



# The forest transformation: Planted tree cover and regional dynamics of tree gains and losses

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## ABSTRACT

Extensions of forest-transition theory to the tropics often depict sustained expansions of planted tree cover and corresponding long-term net gains in total tree cover. To explore the patterns and implications of continued tropical planted tree-cover expansion, we profiled sequences of tree-cover change over 1990–2010 according to Landsat imagery for recently observed (ca. 2014) planted tree-cover areas in 11 tropical countries. Alternative patterns of change emerged from these analyses. Termed the ‘reforestation treadmill’ and ‘forest transformation’ narratives, planted tree-cover change featured relatively ephemeral planted covers, modest net gains, and similar tree-cover change dynamics compared to nearby agricultural-forest mosaics. Planted areas were characterised not by unambiguous reforestation but rather combinations of tree-cover losses and gains, with losses typically being more prominent. Contemporary gains and losses during 5–10-year periods regularly distinguished planted areas from non-planted areas, with losses being 1.5–2.3 times more common than gains. Planted areas were only moderately distinguishable from non-planted areas overall with respect to tree-cover change dynamics. Relationships between tree-cover change and the export orientations of planted tree/tree-crop commodities were also examined. Greater export orientations did not significantly associate with tree-cover loss or larger planted patches, with partial exceptions for Southeast Asia. Regional disparities in planted tree-cover dynamics were apparent. In Southeast Asia, dominated by Indonesia, tree-cover declines in planted areas since 1990 were relatively pronounced (20% of planted areas), particularly with respect to progressive transitions from tree cover to cleared lands. Planted areas there were generally indistinguishable from nearby non-planted areas with respect to historical tree-cover change dynamics. In contrast, in South America, dominated by Brazil, tree-cover increases in planted areas since 1990 were more appreciable (at least 14% of planted areas), with most being progressive, stable, ‘net’ increases (10% of planted areas) and the remainder being dynamic increases entailing short-term losses since 1990 (4% of planted areas). Total tree-cover increases within South American planted areas were equal to or greater than total decreases since 1990. These patterns suggest a forest-transformation narrative in which major planted-area expansion occurs alongside minor net tree-cover change. This narrative appears particularly well suited to Southeast Asia, where planted areas are extensive and expansive but where net tree cover gains are tenuous, reflecting political-economic shifts in forest management and the devaluation of extensive, degraded natural forests.

## 1. Introduction

Planted tree cover is increasingly prominent in the tropics as natural forest cover continues to decline. According to FAO estimates, tropical planted tree cover expanded 87% since 1990 (Keenan et al., 2015) and accounts for 3.2% of total tropical forest area (Payan et al., 2015). In

Southeast Asia, planted forests are reportedly more extensive than naturally regenerating forests (FAO, 2015). Such figures are doubtless under-estimates due in part to their exclusion of various agro-forestry and small-holder forestry systems (Schnell et al., 2015; Midgley et al., 2017). Planted tree-cover expansion is expected to continue due to looming demand shortfalls for forestry and agro-forestry products

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(Carlson et al., 2012; Barua and Lehtonen, 2013; Payn et al., 2015) but also its congruence with various large-scale reforestation schemes (Sloan, 2016; Lewis et al., 2019; Rudel et al., 2019).

Recent expansion of planted cover evokes imagery of new, enduring tree cover. Whether this is the case generally is unknown. To some degree, this outcome depends on the extent to which planted tree-cover expansion has aligned with agriculture, which is widely responsible for tropical deforestation (Gibbs et al., 2010). Planted-cover expansion may, for instance, entail land-use systems or economic processes of landscape change common to agriculture (Sloan, 2016), particularly as many tropical planted areas are dominated by agro-forestry (Petersen et al., 2016; Hurni et al., 2017) that has increasingly adopted intensive monocultural practices (Clough et al., 2016). This conceptualization of planted tree cover recalls Grainger's (1995) land-use transition, a precursor of the forest-transition, in which tree-cover changes in planted areas increasingly depart from the deforesting tendencies of associated agriculture to yield enduring gains in forest cover. A view of tropical planted tree cover as associated with agriculture expansion has emerged in several recent studies (Altamirano et al., 2013; Van Holt et al., 2016) and if generally true would qualify planted-cover expansion as a marginal contributor to stable net reforestation generally. This possibility has nonetheless remained under-appreciated in the forest-transition literature, which has focused on aggregate forest gains and paid relatively little attention to agro-forestry (Rudel et al., 2005; Sloan and Sayer, 2015; Sloan et al., 2019a). Regional transitions from natural to planted tropical tree cover over the last three decades remain unprecedented, but under-studied, aspects of forest change and apparent expansion.

Narratives of tropical forest gain, steeped in forest-transition theory (Rudel, 2009), may fail to reveal critical, dynamic aspects of tropical planted tree-cover expansion. The forest-transition narrative of enduring planted tree cover and its contribution to net forest gains was recently qualified by the reforestation 'treadmill' narrative (Rudel et al., 2016). Contrary to notions of monotonic gains of stable planted cover over long-cleared lands in response to forest scarcity, the treadmill narrative characterises planted covers as 'commodity crops' by which tree-cover gains are relatively dynamic, partially aligned with agricultural activities, and often entail forest conversion. According to this narrative, tropical tree-cover gains are increasingly recurrent but make limited contributions to stable net gains because planted tree covers, while expansive, are ephemeral and progressively replace natural forests (Sloan and Sayer, 2015). Buoyed by political-economic conditions and tropical climates promoting rapid tree reproduction for globalised markets, industrial-scale interests have been advanced as the primary agents of the treadmill, in keeping with trends in enterprise-driven deforestation (Rudel, 2007; Austin et al., 2017a; Curtis et al., 2018) and corporate 'green grabs' (Hall, 2011; Scheidel and Work, 2018; Sloan et al., 2019b). Over time, the treadmill of forest loss and serial tree reproduction would result in *forest transformations*, characterised by extensive new planted tree cover but modest, unstable net tree cover gains within planted areas alongside appreciable forest losses (Hansen et al., 2013; Tropek et al., 2014; Miranda et al., 2015; Rudel et al., 2016). The potential for such transformations is underscored by an increasing integration of commercial agro-forestry and forestry with multi-lateral reforestation programmes targeting tens of millions of hectares for climate-change mitigation and tropical forest restoration (Laestadius et al., 2015; Sloan, 2016; Lewis et al., 2019; Rudel et al., 2019).

The treadmill and forest-transformation narratives depart from the classical forest transition but the evidence for these narratives remains inconclusive. Most large-scale studies of tropical forest gain have conflated planted and natural forest covers (Hansen et al., 2013; Stibig et al., 2014; Zomer et al., 2014) or they employed crude measures of planted areas (Aide et al., 2013; Sloan and Sayer, 2015; Wilson et al., 2017). Provided satellite data on planted tree cover, most such studies are confined to direct transitions from natural to planted tree cover for

a single brief period, specific commodity, and particular region (Gutiérrez-Vélez et al., 2011; Li and Fox, 2012; Gaveau et al., 2016b; Petersen et al., 2016; Van Holt et al., 2016; Austin et al., 2017b; Furumo and Aide, 2017; Hurni et al., 2017). Collectively, such studies tenuously indicate net tree-cover gains within planted areas. The variability and place-specific nature does, however, make it hazardous to generalize their findings to the globe or across regions. Further, their disregard for longer-term dynamics limits conceptual insights regarding how planted-cover expansion proceeds amidst broader, dynamic processes of forest change.

### 1.1. Towards a fuller understanding of tropical planted tree-cover expansion

A generalised understanding of the role of planted tree cover requires long-term, pan-tropical observations of the pathways to planted tree cover. To this end, with a view to elaborate the treadmill and forest-transformation narratives, we explore three key aspects of tree-cover change over 1990–2010 within recent (2014) planted areas pan-tropically:

- *Land-change dynamism*: To what degree are planted areas characterised by simple, monotonic versus dynamic sequences of tree-cover change?
- *Agricultural alignment*: How similar are planted areas to surrounding agricultural-forest mosaics in terms of their tree-cover change dynamics?
- *Commercial nature*: To what degree do export-oriented planted areas associate with unique tree-cover change dynamics, particularly those characterised by losses?

Regarding *land-change dynamism*, a majority of simple, monotonic sequences of increasing tree cover in the absence of losses would indicate stable planted covers and positive contributions towards net reforestation, as per forest-transition narratives. Conversely, dynamic tree-cover changes since 1990, marked over time by alternating tree-cover gains and losses, would indicate ephemeral planted tree covers and suggest possible forest conversion. Such dynamism would accord with treadmill and forest-transformation narratives and undermine the environmental goals of multi-lateral reforestation programmes, which depend on persistent new tree cover over cleared lands (de Jong, 2010; Körner, 2017; Lewis et al., 2019). A disregard of the dynamism of tree-cover change within planted areas (Gibbs, 2012; Meyfroidt et al., 2014; Sloan and Sayer, 2015) would conflate ephemeral gains with enduring gains, as well as temporary losses with longer-term deforestation. Such disregard would also discount 'indirect deforestation' driven by speculation over *later* plantation reforestation (World Bank, 2006; Hoang et al., 2010). In this light, apparent net tree-cover increases due to planted-cover establishment are doubtless inflated (e.g., Altamirano et al., 2013; Gaveau et al., 2016b) and apparent forest-transition trajectories may be otherwise (Zhai et al., 2017). The treadmill narrative anticipates that dynamic sequences of tree-cover increase within planted areas predominate over simple, monotonic increases but only marginally exceed dynamic decreases (*Hypothesis 1*).

The *agricultural alignment* of planted areas, defined as the similarity of their tree-cover change dynamics compared to nearby agricultural-forest mosaic areas, bears on whether a forest transformation or forest transition is more likely. A high similarity between planted areas and surrounding agricultural-forest mosaics would indicate an 'alignment' of their land-change dynamics, if not their land-use systems, rendering planted areas less likely to make enduring contributions towards net reforestation generally. Similarities are likely where planted areas encompass or associate with agricultural activities (Williams, 1986; Farley, 2007; Cramb and Curry, 2012; Potter, 2012; Gaveau et al., 2016a; Sloan, 2016; Sloan et al., 2017); are highly industrialised, e.g., entailing short rotations or limited canopy cover; or where agricultural

areas host recurrent forest regrowth, as due to long fallows, land abandonment, or forest degradation (Sloan, 2016; Reid et al., 2018). The treadmill narrative anticipates that planted areas exhibit similar tree-cover change dynamics compared to agricultural-forest mosaics, albeit with more prevalent tree-cover gains (*Hypothesis 2*).

Regarding the *commercial nature* of tree-cover change, the treadmill narrative expects that planted areas with greater export orientations are more likely to exhibit unique tree-cover change dynamics, centered on tree-cover losses, particularly amongst larger-scale planted patches (*Hypothesis 3*). Such a possibility has been implied by case studies (Gutiérrez-Vélez et al., 2011; Meyfroidt et al., 2014; Sloan, 2016; Scheidel and Work, 2018; Sloan et al., 2019b) and draws upon associations between deforestation and the prices of planted tree commodities in global markets (Williams, 1986; Gaveau et al., 2019). Conversion of degraded forests into planted tree cover endows commercial plantation enterprises with subsidies or efficiencies often crucial to their viability (Barr, 2001a; Gutiérrez-Vélez et al., 2011). Commercial enterprises in turn are relatively responsive to reforestation incentives (Sloan, 2016; Rudel et al., 2019) and able to negotiate access to State production forests (Resosudarmo et al., 2019). The pan-tropical prevalence of industrial-scale planted areas (Petersen et al., 2016) and, increasingly, of larger tropical forest clearings (Austin et al., 2017a), accords with the treadmill and forest-transformation narratives.

Focussing on these key aspects of tree-cover change in planted areas, we introduce the forest-transformation narrative as a complement to the forest transition, with emphasis on Southeast Asia. This narrative generalises land-change and political-economic processes by which significant planted-cover expansion has yielded only limited net tree-cover gains in planted areas and, by extension, limited contributions to reforestation generally. The narrative elaborates emergent perspectives of tropical reforestation as increasingly centrally-organised, accelerated, and centered on planted tree covers (Lewis et al., 2019; Rudel et al., 2019). It also reframes earlier notional 'Asian forest transitions' (Mather, 2007) and 'strong State' reforestation pathways (Lambin and Meyfroidt, 2010), as by underscoring the political-economic basis for limited long-term contributions of planted areas to general reforestation. The narrative likewise supports remote-sensing observations of elevated deforestation in Southeast Asian planted areas (Petersen et al., 2016).

While we discuss the juxtaposition of the forest transformation and forest transition, we do not test the forest-transition model per se. Its particular contexts, drivers, and timeframes are beyond our scope and have been explored elsewhere (Rudel, 2005; Sloan, 2015; Van Holt et al., 2016; Sloan et al., 2019a). Nor do we seek to conclusively quantify the proportion of recent planted tree cover that replaced natural forests pan-tropically. Observations of strictly *natural historical forest cover* are beyond the scope of available pan-tropical satellite data (Sloan, 2012; Feng et al., 2016; Curtis et al., 2018) (Text S5). We focus therefore on the aforementioned aspects of historical *tree-cover* change within recent planted areas, with attention to indicative dynamics, to refine emergent narratives of tropical planted-cover expansion holding implications for broader forest change.

## 2. Methods

### 2.1. Pan-tropical analyses of tree-cover changes

#### 2.1.1. Land-change dynamics in planted areas

To explore the land-change dynamics in planted areas (*Hypothesis 1*, Section 3.1), sequences of tree-cover change over 1990–2010 according to successive Landsat classifications were summarised for 46.7 million hectares of planted areas across 11 tropical countries (Brazil, Cambodia, Colombia, Indonesia, Laos, Liberia, Malaysia, Panama, Peru, Thailand, Vietnam) (Table 1). Planted areas were delineated as of ca. 2014 – the sole year for which their pan-tropical extent is explicitly

observed – such that sequences of tree-cover change outline preceding historical trends for 1990–2010. The 11 countries were selected on the basis of the availability of wall-to-wall spatial data on recent planted tree cover (Table 1). These countries encompass the vast majority of tropical planted extent whilst also including some smaller countries.

Tree-cover change over 1990–2010 was initially observed for three periods separately (1990–2000, 2000–2005, 2005–2010) according to the Landsat classifications of Kim et al. (2015). For each period, one of four tree-cover change classes were defined: *tree-cover gain*, *tree-cover loss*, *persistent tree cover*, or *persistent non-tree cover*. These periodic change classes were classified according to their respective probabilities, derived from estimated percent tree cover at each year bounding each period (1990, 2000, 2005, 2010) (Text S1). Tree cover was defined as parcels of > 1 ha comprising pixels with >30% tree cover.

Planted areas include virtually all forms of planted tree cover, as for timber/fibre, agro-forestry, and agro-industrial tree-crop production (Tables 1 and 2). Visual interpretation of Landsat imagery for 2013/2014 informed manual delineations of planted areas as landscapes of >50% planted cover in seven main countries (Brazil, Cambodia, Colombia, Indonesia, Liberia, Malaysia, Peru) (Petersen et al., 2016; GFW, 2017) accounting for 98% of the total planted extent observed. Landsat interpretations were guided by visual interpretations of coincident high-resolution satellite imagery as well as by field surveys of 7800 sites in Brazil, Indonesia and Malaysia (Petersen et al., 2016) (Text S2.1), which host the vast majority of the planted extent (Table 1). The high-resolution imagery spanned 3% of the seven main countries and had ample coverage of most planted areas. Recently cleared and very young planted areas were included amongst planted areas, accounting for 5–22% of planted areas in six of the seven countries and 35% in Liberia (Figure S1). For Brazil and Cambodia, visual interpretations were complemented by classifications of Landsat (2013/2014) and MODIS (2010) imagery, respectively (Li and Fox, 2012; Petersen et al., 2016). In the remaining four countries, planted areas were classified using Rapideye (5-m) and MODIS (250-m) satellite imagery for 2010–2012 (Li and Fox, 2012; Kindgard, 2013; Ministry of Environment, 2015), as visual interpretations were not available. No minimum planted area patch size was specified for the seven main countries. Patches in these countries were as small as < 5 ha (Figure S2), with such patches generally part of larger planted mosaics or estates. A minimum size of ~5 ha was observed the four other countries (Figure S2, Text S2).

For the analysis of land-change dynamics (*Hypothesis 1*, Section 3.1), all 27 valid temporal sequences of the periodic tree-cover change classes were collapsed into four meta-classes defining sequences of simple or dynamic increase or decrease of tree cover across 1990–2010 within recent planted areas. Sequences of *simple increases/decreases* describe monotonic, stable, net increases/decreases in tree cover at the Landsat pixel scale across the three periods spanning 1990–2010. Simple increases feature a periodic gain in tree cover or a transition to tree cover across periods according to Kim et al. (2015), where the new tree cover is not subsequently lost by 2010. For example, a simple increase may be defined by sequences such as *persistent non-tree cover*→*persistent tree-cover gain*→*persistent tree cover*, or *persistent non-tree cover*→*persistent non-tree cover*→*persistent tree cover*. Simple-decrease sequences are analogous, except that they feature a periodic loss or transition to non-tree cover. In contrast, sequence of *dynamic increase/decrease* entail 'turnover' marked by periodic reversals of tree-cover change trends at the pixel scale. These dynamic sequences terminate in 2010 as either gain/tree cover in the case of increases, or as loss/non-tree cover in the case of decreases. For example, a dynamic increase may be defined by sequences such as *persistent tree cover*→*tree-cover loss*→*tree-cover gain*, or *persistent tree cover*→*persistent non-tree cover*→*persistent tree cover*. Dynamic-decrease sequences are again analogous.

#### 2.1.2. Similarity of tree-cover change in planted and non-planted areas

The similarity of planted areas and surrounding non-planted

**Table 1**  
Coverage and sources of planted areas and non-planted areas for 11 tropical countries.

Country	Sample Points <sup>a</sup>		Extent (ha) <sup>b</sup>		Planted-Area Characteristics <sup>c</sup>	Planted Area Mapping Method <sup>d</sup>	Planted Area Patch Size Observed	Satellite Imagery & Data Source
	Planted	Non-Planted	Planted	Non-Planted				
<i>Central and South America</i>								
Brazil	77,594	967,224	9,522,838	124,756,922	Predominantly timber/fibre stands; complemented with large fruit; predominantly large and corporate planted areas	VI with IC	Yes	Landsat (GFW, 2017)
Colombia	3,543	27,077	498,570	3,805,650	Mix of oil palm, fruit, and other/unknown stands; predominantly large and corporate planted areas	VI	Yes	Landsat (GFW, 2017)
Panama <sup>e</sup>	373	8,581	99,254	2,549,132	Mostly broadleaf timber stands, plus smaller areas of oil palm, banana, coffee, and coniferous stands	IC	No	RapidEye (Ministry of Environment, 2015) Landsat (GFW, 2017)
Peru	611	5,143	103,771	771,899	Mostly oil palm and other/unknown areas; mostly large or corporate areas complemented by medium- and small-sized stands	VI	Yes	Landsat (GFW, 2017)
<i>Southeast Asia</i>								
Cambodia <sup>f</sup>	8,242 (3,543)	33,837 (23,711)	979,713 (914,610)	4,469,536 (2,969,564)	Predominantly rubber stands; mostly medium-sized planted areas	VI with IC (VI only)	Partially (Yes)	Landsat (GFW, 2017); MODIS
Laos	1,058	35,981	132,665	4,230,130	Predominantly rubber stands; conservative delineation	IC	No	Li and Fox, 2012) MODIS classification (Li and Fox, 2012)
Thailand	2,577	56,586	376,687	8,867,608	Predominantly rubber stands; conservative delineation	IC	No	MODIS classification (Li and Fox, 2012)
Vietnam	1,608	31,666	273,096	5,859,228	Predominantly rubber stands; conservative delineation	IC	No	MODIS classification (Li and Fox, 2012)
<i>Insular Southeast Asia</i>								
Indonesia	83,410	1,24,123	24,599,993	48,830,422	Mostly oil palm complemented by rubber and timber/fibre stands; mostly large and corporate planted areas, complemented and interspersed by medium-sized and small-sized planted areas	VI	Yes	Landsat (GFW, 2017)
Malaysia	37,946	50,705	10,010,754	13,433,892	Predominantly oil palm; mostly large and corporate planted areas, complemented and interspersed by medium-sized and recently-cleared planted areas	VI	Yes	Landsat (GFW, 2017)
<i>Africa</i>								
Liberia	305	2,567	145,619	748,281	Largely unknown tree species; mostly large and corporate planted areas, with sizeable extent of recently-cleared planted areas	VI	Yes	Landsat (GFW, 2017)
<b>Total</b>	<b>213,724</b>	<b>1,319,779</b>	<b>46,742,960</b>	<b>218,322,700</b>				

Notes:

- <sup>a</sup> Sampled observation points, excluding illogical tree-cover change sequences over 1990–2010 indicative of Landsat misclassification, as well as built-up areas and cloud, shadow, water, and ‘no data’ pixels.  
<sup>b</sup> Total delineated extents of planted areas and non-planted areas defining the spatial sampling frame.  
<sup>c</sup> Planted-area characteristics were informed by Petersen et al. (2016). See Table 2 for list of planted commodities.  
<sup>d</sup> ‘VI’ is visual interpretation, ‘IC’ is image classification.  
<sup>e</sup> Planted-area observations in Panama are relatively few compared to the total national planted extent due to widespread cloud cover and no-data values in the Landsat data.  
<sup>f</sup> Two overlapping Cambodian planted-area datasets were defined, shown here (Text S2.2). For the more extensive of the two, planted areas were delineated by visual interpretation and an independent semi-automated classification. For the less extensive, planted areas were delineated exclusively by visual interpretation. The latter extent was considered by analyses referencing planted patch size class (Table 3 Analyses 2).

**Table 2**

Tree/tree-crop commodities and products for the estimation of export orientation.

Source: Tree/tree-crop commodities are as per [GFW \(2017\)](#), described by [Petersen et al. \(2016\)](#).

Planted-Area Type	Commodity	Products	
		According to Production Records	According to Trade Records
Agro-Forestry	Banana	Banana	Banana
	Cacao	Cacao beans	Cacao beans; power and cake; paste; butter
	Cashew	Unshelled nuts	Unshelled nuts; shelled nuts
	Citrus Fruit	Lemons and limes; oranges; tangerines, mandarins, and clementines; grapefruits	Lemons and limes; oranges; tangerines, mandarins, and clementines; grapefruits
	Clove	Cloves	Cloves
	Coconut palm	Coconuts	Coconuts; desiccated coconut; copra oil; copra
	Coffee <sup>a</sup>	Green coffee; roasted coffee	Green coffee; roasted coffee
	Fruit <sup>b</sup>	Apples; avocados; bananas; fruit, tropical fresh not nes <sup>c</sup> ; fruit, fresh nes <sup>c</sup> ; mangos, mangosteens, guavas; oranges; papayas; persimmons	Apples; avocados; bananas; fruit, tropical fresh not nes <sup>c</sup> ; fruit, fresh nes <sup>c</sup> ; mangos, mangosteens, guavas; oranges; papayas; persimmons
	Hevea	Natural rubber; dry natural rubber	Natural rubber; dry natural rubber
	Mixture <sup>d</sup>	Average	Average
	Oil palm	Palm oil; oil palm kernel	Palm oil; oil palm kernel
	Palms (mix) <sup>e</sup>	Coconuts; oil palm kernel; palm oil	Coconuts; desiccated coconut; copra oil; copra; oil palm kernel; palm oil
	Pecan	Nuts nes <sup>c</sup>	Nuts nes <sup>c</sup>
Forestry	<i>Acacia</i>	Roundwood (non-coniferous); sawnwood (non-coniferous); wood panels; pulp wood	Roundwood (non-coniferous); sawnwood (non-coniferous); wood panels; pulp wood
	<i>Araucaria</i>	Roundwood (coniferous); sawnwood (coniferous); wood panels; pulp wood	Roundwood (coniferous); sawnwood (coniferous); wood panels; pulp wood
	<i>Areca</i> <sup>f</sup>	—	—
	Bamboo <sup>f</sup>	—	—
	<i>Eucalyptus</i>	Roundwood (non-coniferous); sawnwood (non-coniferous); wood panels; pulp wood	Roundwood (non-coniferous); sawnwood (non-coniferous); wood panels; pulp wood
	Coniferous (general)	Roundwood (coniferous); sawnwood (coniferous); wood panels; pulp wood	Roundwood (coniferous); sawnwood (coniferous); wood panels; pulp wood
	Mixed trees <sup>g</sup>	Roundwood (non-coniferous); sawnwood (non-coniferous); wood panels; pulp wood	Roundwood (non-coniferous); sawnwood (non-coniferous); wood panels; pulp wood
	<i>Paraserianthes</i>	Roundwood (non-coniferous); sawnwood (non-coniferous); wood panels; pulp wood	Roundwood (non-coniferous); sawnwood (non-coniferous); wood panels; pulp wood
	Pine	Roundwood (coniferous); sawnwood (coniferous); wood panels; pulp wood	Roundwood (coniferous); sawnwood (coniferous); wood panels; pulp wood
	Teak	Roundwood (non-coniferous); sawnwood (non-coniferous); wood panels; pulp wood	Roundwood (non-coniferous); sawnwood (non-coniferous); wood panels; pulp wood

Notes:

<sup>a</sup> Planted areas in coffee largely pertain to 'sun-grown' coffee, as shade coffee grown under the canopy of mature trees of irregular spacing and age would often be undetected.<sup>b</sup> Products vary by country.<sup>c</sup> 'nes' means 'not elsewhere specified' amongst national production or export product statistics.<sup>d</sup> Observed in Indonesia only. The corresponding export orientation was the average of those defined for all agro-forestry and forestry commodities in Indonesia.<sup>e</sup> This class pertains to generic, undifferentiated planted palms. Export orientation was estimated on the basis of statistics for either oil palm and coconut palms combined, oil palm only, or coconut palms only, depending on the mix oil palm and coconut palm production in a given country.<sup>f</sup> Assumed to have a nil export orientation. No trade statistics were available.<sup>g</sup> Observed in Brazil only. Assumed the average export orientation of all other Brazilian broadleaf forestry commodities.

agricultural-forest mosaic areas pertains to the degree to which these two areas may be distinguished from one another with reference to their historical tree-cover change dynamics (*Hypothesis 2, Section 3.2*). Similarity was assessed with reference to the tree-cover change classes of [Kim et al. \(2015\)](#), observed separately for the three periods spanning 1990–2010, for both planted areas and non-planted areas across the 11 countries.

Non-planted areas are mosaics of agriculture, pasture, scrub, and natural forest covers within 5 km of planted areas. Built areas (e.g., villages, roadside settlements) observed as of 2014 via Landsat Global Human Settlement Layer smoothed to 1-km resolution were excluded ([Pesaresi et al., 2016b, 2016a](#)). Visual interpretations of non-planted areas using high-resolution imagery indicate that they host natural and managed forest covers (e.g., fallows, degraded forests, agro-forestry) but virtually no planted tree cover (*Text S2.1*). Persistent tree cover during 1990–2010 according to [Kim et al. \(2015\)](#) covered 26% of non-planted areas, with the remainder being cleared or transitioning between tree cover and non-tree cover.

A classification tree determined the similarity of planted and non-

planted areas by classifying these areas as such on the basis of their periodic tree-cover change classes of [Kim et al. \(2015\)](#) as well as country (*Table 3 Analysis 1*). A poorer classification accuracy indicates a greater conflation of planted and non-planted areas and thus greater similarity. By partitioning observations into increasingly homogenous subsets of planted or non-planted areas, the tree also inductively determined those combinations of tree-cover changes that best characterise planted and non-planted areas and so merit comparison (*Section 2.5*). According to the treadmill and forest-transformation narratives, these combinations should prove comparable amongst planted and non-planted area, though planted areas should still feature periodic gains and/or transitions to tree cover relatively prominently, as via the dynamic sequences of tree-cover increase described above. The country variable allowed the tree to branch between countries exhibiting distinct dynamics or chronologies. It also served as proxy for national differences in economic incentives for reforestation or forest conversion ([Rudel, 2009](#)).

**Table 3**  
Pan-tropical and national classification/regression tree analyses.

Analysis	Pan-Tropical Analysis			National Analysis			
	Premise	Response variable	No. Countries	Scale of Observation	Notes	Response variable	Predictor variables
1	Planted and non-planted areas have similar tree-cover change dynamics ( <i>Hypothesis 2</i> )	Area type (planted vs. non-planted)	11	Sample point	Cambodia planted area (and corresponding non-planted area) are defined by Landsat interpretation and MODIS classification 'Recently cleared / young plantations' omitted	Area type (planted vs. non-planted)	Tree-cover change class by period <sup>a</sup>
2	'Tree-cover loss is driven by high export orientations in planted areas ( <i>Hypothesis 3</i> )	Export orientation of planted tree/tree-crop commodities	7	Sample point, except for export orientation, which is defined per commodity per country, with commodities observed per sample point			

Note:.

<sup>a</sup> 'Tree-cover change class by period' comprises four classes (gain, loss, persistently treed, persistently non-treed) observed separately for three periods (1990–2000, 2000–2005, 2005–2010).

### 2.1.3. Commercial nature of tree-cover change in planted areas

To explore the commercial nature of tree-cover change in planted areas (*Hypothesis 3*, Section 3.3), we used a regression tree to associate the export orientations of planted tree/tree-crop commodities with the tree-cover change classes of Kim et al. (2015), observed for the three periods of 1990–2010, as well as planted tree cover patch size (small < 10 ha, medium 10–100 ha, large-industrial 100–1000 ha, corporate estate > 1000 ha; Table 4), plantation type (forestry, agroforestry), and country (Table 3 Analysis 2).

Export orientation is a continuous index of the degree to which a planted tree/tree-crop commodity is oriented towards remunerable international markets. For 23 tree/tree-crop commodities (e.g., coffee, *Acacia*, rubber, fruits, oil palm) (Table 2), this index is defined as the fraction of national production dedicated to international markets, weighted by export price. The measure thus captures both international market demand and export revenues. Tree/tree-crop commodities in planted areas in the seven main countries were identified and delineated by the aforementioned visual interpretations of Landsat and high-resolution imagery guided by field surveys. The export-orientation index is detailed more fully below.

The plantation-type variable distinguished between agro-forestry and forestry planted areas on the basis of commodity (Table 2). The forestry class encompasses ten commodities dedicated to timber/fibre production (e.g., *Acacia*, pine, *Eucalyptus*), while the 'agro-forestry' classes encompasses all other commodities dedicated to tree crops (e.g., citrus, banana). This differentiation reflects the possibility that agro-forestry areas have greater export orientations and ephemeral tree cover, given their dedication to relatively higher value products (e.g., coffee, fruits) and shorter rotations.

Visual image interpretation in the seven main countries defined the four planted-area patch size classes (Table 4). These indicate the scale of planted land uses but not necessarily of land ownership. Patches of a given class are often distributed in extensive configurations and one class may juxtapose another or nest within it. The size classes exclude recently cleared / very young planted areas because these were not categorised by size.

Similar to the classification tree, the regression tree recursively partitioned planted areas to define increasingly homogenous subsets of export orientations on the basis of combinations of periodic tree-cover change classes, planted patch size, plantation type, and country. Combinations were again defined inductively (Section 2.5) such that they characterise subsets of higher export orientations relative to others. Expected combinations should associate higher export orientations with relatively frequent tree-cover losses and/or transitions to non-tree cover, perhaps especially during the first (1990–2000) and second (2000–2005) periods and in conjunction with larger planted patch sizes. Analysis is confined to the seven main countries in which commodities and patch sizes were observed.

### 2.2. Sampling of planted and non-planted areas

Analyses are based on a sample of the planted and non-planted areas defined above (Table 1). Points were randomly sampled from the Landsat tree-cover change data of Kim et al. (2015) at a high initial rate of one point (30-m Landsat pixel) per 100 ha, stratified by country. Subsequently, three subsets of points were discarded to exclude incomplete or erroneous tree-cover change sequences: (i) points classified as cloud, shadow, water, or 'no data' by Kim et al. (2015); (ii) points coincident with built areas in 2014, as observed via Global Human Settlement Layer at 1-km resolution (Pesaresi et al., 2016b, 2016a); and (iii) points with invalid sequences of tree-cover change over 1990–2010 indicative of misclassification during a given period (e.g., *persistent tree cover* → *tree-cover gain* → *persistent non-tree cover*). The final sample is comprised of 0.21 and 1.3 million points for planted and non-planted areas, respectively (Table 1). The final sampling intensity is one point per 169 ha, varying slightly between planted areas and non-planted areas (planted: one per 215 ha; non-planted: one per 162 ha) (Table 1).

**Table 4**  
Patch-size classes for planted tree-cover areas.

Planted Patch Size Class	Description
Small-sized planted mosaics	Planted tree-cover stands of < 10 ha situated within a mosaic of clearings, agriculture, natural forests, and/or settlements, where planted stands account for at least 50% of the delineated mosaic area
Medium-sized planted mosaics	Planted tree-cover stands of 10–100 ha situated within a mosaic area of > 100 ha, where planted stands account for at least 50% of the delineated mosaic area
Large-industrial plantations Corporate estates	Typically monotypic planted-tree cover stands, of 100–1000 ha, which may encompass smaller (<100 ha) associated agricultural areas Typically monotypic planted-tree cover stands, of >1000 ha, which may encompass smaller (<100 ha) associated agricultural areas

Variables for periodic tree-cover change, plantation type, planted patch size, and tree/tree-crop commodity are all locally observed 'per sample point' (Table 3).

### 2.3. Export orientations of tree/tree-crop commodities

We estimated export orientations of 26 observed tree/tree-crop commodities with reference to national production and trade statistics respectively describing 28 and 38 corresponding tree/tree-crop products (FAOSTAT, 2018c, b, a) (Table 2, Text S3). For a given commodity in a given country, estimates are therefore national-scale aggregates, whilst the commodity and its tree-cover change dynamics are observed locally as sample points.

Commodities include undifferentiated commodities such as banana, mixed commodities such as citrus, and industrial commodity tree species such as *Acacia* and oil palm (Table 2). An undifferentiated commodity such as banana yields a single product that is minimally processed (e.g., banana). Its export orientation is given by the exported fraction of this single product, weighted by its export price (Eq. (1)):

$$\text{Export Orientation} = \frac{\text{Exported Production (tonne)}}{\text{Total Production (tonne)}} \times \text{Export Price (USD/tonne\$)} \quad (1)$$

Mixed commodities like citrus entail multiple corresponding products recorded by production and trade statistics (e.g., limes, oranges, grapefruits). Their export orientation is defined by the weighted average of the five most produced products for which national statistics were recorded in a given country (Eq. (2)):

$$\text{Export Orientation} = \sum_{p=1}^5 \frac{\text{Exported Production}_p}{\text{Total Production}_p} \times \text{Export Price}_p \times \frac{\text{Total Production}_p}{\text{Total Production (All)}} \quad (2)$$

where  $p$  denotes a given product in a given country (e.g., limes), 'All' denotes the set of products of the mixed commodity in that country (e.g., citrus), and units are as per Eq. (1). The third term in Eq. (2) weights the first and second terms by the quantity of each product.

For an industrial commodity species such as *Acacia* or oil palm, its multiple processed products were also combined via Eq. (2). For example, the commodity oil palm yields palm oil, palm kernels, and palm cake products, while the commodity *Acacia* yields roundwood, sawnwood, wood panels, and wood pulp products. Processed products were expressed in terms of 'whole units' of a commodity.

For all commodities, export orientations ranged from near nil (Brazilian bamboo) to 18 (Cambodian rubber). Half of all sample-point observations were <0.5, 90% were <1.5, and the global average was  $1.2 \pm 3.3$  standard deviations.

### 2.4. Validations of planted and non-planted areas

Validations of the cover and extent of planted areas and non-planted areas using visual interpretations of high-resolution imagery affirm their high classification and spatial accuracy (Text S2). For the seven main countries, planted areas were verified as hosting planted tree

cover 83–88% of the time, according 3000 validation points randomly sampled and interpreted across planted and non-planted areas (WRI, 2018). The upper accuracy rate is considered most representative and likely an under-estimate (Text S2.1). Non-planted areas hosted planted tree covers only 2–6% of the time, depending on the country. For Malaysia, a separate sample of 1000 validation points similarly interpreted across planted areas indicated a classification accuracy of 87% (Petersen et al., 2016), also considered an under-estimate (Text 2.1). In the remaining four countries, similar validations of land covers in planted areas using high-resolution visual interpretations observed classification accuracies of 90–97% (Li and Fox, 2012; Ministry of Environment, 2015: 88), reflecting a conservativeness of planted-area delineations.

Visual interpretations of tree/tree-crop commodities provide best estimates for which no pan-tropical validation is available. However, comparisons against independent delineations of planted areas of oil palm, *Acacia*, and rubber in Indonesia and Malaysia similarly based on Landsat and high-resolution image interpretation (Gaveau et al., 2014; Sloan et al., 2017) show a very strong concordance for both planted commodities and patch boundaries (Text S2.1). Other studies have also affirmed the accuracy of basic tree-commodity identification using Landsat interpretation (Beaubien, 1986; Austin et al., 2017b; Wijedasa et al., 2018).

### 2.5. Pan-tropical classification/regression trees

The classification tree analysis (CTA) and regression tree analysis (RTA) recursively partitioned the sampled point observations to inductively grow 'branches' of increasingly homogenous subsets or 'nodes'. When read top to bottom, each tree describes interactive combinations of periodic tree-cover change classes and other predictor variables that best distinguish planted from non-planted areas (CTA) or amongst ranges of export orientations (RTA).

For a pan-tropical sample, the CTA and RTA first evaluated the within-node effects of each predictor variable on the response variable using Pearson Chi-Square test (Loh, 2009, 2011). For the RTA, the continuous export-orientation response variable was first categorised by quartiles (Loh, 2009, 2014). The most significant predictor variable was then selected to partition the sample or 'root node' into two 'children nodes'. The selected class(es) of the predictor variable is that which minimises the 'class impurity' of the children nodes (i.e., uniformity of class distribution), indicated by the sum of their Gini indices (Loh, 2011: 15). Partitioning continued recursively in this way for all subsequent nodes of the tree. This two-step approach to partitioning, separating the selection of the predictor variable from that of its key class(es), avoided selection bias amongst variables with relatively many classes.

Explicit pairwise interactions between predictor variables partitioned a given node if no univariate main effects were observed with >95% confidence (Loh, 2009:7–9). In this case, for interacting predictors  $X_1$  and  $X_2$  at node  $n$ , an analysis considered the partition of  $n$  on  $X_1$  and subsequent partition of its children nodes on  $X_2$ , then repeated this process with the roles of  $X_1$  and  $X_2$  reversed, ultimately selecting the sequence yielding the greatest reduction in class impurity (Loh, 2014:333). Implicit interactions amongst predictors are otherwise

revealed by sequences of partitions defining branches in a tree.

Equal *a priori* probabilities of response-variable classes were specified for all CTAs to correct for unbalanced class frequencies in the pan-tropical sample (Table 1). CTAs developed with alternative corrective parameters were variable and greatly inferior in terms of tree growth and accuracy (Text S4). Accuracy pertains to either planted-area classification accuracy (CTA) or the percentage of export-orientation variance explained ( $R^2$ ) (RTA).

Node partitioning continued recursively until a partition would have produced a node with less than a specified number of observations. For the CTAs, this was <3000, being approximately the number of planted and non-planted observations in the country with the fewest observations. For the RTA, the threshold was <1800, being 1% of the pan-tropical sample of planted areas for the seven-country set. Resultant 'full' trees were pruned to produce the smallest tree possible within specified accuracy thresholds. Following CART protocols (Breiman et al., 1984), a pruned tree had an accuracy of <0.5 standard errors of the mean accuracy of 10 full trees grown and validated via 10-fold cross-validation (Loh, 2011). Trees were grown using the GUIDE algorithm (Loh, 2014, 2017).

### 2.5.1. National classification trees

National CTAs were developed to complement the pan-tropical CTA. Each national CTA considered the same observations and variables as the pan-tropical CTA (excepting country; Table 3 Analysis 1), but on a national basis, presenting relatively conservative estimates of the similarity of tree-cover change dynamics between planted and non-planted areas. The set of national CTAs also reveal pan-tropical variability in the strength of association between planted areas and their tree-cover change dynamics.

Each national CTA was defined by a random-forest 'ensemble' of 500 unpruned trees derived from 500 bootstrapped national samples (Loh, 2009:24). Ensemble analyses approximate maximal class discrimination without overfitting the data (Segal, 2004; Prasad et al., 2006). Two-step partitioning and explicit interactions were again allowed. A random set of two variables was considered when partitioning each node to weaken dependence amongst trees of an ensemble (Loh, 2014: 334). Tree growing stopped if new nodes would host <1% of the national observations or produce a tree with > 7 levels. The majority 'vote' of an ensemble classified observations as either planted or non-planted areas and defined nodal probabilities of class membership (Loh, 2014:334).

## 3. Results

### 3.1. Simple and dynamic tree-cover change for planted areas, 1990–2010

Tree-cover changes occurred over one-third of recent (ca. 2014) tropical planted areas during 1990–2010 (Fig. 1). Overall, sequences of tree-cover decrease were more prevalent than sequences of tree-cover increase within planted areas, with simple decreases being particularly prevalent. Over 1990–2010, simple and dynamic decreases in tree cover characterised 15% and 2.7% of planted areas, respectively, whereas simple and dynamic increases in tree cover characterised 7% and 3.6%, respectively (Fig. 1). This imbalance between decreases and increases, as well as the prevalence of simple decreases, is most acute in Indonesia, Malaysia, and Cambodia and suggests widespread forest conversion there preceding planted-area establishment. In these three countries, simple decreases (18.2% of planted areas) were more than twice as extensive as simple and dynamic increases combined (8.9%). Similarly, in these countries the ratio of simple decreases to dynamic decreases (12.2) is more than five-fold that for other countries (2.3).

South America presents a contrasting picture of tree-cover change more aligned with increases in planted areas. Here, simple and dynamic tree-cover increases occurred over 14% of planted areas during 1990–2010. Of these increases, some two-thirds were simple increases

indicative of net reforestation since 1990 (10% of South American planted areas), notably spanning nearly 20% of Colombian planted areas. The remaining increases within South American planted areas were dynamic increases entailing short-term losses of tree cover (4% of South American planted areas), as per serial reforestation or natural forest replacement. In Colombia and Brazil, the extent of simple and dynamic increases combined were much greater or effectively equal to that of simple and dynamic decreases combined, respectively (Fig. 1).

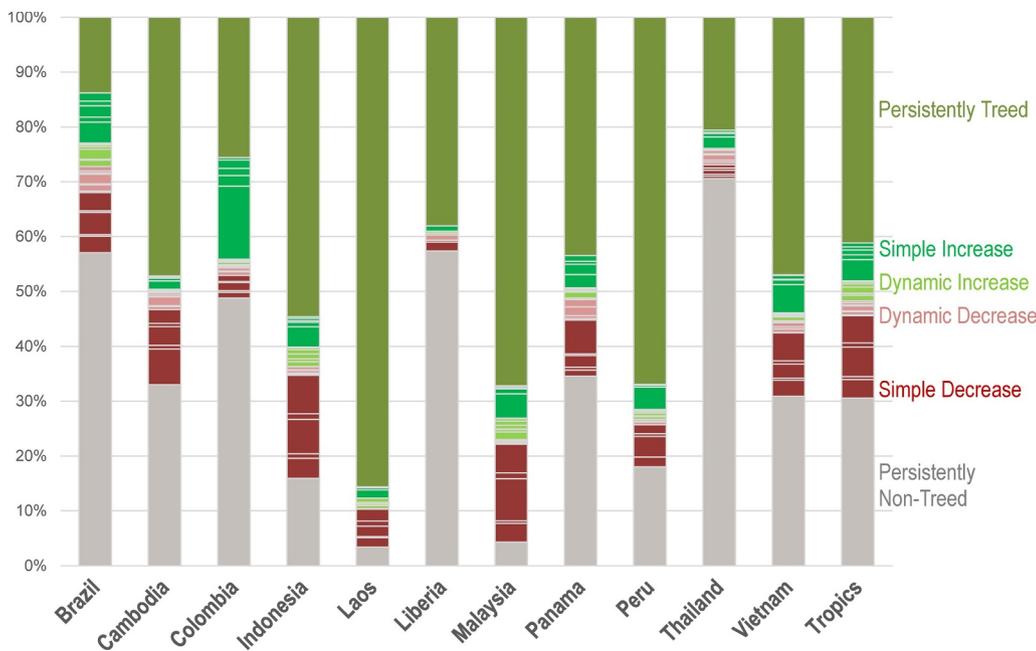
The extent of increases across the tropics and particularly in Brazil is potentially greater than indicated above. Some 30% of recent planted areas were apparently devoid of tree cover over 1990–2010, with two-thirds of these being in Brazil and one-fifth in Indonesia (Fig. 1). The degree to which these planted areas represent very recent reforestation of long cleared lands, or rather serial clearing and replanting of sparse tree cover undetected by Landsat imagery, remains uncertain. Visual interpretations of these Brazilian planted areas using recent (ca. 2009–2018) high-resolution imagery in Google Earth qualify most as industrial, short-rotation tree farms hosting sparse and/or young tree cover with limited canopy cover as well as bare earth (Fig. 2). Tree-cover changes in these tree farms over brief periods (Fig. 2) support the possibility that some may have been established earlier but not detected as treed. This possibility cannot be directly confirmed here due to the paucity of high-resolution imagery coverage before ca. 2005 (Text S5).

### 3.2. Similarity of tree-cover change dynamics in planted and non-planted areas

Planted areas were similar to non-planted areas with respect to tree-cover change dynamics since 1990 and were generally not distinguished by a prevalence of periodic gains and/or transitions to tree cover alone. Rather, where the pan-tropical CTA referenced tree-cover change (i.e., from Node 3 onwards in Fig. 3), planted areas were distinguished largely by extensive, contemporary occurrences of tree-cover gains and losses during a given period between 1990 and 2010 (nodes 6, 28, 120 Fig. 3). Gains were less extensive than losses in each instance. Gains were 66% as extensive as losses during 2000–2005 for planted areas in the large and geographically general node 6 (Figs. 4a and 3). Similarly, gains were 42% as extensive as losses during 2005–2010 for the smaller South American node 28 (Figs. 4a and 3). Such co-occurrences of gains and losses within planted areas are indicative of high levels of tree-cover turnover and robust to variations to sampling and classification parameters (Text S4).

Certain tree-cover expansion dynamics also distinguished planted areas from non-planted areas but were less extensive than the turnover dynamics above. These expansion dynamics in planted areas were defined by persistent tree cover during 2005–2010 interacting with various trends during 1990–2000, mostly in South America (nodes 120, 124, 504 Figs. 4c and 3) and to a limited degree in Indonesia and Malaysia (node 13b) (Figs. 4d and 3). Of these South American expansion dynamics, the 1990–2000 trends were, in order of prominence, (i) contemporary gains and losses (node 120), in keeping the turnover above; (ii) persistent tree cover mixed with occasional persistent non-tree cover (node 124, mostly Colombia); and persistent non-tree cover alone (node 504, Brazil). These South American expansion dynamics equate to 36% of the planted areas involved in the aforementioned South American turnover dynamics (nodes 6, 28 Figs. 4a and 3). In Indonesia and Malaysia, the nominal 'expansion dynamic' (node 13b Figs. 4d and 3) equates to 20% of the area of nominal turnover dynamic (node 7b Figs. 4b and 3). Geographical distributions of these expansion dynamics were similar to turnover dynamics, particularly in South America (Fig. 4a and c), suggesting a spatio-temporal association between contemporary gains and losses and longer-term net tree-cover increases within planted areas.

In Indonesia, Malaysia, and Cambodia, tree-cover change dynamics were generally highly similar between planted and non-planted areas. At the pan-tropical level, the relatively high proportion of pan-tropical



**Fig. 1.** Sequences of tree-cover change over 1990–2010 in recent tropical planted areas, as a percentage of national planted area.

Note: 27 temporal sequences of tree-cover change were collapsed into six meta-classes. ‘Simple Increases’ / ‘Simple Decrease’: Monotonic, stable increases / decreases in tree cover over three periods (1990–2000, 2000–2005, 2005–2010), defined by periodic gains / losses and/or transitions to tree cover / non-tree cover. ‘Dynamic Increase’ / ‘Dynamic Decrease’: Turnover entailing periodic transitions between gains, losses, tree cover, and non-tree cover, ultimately terminating gains/tree cover in the case of dynamic increases, or in losses/non-tree cover in the case of dynamic decreases. ‘Persistently Treed’ / ‘Persistently Non-Treed’: The presence of tree cover / non-tree cover in each of the three periods, respectively. Percentages areas of a sequence are according to the pan-tropical sample of point observations.

planted areas within these three countries (Table 1) and their high ratios of planted to non-planted areas (mean 0.56 vs. 0.07 for other countries, ANOVA  $p < 0.001$ ) separated the three countries from all others (node 2 in Fig. 3). Within these three countries, however, tree-cover change dynamics afforded no additional discrimination of their planted areas from non-planted areas. Hence, a separate CTA for Indonesia, Malaysia, and Cambodia failed to distinguish their planted areas better than random overall (Fig. 3 red nodes), notwithstanding two nodes (7b and 13b in Fig. 3) whose planted areas were distinguished with reasonable local classification accuracy (65–73%). The latter node (13b) is characterised by persistent tree cover in 2005–2010 following various tree-cover changes during 1990–2000 and accounts for only 7% of planted areas in Indonesia and Malaysia. The former node (7b) is characterised by non-tree cover during 2005–2010 and accounts for almost one-third of planted areas in Indonesia and Malaysia.

Overall, planted areas were only moderately distinguishable from non-planted areas on the basis of tree-cover change dynamics. Combinations of periodic tree-cover changes during 1990–2010 and country correctly distinguished 69% of planted areas from non-planted areas across the 11-country set (Fig. 3). Amongst the more conservative national-scale CTAs, the mean national planted-area classification accuracy fell to 56% (std. dev. 18%) for the 11 countries. This decreased to 44% (std. dev. 6%) if weighted by national planted extents. The mean national predicted probability of being planted was similarly modest for observed planted areas in the 11 country-set, at 0.52 (std. dev. 0.02) (Fig. 5). This increased to 0.53 (std. dev. 0.07) if weighted by national planted extents.

### 3.3. Export orientation and tree-cover change in planted areas

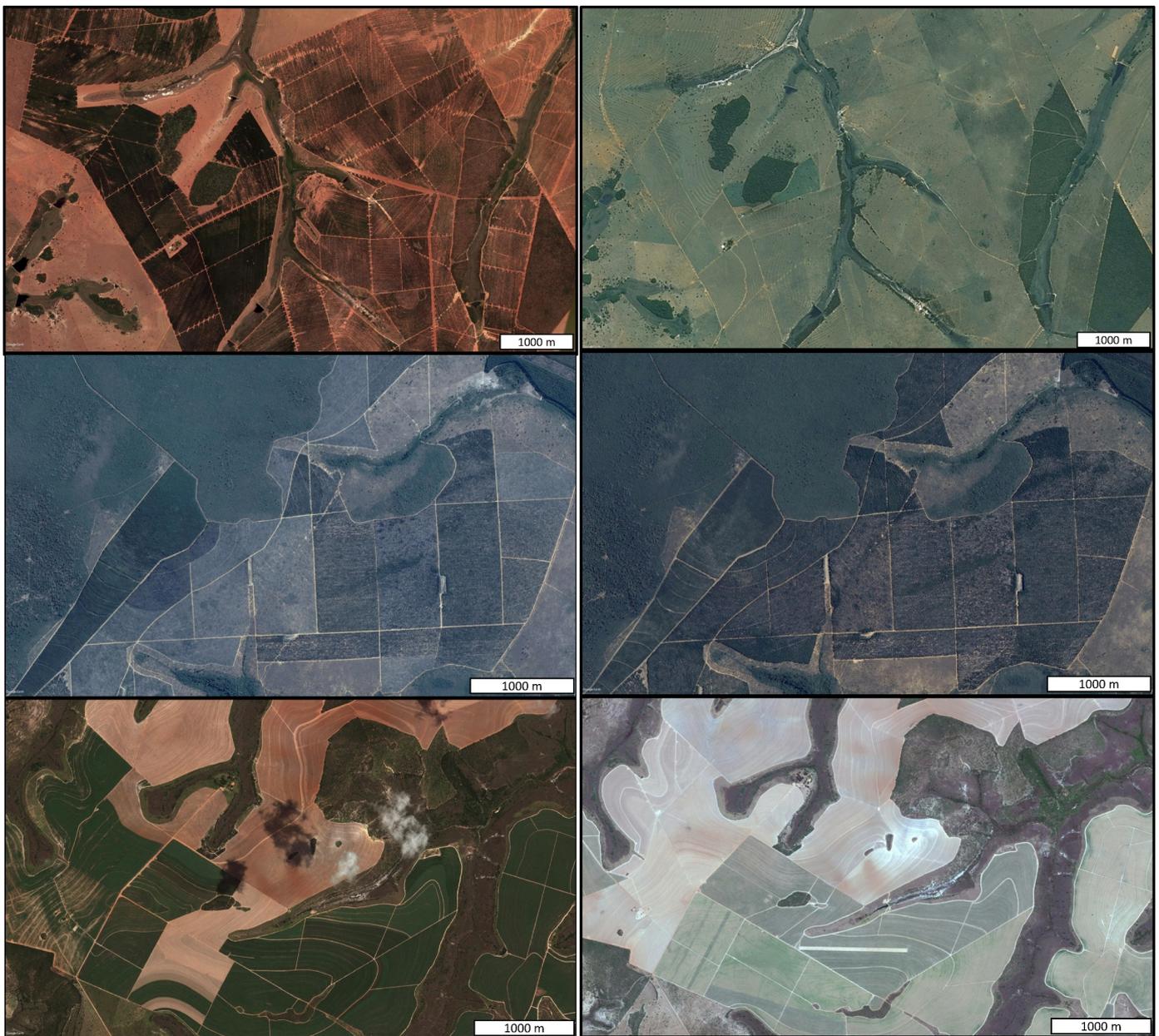
Tree cover losses in planted areas did not clearly associate with higher export orientations (EOs) or larger planted patches. In general, recent planted areas that were *cleared* as of 1990 had *higher* EOs than those planted areas experiencing significant losses since 1990, at least amongst small- and medium-sized patch classes. Insular Southeast Asia again stands out as a partial exception. There, planted areas that were *treed* as of 1990 had *higher* EOs by pan-tropical standards, suggesting a possible linkage between forest loss and the export of tree/tree-crop commodities.

Higher EOs of planted commodities did not generally associate with

larger planted patches (Fig. 6). Instead, EOs for small and medium patch classes were orders of magnitude greater than those of large-industrial and corporate patch classes. For the small and medium classes, prominent EO groupings in the regression tree ranged between means of 1.07 (near pan-tropical average) and 2.21 (top 10th percentile) (nodes 24, 26, 27, 50, 51 of Fig. 6). EOs of medium patches were greatest overall (nodes 27, 50, 51 Fig. 6), at ~30–100% more than those of small patches. High EOs amongst medium patches reflect widespread production of rubber, which commands a high export price and is largely exported, mostly from Indonesia, Malaysia and Brazil. The high EO for medium-sized rubber patches (average 8.6) is tempered by lower EOs for medium-sized oil palm patches (average 0.55), which is equally extensive amongst medium-sized patches. Low EOs amongst large-industrial and corporate patches reflect their greater orientation towards domestically-consumed commodities with lower export prices. Thus, EOs for large-industrial and corporate-estate patches ranged from 0.07 for forestry commodities (node 15 Fig. 6) to only 0.69 for agroforestry commodities (node 14), the latter being composed mostly of oil palm (average EO 0.49–0.53 depending on country).

Higher EOs also did not generally associate with greater tree-cover loss since 1990. Interactions between EOs, tree-cover change, and patch size center instead on whether planted areas were cleared as of 1990 (node 6 Fig. 6). Where recent planted areas were already cleared as of 1990, EOs of small- and medium-sized patches are significantly higher than otherwise (+0.23; node 12 vs. 13 Fig. 6). This is due to high EOs amongst Indonesian medium-sized patches – mostly rubber producers plus oil-palm producers to a lesser extent – who reforested cleared lands after 1990 (node 51 Fig. 6).

An exception is observed for Southeast Asia upon excluding the patch-size variable to yield a complementary RTA focused more explicitly on tree-cover change dynamics (Figure S4). For this RTA, EOs were similarly higher (+0.25) for cleared lands as of 1990 (node 14 vs 15 Figure S4), yet Southeast Asian planted areas assumed an additional, contrary relationship with 1990 tree cover. For areas *treed* as of 1990, EOs of Indonesian and Malaysian planted areas were six times *greater* than for comparable planted areas in South America (0.98 vs. 0.16) (nodes 31 vs. 30 Figure S4). For Indonesia and Malaysia, such *treed* areas as of 1990 were very likely mostly natural forest, considering that most planted areas there were not established until after 1990 (Section 4.3.1). This association for Indonesia and Malaysia is the inverse of that noted above, both with respect to the relationship



**Fig. 2.** Brazilian industrial tree farms labelled as persistently non-tree cover during 1990–2010, as of the early 2010s (left) and late 2010s (right). Note: Image years: top left, 2013; top right, 2018; middle left, 2013; middle right, 2016; bottom left, 2010; bottom right, 2015. Image coordinates: top 19°43'53" S, 52°09'17" W; middle 15°7'28" S, 42°15'13" W; bottom 18°8'18" S, 46°3'02" W.

between EOs and 1990 tree cover and the fact that oil palm, rather than rubber, is the predominant commodity. The fact that this RTA failed to further distinguish Indonesia and Malaysian oil palm (moderate EO, mostly larger patches) from rubber (high EO, mostly medium and small patches) on the basis of tree-cover change is not entirely surprising, as modes of rubber production have increasingly approximated those for oil palm since the early 1990s (Clough et al., 2016).

## 4. Discussion

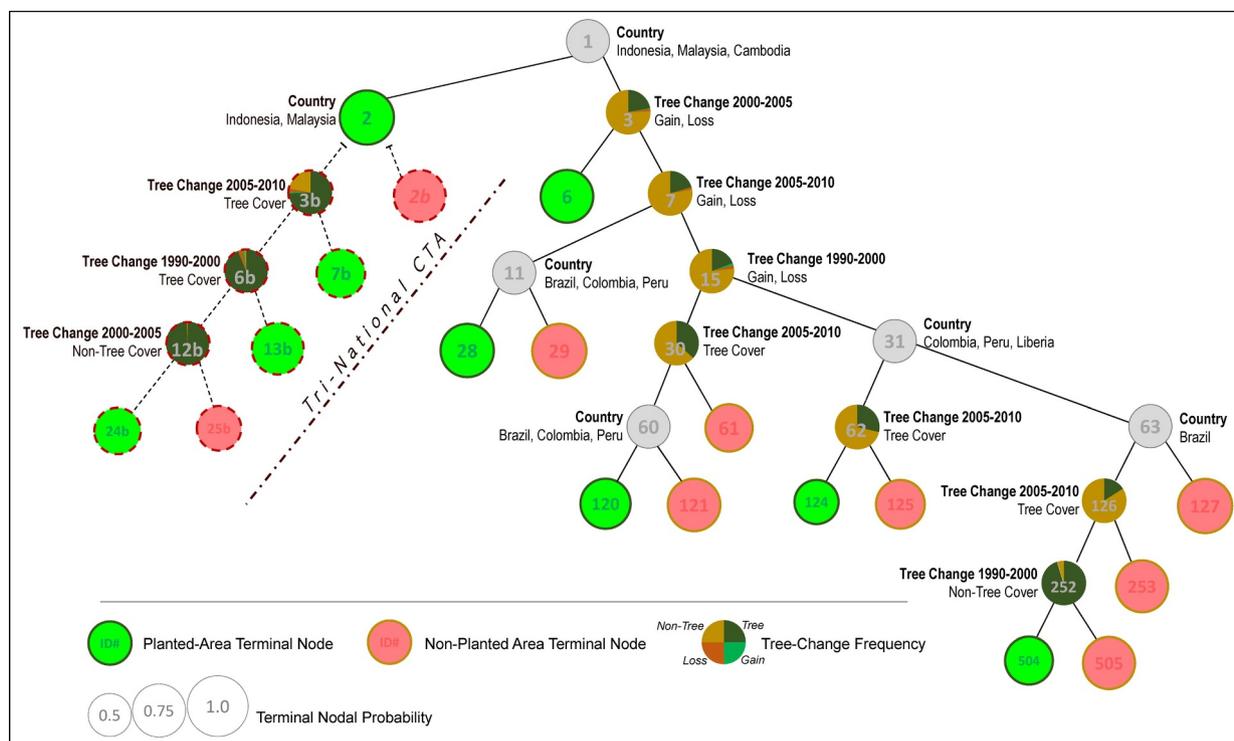
### 4.1. A treadmill of planted-cover expansion

This study profiled tree-cover changes during 1990–2010 within recently-observed areas of planted tree cover. Our findings are generally consistent with the emergent treadmill and forest-transformation narratives of forest turnover and loss associated with planted tree-cover expansion, although they do not preclude forest-transition narratives of

monotonic, enduring tree-cover gains in specific contexts.

In support of these emergent narratives, tropical planted areas experienced dynamic sequences of tree-cover increase and decrease since 1990 far more often than simple, enduring increases (*Hypothesis 1*), with partial exceptions for Eastern Brazil and Colombia (Fig. 4). This characterisation of planted areas is remarkable considering that planted areas have more than doubled in extent since 1990 – a trend evoking imagery of new tree cover where none previously existed. Instead, recent planted areas experienced greater tree-cover decreases than increases since 1990, due largely to extensive decreases in Southeast Asia (Fig. 1). Historical tree-cover decreases in recent planted areas are challenging to interpret due to the lack of data distinguishing ‘natural’ from planted tree covers as of 1990 (Text S5). Regardless, the ephemerality of tree covers in planted areas due to frequent periodic losses lends support to the treadmill-transformation narrative.

In further support of the treadmill – transformation narratives, planted areas were distinguished from surrounding mosaic areas only



**Fig. 3.** Classification tree of recent planted areas and non-planted areas according to tree-cover changes over 1990–2010 and country, for 11 tropical countries.

Notes: The classification tree pertains to pan-tropical Analysis 1 in Table 3. At each partition, an observation goes to the left node if the stated condition is satisfied. Terminal node sizes are scaled according to the estimated probability that its observations correspond to the target class (planted area, non-planted area) (Text S4). Intermediate nodes referencing tree-cover change for a given period show the relative frequencies of tree-cover change classes for the period referenced. Nodal ID numbers are for reference only. Nodes outlined in red dashes correspond to the supplemental CTA for Indonesia, Malaysia and Cambodia, which disaggregates node 2 of the pan-tropical CTA. The parameters of this tri-national CTA were equivalent to the pan-tropical CTA, save that its minimum potential node size was reduced to 1000 to account for its smaller sample size. As noted, the tri-national CTA should be consulted with caution due to its negligible overall accuracy.

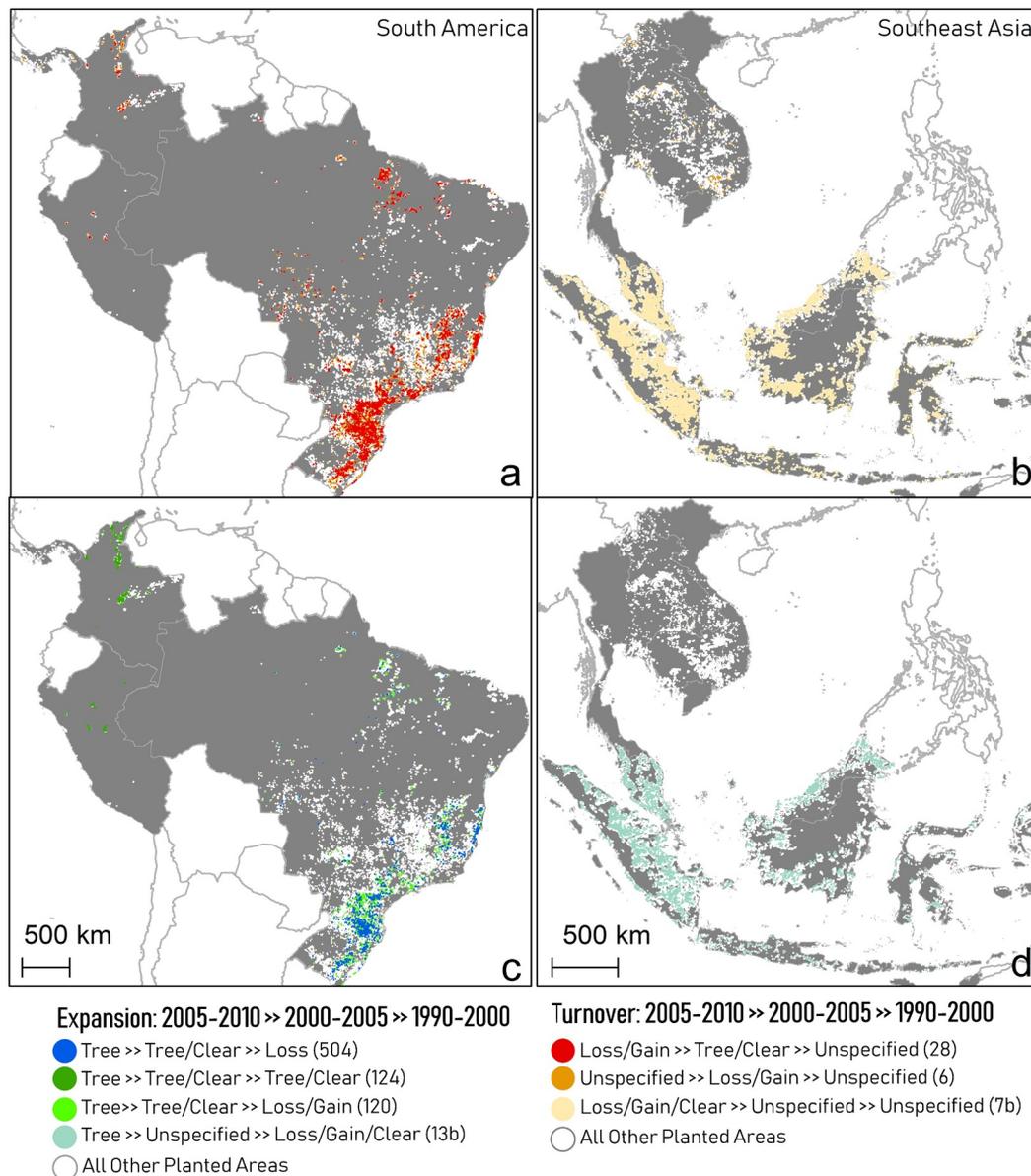
moderately and by contemporary periodic losses and gains, with such losses being more frequent. The manner and moderate degree to which planted areas were differentiated from mosaic areas qualifies planted tree covers as similar to agriculture in terms of negative effects on forests (Gibbs et al., 2010), if not underlying processes or systems of land use, notwithstanding any ‘forest sparing’ effects of timber/fibre planted covers (Heilmayr, 2014; Jadin et al., 2016, 2017). This viewpoint is emergent in the tropical forest-change literature (Altamirano et al., 2013; Van Holt et al., 2016) but under-appreciated in the forest-transition literature, which has dealt more generally with aggregate forest-change and neglected agro-forestry (Rudel et al., 2005; Sloan and Sayer, 2015; Meyfroidt et al., 2018).

The expected relationship between higher export orientations (EO) and elevated historical tree-cover losses and turnover in planted areas was weakly supported (Hypothesis 3). Higher EOs associated rather with lands cleared as of 1990, and it cannot be confidently assumed, using data from as late as 2014, that such lands were cleared to facilitate the observed planted commodities. However, in partial support of the narratives, relatively high EOs amongst Indonesian and Malaysian planted areas treed as of 1990 and dominated by oil palm by 2014 resonate strongly with the well-known conversion of forest to oil palm there over the 1990s and 2000s (Miettinen et al., 2012; Wijedasa et al., 2018).

Contrary to the treadmill-transformation narratives, higher EOs were rare amongst large-industrial and corporate estates, which showed no distinguishing affinities with tree-cover losses (Hypothesis 3). Still, such larger-scale planted patches account for most of the planted extent (Figure S1), which as a whole was characterised by tree-cover decreases (Fig. 1) and contemporary gains and losses (Fig. 3). Ambiguity in the relationship between the nominal scale of planted patches and

tree-cover change reflects complicating factors. Most prominently, larger-scale patches were so common as to arguably define ‘average’ tree-cover change dynamics across planted areas. Particular relationships between larger patch-size classes and specific tree-cover change dynamics may therefore not have been distinguishable against this backdrop of ‘average trends’. Other complicating factors include (i) the use of patch size as a proxy for landholder type; (ii) the difficulty in ascertaining whether historical tree cover was ‘natural’ or planted (Text S5); and (iv) the changing modes of tree/tree-crop production.

Accounting for changing modes of tree/tree-crop production alongside its scale is a priority for the emergent narratives as well as forest landscape restoration (FLR) schemes entailing planted reforestation. Indonesian rubber and oil palm are illustrative in this respect. As noted, Indonesian rubber was distinguished from oil palm and characterised by net gains since 1990 on the basis of tree-cover changes and patch size combined, but not on the basis of tree-cover changes alone. In Indonesia, smaller landholders produce ~80% of rubber (Susila, 1998; GAPKINDO, 2015) but just less than half of oil palm (Kartodihardjo and Supriono, 2000; Cramb and Curry, 2012). Key to the historical sustainability of rubber was the “independence of ... [rubber] smallholders from external economic and political influences” (Dove, 1993), allowing for low-intensity ‘jungle’ rubber production. Jungle rubber and its recovery of cleared lands in Indonesia has become rare, however, as modes of production have shifted. Since the 1990s, smaller producers have increasingly adopted monocultures for rubber and oil-palm alike (Clough et al., 2016). Smaller-scale rubber and oil-palm monoculture now have comparable forest-conversion rates exceeding that of jungle rubber (Clough et al., 2016). This shift reflects greater labour returns and profitability (Schwarze et al., 2015; Clough et al., 2016) but also the steady promotion of ‘development’ and



**Fig. 4. Tree-cover change sequences over 1990–2010 defining tree-cover turnover (a, b) and expansion (c, d) distinguishing recent planted areas from non-planted areas.**

Note: Point observations and tree-cover change sequences correspond to planted-area terminal nodes of Fig. 3. Node numbers are indicated in parenthesis in the legend. All sequences are mutually exclusive. Only planted-area observations are shown. The terms ‘tree’ and ‘clear’ refers respectively to tree cover and non-tree cover persistent over a given period. Sequences with relatively few planted areas are visually exaggerated relative to adjacent sequences with more extensive planted areas because symbols for sequences with relatively few planted areas partially overlap symbols of relatively-extensive planted areas. The ordering of sequences in the legend depicts which sequences are exaggerated. In the legend, top-most sequences are the least extensive and so most exaggerated (nodes 504, 28), while bottom-most sequences are the most extensive and so least exaggerated (nodes 13b, 7b). ‘All other planted areas’ pertain to point observations not defined by nodes denoted in the legend. Node 24b of Fig. 3 is omitted due to its negligible number of planted areas (564) and moderate probability of being planted (0.58). Far eastern Indonesia is not shown because it has a negligible extent of planted areas.

declines in mixed and subsistence land-use systems.

#### 4.2. Tropical planted-cover expansion for forest landscape restoration

Reframing planted tree-cover expansion as a dynamic land-use trend holds implications for forest FLR schemes. Numerous multi-lateral FLR schemes are underway, with over 59 countries and regional entities having committed to restore 170 million hectares to date (Chazdon et al., 2017; Lewis et al., 2019; The Bonn Challenge, 2019). The Bonn

Challenge – a global initiative integrating myriad restoration programmes – aspires to restore 350 million hectares by 2030. The means of FLR are varied and debated. Planted tree cover, while typically more costly than natural regeneration (Chazdon and Uriarte, 2016; Chazdon, 2017), may ultimately account for most FLR commitments due to its affinity with government incentives and accounting schemes, its alignment with political-economic interests, and the need for forestry revenues (Sloan, 2015, 2016; Lewis et al., 2019; Rudel et al., 2019). Progressive and enduring conversions of cleared land into planted tree

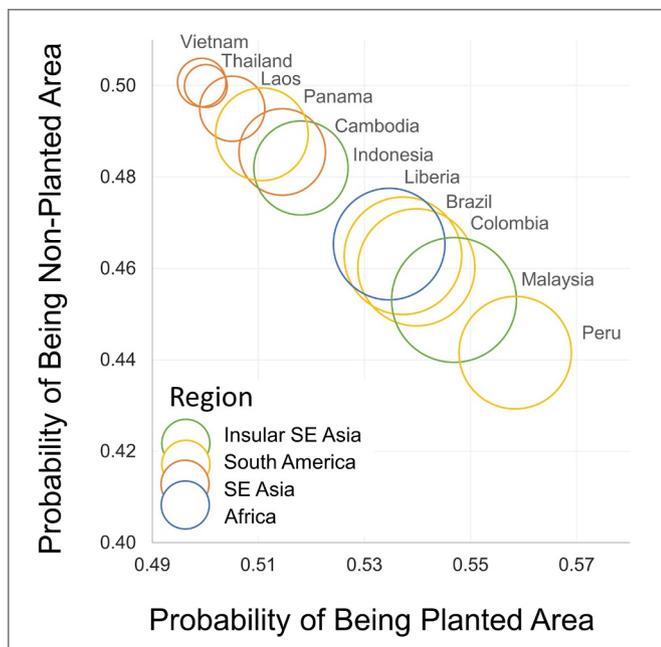


Fig. 5. Mean probability of being planted-tree cover, for recent planted areas, according to national classification trees referencing tree-cover change for 1990–2010.

Notes: The random-forest classification trees pertain to the national Analysis 1 of Table 3. Probabilities were estimated for observed planted areas only. Data point ‘bubbles’ are scaled according to the standard deviation of predicted probabilities of being a planted area, calculated for observed planted areas. The largest and smallest bubbles are respectively 0.174 for Malaysia and 0.021 for Thailand.

covers are possible, provided good governance and well-devised incentives (Sloan, 2008; Yui and Yeh, 2013, 2016); yet the present analysis suggests that such conversions are not prevalent (see also Szulecka and Monges Zalazar, 2017).

Tree-cover loss and turnover undermine the cost/benefit balance of FLR schemes (Verdone and Seidl, 2017). These trends may reflect relationships between planted cover and agriculture. In the tropics, major areas of planted tree-cover often arise out of or envelop agricultural landscapes (Potter, 2012; Gaveau et al., 2016a; Van Holt et al., 2016; Sloan et al., 2017), which in turn may grow by providing labour and plots for planting (Farley, 2007; Cramb and Curry, 2012). Subsidising planted tree cover may therefore indirectly stimulate agriculture and associated impacts on forests and environmental services (Locatelli et al., 2017; Wilson et al., 2017). This is most likely in the probable scenario in which commercial agro-forestry (e.g., coffee, *acai*, rubber) is promoted as feasible, marketable, self-sufficient FLR. Extensive tree-cover turnover in planted areas raises serious questions over the efficacy of planted cover as a means of long-term carbon sequestration, considering the importance of tree-cover longevity to achieve this end (Körner, 2017). Managed commercial tree covers also fail to foster native biodiversity, compared to multi-species restoration plantings or assisted natural regeneration (da Rocha et al., 2013; Hua et al., 2018; Wang et al., 2019). Though a careful selection of tree species might enhance both biodiversity and carbon sequestration within plantations, frequent forest losses would seriously negate both of these environmental services (Hall et al., 2012).

Preliminary observations suggest that small planted patches are associated with tree-cover gains (Text S7). Others have similarly described smallholders as relatively amenable to reforestation programs (Szulecka and Monges Zalazar, 2017) or as stewards of planted forest (Dove, 1993; Midgley et al., 2017). Enthusiasm for such possibilities is

tempered by the fact that small-scale planted patches as observed here comprise only 5% of planted areas pan-tropically. The utility of smallholders for large-scale FLR is therefore debatable and would probably entail trade-offs in the efficiency and effectiveness of FLR schemes (Sloan, 2016). Our study suggests that the mode of production, rather than solely the scale of production, matters for land-change outcomes.

#### 4.3. The forest transformation – a conceptual elaboration

Historical tree-cover change dynamics in recent planted areas distinguished insular Southeast Asia, dominated by Indonesia, from South America, dominated by Brazil. In South America, planted areas were relatively inclusive of tree-cover expansion and distinguishable from agricultural-forest mosaic areas in terms of their historical dynamics. In contrast, Southeast Asian planted areas had a greater tendency towards tree-cover loss – particularly simple declines in tree cover since 1990 (Fig. 1) – and were not readily distinguishable from surrounding agricultural-forest mosaic areas (Fig. 3). This regional dichotomy informs distinctions between the forest transition and forest transformation. The forest transition emphasises forest scarcity and reforestation incentives (Baptista and Rudel, 2006). These factors promote appreciable net reforestation at large scales, including via spontaneous tree planting, while limiting losses of increasingly-valued forests. In contrast, the forest transformation emphasises changing forest valuation given shifting opportunity costs underlaid by political-economic trends in forest management. It is characterised by widespread planted-cover expansion, but modest net tree cover gains across planted areas, given ephemeral planted covers and significant losses of devalued forests. Accordingly, planted-cover expansion would make only minor contributions to broader forest-change trajectories leading towards net reforestation.

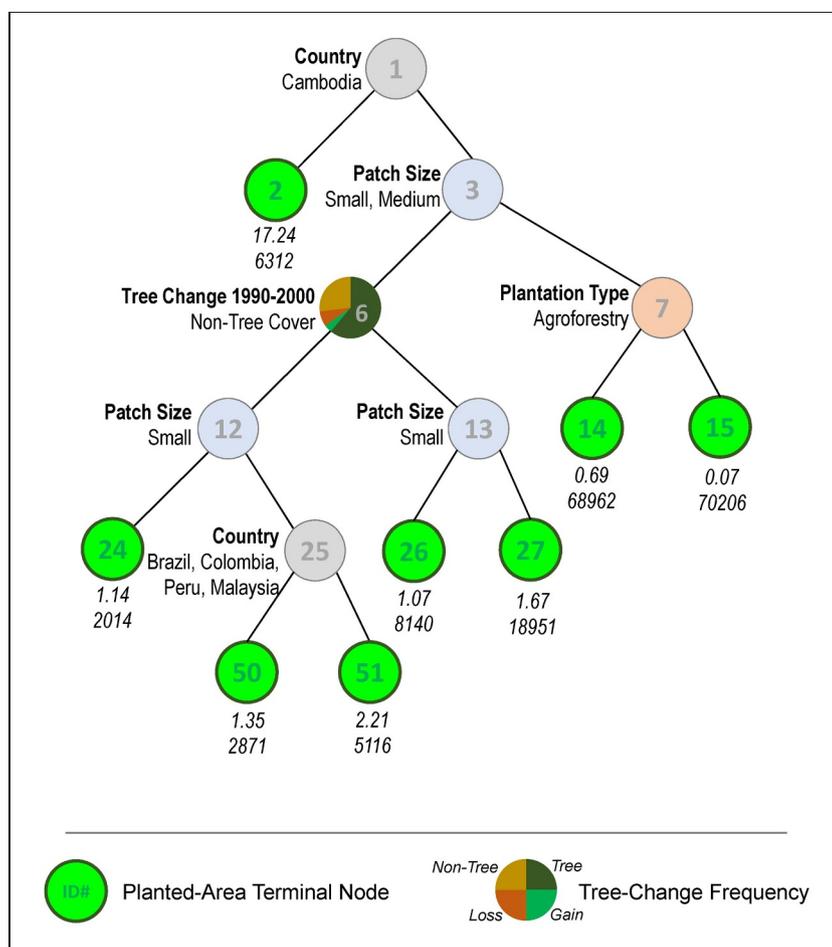
The forest-transformation and forest-transition narratives are not necessarily mutually exclusive. Conceivably, the forest transformation may represent an early stage of the classical forest-transition curve during which extensive forest losses occur. Yet it may equally represent a modern economic or regional variation of the forest transition, which was originally a historical narrative for western Europe (Mather et al., 1999).

Indeed, the two narratives differ at critical junctures. The focus and limitations of the forest-transition narrative are well illustrated by Pirard et al.’s (2016) review of plantation reforestation and forest change. Their review concludes that timber/fibre reforestation driven by forest scarcity and price incentives likely increased overall forest cover, including where plantations replaced natural forests. This effect depends on various assumptions, namely that:

- (i) Natural forests spared from exploitation by plantations retain a value (economic or otherwise) discouraging conversion;
- (ii) Plantations are established before natural forests are widely degraded or converted, but not so early as to widely replace them; and
- (iii) Greater reforestation means natural forests are exploited and converted less widely.

Elaborating the latter assumption, Heilmayr (2014) suggests that timber/fibre plantation reforestation increases ‘unharvested natural forest’ and reduces pressure on production forests. This outcome rests on a further assumptions, namely that:

- (iv) Plantation production may readily substitute for natural-forest production (Jadin et al., 2017), and
- (v) Timber/fibre demand is steady should prices decline due to increased supply from plantations.



**Fig. 6.** Regression tree of planted tree/tree-crop commodity export orientation as a function of tree cover change of 1990–2010, patch size, plantation type, and country, for 7 tropical countries.

Notes: The regression tree pertains to Analysis 2 in Table 3. At each partition, an observation goes to the left node if the stated condition is satisfied. The average export orientation and total number of corresponding planted-area observations (italicised) are reported under each terminal node. The intermediate nodes referencing tree-cover change for 1990–2000 show the relative frequencies of tree-cover change classes for that period. Nodal ID numbers are for reference only.  $R^2 = 0.87$ . Tables.

Political economies of forest management in Southeast Asia have challenged these assumptions (Barr and Sayer, 2012; Scheidel and Work, 2018), as discussed below for Indonesia. For instance, rather than reducing natural-forest exploitation, planted reforestation has also often constituted a ‘second stage’ of such exploitation, as by converting degraded forests undervalued by rent-seeking regimes or raising incentives for illegal logging (Barr et al., 2010). Accordingly, significant planted-cover establishment in Indonesia, Malaysia, and Cambodia diverges from forest-transition narratives and adheres more closely to a forest-transformation narrative (e.g. Gaveau et al., 2016b; Austin et al., 2017b; Scheidel and Work, 2018).

The forest-transformation narrative has been articulated only implicitly with reference to ‘Asian forest transitions’ or ‘strong-state reforestation pathways’ (Mather, 2007; Lambin and Meyfroidt, 2010). It is developed here by describing the political-economic forces underlying the land-change dynamics profiled above. The following discussion focuses on Indonesia, given its prominence amongst Southeast Asian planted areas (Table 1) and the pan-tropical trends observed above (Fig. 3).

#### 4.3.1. The Indonesian forest transformation

Indonesian tree plantations arose amongst abundant forests and extensive exploitation. Since the 1960s, forest management sought to maximise rents for the State’s elite (Barr, 1999) and secure political patronage (Gillis, 1987; Ross, 2001; Burgess et al., 2011). In this context, strong global demand for forest products (Barr, 2001b) aligned with incentives for rapid forest exploitation (Kartodihardjo and Supriono, 2000). In Borneo, for example, the volume of wood exported during its agro-forestry industrialisation phase of 1980–2000 reportedly exceeded those from tropical Africa and Latin America combined (Curran et al., 2004). By the late 1990s, Indonesian merchantable timber stocks were increasingly rare and remote (Gaveau et al., 2014) and production forests

were increasingly degraded. From 1990 to at least 2011, logging steadily contracted in terms of total concession area (–40 Mha), average concession size (–0.3Mha), and new logging roads (Ministry of Forestry, 2012; Gaveau et al., 2014). Legal logged timber volumes declined 37.5% during 1990–1997 alone, and often much more in Indonesia’s major producing regions (Ministry of Forestry, cited by Barr, 2001c: 41–42).

In this context, plantation expansion was catalysed by economic scarcity, characterised by abundant but commercially-depleted forests set against strong demand and preoccupation over forest rents. The decline of logging was mirrored by an expansion of timber/fibre plantation concessions within ‘degraded’ logged forests, rising from near nil in 1990 to over 10 Mha by 2010 (Ministry of Forestry, 2012). The vast majority of these plantations were managed by a relative small number of commercial interests (Kartodihardjo and Supriono, 2000: 2). In contrast to the export-orientation of logging, plantation expansion also reflected growing domestic demands (hence the lower export orientations for timber/fibre commodities in Fig. 6) as well as structural incentives partly inherited from the prior forest regime. Key amongst these were (i) low timber prices/values due to depressed royalties (Barr, 2001d: 80), cheap credit, and illegal logging (Kartodihardjo and Supriono, 2000); (ii) over-investment in milling capacity, which created ‘committed’ demand, including for illegal timber (Barr, 2001d; Margules Groome, 2017); and (iii) reforestation subsidies (Kartodihardjo and Supriono, 2000; Barr, 2001b).

The effect was that Indonesian timber/fibre plantation concessions were under-planted, over-extended, and associated ironically with tree-cover decreases since 1990 (Fig. 1). Concessionaires frequently overstated the area to be planted to maximise subsidies (Ernest & Young, 1997; Barr, 2001d: 80) and then legally clear natural forest therein. Of the total extent of timber/fibre concessions during the 1990s, 22% was estimated to be natural forests (Kartodihardjo and Supriono, 2000). Forest conversion rates for such concessions were apparently similar

**Table 5**  
The forest transformation (FTf) and the forest transition (FT): Two modes of tree-cover expansion.

Tenet of Forest Transformation	Description of Tenet	Contrast with Forest Transition
Economic scarcity precipitates reforestation.	Economic scarcity coincides with extensive forests that are uneconomical, or 'under economical', particularly given rent-seeking behaviour and high exogenous demand.	Unlike the FT, the FTf is not driven by forest scarcity but merely shortfalls relative to high, often exogenous (international) industrial demands. Forests may be abundant during the FTf.
Net tree-cover gains are rapid, arising over one to two decades, but remain modest relative to gross reforestation and corresponding forest loss.	Over this timeframe, Indonesia suggests a ratio of ~2–5 hectares of gross planted tree-cover gain for each hectare of direct conversion of natural forest to planted cover (Kartodihardjo and Supriono, 2000; Gaveau et al., 2016b; Austin et al., 2017b; Fig. 1 of this study). Over three to five decades, virtually all planted areas may be traced back to natural, often intact forests (Gaveau et al., 2016b).	Gross gain rates for FT planted reforestation on cleared lands (e.g., China's Grain for Green program) may compare to the FTf, but net gain rates will be greater. For FT natural reforestation, gains are slower and less extensive but may again result in greater net reforestation over one or two decades (Sloan et al., 2019a). In both cases, many reforested areas are unlikely to trace back to natural, intact forests within three-to-five decades (Thomlinson et al., 1996; Arroyo-Mora et al., 2005), often arising instead in long cultivated/cleared areas.
Natural forests are devalued and/or undervalued	Devaluation is economic and acute for elites. It may reflect political-economic and market distortions depressing prices and discouraging sustainable management. Value for ecological services is overlooked.	During a FT, extant forests enjoy a valuation and protection premium, given forest scarcity and declining ecological services (Locatelli et al., 2017). Natural reforestation often concentrates around these forests (da Silva Mariano Pereira et al., 2013; Sloan et al., 2015).
Reforestation subsidisation is common and generous	Generous subsidisation (Barr and Sayer, 2012) reflects a supposed lack of incentive in their absence (Bull et al., 2006). It entails coordination, even loose central control, over a network of agents. Subsidies, either direct (e.g., tax breaks, carbon-forestry subsidies) or indirect (e.g., political protection), aggravate moral hazards (little reforestation per subsidised dollar or per forest loss), patronage, and land-use distortions (Barr et al., 2010).	FT reforestation is relatively spontaneous, entrepreneurial, uncoordinated, even individual, for both planted and natural forest cover. This reflects local responses to forest scarcity and declining ecological services (Wilson et al., 2017) but also the relatively non-economic or indeed 'personally economic' nature of much FT reforestation.
'Degraded forest' or similar becomes a fluid, constructed concept	This defines the scope for forest transformation (Barr and Sayer, 2012; Chazdon et al., 2016) and is inherently political. Noting the ambiguity of this concept for designating production forest in Southeast Asia, Barney (2008) remarks that it is unclear whether "it is based on... scientific criteria or whether degraded forests are simply those forests declared and legally classified as such".	For the FTf, elites define natural 'political forests' while negotiating the scope of reforestation. For the FT, reforestation is politicized <i>post-hoc</i> , largely by environmentalists and scientists. This pertains to the recognition, status, and management of reforestation (Hecht, 2004; Hecht et al., 2006; Chazdon et al., 2016), not possibilities for same.
Control over access to, and rents from, 'degraded forest' becomes politicised	This reflects moves to consolidate or maintain power within bureaucracies, commonly a Ministry of Forestry (Ross, 2001; Sloan et al., 2019b). Degraded forests may be retained, re-exploited, and planted via inventive schemes, especially where the continued status of 'forest' is contested, e.g., forest restoration licenses (Buergin, 2016; Sloan et al., 2018), REDD+ reforestation (Verhot et al., 2010; Scheidel and Work, 2018).	Contrarily, FT reforestation politicization (above) can depreciate reforestation and dissuade investment in the same. Politicisation affects conservation funding allocation by privileging formal or private schemes attending to intact forests over less formal / community schemes attending to 'ecologically uninteresting' reforestation (Hecht et al., 2006; Sloan, 2016).
Timber/fibre plantations and agro-forestry plantations become inter-related	Inter-relation is geographic, temporal, financial, and with respect to agents and drivers. Both timber/fibre and agro-forestry plantation types may entail rent-seeking and forest mining, especially where conditions favour forest asset liquidation over plantation growth.	Inter-relation in the FT is uncertain and likely less commercial. Local responses to forest scarcity would however often entail mixed planted tree covers, e.g., timber plus agro-forestry.
Synergies between tree planting and forest degradation are strengthened	Synergies strengthen especially amidst weak governance, as where loggers overharvest to later clear-fell for reforestation (Barr, 2002). Synergies exacerbate uneven access to, and benefits from, tree planting, as by deepening commercial advantages or excluding non-elites.	Local exploitation-reforestation synergies are unlikely for FTs as reforestation arises over long cleared lands. Conceptually similar FT synergies instead entail non-local 'displacement deforestation' – the intensification of distant forest or agricultural production permitting greater local reforestation (Meyfroidt and Lambin, 2009; Jadin et al., 2016).
New tree cover is relatively persistent	Planted tree cover associates with mixed gain-loss dynamics (Rudel et al., 2016) but, with few exceptions, it may be expected to persist for decades once established, notwithstanding serial tree harvests and replanting.	FT planted reforestation may persist comparatively to the FTf. However, FT natural reforestation is relatively ephemeral (Sloan and Pelletier, 2012; Reid et al., 2018). In this case, compared to the FTf, relatively large portions of landscapes may therefore have to endure FT dynamics/ reforestation to yield a given, possibly lesser area of persistent reforestation.

during the 2000s (Abood et al., 2015). Further, reforestation within concessions achieved only 23% of the intended extent as of the late 1990s. Partly reflecting structural conditions, timber/fibre plantations were apparently not financially attractive independent of subsidies, much of which came from natural forest exploitation.

The rise of Indonesian agro-forestry plantations was also linked to rent seeking amongst declining forest revenues and contributed heavily to prominent sequences of tree-cover decrease within Indonesian planted areas since 1990 (Fig. 1). Over the 1990s, Indonesian agro-forestry was dominated by smallholder rubber (Susila, 1998) that

induced relatively little forest conversion despite its high export orientation (Dove, 1993). By the 2000s, the extent of oil palm had surpassed that of rubber and timber/fibre concessions (Abood et al., 2015), including amongst smaller landholders (Clough et al., 2016). Like timber/fibre plantations, oil palm was valued not only for its own production but also for the ability to 'mine' residual forest rents (Kartodihardjo and Supriono, 2000:8) and thus meet 'committed' demand for waning forest production. Larger oil-palm stands were established initially over natural 'conversion forests' – nominally the most degraded of logged areas – from which timber volumes more than

doubled over the 1990s to comprise 40% of total timber and pulpwood by the early 2000s (Barr, 2001a: 42, 44).

Oil palm and timber/fibre planted tree covers as well as logging became increasingly integrated, commercially, logistically, and with respect to effects on forests (Kartodihardjo and Supriono, 2000:4,6). This integration culminated in the expansion of oil palm into 'permanent' natural production forests during the late 1990s (Potter and Lee, 1998), thereto the sole legal domain of timber/fibre plantations. Both the oil palm and timber/fibre sectors thus occupied overlapping realms of logged forests, the value and volume of which being integral to these enterprises. Rates of nominal forest loss were comparable between oil palm and timber/fibre concessions over the 2000s (Abood et al., 2015).

#### 4.3.2. Tenets of the forest transformation

Taking the Indonesia case study as indicative, a set of tenets of a forest-transformation narrative emerge to distinguish it from the forest-transition narrative (Table 5). These tenets outline a political-economic process whereby economic scarcity, not forest scarcity, reframe forest (re)production and control of increasingly devalued, abundant forests. Forest transformations are relatively centrally coordinated with respect to their organisation and incentives. They are less aligned with local demand and land users. Instead they emerge to meet global and national demands (including structural demands) through state political-economic interests and commercial agents. The transformations occur amidst extensive and devalued forests and are relatively rapid and widespread, particularly if subsidised; yet they yield comparatively modest contributions towards enduring net reforestation. The transformation process politicises forests earlier in the reforestation process and 'from the top down' within bureaucracies. In this way, the ascendancy of an integrated plantation sector may rapidly yield ~2–5 ha of gross tree-cover gain per hectare of related forest conversion.

While we elaborate these tenets with reference to Indonesia, they likely apply readily to Malaysia (Peluso and Vandergeest, 2010) and to Southeast Asia to a degree. In Cambodia, a recent reforestation scheme for carbon sequestration entailed forest mining and conversion to agroforestry (Scheidel and Work, 2016, 2018), recalling business-as-usual couched as restoration. In Vietnam, half of recent net reforestation was attributable to plantation reforestation (Meyfroidt et al., 2014), including for REDD+ carbon forestry (Ingalls et al., 2018) that prioritised quick monetary returns and forest-territorial consolidation via politically-aligned, self-interested agencies (Lambin and Meyfroidt, 2010:115). Still, differences amongst contexts urge the refinement of the forest-transformation narrative for general application.

To this end, we highlight a key dichotomy between the forest-transformation and forest-transition narratives relevant for theoretical development. Forest transitions are conceptually generalised insofar as they describe regional or national forest-change trajectories even whilst reforestation concentrates in specific contexts, e.g., peripheral uplands. The forest transformation, in contrast, remains relatively conceptually confined to planted areas, such that its equivalence with broader forest-change trajectories remains unspecified. At a minimum, limited net tree-cover increases in planted areas equate with limited contributions of planted-area expansion to reforestation generally – a point relevant to the increasing prominence of planted areas in discussions of emergent tropical reforestation trajectories and schemes (Sloan, 2016; Lewis et al., 2019; Rudel et al., 2019). Still, other associations between planted-cover expansion and broader forest-change trajectories are conceivable and merit exploration.

## 5. Conclusion

Planted tree cover is expanding across the tropics but it does not generally appear to be associated with progressive, enduring increases in tree cover. Exceptions apply to eastern Brazil and Colombia, where tree-cover increases in planted areas were relatively prominent. Expectations of unambiguous reforestation within planted areas were qualified by

characteristic combinations of contemporary losses and gains of tree cover, with losses being more prominent, as well as by an appreciable similarity between planted and non-planted areas in terms of their historical tree-cover change dynamics. The forest-transformation narrative was presented as a complement to the forest-transition narrative, appearing particularly suited to Southeast Asia. There, planted areas are extensive and expansive, but related net tree cover gains are rare and tenuous, reflecting political-economic trends in forest management.

## Declaration of Competing Interest

The authors declare no conflict of interest.

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## Supplementary materials

Supplementary material associated with this article can be found, in the online version, at [doi:10.1016/j.gloenvcha.2019.101988](https://doi.org/10.1016/j.gloenvcha.2019.101988).

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