximum (LGM) around 20 ka (Wang *et al.* 2017).
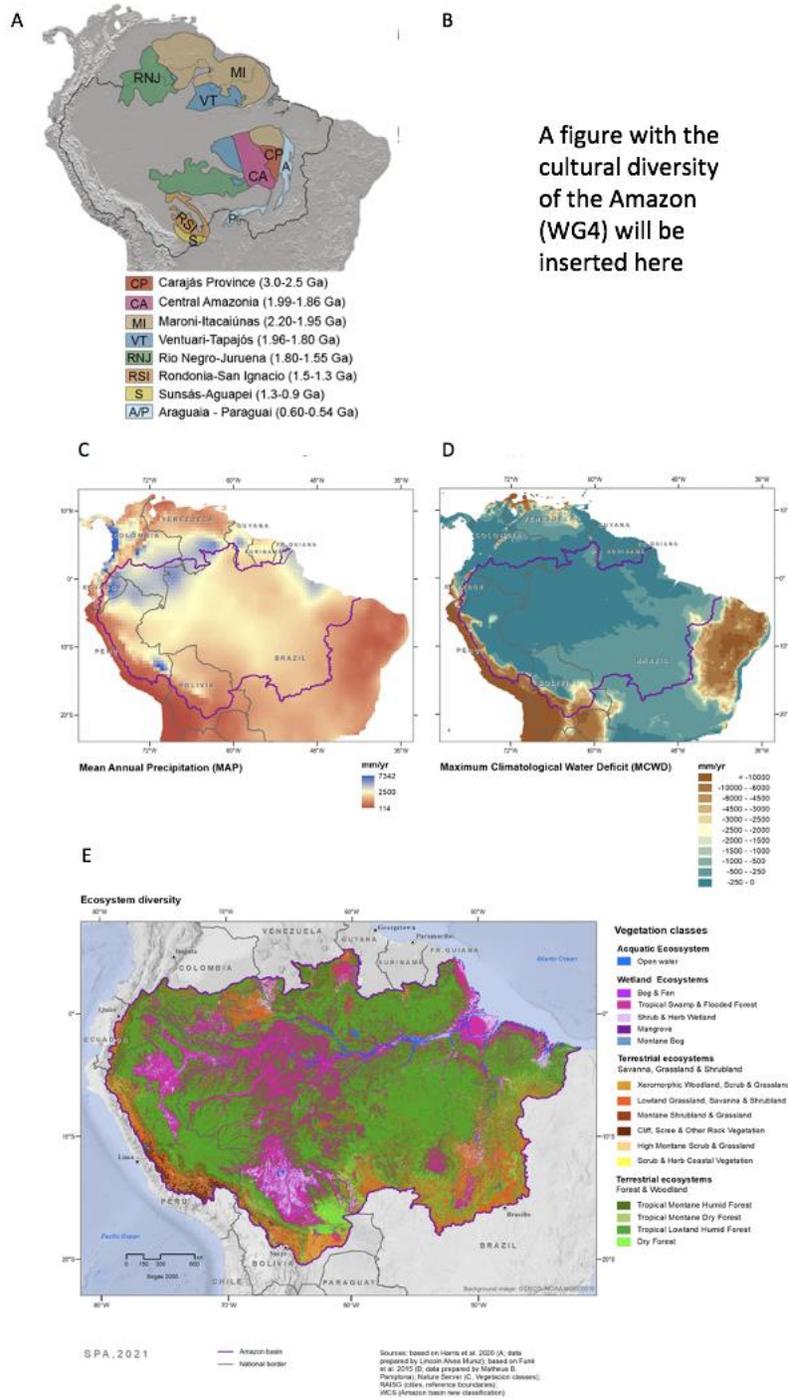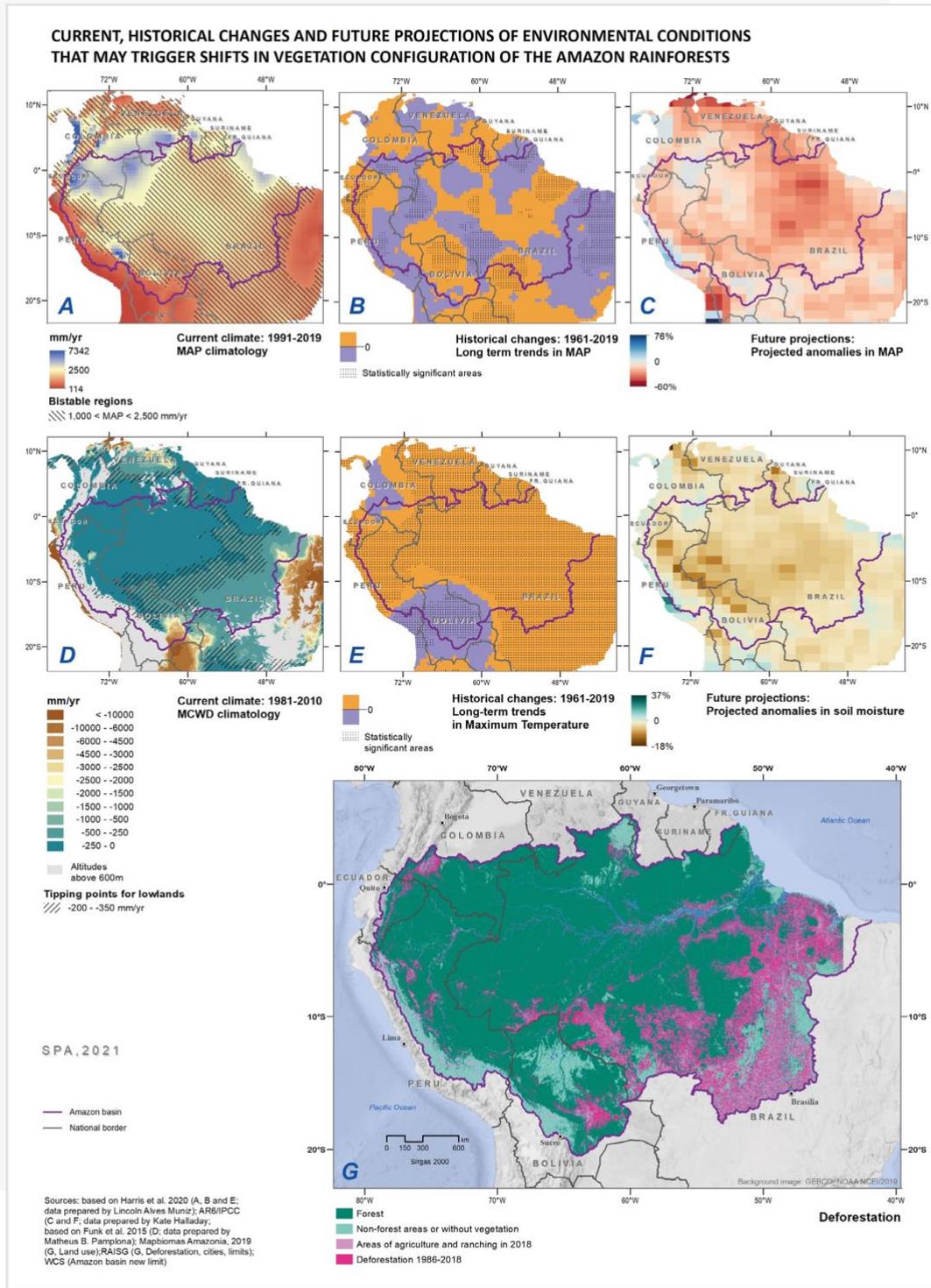
1

2 **Figure 24.1.** Heterogeneity in abiotic and biotic conditions throughout the Amazon system,

3 which shaped a range of plant community assemblages within different time and spatial

4 scales. (a) geochronological map of South America with the main provinces of the Amazon

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1 Craton (see Chapter 1); (b) past cultural diversity (to be defined either early ceramics

2 complexes or Holocene ceramic complexes - still to be built by the map team) (see Chapter

3 8); (c) 1981-2010 climatology mean annual precipitation (MAP, mm/yr) from CHIRPS

4 dataset (Funk *et al.* 2015); (d) 1981 - 2010 maximum climatological water deficit (MCWD,

5 mm/yr) from CHIRPS dataset (Funk *et al.* 2015); (e) wetland and terrestrial ecosystems of

6 the Amazon to show the tremendous biodiversity embedded within the system (See Chapter

7 4).

8 In the last century, however, the Amazon system began to change faster, mostly due to

9 local, regional, and global human activities that intensified particularly since the 1970s

10 (See Chapters 14-21, and Figs. 24.2c-e). Within the last two decades, extreme droughts

11 have become more frequent, and extremes in precipitation during the wet and dry seasons

12 have intensified (see Chapter 22; Marengo *et al.* 2011; Gloor *et al.* 2013; Jiménez-Muñoz

13 *et al.* 2016). Mean, maximum and minimum temperatures have also risen (see Chapter 22;

14 Jiménez-Muñoz *et al.* 2013), particularly on fragmented landscapes due to deforestation

15 (Zeppetello *et al.* 2020). As a result, mature Amazonian forests are now losing drought-

16 sensitive species and becoming more dominated by drought-tolerant species (Esquivel-

17 Muelbert *et al.* 2016, 2019; see also Chapter 23), with higher mortality rates of the drought-

18 sensitive ones particularly along the southern fringes of the Amazon boundaries (Esquivel-

19 Muelbert *et al.* 2020). In central Amazon, interactions between extremely wet and dry

20 periods are increasing tree mortality rates and reducing growth (Aleixo *et al.* 2019; Esteban

21 *et al.* 2021).

CURRENT, HISTORICAL CHANGES AND FUTURE PROJECTIONS OF ENVIRONMENTAL CONDITIONS THAT MAY TRIGGER SHIFTS IN VEGETATION CONFIGURATION OF THE AMAZON RAINFORESTS

Sources: based on Harris et al. 2020 (A, B and E; data prepared by Lincoln Alves Muniz); AR6/IPCC (C and F; data prepared by Kate Halladay; based on Funk et al. 2015 (D; data prepared by Matheus B. Pamplona); Mapbiomas Amazonia, 2019 (G, Land use);RAISG (G, Deforestation, cities, limits); WCS (Amazon basin new limit)

1

Figure 24.2. Tipping points (section 2) and disturbances/perturbations which may affect the resilience of the Amazon. (A) 1991 - 2019 climatology of mean annual precipitation (MAP, mm/yr) showing bistable areas for tipping point range (tipping point 1) using CRU 4.04 dataset (Harris *et al.* 2020); (B) long-term trends (Kendall t) in MAP (hatched areas are statistically significant) using CRU 4.04 (Harris *et al.* 2020); (C) projected relative changes in MAP at 4°C global warming with the UKESM1 climate model (Sellar *et al.* 2019); (D) 1981-2010 MCWD climatology showing tipping points (-200 and -350 mm/yr for lowlands) (tipping point 3); (E) long-term trends (Kendall t) in maximum temperatures (hatched areas are statistically significant) using CRU 4.04 dataset; (F) projected relative changes in soil moisture at 4°C global warming with the UKESM1 climate model (Sellar *et al.* 2019); (G) deforestation according to MapBiomas.

Moreover, wildfire regimes are intensifying (Alencar *et al.* 2015, see also Chapter 22), causing unprecedented levels of tree mortality (Brando *et al.* 2014). The expansion of cattle production has introduced invasive alien grasses, increasing the flammability of degraded and regenerating forests (Cochrane 2003). Moreover, deforestation disrupts forest-rainfall interactions across the Amazon by interrupting the moisture recycling by forest trees, and consequently the east-west moisture flow (see Chapter 7); a process that may accelerate forest loss (Zemp *et al.* 2017; Staal *et al.* 2020). Wildfires and deforestation also threaten species located along the southern edge of the system (Steege *et al.* 2015), particularly where forests are likely to be more resilient to climate change (Ciemer *et al.* 2019). On the other hand, changes in wildfire regimes may affect areas away from the southern edges given that species may have less adaptations to thrive under more frequent and intense wildfire events (Staver *et al.* 2020). In the case of Brazil, the Amazonian country that holds the largest deforestation rates (see Chapters 17 and 19), deforestation rates that had been previously slowing down began to rise since 2012, due to political changes that led to the weakening of Brazilian environmental governance (Levis *et al.* 2020; Rajão *et al.* 2020, see Chapters 14 and 17). All these changes imply that the Amazon now has to deal with unprecedented levels of stressing conditions and disturbance regimes.

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1   A topic that has raised concern is the potential existence of an ecological tipping point that

2   could affect the stability of the Amazon, causing large scale forest dieback or collapse (Box

3   24.1). Despite increasing evidence of tree mortality caused by extreme rainfall events (both

4   dry and wet), fire, deforestation, and the potential of their combined effects (Cochrane *et al.*

5   1999; Aragão *et al.* 2007, 2008; Phillips *et al.* 2009; Brando *et al.* 2014; Esquivel-Muelbert

6   *et al.* 2020; Staal *et al.* 2020; Esteban *et al.* 2021), the actual behavior of the Amazon

7   system still remains uncertain. For instance, with increasing water deficit levels and aridity,

8   the Amazon forest may not necessarily shift abruptly across the whole basin, but instead,

9   shift gradually as least resilient forests are affected first, followed by the more resilient ones

10   (Levine *et al.* 2016; Figure 24.1). On the other hand, human-induced changes are likely to

11   occur faster than the time forest communities would need to recover. Moreover, a long

12   lasting hypothesis is that the Amazon forests that collapse may undergo a "savannization"

13   process, i.e., forests would turn into a savanna-like vegetation cover (Nobre *et al.* 1991).

14   Nevertheless, evidence suggests that native savannas are unlikely to replace all portions of

15   the Amazon forest, since most stressors are associated with human activities that would

16   introduce invasive alien grasses instead of native savanna species (Veldman and Putz

17   2011), maintaining forests trapped in a degraded and early successional stage (Barlow and

18   Peres 2008).

19   Grounded on the theory of complex dynamical systems, we review and discuss existing

20   evidence of ongoing changes that may reduce forest resilience and potentially lead to

21   tipping points (Box 24.1), in which the Amazon forest may shift into other types of

22   configurations. By analyzing the state of the Amazon forest and its potential trajectories in

23   the 21st-century, we expect to provide critical information that will support a science-based

24   management scheme for enhancing the resilience of this iconic system.

25   **2. POTENTIAL TIPPING POINTS AND NEW CONFIGURATIONS**

26   The tipping points that have been proposed for the Amazon rainforests so far are: (1)

27   annual rainfall totals below 1,000 mm/yr, inferred from satellite observations of tree cover

28   distributions (Hirota *et al.* 2011; Staver *et al.* 2011; Figs. 24.2a-d) or 1,500 mm/yr inferred

1   from global climate models (Malhi *et al.* 2009), (2) the dry season length longer than seven

2   months, inferred from satellite observations of tree cover distributions (Staver *et al.* 2011),

3   (3) for the Amazon lowlands, larger maximum cumulative water deficit values than 200

4   mm/yr or 350 mm/yr, inferred from different analysis with global climate models

5   (respectively, from Malhi *et al.* 2009; Zelazowski *et al.* 2011; Figure 24.2e); (4) an increase

6   of 2°C on the equilibrium temperature of the Earth, inferred from a coupled climate–

7   vegetation model (Jones *et al.* 2009; for instance, with consequences shown in Figs.

8   24.2d,g), and (5) surpass 20-25% accumulated deforestation, inferred from a combination

9   of environmental changes (i.e., increases in dry season length, see Chapter 22), climate

10 projections for the most pessimistic pathway of the Intergovernmental Panel on Climate

11 Change (IPCC; Figs. 24.2d,g), and human-induced degradation via deforestation (Figure

12 24.2h) (Nobre *et al.* 2016; Lovejoy and Nobre 2019). The main concern is that beyond

13 these possible tipping points, the system would enter a loop of rainfall reduction, fire and

14 forest mortality.

15 Given the constraints to acquire sufficiently long time series to effectively and directly

16 account for temporal changes, their impacts on vegetation cover, and consequent tipping

17 points (Box 24.1), the ones mentioned above have been inferred and proposed by different

18 types of modeling and observational approaches. The first two use a space-for-time

19 substitution method, which replaces temporal information on changing conditions and their

20 impacts (e.g., changes in precipitation intensity leading to changes in vegetation cover) by

21 observational data of vegetation status (canopy closure using tree cover values) along a

22 gradient of precipitation (e.g., 1,000 to 2,500 mm/yr) at a single snapshot in time. Tipping

23 points (3) to (5) are based on coupled climate-vegetation models, which are able to

24 simulate long time series with their integrative structure, but depending on a set of

25 parameterizations that may fail to adequately represent soil-plant-atmosphere interactions.

26 Thus, even having a glimpse of thresholds that may trigger irreversible changes, the

27 trajectories leading to stable and transient configurations of the Amazon basin need to be

28 further explored and studied by a combination of experimental and modeling studies. For

29 instance, a recent study has shown that, after correcting for models' biases identified using

7

1   observation data, a basin-wide Amazon dieback is unlikely to occur, even under the most

2   pessimist IPCC pathway (Chai *et al.* 2021).

3   Based on existing evidence, we identify four main configurations Amazonian forests may

4   shift and persist due to self-reinforcing feedbacks (Figure 24.3) to: (i) a closed-canopy

5   seasonally dry tropical forest, with increasing abundance of deciduous tree species; (ii) a

6   savanna state, dominated by native grass and tree species; (iii) an open-canopy degraded

7   state, dominated by invasive alien grasses and native fire-tolerant tree species; and (iv) a

8   closed-canopy secondary forest, dominated by native early successional tree and other

9   plants species. In the following subsections, we explain how current environmental changes

10 in the Amazon system (see Chapters 14-21 and Figure 24.2) may alter forest dynamics, as

11 well as feedback mechanisms (Box 24.1) that could arrest Amazonian ecosystems in the

12 configurations (i) to (iv), and illustrate these trajectories with evidence on past and current

13 changes.
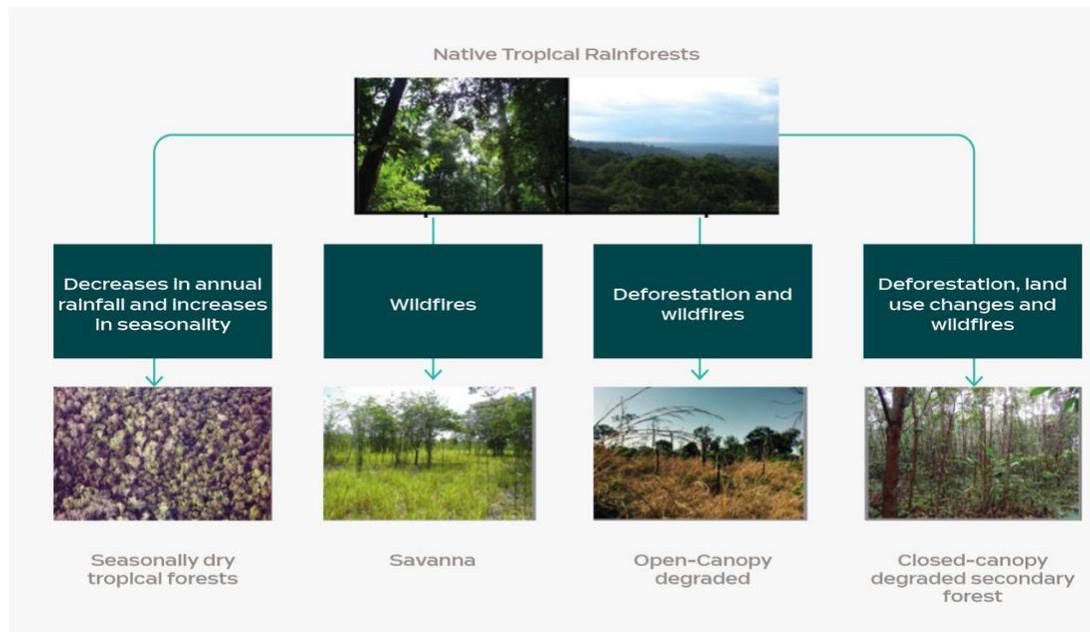
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15 **Figure 22.3.** Potential alternative configurations and drivers. Photo credits: Native tropical

16 rainforests at ZF2 Station (AM, Brazil) by Marina Hirota; seasonally dry tropical forests at

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1 Maracá Island (RR, Brazil) by Marcelo Trindade Nascimento; savanna at Barcelos (AM,

2 Brazil) by Bernardo M. Flores; open-canopy degraded at Fazenda Tanguro (MT, Brazil) by

3 Paulo Brando; closed-canopy degraded secondary forest at Tefé (AM, Brazil) by Catarina

4 Jakovac.

5

2.1 Forest shift to a closed-canopy, seasonally dry tropical forest

7 Considering the observed trends towards a drier climate in some parts of the Amazon (see

8 Chapter 22), there is a possibility that forests over nutrient-richer soils may shift into a

9 closed-canopy state that resembles in terms of structure and functioning a seasonally dry

10 tropical forest (SDTF) (Malhi *et al.* 2009; Dexter *et al.* 2018), dominated by deciduous

11 fast-growing trees, with high tolerance to drought conditions, and a higher demand for

12 nutrients. This type of semi-deciduous forest (i.e., with varying abundances of deciduous

13 species) is very common in transitional zones of the Amazon boundaries, and under drier

14 climatic conditions (Silva de Miranda *et al.* 2018), they may expand over wet Amazonian

15 forests (Dexter *et al.* 2018). For instance, drought tolerant species are widely distributed

16 across the Amazon region (Esquivel-Muelbert *et al.* 2017), which would allow them to

17 dominate in case of a shift of climatic regime (Esquivel-Muelbert *et al.* 2019). However,

18 drought-tolerance is not only expressed in terms of deciduousness and alternative

19 phenotypes may include trees with more resistant water-transporting systems (Barros *et al.*

20 2019) and/or deeper-rooted species. Nonetheless, a shift to semi-deciduous forest would

21 probably not follow a catastrophic non-linear dynamics, with associated *tipping points*

22 (Box 24.1, Figure 24.B1) because rainforests and STDFs occupy separate climatic niches

23 (Silva de Miranda *et al.* 2018), implying that tree species may have to migrate long

24 geographical distances. Hence, such changes might occur smoothly and more gradually

25 with increasing aridity and seasonality (Oliveira *et al.*).

26 ### *2.2 Forest shift to a native savanna state*

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1 The Amazon forest is often assumed to shift into a savanna-like state, once it passes tipping

2 points such as the ones mentioned in (1) – (5) (Cox *et al.* 2004; Jones *et al.* 2009; Hirota *et*

3 *al.* 2011; Staver *et al.* 2011; Lovejoy and Nobre 2019). However, evidence for such shifts

4 at the local scale is lacking, mostly because disturbed forests are commonly invaded by

5 alien grasses (see section 2.3) instead of native grass species from South American

6 savannas (Veldman 2016). This happens particularly in landscapes where forest is widely

7 converted into pastures, from where such invasive grasses escape and become dominant in

8 disturbed forests. Nonetheless, far from the agricultural frontier (arc of deforestation), and

9 far from small-scale pastures at the core of the Amazon forest system, black water

10 floodplain forests disturbed by wildfires are being replaced by native savanna vegetation

11 (Flores and Holmgren 2021) (Figure 24.3). In floodplain landscapes of the Rio Negro, fires

12 are highly destructive, killing practically all trees, and allowing the ecosystem to shift to a

13 savanna state within only 40 years. After the first wildfire, soils start to change from clayey

14 to sandy, while tree composition shifts from forest to white-sand savanna species, and the

15 herbaceous community remains dominated by native opportunistic plants (Flores and

16 Holmgren 2021). This local abrupt shift from forest to white-sand savanna seems to be

17 driven by repeated wildfires and a strong flood erosion mechanism that alters plant-soil

18 interactions, favoring savanna species. Previous analyses at the basin scale have shown that

19 these floodplain forests are less resilient than the upland forests (Flores *et al.* 2017),

20 including in the watersheds of large white-water rivers, such as the Madeira and Solimões.

21 Hence, as in other forest-savanna transition zones, evidence suggests that savannas of the

22 Amazon system may expand and persist due to feedback mechanisms involving repeated

23 wildfires and soil erosion processes (Flores *et al.* 2020; Flores and Holmgren 2021).

24 *2.3 Forest shift to an open-canopy, degraded state*

25 When forests are repeatedly disturbed and native savanna species are not able to colonize,

26 the ecosystem often becomes trapped in an open vegetation state, dominated by fire-

27 tolerant tree and palm species that usually occur in the forest, together with invasive alien

28 grasses and opportunistic herbaceous plants (Perz and Skole 2003; Veldman and Putz

1 2011), as well as vines and lianas (Tymen *et al.* 2016; Maia *et al.* 2021; Medina-Vega *et al.*

2 2021) (Figure 24.3). Below, we describe the feedback mechanisms that have been proposed

3 to explain how the ecosystem can be trapped in this state.

4 Numerous disturbances that open forest structure immediately increase light availability at

5 ground level, allowing herbaceous plants to invade (Cochrane and Schulze 1999; Silvério

6 *et al.* 2013; Longo *et al.* 2020). Satellite observations of fire occurrences from across the

7 global tropics reveal that when tree cover is reduced below 50%, ecosystem flammability

8 rises steeply (van Nes *et al.* 2018). Because most trees in the Amazon forest are fire-

9 sensitive, repeated fires often kill most of the tree community (Cochrane and Schulze 1999;

10 Barlow and Peres 2008; Balch *et al.* 2011; Brando *et al.* 2012; Staver *et al.* 2020),

11 particularly the younger individuals, reducing tree recruitment (Balch *et al.* 2011). As a

12 result, disturbances that reduce forest cover below this threshold may cause the ecosystem

13 to be trapped in an open state by repeated wildfires. Such consequences have been reported

14 in multiple studies in the Amazon, showing that shifts to an open degraded state are already

15 occurring (Barlow and Peres 2008; Brando *et al.* 2012; Flores 2016).

16 Other feedback mechanisms are also known to contribute to this ecosystem shift at the

17 landscape scale. For instance, the expansion of invasive alien grasses may also directly

18 reduce tree recruitment due to light competition with young seedlings (Hoffmann *et al.*

19 2004), which maintains low tree cover and grass dominance. Forest loss, degradation and

20 fragmentation inhibit the movement of many mobile animal species, particularly the ones

21 that are sensitive to open habitats (Laurance *et al.* 2004), causing many species to disappear

22 from the system (Barlow *et al.* 2016). In the case of frugivore species, by avoiding the use

23 of open disturbed habitats, tree seed dispersal in those sites may become limited, reducing

24 tree recruitment and forest regrowth. This dispersal limitation feedback is expected to be

25 stronger where disturbances are most severe (Turner *et al.* 1998). Evidence from the

26 tropical Atlantic Forest suggests that 30% tree cover could be a threshold in which many

27 forest adapted animal species disappear, and are replaced by disturbance adapted species

1 (Banks-Leite *et al.* 2014), potentially disrupting plant-animal interactions that are critical

2 for forest recovery.

3 The current expansion of open degraded ecosystems across vast portions of the

4 southeastern Amazon forest is triggering other types of feedback mechanisms at the

5 regional and global scales. Forests play a major role in maintaining the rainfall regime of

6 the Amazon by allowing moisture that originates in the Atlantic Ocean to be transported

7 across the basin; a process that may involve up to seven cycles of rainfall and re-

8 evapotranspiration (Spracklen *et al.* 2012; Zemp *et al.* 2017; Staal *et al.* 2018; see also

9 Chapter 7). Hence, by interrupting this process, deforestation and forest degradation will

10 likely reduce rainfall at the central and western Amazon, with stronger potential impacts

11 particularly during the dry season. This process also involves a feedback between drought

12 and deforestation that is already strengthening with accumulated deforestation, in which the

13 more forest area is lost, the stronger dry seasons will be, further increasing deforestation

14 rates (Staal *et al.* 2020) and forest fires (Xu *et al.* 2020). In addition to its effects on

15 precipitation, deforestation also affects regional temperatures, with fragmented landscapes

16 being considerably hotter than non-fragmented ones (Zeppetello *et al.* 2020). Due to this

17 large-scale feedback, a tipping point (5) has been proposed to cause a major forest dieback

18 within the Amazon basin (Nobre *et al.* 2016; Lovejoy and Nobre 2019). A previous model

19 study had estimated this deforestation tipping point at 40% (Sampaio *et al.* 2007), yet,

20 recent evidence based on a climate-vegetation model that accounts for the combined effects

21 of climate change, deforestation and wildfires (Nobre *et al.* 2016; Lovejoy and Nobre

22 2019), suggests that this threshold might indeed by closer to the 20-25%. In sum,

23 considering these broad-scale interactions, the more Amazonian forests become trapped in

24 an open-canopy degraded state, the more likely that a 20-25% threshold is sufficient to

25 accelerate a critical systemic transition.

26 *2.4 Forest shift to a closed-canopy, secondary forest state*

27 Differently from the previous cases, in which the forest is trapped in a contrasting open-

28 canopy state, here, disturbed forests recover their closed-canopies but do not progress

towards a mature forest state. Instead, they persist in an early successional stage, trapped by

different feedback mechanisms (Figure 24.3). Such secondary forests may not be identified

through satellite monitoring of canopy conditions, as high levels of greenness and leaf area

index may be interpreted as if the ecosystem has recovered its original forest state, when

aspects such as biodiversity and carbon storage remain at much lower values (Poorter *et al.*

2016; Rozendaal *et al.* 2019). In the Brazilian Amazon, for example, around 21% of the

previously deforested land is currently covered by secondary forests (INPE and

EMBRAPA 2011), but the ecological state of such regrown vegetation is unknown.

Under optimal conditions, during regrowth, environmental conditions in the understory

gradually change along with species taxonomic and functional composition, in a transition

from a state of an open-canopy with light demanding species towards a closed-canopy with

mature-forest species. With time, species diversity increases and plant-animal interactions

recover complexity and biomass (Poorter *et al.* 2016; Rozendaal *et al.* 2019). Nonetheless,

secondary forests are almost two times more likely to be cleared for land use than mature

forests, possibly due to lower governmental restrictions and higher accessibility (Wang *et

al.* 2020). As a result, most secondary forests are cleared again before 20 years of regrowth

(Chazdon *et al.* 2016; Jakovac *et al.* 2017; Schwartz *et al.* 2020). Such feedback causes

secondary forests to persist in the landscape only at an early-successional state (Barlow and

Peres 2008).

A combination of socio-economic and biophysical factors defines where and when forests

recover back to their previous state in terms of structure and composition. Within

traditional shifting cultivation systems that dominate riverine landscapes of the Amazon,

forest regrowth constitutes the fallow period that supports repeated crop yields, being an

essential element of the rotational system. In contrast, throughout the extensive

pasturelands that dominate Amazonian landscapes at the "arc of deforestation", forest

regrowth constitutes an obstacle to pasture productivity and is often managed with

prescribed burning. Eventually, regrowth may occur in abandoned areas when landowners

do not have the means to continue managing the land or when land productivity is reduced

1 by soil degradation (Vieira *et al.* 2014; Nanni *et al.* 2019). Therefore, feedback

2 mechanisms between social and ecological elements partly determine whether the

3 ecosystem will become arrested in a closed-canopy secondary forest state.

4 The capacity of secondary forests to fully recover depends on the management practices

5 applied prior to abandonment and on the landscape context where it occurs (Jakovac *et al.*

6 2021). The repeated fire use to clean pastures and fertilize cropping fields reduces soil

7 fertility and consequently the rates of forest recovery, particularly when return-intervals

8 between slash-and-burn events are shortened (Zarin *et al.* 2005; Jakovac *et al.* 2015;

9 Heinrich *et al.* 2020). Under a high disturbance regime, survival strategies are favored over

10 growth strategies and a plant community with conservative traits is more likely to thrive.

11 Survival traits include high sprouting ability and low nutrient demand (Jakovac *et al.*

12 2015), high wood density and high leaf toughness (Fernandes Neto *et al.* 2019), all of

13 which are traits associated with resistance to disturbance and often with slow growth rates

14 (Poorter *et al.* 2010). Lianas and grasses are also favored by disturbances (Roeder *et al.*

15 2010; Veldman and Putz 2011), contributing to arrest succession by competing with trees

16 and leading to reduced growth rates and higher tree mortality (Schnitzer and Bongers

17 2002). Combined, these feedbacks arrest forest succession, maintaining lower basal area,

18 biomass, canopy height and species diversity, as well as higher density of stems, lianas in

19 the canopy and grass cover in the understory.

20 Furthermore, forest fragmentation associated with deforestation causes tree seed dispersal

21 limitations, consequently reducing tree recruitment (Arroyo-Rodríguez *et al.* 2015),

22 representing another amplifying feedback that can hinder secondary forest succession. The

23 seed rain in such landscapes is mainly composed by early successional pioneers dispersed

24 by wind or by generalist bats and birds that are able to cross large extents of pasture or

25 cropping fields (Cubiña and Aide 2001; Wieland *et al.* 2011). Overhunting in degraded

26 forests embedded within human-modified landscapes further contributes to reduce the

27 availability of animal dispersers and increase dispersal limitation (Bagchi *et al.* 2018). The

1   slow inputs of seeds from mature forests results in consistently slow species accumulation

2   over time and therefore a slow species turnover during regrowth (Mesquita *et al.* 2015).

3   In sum, different combinations of drivers and feedback mechanisms can cause Amazonian

4   forests to be trapped in different configurations, some of which are alternative states (Box

5   24.1). Here, we have identified four of these configurations, based on existing evidence: (i)

6   a closed-canopy semi-deciduous forest state, potentially with a more gradual shift; (ii) a

7   native savanna state; (iii) an open-canopy degraded state; and (iv) a closed-canopy

8   secondary forest state. Shifts to these alternative configurations may occur locally, but

9   depending on the scale of the feedbacks, they may become contagious and spread

10 disturbances across large parts of the basin, increasing the probability of a systemic forest

11 dieback. Moreover, other types of configurations are possible, such as bamboo-dominated

12 (*Guadua sarcocarpa*) forests of the southwestern Amazon that self-perpetuate facilitated

13 by a fire feedback; however, we have focused on four general types that are more likely to

14 expand in the coming future.

15 **3. PAST EVIDENCE OF THE DYNAMICS OF AMAZONIAN ECOSYSTEMS**

16 **SINCE THE LAST GLACIAL MAXIMUM (20 KA)**

17 Studies focusing on past vegetation changes have documented several of the forest change

18 scenarios outlined in section 2 (see also Chapters 1 and 2). For instance, an expansion of

19 savannas in the northeastern portions of the basin during a climatically unstable period

20 with increasing temperatures was registered at the beginning of the Holocene, i.e.,

21 approximately 11 ka (Rull 2015). However, the changes observed in sedimentary archives

22 have not always shown a change towards savannization, but depended on the nature of the

23 environmental driver. For instance, pollen analysis revealed a rainforest expansion during

24 the last 3,000 years in forest-savanna boundaries of the southern Amazon, driven by wetter

25 conditions related to changes in the location of the intertropical convergence zone (Mayle

26 *et al.* 2000). Hence, these forests have likely reached their maximum potential southern

27 limit expansion for the past 50 ka (Mayle *et al.* 2000), with a 22% increase in the CO_2

28 storage budget since the mid-Holocene (6 ka) (Mayle and Beerling 2004). Given the

1   historical observations registered during the last decades (see Chapter 22), and the climate

2   projections forecasted for this region towards drier conditions (Magrin *et al.* 2014), it is

3   unlikely that this forest expansion and consequent increased carbon sequestration will

4   continue. Instead, combined evidence suggests that these forests are more likely to recede,

5   being replaced by open vegetation types.

6   Empirical data of long-term forest dynamics have shown the differential sensitivity to past

7   climate change across the Amazon basin. Regions like the southern and southeastern

8   Amazon have shifted between forest and open savanna vegetation in relatively recent

9   periods of colder and drier LGM climate (Absy and Hammen 1976), whereas the Andean

10   flank in the western (van der Hammen and Absy 1994) and eastern portions of the Amazon

11   (Wang *et al.* 2017) seem to have persisted as forest. Long-term ecological data from pollen

12   analysis have shown the prevalence of various types of rainforests, both in southwestern

13   cloud forests and northwestern pre-montane forests of Amazonian highlands, showing the

14   importance of the cloud cover in buffering forests when facing climate change (Urrego *et*

15   *al.* 2010; Montoya *et al.* 2018). Presence of forests with distinct composition during LGM

16   has also been observed in northwestern Brazilian Amazon (Bush *et al.* 2004; D'Apolito *et*

17   *al.* 2013). All these regional evidence of a persistently forested Amazon system are

18   consistent with large-scale speleothem analyses showing a remarkable stability of the

19   Amazon rainforest for the past 45 ka, even under a 60% decrease in precipitation totals

20   (Wang *et al.* 2017).

21   The Mid-Holocene Dry Event (MHDE; 9-4 ka) has been proposed as a potential past

22   analog of current and future trends of decreased precipitation, yet there is still limited

23   evidence covering the entire duration of MHDE throughout the basin. Currently available

24   paleo-records, however, suggest a higher vulnerability of tropical forests to extended

25   droughts in peripheral transitional zones (Mayle and Power 2008; Smith and Mayle 2018).

26   In addition, changes in plant functional traits spanning the termination of the MHDE (i.e., a

27   period of increasing rainfall amount) suggest that rainfall increases led to a replacement of

28   slow-growing, drought-tolerant taxa by fast-growing, drought-vulnerable taxa (van der

1 Sande *et al.* 2019). Indeed, secondary forest species usually differ in their ecological

2 strategies from mature forest species, changing the forest functioning and stability. In

3 southeastern Venezuela, for instance, rainforest taxa were replaced by secondary dry

4 forests around 2.7 ka, a shift that persisted for more than 1,000 years. These secondary

5 forests were finally replaced 1.4 ka under a period of high fire occurrence by the current

6 vegetation consisting of open savanna (Montoya *et al.* 2011).

7 When a forest is disturbed, the rates of ecosystem change observed in sedimentary archives

8 depend on the ecological scale, being abrupt (decadal) at the species level, but gradual

9 (centennial) at the community level (Montoya *et al.* 2018, 2019). In a tropical meta-data

10 analysis of forest recovery rates after disturbances based on pollen records, Cole *et al.*

11 (2014) observed that South American forests required an average of 325 years to recover

12 from disturbances (natural and human-induced). The recovery rate was calculated in terms

13 of attaining a forest cover (expressed in % of tree pollen) similar to that prior to the

14 disturbance, without differentiating changes in the forest composition, structure or function.

15 Forests exposed to natural large infrequent disturbances (i.e., hurricanes or volcanic

16 eruptions) recovered faster compared to those affected by post-climatic and human

17 impacts. However, forests exposed to more frequent disturbances usually recovered faster,

18 suggesting that repeated disturbances may increase forest adaptive capacity and resilience,

19 yet over multi-centennial time scales (Cole *et al.* 2014). In the Andean Amazon region,

20 Loughlin *et al.* (2018) studied lands that were managed by Indigenous populations, but

21 following European conquest, forests recovered structurally (not compositionally) in only

22 130 years, possibly because the higher soil productivity of this region boosted tree growth.

23 Despite differences in these estimates, both studies manifest that the temporal range

24 required for forest to potentially recover is multi centennial (Cole *et al.* 2014; Loughlin *et*

25 *al.* 2018).

26 In summary, paleoecological evidence hints at two main directions. Firstly, the Amazon

27 forests have undergone local to regional shifts to dry secondary forests or savannas

28 (Montoya *et al.* 2011) depending on the disturbances at play (climate- or human-induced

changes), but not a basin-wide abrupt dieback, even during intense drier and warmer

periods that could well represent analogs of the hypothesized climate-related *tipping points*

(1) – (4). Secondly, the recovery ability of Amazonian forest ecosystems depends on their

disturbance histories: the more disturbance-adapted, the faster the recovery rates.

Nevertheless, long-term ecological data are still limited in the basin and concentrate

primarily along the Amazon margins; more work is still needed for unraveling the

dynamics of such heterogeneous ecosystems (Lombardo *et al.* 2018). In addition, some

important caveats need to be addressed when using paleo-data as reference for future

dynamics: (1) the increased rates and magnitudes of the changes projected for the near

future, with combined disturbance events (climatic and human-induced) acting

synchronously are unprecedented and may hamper forest recovery due to novel

mechanisms; and (2) the baseline conditions we have shown are no-analog of

ecophysiological drivers such as the enhanced atmospheric CO_2 concentrations of the 21st

century (section 5.3).

4. DRIVERS OF THE AMAZON FOREST RESILIENCE

Across the Amazon forest system, biotic diversity and abiotic heterogeneity promote a

huge variety of responses to disturbances such as extreme droughts and wildfires

(Feldpausch *et al.* 2016; Longo *et al.* 2018). This spectrum of responses affects the balance

between plant growth, survival, and mortality, and therefore, the resilience of ecosystems.

Below, we discuss the main environmental factors that affect plant growth and mortality at

different spatial and temporal scales.

The resilience of the Amazon forest is directly linked to the functional characteristics of

individual trees and their capacity to resist adverse conditions and disturbances. Thus,

processes that exert pressure on the capacity of trees to maintain their functioning and

survival are critical. Water deficit associated with increasing length of the dry season or

extreme droughts (i.e., related to *tipping points* (2) and (3), section 2), is likely to be the

major climatic threat to Amazonian trees, as suggested by observational and experimental

studies, showing that droughts increase tree mortality rates of individual trees (Nepstad *et*

al. 2007; DaCosta *et al.* 2010; Phillips *et al.* 2010; Rowland *et al.* 2015; Zuleta *et al.* 2017;

Aleixo *et al.* 2019; Janssen *et al.* 2020b). At least 50% of the Amazon forest is exposed to

seasonal droughts of three months or more (Nepstad *et al.* 1994), and contrasting rainfall

regimes have selected for species with different drought resistance mechanisms (Oliveira *et*

al.; Barros *et al.* 2019; Brum *et al.* 2019). In many cases, extreme drought events may not

necessarily cause the death of trees, but reduce their growth and capacity to maintain

transpiration rates. However, a recent meta-analysis of field observations reveals that

highly diverse Amazonian tree communities seems to buffer this effect, conferring higher

ecosystem resistance in terms of evapotranspiration rates (Janssen *et al.* 2020a).

Examples of functional characteristics of Amazonian trees to cope with seasonal water

deficit include: (1) investment in deep roots (Nepstad *et al.* 1994; Brum *et al.* 2019); (2)

roots that allow hydraulic redistribution during the dry season (i.e., passive movement of

water from deep to shallow soil through roots) (Oliveira *et al.* 2005); (3) high embolism

resistance, particularly in shallow-rooted understory trees and trees over plateaus far away

from the water table (Oliveira *et al.*; Brum *et al.* 2019); (4) strong stomatal control in the

dry season resulting in high water use efficiency (Barros *et al.* 2019; Brum *et al.* 2019); (5)

leaf shedding capacity by deciduous species (Wolfe *et al.* 2016). Although these traits do

not guarantee survival under future increasingly drier and variable climates, in locations

where the dry season has been intensified, changes in forest composition dynamics are

already underway through the recruitment of more dry-affiliated species and the mortality

of more wet-affiliated species (Esquivel-Muelbert *et al.* 2019). Also, life-history strategies

(e.g., fast-slow continuum in growth rates) have been shown to determine species-level

mortality, i.e., the faster you grow, the higher is the mortality risk (Esquivel-Muelbert *et al.*

2020).

There is also evidence that temperature changes (see Chapter 22; Figure 2e) could already

be changing forest functioning. Warmer temperatures tend to reduce forest productivity

rates (Sullivan *et al.* 2020), particularly by intensifying the atmospheric vapor pressure

deficit (Smith *et al.* 2020), indicating that rising temperatures may eventually impact forest

1 functioning and persistence (Araújo *et al.* 2021). Additional CO_2 is expected to buffer the

2 effect of water stress by increasing plant water-use efficiency and accelerating tree growth

3 (section 5.3). Elevated atmospheric CO_2 may be the cause of the increase in woody

4 biomass and productivity observed across Amazonian forests (Brienen *et al.* 2015).

5 However, CO_2 driven accelerations of tree growth have come at the cost of decreasing tree

6 longevity across the basin, further contributing to increase tree mortality rates (Brienen *et*

7 *al.* 2015; Hubau *et al.* 2020).

8 Despite the uncertainties regarding forest responses to climate change, current findings

9 suggest that, in the absence of fire, Amazonian forests may change both compositionally

10 and functionally in response to climatic changes, but still remain as closed-canopy forests.

11 Furthermore, if climate-related *tipping points* (2) – (4) (section 2) are crossed, shifts are

12 likely to be sparse and local because of the high heterogeneity and diversity of forest types.

13 Increased tree mortality caused by human-induced disturbances (e.g., wildfires and

14 deforestation), however, may contribute to destabilize the Amazon forest (Silva *et al.*

15 2018), increasing the likelihood that forests will be trapped in an open-canopy degraded

16 state, and that the system as a whole will cross the *tipping point* (5) (section 2.3).

17 **5. UNCERTAINTIES ASSOCIATED WITH TIPPING POINTS WITHIN THE**

18 **AMAZON SYSTEM**

19 *5.1 How does forest heterogeneity affect large-scale tipping points?*

20 Amazonian forests are home to ca. 15,000 tree species (Ter-Steege *et al.* 2020). Most of

21 these species are rare, and many remain unknown to science (ter Steege *et al.* 2013),

22 implying that this huge diversity imposes an enormous challenge to the understanding of

23 how the system functions. In particular, dominant species are responsible for most of the

24 ecosystem functions, such as carbon cycling (Fauset *et al.* 2015). Yet, the many non-

25 dominant and rare species that exist in a forest theoretically also play a fundamental role in

26 ecosystem resilience (Walker *et al.* 1999). When stressing conditions and disturbance

27 regimes change, these rare species can offer new possibilities of functioning, thus

increasing the capacity of the ecosystem to adapt and persist (Elmqvist *et al.* 2003). For instance, if a tree species is rare in waterlogged forests, but common on drier climatic conditions, due adaptations such as deep roots, it could emerge as a dominant species if climate becomes drier. As a general rule, species diversity is therefore expected to increase the resilience of Amazonian ecosystems. First, because diversity has a positive impact on forest productivity (Coelho de Souza *et al.* 2019) and carbon storage (Poorter *et al.* 2015), potentially accelerating regrowth after disturbances. Moreover, as the number of species is related to the number of strategies and potential responses to disturbances, diversity increases stability at the community and ecosystem levels, and the overall forest resilience (Elmqvist *et al.* 2003; Sakschewski *et al.* 2016; Anderegg *et al.* 2018). For instance, disease and herbivore outbreaks have been causing large-scale tree mortality in temperate regions, yet such events have not been observed in the tropics, likely because the high species diversity of tropical ecosystems reduces the spread of contagious diseases. Drought-tolerant species are often distributed across a vast range of precipitation conditions, hence they may occur as rare species in the wet parts of the basin (Esquivel-Muelbert *et al.* 2016). This pattern implies that if climate becomes drier in the more diverse wet forests, drought-affiliated species may be already present and could increase in abundance, maintaining forest cover, while altering forest functioning.

Rainfall variability (intra- and inter-annual fluctuations) may also add more heterogeneity to the system, as forests that experience more variability seem to be more resilient, likely due to a training-effect after experiencing multiple wet and dry periods (Ciemer *et al.* 2019). For instance, tree communities embedded within a more seasonal rainfall regime are more diverse in terms of their tolerance strategies to cope with drought, when compared to communities within a less seasonal rainfall regime (Barros *et al.* 2019). In other words, while higher mean annual precipitation (above 2,500 mm/yr) increases forest resilience (e.g., northwestern Amazon; Hirota *et al.* 2011; Staver *et al.* 2011), forests exposed to higher average seasonality and interannual variability seem to be more drought tolerant, compensating the lower resilience (e.g., eastern x northwestern forests). Valley forests may also be less resistant to droughts than plateau forests due to a similar mechanism, due to a

1   training-effect related to water table fluctuations selected for tree communities with

2   contrasting hydraulic traits (Zuleta *et al.* 2017; Cosme *et al.* 2017; Oliveira *et al.* 2019).

3   Nonetheless, *tipping point*s (2) and (3), related to dry season increases in length and

4   intensity, imply that in forests where climate is already drier, increases in rainfall

5   seasonality could potentially cause forest loss. Also, increases in frequency of extreme

6   drought events may prevent a proper forest recovery (Anderson *et al.* 2018; Longo *et al.*

7   2018).

8   Another heterogeneity that may affect the probability of *tipping point* (1) (1,000 mm/yr;

9   section 2) is related to seasonal flooding. Amazonian floodplains cover 14% of the basin

10   and the forests in these ecosystems were shown to be less resilient than the dominant

11   upland forests, with a potential tipping point of forest collapse at approximately 1,500

12   mm/yr (Flores *et al.* 2017). Therefore, exploring sources of heterogeneities in forest

13   responses to different types of disturbances is key to understand whether the Amazon could

14   shift gradually or abruptly from local to basin-wide scales (e.g., Higgins and Scheiter 2012;

15   Levine *et al.* 2016).

16   *5.2 How does forest connectivity affect large-scale tipping points?*

17   Spatial heterogeneity implies reduced connectivity (less interactions) and may have a huge

18   influence on the systemic resilience of the Amazon, altering how the forest responds to

19   changes in climate change and human pressures (Levine *et al.* 2016; Longo *et al.* 2018).

20   For instance, the climatic, hydrological and biogeochemical connections between the

21   Andes and the low-lying Amazon are undeniably key factors in determining the

22   functioning of the current and future of the entire system on the large scale (see Chapter 22;

23   Builes-Jaramillo and Poveda 2018). Nonetheless, theoretically, connectivity may still be

24   high even in heterogeneous environments, with different processes linking parts of the

25   system (Scheffer *et al.* 2012). Although forests with contrasting geomorphological,

26   climatological, biological and cultural histories have formed the Amazon (see Chapters 1-

27   13; Figure 24.1), these forests can interact. For instance, biogeochemical cycles involve

28   fluxes that transport water vapor from plateau to valley forests on a landscape scale. At

broader scales, large white-water rivers transport huge loads of sediment rich in nutrients from the west to the east of the basin (see Chapter 1), depositing them along floodplains where forests can grow faster. Eastern Amazonian forests are also connected to western forests through rainfall recycling (Zemp *et al.* 2017, see also Chapter 7); a mechanism that enhances the resilience of western forests but may be losing strength due to deforestation (Staal *et al.* 2020). When a forest is disturbed locally, mobile animals may transport tree seeds and propagules from surrounding forests and accelerate its recovery (Lundberg and Moberg 2003). However, mobile animals may also transport seeds of alien invasive grasses from open areas to degraded forested landscapes, increasing their flammability. Local human populations of different Amazonian regions may share ancient knowledge of forest management practices (Levis *et al.* 2018, see also Chapters 8 and 10), potentially changing tree species composition and re-shaping forest resilience.

In sum, connectivity may theoretically increase systemic forest resilience, because spatial interactions facilitate recovery of disturbed sites, but as conditions change and disturbance regimes intensify, increasing for instance landscape fragmentation and wildfires, disturbances may become contagious, resulting in systemic collapse (Scheffer *et al.* 2012) Managing the various processes that connect different parts of the Amazon is therefore critical for enhancing its resilience.

5.3 The interplay between CO2 fertilization effect and nutrient availability

Two of the most pressing uncertainties regarding the resilience of the Amazon forest to climate change and other anthropogenic disturbances are the potential physiological effect of increased atmospheric CO_2 (also known as "CO_2 fertilization effect", eCO_2) and the hypothetical limitations to forest productivity and biomass accumulation imposed by soil nutrient constraints, notably phosphorus (P; see Chapter 1). The current generation of ecosystem models (namely standard Dynamic Global Vegetation and Earth System Models), are constrained in their ability to provide more trustful projections on the impact of climate change on the forest, due mainly to the acute lack of evidence about the

1   existence, magnitude and duration of a CO_2 fertilization effect and associated limitations

2   imposed by soil nutrients (Lapola 2018).

3   On the one hand, the CO_2 fertilization effect could, theoretically, increase forest

4   productivity, biomass accumulation rates (Ainsworth and Long 2005) and water use-

5   efficiency (Kauwe *et al.* 2013). On the other hand, the lack of key nutrients for plant

6   metabolism refrains further biomass gains under elevated CO_2 conditions (Norby *et al.*

7   2010). There are preliminary (i.e., short-term) indications from other phosphorus-limited

8   forests (in sub-tropical Australia), subjected to increased atmospheric CO_2 concentrations,

9   that did not significantly increase biomass (Jiang *et al.* 2020) given that phosphorus is

10   needed especially for making the cell membrane, and also for energetic (ATP) and genetic

11   (DNA and RNA) plant molecules. As such, trees might increase their photosynthetic rates

12   under enhanced CO_2 but do not allocate these extra photosynthates in additional plant

13   biomass, possibly simply increasing biomass turnover rates across the forest ecosystem.

14   Nevertheless, this evidence comes from single-species forests and the response of highly

15   diverse forests such as the Amazon to enhanced CO_2 is yet to be unveiled. In this sense,

16   observational data along a P availability gradient in Panamanian tropical forests revealed

17   that, although such P limitation exists, it does not affect different species in the same way

18   (Turner *et al.* 2018). This latter finding is of particular relevance for the Amazon forest

19   given that climate change and other anthropogenic disturbances may imply significant

20   alteration of the forest tree community composition and dominance relations, both in

21   taxonomic and functional terms (Norby *et al.* 2016). Alternatively, it is hypothesized that

22   Amazon forest trees could change symbiotic exchanges of carbohydrates and nutrients with

23   mycorrhyizae fungi to access currently unavailable soil P pools.

24   Besides the implications to the Amazon forest carbon budget and functional diversity, the

25   physiological effects of elevated CO_2 have the potential to interfere in the flux of humidity

26   from trees to the atmosphere, which is especially relevant for the region, where from 20%

27   to 80% of the rainfall is recycled locally (Zemp *et al.* 2017). In that sense, free-air

28   concentration enrichment (FACE) experiments in temperate forests in the United States and

in an Eucalyptus-dominated woodland in Australia have found a reduction of stomatal conductance and canopy transpiration in the order of -20% (Kauwe *et al.* 2013; Gimeno *et al.* 2016). That is the same magnitude of reduction in transpiration found in recent coupled climate-vegetation modeling studies for the region, which is ultimately related to a basin-wide reduction of 15% to 20% in rainfall (Kooperman *et al.* 2018). Such a rainfall reduction possibly caused by the physiological effect of elevated CO_2 is equivalent to the rainfall reduction in a scenario with complete deforestation of the Amazon (Sampaio *et al.* 2020).

Without an enhancement of productivity and with a reduction of forest canopy transpiration due to increased atmospheric CO_2, the actual Amazon forest and its current community compositions and functional relations are thought to become less resilient to climatic changes, deforestation, degradation and other anthropogenic disturbances, with pervasive impacts on the regional socio-economy (Lapola 2018). Two ongoing ecosystem-scale experiments - the AmazonFACE experiment and the Amazon Fertilization Experiment (AFEX) - will soon provide valuable information about the CO_2 fertilization effect and the limitation of forest productivity and biomass stocks by soil nutrients in the Amazon forest (Hofhansl *et al.* 2016).

6. MODELING THE RESILIENCE AND THE TIPPING POINTS OF THE AMAZON FOREST

Dynamic Global Vegetation Models (DGVMs) and Land Surface Models (LSMs) are the most used tools to mimic the behavior of vegetation dynamics (Sato *et al.* 2015; Fisher and Koven 2020). As those models are capable of simulating long time series of vegetation under global change, they are key to project the future of the Amazon system (e.g., White *et al.* 1999; Cox *et al.* 2004). Often DGVMs and LSMs are the vegetation component in Earth System Models (ESMs), and their success in comprehensively representing processes of vegetation growth and interactions with other Earth System components relies on empirically-derived evidence. This means those models need to make use of the information described in section 4. Given the extreme complexity involved in soil-plant-

1 atmosphere interactions at different temporal and spatial scales, selecting the most relevant

2 processes and implementing them into models are very challenging tasks (Fisher and

3 Koven 2020), and leads to substantial uncertainties (e.g., Rammig *et al.* 2010).

4 As stated above, model simulations can be performed a) offline, meaning the vegetation

5 model is driven stand-alone by externally generated climate data or b) coupled, meaning

6 that the vegetation model is part of an ESM in which different compartments of the Earth

7 System (e.g. the vegetation and the atmosphere) can interact. Such a coupling increases the

8 amount of accounted feedback mechanisms (Box 24.1) which are theoretically necessary to

9 identify *classical tipping points*, besides the prerequisite that the DGVM/LSM allows for

10 the existence of two or more alternative vegetation cover configurations under the same

11 underlying conditions (e.g., climatic) (Box 24.1). For the Amazon ecosystems, tipping

12 point simulations performed so far rely on both offline and coupled runs (*tipping points* (4)

13 and (5) from section 2). Taking the inherent limitations of simulating alternative stable

14 states into account, below we present a summary of what such models can already tell us

15 about dieback, thresholds (Box 24.1) and resilience within the Amazon basin.

16 More than 20 years ago, modeling studies pointed out to a potential Amazon dieback under

17 climate change (White *et al.* 1999; Cox *et al.* 2000, 2004; Cramer *et al.* 2001). Up to now a

18 substantial amount of literature has painted a complex picture with key uncertainties

19 regarding the resilience and potential tipping points of the Amazon under global and

20 regional environmental changes. The results span from clear identification of crossing

21 tipping points in time, as represented by levels of tree cover or biomass stock (e.g., Cox *et*

22 *al.* 2004; Sitch *et al.* 2008), up to an overall increase of biomass and forest cover

23 (Schaphoff *et al.* 2006; Lapola *et al.* 2009; Rammig *et al.* 2010; Huntingford *et al.* 2013).

24 Such a large variety of results can be explained by: 1) whether the DGVM/LSM was

25 coupled (e.g., Cox *et al.* 2004); 2) the existing variety of underlying model assumptions

26 and processes; and 3) general uncertainties on future climate changes in the region.

27 Moreover, DGVMs and LSMs represent vegetation using a limited set of plant functional

28 types (PFTs), which are still not capable of comprising the entire range of plant strategies

1 that confer more or less resilience to the Amazon forests (Oliveira *et al.*). In this sense, a

2 more comprehensive representation of different vegetation ecosystems is needed to

3 improve the simulation of the gradual and abrupt shifts to alternative configurations for the

4 Amazon forests described in section 2. Hence, so far, the is a fairly binary possibility:

5 either the current configuration or a complete replacement of forest by another simulated

6 vegetation type and presented by current models.

7 The main drivers behind this original modeled forest dieback (Cox *et al.* 2004) are the

8 acute reduction in regional rainfall that affects photosynthetic rates and the accompanying

9 increase in temperature that further increases plant respiration, resulting therefore in a

10 considerable reduction of plant productivity and growth. The effects on carbon assimilation

11 also impact the flux of water from the surface vegetation to the atmosphere through

12 transpiration, reinforcing the moisture limitation and ultimately leading to a shift of PFTs,

13 from predominantly tropical broadleaf trees to C_4 grasses with about 30% of broadleaf tree

14 cover, resembling a savanna vegetation (Betts *et al.* 2004; Cox *et al.* 2004). Even without

15 the implementation of such feedbacks through coupling with ESMs, previous offline

16 simulations support such "savannization" processes (section 2.2) under future scenarios of

17 precipitation and temperature changes (Nobre *et al.* 1991; Oyama and Nobre 2003).

18 Importantly, the feedbacks are magnifying the regional climate and vegetation response,

19 and a long-term commitment to Amazon dieback occurs at 2°C global warming,

20 determining an actual *tipping point* (4) from section 2 (Jones *et al.* 2009). Therefore, it is

21 clear that the Amazon dieback is an issue about feedbacks (i.e., interactions within a closed

22 loop) between the regional climate and the forest vegetation functioning. In this sense, a

23 key component is the regional climate response to global warming and the role of non- or

24 dysfunctional forest states in magnifying this process; in other words, whether the regional

25 climate moves from a configuration supporting the rainforest to another, which it does not.

26 This depends on the availability of soil moisture, which itself depends on precipitation and

27 evaporation, both of which change with global warming (see Figure 24.2 for historical and

28 projected changes in some of these variables). If the regional climate reaches a critical

1 state, the resulting forest dieback magnifies the regional climate change and causes further

2 forest dieback.

3 However, as in many regions of the world, the projected changes in precipitation in the

4 Amazon due to anthropogenic climate change are highly uncertain (e.g., Jupp *et al.* 2010).

5 While the majority of the current generation of climate models project a decrease in annual

6 mean precipitation with global warming (see Chapter 22), the rate of the Amazon

7 precipitation decrease in relation to global warming varies widely between the models. A

8 family of climate models notable for their projection of severe Amazon drying, HadCM3

9 (Gordon *et al.* 2000), project the annual precipitation in the eastern Amazon to reduce to

10 below 1,500 mm/yr at approximately 3°C global warming (Betts *et al.* 2012). This

11 precipitation level has been identified as one of the climatic thresholds critical to the

12 support of rainforests (Malhi *et al.* 2009), with empirical evidence suggesting this seems to

13 be the *tipping point* (1) for floodplain forests (Flores *et al.* 2017). The largest decrease in

14 precipitation in the HadCM3 model family was largely a result of atmospheric circulation

15 changes driven by particular patterns of sea surface temperature (SST) (Harris *et al.* 2008).

16 The variation in precipitation change between the models was found to be related to the

17 strength of the SST changes in the equatorial Atlantic (Good *et al.* 2008, 2013). Most other

18 models also project decreased precipitation, but less severely.

19 On the one hand, there are three main underlying drivers to the aforementioned climatic

20 changes that can trigger or reinforce a modeled threshold crossed in the region, even under

21 less severe decreases in precipitation: global climatic changes due to higher atmospheric

22 greenhouse gas (GHG) concentration (Cox *et al.* 2004; Schaphoff *et al.* 2006; Lapola *et al.*

23 2009; Jupp *et al.* 2010; Huntingford *et al.* 2013), deforestation and forest degradation

24 (Sampaio *et al.* 2007; Staal *et al.* 2020), and forest fires (Burton *et al.*; Barlow and Peres

25 2008; Cochrane and Barber 2009; Nobre *et al.* 2016). The occurrence of the climate tipping

26 point for Amazon forest dieback projected in the models therefore depends partly on the

27 nature of the regional climate response to global warming and the impact of CO_2

28 fertilization, wildfires and deforestation (Figure 24.4). If the regional climate response is

28

1 relatively small, forest dieback does not occur. However, if the regional climate response is

2 large, forest dieback could in principle occur and magnify itself through local and global

3 climate feedbacks.
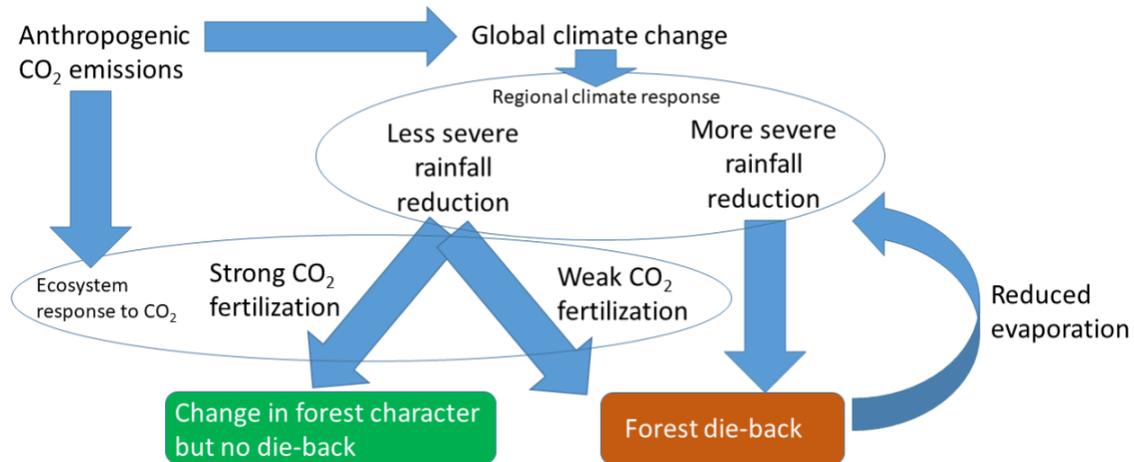
4

5

6 **Figure 24.4.** Simplified sketch of processes involved in the potential Amazon dieback

7 tipping point due to climate change.

8 Regardless of the feedbacks involved, after correcting for biases (found in climatic

9 projections under climate change conditions) identified using observation data, a basin-

10 wide Amazon dieback would be unlikely to occur, even under the most pessimist IPCC

11 pathway (Chai *et al.* 2021). Furthermore, there are some ecological processes that can

12 potentially damp, offset or prevent Amazon dieback, namely the CO_2-fertilization effect

13 under enhanced CO_2 (section 1.4.3) (Hickler *et al.* 2008; Huntingford *et al.* 2013;

14 Kooperman *et al.* 2018), the acclimation of tree physiology to warmer and drier climates

15 (Kumaranthunge *et al*. 2018), as well as the reorganization of forest communities and/or

16 their functional characteristics such that biomass and other broad characteristics that define

17 crucial ecosystem functions are maintained (Sakschewski *et al.* 2016).

18 Processes related to functional diversity (e.g., Fyllas *et al.* 2014; Fischer *et al.* 2016;

19 Sakschewski *et al.* 2016), including plant hydraulics (e.g., Christoffersen *et al.* 2016; Xu *et*

1 *al.* 2016; Eller *et al.* 2020) and rooting depth (Langan *et al.* 2017; Sakschewski *et al.* 2020),

2 have already started to be implemented in current vegetation models to improve the

3 representation of local-scale heterogeneity of the Amazon basin and consequently the

4 ability models have to capture resilience increases due to biotic and abiotic heterogeneity

5 (section 5.1) (Levine *et al.* 2016; Sakschewski *et al.* 2016; Longo *et al.* 2018).

6 Furthermore, models demand a high amount of observational, field-based and/or

7 experimental data, which are still scarce. Kooperman *et al.* (2018), for example, point out

8 that stomatal closure under enhanced CO_2 (as part of the CO_2-fertilization effect) can drive

9 significant modeled rainfall reduction in the Amazon through reduced forest transpiration

10 and moisture recycling (Zemp *et al.* 2017), even though ecosystem-scale evidence on the

11 interplay between enhanced CO_2 and stomatal conductance are very scarce. Adding to that

12 complexity, other studies suggest that stomatal closure under enhanced CO_2 might not turn

13 out as strong as anticipated by models, since leaves need to increase transpiration cooling

14 under elevated temperatures (Dong *et al*. 2014). Another example is that modeled

15 phosphorus limitation (present in about 60% of Amazonian soils (Quesada *et al.* 2012; see

16 Chapter 1) might reduce or even eliminate any gains in primary productivity arising from a

17 supposed CO_2 fertilization effect in the Amazon (Fleischer *et al.* 2019); but, again, there is

18 lack of field data and knowledge on the Amazon phosphorus cycle to corroborate such a

19 result (section 5.3).

20 As such, the way forward for modeling and evaluating the likelihood and mechanisms

21 behind an Amazon tipping point passes first through a closer integration between models,

22 data and field experiments. Field data show us for example that community dynamics – tree

23 recruitment and mortality – plays a key role in the impact of climate change and climatic

24 extremes in the Amazon (section 4) (Esquivel-Muelbert *et al.* 2019, 2020; Hubau *et al.*

25 2020). Thus, improving the representation of such recruitment and mortality dynamics and

26 its driving causes is one priority for modeling. Other processes such as the role of plant

27 hydraulics (Eller *et al.* 2018) and increased plant functional diversity (Scheiter *et al.* 2013;

28 Sakschewski *et al.* 2016), as well as large scale heterogeneities related to climate,

29 hydrology and soil chemistry, for instance, should be explored in more depth by other

1 models. The potential CO_2 fertilization effect on photosynthesis and water use and possible

2 limitation of forest productivity by soil nutrients (section 5.3) represent a *quasi*-complete

3 gap in existing models of the Amazon forest vegetation due to the lack of mechanisms

4 understanding and field data. Last, but not least, narrowing down the uncertainties of

5 rainfall projections for the region would also be very important for better constraining

6 modeling studies on the Amazon tipping point.

7. CONCLUSIONS

8 The pressure of intensified anthropogenic activities has promoted the appearance of new

9 stressing factors operating in the Amazon forests, as well as an intensification of some

10 environmental drivers at different spatial and temporal scales. It has been hypothesized that

11 the cumulative effect of disturbances such as deforestation, droughts, and fires may

12 unbalance the natural dynamics of these globally important ecosystems due to the systemic

13 loss of forest resilience. The analysis of the existing literature performed in this chapter has

14 highlighted five different scenarios of tipping points to which the Amazonian forests could

15 be sensitive to (Figure 24.2), namely: (1) the annual rainfall between 1,000 mm/yr and

16 1,500 mm/yr inferred from global climate models, (2) the dry season length of seven

17 months, inferred from satellite observations of tree cover distributions, (3) for the Amazon

18 lowlands, the maximum cumulative water deficit values between 200 mm/yr and 350

19 mm/yr, inferred from global climate models; (4) an increase of 2°C on the equilibrium

20 temperature of the Earth, inferred from a coupled climate–vegetation model, and (5) the 20-

21 25% accumulated deforestation of the whole basin, inferred from a combination of

22 environmental changes and human-induced degradation via deforestation. Based on

23 empirical evidence, four different ecosystem configurations, some of which could be

24 alternative stable states, have been proposed for Amazonian forests if a tipping point or

25 threshold is crossed, including: (i) a closed-canopy seasonally dry tropical forest state; (ii) a

26 native savanna state; (iii) an open-canopy degraded state; and (iv) a closed-canopy

27 secondary forest state. However, due to the existence of novel feedbacks associated with

28 invasive plants and human-modified landscapes, we consider the open degraded state and

the closed-canopy secondary forest state as more likely to occur over broad areas, particularly across the 'arc of deforestation'. New evidence, however, indicates that in remote parts of the Amazon basin far from the agricultural frontier, the native savanna state could be replacing seasonally inundated forests disturbed by wildfires. Ecological features including differential tree growth, recruitment, and survival among Amazonian species are key to promote forest resistance to, as well recover from, disturbances at local scales. We identify three mechanisms that may affect the risk of a large-scale tipping point due to contagious forest dieback: (a) the environmental heterogeneity and connectivity among forests across the basin; (b) the functional diversity and adaptive capacity of the species present in the different forest types; and (c) the uncertain effect of enhanced CO_2 and nutrient limitation. The lack of this ecological information for many Amazonian species, the uncertainty of the potential feedbacks operating, as well as the need for further improvements in climate change projections hamper the development of robust models for anticipating the potential shifts that Amazonian forests may undergo in the near future. The way forward for modeling and evaluating the likelihood and mechanisms behind an Amazonian tipping point passes first through a closer integration between models, observational data and/or field experiments. Even with models where a tipping point is not met, and accounting for the uncertainty due to the limited data available, we need to urge the international community within and outside academia to protect, maintain and sustainably manage the resilience of these complex and dynamic entities that are the Amazonian forests.

9. RECOMMENDATIONS

- Combining analysis of future environmental change scenarios with past and present dynamics can help improve our understanding of alternative ecosystem configurations;

Chapter 24

1 • Building a holistic and integrative scientific framework with all Working Groups is

2 needed to assess the main heterogeneities, drivers and ways to manage the

3 resilience of the Amazon forest system;

4 • Understanding the heterogeneities of the Amazon is key to assess the risk of a

5 large-scale tipping point and manage the resilience of the system;

6 • Planning an effective transnational monitoring system to improve our knowledge on

7 the dynamics of different Amazon ecosystems (embedded in a wider range of

8 environmental conditions), and potentially their heterogeneous response to various

9 types of disturbances (e.g., climatic extremes, wildfires, deforestation);

10 • Managing Amazonian resilience locally can help reduce the risk of reaching a

11 tipping point. This requires protecting and restoring forest cover, biodiversity,

12 agrobiodiversity, and cultural diversity, as well as improving fire management and

13 fire early-warning systems;

14 • Managing Amazonian resilience also requires global action to halt GHG emissions.

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Chapter 24

1 **CORE GLOSSARY**

2 **Disturbances**: Changes in conditions of a system.

3 **Hysteresis:** Path-dependent behavior of a system.

4 **Perturbations**: Changes in the state of a system.

5 **Positive feedback mechanisms:** Determined by the internal dynamics of the system in a

6 closed loop, i.e., the initial perturbation is self-reinforced and amplified.

7 **Stabilizing (negative) feedback mechanisms:** Occur when system dynamics dampen the

8 initial disturbance/perturbation.

9 **Tipping points:** A critical threshold from which accelerating change caused by positive

10 feedback drives the system to a new state.

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Chapter 24

1 **BOXES**

2 BOX 24.1 – Main concepts and definitions based on the theory of dynamical systems.

3 The theory of dynamical systems suits as a model to any types of system that evolve in
4 time. The dynamics of such systems may have linear, nonlinear, chaotic and complex
5 behaviors, depending on the underlying conditions or the control/explanatory variables, and
6 the response or state variable (Strogatz 2015). For ecosystems such as the Amazon
7 rainforests, conditions would be, for instance, the total amount of precipitation or nutrient
8 availability; the state variable would represent the status of vegetation cover, e.g., tree
9 cover percentage or productivity. When the system presents nonlinear dynamics, we can
10 have a steep but still gradual shift from one ecosystem state to another (Figure 24.B1, left
11 panel b), meaning that for each condition there is one and only one ecosystem state
12 associated; and a more abrupt or catastrophic shift (Figure 24.B1, left panel c), when two
13 (or more) ecosystem states can exist under the same set of conditions (the reason why the
14 sigmoid from panel b turns into an s-shaped curve in panel c). The two possible
15 configurations (continuous blue line on left panel c) represent stability and are called
16 *alternative stable states* or *attractors*; and the dashed blue line in the middle represents the
17 transient behavior of the system and is called the *unstable states* or *repellors* (from there
18 the system could move either upwards to the higher stable state or downwards to the lower
19 state - see red arrows pointing up and downwards).

20 [Figure 24.B1 is being developed and will be inserted here]

21 **Figure 24.B1** - (Left panels) Linear and nonlinear responses of ecosystem state (y-axis)
22 depending on underlying conditions (x-axis). (Right panels) Illustration of how catastrophic
23 shifts can occur under changes in conditions (e.g., climate changes) and in the state variable
24 (e.g., human activities). Modified from Scheffer *et al.* (2001).

25 The two black open circles (*F1* and *F2*) are named *bifurcation points*, *tipping points* or
26 *critical thresholds.* In this sense, such *tipping points* exist only when two or more
27 *alternative stable states* occur (Scheffer *et al.* 2001). *Tipping points* can be reached if either

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disturbances (changes in conditions) or *perturbations* (changes in the state), or both occur

(Fig. 24.B1, right panels) (Van-Nes *et al.* 2016). First, **if conditions change** and *F2* is

crossed (Fig. 24.B1, right panel a), a sudden drop (downwards) can occur towards a

different state. Interestingly, to return to the original state, the system would need to

undergo a much stronger change in conditions, in this case, to reach the other bifurcation

point *F1*, which could lead the system upwards again. This *path-dependence* behavior is

called *hysteresis*. Such a feature defines the likelihood of irreversibility after crossing a

tipping point. Eventually, it is so challenging to return conditions to *F1* levels, and thus

return to the original state, that reaching a *tipping point* can indeed cause irreversible

changes. In the case of the Amazon rainforests, climate change translated into extreme

drought events or increases in dry season length could represent changes in one of

underlying conditions that maintain Amazon ecosystems in the current configuration.

Secondly, **if changes occur in the ecosystem state**, e.g., decreases in tree cover after

deforestation and/or wildfire events, the system could reach the instability region (blue

dashed line), causing either a return to the original state or a (irreversible) change in the

system configuration. In either case what drives the accelerated shift to a new state are

positive *feedback mechanisms* (DeAngelis *et al.* 1986), determined by the internal

dynamics of the system in a closed loop, i.e., the initial perturbation is **self-reinforced and**

amplified. For instance, deforestation leads to less tree cover, which, in turn, leads to less

evapotranspiration, less precipitation and thus less tree cover, i.e., in this case, the initial

perturbation is reinforced and amplified. On the other hand, (**negative**) **stabilizing** *feedback*

mechanisms occur when they dampen the initial disturbance/perturbation (DeAngelis *et al.*

1986). Therefore, in broader context, *tipping points* can refer "to any situation where

accelerating change caused by a positive feedback drives the system to a new state" (Van-

Nes *et al.* 2016).

The connection between *tipping points* and *resilience* is more easily observed when

building *stability landscapes* (or ball-in-a-cup diagram) using the concept of *basins of*

attraction (Fig. 24.B2a, b) (Scheffer *et al.* 2001; Strogatz 2015). In this sense, theoretically

resilience can be qualitatively understood as the **size** of the *basin of attraction* (valleys on

1   Fig. 24.B2a). Each cross-section of the ecosystem state *vs.* conditions graph corresponds to

2   a different *stability landscape*, showing potential *alternative stable states* and the **size** of the

3   *basin of attraction* separating them. Particularly, for tropical forests, Fig. 24.B2b shows

4   five condition cross-sections (for increasing precipitation): 1) only a treeless state, i.e., only

5   one *basin of attraction* representing one state possible; 2) two *alternative stable states*,

6   namely treeless and savanna, with a higher resilience (deeper valley) associated with the

7   treeless state; 3) and 4) forests and savannas as *alternative states* with higher forest

8   resilience related to higher levels of precipitation; 5) only forests as a stable state with the

9   highest levels of precipitation. Note that this diagram shows only precipitation as a driving

10   condition. We can go further and think about changes in the conditions or in the ecosystem

11   state (Fig. 24.B1) using this type of diagram (Figs. 24.B2c-e). For instance, increases in the

12   frequency of extreme droughts and/or in dry season length could **erode** the *basin of*

13   *attraction* of the forest state, i.e., forests lose resilience up to a point that a relatively lower-

14   intensity drought could trigger a shift towards another *basin of attraction* easier than if

15   climate change impacts would not occur (Fig. 24.B2d). Human-induced changes affecting

16   the ecosystem state directly (e.g., wildfires or deforestation) would provoke a state flip

17   independently on whether forests had lost resilience or not (Fig. 24.B2e).

18   [Figure 24.B2 is being developed and will be inserted here]

19   **Figure 24.B2** - The connection between *tipping points* and *resilience* using *stability*

20   *landscapes*. Modified from Scheffer *et al.* (2001); Hirota *et al.* (2011); van Nes *et al.*

21   (2016).

22   Based on the ball-in-a-cup diagram, we use the qualitative definition of resilience as the

23   capacity of the Amazon region to persist as a tropical rainforest, maintaining similar

24   interactions and functioning, despite being constantly pushed away from its stable states by

25   disturbances and perturbations (Holling 1973).