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Dissecting forest transition: Contribution of mature forests, second-growth forests and tree plantations to tree cover dynamics in the tropics

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ABSTRACT

Forest Transition (FT) is a theoretical framework for understanding tree cover changes but often overlooks differences within countries, across forest types (e.g., second-growth forests, tree plantations replacing natural forests), regions, and climates. We quantified tropical tree cover dynamics across eight regions in four tropical countries, examining how these patterns relate to FT and how they vary between climates and forest types. Each country represented a different stage in the FT trajectory. We combined Landsat-derived time-series from 1990 to 2020 with Sentinel-2-based land cover classification to distinguish between mature natural forests (MF), second-growth forests (SF), tree plantations (TP), and their dynamics. During this period, 50 % of MF was lost, while tree cover gains averaged 16 % across regions; SF contributed 23 % and TP 12 % of total tree cover by 2020. SF steadily increased, yet its average lifespan was only 10 years, limiting its ecological contributions compared to MF. The studied regions followed the theoretical FT trajectory: the Ghanaian regions were in early transition (pre-inflection), Mexican regions were in late transition (pre-inflection), and the Australian and Brazilian (São Paulo state) regions were in post-transition (post-inflection). Evaluating FT while including or excluding TP results in different conclusions about the FT trajectory of a region or country. MF was lower in dry (from 55 % in the 1990s to 23 % in 2020) than in wet (from 73 % in the 1990s to 35 % in 2020) forest regions. SF gains were higher in dry (31 %) than in wet (23 %) regions, though SF increases did not compensate for MF loss, resulting in reduced biodiversity and ecological functioning. Hence, halting deforestation and protecting young forests are equally crucial. Evaluating FT excluding TP and quantifying SF persistence may have far-reaching consequences for how to evaluate tree cover by not only evaluating tree cover quantity, but also tree cover quality. Our findings can inform policymakers to design smart policy mixes that sequence the right policy instruments at the right time. Local people must participate in forest restoration strategies and issues of equity, justice and power imbalances must be addressed to facilitate FT.

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Dissecting FT increases our understanding of the underlying forest cover dynamics, which can lead to better policies for protecting local people`s livelihoods, halt deforestation, and facilitate FT to restore the natural world upon which people`s lives and society depend.

1. Introduction

Tropical forest loss has accelerated recently, after a decline in annual deforestation since 2000, increasing greenhouse gas emissions, and accelerating the loss of biodiversity and ecosystem services. Tree cover gains, however, have continued as a result of expanding natural second-growth forest (SF) and tree plantations (TP) (Chazdon, 2014; Fagan et al., 2022; Song et al., 2018; Vancutsem et al., 2021). Forest transition (FT) is a conceptual framework for understanding these tree cover dynamics (Mather, 1992; Meyfroidt and Lambin, 2011; Rudel et al., 2005). The FT framework postulates that in the early phases of FT a country`s net forest cover decreases until it reaches a lowest forest cover point, called the FT inflection point (Fig. 1). Hereafter net tree cover increases because reforestation outpaces deforestation (Meyfroidt and Lambin, 2011; Garcia et al., 2020; Barbier et al., 2010). Although FT describes net changes in forest cover, this does not necessarily result in increase in natural forest nor does it mean that the FT trajectory is deterministic (i.e. that a country or region follows this theoretical u-shaped trajectory by definition, but may undergo different transitions or unexpected forest cover change events). The FT literature often conflates tree cover types (e.g. natural regrowth and TP) and tends to focus on total forest cover, thereby overlooking the underlying dynamics of different tree cover types (Rudel et al., 2016; Sloan et al., 2019) and their contributions to forest restoration goals. Tropical SF and TP are frequently short-lived and how their dynamics link with FT is poorly understood (Rosa et al., 2021; Piffer et al., 2022a; Reid et al., 2019; Breugel et al., 2013). However, it is important to distinguish between natural forests and TP because they differ fundamentally in their structure, diversity, ecosystem processes and services, and the underlying drivers that shape their dynamics. Therefore, we use the term “forest cover” to describe natural forest and the term "tree cover" to describe the combination of TP and natural forests (de Jong, 2010; Heilmayr et al., 2016). Assessments of whether a country went through a FT inflection point or not is typically evaluated at a country level (Rudel et al., 2005), but also at sub-national and supra-national scale (Youn et al., 2017; Ashraf et al., 2017; Rudel, 2019). Large variation in tree cover dynamics within countries are common. These large variations may be associated, for example, with differences in climate (dry, wet) (Lambin and Meyfroidt, 2010) and the variation in FT drivers (such as China, Vietnam, Mexico) (Youn et al., 2017; Bonilla-Moheno and Aide, 2020; Hernández-Aguilar et al., 2021; Cochard et al., 2017). To understand the processes underlying FT, we evaluated tree cover dynamics variation within and between four dry and four wet regions in four tropical countries at different FT stages, and evaluated the types of tree cover that were lost and gained.

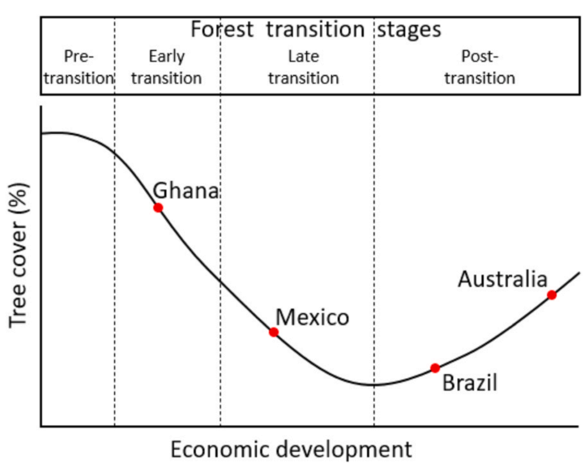

Fig. 1. Forest transition conceptual framework with four transition stages. The positions of the countries in which our study regions are located are projected on the theoretical FT trajectory. The pre-transition phase is characterized by high natural forest cover and low losses, while the early transition phase is characterized by rapid forest cover losses, and the late transition is characterized by a slowdown of forest cover loss and increasing tree cover gains. The post-transition phase is characterized by tree cover gains outpacing forest cover losses, and hence, net tree cover increases. The FT framework helps understand which policies can be most relevant and effective at different stages (Furumo and Lambin, 2021; Angelsen and Rudel, 2013).

Tropical tree cover dynamics vary across and within countries. For example, net forest cover change is currently close to zero in the Brazilian Atlantic Forest (Rosa et al., 2021), whereas the country’s Amazon rainforest continues to decline (Silva Junior et al., 2021; Qin et al., 2021). In other tropical countries, similar trends have been identified. Such as in Vietnam, where total forest cover has been increasing for several decades, although in some regions forest cover is declining (Cochard et al., 2017, 2023).TP cover dynamics show similar within-country variation (Nanni et al., 2019). Major deforestation drivers in the tropics are both commercial crop and commodity production, i.e., cattle, Eucalyptus, soy, palm oil, and rubber production, but also smallholder (subsistence) agriculture (Song et al., 2018; Pendrill et al., 2022; Curtis et al., 2018). Drivers of tropical tree cover gain are less well understood than drivers of deforestation (Piffer et al., 2022a; Nanni et al., 2019). In more recent years, FAO’s Forest Resource Assessments differentiate between natural forest cover gains and TP cover gains, and scientific studies of both tropical deforestation and FT in tropical countries increasingly differentiate between natural forest cover and TP cover dynamics. However, the literature rarely considers the contributions of SF and TP cover to the trajectory of FT (but see, for example, Rosa et al., 2021).

Different tree cover types (e.g. natural forests or TP) contribute differently to forest restoration goals, biodiversity conservation, climate change mitigation and ecosystem services (Fagan et al., 2022; Rudel et al., 2016; Nanni et al., 2019; Hua et al., 2022). Commercial TP, such as Eucalyptus, oil palm, and rubber plantations, have a much narrower set of valued ecosystem services because they are optimized for large-scale, intensive production (Hua et al., 2022; Altamirano et al., 2020; Wilson et al., 2017). In contrast, SF are important for forest restoration targets because recovering natural forests host more biodiversity and provide a broad range of valued ecosystem services compared to TP (Chazdon, 2014; Hua et al., 2022). These valued ecosystem services can provide a wide range of benefits such as food, fuel, wood, medicine, and timber to meet local livelihood needs. However, SF contributions depend, among others, on their persistence.

SF in many tropical regions are allowed to grow for a limited time only. For example, smallholder fallow agriculture (Breugel et al., 2013; Martin et al., 2023), where the fallow period is used to suppress weeds and recover soil fertility, and then slashed again and used for the next cropping cycle. Land under SF may therefore be used again for agricultural production, and, not uncommonly, SF may be slashed after some years to prevent it from becoming forests that are protected under law, causing farmers to lose ownership over the land (Rosa et al., 2021; Calzada et al., 2018; Chazdon et al., 2020; Román-Dañobeytia et al., 2014). For example, in the Brazilian Atlantic Forests, SF that regenerated between 1985 and 2019, on average were cleared again after only 8 years (Piffer et al., 2022b). Similar trends have been observed elsewhere in the tropical forest biome. Hence, because of their short permanence, the long-term contribution of SF to regulating ecosystem services and especially biodiversity conservation is limited (Rosa et al., 2021; Breugel et al., 2013; Lohbeck et al., 2022). FT literature does not differentiate between short-lived SF and permanent SF and what roles these SF play in FT.

Tree cover dynamics also differ between dry and wet tropical forests. Between 2000 and 2020, tropical dry forest lost 33 % of its cover (between 56 and 71 Mha) (Ocón et al., 2021; Buchadas et al., 2022), whereas tropical wet forest cover decreased by 12 % (138 Mha) (Vancutsem et al., 2021). This difference could be resulting from higher soil fertility in drier regions because of less rainfall and related nutrient leaching, but also from the utilization of irrigation systems that incentivize more intense agriculture. In general, drier regions are more suitable for mechanized agriculture, and their population densities are higher (Sánchez-Azofeifa and Portillo-Quintero, 2011; Sánchez-Azofeifa et al., 2005; Siyum, 2020). Lower incidence of pests and diseases in drier climates may also explain more intensive land use in dry tropical forest regions (Sánchez-Azofeifa and Portillo-Quintero, 2011). Hence, agricultural land use may be generally more permanent. Forest recovery on abandoned agricultural lands is much faster in wet tropical forests than in dry tropical forests because of the higher rainfall and longer growing season (Poorter et al., 2016).

Transitioning from net forest cover loss to net forest gain, in short forest transition, can be achieved by halting deforestation and forest degradation, and increasing forest restoration (Garcia et al., 2020). FT partially results from a mix of (sometimes conflicting) policies that increase forest cover losses (e.g. agricultural subsidies), slow down forest cover losses (e.g. forest certification), and forest restoration (e.g. payment for ecosystem services). To facilitate FT and reverse deforestation lock-ins, purposeful sequencing of smart policies and instruments can address conditions evolving along FT curve (e.g. the right policy at the right time) (Furumo and Lambin, 2021; Angelsen and Rudel, 2013). For example, in early FT stages, policies can avoid infrastructure projects close to intact forests, while in later FT stages, policies can accelerate forest restoration (Furumo and Lambin, 2021).

This study advances our understanding of tree cover dynamics in tropical forest regions and assesses their implications for regional FT. We compared tree cover dynamics across tropical regions with different socioeconomic development, differentiated between different types of tree cover dynamics, and compared these dynamics between dry and wet tropical forests. To achieve this, we combined remote sensing time-series models based on all available Landsat images from 1990 to 2020 and land cover classification based on Sentinel-2. Specifically, we ask (1) how do tree cover loss and gain change between 1990 and 2020 in a selected number of globally distributed tropical forest regions, (2) how do these dynamics relate to the theoretical FT curve, (3) how do tree cover dynamics vary between dry and wet tropical forests, and (4) what role do SF cover dynamics play in regional FT? The four dry tropical forest and wet tropical forest regions are located in Ghana, Mexico, southeastern Brazil, and Australia; four countries in different socioeconomic development phases which represent different FT stages) (Fig. 1).

2. Methods

2.1. Study areas

To answer our research questions, we identified eight focal study regions using three selection criteria. First, we selected four countries (Ghana, Mexico, Brazil, and Australia) that differ in socioeconomic development and therefore represent different stages of the FT curve (Figs. 1 and 2; Table 1). Second, within each country we selected a dry and wet tropical forest biome. We should emphasize that our definitions of dry and wet are relative and not absolute. For example the Ghanaian wet region has a lower precipitation (1690 mm/year) than the Australian dry region (2080 mm/year), because in Ghana there are no forests with an annual precipitation higher than 2000 mm. Third, to ensure comparability between regions we selected within each biome a rectangular area of 20 × 20 km (40,000 ha) where we had field experience. This experience enables an in-depth understanding of local forest tree cover changes and their underlying dynamics. We selected a size of 20 × 20 km because this is still a relatively homogeneous area from a socioeconomic and biophysical point of view. We did not increase the number of study regions because of logistic and time limitations). For a more detailed context description per region, see the Supplementary Materials (Supplementary Materials A).

Ghana. In Ghana's tropical forest region, smallholder agricultural landscapes with agricultural fields and SF fallows occur next to the production of coffee, cocoa, and gold. Legal and illegal logging, often in forest reserves, has had a large impact in the past but has slowly decreased in recent years. In the dry region of Ghana, the region of Abofour is close to the savanna transition zone, and people depend on fallow agriculture, hunting, and logging (Murphy and Lugo, 1986). However, owing to the increasing demand for land, farmers have been moving away from fallow agriculture to permanent agriculture on the same land every year. Deforestation is driven by agricultural land expansion and legal and illegal logging of timber and other forest products. Fires, escaping from agriculture, and uncontrolled bush fires from hunters also drive forest losses (Owusu and Essandoh-Yeddu, 2018). Increased fuel load and flammability in combination with wildfires result in a positive feedback loop of building-up fuel load and fire, accelerating forest loss (Amissah et al., 2011). In particular, since the El Niño drought event of 1983, annual forest fires have had a greater impact. Consequently, farmers moved away from perennial crops, such as cocoa, to annual crops, such as maize (Amissah et al., 2011). In the wet region of Ghana, the region of Bonsa, the expansion of gold mining has accelerated deforestation. Cocoa farming traditionally occurs in dry regions, but has increasingly moved to wet regions to reduce forest fire risks. Rubber and palm oil plantations are expanding too.

Mexico. In the past half-century, Mexico went through a process of industrialization, agricultural expansion, and intensification with the help of governmental subsidies. These subsidies supported subsistence farmers to intensify their production, and remote areas gained better access to national and international markets. More recently, market-driven programs for conservation and restoration were implemented (Lohbeck et al., 2022; Berget et al., 2021). The Mexican tropics, mostly located south of the Tropic of Cancer, are more humid in the Atlantic watershed and drier in the Pacific watershed because of the strong influence of the easterlies, which bring humidity inland from the Gulf of Mexico. In the dry region of Mexico, in the Isthmus of Tehuantepec, the government has for decades promoted economic development of the flat coastal plain because of the strategic position of the region connecting the Atlantic and Pacific oceans, including large scale irrigation, petrochemical industry, and recently the expansion of wind farms. Oaxaca is one of the most economically underdeveloped states in Mexico. In the hilly areas of the Isthmus, the soils are shallow, and mechanized agriculture is not feasible. Therefore, after one or two cycles of crop cultivation, the land is left fallow. Cattle raising has been in the region for decades but has increased steadily since the 1990s. In the wet region, Marqués de Comillas, in the underdeveloped state of Chiapas, smallholder subsistence farmers colonized the region during the 1970s-1980s. A large forest reserve since 1978 has slowed encroachment and conserved parts of the tropical rainforest. Since the 1980s, deforestation has accelerated because Guatemalan refugees were hired as cheap laborers. The severe drought caused by the 1997–1998 El-Niño year enabled the spread of human-induced fires and caused large-scale loss of forests (Lohbeck et al., 2022; Berget et al., 2021). During the mid-2000s, palm oil and rubber production were introduced. The region was shaped by sometimes conflicting government subsidies and policies, subsistence farming, cattle raising, and expanding market access, resulting in a landscape with a mix of SF, plantations, old-growth forest remnants, cattle pastures, crop fields, and settlements (Zermeño-Hernández et al., 2015).

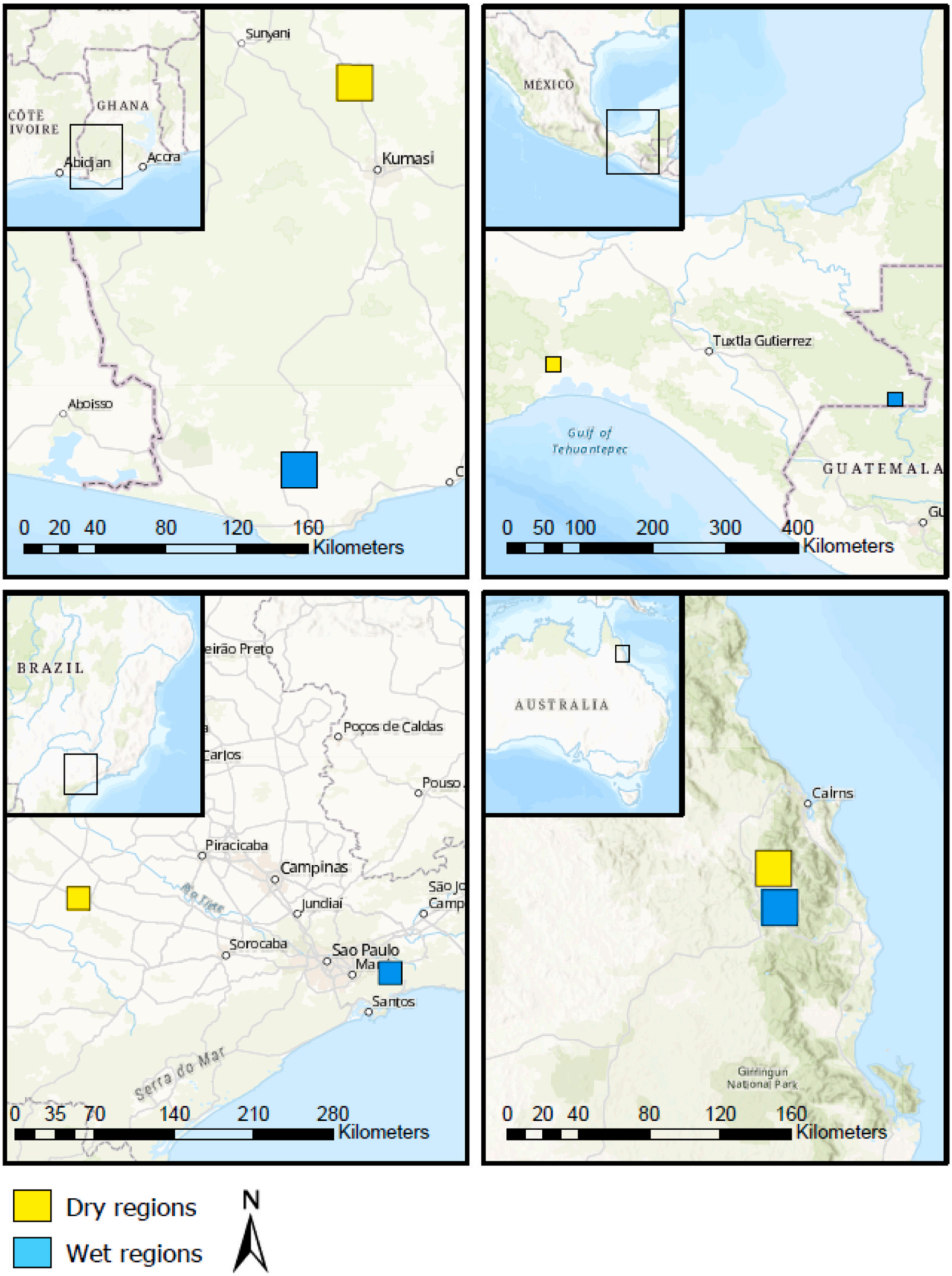

Fig. 2. Overview of studied regions and their respective locations. Yellow and blue regions correspond to dry and wet regions. Each study region has an extent of 20 × 20 km. Maps made in ArcGIS Pro 2.4.

Table 1
Bioclimatic overview of all study regions.

Bioclimate	Ghana		Mexico		Brazil		Australia	
	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
Ecoregion (WWF)	Tropical & Subtropical Moist Broadleaf Forests	Tropical & Subtropical Moist Broadleaf Forests	Southern Pacific dry forest	Petén-Veracruz moist forest	Cerrado	Serra do Mar coastal forest	Queensland tropical rainforest + Einasleigh upland savanna	Queensland tropical rainforest
Köppen-Geiger classification	Tropical Monsoon	Tropical Savanna (dry winter)	Tropical Rainforest	Tropical Savanna (dry winter)	Temperate, no dry season, hot summer	Temperate, no dry season, hot summer + warm summer	Temperate, dry winter, hot summer	Temperate, dry winter, hot summer + Temperate, no dry season, hot summer
Temperature (degree Celsius)	25.6	26	26	25	19.8	19.1	20.8	20.5
Annual rainfall (mm/year)	1380	1690	1050	2670	1380	2010	2080	2800
Elevation range (m a.s.l.)	250–500	5–250	10–900	100–300	700–1000	800–1200	700–1000	700–1000

Brazil. The Atlantic Forest (wet and partially dry regions) and Cerrado (partially dry region) in the state of São Paulo have a long history of deforestation and fragmentation starting when colonizers arrived. Sugarcane production in the 16th century was followed by coffee production in the 19th and early 20th centuries, when most deforestation occurred (Dean, 1997). In addition, pastureland for beef and milk production, cash crops (e.g., sugarcane, corn, and soy production, and *Eucalyptus* plantations) have become additional causes of deforestation. Cash crops are mostly grown on flat terrain because they are more suitable for mechanized agriculture. SF and monoculture TP, as well as pastures, are located on hilly terrains. Due to the Brazilian Forest Code from 1965, landowners of this region are required to have 20 % of their lands, and additionally riparian buffers and other fragile environments like mountain tops and steep slopes, covered with native forests. If current forest cover is not enough, farmers have to restore forests to supply their legal deficit. Enforcement of the code was sparse but has increased in the state of São Paulo since the early 2000s (Brancalion et al., 2016). This contributed to a steady increase in SF cover. Federal legislation protects Atlantic Forest remnants of intermediate to advanced successional stages from deforestation. Because of the long history of the Atlantic Forest, land use in the state has become a complex matrix of industrial agricultural production and nature conservation.

Australia. Deforestation of the Australian wet tropics, specifically the Atherton Tablelands in Queensland started in the late 19th century with the logging of natural forests (Gilmore, 2005). Agriculture followed logging as transport routes were established, and cropping and dairy grazing lands expanded into flat and fertile uplands (Frawley, 1987). Since the 1970s, the accelerated conversion of intensively used land to forests in uplands has led to a decline in the dairy industry. The rate and area of SF recovery have been influenced by several factors, such as time since clearing, proximity to mature natural forests, abandonment, and soil fertility (Goosem et al., 2016; Sloan et al., 2016). In contrast, deforestation surged again since the late 1980s, prior to the protection of the region's mature natural forest with the demarcation of the World Heritage Area, and more recently with increases in beef prices and droughts in other regions (S. Laurance, pers. comm). Forest restoration through tree-planting programs began 40 years ago as an effective way to facilitate forest recovery on lands with aggressive pasture grasses (Goosem and Tucker, 2013). Commercial TP were only of minor importance in this region. Cyclones have been an important cause of forest disturbance, such as Cyclone Larry in 2006, which hit the Atherton Tablelands directly and caused extensive damage to rainforests (Turton, 2008, 2012).

2.2. Tree cover dynamics

For a detailed description of data analyses, see the Supplementary Materials (Supplementary Materials B-E).

2.2.1. Land cover classification

To distinguish between different tree cover types, we classified land covers in each region based on Sentinel-2 satellite imagery from 2020 and RandomForest machine learning in Google Earth Engine. We adopted the publicly available algorithm from Tassi and Vizzari (2020) (Tassi and Vizzari, 2020), which is a Geographic Object-Based Image Analysis (GEOBIA) that clusters similar pixels into a single object, instead of classifying individual pixels (Tassi and Vizzari, 2020; Kucharczyk et al., 2020; Ye et al., 2018). We used 25 training points per land-cover class and approximately 50 validation points per land-cover class (stratified sampling) per region (Table SB.2). The sample size for training was based proportionally to the area of the study area of Tassi and Vizzari (2020) (Tassi and Vizzari, 2020). The sample size for validation is based on common practice where 50 samples per class is a rule-of-thumb (Ye et al., 2018; Congalton and Green, 2019). For validation, we performed an accuracy assessment for forests/non-forest cover and thereafter an accuracy assessment for the separate land cover classes. We used a mix of high-resolution satellite images from Google Earth Pro timelapse, BING imagery, and Planet imagery (from the year 2020) for the validation procedure. To avoid challenges with validation points at the edges of the images, a 50 m buffer was used for creating the validation points. To ensure consistency, the validation procedure was performed by the same person. The average accuracy level of the assessment for the forest/non-forest was 86 % and of the assessment for all land cover classes was 76 % (Supplementary Materials C). To match the pixel resolution of the land cover maps with the resolution of the Landsat-derived time-series, we resampled the land cover maps with the nearest neighbor approach before the validation procedure with the terra package in R.

2.2.2. Landsat-based disturbance-regrowth time series

To analyze tree cover dynamics, we used the Anomaly Vegetation Change Detection (AVOCADO) algorithm with all available Landsat-derived Normalized Difference Moisture Index (NDMI) vegetation index (Decuyper et al., 2022). NDMI is sensitive to canopy moisture content and is therefore sensitive to changes in canopy structure, and is especially useful for detecting forest disturbances and regrowth (Decuyper et al., 2022; Requena Suarez et al., 2023). AVOCADO was designed to consider seasonal patterns (e.g., in the case of deciduous forests). We downloaded all available level-1 Landsat satellite images in Google Earth Engine. The satellite imagery-stacks ranged from January 1st 1990 to December 31st 2020. However, we started each time-series from the moment that satellite image retrieval was consistent for each region. Hence, the starting time differed from region to region (Table SB.3).

We selected mature forest (MF) cover patches to construct a reference phenological baseline (Figure SB.1). MF was defined as a forest that was visually mature at the beginning of the monitoring period. We verified that these patches matched the definition with all high-resolution Google Earth Pro satellite images available, visually inspected the AVOCADO time-series, and consulted experts with extensive local field experience in the study areas (also authors of this article).The phenology baseline uses all available NDMI pixel values within the MF cover patch of the Landsat reference layers between 1990 and 2020 to construct a confidence interval.

After constructing the phenology baseline with a confidence interval, all pixels in the study areas between 1990 and 2020 were compared to the reference baseline. If the NDMI value of a pixel was within the confidence interval of the phenological baseline, it was classified as mature. Consequently, at the start of the study period, old(er) second-growth forests with NDMI values comparable to mature forest NDMI values could not be separated from mature forest. If three consecutive pixel values were below the baseline confidence interval, they were classified as being disturbed. The confidence interval of the reference phenology baseline, which was created from all reference years, was set to 95 % or 99 %. The choice depended on the consistent availability of Landsat images. For Ghana, Landsat availability was more inconsistent. Hence, we set the confidence interval to 95 % for Ghana. For the other countries, we set the confidence intervals to 99 %. Pixels were classified as regrowth when three consecutive pixel values of a disturbed pixel were within the confidence interval of the reference phenology baseline and no disturbance was detected within two years after the first regrowth. AVOCADO can detect up to four cycles of disturbance and regrowth, which is relevant for landscapes with high forest turnover. To construct the time-series we followed the procedure of Decuyper et al. (2022) (Decuyper et al., 2022). An example of the AVOCADO main output can be found in the supplementary materials (Figure SB.2).

To validate the results from the AVOCADO time-series, we performed an accuracy assessment for Australia dry. We randomly selected 200 samples equally divided over four strata: stable forest, stable non-forest, disturbance and regrowth. Australia dry was chosen because to accurately validate disturbances, and especially regrowth, very high resolution satellite images are needed close to the actual change events. We used the available very high resolution images in Google Earth Pro and aerial photographs from the Queensland government for the accuracy analysis. Stable forest and stable non-forest were classified with high user`s accuracies (100 % and 94 %, respectively), and disturbance and regrowth with low user`s accuracy (40 % and 22 %, respectively). The overall accuracy of the Australia dry time-series was 64.0 %. Disturbances and regrowth are actually often actually stable forest. One reason for this misclassification is natural events (droughts or cyclones/storms) that have prolonged effects on the tree cover to recover, however, these disturbances are difficult to detect with very high resolution imagery and cannot be classified as deforestation, hence it is not always possible to distinguish deforestation events from disturbances. These misclassifications result in an overestimation of stable forest disturbance and an overestimation of regrowth (which results from these initial misclassified disturbances). See supplementary materials for a more detailed description (Supplementary materials SE).

2.2.3. Differentiating tree cover classes

To distinguish between different types of tree cover gain, we masked regrowth pixels (AVOCADO) with (monoculture) TP from the land cover maps. Furthermore, land cover maps were used to classify and mask non-tree cover classes (e.g., built-up) and water bodies. A disturbed pixel without regrowth (AVOCADO) was assigned to the land cover class of the matching land cover pixel (e.g., agriculture). A regrowth pixel (AVOCADO) was classified as SF if it overlaid natural forest on the land cover map. The same was done for TP classes (i.e., oil palm, *Eucalyptus*, rubber, cocoa, and timber). If a regrowth pixel was assigned to a TP class, then all consecutive regrowth pixels were assigned to that TP class. Regrowth pixels that did not match any tree-cover class, because, for example, these regrowth pixels are an outcome of an agricultural land cover type where the NDMI pixel value was within the phenological baseline and thus classified as regrowth, were masked out (see Supplementary Materials SF).

2.3. Survival analysis of second-growth forests

To assess the persistence and longevity of SF, we analyzed survival probabilities of SF in all study regions with Kaplan-Meier survival analysis of the survival in R-package (Lohbeck et al., 2022; Jakovac et al., 2017; Therneau, 20092023). Furthermore, we analyzed temporal changes of survival probabilities by dividing SF pixels in 'early' (~1990–2005) and 'late' (~2005–2020) groups, based on the year of establishment. All SF pixels that survived until the end of the study period (i.e. no event happened until that moment) were right-censored. Right censoring is applied in survival analysis because at the time the study period ends, we do not have any information if or when the event will happen (i.e. non-informative). To obtain a more spatially unbiased subset of all the SF pixels, we randomly selected 2000 pixels from each region. Furthermore, we analyzed the persistence of SF in two groups: persistent SF (i.e., SF that was established during the monitoring period and survived until the end of the monitored period) and ephemeral SF (i.e. established during the monitoring period but cut before the end of the monitoring period) (Piffer et al., 2022a). All analyses were performed using R version 4.2.2 (Core Team. R, 2022).

3. Results

3.1. Changes in tree cover losses and tree cover gains

Overall, we observed steep tree cover losses, but with high variability between regions and countries. At the end of 2020, the average total forest area, including MF and SF, for all regions was 39.9 % (SD = 11.2 %, range 9.1–50.8 %) per region (Fig. 3; Table 2). Compared to the total land area per region, MF cover loss averaged 34.5 % (SD = 15.7 %, range 12.0–58.2 %), but of the total mature forest at the beginning of the monitoring period, 46.9 % (SD = 13.9 %, range 24.7–63.6 %) was lost by the year 2020 (Fig. 3; Table 2). Tree cover gain averaged 15.8 % (SD = 8.6 %, range 8.6–36.8 %) (Fig. 3; Table 2). Contributions to total tree cover gain for SF averaged 10.6 % (SD = 3.6 %, range 4.6–16.7 %) and for TP averaged 5.3 % (SD = 7.3 %, range 0–24 %) (Fig. 3; Table 2). Our results showed that SF cover is highly dynamic and generally short-lived (see Section 3.4).

3.2. Variations between regions with different positions along the forest transition

Trends in forest cover dynamics, i.e., natural forest cover gains and losses but not TP cover, showed that the studied regions occupied different positions along the theoretical FT curve. The Ghanaian regions were in an early-transition phase, as they experienced rapid forest cover losses. The Mexican wet region was close to the FT inflection, caused by decreasing forest cover losses and increasing SF cover. The Mexican dry region was just past the inflection point. The Brazilian regions experienced clear mature forest cover losses and SF recovery, hence, they were in the post-transition phase (Fig. 3; Table 2). The Australian regions were past FT inflection too, but their trajectory was flatter than that of the Brazilian regions. Total natural forest cover changes (mature and SF) were highest in Ghana (dry = −42.1 %, wet = −39.9 %), followed by Mexico (dry = −24.1 %, wet = −50.5 %), Brazil (dry = −42.1 %, wet = −19.3 %), and Australia (dry = −1.3 %, wet = +1.4 %) (Fig. 3; Table 2).

When TP cover was included in the evaluation of tree cover dynamics, then the trends were different for three regions. When including TP, Ghana dry approached the FT inflection point, and Mexico's wet region was in a post-transition phase. In Brazil, dry TP cover contributed

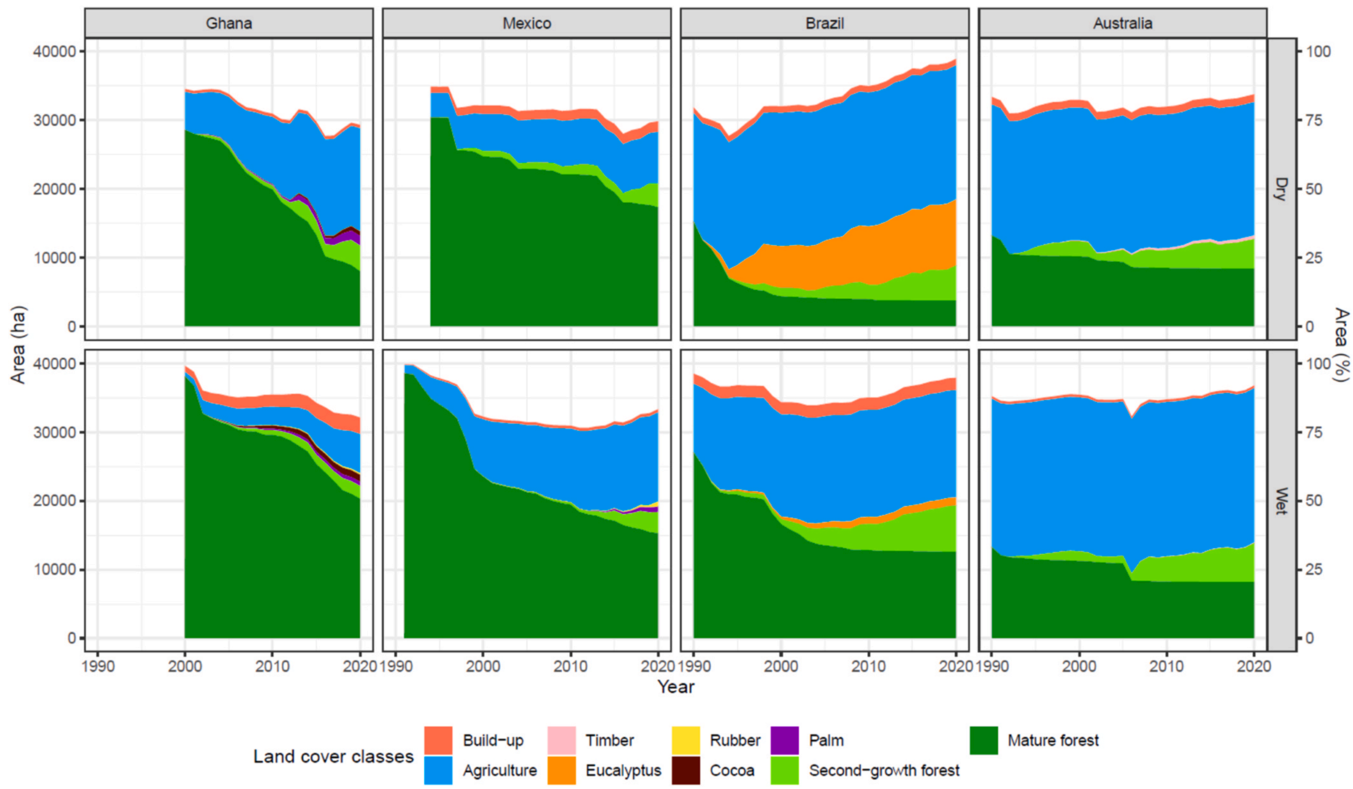

Fig. 3. Trends of land cover changes by study region in absolute (in ha, left axis) and relative (in %, right axis) area.

Table 2
Tree cover changes between the start and the end of the study period. The end of the study period is at the end of the year 2020. The start of the study period is 1990, however because of limited Landsat image availability in Ghana and Mexico, starting dates were later.

Study regions (start of monitoring period)	Ghana		Mexico		Brazil		Australia	
	Dry (2000)	Wet (2000)	Dry (1994)	Wet (1991)	Dry (1990)	Wet (1990)	Dry (1990)	Wet (1990)
Mature forest cover in 2020 (ha)	8024	20,307	17,358	15,282	3764	12,592	8411	8186
Second-growth forest cover in 2020 (ha)	3707	1822	3431	3075	5112	6698	4275	5643
Tree plantation cover in 2020 (ha)	2101	1868	0	1513	9600	1191	546	109
Total tree cover in 2020 (ha)	13,832	23,977	20,789	19,870	18,476	20,481	13,232	13,938
Mature forest cover loss between start and end of study period in ha (and %)	20,563 (51.4)	17,796 (44.4)	13,058 (32.6)	23,262 (58.1)	11,446 (28.6)	14,415 (36.0)	4807 (12.0)	5078 (12.7)
Total natural forest cover change in ha (and in %)	−16.856 (−59.0)	−15.974 (−41.9)	−9.627 (−31.7)	−20.187 (−52.4)	−6.334 (−41.6)	−7.717 (−28.6)	−532 (−4.0)	656 (4.3)

similarly to tree cover gain as SF cover. Hence, considering TP, the region passed the FT inflection ~10 years earlier. Total tree cover changes, including TP cover, were highest in Ghana (dry = −11.9 %, wet = −35.3 %), followed by Mexico (dry = −24.1 %, wet = −46.7 %), then Brazil (wet = −16.3 %, dry = +8.2 %) and Australia (dry = 0.0 % ha, wet = +1.7 %) (Fig. 3; Table 2).

In line with the FT curve, SF cover increased in all regions, but differed by region. SF cover increase was highest in the Brazilian and Australian regions, while SF cover increase was only half of that for the Ghanaian and Mexican regions. SF cover increases partially compensated for MF losses and fully compensated for losses in the Australian regions.

3.3. *Differences between dry and wet regions*

By the end of the study period, the dry regions had a lower average natural forest cover (33.8 %, SD = 11.1 %) than the wet regions (46.0 %, SD = 7.5 %) (Fig. 3; Table 2). The average dry MF cover loss was lower than the wet MF cover loss (31.2 % with SD = 14.0 % versus 37.8 % with SD = 16.5 %) (Fig. 3; Table 2). The average SF cover gain in dry tropical forests was similar to that in wet forests (10.3 % with SD = 1.6 % versus 10.7 % with SD = 4.9 %). However, tree cover gains, including SF and TP, in dry tropical forests were higher (10.8 %, SD = 11.1 %) than in wet tropical forests (13.7 %, SD = 3.9 %).

3.4. *Role of second-growth forest cover dynamics*

SF cover dynamics differed across regions and time. SF cover was short-lived in all regions, although the average longevity of secondary forests increased over time. Over the whole study period, SF survival (expressed as 50 % probability of survival in years) was 10 years, but it increased from 7 years in the first half of the study period to more than half of SF cover surviving during the second half of the study period (i.e., median probability not dropping below 50 %) indicating a slowdown of SF turnover rate (Figs. 4, 5). Over the entire study period, more than 50 % of SF did not survive after 20 years. SF survival was lowest in Brazil's dry region (5 years). SF survival decreased drastically in the two Ghanaian regions and slightly in Mexico's dry region (Fig. 5). SF cover

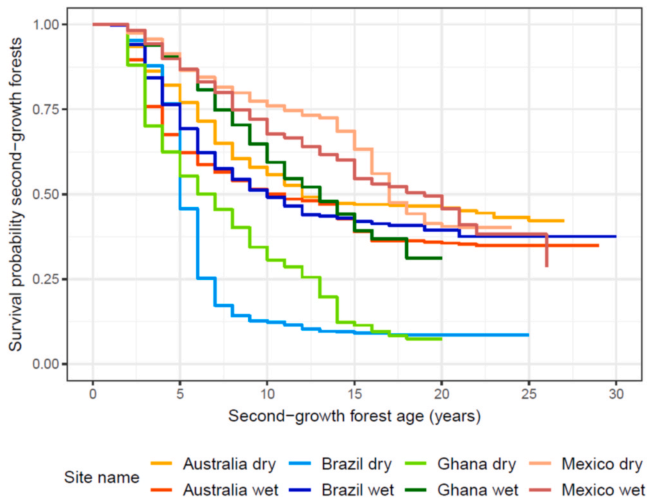

Fig. 4. Probability of second-growth forest survival by study region. Sample of 2000 randomly selected pixels per region.

that was established within the study period and survived until the end of the study period (i.e., 'persistent' SF) had an average age of 8.2 years (SD = 5.9), while SF cover that was established within the study period and cut before the end of the study period (i.e., 'ephemeral' SF) had an average age of 5.8 years (SD = 3.6).

Comparing the dry and wet climate regions, SF survival was lower in the dry tropical regions (8 years) compared to the wet tropical regions (13 years) (Fig. 6). Persistent SF cover in the dry regions was of similar age as in the wet regions (7.1 years +/- 4.9 versus 7.1 +/- 5.5). Similarly, ephemeral SF of the dry regions was of similar age as in the wet regions (5.4 years +/- 2.8 versus 5.1 +/- 3.11).

4. Discussion

We analyzed tree cover changes in eight regions in four tropical countries that differed in level of economic development. We quantified losses of mature forest cover, gains of SF and TP covers, and SF cover persistence for the past three decades.

We found that forest cover losses and tree cover gains varied tremendously between regions, but forest cover losses were generally greater than tree cover gains. Natural forest cover dynamics were generally in line with the FT conceptual framework. The results were different when TP was included or excluded from overall tree cover gains. Dry forest regions lost mature forests faster than wet forest regions did. SF cover increased in area and, thus, partially replaced mature forests in area. However, overall SF was short-lived.

4.1. Overall trends in tree cover

Over the past three decades, mature forest cover in our study regions has declined by almost half. This was much higher than the overall values reported for the studied countries (Fig. 3; Table 2). The Forest Resource Assessment (FRA) reported an approximately 17 % net forest cover loss (including MF, SF and TP) in the tropics between 1990 and 2020 (compared to 29.2 % in this study when including MF, SF and TP), with large differences among countries and regions (FAO, 2020). For example, FRA net forest cover changes (excluding TP) were −22 % in Ghana and −16 % in Brazil (and −30 % in Brazilian Atlantic Forest), while only −5 % in Mexico and a slight gain (<1 %) in Australia (FAO, 2020). Estimates for tree cover increase (i.e., natural forests and TP) are difficult to obtain from FRA because data availability and data detail are inconsistent across time and countries. Remote sensing studies based on

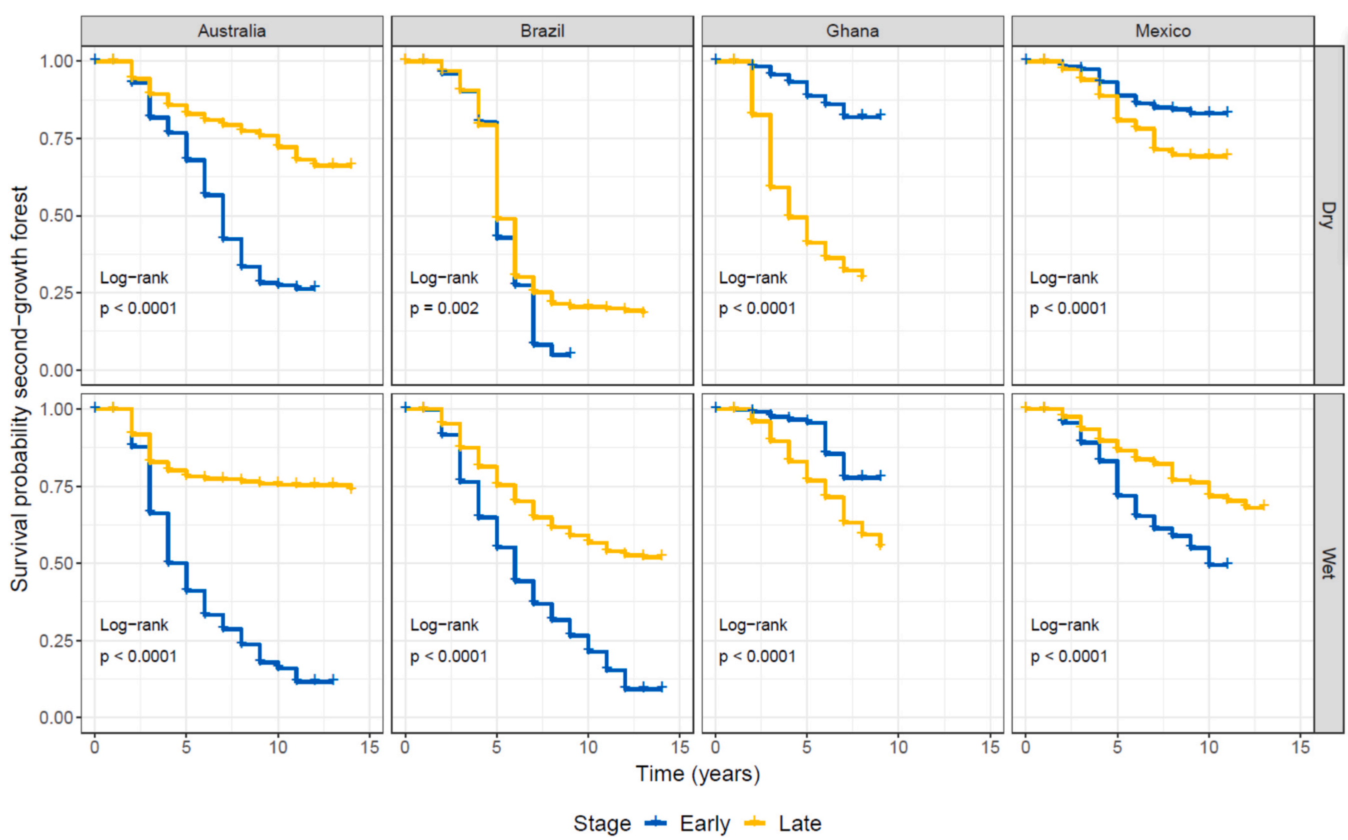

Fig. 5. Shifts in survival probabilities between early and late second-growth forests for each region, including confidence intervals. Sample of 2000 randomly selected pixels per region. Significant difference between early and late second-growth forests per region (Log-rank test for each region).

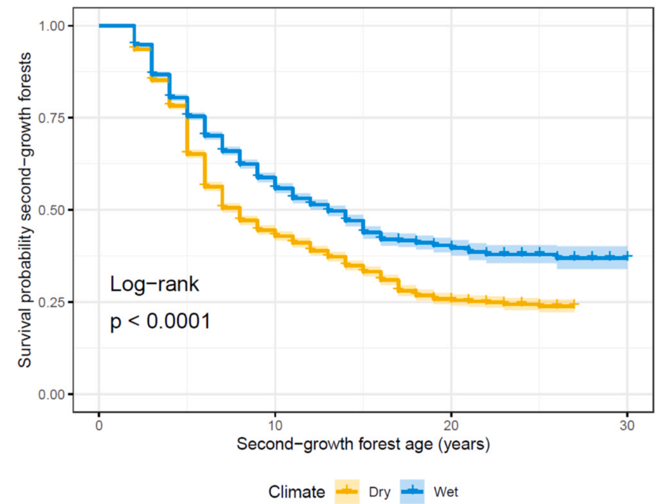

Fig. 6. Probability of second-growth forests survival between dry and wet climate, including confidence intervals. Sample of 8000 randomly selected pixels per climate strata. Significant difference between climates (Log-rank test).

time-series analyses may provide more detailed results on tree cover dynamics. For example, Vancutsem et al. (2021) (Vancutsem et al., 2021) estimated that 17.2 % of tropical MF cover was deforested between 1990 and 2019, and only 13.2 % of the deforested area recovered as natural regrowth. Furthermore, deforestation of MF (without natural regrowth) was high in Ghana (33 %) and Mexico (40 %), whereas Brazil and Australia both experienced 17 % deforestation of MF.

Compared with the 2020 FAO-FRA country reports, our results showed substantially higher forest cover changes (only natural forest cover and not including TP) in the Ghanaian (-42 % and −40 %) and Mexican (-24 % and −50 %) regions (FAO, 2020). Forest cover changes in the Brazilian Atlantic Forest regions (-16 % and −19 %) were in line with the Brazilian national FAO data (-16 %) but different to the Brazilian Atlantic Forest FAO data (-29 %). The Australian regions (+/-1 %) showed similar changes to the FAO results (FAO, 2020). Compared to the national estimates of Vancutsem et al. (2021) (Vancutsem et al., 2021), our results showed higher deforestation of MF in Ghana (51 % and 44 %), Mexico (33 % and 58 %), and Brazil (29 % and 36 %), but lower deforestation in Australia (12 % and 13 %). A comparison of our regional MF losses and the estimates of the Global Forest Change product (Hansen et al., 2013) for our eight regions shows that their forest loss estimates were on average 15 %-point lower than ours. The lower estimates of the Global Forest Change product may partly be explained by their shorter monitoring period (2000–2020) compared to our study (on average 1993–2020). In summary, most of our regions experienced substantially higher forest cover changes (excluding TP) or deforestation than suggested by national reports, indicating that regional forest cover dynamics vary widely within countries and can locally be much higher. Country wide statistics that average out across broad spatial and temporal scales might therefore conceal large differences in forest cover dynamics within countries.

We found that new forest cover (naturally regenerated SF only) increased on average from 0 % to 30 % (range 17 – 46 %) of final total forest cover in 2020. This is in line with other studies showing SF increases throughout the tropical biome (Vancutsem et al., 2021; Rosa et al., 2021; Schwartz et al., 2020). For example, SF cover constituted as much as 2.4 million km (Fagan et al., 2022) (28 %) of Latin America in 2008 (Chazdon et al., 2016). Hence, SF may potentially play an important role for nature (Poorter et al., 2016; Chazdon et al., 2016; Rozendaal et al., 2019) (e.g., for climate, biodiversity, and water) and people (Hua et al., 2022) (for providing goods and services) where FT is taking place or where forest restoration is pursued. Although SF sequester 11 times the CO_2 of old-growth forests and species richness reaches MF levels within 20 years, their aboveground biomass stock and species composition may take many decades to centuries to reach MF levels (Poorter et al., 2016; Rozendaal et al., 2019; Poorter et al., 2021). Therefore, SF cannot be expected to offset carbon emissions from deforestation. For example, between 1984 and 2019, SF and recovering degraded forests counterbalanced only 26 % of the emissions from tropical moist MF loss (Heinrich et al., 2023).

We found that (monoculture) TP cover increased on average from 0 % to 11 % (range 0 – 52 %) of total tree cover in 2020 (Fig. 3; Table 2). In the study regions, these TP were not only for timber (e.g., *Eucalyptus* plantations in Brazil) but also for agricultural commodities (palm oil and rubber in wet Mexico and Ghana, and cocoa in Ghana). Vancutsem et al. (2021) (Vancutsem et al., 2021) found a similar increase in TP (4.7 %), estimates that did not include cocoa, *Eucalyptus* and oil palm plantations, while Fagan et al. (2022) (Fagan et al., 2022) found that the newly gained area of SF and monoculture TP between 2000 and 2012 were similar (31.6 Mha vs. 32.2 Mha, respectively). This demonstrates that TP play an important role in tropical tree cover gain and, for instance, contributes to achieving the goal of restoring 350 Mha with the Bonn Challenge. However, their total extent may be underestimated. The role of TP in biodiversity conservation and the regulation of ecosystem services provision is often low because TP are predominately optimized for production goals (Hua et al., 2022; Bond et al., 2019; Lewis et al., 2019; Malkamäki et al., 2018; Qaim et al., 2020; Quezada et al., 2019).

In summary, mature tropical forest cover has decreased dramatically in the study regions over the past three decades. SF cover increases compensate for some of the MF loss, but simultaneously lead to rejuvenation of forest cover and thus reduces biodiversity conservation and provisioning of ecosystem services (i.e. may reduce forest quality) (Heinrich et al., 2023). Unless SF cover persists for many decades so SF attributes reach MF levels, SF also cannot offset the biodiversity decline caused by MF loss (Rozendaal et al., 2019; Poorter et al., 2021), for which protection of the remaining tropical forests should remain a priority. Establishing plantations or agroforestry systems on former or abandoned pastures and crop fields instead of replacing natural forest cover can relieve pressure on natural forests. The discrepancy between national reports on tree cover dynamics and our results shows the importance of understanding both national and regional-local spatial scales.

4.2. Where are the studied regions on the FT curve?

FT framework postulates that with social and economic development, total tree cover will initially decrease because of the conversion of MF to other types of land use, and then will increase when new tree cover gains outpace forest cover losses. Such a transition facilitates forest (cover) restoration and expansion of TP which can result from five different mutually non-exclusive FT pathways: economic development, state policy, forest scarcity, globalization, and smallholder intensification (Lambin and Meyfroidt, 2010; Marcos-Martinez et al., 2018; Gingrich et al., 2019). When evaluating FT, studies often do not distinguish between types (hence, the quality) of tree cover gain, such as natural regrowth and monoculture TP. However, this has consequences for how to interpret FT when an inflection point is passed and tree cover in a given area increases. Here, we discuss FT in the eight study regions, including and excluding (monoculture) TP cover change.

Our results show that the Ghanaian regions were in an early transition phase, the Mexican wet region was close to the FT inflection point, and the Mexican dry region just past its inflection point. The Australian and Brazilian cases were in the post-transition phase (Fig. 3; Table 2).

4.2.1. Natural forest cover

In the two regions in Ghana, net forest cover loss was high and did not yet slow down. Other Ghanaian regions also showed similar high forest cover losses, confirming our findings that the Ghanaian regions are in early FT phases (Kusimi, 2008; Sarfo et al., 2022a, 2022b), which was also found by other studies focusing on the entire country (Vancutsem et al., 2021; FAO, 2020; Oduro et al., 2015). Both Mexican regions were near the FT inflection point. In the wet region of Mexico net forest cover losses slowed down to close to zero, but MF cover losses continued (Lohbeck et al., 2022; Berget et al., 2021; Vaca et al., 2012). In the dry Mexico region, net forest cover increased slightly, but contiguous MF cover declined. Hence, the dry region of Mexico has passed the FT inflection point, i.e., is in a post-transition phase. Other studies in the same Mexican dry region reported minimal net forest cover losses (~10 %) (Calzada et al., 2018; Figueroa et al., 2020). Compared to the results from national reports (see Section 4.1), our regions show a different picture.

Although the Australian and Brazilian regions passed their FT inflection points, the Brazilian regions showed slightly higher net forest cover gains than the Australian regions did. The Australian regions experienced low net forest cover increases, with MF disturbed by cyclones in Australia's wet region and SF slowly increasing due to pasture abandonment (Sloan et al., 2016). Compared to both regions, trends in all tropical forests in Australia indicate that the Australian moist tropical forest biome is still not expanding (Vancutsem et al., 2021), although different studies present different results (Marcos-Martinez et al., 2018; Calderón-Loor et al., 2021; Evans, 2016). The two Brazilian regions showed net forest cover losses in the first half of the study period, after which these regions reached their FT inflection point and thereafter experienced a net forest cover increase (Fig. 3; Table 2), probably reflecting legal requirements to have 20 % of the land forested, stricter law enforcement, and the result of restoration initiatives such as the Atlantic Forest Restoration Pact. The same trend was observed for the Brazilian Atlantic Forest as a whole (Rosa et al., 2021). Other studies have reported that both the Atlantic Forest and Brazil as a whole have not yet passed their FT inflection point (FAO, 2020; Bicudo da Silva et al., 2023).

4.2.2. Tree cover

When including TP, the interpretations of FT were different for three regions. The Ghana dry region approached its FT inflection point when including increases in cocoa and palm oil plantations. Although Ghana did not pass the FT inflection point, regional differences were observed. In particular, regions with steadily increasing TP are approaching FT inflection (Oduro et al., 2015). The Mexican wet region passed the FT inflection point when oil palm and rubber plantations were included. In this region, palm oil, rubber, and other tree commodities were introduced in the mid-2000s (Lohbeck et al., 2022; Berget et al., 2021). When including *Eucalyptus* plantations, the Brazilian dry region went through FT inflection 10 years earlier because E*ucalyptus* plantations increased as much as SF cover in the same study period (Fig. 3; Table 2). In recent decades, TP, predominantly *Eucalyptus* and pine, has increased in the Brazilian Atlantic Forest region (Rosa et al., 2021).

According to FT framework and modernization theory economic growth at country level should result in relieving pressure on natural systems, including a decline in forest cover loss and increase in tree cover (SF and plantations) (Liebman and Gagliano, 2021; Rudel et al., 2021). Other studies have found evidence that forest cover losses in the last couple of decades primarily occur in lower-income countries and forest cover gains in high-income countries (Estoque et al., 2022; Rodríguez García et al., 2021; Winkler et al., 2021). However, there are exceptions (e.g. Zagros forest in Iran) (Heidarlou et al., 2024). Our results are largely in line with FT framework and evidence found in these other studies, although the difference between the Brazilian and Australian regions is an exception. This may suggest that socioeconomic development, but more importantly the underlying institutional factors and compliance with legislation, can slow down deforestation, whereas rigorous institutional instruments for conservation and restoration can increase forest restoration to bend the net forest cover curve at national, regional or local scales (Lambin and Meyfroidt, 2010; Furumo and Lambin, 2021; Heidarlou et al., 2024; Bhattarai and Hammig, 2001).

Furthermore, deforestation can be displaced from one region or country to another, when natural resource extraction and agricultural production for (inter)national markets move to other countries or regions within countries at the expense of their forest cover (e.g. teleconnections) (Hoang and Kanemoto, 2021; Pendrill et al., 2019; Yu et al., 2013; Corbera et al., 2019). This may lead to FT inflections in (post-)industrialized regions (e.g. Atlantic Forest vs. Amazonia in Brazil) or countries, but at the same time leads to deforestation elsewhere (e.g., from Australia or China to Southeast Asia) (Gingrich et al., 2019; Pendrill et al., 2019; Yu et al., 2013; Meyfroidt et al., 2010). Increased globalization and free market mechanisms have accelerated deforestation displacement in recent decades (Hoang and Kanemoto, 2021; Pendrill et al., 2019).

In summary, our studied regions were in line with the theoretical FT trajectory, but we observed some variations between regions and some differences when excluding or including TP in the evaluation of FT. Furthermore, we observed discrepancies between regional and national FT trajectories, suggesting that effective implementation of restoration and conservation policies should consider regional differences in forest cover dynamics to prevent displacement of deforestation within or between countries.

4.3. Differences between dry and wet forests

Many studies have focused on humid tropical forests because of their high biodiversity, whereas tropical dry forests have been overlooked by researchers and policymakers. Recently, there is an increased interest in tropical dry tree cover changes because of their threatened status, unique flora, and their importance for people's livelihood (Pendrill et al., 2022; Buchadas et al., 2022; Siyum, 2020; Buchadas et al., 2023).

We found that tropical dry mature forest losses were higher than wet mature forest losses and that their overall MF cover was lower as well, but forest cover gains were higher (in %, but not in ha; Fig. 3; Table 2). Others found that, in relative terms, in the past two decades, tropical dry forest cover loss is almost three times higher than moist tropical forest cover loss (Vancutsem et al., 2021; Ocón et al., 2021; Buchadas et al., 2022). These trends suggest that large areas of tropical dry forests are deforestation frontiers too.

An increasing driver of tropical dry forest losses is the expansion of commodity-driven agriculture and intensive land use (Buchadas et al., 2022, 2023). For instance, in Brazil, commodity production (such as E*ucalyptus*, soybean, and sugarcane) is located in the drier regions (especially in the Cerrado and inland Atlantic Forest biomes) (Florêncio et al., 2022; Song et al., 2021; Zheng et al., 2022) because of lower water deficits during the growing season and flat topography that allow for mechanized agriculture (Cattelan and Dall'Agnol, 2018; Granco et al., 2017; Sentelhas et al., 2015). Drier climates have higher soil fertility because of lower rainfall-related nutrient leaching and higher population densities (Sánchez-Azofeifa and Portillo-Quintero, 2011; Sánchez-Azofeifa et al., 2005; Siyum, 2020). In contrast, in the humid tropics, the production of oil palm, rubber, and cocoa is more common, and pastures for cattle grazing are increasing (Meijaard et al., 2020; Priyadarshan, 2017). Overall, the expansion of such TP predominantly occurs in humid tropics (Fagan et al., 2022), possibly because the dominant tree crops favor humid climatic conditions (i.e., palm oil and rubber).

The risk of forest fires is higher in tropical dry forests because of warmer and drier climatic conditions, in combination with higher fuel loads. Fires, but also forest degradation in general, are precursors of human-caused deforestation (Vancutsem et al., 2021; Siyum, 2020). For example, forest cover in the Ghanaian dry region steeply declined in the second half of the 2010s, when farmers entered a forest reserve after widespread forest fires. Although we found similar SF increases in dry and wet forests, a pantropical study found that SF regenerates predominantly in humid tropics (Fagan et al., 2022), probably because of favorable climatic conditions.

In summary, the total forest cover and net forest cover decline (excluding TP) in tropical dry forests was lower than that in tropical wet forests. This is an outcome of intensive land use, such as commodity driven agriculture, anthropogenic (fires) and natural (drought) disturbances. Differences in dry and wet tropical forest cover dynamics may have resulted from different drivers. This is critical to recognize for evaluating FT trajectories in countries. In FT literature, countries are recognized as climatically homogeneous, although this is frequently not the case and thus has implications for understanding FT trajectories. Therefore, we argue that climatic differences within countries should be considered amongst other factors to understand forest cover dynamics and for designing smart policy mixes to bend the curve of forest cover loss.

4.4. The role of secondary forests in the FT curve

SF play an important role in FT, as they expand forest cover and result from complex socioeconomic and demographic processes, or managing and planning land use, which may or may not include abandonment of agricultural fields (Holl et al., 2022, 2023; Jakovac et al., 2021; Crawford et al., 2022). We were able to analyze SF cover dynamics, i.e., persistence and turnover. Our results show that SF are critical in the FT trajectory because over time, their cover increased on average to one-third of net forest cover, and their persistence increased as well (Figs. 3 and 4; Table 2). While this is the case, SF cover survived for a relatively short period (50 % survival probability of 10 years). SF survival in wet forests (13 years) was 1.5 times higher than in dry forests (8 years).

The clearance of young SF result from socioeconomic dynamics; to use the land for agricultural production (in Ghana and Mexico), to claim land ownership (Brazil), and to prevent young forests from maturing and thus become protected by law (in Australia and especially in Brazil) (Rosa et al., 2021; Chazdon et al., 2020; Román-Dañobeytia et al., 2014). Furthermore, fallow agriculture is an important form of low-input subsistence agriculture, where the SF fallow phase is an inherent part of the agricultural system (Martin et al., 2023). During the fallow period, weeds are suppressed and soil fertility recovers, after which the SF fallow is slashed again (Jakovac et al., 2017). Temporal changes in SF persistence can be partially explained by the change in fallow agriculture. Increases in SF persistence can result from agricultural transitions away from fallow agriculture to more permanent and intensive agriculture due to changes in agricultural practices (e.g., decreased fallow time, abandonment of depleted soils or marginal lands) and stricter conservation policies (e.g., protection of forests and forest restoration, land tenure) (Martin et al., 2023; Heinimann et al., 2017; Rasmussen et al., 2018; van Vliet et al., 2012).

SF cover in Latin America shows similar turnover as found in this study, where large proportions of SF cover are short-lived. The average age of SF cover in the Atlantic Forest was 14 years, while ephemeral SF was less than 8 years (Piffer et al., 2022b). Our results of average SF age of the Atlantic Forest regions were higher than those of the Brazilian Amazon (5–8 years) (Jakovac et al., 2017) and Mexican wet tropics (8 years) (Lohbeck et al., 2022), while almost 60 % of SF in the Brazilian (Nunes et al., 2020) and Peruvian Amazon (Schwartz et al., 2017) was cleared after less than 5 years. In Costa Rica, which passed its FT inflection point in the 1980s and is considered a model of environmental sustainability (Jadin et al., 2016), 50 % of the SF patches are still cleared within 20 years (Reid et al., 2019). Scrutinizing SF cover dynamics is critical for understanding the contributions of SF to climate and biodiversity conservation; however data on SF dynamics are not available in most tropical countries. In summary, SF cover has a high turnover rate, and as a result, its contribution to carbon stocks and biodiversity conservation remains low (Figs. 4, 5 and 6) (Piffer et al., 2022b; Nunes et al., 2020; Schwartz et al., 2017).

Conserving SF and increasing their persistence is important, as they have the potential to contribute to biodiversity conservation, climate change mitigation, and provisioning of ecosystem services (Cook-Patton et al., 2020; Strassburg et al., 2020; Williams et al., 2023; Mo et al., 2023). For example, after 20 years, SF attributes attain an average of 80 % of the values of neighboring MF. However, some attributes, such as soil fertility and species diversity, recover fast, whereas other attributes, such as biomass and species composition, recover slower (Poorter et al., 2016; Rozendaal et al., 2019). Similarly, for other biota, such as amphibians, birds, mammals, and reptiles, the recovery of species richness is fast, but the recovery of species composition is slow (Acevedo-Charry and Aide, 2019). Hence, while protecting MF is a priority, it is also crucial to protect and increase the persistence of SF, because the older they are, the more they contribute to biodiversity conservation and the provisioning of regulating ecosystem services (Chazdon et al., 2016). Increasing the persistence of SF can be achieved by introducing and enforcing laws (such as the Brazilian Native Vegetation Protection Law from 2012), economic incentives (e.g. governmental subsidies or societal initiatives, or payment for environmental services), and liberating land (e.g. shifting to more plant-based diets because plant-based food production requires much less land than animal-based food production) (Rosa et al., 2021; Chazdon et al., 2020; Dockendorff et al., 2022; Sun et al., 2022). Furthermore, clear definitions of natural regeneration are essential and, to monitor SF recovery and for effective restoration strategies, concrete indicators and reference values such as recovery of biomass and biodiversity, are important (Vieira et al., 2024). As SF are shaped by people and are part of a socio-ecological system (Balvanera et al., 2021), it is of utmost importance that policies for protecting SF include and benefit local people (Chazdon et al., 2020).

4.5. How to move forward with dissecting the Forest Transition?

We showed that FT is a complex process which is comprised of different dynamics that happen simultaneously, but typically these dynamics are lost in simplified statistics or graphs. To better understand forest transitions and to design effective, equitable and socially just policy strategies to achieve forest restoration goals and facilitate forest transition, it is critical to scrutinize these fundamental dynamics that are underlying FTs. We identified two FT dynamics in this paper: 1) different but conflated types of tree cover gain, and 2) short-lived permanence and high turnover rates of SF. These dynamics are often overlooked in science and policy, although they are fundamental to our understanding of forest transitions and forest restoration, and may have important impact on how policy strategies are designed. Here, we discuss how to move forward with dissecting FT and we discuss potential policy implications and discourses.

In FT literature, forest cover gains, or here tree cover gains, are typically conflating tree cover types that reflect different tree cover qualities (e.g. natural forests and TP) which results in a misunderstanding of tree cover dynamics or forest conditions (Rudel et al., 2016; Sloan et al., 2019; Schwartz et al., 2020; Sloan, 2022; Zhai et al., 2017). Although the FAO definition of forests applies to TP, TP have significantly different contributions to regulating ecosystem services compared to natural forests (Fagan et al., 2022; Hua et al., 2022; Lewis et al., 2019; Holt et al., 2016). In fact, TP are increasingly referred to as commodity crops because they have limited contributions to carbon sequestration and conservation due to their frequently ephemeral nature, focus on optimizing production and profit, and TPs replacing natural forests (Rudel et al., 2016; Sloan et al., 2019; Holt et al., 2016; Sloan and Sayer, 2015). Currently, natural regeneration as a restoration strategy is often invisible (Chazdon et al., 2020), hence, distinguishing tree cover types will contribute to recognizing natural regeneration as a crucial restoration strategy. Excluding TP from the FT may have far-reaching consequences for evaluating at which stage of the FT trajectory countries or regions are and has important implications for restoration efforts. Currently, countries or regions with large TP covers may be past inflection points, whereas they might actually be at an earlier transition stage when TP are excluded from the analysis. For example, Chile's most recent FT relies heavily on TP, although Chile did not go through a natural forest FT yet (Heilmayr et al., 2016).

Seemingly steady increases of tree cover, specifically increasing SF cover, may be misleading, because in reality SF may have high turnover rates. Therefore, we argue to specifically quantify SF cover dynamics in FT assessments and monitoring forest restoration initiatives because high turnover rates impact SF`s contributions to regulating ecosystem services and people.

Until recently, distinguishing tree cover gains in different tree cover types and quantifying their dynamics have been complex. However, this study successfully demonstrated how to distinguish different tree cover types at larger spatial scales and unravel SF cover dynamics resulting from advances in earth observation, time-series analyses, and data availability. Our results clearly show differences in how to interpret tree cover gains in relation to the FT trajectory, and specifically tree cover gains, into natural forests and TP. We stress that distinguishing tree cover types and their dynamics is critical because they make significantly different contributions to nature and people.

Studies suggest tropical deforestation is locked-in an unsustainable land-use regime (e.g. depending on deforestation), because tropical deforestation has continued in the past decades despite international commitments and policy instruments to halt deforestation and stimulate forest restoration (Garcia et al., 2020; Furumo and Lambin, 2021; Russo Lopes and Bastos Lima, 2022). No single driver is responsible for continued deforestation, but instead a self-reinforcing system which operates within a corridor of the probable and makes continuation of the system`s pathway more likely (Garcia et al., 2020; Russo Lopes and Bastos Lima, 2022). Halting deforestation is a critical element for facilitating FTs, however, current policies to halt deforestation have not resulted in sufficiently effective outcomes. Partially, this is a result of a complex mix of policies that often counteract each other and result in a negative impact on forest cover. These policies (or different policy strategies) individually aim to either reduce deforestation (e.g. forest certification programs), stimulate forest restoration (e.g. payment for ecosystem services), or change land-use for other purposes (e.g. subsidies to increase agricultural production) and do not deliberately and holistically address forest cover losses (Furumo and Lambin, 2021; Russo Lopes and Bastos Lima, 2022). Unlocking deforestation lock-ins and transitioning towards sustainable land-use requires more than the current strategy of patching and/or layering policies. Instead, a smart policy mix sequencing policy instruments at the right time is critical (Furumo and Lambin, 2021; Russo Lopes and Bastos Lima, 2022). These policy mixes should be adequately tuned-in with the prevailing forest cover dynamics, as explained here. Transformative sequencing of policies weakens (destabilizes) status-quo vested interests and power structures while concurrently establishing new institutions and governance strategies for sustainable land-use alternatives (Furumo and Lambin, 2021; Russo Lopes and Bastos Lima, 2022), and thus can facilitate forest transition. Here, states play a critical role in formulating long-term visions, creating an enabling environment, and including local people in equitable and just participation processes. Apart from such top-down approach, bottom-up community-based initiatives with access to social capital and local governance may play important roles in forest cover recovery and forest transition as well (Hernández-Aguilar et al., 2021).

In summary, to design policy strategies and instruments for facilitating land use transition and forest restoration we urge future studies to distinguish between different types of tree cover gains and losses when assessing FT trajectories of any given region. Quantifying tree cover gain dynamics, specifically SF dynamics, and their types is essential for understanding FT characteristics, which, as we have demonstrated, may have serious consequences for understanding and evaluating FT in countries and regions. However, for many tropical countries, these data are not available or have limitations and therefore should be a research priority. Furthermore, we urge that policy and restoration initiatives include local people and address issues of equity, justice and power imbalances.

4.6. Limitations methodological approach: remote sensing time-series and land cover classification

Using the AVOCADO algorithm gives a robust phenological baseline to assess forest cover disturbances and regrowth events through time. Using time-series algorithms come with some uncertainties and imitations that we want to briefly discuss here. At the start of a study period it is difficult to distinguish mature forests and old(er) second-growth forests. Frequently, time-series algorithms are limited to the differences between the spectral reflectance of different land cover types. Hence, any land cover type that is similar enough to the values that the time-series algorithm was trained on, may be detected as mature forest (such as older second-growth forest and tree plantations). Therefore, it is critical to select reference forests carefully. This does, however, not prevent misclassifications as we have encountered in this study. One solution, as we have demonstrated in this work, is to filter AVOCADO results with land cover classification maps. Furthermore, AVOCADO allows to change parameter settings (e.g. how many days should a detected regrowth stay regrowth, see for more detail Decuyper et al., 2022) to match the algorithm better to the local conditions and with the questions asked. While we matched parameters as much as possible, it is not always possible to disentangle deforestation from disturbances that caused multi-year effects on the tree vitality (e.g. extreme droughts or storms). These confusions may result in an overestimation of detecting disturbance and regrowth events. While temporal satellite imagery gaps and clouds pose challenges to utilizing time-series algorithms, AVOCADO suffers less from these challenges because change events will be detected with a delay instead of missing the change events altogether.

In our approach, we combined AVOCADO time-series with 2020 land cover classification maps to distinguish different land cover types, especially different tree cover types. We showed that this approach is useful to assess different tee cover dynamics. We acknowledge that this study would benefit from more robust analyses to exactly estimate how uncertainties and errors propagate from one product to the other. Although uncertainties and misclassifications remain, the errors are expected to be limited.

5. Conclusions

The studied regions followed the theoretical FT trajectory, however, with some exceptions. Overall, mature forest cover declined by more than half (from 25,544 ha to 11,741 ha per study region) and total forest cover declined by almost 40 % (from 25,544 ha to 15,963 ha per study region) within three decades, and these declines were higher in tropical wet forests than in tropical dry forests. SF cover substantially increased in all areas, but this new forest cover increase was not enough to compensate for the lost areas of mature forest. Although SF age increased over time, SF were short-lived, because of high turnover rates. Therefore, SF were limited in contributing to carbon sequestration, biodiversity conservation, and other regulating ecosystem services provisions. Tree plantation cover increased steadily over time. Given that FT is the outcome of different underlying processes, we urge future research to distinguish between different tree cover types (natural forests and TP) and unravel SF dynamics. Evaluating FT with only MF and SF, thereby excluding TP, and their dynamics likely results in different interpretations of socioecological outcomes as well. This may have far-reaching consequences for restoration and afforestation policies and initiatives because we consider not only tree cover quantity but also quality. Our study and other studies increasingly show that SF are predominantly short-lived; consequently, their contribution to biodiversity and ecosystem services is limited. Dissecting FT, as we propose here, increases our understanding of its underlying forest cover dynamics, that can lead to better policies to halt deforestation and achieve forest restoration, and to restore the natural world upon which peoples` lives and society depend.

CRediT authorship contribution statement

Jakovac Catarina C.: Writing – review & editing, Methodology, Conceptualization. **Brancalion Pedro H.S.:** Writing – review & editing, Funding acquisition, Conceptualization. **Amissah Lucy:** Writing – review & editing, Conceptualization. **Martínez-Ramos Miguel:** Writing – review & editing. **Bongers Frans:** Writing – review & editing, Funding acquisition, Conceptualization. **Lohbeck Madelon:** Writing – review & editing, Methodology, Conceptualization. **Veenendaal Elmar:** Writing – review & editing, Conceptualization. **Meave Jorge A.:** Writing – review & editing, Conceptualization. **Bartholomeus Harm:** Writing – review & editing, Methodology, Conceptualization. **de Jong Johan:** Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Funding acquisition, Data curation, Conceptualization. **Laurance Susan G.W.:** Writing – review & editing. **Poorter Lourens:** Writing – review & editing, Writing – original draft, Validation, Supervision, Methodology, Funding acquisition, Conceptualization. **Brown William Hagan:** Writing – review & editing. **de Jong Wil:** Writing – review & editing, Conceptualization. **Decuyper Mathieu:** Writing – review & editing, Validation, Software, Resources, Methodology.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.landusepol.2025.107545.

Data availability

Data will be made available on request.

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