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Multi-fold increase in rainforests tipping risk beyond 1.5-2^{o}C warming

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Abstract. Tropical rainforests invest in their root systems to store moisture in their root zone from water-rich periods for use in water-scarce periods. An inadequate root-zone soil moisture storage predisposes or forces these forest ecosystems to transition to a savanna-like state, devoid of their native structure and functions. Yet changes in soil moisture storage and its influence on the rainforest ecosystems under future climate change remain uncertain. Using the (mass-balance-based) empirical understanding of root zone storage capacity, we assess the future state of the rainforests and the forest-to-savanna transition risk in South America and Africa under four different shared socioeconomic pathway scenarios. We find that under the end-of-the-21st-century climate, nearly one-third of the total forest area will be influenced by climate change. Furthermore, beyond 1.5-2°C warming, ecosystem recovery reduces gradually, whereas the forest-savanna transition risk increases several folds. For Amazon, this risk can grow by about 1.5-6 times compared to its immediate lower warming scenario, whereas for Congo, this risk growth is not substantial (0.7-1.65 times). The insight from this study underscores the urgent need to limit global surface temperatures below the Paris Agreement.

1. Introduction

Tropical rainforests in the Amazon and Congo basins are critical to the Earth system since they store and sequester a large amount of carbon, host vast biodiversity, and regulate the global water cycle (Malhi et al., 2014). However, these forests are under severe pressure from climate change and land-use change (Davidson et al., 2012; Lewis et al., 2015; Malhi et al., 2008), which risk amplifying further warming and forest degradation (Lawrence et al., 2022). Climate change and land-use change lead to a decrease in precipitation, an increase in seasonality and atmospheric water demand (Malhi et al., 2014). This causes a deficit in soil moisture availability that inhibits plant growth (Singh et al., 2020; Wang-Erlandsson et al., 2022). Furthermore, climate-induced hydroclimatic changes, including the projected increases in drought frequency, severity, and duration (Dai, 2011; Liu et al., 2018), present imminent threats to the capacity of rainforests to maintain their native ecological structure and functions (i.e., resilience) (Bauman et al., 2022; Grimm et al., 2013; Jones et al., 2009).

Under water-deficit, rainforests adapt by investing in their root systems to increase their capacity to access soil moisture (Singh et al., 2020, 2022a). At the same time, the availability of surplus moisture at shallow depths minimises the need for ecosystems to invest in extensive (deeper and lateral) root systems (Bruno et al., 2006). Since the rooting structure is challenging to measure at the ecosystem scale (Fan et al., 2017), previous studies have found that empirical mass balance-derived root zone storage capacity (S_r) correlates well with ecosystems' capacity to store water in its roots and their above-ground transition dynamics (de Boer-Euser et al., 2016; Singh et al., 2020; Stocker et al., 2023; Wang-Erlandsson et al., 2016) – thus serves as a proxy for ecosystems ability to store, utilize and adapt based on available subsoil moisture. Here, S_r constitutes a hydrological buffer required by the ecosystem for the collection of surplus precipitation from wet periods to be stored and used for evaporation throughout the dry periods (when total evaporation is greater than precipitation) (Grossiord et al., 2020; Singh et al., 2020; Wang-Erlandsson et al., 2016). Therefore, a lowly water-stressed (defined based on the magnitude of deficit in soil moisture availability inhibiting plant growth) ecosystem will need the least investment to access stored moisture.

In contrast, a highly water-stressed ecosystem will require extensive subsoil investment (Singh et al., 2020). However, S_r investment is costly, and there exists a ceiling up to where ecosystems cannot maximise their S_r any further (Singh et al., 2020, 2022a). Approaching this ceiling also implies that forest ecosystems are depleting their adaptive capacity towards further future hydroclimatic changes (Fan et al., 2017; Guswa, 2008; Kleidon and Heimann, 1998; Singh et al., 2020), with forests that have extended their S_r close to their maximum storage limit being most vulnerable to increases in water-stress (Singh et al., 2022a). Excessive short-term water deficits in these forests lead to tree mortality, loss of carbon sink strength, and an increase in the risk of fire (Aleixo et al., 2019; Bauman et al., 2022; van Nes et al., 2018; Singh et al., 2022b), whereas long-term water deficits can lead to large-scale tipping to a savanna-like state (Hirota et al., 2011; Staal et al., 2020; Staver et al., 2011; Zemp et al., 2017).

However, such ecohydrological dynamics remain challenging to incorporate in the Earth System Models (ESMs) (Lenton, 2011; Maslin and Austin, 2012; Valdes, 2011) – complex mathematical representations of Earth system processes and interactions across different biospheres. This limits ESM's capacity to simulate tipping points as an emergent property of the system (i.e., properties that emerge due to multiple interactions between several system components, and is not the property of an individual component) (Hirota et al., 2021; Reyer et al., 2015). This constraint is mainly due to our poor understanding of complex mechanisms governing the ecosystem, which are not well represented in ESMs. This includes a limited understanding of vegetation-climate feedbacks (Boulton et al., 2013, 2017; Chai et al., 2021), subsoil moisture availability (Cheng et al., 2017), adaptation dynamics (Yuan et al., 2022), the response time of forest ecosystems to climate change perturbations, and assumptions about future (i.e., prescribed) land-use change (Hurtt et al., 2020) in the ESMs. Furthermore, in the Earth system, some interactions still remain largely unknown, thereby making the prediction of (abrupt) forest-to-savanna transition (referring to changes in the dense-canopy structure of forests to one that mimics an open-canopy structure similar to savanna; hereafter referred to as forest-savanna transition) challenging (Drijfhout et al., 2015; Hall et al., 2019; Koch et al., 2021).

In addition, the risk of forest-savanna transitions under various possible climate future scenarios is relatively under-investigated. As a result of the conflicting findings and scenario-dependent uncertainties, the Intergovernmental Panel on Climate Change (IPCC) has only low confidence about the possible tipping of the Amazon forest by the end of the 21st century (Canadell et al., 2021). However, with mounting empirical evidence on how climate change influences rainforest ecosystems (Boulton et al., 2022; Küçük et al., 2022; Singh et al., 2020, 2022a), the research on rainforest resilience loss has accelerated considerably in the recent decade (Ahlström et al., 2017; Huntingford et al., 2013). Yet, forest resilience is often assessed based on changes in forest carbon stocks (Huntingford et al., 2013; Parry et al., 2022) or precipitation (Hirota et al., 2011; Staal et al., 2020; Zemp et al., 2017); and rarely on actual moisture storage capacity in the root zone (Singh et al., 2022a). Further, there is a need to assess and contrast the forest resilience consequences of low-emission and current commitment trajectories with the more commonly used high-emission scenario (Jehn et al., 2022).

This study aims to assess the rainforest resilience and risk of a forest-savanna transition by the end of the 21st century based on an empirical understanding of ecosystems' root zone storage dynamics. For this, we use hydroclimate-derived root zone storage capacity (S_r) – representing the maximum amount of soil moisture vegetation can access for transpiration (Gao et al., 2014; Singh et al., 2020; Wang-Erlandsson et al., 2016) – to classify the ecosystems under current and future climates, and assess potential forest transitions. It should be noted that we refer to rainforest tipping as forest-savanna transition 'risk' since the timing of such transitions depends on the stochastic fluctuations of other environmental factors, beyond just hydroclimate (e.g., fire, human influence, species composition) (Cole et al., 2014; Cooper et al., 2020; Higgins and Scheiter, 2012; Poorter et al., 2016). Therefore, to project if an ecosystem is a forest or has tipped to savanna in the future, we

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assume the hydroclimate projected by the end of the 21st century (i.e., 2086-2100) and ecosystem are in equilibrium. However, we do not account for the time required for ecosystems to reach their (long-term) equilibrium state.

2. Methodology

2.1 Study Area

This study focuses on forest ecosystems (i.e., excluding savanna/grassland and vegetation in human-influenced ecosystems) extending between 15°N–35°S for South America and Africa.

2.2 Data

This analysis uses both empirical and ESM-simulated datasets of precipitation and evaporation. Empirical datasets include remotely sensed and observation-corrected precipitation and evaporation time-series. Empirical precipitation estimates at daily timestep are obtained from the Climate Hazards Group InfraRed Precipitation with Station data (CHIRPS; 0.25° resolution) (Funk et al., 2015). Furthermore, empirical evaporation is derived using an equally-weighted ensemble of three different datasets – (i) Breathing Earth System Simulator (BESS; 0.5° resolution) (Jiang and Ryu, 2016) (ii) Penman-Monteith-Leuning (PML; 0.5° resolution) (Zhang et al., 2016) and (iii) FLUXCOM-RS (0.083° resolution) (Jung et al., 2019) – at monthly timestep. Here, evaporation represents the sum of all evaporated moisture from the soil, open water and vegetation, including interception and transpiration. We only selected evaporation datasets free from biome-dependent parameterisation (such as plant function types, stomatal conductance, and maximum root allocation depth) and soil layer depth (represents maximum depth of moisture uptake). Ultimately, all evaporation datasets are bilinearly interpolated to 0.25° resolution and downscaled to daily timestep using ERA5 evaporation (0.25° resolution) estimates (Hersbach et al., 2020). All empirical datasets are obtained for 2001-2012.

We also obtained precipitation and evaporation estimates from 33 ESMs (from 22 different institutes) which included CMIP6-historical and four SSP scenario simulations (SSP1-2.6 leads to approx. 1.3-2.4°C warming; SSP2-4.5 corresponds to 2.1-3.5°C warming and is closest to the current trajectory according to the nationally determined contributions (Anon, 2015); SSP3-7.0 around 2.8-4.6°C warming; and SSP5-8.5 represents 3.3-5.7°C warming; °C warming represents an increase in mean global surface temperature change by the end of 21st century relative to 1850-1900 (IPCC, 2021) (Fig. 1; Supplementary Table 1 and 2). The historical estimates are obtained at a monthly timestep for 2000-2014, and the estimates under different SSPs are obtained for 2086-2100. Though obtained estimates from different ESMs are at different spatial resolutions, we bilinearly interpolated them to 0.25° for this analysis.

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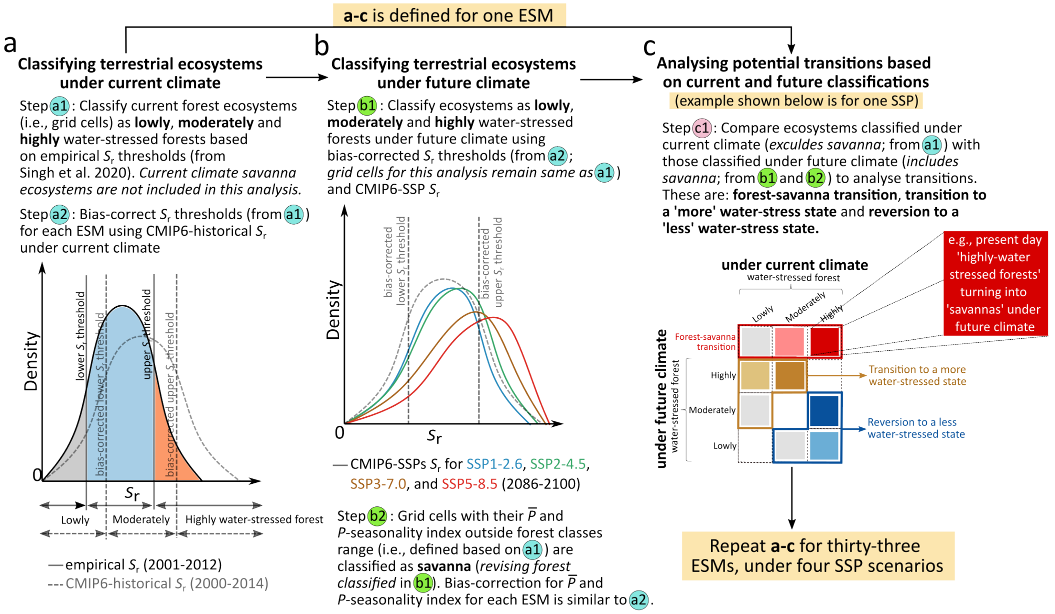
Finally, to minimize the influence of human activity and non-forest land cover on the natural water cycle, we utilized land-cover data to remove pixels with such features from our analysis. We began by removing human-influenced and non-forest land cover, such as savanna, grasslands, and water bodies, from Globcover, a global land-cover classification dataset by the European Space Agency (ESA) at 300m resolution (GlobCover land-use map, 2022). We then performed majority interpolation to convert the dataset to a 0.25° resolution and to mask grid cells with less than 50% forest cover. This step ensured that only grid cells with over 50% forest cover were classified as forests for further analysis.



Step a1 : Classify current forest ecosystems (i.e., grid cells) as **lowly**, **moderately** and **highly** water-stressed forests based on empirical S_r thresholds (from Singh et al. 2020). *Current climate savanna ecosystems are not included in this analysis.*

Step a2 : Bias-correct S_r thresholds (from a1) for each ESM using CMIP6-historical S_r under current climate

Step b1 : Classify ecosystems as **lowly**, **moderately** and **highly** water-stressed forests under future climate using bias-corrected S_r thresholds (from a2 ; *grid cells for this analysis remain same as* a1) and CMIP6-SSP S_r

Step b2 : Grid cells with their $\overline{P}$ and *P*-seasonality index outside forest classes range (i.e., defined based on a1) are classified as **savanna** (*revising forest classified* in b1). Bias-correction for $\overline{P}$ and *P*-seasonality index for each ESM is similar to a2 .

Step c1 : Compare ecosystems classified under current climate (*exculdes savanna*; from a1) with those classified under future climate (*includes savanna*; from b1 and b2) to analyse transitions. These are: **forest-savanna transition**, **transition to a 'more' water-stress state** and **reversion to a 'less' water-stress state.**

Figure 1 | Methodological framework for analysing the potential transitions in tropical terrestrial ecosystems using empirical and CMIP6-Earth System Models (ESMs) hydroclimate estimates. (a) We use root zone storage capacity (S_r)-based classification thresholds (obtained from Singh et al., 2020) – calculated using empirical precipitation (*P*) and evaporation (*E*) estimates (Supplementary Fig. 1; see methods) – to classify terrestrial ecosystems under the current climate. Savanna ecosystems under the current climate are excluded from this analysis. We bias-correct these S_r thresholds for all ESMs using the histogram equivalence method(Piani et al., 2010) (Supplementary Table 1). **(b)** We then use these bias-corrected S_r thresholds to classify ecosystems under future climate conditions (Supplementary Figs. 2-3). Furthermore, we use mean annual precipitation ($\overline{P}$) and *P*-seasonality index range (S_r-based forest classes from a) – as a proxy for ecosystem state – to revise our classification under future climate (Supplementary Fig. 4). **(c)** We then analyse the potential transitions by comparing ecosystems classified under the current climate (analysed in a) with those classified under future climate (analysed in b) individually for all ESMs (Supplementary Figs. 5-6 and Supplementary Data). The transition analysis is based on the assumption that the vegetation and hydroclimate are in equilibrium, and does not account for the time required for transitions to occur. A detailed description is provided in the 'Methods' section. An exemplification of this methodological framework is shown in Supplementary Fig. 7.

2.3 Root zone storage capacity-based framework for projecting forest transitions

Previous studies have shown that forest ecosystems adapt to water deficit by investing in roots to store and access subsoil water (Brum et al., 2019; Fan et al., 2017; Nepstad et al., 1994), efficiently distribute water through their roots for transpiration during dry-periods (e.g., hydraulic redistribution) (Liu et al., 2020; Oliveira et al., 2005), maximise water-use efficiency (by regulating stomatal conductance) or minimise moisture loss (by shredding leaves (Wolfe et al., 2016)) to reduce root zone moisture storage (Barros et al., 2019; Brum et al., 2019; Lammertsma et al., 2011). Since vegetation uptakes moisture from its roots, root zone available moisture is a key element that mediates the interaction between vegetation and climate (Brooks et al., 2015; Küçük et al., 2022; Rosas et al., 2019; Wang-Erlandsson et al., 2022). However, these root-relevant characteristics (e.g., vertical and lateral extent, and soil moisture uptake profile) are challenging to measure at ecosystem scales (Bruno et al., 2006). Furthermore, models tend to oversimplify the transfer and storage of water in roots due to insufficient knowledge about soil-vegetation-climate interactions (Albasha et al., 2015; Hildebrandt et al., 2016; Wang et al., 2004). In such cases, the mass-balance approach-based S_r provides a tangible and comprehensive understanding of ecosystem subsoil dynamics (de Boer-Euser et al., 2016; Gao et al., 2014; McCormick et al., 2021; Stocker et al., 2023).

2.3.1 Root zone storage capacity (S_r)

Derived using the mass-balance approach, S_r represents the maximum amount of moisture accessed by vegetation for transpiration (Singh et al., 2020; Wang-Erlandsson et al., 2016). This moisture is stored within reach of roots, beyond which the gravity-induced gradients control subsoil moisture (de Boer-Euser et al., 2016). This methodology assumes that ecosystems do not invest in expanding their storage more than necessary to bridge the water-deficit experienced by the vegetation in dry periods (i.e., periods in which evaporation is greater than rainfall, irrespective of the seasons). Since investment requires carbon allocation, extensive investment (i.e., more than necessary) will be inefficient from the perspective of the plants (Gao et al., 2014; Schenk, 2008). This mass-balance approach only requires precipitation and evaporation estimates to determine S_r. It is, thus, independent of any prior vegetation, soil or land cover-based information (Wang-Erlandsson et al., 2016). Furthermore, using empirical (observation-based) datasets, we capture the actual state of the ecosystems – reflecting the actual soil moisture availability for the ecosystems (Singh et al., 2020).

Previous empirical studies on mass-balance derived S_r have provided a comprehensive understanding of catchment characteristics and the ecosystem's subsoil dynamics (de Boer-Euser et al., 2016; Gao et al., 2014). However, a recent study by Singh et al. (2020) found them to dynamically represent the ecosystem's above-ground structure and below-ground root zone adaptation dynamics. In their study, Singh et al. (2020) classify drought coping strategies of forest and ecosystems under changes to their hydroclimate – which are not apparent from just precipitation and tree cover data, but also highlight (and validate) thresholds beyond which

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forest-savanna transitions occur. These adaptation dynamics, thus, align with the alternative stable state theory (i.e., forest's stabilising feedback under hydroclimatic changes and tipping risk beyond certain hydroclimatic extremes) (Hirota et al., 2011), which makes S_r more representative of the transient state of the ecosystem than precipitation (Singh et al., 2022a). Due to their strong influence over hydroclimate and ecological systems (Tumber-Dávila et al., 2022; Wang-Erlandsson et al., 2022), we use mass-balance derived S_r to project rainforest tipping risk under future climate change. A detailed description of how previous studies have projected rainforest tipping (Supplementary Table 3), and how S_r-based framework builds upon their shortcomings is mentioned in Supplementary description.

2.3.2 Projecting forest transitions under future climate change

Some previous studies have directly used the ensemble of hydroclimatic estimates to analyse tipping (Staal et al., 2020; Salazar et al., 2007). However, ESMs simulate Earth system processes based on their unique parametrisations and biases. Since an ensemble aggregates hydroclimatic estimates from different ESMs, it generalises the understanding of Earth system processes between different ESMs (Baker et al., 2021; McFarlane, 2011; Yuan et al., 2022). Therefore, we qualitatively assessed (i.e., by classifying) the ecosystem's water-stressed state using S_r (Singh et al., 2020) for each ESM individually. For this, we follow three steps: (i) calculating S_r under the current climate and classifying terrestrial ecosystems based on S_r thresholds, (ii) calculating S_r under future climate and using S_r thresholds to classify terrestrial ecosystems in the future, and (iii) analysing potential transition for each ESM using current and future ecosystem classification (Fig. 1).

First, using a mass-balance approach, we determine the maximum annual accumulated water deficit from (empirical) daily estimates of CHIRPS precipitation and ensemble evaporation over several years (2001-2012, in this case) (see Supplementary Methods). We then use a 20-year drought return period based on Gumbel extreme value distribution to simulate S_r under the current climate. Thus, S_r refers to the maximum deficit expected to occur every twenty years under static climate conditions. We acknowledge that grasslands and savanna adapt to shorter drought return periods (i.e., <10 years and 10-20 years, respectively). In contrast, forests adapt to long drought return periods (>40 years) (Wang-Erlandsson et al., 2016). However, to avoid artificially introduced S_r transitions between landscapes, we chose a uniform 20-year drought return period (following Bouaziz et al., 2020; Nijzink et al., 2016), rather than assigning different drought return periods to different land cover types (i.e., forest, savanna and grassland) (Singh et al., 2020).

In this study, we use Singh et al. (2020) S_r-based classification scheme, which classifies terrestrial ecosystems as lowly water-stressed, moderately water-stressed, highly water-stressed forests and savanna-grassland regime. This classification is based on empirically observed patterns in the ecosystem's above- and below-ground structure (i.e., the statistical relationship between tree cover and S_r), hydrology, and hydroclimate; and validated against empirical rooting depth (Fan et al., 2017) and ecoregion datasets(Dinerstein et al., 2017). Here, lowly, moderately and highly correspond to the state of the forest under different levels of water stress (i.e., quantifying the magnitude and duration of water-deficit experienced by vegetation which can inhibit plant growth. We directly used the S_r-based thresholds estimated by Singh et al. (2020) for ecosystem classification under current climate conditions (detailed description in Table 1 of Singh et al., 2020) (Fig. 1a). It should be noted that the savanna-grassland ecosystems under the current climate are excluded from this analysis. For South America, these empirical S_r thresholds are ≤100 mm (for lowly; also referred to as 'lower S_r threshold'), 100-400 mm (for moderately) and >400 mm (for highly water-stressed forests; also referred to as 'upper S_r threshold') (Fig. 1a). For Africa, these S_r thresholds are ≤100 mm (for lowly), 100-350 mm (for moderately) and >350 mm (for highly water-stressed forests). Higher S_r implies a need for larger storage to buffer water deficit, which previous studies found corresponds to plants expanding their roots vertically and laterally to maximise storage (Singh et al., 2020). The difference in S_r thresholds between both continents is due to the presence of water-use efficient C4 grasses in Africa, which reduces the competitiveness for moisture uptake between tree species and grasses – leading to a lesser need for root zone storage in the ecosystem(Singh et al., 2020).

Second, for determining future S_r, we follow the same mass-balance approach as empirical (see Supplementary Methods). However, as mentioned previously, different ESMs are based on different research groups' understandings of Earth system processes and are therefore parametrised differently(McFarlane, 2011; Baker et al., 2021). Therefore, it does not make sense to directly use the empirical (2001-2012) S_r thresholds obtained under current climate conditions to classify S_r from future CMIP6-ESMs simulations. Furthermore, since daily estimates of precipitation and evaporation are not publicly available for all CMIP6-ESM simulations, it would not be logical to directly compare them with monthly precipitation and evaporation derived S_r (Supplementary Method and Supplementary Fig. 8). To resolve this, we used the histogram equivalence method (Piani et al., 2010) (i.e., a bias-correction method). Here, the percentile-equivalent for the empirical S_r thresholds is calculated individually for thirty-three CMIP6-ESMs under the current climate (CMIP6-historical between 2000-2014). For example, if a S_r of 100 mm corresponds to the 10th percentile in the empirically-derived S_r sample (n = 20% of total pixels), the 10th percentile equivalent in CMIP6-historically-derived S_r is considered its equivalent, but this can be higher or lower than 100 mm (Fig. 1 and Supplementary Table 1).

Percentile-equivalent S_r thresholds are calculated for all ESMs individually under current climate conditions (i.e., using CMIP6-historical estimates between 2000-2014) (Fig. 1a). These histograms analysed thresholds are referred to as percentile-equivalent lower and upper S_r thresholds. To classify ecosystems under future climate (2086-2100), we directly overlay the CMIP6-historical (2000-2014) percentile equivalent lower and upper S_r thresholds to CMIP6-SSP derived S_r (2086-2100) (Fig. 1b and Supplementary Table 1). For our analysis, we chose the same period of 15 years for both CMIP6-historical (2000-2014) and CMIP6-SSP (2086-2100) hydroclimate, representing the epoch of current and future climate. Furthermore, the time-period for

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CMIP6-historical (2000-2014) is chosen such that it will be close to the simulation period of empirical S_r (i.e., 2001-2012).

Under future climate change, some ecosystems will remain forest, while others will transition to savanna. However, despite the analysis above capturing the change in S_r, it is difficult to discern whether an ecosystem is a forest or savanna just with S_r (the above analysis only classifies ecosystems as forests). Savanna and grassland ecosystems experience considerably less precipitation than forests and can show high precipitation seasonality. However, because they also have lower evaporation than forests, their accumulated deficit and, therefore, S_r is also less(Singh et al., 2020). These low S_r patterns might conceal savannas as a lowly or moderately water-stressed forests (since these forests also have low water-deficit and therefore lower S_r) (Supplementary Fig. 4a). Segregating them is easier under the current climate, where we have several remote sensing products capturing vegetation structure (e.g., tree cover, tree height, floristic patterns)(Hirota et al., 2011; Aleman et al., 2020; Xu et al., 2016). However, under future climate, we must find a proxy. Previous studies have either depended on some vegetation structure metric derived from the models themselves (e.g., the fraction of forest cover or net primary productivity)(Jones et al., 2009; Boulton et al., 2013), or had assumed that no other vegetation would replace forests. Therefore, the latter assumption assesses the possible extent of forest ecosystems under future hydroclimatic changes(Staal et al., 2020). Since ESMs use IAM-derived land-use scenarios solely based on macro-socioeconomic factors(Ma et al., 2020), and not biophysical processes, the ecosystems are not in equilibrium with their climate. We went with the latter assumption of assessing the theoretically possible extent of forest ecosystems under hydroclimatic changes (i.e., the ecosystem is in equilibrium with their climate).

We did this using mean annual precipitation and the precipitation seasonality index range obtained under the current climate (2001-2012) for forest classes (i.e., lowly, moderately, and highly water-stressed forest) and savanna-grassland regime (Supplementary Fig. 4). The ecosystems (i.e., grid cells) falling outside the range of bias-corrected mean annual precipitation and precipitation seasonality index of forest classes are classified as 'savanna' (this includes grassland). This way, not only do we segregate forests and savanna ecosystems under future climate conditions (revising the forest classification analysis under future climate), but using S_r, we also include the magnitude and duration of water-deficit, which previous studies suggest is important(Staal et al., 2020; Zemp et al., 2017), but were unable to operationalise in projecting rainforest tipping.

As an additional check, since we do not question whether the forest regions classified as savanna in the future are, in fact, forests under the current climate, or more specifically, have the hydroclimate characteristics of forest under the current climate (since we are not using prescribed land-use scenarios as proxy); we again use the bias-corrected range of mean annual precipitation and the precipitation seasonality index (Supplementary Fig. 4). With this check, if a future savanna ecosystem (i.e., grid cell) has its mean annual precipitation and precipitation seasonality index falling outside the range of forest classes under the current climate conditions, we exclude it from further analyses (the black region in Supplementary Fig. 7). This way, we avoid over-estimating tipping risk under future climate change.

Third, we evaluate potential transitions by comparing ecosystems classified under current climate conditions (*this excludes savanna*) with those under future climate conditions (*this includes savanna*). This classification allows us to evaluate tree types of transitions (Fig. 1c): (i) current forest ecosystems that will transition to savanna in the future – referred to as forest-savanna transition, (ii) forest ecosystems that become 'more' water-stressed in the future, and (iii) forest ecosystems that become 'less' water-stressed in the future. For example, a lowly water-stressed forest transitioning to a moderately or highly water-stressed forest is considered as a 'transition to a more water-stressed state'. Whereas a transition from highly to a moderately or lowly water-stressed forest is considered a 'transition to a less water-stressed state'. We synthesise the results from all CMIP6-ESMs under different SSP scenarios (Fig. 1). Here, grid cells with > 50% of model convergence are referred to as 'moderate-high model agreement', 20-50% as 'moderate model agreement' and ≤ 20% as 'low model agreement'.

However, this methodology has two major assumptions: (i) We assume that the empirically derived S_r thresholds remain valid in the future, and (ii) ESM projected hydroclimatic estimates represent the actual climate (this includes the influence of atmospheric moisture flows on hydroclimate). To address the uncertainty due to the former case, we do several sensitivity analyses to check its influence on potential transitions. Whereas in the latter, we explicitly apply this methodology across a wide range of available ESMs under four SSP scenarios to highlight the agreements and conflicts between the end results. The implications of both these assumptions are discussed in the Discussion sub-section 'Limitation and sensitivity analyses'. Furthermore, the discrepancies between the prescribed land-use and our methodology-derived transitions are discussed in Fig. 5.

2.4 Sensitivity analyses

Since our study relies on ecosystem classification thresholds based on S_r, to evaluate the robustness of our analysis, we conducted four sensitivity analyses: (i) by assuming that the regions that exceed the 99th percentile S_r are prone to a forest-savanna transition as high S_r investment could be unrealistic from the perspective of plants, (ii) evaluating forest transitions individually for all three evaporation datasets, (iii) evaluating forest transitions under 10- and 40-year drought return periods and (iv) by changing the forest-savanna transition thresholds.

3. Results

In this study, we focus on three specific transitions: (i) Forest-savanna transition, (ii) forests' transition to a more water-stressed state, and (iii) reversion to a less water-stressed state (Fig. 2a). We find that under future climate conditions (2086-2100), considering >50% models' agreement, about one-fourth of the forests in both South America and Africa are projected to transition (Fig. 2b-g). With >20% models' agreement, these transitions are projected to occur for about three-fourths of the forests for both continents. Considering a lower threshold for models' agreement causes double or triple counting of some transitions (Fig. 2b-g). To minimise this in further analyses, we only consider >50% models' agreement for forests that transition to a more and less water-stressed state. Furthermore, because (abrupt) forest-savanna transitions are under-represented in ESMs(Lenton, 2011; Maslin and Austin, 2012; Valdes, 2011; Drijfhout et al., 2015), we consider >20% models' agreement for them. Considering this, we not only reduce the overlap to <0.4% of the total forest area (Supplementary Fig. 9), but we also maximise highlighting forest-savanna transition risk for both continents.

We find that forest-savanna transitions mainly occur in the Guiana Shield of South America, and the southern and south-eastern regions of Africa (Fig. 3). Compared to Africa, forest-savanna transitions are more prominent in South America under warmer climates (i.e., higher SSPs; Figs. 2b and 3). Our analysis reveals that the extent of forest-savanna transitions in South America decreases from almost 1.32 × 10^6 km^2 (16.3% of total forest area in South America) under the highest emission scenario to 0.04 × 10^6 km^2 (0.5%) under the lowest emission scenario (Fig. 2b). Interestingly, for Africa, the extent of forest-savanna transition did not change much for different SSPs, i.e., (median) 0.25 × 10^6 km^2 with a maximum deviation of ±0.11 × 10^6 km^2 (minimum and maximum extent of transition between 3-6.6% of total forest area in Africa) (Fig. 2c). By evaluating changes to their hydroclimate, we find that under warmer climates, forest-savanna transition regions in both continents are projected to experience a decrease in precipitation. Furthermore, we observe an increase in precipitation seasonality for South America, whereas Africa shows a decrease (Supplementary Fig. 12). Here, an increase in precipitation seasonality (seasonal variability in precipitation over the year) corresponds to water-deficit induced stress, whereas a decrease in seasonality and precipitation in Africa corresponds to a lower moisture availability altogether. Nevertheless, for both these continents, this transition seems to occur for the previously highly water-stressed forests under the current climate, followed by moderately, with the least contribution from lowly water-stressed forests (Fig. 3). This highlights the looming risk on highly water-stressed forests to experience a forest-savanna transition under warmer climates.

Forests that transition to a 'more' water-stressed state in South America are spatially aggregated towards the border between Brazil, Colombia, and Peru – covering a considerable portion of the Central Amazon (Fig. 3). Whereas for Africa, these forests exist in moderate to small patches towards the northern and southern extent of central Congo rainforests. We observe that these transitions account for most of the projected changes to forests' states across both continents (Fig. 2d,e), with the transition to just the 'highly water-stressed forest' accounting for more than three-fourths of all such transitions (Fig. 3). We observe that South American forests gradually become increasingly water-stressed under warmer climates, with maximum and minimum projected transition of 1.89 × 10^6 km^2 (23.4%) and 1.61 × 10^6 km^2 (19.9%) observed under the highest and lowest emission scenarios, respectively (Fig. 2d,e). Whereas for Africa, the change in the water-stressed state of the forests under different SSP scenarios remains almost similar (i.e., median 1.14 (±0.06) × 10^6 km^2; 19.6-22.2%). Analysis of their hydroclimatic changes reveals that water-stress is induced by both a decrease in precipitation and an increase in seasonality in South America (Supplementary Fig. 13). In contrast, water-stress in Africa is driven solely by an increase in seasonality. We observe that these newly water-stressed forests seem to have permeated to regions that were previously (under the current climate) dominated by lowly and moderately water-stressed forests (Fig. 3). Here, this shift only signifies the changes to hydroclimatic conditions allowing forests to transition to a more water-stressed state, rather than the changes to the floristic composition of terrestrial species from one location to another. Although such a shift under changing climate is not unlikely (Esquivel-Muelbert et al., 2019), they are not analysed in this study.

Forests that revert to a ‘less’ water-stressed state in South America are primarily observed in the south-eastern Amazon, with small patches observed towards eastern Brazil and the western coast of Equatorial Guinea and Gabon (Fig. 3). For Africa, the reverted forests exist in patches in the northern and southern regions of the Congo rainforest. Furthermore, for South America, we observe a gradual decrease in these reversions with an increase in warming. Here, we observe the lowest reversion of 0.23 × 10^6 km^2 (2.8%) under the highest emission scenario and the highest reversion of 0.67 × 10^6 km^2 (8.4%) under the lowest emission scenario (Fig. 2f,g). For Africa, these trends remain almost similar under all SSPs (i.e., median 0.18 (±0.05) × 10^6 km^2; 2.2-3.5%). Comparing these transitions with their hydroclimatic changes reveals an overall increase in precipitation (Supplementary Fig. 14). Interestingly, we observe a much higher precipitation increase for South America under high-emission scenarios than those in lower-emission scenarios. However, we find that precipitation seasonality is also higher for these ecosystems under warmer climates. This suggests that increased precipitation without changes to precipitation seasonality help decrease the water-stress of the ecosystem, compared to the ecosystems that experienced a simultaneous increase in both.

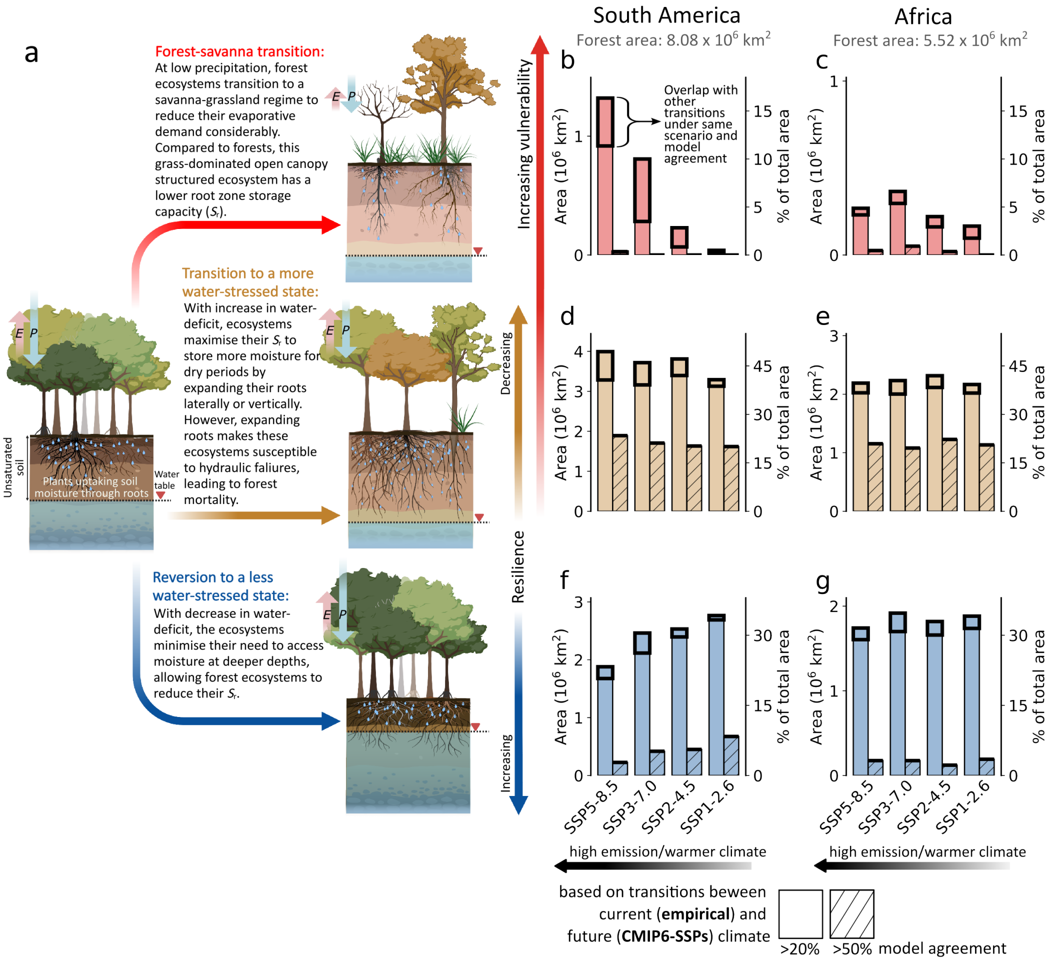

Figure 2 | Comparing the potential transitions under different SSP scenarios. (a) The state of the ecosystem, both above- and below-ground, (post-transition) under future climate, quantifying **(b,c)** forest-savanna transition, **(d,e)** forests' that transition to a more water-stressed state and **(f,g)** revert to a less water-stressed state for South America and Africa (present forest area mentioned on the top of (b,c)), respectively. For the analysis above, transitions are calculated for grid cells with model agreement >20% (plain bar plot) and >50% (hatched bar plot). These quantifications show changes in the forest area based on ecosystem transitions under empirical-current (2001-2012) and future (2086-2100) climate conditions. For all these transitions, we assume that the hydroclimate and vegetation are in equilibrium. Analyses comparing ecosystem transitions based on CMIP6-historical (2000-2014) and future (2086-2100) climate conditions are shown in Supplementary Figs. 10 and 11. For each transition, the total area of spatial overlap with other transitions under the same SSP scenario and model agreement is highlighted with thick black bars. The *P* and *E* arrows in (a) describe the relative magnitude of precipitation and evaporation fluxes. The illustration in (a) is adapted from Singh et al. (2020) and created with BioRender.com.

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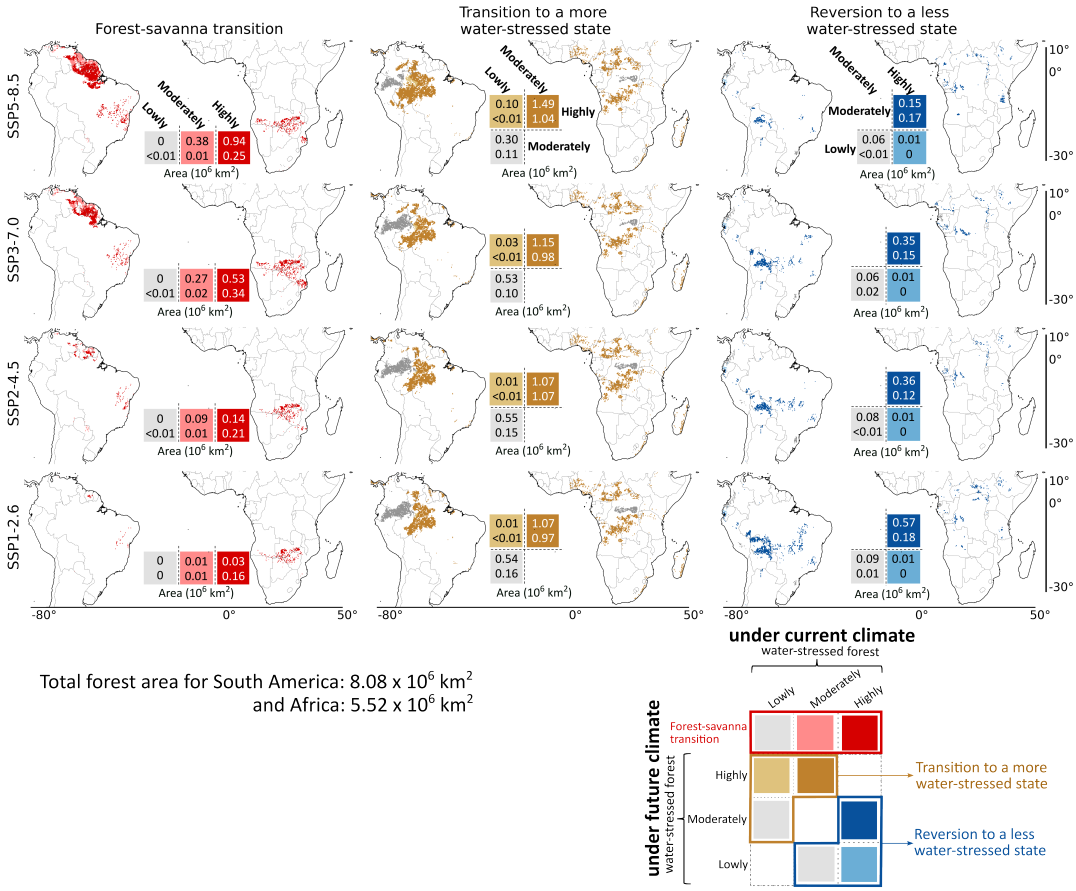

Figure 3 | Spatial extent of potential transitions with respect to their current state under different SSP scenarios. We analysed transitions, explicitly focusing on forest-savanna transition, transition to a more water-stressed state, and reversion to a less water-stressed state, by comparing different ecosystem classes under current (empirical; 2001-2012) and future (SSPs; 2086-2100) climate conditions (as defined in Fig. 2). All transitions shown above are analysed for moderate-high (>50%) model agreement, except forest-savanna transition, for which moderate (>20%) model agreement is considered. Values overlaying the legends correspond to the total area of transition for South America (top values) and Africa (bottom values).

4. Discussion

4.1 Asynchronous resilience risks under future climate change

Our analysis reveals the spatial extent of potential ecosystem transitions in South America and Africa and their vulnerability to future climate change (Fig. 2 and 4). For South America, we find a clear indication of a decrease in forest resilience (i.e., an increase in water-stressed forests) and an increase in forest-savanna transition risk under warmer climates (Fig. 2b,d,f). In contrast, these trends are not symmetric for Africa, where transition risk shows only slight variation across the different SSPs (Fig. 2c,e,g). Similar to the results of this study, previous studies on rainforest tipping have also suggested that exceeding 1.5-2°C will considerably increase the tipping risk (Jones et al., 2009; Parry et al., 2022), with the Guyana Shield in the Amazon being the most susceptible under future climate change (Cox et al., 2004; Parry et al., 2022; Staal et al., 2020) (Fig. 3 and Supplementary Table 3). Previous studies also agree that, in contrast to the Amazon, the projected risk to Congo rainforests is not substantial (Higgins and Scheiter, 2012; Staal et al., 2020) (Fig. 2). Despite it being unclear to what extent the ESMs represent the correct carbon dynamics (Koch et al., 2021), our results show a further divergence between Amazon's and Congo's responses to different SSPs (Fig. 2 and Supplementary Figs. 12-14). This could either be caused simply by a different response of precipitation patterns in the respective regions (Kooperman et al., 2018; Li et al., 2022) or by a different response to increased CO_2 levels in the atmosphere (Brienen et al., 2015; Hubau et al., 2020; Trumbore et al., 2015).

Previous empirical studies have found that forest ecosystems in the Amazon are more dynamic – grow faster due to high CO_2 levels in the atmosphere – than those in the Congo rainforests. However, these fast-growing trees also die young due to them investing substantially less in their adaptive strategies against perturbations than (less dynamic) old-growth forests (Brienen et al., 2015; Körner, 2017). This, combined with accelerated warming and frequent droughts faced by Amazon in recent decades, has made them more vulnerable to climate change than the Congo rainforests (Yang et al., 2018). For these ecosystems, the positive influence of CO_2 fertilisation-induced growth is counteracted by the negative impact of warming and droughts – thereby making the Amazon rainforest especially sensitive to CO_2 emissions pathways, which can exacerbate forest mortality under high emission scenarios (Brienen et al., 2015; Hubau et al., 2020). In this case, the projected changes to the future hydroclimate could be an artefact of forest mortality decreasing transpiration and precipitation over the rainforest. Previous studies also hint that these asynchronous resilience risks in the rainforest could be due to evolutionary biogeographical differences in the ecosystems leading to divergent species pools and resulting differences in ecosystems' functional attributes (Hubau et al., 2020; Singh et al., 2020; Slik et al., 2018), and nutrient limitation (Fleischer et al., 2019). According to them, the terrestrial species in Congo rainforests have already experienced severe droughts in the glacial periods, which makes them more adaptive to episodic water-induced perturbations than Amazon (Cole et al., 2014). Nevertheless, with compounding influence from land-use and climate-induced hydroclimatic changes (Davidson et al., 2012), these rainforests risk tipping to a savanna state. Our results highlight that by keeping the mean global surface temperature below 1.5-2^{o}C warming (which in this case is equivalent to SSP1-2.6 relative to the pre-industrial), we minimise forest-savanna transition risk and maximise recovery – thereby improving the resilience of rainforest ecosystems (Fig. 2).

4.2 Inferring adaptations from root zone storage capacity

We analyse S_r to relate changes in precipitation, precipitation seasonality and atmospheric water demand (Figs. 1-2 and Supplementary Figs. 12-14) with the ecosystem's dynamic subsoil adaptation (Singh et al., 2020). Here, the observed transitions are the aftermath of the ecosystem's minimising and (as observed in most cases) maximising their subsoil storage capacity to offset water-deficit and efficiently utilise available subsoil moisture under future climate change (Fig. 2a). Since plants prefer moisture uptake from the shortest pathway with the least resistance, a decrease in water deficit – increase in precipitation, decrease in seasonality and atmospheric water demand – will enhance the availability of moisture at shallow depths and motivate vegetation to utilise shallow roots for moisture uptake (Bruno et al., 2006). This allows the forests to reduce their total subsoil storage capacity while transitioning towards a less water-stressed state (Singh et al., 2020; Bruno et al., 2006).

However, an increase in water deficit forces forest ecosystems to invest in their subsoil structure and adapt strategies to store surplus moisture from wet seasons to ensure their survival during dry seasons, meaning that ecosystems transition to a more water-stressed state (Singh et al., 2020). Furthermore, higher investment in deeper and extensive lateral roots exposes plants to embolism-related hydraulic failures (Liu et al., 2022), thus increasing forest mortality risk under droughts (Aleixo et al., 2019; Anderegg et al., 2016; Bittencourt et al., 2020). This, along with other biotic and abiotic factors, including the maximum rooting extent of individual tree species (Canadell et al., 1996; Jackson et al., 1996), geological factors limiting roots to utilise deeper subsoil water and nutrient resources (Schenk and Jackson, 2002), and anaerobic conditions influencing microbial population at deeper depth (Dittert et al., 2006), among others (Alvarez-Uria and Körner, 2007; Brunner et al., 2015); caps the maximum adaptive capacity of the ecosystems to invest(Singh et al., 2020) and may influence diverse adaptive behaviour between ecosystems (Bonal et al., 2016). Under further episodic changes in soil moisture availability, i.e., beyond their maximum adaptive capacity, ecosystems survive by adapting to a new regime with relatively low moisture demand and more drought tolerance (Sankaran, 2019), which in this case is similar to a savanna ecosystem (Singh et al., 2020, 2022a).

4.3 Changes in atmospheric moisture flow drive forest-savanna transitions

Among all transitions, the most noticeable and catastrophic (since it is difficult to revert) is the forest-savanna transition projected in the Amazon's Guiana Shield of South America, and over the southern and south-eastern part of Africa (Figs. 2b,c and 3). These transitions are associated with the shifting of the inter-tropical convergence zone (ITCZ) (Mamalakis et al., 2021), which decreases precipitation and increases precipitation seasonality over the continents. For South America, the creation of these low-pressure bands allows the trade winds to bring in considerable moisture from the equatorial Atlantic Ocean over to Amazon by passing through the Guiana Shield and ultimately carrying it across the La Plata Basin via the South American low-level jet (Bovolo et al., 2018; van der Ent et al., 2010; Zemp et al., 2014). Similarly, for Africa, where south-eastern trade winds bring moisture from the Indian Ocean over the centre of the African continent (Mamalakis et al., 2021).

Under a warmer climate, sea surface temperature over the equatorial Atlantic and the northern Indian Ocean is projected to increase (Pascale et al., 2019; Zilli et al., 2019), leading to a southward shift in ITCZ over the eastern Pacific and Atlantic Oceans, and northward over east Africa and the Indian Ocean (Mamalakis et al., 2021; Xie et al., 2010). Previous studies also acknowledge that the intense surface warming over the Sahara under future climate can also attract ITCZ northwards in Africa (Cook and Vizy, 2012; Dunning et al., 2018; Mamalakis et al., 2021). Since these shifts in ITCZ can potentially both counteract and aggravate (especially critical for highly water-stressed forests) the impact of water-deficit, including those impacted by the localised deforestation (Leite-Filho et al., 2021; Schumacher et al., 2022; Staal et al., 2018; Wunderling et al., 2022). It warrants the need to include changes in atmospheric circulation for studies analysing the impact of future climate on the resilience of natural and human-influenced systems (Staal et al., 2020; Zemp et al., 2017).

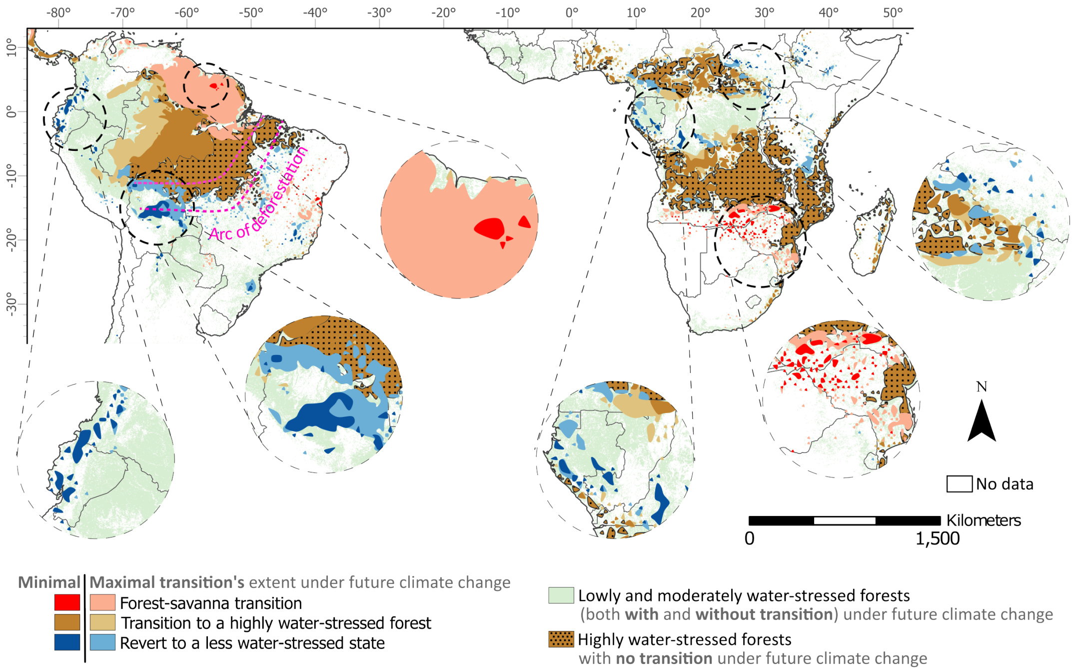

Figure 4 | Minimal and maximal extent of potential ecosystem transitions under future climate change in the entire study region over South America and Africa. The three transition types are: forest-savanna transition, from any class to highly water-stressed forests, and to a less water-stress state (see definitions in Figs. 2-3). For better visualisation of these transitions, in this figure, we first converted all grid cells to shape, merged them, and then smoothed using the 'polynomial approximation with exponential kernel' function (with a tolerance value of 1) in ArcGIS pro. The unsmoothed version of the transitions is shown in Fig. 3. The minimal and maximal represents the minimum and maximum possible extent of transitions (as quantified in Fig. 3) based on changes between current (empirical; 2001-2012) and future (SSPs; 2086-2100) climate conditions regardless of the SSP scenarios.

4.4 Comparing prescribed future land-use with projected transitions

Besides different radiative forcing, the CMIP6-ESMs also use prescribed land-use scenarios for each SSP (Hurtt et al., 2020). Therefore, it is interesting to check whether these land-use scenarios agree or conflict with the changes projected from our S_r-based ecosystem (classification and) transitions (Fig. 5 and Supplementary Figs. 15-18). Our analysis reveals that the extent of forest-savanna transitions is often underestimated in prescribed land-use compared to those projected in this study (i.e., prescribed land use predicts less non-forested areas; Fig. 5a). Whereas forests that revert to a 'less water-stressed state' is overestimated in our analysis (i.e., our analysis projects more forested areas; Fig. 5c). This discrepancy is because the land-use categories in CMIP6-ESMs are prescribed (simulated in Integrated Assessment Models (IAMs)) (Hurtt et al., 2020) and do not shift in response to hydroclimatic change. Also, the land-use scenarios used in the ESMs are assumed to evolve in time (as a function of macro-economic processes (Ma et al., 2020)), whereas our study does not account for the time required for transitions to unravel, and assumes an equilibrium between the 21st century hydroclimate (i.e., 2086-2100) and the ecosystems. In ESMs, this prescribed land use that does not dynamically respond to hydroclimatic changes could introduce errors in subsequent biophysical processes (Ma et al., 2020). For example, prescribing forest to a (projected) savanna region in ESM (e.g., Fig. 5a) will lead to extraction of deeper subsoil moisture, which ecosystems do not have the capacity to access (Ahlström et al., 2017; Yu et al., 2022), and, therefore, will lead to overestimation of ecosystem's evaporation and associated precipitation downwind. By analysing potential transitions based on hydroclimate-derived S_r, we highlighted the inconsistencies in prescribed land-use solely based on IAMs (Fig. 5).

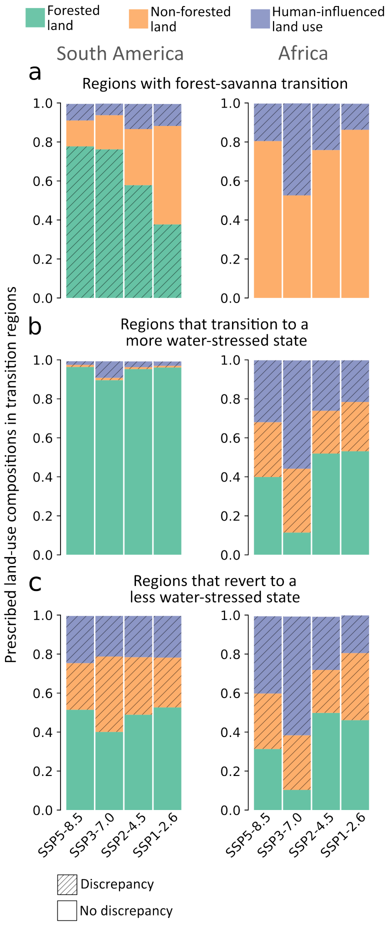

Figure 5 | Prescribed land-use composition for each transition region under different SSP scenarios (median 2086-2100), calculated as the ratio between the prescribed land use area and the projected transition area. Regions where IAM prescribed land use are same as projected transitions (from Fig. 3) are shown in plain colours (i.e., no discrepancy). Whereas regions where IAM-prescribed land use are different from projected transitions are hatched (i.e., discrepancy).

4.5 Limitations and sensitivity analyses

We assume that the S_r-derived thresholds – used to classify terrestrial ecosystems under the current climate – remain valid under future climate change. However, in some cases, this might lead to an over- or under-estimation of forests' adaptability to future climate change. For example, under the CO_2 fertilisation effect, forests may become more water-use efficient (i.e., less transpiration and therefore need for a lower S_r) (Xue et al., 2015), which will change the stability landscape in the future – implying an underestimation of forest resilience in this study. Whereas nutrient limitation (Condit et al., 2013) or extensive human influence (van Nes et al., 2016) in the ecosystems might lead to an earlier tipping – an overestimation of forest resilience. However, the uncertainty surrounding the effect of CO_2 fertilisation, nutrient limitation and human influence on vegetation makes them a research frontier for improving rainforest tipping projections under future climate change (Fleischer et al., 2019; Hofhansl et al., 2016). Furthermore, local scale factors, such as precipitation variability, species composition, and topography, might lead to a heterogeneous forest response (i.e., resilience) under future climate change (Staal et al., 2020). It should also be noted that though these uncertainties may hinder our understanding of local-scale forest resilience, the influence of future hydroclimatic changes on forests also constitutes major prediction uncertainties. Therefore, in this study, regardless of how these influences are parametrised or simulated in each ESM, we assume that hydroclimatic estimates projected by the ESMs represent the actual climate. Of course, this opens us (and other studies projecting forest conditions) in the future to certain limitations. For example, our capacity to project forest-savanna transitions (or any transition) depends on the model's capability to simulate complex feedbacks. On the one hand, some models have a complex vegetation-atmosphere interaction – simulating local and regional scale feedbacks across time (Ferreira et al., 2011; Jach et al., 2020). On the other, some models simulate these interactions based on simple parametrisation (Nof, 2008). Furthermore, some models might have a stronger CO_2 fertilisation effect, whereas some weaker (Koch et al., 2021). However, caution should be taken to not overgeneralise the functioning of tropical forests just from the analysis presented in this study, and also realise the current potential of ESMs to simulate them (Staal et al., 2020). To address this, rather than using just one ESM, we consider multiple ESMs under multiple SSP scenarios (Fig. 2, Supplementary Figs. 5-6, and Supplementary Figs. 10-11). This way, we highlight not only the rainforest tipping risks, but also the agreements and conflicts between potential transitions that will allow future studies to disentangle vegetation-climate feedbacks and improve the modelled local-scale interactions (e.g., parametrisation of rooting/soil depth, CO_2 fertilisation) in the ESMs.

Moreover, since the projected transitions are sensitive to ecosystems' hydroclimatic changes and adaptive strategies, we perform sensitivity analyses on S_r (representing S_r-based adaptation) and forest-savanna transition thresholds to check the robustness of the projected transitions (Figs. 1-2 and Supplementary Figs. 18-22). Fixing an extreme S_r threshold – signifying forest-savanna transition for ecosystems that cannot maintain their above-ground structure at high S_r – we observe some shifts close to the already projected risk regions and coastal regions (Fig. 3 and Supplementary Fig. 18). However, this transition risk in the coastal regions could be an artefact of interpolating hydroclimate estimates to higher resolution, and since oceans have more prevalent evaporation than land – it could lead to high S_r and, therefore, projection of tipping risk in coastal regions. We also discover that variations in the evaporation datasets and return periods utilized for calculating S_r have only a minimal effect on forest transitions (Supplementary Figs. 19-20). This is because, even though the forest classification thresholds may change due to different evaporation products under current climate (Singh et al., 2020), our histogram equivalence method ensures that forest classification under future climate adjusts relatively, resulting in minor alterations to the final outcome (Fig. 1b and Supplementary Figs. 19-20). Furthermore, while S_r values increase with increase in return periods, the impact of these changes is more noticeable for shorter return periods and less substantial for longer return periods (Wang-Erlandsson et al., 2016). Moreover, lowering the forest-savanna transition thresholds can lead to a reduction in forest-savanna transition risk due to an increase in precipitation range and seasonality for forest ecosystems (Supplementary Figs. 21-22). Despite these sensitivity analyses, the differences in transition magnitudes are minor, and the trends across different SSP scenarios for both continents remain similar (Fig. 2 and Supplementary Figs. 18-22). Therefore, the conclusions drawn from this study remain robust, even with variations in factors that could potentially affect forest transitions.

5. Conclusions

Classifying terrestrial ecosystems based on empirical and CMIP6 ESMs-derived S_r – the ecosystem's capacity to store surplus moisture and access moisture during dry periods – allowed us to assess the future transitions in the rainforest ecosystems. The lowest emission scenario minimises rainforest tipping risks and maximises reversion to less water-stressed states, whereas the opposite is achieved in the high emission scenario. In the Amazon rainforest, forest-to-savanna transition risks increase non-linearly with each degree of warming. In contrast, the risk increase between different emission scenarios is not significant for Congo. We believe that the results from this study can be used to further assess the direct and cascading influence of ecosystem transitions under future climate change on other natural and human-influenced systems (e.g., the influence of rainforest tipping on downwind rainfall, agricultural production and global food supply chain). We find very limited tipping risk that is 'unavoidable', whereas the vast majority of potential transition risks can still be avoided by steering towards a less severe climate scenario, highlighting the window of opportunity. Furthermore, regions projected to transition to less water-stressed state can potentially become easier to restore and respond well to deforestation prevention measures. This study highlights the importance of restricting temperature change below 1.5-2°C warming relative to the pre-industrial global surface temperatures to prevent potential forest transition risks and provide the best conditions for effective ecosystem stewardship.

Data availability

All the data generated during this study is made publicly available at Zenodo: https://zenodo.org/record/7706640. Other datasets that support the findings of this study are publicly available at: (CMIP6; citations referred to in Supplementary Table 2) https://aims2.llnl.gov/, (Root zone storage capacity; empirical) https://github.com/chandrakant6492/Drought-coping-strategy, (P-CHIRPS) https://data.chc.ucsb.edu/products/CHIRPS-2.0/, (E-BESS) ftp://147.46.64.183/, (E-FLUXCOM) ftp.bgcjena.mpg.de, (E-PML) https://data.csiro.au/collections/#collection/CIcsiro:17375v2, (E-ERA5) https://cds.climate.copernicus.eu/cdsapp#!/dataset/reanalysis-era5-single-levels, (Globcover) http://due.esrin.esa.int.ezp.sub.su.se/page_globcover.php. Potential transitions for each ESM based on the comparison between empirical (2001-2012) and SSP (2086-2100) scenarios are presented in Supplementary Data.

Code availability

The python-language scripts used for the analyses presented in this study are available from GitHub: https://github.com/chandrakant6492/Future-forest-transitions-CMIP6. The python-language code for calculating (empirical) root zone storage capacity is available from GitHub: https://github.com/chandrakant6492/Drought-coping-strategy.

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Author contributions

All authors contributed to the conceptualisation of this research. CS performed the analyses and wrote the initial draft. All authors contributed to the discussion and revision, leading to the final version of the manuscript.

Competing interests

The authors declare no competing interests.

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