Chapter 9
Long-Term Vegetation Change in Central Africa: The Need for an Integrated Management Framework for Forests and Savannas

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9.1 Introduction

Tree-dominated forests and grass-dominated savannas represent the two main tropical biomes covering the overwhelming majority of sub-Saharan Africa (SSA) (White 1983). Forests and savannas have a different, even antagonistic, ecological functioning (Staver et al. 2011a), but they both provide critical services to local populations (see Chaps. 7 and 10, Vol. 1; Chaps. 5 and 6, Vol. 2). Tropical forests are dominated by trees, forming closed canopies and complex vertical structures, and contain in their understory C3 grasses that are more adapted to humidity and shadow. Such forests are encountered in areas of high annual rainfall and limited seasonality (Malhi et al. 2009), and are very sensitive to disturbances. In contrast, trees and C4 grasses in tropical savannas coexist, are more adapted to aridity, are shade intolerant, and are found in areas that are drier and have higher seasonality (Ratnam et al. 2011). Savannas rely on frequent disturbances due to fires and/or mega-herbivores that maintain an open canopy and species diversity (Bond et al. 2005; Sankaran et al. 2005; Staver and Bond 2014).

The occurrence of tropical forests and savannas is, however, not rigidly determined by climatic conditions. Recent analyses of remotely sensed tree cover at the global (Hirota et al. 2011; Staver et al. 2011b) and at the regional (Favier et al. 2012) scales, supported by theoretical work (Staver et al. 2011a; Staver and Levin 2012),

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have demonstrated that for intermediate rainfall regimes,[1] forests and savannas can both occur in the same regions. Under these climatic conditions they represent alternative stable states maintained by a positive feedback between fire and canopy cover (Hirota et al. 2011; Staver et al. 2011b; Favier et al. 2012). These two ecosystems are unique, highly biodiverse, and can provide multiple ecosystem services that are directly related to their unique functioning (see Chaps. 7 and 10, Vol. 1; Chaps. 5 and 6, Vol. 2).

Tropical forests harbor more than 60% of all known species (Laurance and Useche 2009), are characterized by unique food webs and high endemism and diversity (Malhi et al. 2014). Tropical forests are also critical for the global climate system (Spracklen et al. 2018) as they represent one of the largest carbon sinks, which constitutes 40–50% of terrestrial carbon stocks (Pan et al. 2011; Achard et al. 2014).

The Congo basin is the largest forest block in SSA in terms of size, and is the world's second largest behind the Amazon. However, tropical forests in SSA have been heavily used for a very long time, and can hardly be considered as pristine (Chap. 10, Vol. 1), see also Willis et al. (2004) and Roberts et al. (2017) for evidence of human activity across the tropics, and Morin-Rivat et al. (2017) across central Africa). At the same time, tropical forests are facing the new pressures of the Anthropocene (Malhi et al. 2014), such as accelerated deforestation and forest degradation due to slash-and-burn subsistence agriculture, and more recently conversion for cash crops (e.g., cocoa, hevea) production (Sonwa et al. 2011), among many other pressures (IPBES 2018). Land use change, that is, the conversion of forest land into another land use, is the most important threat for tropical forests worldwide (Sala et al. 2000; Malhi et al. 2014; Kehoe et al. 2017), and in particular for SSA (Aleman et al. 2016, 2017). Deforestation occurs particularly around big cities, and along major road axes and rivers (Verhegghen et al. 2012), and is expected to continue along such patterns in the near future (Laurance and Arrea 2017).

The logging industry is dominated by industrial scale operations, also representing a major driver of forest fragmentation across central SSA (Laporte et al. 2007). Timber production is of extreme importance for the national economies in the region, with production forests spanning 180 million ha (De Wasseige et al. 2012) (Chap. 1, Vol. 1). However, timber production does not necessarily induce land use change (Pan et al. 2013). Indeed, temporal changes in forest cover due to forest exploitation do not imply a conversion to another land use due to the quick recovery of tropical forests after disturbance (Poorter et al. 2016) or silvicultural interventions (Gourlet-Fleury et al. 2013). Moreover, forest management in central SSA is highly selective with only few trees per hectare extracted over 25–30 year rotations (Ruiz Pérez et al. 2005). This allows for the maintenance of a permanent forest cover after logging, and offers the possibility to manage forests for timber

[1]Intermediate rainfall regimes are 1000–2000 mm for SSA (Staver et al. 2011a) and 1000–2500 mm for the global tropics (Staver et al. 2011b).

extraction and carbon storage (Nasi et al. 2012). Yet, hunting is a non-negligible side-effect, as it nearly always accompanies logging in tropical forests of SSA, having dramatic consequences on animal populations and diversity (Poulsen et al. 2011). Defaunation is indeed occurring at an alarming rate in central SSA (Nasi et al. 2011; Abernethy et al. 2016), potentially having substantial impacts on ecosystem functioning, and specifically tree regeneration and recruitment as demonstrated in other parts of the world (for example, see Terborgh et al. (2008) for southeastern Peru).

Tropical savannas cover 20% of the global land surface, account for 30% of terrestrial net primary productivity, and sustain 20% of the human population, while hosting most of the remaining megafauna (Scholes and Archer 1997; Veldman et al. 2015a; Bond 2016). Tropical savannas cover the overwhelming majority of SSA, while provide important services to local populations (IPBES 2018). Furthermore, savannas and other grassy ecosystems harbor important biodiversity, including the largest herbivores on Earth (Parr et al. 2014; Bond 2016).

Savannas are commonly used in SSA for agriculture, pastoralism, and fuelwood extraction, catering for the livelihoods of many local communities (Scholes and Archer 1997; IPBES 2018) (see also Chaps. 3 and 5, Vol. 2). Recent land use change has been threatening savanna ecosystems across SSA, with many savannas identified as potentially interesting areas for large-scale conversion to industrial/cash crops (Searchinger et al. 2015) (Chap. 3, Vol. 1), biofuel feedstock production (Alexandratos and Bruinsma 2012) (Chap. 2, Vol. 1; Chap. 5, Vol. 2), and forest plantations for forest restoration and CO_2 sequestration (Laestadius et al. 2012). Contrary to tropical forests, savannas rely on chronic disturbances for maintaining an open and diverse ecosystem (as detailed above). In fact, savannas require antagonist management strategies such as recurrent or prescribed fires in high rainfall areas (Lehmann et al. 2014; Osborne et al. 2018), which are not taken into account by the forest scientific community (Veldman 2016).

Mesic savannas are located in areas where forest can also occur, and are especially targeted for the types of land conversion outlined above (Searchinger et al. 2015; Aleman et al. 2016). These types of savannas are indeed often mistaken as "degraded" landscapes, or even as secondary successional stages following deforestation, which are of little value for biodiversity conservation, see arguments in Bond and Parr (2010), Parr et al. (2014), and Veldman et al. (2015b). This is mainly due to the fact that there is no recognition regarding their status as an alternative stable state (Bond and Zaloumis 2016). This is clearly illustrated in the ongoing debate on how to define forests.[2] A recent study that reassessed the extent of forests in dryland biomes worldwide (Bastin et al. 2017), mistakenly considered most

[2]The FAO defines forests as areas of more than 0.5 ha that have tree cover >10% (Pan et al. 2013). Other authors have proposed higher thresholds such as tree cover >65% (e.g., Hirota et al. 2011; Staver et al. 2011a).

savannas as dry forests[3] (Bond and Zaloumis 2016; Griffith et al. 2017). The above suggest that the FAO definition of forest is problematic, as most mesic savannas would be consequently considered as forests (more specifically degraded forests), and could thus be targeted for reforestation (Laestadius et al. 2012).

For example, the Miombo woodlands that cover much of southern Africa has been identified as a particular ecoregion (Olson et al. 2001), and is considered as forest by Bastin et al. (2017) or other type of woodland by Achard et al. (2014). However, Miombo woodlands in which trees are widely spaced and do not form a closed canopy cover are associated to the savanna flora (Linder 2014). They are also associated to the savanna biome as they have a grassy understory and experience regular fires (Ratnam et al. 2011; Dexter et al. 2015; Pennington et al. 2018). The Miombo woodlands host a high floristic richness and specificity, and are dominated by species of the *Brachystegia*, *Julbernardia*, and *Isoberlinia* genera (White 1983). They also provide many different types of natural resources and ecosystem services such as timber products, fuelwood, and a huge variety of non-timber forest products including beeswax and honey, mushrooms, edible caterpillars, wild fruits, and livestock grazing (Lawton 1982; Malaisse 2010) (see Chap. 5, Vol. 2). With millions of people living in (and depending on) the Miombo woodlands (Bradley and McNamara 1993; Dewees 1994), these landscapes are heavily affected by human activity and are under threat from increasing fragmentation and deforestation (Chazdon 2008).

The above suggests that forests and savannas are both two very important tropical biomes in SSA and face the new threats posed by the Anthropocene. Land use change (e.g., conversion to agriculture in forest areas, and industrial plantations in savanna areas) and mismanagement (e.g., extensive logging and woodfuel harvesting, uncontrolled fire regimes) threaten biodiversity and ecosystem services from these biomes. Furthermore, they can compromise the critical role that tropical forests and savannas play for local livelihoods and global climate regulation.

In this respect the loss of tropical forests and savannas and the need for their effective management have become major sustainability challenges in SSA, and especially in central Africa. In particular, there is a need for their integrated management, taking into account that they can be alternative stable states in some areas (i.e., under intermediate rainfall) (Staver et al. 2011a; Favier et al. 2012), and that savannas are not simply degraded forests only useful for reforestation and afforestation projects. In this context, it is important to identify the appropriate management responses to the drivers of vegetation change in forests and savannas of central Africa.

The aim of this chapter is to provide a comprehensive analysis of the past, present, and future vegetation changes in central African forests and savannas. We focus particularly on historical and future changes related to climate change and land

[3]Tropical dry forests are an important biome in the Neotropics, the third tropical biome (Dexter et al. 2018). However in SSA tropical dry forests are supposed to be restricted only to a very small area in East Africa (Pennington et al. 2018).

use change. We start by reviewing the vegetation changes since the Last Glacial Maximum (approximately 21,000 years ago), and assessing the current distribution of major vegetation types in relation to different drivers, mainly related to climate and fire regimes. We then project future vegetation cover based on rainfall variability and agricultural activity, using different scenarios. Section 9.2 outlines the main historical milestones shaping vegetation change in central Africa, and the main data collection and analysis approaches. Section 9.3 highlights the main past (Sect. 9.3.1), current (Sects. 9.3.2), and future (Sect. 9.3.3) vegetation changes, and their drivers. Section 9.4 synthesizes the main findings and discusses some of the main policy implications for restoring ecosystems, expanding protected areas, and designing sustainable forest management approaches in the region.

9.2 Methodology

9.2.1 Central Africa and Its History

9.2.1.1 Study Area

This chapter focuses on vegetation changes in central Africa, an area that covers the Congo Basin, and adjacent savannas and woodlands (including the Miombo). For practical reasons, the study area is defined by the boundaries of Cameroon, the Central African Republic (CAR), the Democratic Republic of Congo (DRC), Equatorial Guinea, Gabon, and the Republic of Congo, and spans more than 4 million km^2. According to the Tropical Rainfall Measuring Mission (TRMM, Nicholson et al. 2003), the environmental conditions vary widely across this vast study area. Mean annual rainfall ranges between 440 and 3220 mm, from the driest sites in North Cameroon to the wettest sites in the Coast of Cameroon.

The distribution of forests and savannas in this area is the legacy of a long history of climate changes and human impacts. Table 9.1 summarizes the main periods of climate and vegetation changes since 21,000 cal yr BP (i.e., calibrated years before present, the present being 1950 by convention).

9.2.1.2 Paleo-History

Paleo-environmental reconstructions[4] suggest that central Africa has experienced a succession of dry and humid periods, with tropical forests contracting or expanding in response (Maley 1989, 1996; Vincens et al. 1999).[5] The Last Glacial Maximum

[4]Paleoenvironmental reconstructions entail the reconstruction of past vegetation, climate and disturbances using bio-proxies (e.g., pollen, phytoliths, diatoms, charcoal particles) and the biogeochemical analysis of natural archives (e.g., sediments or paleosols).

[5]In comparison the Amazon remained much more stable during the same period (Anhuf et al. 2006).

Table 9.1 Main periods of climate and vegetation changes in central Africa

Period	Climate and vegetation
Last glacial maximum (~21,000 cal yr BP)	Low rainfall and temperature. Open vegetation, Afromontane species, the Congo Basin is supposedly reduced to few forest refuges.
Younger Dryas (from ~12,900 to ~11,700 cal yr BP)	Intensely dry period. Open vegetation.
African humid period (from 11,700 to 4000 cal yr BP)	Higher rainfall than currently, increase in temperature. Largest forest extent of the Congo Basin.
Third millennium rainforest crisis (~4000 to 1200 cal yr BP)	Rainfall modification, droughts. Contraction and fragmentation of the Congo Basin.
Since 1200 cal yr BP	Increase in rainfall. Forest transgression.

Note: cal yr BP stands for calibrated years before present, the present being 1950 by convention

(~21,000 cal yr BP) was a period of very low rainfall and temperature, while the subsequent Younger Dryas period (from ~12,900 to ~11,700 cal yr BP) was a short, but intensely dry, period. During these two periods, forests in central Africa were supposedly reduced to small patches (Maley 1989, 1996). Rainfall started increasing at the beginning of the Holocene (~11,700 cal yr BP), reaching higher rainfall levels than currently observed, with this period commonly referred to as the African Humid Period (de Menocal et al. 2000; Shanahan et al. 2015). During this period, tropical forests were more widespread across central Africa than currently (Vincens et al. 2010, 1998; Lebamba et al. 2016).

The African Humid Period ended abruptly at ~4000 cal yr BP, and was followed by the "third millennium rainforest crisis." This period was characterized by low rainfall and major droughts that lasted until 1200 cal yr BP (Vincens et al. 1999), resulting in major perturbations across the Congo forest block. The relative roles of climate and people during the third millennium crisis have been hotly debated in the literature. Some scholars favor the climate hypothesis (Maley et al. 2012; Neumann et al. 2012b; Lézine et al. 2013; Giresse et al. 2018), and others suggest that the migration of Bantu people from the border of Cameroon and Nigeria also contributed to large-scale forest disturbances (Bayon et al. 2012; Garcin et al. 2018). The Bantu people were farmers and metallurgists (Bostoen et al. 2015) who used slash-and-burn farming techniques and required large quantities of wood for metal processing. They tended to farm pearl millet (Neumann et al. 2012a) and raise cattle (Grollemund et al. 2015) within the tropical forest of western and central Africa. Even if they were not responsible for this large-scale event, they may have caused more localized perturbations (Garcin et al. 2018), through canopy opening and wood collection (Van Gemerden et al. 2003; Neumann et al. 2012a). Additionally, charcoal has been found in lakes, wetlands and soils, in the deepest parts of the forest, suggesting localized forest burning (Hubau et al. 2013; Tovar et al. 2014; Biwolé et al. 2015; Morin-Rivat et al. 2016). There was an increasing occurrence of charcoals, as seasonality increased drastically in ~2500 cal yr BP (Hubau et al.

2015). Moreover, people may have played a role in maintaining the newly formed savannas in areas located in the periphery of the Congo forest.

Rainfall started increasing again after 1200 cal yr BP, leading to forest expansion, a trend still observed today in some areas of Cameroon (Youta Happi 1998) and CAR (Guillet et al. 2001). However, the more recent human activity is also important for explaining the current distribution and composition of forests and savannas in central Africa (Willis et al. 2013; Morin-Rivat et al. 2017).

9.2.1.3 Recent History

Human occupation patterns were strongly modified following the colonization of the central African region in the 1900s, which had impacts on current forest composition (Van Gemerden et al. 2003; Engone Obiang et al. 2014; Morin-Rivat et al. 2017). Indeed, even though local communities used to live deeper into the forest before the colonial period, they were then forced to gather in the European trading centers and villages along the main roads. This demographic change is now reflected in forest composition, as the populations of tree species that dominate the canopy are now aging (Engone Obiang et al. 2014; Morin-Rivat et al. 2017).

Following European colonization, deforestation and forest degradation also started to increase. Indeed, the industrialization of the region (starting in ~1920) was associated with the development of extensive oil palm, coffee, and rubber plantations (Van Reybrouck 2012) (see Chap. 3, Vol. 1). This process is anecdotally recorded in lake sediments, where forest degradation has been associated with the increasing prevalence of fire within forest areas (Aleman et al. 2013). This supports the notion that plantations and logging activities, coupled with the development of roads, has been a major cause of forest degradation in the region.

However, despite these recent changes (and their impacts on the vegetation), the tropical forests of central Africa have experienced relatively low deforestation rates since the 1900s (Aleman et al. 2018), and even since the 1980s (Pan et al. 2011; Achard et al. 2014), compared to the other tropical regions (Sodhi et al. 2010; ter Steege et al. 2015). The socioeconomic and political instability that has plagued most countries in the region since their independence from colonial powers have prevented the large-scale expansion of agricultural activities and deforestation. More recently, forest degradation, and specifically forest fragmentation (Malhi et al. 2014) due to road development (Laurance et al. 2017) and logging (Laporte et al. 2007) have been identified as a major threat for the biodiversity (Poulsen et al. 2011; Abernethy et al. 2016; Ziegler et al. 2016, as also reported in the IPBES report for Africa). If the broad region eventually stabilizes, a surge of land conversion may be expected, particularly linked to the expanding oil palm sector (IPBES 2018).

The recent history of land use change also affected vegetation structure and composition within savannas. First, fire was seen as a destructive force during the colonial period that was responsible for the deforestation of extensive areas. Indeed, most mesic savannas are located in areas where the rainfall range allows forests and savannas to represent alternative stable states (i.e., 1000–2000 mm), and these savannas contain gallery forests and forest patches. Mesic savannas were then seen as relatively new and formed through anthropogenic activity (Aubréville 1939, 1947; Bond and Zaloumis 2016). During this period, many fire suppression policies were developed in an attempt to protect tropical forests (see Bond and Zaloumis 2016). Fire practices were heavily modified, triggering changes in fire regimes, and ultimately leading to woody encroachment in savannas (Aubréville 1947).

More recently, Sahelian pastoralists from west and central Africa started migrating in mesic savannas in the 1960s, in areas where Mbororo farmers dominated historically the agricultural landscape (Bassett and Boutrais 2000; Ankogui-Mpoko 2003). This has led to important land conflicts between pastoralists and farmers in the CAR (Ankogui-Mpoko 2003) and Cameroon (Ouikon 2003), and ultimately changed the vegetation structure (Bassett and Boutrais 2000). In particular, changes in agricultural practices modified the fire regime (i.e., from large fires for hunting, to small and early fires for cattle herding), which, coupled with overgrazing, resulted in woody encroachment (Bassett and Boutrais 2000).[6] Pasturelands became less productive "forcing" cattle herders to migrate southwards (Ankogui-Mpoko 2003).

9.2.2 Data Collection and Analysis

9.2.2.1 Reconstruction of Past Vegetation Changes

In this chapter, we conduct a meta-analysis of published studies to better understand and illustrate past changes on vegetation cover, and their determinants (e.g., fires, human presence, and climate) since the Last Glacial Maximum (~21,000 cal yr BP) (see Sect. 9.3.1). We specifically focus on vegetation reconstruction studies from 14 paleo-ecological sites (Table 9.2), for which palynological information is available.

Pollen preserved in lake sediments can be used to infer past vegetation composition and structure (Vincens et al. 1999). For example, we used data downloaded from the African Pollen Database[7] to explain composition changes at Lake Barombi Mbo. Each of the different pollen sites was recorded, and then plotted on a map of central Africa. Dated charred material (e.g., oil palm endocarps, charcoals) can be used as a proxy for local past fire events (Vleminckx et al. 2014). We plotted this information on the same map described above to illustrate the location of past fire events, even those located very deeply within the forest block.

We also reviewed the evidence of past human presence and activity in central Africa, using all of the dated information on human presence assembled by Oslisly

[6]Woody encroachment in savannas is currently widespread throughout SSA, and even in other parts of the world (Stevens et al. 2017). While local encroachment has been directly linked to land use change, global climate change, and specifically CO_2 fertilization, has also been identified as a possible driver (Buitenwerf et al. 2012).

[7]For more information refer to: http://fpd.sedoo.fr/fpd/

Table 9.2 List of palynological sites used to explain vegetation change

#	Site	Latitude	Longitude	References
1	Mbalang	7.32	13.73	Vincens et al. (2010)
2	Tizong	7.25	13.58	Lebamba et al. (2016)
3	Bambili	5.94	10.24	Assi-Kaudjhis (2012)
4	Barombi Mbo	4.67	9.40	Giresse et al. (1994)
5	Ossa	3.80	10.75	Reynaud-Farrera et al. (1996)
6	Nyabessan	2.67	10.67	Ngomanda et al. (2009)
7	Maridor	−0.17	9.35	Ngomanda et al. (2007)
8	Nguene	−0.20	10.47	Ngomanda et al. (2009)
9	Mopo Bai	2.23	16.26	Brncic et al. (2007, 2009)
10	Goualougo	2.16	16.51	Brncic et al. (2007, 2009)
11	Sinnda	−3.83	12.80	Vincens et al. (1994)
12	Kitina	−4.25	11.98	Elenga et al. (1996)
13	Coraf	−4.75	11.85	Elenga et al. (2001)
14	Tilla	10.39	12.13	Salzmann (2000)

et al. (2013) and complemented by Morin-Rivat et al. (2014). The combined dataset consists of 585 ^{14}C-dated records from archaeological sites covering the period 5000 to 100 cal yr BP. Each record is calibrated using the IntCal13 curve and a probability density is obtained through a Bayesian modeling procedure in the “Bchron” package (R Core Team 2015). For each date, we used the median of the probability density, and bin the number of dates every 50 years.

Finally, we illustrated climate change effects during this period, by reporting sea surface temperature (SST) reconstructed from a core in the Guinea Gulf (Weldeab et al. 2007).

9.2.2.2 Assessment of Current Vegetation Cover and Change

We used remotely sensed information about vegetation types (Verhegghen et al. 2012) at the regional scale with a pixel resolution of 900 m, to understand current vegetation types and recent vegetation cover changes across the study region (i.e., between 2000 and 2015) (see Sect. 9.3.2). We aggregated the different vegetation types into the following eight classes: (1) tropical forest; (2) montane forest; (3) swamp forest; (4) mangrove; (5) savanna (containing the different savanna types of the original map); (6) forest–savanna mosaic; (7) Miombo woodland; and (8) areas impacted by humans. The areas affected by humans contain all such classes from the original map, including agricultural areas, human settlements, and roads, among others. To delineate the main biogeographic zones (i.e., phytochoria) in central Africa, we used the biogeographic regions from White (1983) and Linder et al. (2012).

Altitude and topography are important determinants of vegetation distribution across the Tropics, including in central Africa. A very important distinction in the study area is between mountain and lowland forests (Letouzey and Aubréville 1968; White 1983). Within the lowland forest category, swamp forests and mangroves are distinguished from *terra firme* forests (Letouzey and Aubréville 1968), and can be remotely sensed (Verhegghen et al. 2012). Even though such forests occupy vast areas in the Congo basin, and hold huge amount of carbon (Dargie et al. 2017), we focused here only on *terra firme* vegetation, excluding mangrove and swamps from our analysis.

We further studied how tree cover and tree cover changes are distributed across forests and savannas (grouping classes 5–7 outlined above). This is because tree cover is a key variable for both forest and savanna functioning, enabling the differentiation of forests and savannas at a broader scale (Staver et al. 2011a; Aleman et al. 2016; Aleman and Staver 2018). Tree cover within savannas has been shown to be highly variable (Sankaran et al. 2005) and largely determined by annual rainfall (Bucini and Hanan 2007). However, even though annual rainfall indeed constraints maximum tree cover, disturbances (especially fire and herbivory), complex interactions between disturbances (Hanan et al. 2008; Staver and Bond 2014) and soil composition (Sankaran et al. 2008) can also reduce tree cover from its climatic potential in a less predictable way (Sankaran et al. 2005, 2008; Staver 2018).

We used remotely sensed information on tree cover, initially available at 30-m resolution, and aggregated at the resolution of the vegetation types (900 m) (Hansen et al. 2013). Tree cover losses between 2000 and 2015 are also available from the same dataset (Hansen et al. 2013) and were used to identify the forest and savanna areas that have been heavily modified during that period.

9.2.2.3 Identification of Natural and Anthropogenic Drivers of Vegetation Change

A wide array of drivers is currently influencing vegetation structure (Aleman et al. 2017) and composition in forests (Fayolle et al. 2012, 2014b) and savannas (Fayolle et al. 2019). We focused here on annual rainfall and fire as drivers of vegetation change (Sect. 9.3.3), as they are important in determining forest and savanna as alternative stable states in the climate zone where both forests and savannas co-occur (Staver et al. 2011a).

We used for annual rainfall the 3B43 Monthly gridded rainfall product from TRMM, available at 0.25° resolution (Nicholson et al. 2003). For fires, we used the burned area monthly product from the MODerate-resolution Imaging Spectroradiometer (MODIS) sensor available at 500-m resolution (Giglio et al. 2010). Both datasets are resampled at the resolution of the vegetation types (900 m) (Sect. 9.2.2.2), using a nearest neighbor procedure and the package “raster” in the R environment (R Core Team 2015). We computed annual rainfall between 2002 and 2015, and the number of times an area represented by a pixel was burned between 2002 and 2015 as an index of fire frequency.

As mentioned in Sect. 9.2.1, ecosystems in central Africa are facing an increasingly high pressure from human activity, but are also threatened by ongoing climate change (see also Chap. 10, Vol. 1). Through a literature review, we identified the main threats that forests and savannas are experiencing in central Africa. Additionally, we extracted spatial information on land allocated for biodiversity conservation (i.e., protected areas) and to economic exploitation, specifically logging concessions. We used protected area maps[8] for the six countries of central Africa mentioned (Sect. 9.2.1) and data from the World Resources Institute to produce a map of the areas under logging concessions and identified as having forest restoration opportunities.

9.2.2.4 Prediction of Future Vegetation Change

We estimated the area of current forest and savanna that may be impacted by future rainfall and agricultural activity (Sect. 9.3.3). In particular, we used the projected annual rainfall and cropland area for 2070 in central Africa derived from the two most contrasted scenarios of future global change: the Representative Concentration Pathways (RCP) 2.6 and 8.5. These RCPs represent different trajectories for greenhouse gases (GHG) concentrations and radiative forcing for the year 2100 (Van Vuuren et al. 2011). RCPs contain different assumptions about socioeconomic trajectories (e.g., demography, economy, land use).

RCP 8.5 is the most pessimistic scenario (Van Vuuren et al. 2011). In this scenario, radiative forcing and GHG concentration continue to increase at the current rate, while cropland and pasture land also expand significantly due to large population increase (Hurtt et al. 2011). Conversely, RCP 2.6 represents the most optimistic scenario in terms of GHG emissions, as they are expected to eventually decline due to declining fossil fuel consumption and aggressive climate policies implemented to reduce climate change. A crucial element of RCP 2.6 is the use of bioenergy, and carbon capture and storage technologies, which are expected to result in negative emissions (see Chap. 2, Vol. 1). However, the extensive bioenergy expansion is expected to cause a large increase in the cropland dedicated to biofuel production (Hurtt et al. 2011).

Future climate projections for the year 2070 (average 2061–2080) were taken from all general circulation models (GCM) for the two RCPs, which are available downscaled and calibrated against Worldclim 1.4 as baseline climate (Hijmans et al. 2009), using relative change for rainfall. Future land use projections for cropland area for the year 2070 are averaged between 2061 and 2080 for the two RCPs, in order to be consistent with the climate data.

We compared areas that are currently forests and savannas, and that will be impacted by future changes in annual rainfall (>100 mm year^{-1}) and by changes in the fraction of pixels affected by cropland (>0.25). This analysis is conducted for forests that are currently under logging concession and for forests/savannas areas that are currently protected areas. The rationale is that protected areas can potentially buffer against climate and/or land use change (Loarie et al. 2009; Aleman et al. 2016).

[8]For more information refer to: https://www.protectedplanet.net (accessed August 2017).

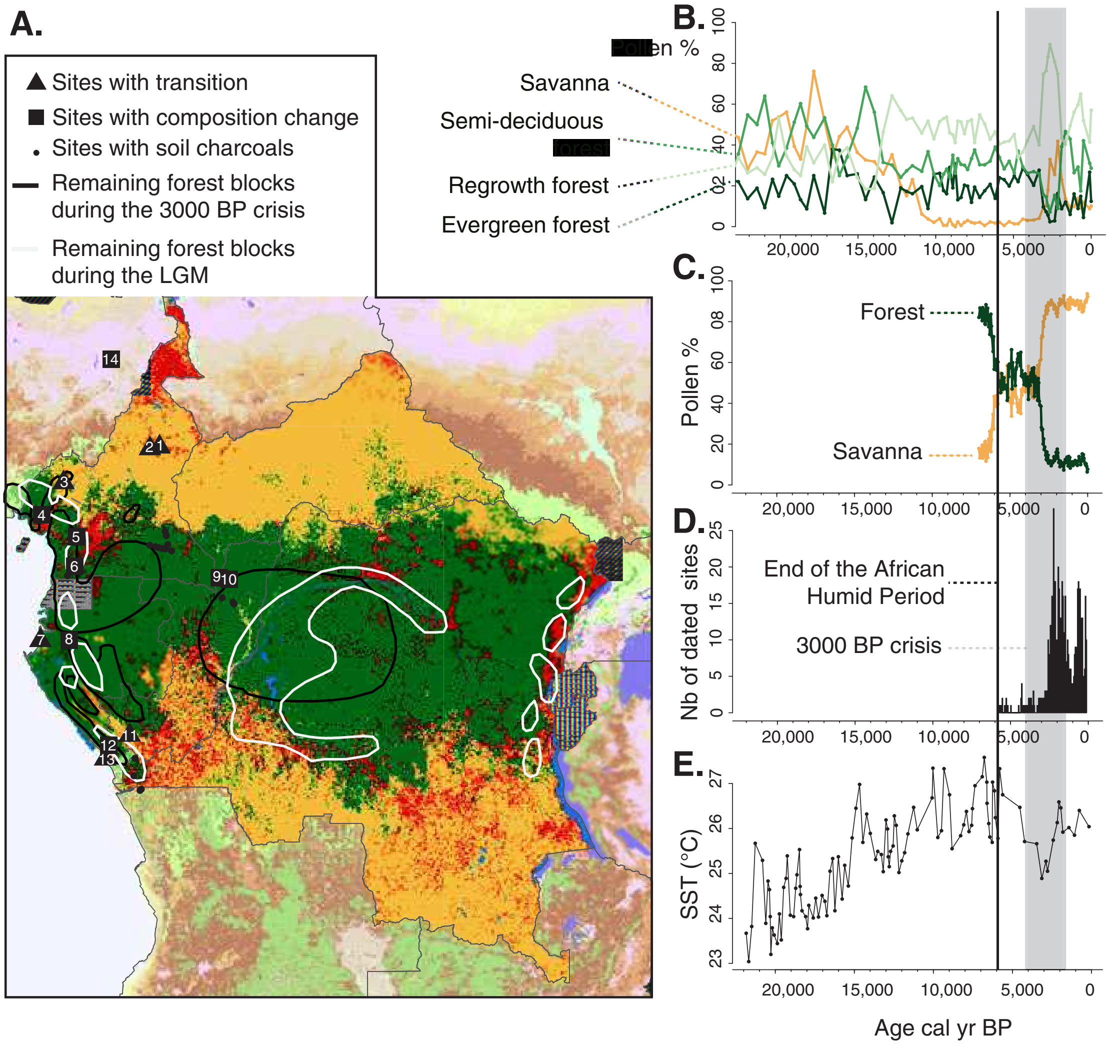

Fig. 9.1 Location of 14 pollen sites and other proxies in central Africa (**a**), pollen assemblages for Lake Barombi-Mbo (**b**, #4 in Table 9.2) and Lake Mbalang (**c**, #1 in Table 9.2), archeological sites (**d**) and sea surface temperature (**e**). Note: In (**b**)–(**e**), the black line corresponds to the end of the African Humid Period, and the beginning of the dryer conditions that led to the 3000 cal yr BP crisis (gray highlight)

9.3 Results

9.3.1 Past Vegetation Changes and Its Drivers

9.3.1.1 Trends from the Last Glacial Maximum to the African Humid Period

Figure 9.1a illustrates the hypothesized forest extent during the Last Glacial Maximum (in light gray) from Maley (1989), and during the 3000 cal yr BP crisis (in black) from Maley (2002). The vegetation map is a simplified version of the map included in Verhegghen et al. (2012) that contains forests, savannas, and human-impacted ecosystems in central Africa, using as a background the land-cover map of Mayaux et al. (2004) for Africa.

After the Last Glacial Maximum (21,000 cal yr BP), tropical forests were reduced to only few refugia in central Africa. This is suggested both through the direct paleo-data (Dechamps et al. 1988; Maley 1989) and indirect information from endemism centers (Maley 1989). After the Younger Dryas, which was a short but intensely dry period (from ~12,900 to ~11,700 cal yr BP, see Table 9.1, Sect. 9.2.1), rainfall levels started increasing at the early stages of the Holocene (Fig. 9.1e). During that period, rainfall levels were even higher than current levels, with this period called the African Humid Period (de Menocal et al. 2000; Shanahan et al. 2015). As a result, tropical forests were more extensive across central Africa, even reaching the Adamawa Plateau in Cameroon (sites 1–2, Fig. 9.1a (Vincens et al. 2010; Lebamba et al. 2016)) and the Niari Valley in the Republic of Congo (site 11, Fig. 9.1a (Vincens et al. 1998)).

Figure 9.1b–e presents information about the different proxies and determinants outlined in Sect. 9.2.2.1. Figure 9.1b, c highlights two examples of changes in pollen assemblages, (a) a site that had undergone major changes in vegetation composition during the LGM and the 3000 cal yr BP crisis (Fig. 9.1b): Lake Barombi Mbo from Giresse et al. (1994) (site 4, Fig. 9.1a) and (b) a site that had experienced a transition from forest to savanna during the 3000 cal yr BP crisis (Fig. 9.1c): Lake Mbalang from Vincens et al. (2010) (site 1, Fig. 9.1a).

Figure 9.1d presents the frequency of dated archaeological evidence in sites binned every 50 years. This is used to illustrate the changes in human impact across the Congo basin using available data since 5000 cal yr BP (Oslisly et al. 2013; Morin-Rivat et al. 2014). The period indicated in red corresponds to a period of population increase across the Congo basin.

The long-term dynamics of sea surface temperature (SST) illustrates changes in climate, and specifically rainfall (Fig. 9.1e) (Weldeab et al. 2007). It is worth noting the period spanning the 3000 cal yr BP crisis, which is a period characterized by abrupt changes in SST as indicated in the gray area (Fig. 9.1e).

9.3.1.2 Trends During the Late Holocene

The African Humid Period ended abruptly ~4000 cal yr BP (Fig. 9.1e). The subsequent period was characterized by low rainfall and major droughts that lasted until 1200 cal yr BP (Vincens et al. 1999). This period is called the “third millennium rainforest crisis” and is divided into two major phases (Maley 2002).

The first phase started shortly after 4000 cal yr BP, and is associated with an abrupt decrease in rainfall (see increase in SST in Fig. 9.1e). This trend affected the peripheral areas of the Congo basin (see sites 1–2 and the Niari Valley in Fig. 9.1a, c), and was responsible for the opening of the coastal savannas in central Africa (see site 7, Fig. 9.1a) (Elenga et al. 1992; Ngomanda et al. 2009) and the Dahomey Gap in west Africa (Salzmann and Hoelzmann 2005, not seen in Fig. 9.1a). During the same period, savanna vegetation was also heavily modified, as suggested by the pollen record of Lake Tilla (site 14, Fig. 9.1a), with a gradual (during the African Humid Period termination) and abrupt (during 3000 cal yr BP crisis) floristic shifts from Guinean to Sudano-Guinean savanna (Salzmann 2000). As illustrated in Lake Mbalang (site 1, Fig. 9.1a) (Vincens et al. 2010), the first phase of this abrupt climatic change triggered a transition from forest to savanna taxa (Fig. 9.1c) in some sites.

The second phase was rather short and abrupt, lasting between 2500 and 2000 cal yr BP. The SST reconstructions (Fig. 9.1e) and geological limestone zones (Maley et al. 2012, 2018) suggest a strong climatic seasonality. During this phase (in gray, Fig. 9.1b–e), vegetation reconstructions from pollen data show an increasing abundance of pioneer and secondary forest trees, and even in grasses within forest areas (Vincens et al. 1999) (Fig. 9.1b). This suggests that forests were highly disturbed during this period, probably exhibiting mosaics of open patches and closed forests. Some authors even suggest the opening of a north–south savanna corridor in the Sangha River Interval (Maley and Willis 2010) that could have permitted the migration of Bantu-speaking people. However, the savanna corridor has been controversial (Bremond et al. 2017). The evidence of increased human population and activity in the forest zone is dated from ~2500 cal yr BP, with the first iron-age settlements (Fig. 9.1d) (Wotzka 2006). Human presence in the region generally shows a bimodal pattern (Fig. 9.1d, red highlight), with two phases of human population expansion, and an intermediate phase of depopulation, between ~1300 and ~700 cal yr BP, but of unknown origin (Morin-Rivat et al. 2014, 2017).

9.3.2 Current Vegetation Distribution

9.3.2.1 Vegetation Types

Currently, tropical forest (in green, Fig. 9.2a) is the most important vegetation type in central Africa, covering 42% of the study area, followed by savanna (yellow, 28.4%), Miombo woodland (brown, 7%), mosaics of tropical forest and mesic savanna (light green, 5%), and other types of forested vegetation, such as swamp and swamp forest (3.0%), montane forest (1.3%), and mangrove (1%). The area affected by human activity amounts to 13.3% of the study area (red, Fig. 9.2a).

Terra firme forests occupy a vast and continuous area across the Congo basin, forming mosaics with southern savannas, such as the Nyanga river area and the Batéké Plateau. Deforested areas (in red, Fig. 9.2a) are prevalent around major cities such as Douala and Yaoundé in Cameroon, Brazzaville, and Ouesso in Congo, Kinshasa, and Kisangani in DRC, and along major road networks and rivers that connect them (Laurance et al. 2017). Deforestation is also prevalent in the densely populated areas in the mountains of eastern DRC. Large savanna areas are located in the northern and southern edge of central Africa, and are associated with high human impacts, for example, in northern Cameroon, southern Congo, and DRC (Fig. 9.2a).

There is substantial floristic variation within these vegetation types. *Terra firme* forests can broadly be divided into (a) evergreen forests under wet and aseasonal

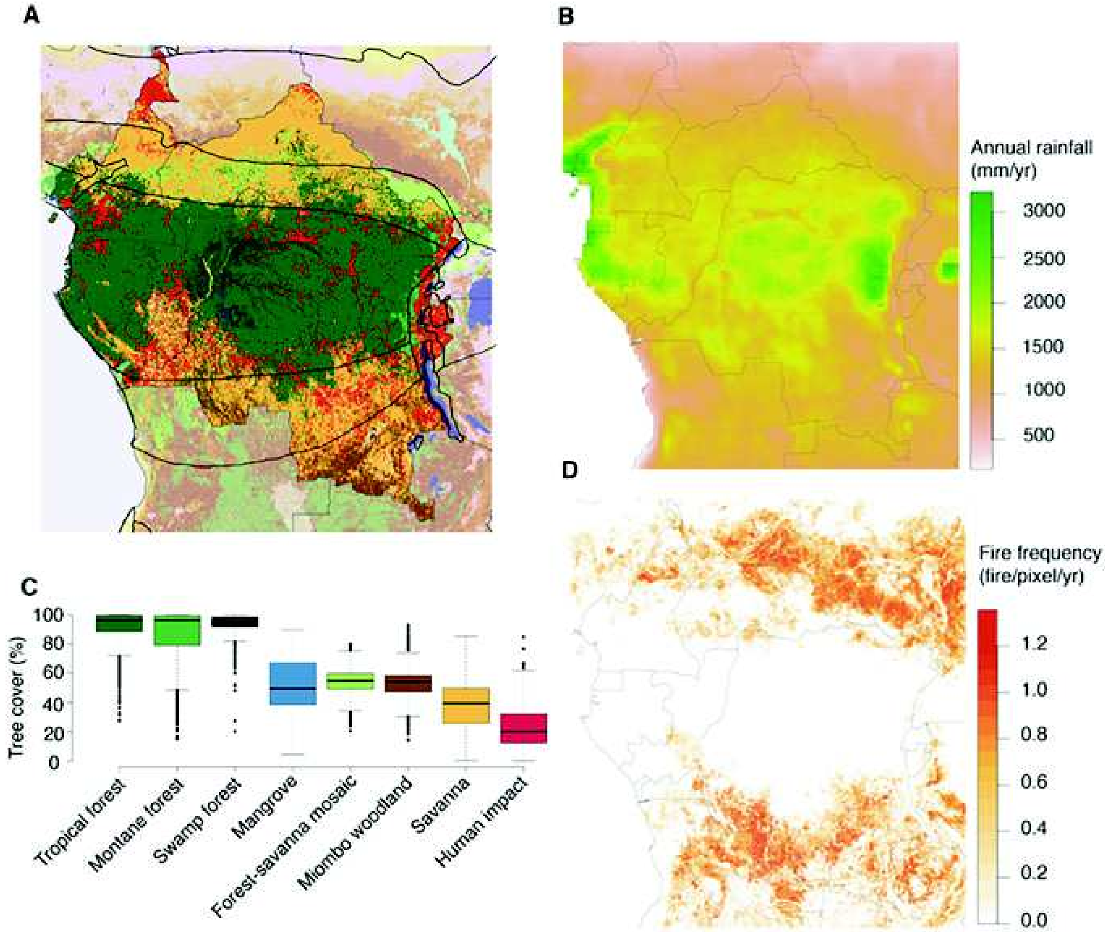

Fig. 9.2 Vegetation types (**a**), fire frequency (**b**), tree cover (**c**), and rainfall frequency (**d**) in central Africa. Note: Figure (**a**) is modified from Verhegghen et al. (2012). The background of the vegetation map (Fig. 9.1a) is based on the land-cover map of Mayaux et al. (2004) and the limits of the major phytochoria (black lines) according to White (1983). Mean annual rainfall (**b**) is expressed as millimeter per year, averaged over the period 2002–2015. Fire frequency (**d**) is expressed as the number of times that the area of each pixel is burned per year during the period 2002–2015

climates (see rainfall in Fig. 9.2b), (b) semi-deciduous forests under moist and seasonal climates (Fayolle et al. 2014a), (c) a transition type generally recognized in between the previous two types (De Namur 1990). Similarly, savannas can be divided into several types based on physiognomy, from woodlands to grass savannas (Aubréville 1957), or based on tree species composition (Fayolle et al. 2019; Osborne et al. 2018).

Tree cover can be used to distinguish the physiognomy of broad vegetation types, such as forest and savanna. We find that although highly variable, tree cover is (a) high in forested vegetation types such as tropical forests, montane forests, and swamp forests (>90%, Fig. 9.2c), (b) intermediate in mangroves, forest–savanna mosaics, and miombo woodlands (~50%, Fig. 9.2c), and (c) very low in savannas (<40%, Fig. 9.2c) and areas modified through human activity (~20%, Fig. 9.2c).

9.3.2.2 Determinants of Vegetation

The spatial distribution of mean annual rainfall and fire occurence highlights the different ecological functioning of forest and savanna in central Africa (Fig. 9.2b, d). The continuous forest block experiences only a few fire events each year (Figs. 9.2b and 9.3b), and encompasses wet (>2000 mm) and moist sites (1500–2000 mm) (Figs. 9.2d and 9.3a). In contrast, northern and southern savannas and woodlands experience frequent fires (Fig. 9.3d), and are generally encountered under drier conditions (Figs. 9.2d and 9.3c).

As expected, forests and savannas are located on the two extremes of a rainfall gradient, ranging from evergreen forests (high rainfall: >2500 mm year^{-1}) to open savannas (low rainfall: <500 mm year^{-1}). Tree cover increases with rainfall (Bucini and Hanan 2007), at least in the lower part of the rainfall gradient, interacting with the fire regime (Staver et al. 2011a). For intermediate rainfall (here, 1000–2500-mm year^{-1}) forests and savannas represent alternative stable states (Staver et al. 2011a, b), meaning that both biome types can co-occur (Fig. 9.3a–c, gray area). However, in central Africa, forests and savannas are spatially separated (Fig. 9.2a) (Aleman and Staver 2018). Forests in Congo and forest–savanna mosaics in the CAR occur under the same rainfall conditions, but one biome is dominated by trees

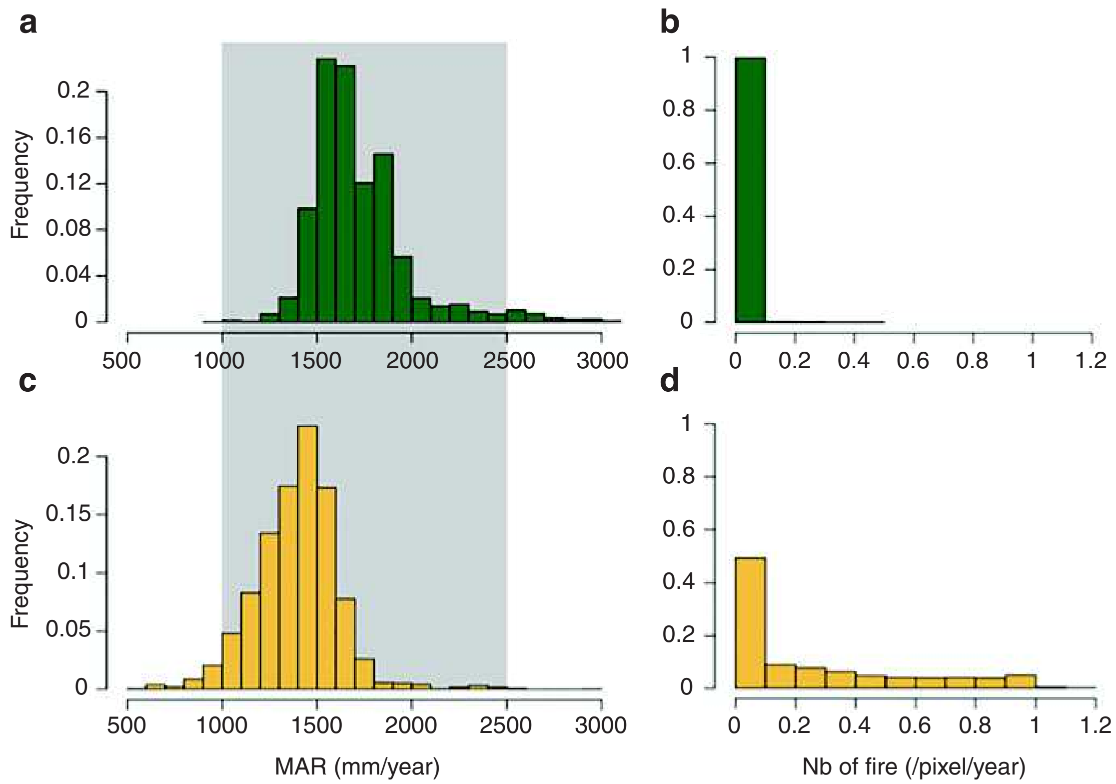

Fig. 9.3 Mean annual rainfall (in mm) (**a**, **c**) and fire occurrence between 2002 and 2015 (**b**, **d**). Note: Fire occurrence was estimated through the number of fires per pixel between 2002 and 2015. Green figures illustrate the mean annual rainfall and fire occurrence for tropical forest (**a**, **b**) and the yellow for savannas (**c**, **d**). The mean annual rainfall range for which forest and savanna co-occur is highlighted in gray (i.e., 1000–2500 mm year^{-1})

(forest), while the other is dominated by grasses (savanna). The two alternative states are maintained through a positive feedback between fire and an open canopy, with fire being common in savannas (Fig. 9.3d), while nearly absent in forests (Fig. 9.3b). This seems to be the legacy of long-term vegetation distribution, and not some kind of founder effect.

Finally, even though rainfall and fire are important determinants of the distribution of tropical forests and savannas, other drivers can also become significant, including soils (Lloyd et al. 2015), and herbivores, specifically mammal browsers (Charles-Dominique et al. 2016). However, we have not addressed these effects in this chapter.

9.3.3 Changes in Vegetation

9.3.3.1 Ongoing Vegetation Changes

The remotely sensed data indicate strong ongoing vegetation changes, and specifically deforestation and forest degradation in the forests and woody encroachment in savannas (see Mitchard et al. 2011). At the regional scale, tree cover loss is far more prevalent in forests than in savannas (Fig. 9.4a). Overall, although deforestation is relatively low in central Africa (Pan et al. 2011; Achard et al. 2014; Aleman et al. 2018), tree cover loss can be extremely important locally, especially along the main roads and the surroundings of major cities, specifically in the DRC (Fig. 9.4a). This trend is due to the increasing fuelwood demand and need for agricultural lands, as population grows (Schure et al. 2014; see also Chaps. 7 and 10, Vol. 1). Logging is also responsible for forest degradation and fragmentation, as tree cover loss is much more prevalent in logging concessions than in protected areas (Fig. 9.4b). Approximately 29% of the forests in central Africa are currently granted to logging companies, ranging, however, from 13% in DRC to more than 60% of Gabon and Congo (Fig. 9.4b, Laporte et al. 2007). Although selective logging (Ruiz Pérez et al. 2005), specifically under Forest Stewardship Council (FSC) criteria, might not be very detrimental to forest biodiversity (Putz et al. 2012), the sustainability of conventional logging practices elsewhere in the region is certainly questionable, especially due to the progression of logging roads (Laporte et al. 2007).

In this chapter, we only analyzed tree cover loss between 2000 and 2015. However, there is a longer term trend of woody encroachment in savannas areas across SSA (Stevens et al. 2017), and also more regionally in central Africa (Youta Happi 1998; Mitchard et al. 2009). Soil carbon isotopes signature analysis has demonstrated forest encroachment into savanna areas over the past century in the CAR (Runge 2002), Gabon (Delègue et al. 2001), and Cameroon (Guillet et al. 2001). However, the causes of this widespread woody encroachment in savannas are hotly debated in the academic literature (Devine et al. 2017). Some scholars have proposed that this is a response to fire suppression policies during the last century

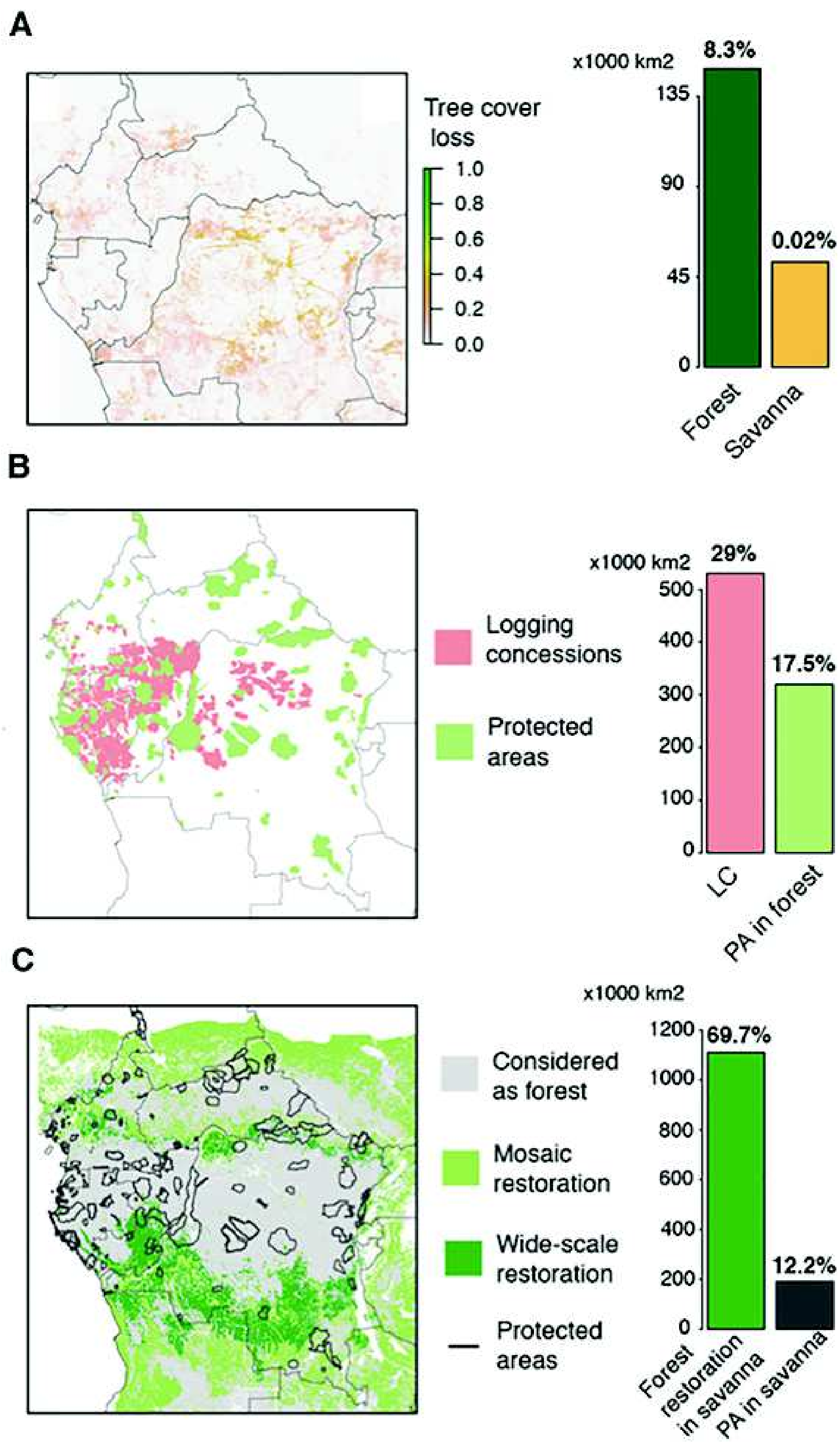

Fig. 9.4 Areas of tree cover loss between 2000 and 2015 in forests and savannas (**a**), in logging concessions and protected areas (**b**) and according to restoration potential (**c**). Note: Map in (**a**) shows tree cover loss at the pixel scale. Map in (**b**) delineates protected areas and areas granted to logging companies. Map in (**c**) highlights the location of forest restoration opportunities according to the WRI. Fractions in bar charts indicate (**a**) the percentage of pixels with tree cover loss above 10% for forest and savanna pixels, (**b**) for forest areas granted to logging concessions or falling within protected areas, and (**c**) for savanna areas protected or granted for reforestation

(Veldman et al. 2017; Bond and Zaloumis 2016), with more recent acceleration possibly being due to increasing CO_2 levels (Buitenwerf et al. 2012).

Restoration and reforestation efforts have been aiming to reverse deforestation and forest degradation in the region (Fig. 9.4c). However, such efforts are sometimes misplaced, with large savanna areas being targeted for tree plantation purposes (Fig. 9.4c). Approximately 30% of savanna areas are currently identified as forest (in gray, Fig. 9.4c), while 70% are targeted for restoration, including both savannas in mosaic and large extent of savannas. However, these areas have actually been savannas since the Last Glacial Maximum, and essentially represent a strongly stable state, considering that savannas and forests are alternative stable states in areas of intermediate rainfall (Staver et al. 2011a, b; Veldman et al. 2015a; Bond and Zaloumis 2016; Veldman 2016) (Sect. 9.1).

Protected areas are crucial for sustaining biodiversity and ecosystem processes and services, in both forest and savanna areas. They can also protect landscapes against misplaced reforestation projects and unsustainable logging practices. Currently, protected areas in central Africa cover approximately 17.5% of forests and 12.2% of savannas (Fig. 9.4b, c). This represents a relatively lower fraction of protected area coverage, compared to other parts of SSA such as East and South Africa (Aleman et al. 2016; IPBES 2018).

9.3.3.2 Predicted Vegetation Changes

We then explore the effects of future land use and climatic change in the area currently covered by forests and savannas in central Africa. Cropland expansion is expected to affect significantly savanna areas, considering that more than 40,000 and 150,000 km^2 of land will be converted for crop production by 2070 according to RCP 8.5 and RCP 2.6, respectively (Fig. 9.5a, b). The changes according to the most "optimistic" scenario in terms of emissions and radiative forcing (RCP 2.6) are projected to be more significant in savanna areas (14% converted area) compared to RCP 8.5 (2.8% converted area) (Fig. 9.5a). Forests are expected to be less affected, with only 0.5% and 0.2% of their area converted, respectively (Fig. 9.5a). This is mainly due to the fact that forests are much more protected under these scenarios than savannas (IPCC 2014). This large increase in cropland under RCP 2.6 reflects the large expansion of biofuel production (Van Vuuren et al. 2011) (see Chaps. 2–4, Vol. 1; Chap. 5, Vol. 2).

When it comes to climate change impacts, it is projected that rainfall patterns will change and the probability of drought and extreme events will increase. Rainfall

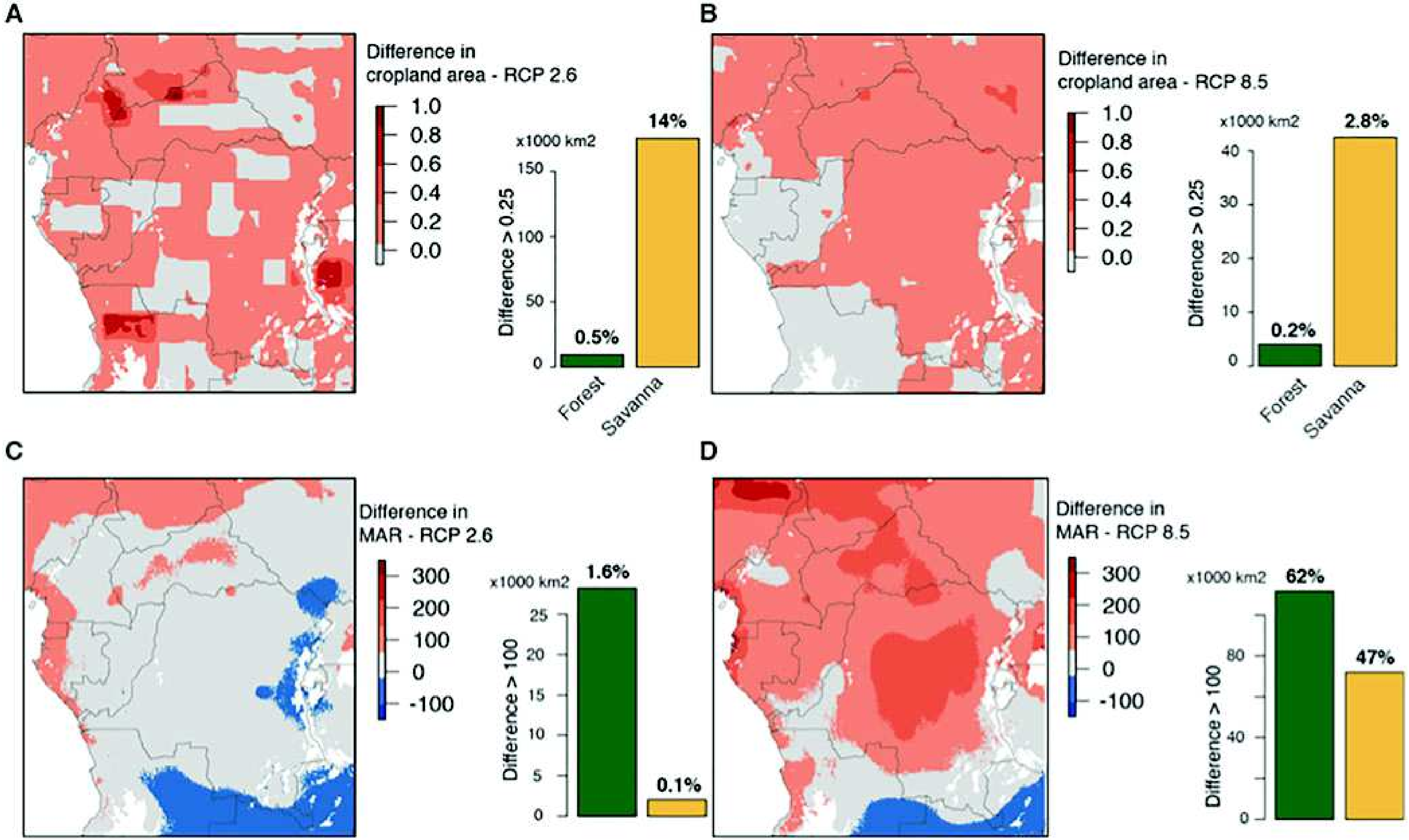

Fig. 9.5 Change in cropland area and rainfall according to RCP 2.6 (**a**, **c**) and RCP 8.5 (**b**, **d**). Note: Change in cropland area in each pixel is estimated as the difference between current and future (in 2070) cropland area in each pixel for the scenarios RCP 2.6 (**a**) and RCP 8.5 (**b**). Barplots represent the area for which this difference is expected to exceed 25% for forest and savanna areas in central Africa. Changes in rainfall (mean annual rainfall in mm year^{-1}) are estimated as the difference between current and future (in 2070) rainfall for the scenarios RCP 2.6 (**c**) and RCP 8.5 (**d**). Barplots represent the area for which this difference is expected to exceed 100 mm year^{-1} for forests and savannas in central Africa

change within RCP 2.6 is expected to be less intense and extensive, with only 1.6% of forest areas (located at the western part of central Africa) expected to be affected by rainfall change (Fig. 9.5c). On the contrary, according to RCP 8.5, as much as 62% of forests and 47% of savannas will be impacted by a decrease (or increase) in annual rainfall, with a massive drying trend over central Africa, and a localized increase in rainfall in the extreme south (Fig. 9.5d).

9.4 Discussion

9.4.1 Changes in Central African Forests and Savannas

The evidence reviewed in this study highlights that since the Last Glacial Maximum (21,000 cal yr BP), the area covered by tropical forests experienced a succession of dry and humid periods, which resulted in forest expansion and contraction (Maley 1989, 1996; Vincens et al. 1999) (Sect. 9.3.1). In contrast, forests in Amazonia remained much more stable during that same period (Anhuf et al. 2006). Savannas were more widespread and changed both in terms of extent and composition following climate fluctuations. However, for some areas, such as Lake Bambili in Cameroon (Site 4, Fig. 9.1a), there is accumulated evidence that savanna vegetation has been maintained since the LGM, indicating a very stable state at this relatively high altitude (>2000 m above sea level).

When analyzing the current extent and distribution of forest and savanna according to the most recent vegetation types available for central Africa (Verhegghen et al. 2012), we confirm that forests dominate the Congo basin, are found under moist and wet climates, and experience only few fire events (Sect. 9.3.2). In contrast, savannas occupy drier areas in the northern and southern edge of central Africa, and correspond to the Sudanian and Zambezian Savanna Regions (White 1983). These savannas occur in areas of relatively high rainfall and experience frequent fires (Lehmann et al. 2014; Osborne et al. 2018). Forests and savannas can both occur at intermediate rainfall (1000–2500 mm year^{-1}) (Sect. 9.3.2). This seems to be the outcome of feedback mechanisms between canopy closure and fire, past climatic changes, and, to a lesser extent, human occupation, as shown at the continental (Sankaran et al. 2005; Aleman and Staver 2018) and the global (Staver et al. 2011b) scales.

As the ecological functioning of forests and savannas depend on factors related to both climate and land use, we then examined such ongoing changes in areas currently covered by forests and savannas (Sect. 9.3.3). Over the past decades, woody encroachment has been reported in savannas worldwide (Stevens et al. 2017), probably due to land use change and/or CO_2 fertilization effects. More recently (i.e., between 2000 and 2015), there has been extensive tree cover loss in the forest area (Hansen et al. 2013) and forest fragmentation due to industrial logging (Laporte et al. 2007). This has been associated to hunting (Poulsen et al. 2011; Abernethy et al. 2013) (Sect. 9.3.3).

Both ongoing climate change and land use change threaten the functioning and resilience of forests and savannas in central Africa. Increasing annual rainfall will certainly favor forests (Malhi et al. 2009) and could even improve crop yields in savanna areas (Sonwa et al. 2011; IPBES 2018). Even though we have not explored how seasonality and temperature are projected to change by 2070, other studies have predicted important shifts in composition (Sala et al. 2000) and vegetation structure (Aleman et al. 2016, 2017). In addition, even though forest in central Africa are carbon sinks that sequester large volumes of carbon annually (Lewis et al. 2009), massive emissions could be expected in specific years due to drought-related tree dieback (Allen et al. 2010; Bennett et al. 2015), as reported for the Amazon (Phillips et al. 2009). This phenomenon has, however, not yet been reported in central Africa.

Section 9.3.3 outlined how savannas are specifically targeted for reforestation and the production of biofuel and cash crops, largely because they are perceived to have lower conservation value than forests (see Alexandratos and Bruinsma 2012; Searchinger et al. 2015). Currently >30% of forests in central Africa have been granted to logging companies. Even though increasing protected area coverage may help prevent the further degradation of both forests and savannas (especially savannas that only have 12.2% of their area under protection), there is a widespread concern that protected areas are not effective in the region, primarily due to conflicts and poor governance (Chape et al. 2005) (Chap. 1, Vol. 1). This potentially compromises their effectiveness in maintaining forest and savanna resilience. Indeed, protected areas in tropical areas are increasingly threatened due to population increase (Laurance et al. 2014) and especially funding constraints (Bruner et al. 2004) (Chap. 1, Vol. 1). Illegal logging, grazing, and agriculture commonly occur inside poorly enforced protected areas (Laurance et al. 2012), especially in west and central Africa (Tranquilli et al. 2014). Agricultural expansion near protected areas tends to erode biodiversity and ecosystem services due to edge effects (Wittemyer et al. 2008; Laurance et al. 2014). However, if the integrity of protected areas can be maintained, for example, by involving local communities (Vodouhê et al. 2010), then central African forests and savannas may remain relatively resilient in the face of these threats, and future climate change (Chap. 10, Vol. 1).

9.4.2 Policy Implications and Recommendations

Based on the comprehensive analysis of the past, present, and future vegetation patterns in central African forests and savannas, we identify two main policy relevant findings: (a) mesic savannas are not degraded forests and (b) forests and savannas have experienced a long history of climate changes and human activity, which offers insights for their effective management.

Considering that forests and savannas in central Africa offer habitat for biodiversity and contribute manifold to the livelihoods of local communities (see

Chap. 10, Vol. 1), these findings carry important implications for meeting the sustainable development goals (SDGs) and especially SDG 15 "Life on land." Below, we discuss some important aspects related to reforestation (Sect. 9.4.2.1), designation of protected areas (Sect. 9.4.2.2), and sustainable forest management (Sect. 9.4.2.3).

9.4.2.1 Target Degraded Areas for Ecosystem Restoration

Our findings indicate that savannas have unique ecological functioning, based on chronic disturbances (Sankaran et al. 2005). These savannas exist for millennia (Salzmann and Hoelzmann 2005), but are currently threatened by recent land use change, especially crop expansion and reforestation projects (Sect. 9.3.3). This is often done because savannas are mistaken as degraded forests (Sect. 9.1), despite the fact that they are alternative stable states to forest and are maintained through a positive feedback between vegetation structure and fire events (Sect. 9.3.2).

Due to this mistaken view of mesic savannas being degraded forests, many studies have suggested to target these savannas for forest restoration and tree planting for carbon sequestration (Laestadius et al. 2012; WRI 2014; Chazdon and Laestadius 2016). Even though many studies have suggested that the ecological restoration of degraded savanna and forest landscapes can have substantial ecological benefits (Lamb et al. 2005; Buisson et al. 2019), it is important to focus restoration efforts only to those areas that have actually been degraded, and are not mistaken as such. This is particularly important in the context of SDG15, and especially Target 15.2, where forest restoration is seen as an important intervention.

To this end, there is a need to develop integrated restoration frameworks that both consider forest and savanna ecosystems, their respective ecological functioning, and their long-term location and history. This could help avoid misrepresenting savannas as degraded forest (Veldman et al. 2015a, c), and thus implementing restoration interventions that could have counterproductive conservation outcomes. This would require more retrospective studies to accurately reconstruct the past locations of forests and savannas (Aleman et al. 2018).

In those cases where reforestation and forest restoration are deemed appropriate, then it would be crucial to favor native species (Chazdon 2008; Doucet et al. 2016). Indeed, exotic tree species such acacia and eucalypt currently dominate timber plantations in central Africa, following a systematic replication of the same experimental design (Proces et al. 2017) (Chap. 10, Vol. 1).

9.4.2.2 Expand Protected Areas

Section 9.3.3 highlighted that protected areas in central Africa cover approximately 17.5% of tropical forests and 12.2% of savannas. According to the Aichi Biodiversity Target 11, 17% of terrestrial ecosystems need to be conserved through protected areas by 2020. In this respect, it can be argued that the protection of savanna areas falls well below established targets, possibly due to the fact that savanna areas can be mistaken for degraded forests (Sect. 9.4.2.1). In view of the renewed conservation efforts through SDG Target 15.1 and the currently negotiated post-2020 global biodiversity framework, it can be argued that both savannas and forests in central Africa may require better protected areas networks.

Boosting formal conservation efforts (through national parks and otherwise) within savanna and forest ecosystems may help mitigate the negative future outcomes of land use and climate changes (Parr et al. 2014; Veldman et al. 2015a). However, it is interesting to point that savanna ecosystems may also lend themselves to alternative conservation practices. For example, protected areas in savannas are often associated with lower plant diversity compared to communally managed areas (Dahlberg 2000; Nacoulma et al. 2011). This is because savannas are intrinsically associated with disturbances such as fires (Veldman et al. 2015c) (Sect. 9.3.2). In this sense, traditional land management might not degrade savanna habitats (Nacoulma et al. 2011) (see Chap. 10, Vol. 1; Chap. 3, Vol. 2).

In such contexts, there might be a need to reevaluate land use change planning practices to explicitly include the conservation and agricultural value of savanna ecosystems (Veldman et al. 2015c). Some scholars have even argued that effective biodiversity conservation will rely on associating traditional and communally managed areas, with reserves (Abel and Blaikie 1989; Nacoulma et al. 2011). Such approaches could potentially create important synergies between SDG15 (especially as it pertains to the sustainable use of biodiversity), SDG13 (Climate action), and SDGs directly related to human livelihoods such as SDG2 (zero hunger) and SDG1 (no poverty).

There would also be a need to develop appropriate management frameworks for grass-dominated biomes (Veldman et al. 2015a, c), and to better estimate the environmental value of such biomes. Historically, the vast majority of conservation efforts have focused on forests (Laurance et al. 2014; Tranquilli et al. 2014), but savannas are at equal, if not greater, risk of change due to land use change (Aleman et al. 2016). Mesic savannas and transition zones between forests and savannas in west and central Africa may be particularly susceptible, not the least due to the fact that they contain few protected areas (Sect. 9.3.3). Sustainable management plans for such ecosystems should include strong monitoring components that serve both biodiversity conservation and human development needs.

9.4.2.3 Promote Sustainable Forest Management Approaches

Finally, some of the new directions in biodiversity conservation and ecosystem management suggest to take into account the long-term history of ecosystems (Gillson 2015; Gillson and Marchant 2014; Barnosky et al. 2017). The peculiar long-term history of forests and savannas in central Africa, especially in relation to climate change and human activity (e.g., migration and activity of Bantu people), could inform the development of more sustainable logging practices.

Logging concessions in central Africa constitute a major land use, accounting for more than 60% of forest area in some countries, while at the same time being an engine of economic growth (Bayol et al. 2012; De Wasseige et al. 2012) (Chap. 1, Vol. 1). Even though current logging practices are not entirely sustainable, considering that the initial stock of timber accumulating for centuries cannot be recovered in 25–30 years of a timber rotation, some good logging practices such as selective logging following Forest Stewardship Council standards could limit negative biodiversity outcomes (Putz et al. 2012) and ensure at least 50% recovery of the timber resource for the targeted species (Bayol et al. 2012). With logging roads closing within only a few years (Kleinschroth et al. 2015) and biomass recovering within approximately 20 years following logging (Gourlet-Fleury et al. 2013), it is possible that production forests could provide multiple ecosystem services, at levels even comparable to "intact" or old-growth forests.

Nevertheless, logging can be a major driver of forest fragmentation, having negative biodiversity outcomes through habitat fragmentation and forest accessibility (Poulsen et al. 2013; Ziegler et al. 2016). Thus enhancing conservation actions and preventing poaching/hunting are key priorities in logging concessions. In addition, even though timber recovery rates are supposed to reach 50%, regeneration deficits have been observed for many timber species (Engone Obiang et al. 2014; Morin-Rivat et al. 2017). Thus prior to logging management inventories should be used to identify target species and how the regeneration is organized. Indeed, dissimilarities between canopy species and regeneration have been observed in different forest types (e.g., Swaine and Hall 1988 in Ghana along a rainfall gradient), which can be explained by forest successional dynamics. The conditions that promoted the regeneration of canopy species may no longer be encountered. As a result (and in order to be logged in the future) the species targeted for timber in central Africa will need to be favored, and specific silvicultural interventions will be required. Among them, forest enrichment (Doucet et al. 2009) and plantations (Doucet et al. 2016) have already provided good results at a relatively small scale.

The above suggest that designing nuanced sustainable management approaches based on the long-term understanding of forest and savanna dynamics could contribute substantially in meeting SDG Target 15.2 (sustainable forest management) and 15.7 (end poaching). More importantly such interventions can have important synergies with some of the targets related to SDG12 (responsible consumption and production) and SDG8 (decent work and economic growth).

9.5 Conclusions

In this chapter, we analyzed the distribution of the two main biomes in central Africa, tropical forests and savannas, as well as the underlying factors affecting their distribution. Interestingly, in intermediate rainfall regimes, forest and savanna represent alternative stable states maintained by a positive feedback between vegetation structure and fire events. Forest-savanna bistability spans a huge portion of central

Africa, with 36% of African savannas currently occurring where rainfall is sufficient for forest canopy closure. Consequently, these mesic savannas occurring in areas of annual rainfall between 1000 and 2500 mm have often been mistakenly considered as degraded forests.

However, forests and savannas in central Africa have experienced a long history of climate events and human activity that have both triggered important contractions and expansions in the Congo Basin, as well as modifications in forest floristic composition. During the past century, new threats have emerged for these biomes, as we have entered the Anthropocene. Apart from climate change, the alarming rates of land use change related to cropland expansion, logging concessions, and misplaced forest plantations may represent an even more substantial threat to forest and savanna structure, composition, and ultimately ecosystem services provision.

As a conclusion, we urge academics and practitioners to develop integrated management frameworks for both forest and savanna ecosystems, to avoid further ecosystem degradation. Furthermore, we point the need to restore the importance of savannas in biodiversity conservation efforts. We advice taking into account knowledge from the long-term history of these ecosystems to build more sustainable management plans that can guide ecosystem restoration, protected area expansion, and the design of sustainable forest management approaches in the region, and beyond.

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