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The nature of a 'forest transition' in Thừa Thiên Huế Province, Central Vietnam – A study of land cover changes over five decades

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ABSTRACT

Informed from historical case studies of land cover change and development in northern countries, forest transition (FT) theories have a tendency to precast specific conclusions. Considering the case of a so-called FT in Thừa Thiên Huế Province in tropical Central Vietnam, we investigated 1.) whether such a 'FT' indeed reflects a resurgence of genuine forest, 2.) whether the land cover changes can be explained through conventional 'pathways' of FT, and 3.) in which ways the changes may or may not portend 'sustainable development'. Using satellite imagery and topographic maps, we produced maps for twenty land cover types for the years 1966, 1973, 1979, 1988, 1998, 2008, 2016 and 2019 and analysed land cover change over time. We contextualize these results with reference to the historical and scientific literature on Vietnam, and find that 1.) the forestlands represent a historically rich bio-cultural landscape; 2.) considerable forest destruction resulted from the Second Indochina War rather than classical degradation pathways; 3.) in the post-war period altered forestland spaces and re-emerging land uses interacted with state-led re-territorialization and socialist plans for land resource development, influencing shifts in forest cover; 4.) during 1979–1988 state-led intensive timber logging in remaining rainforests (causing widespread forest degradation) somewhat paradoxically (in terms of conventional FT models and theories) coincided already with a slight increase in lower-biomass tree cover; 5.) after 1988 logging in natural forests was officially prohibited (logging bans), and forestry shifted to a reliance on wood produced in acacia-based plantations (largely on lands officially allocated to households); and 6.) this shift went along with a significant state-led restructuring and development of land use policies and the promotion of forest-relevant economic industries. 'Restoration' of tree cover mainly consisted of the expansion of exotic tree plantations, but – at least intermittently – this may also have mitigated impacts on natural forests. We conclude with some reflections on the FT-outcome, its 'sustainability', and future trajectories and possibilities of land cover changes in TTHP.

1. Introduction

While rapid deforestation continues in much of Southeast Asia (Stibig et al., 2014), increases of forest cover have reportedly occurred in Vietnam since the 1990s, leading to suggestions that a 'forest transition' (FT) has taken place (Mather, 2007; Meyfroidt & Lambin 2008a; Lestrelin et al., 2013; Liu et al., 2017; Cochard et al., 2017). Within the tropical context of Vietnam, what does it however mean to say "there

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has been a FT"? Would a FT – if adequately understood – be indicative of more sustainable patterns of environmental management?

By conventional definition (Mather, 1992), a FT is an incisive historical turnaround from deforestation (net forest cover loss) to reforestation (net gain) within a specified territory, usually an entire country. In temperate countries, FTs (through spontaneous or human-aided reforestation with native trees) were aptly depicted as a simple U-shaped curve, as shown in Fig. 1 (black trend line). Such FTs were commonly also seen to epitomize a transition to more sustainable forest and landscape management[1] (Notes see Supplementary Materials A). In the case of species-rich tropical forests, conversely, so-called FTs are often owing to newly arising species-poor types of woody vegetation, including mono-species 'forests' composed of alien plantation trees, invasive woody species, or specific pioneer species (de Jong, 2010; McConnell et al., 2015; Nicolic et al., 2008; Heilmayr et al., 2016). Rather than just looking at cover[2] of 'forest' one therefore needs to take account of important 'transitions' inherent within and alongside any stands of trees (Sloan et al., 2019; van Holt et al., 2016; Chazdon et al., 2016; Chazdon, 2014; Putz and Redford, 2010; Fig. 1). Such integral 'transitions' imply complex ecological shifts and dynamics that are linked to a multitude of easily overlooked human land uses. Correspondingly, such 'transitions' are set to alter numerous parameters pertaining to social-environmental sustainability (e.g. specific environmental risks, livelihood options, biodiversity, ecosystem services, etc.).

The notion of FT not only enunciates a somewhat questionable blueprint model of forest resurgence. Within mainstream academic discourse, FTs are usually conceived in tandem with specific theoretical causal explanations that are linked to broader societal, political, and economic processes (Perz, 2007). Largely informed from historic country case studies in the Northern hemisphere[3], the classic FT theories have an affinity with modernization theories, setting a strong focus on interpretations of macro-economic factors. Accordingly, the 'transition' has often been presented as a quasi-deterministic socio-environmental development process, and as a 'turnaround' attesting to processes of a wider 'sustainability transition' (Meyfroidt and Lambin, 2011, cf. Mather et al., 1998, Mather and Fairbairn, 2000). More recent studies, by contrast, have brought into focus more material-grounded realities, accentuating that original FT theories have paid limited attention to 1.) overall 'energy transitions' (or other material transitions) inherent in historic industrialisation of Northern countries (Gingrich et al., 2019, 2021; Magerl et al., 2022)[4] and more recently in some Southern countries (Sloan et al., 2019), and/or 2.) 'transitions of spatial displacement' which became possible under increasingly globally-networked (yet unequally weighted) economies, from colonial to post-colonial eras (Walker, 2008; Meyfroidt and Lambin, 2009; Meyfroidt et al., 2010; Pfaff and Walker, 2010; Pendrill et al., 2019)[5].

In brief, FT is a notion in socio-environmental discourse which usually implies 1.) a seemingly plain and recurrently observable developmental pattern (U-shaped curve, Fig. 1), and 2.) notably a pattern that is typically seen as positive on a normative scale (~sustainability transition under modernization). In reality, however, a FT is neither a 'fact of nature' per se nor does it 'naturally' imply positive outcomes across the board. Whilst one may recognise specific functions of the 'concept of FT' within the dominion of discourse, one may thus equally recognise that FT essentially constitutes a 'forest cover change' of specific natures. An appreciation of such natures is of fundamental importance for any understanding of FTs in relation to socio-environmental sustainability.

Forest cover changes in Vietnam (~FT?) were previously described and analysed mostly on the basis of official provincial or district forest statistics (Cochard et al., 2017, 2020; Meyfroidt and Lambin, 2008a, 2008b, 2009). Such official data are relatively rough (in terms of describing distinct types of vegetation cover) and of limited extent and precision (data exists only since the 1990's and is based on field inventories; cf. Cochard et al., 2017). In the present study, we take an empirically-based deeper and longer-term focus on the case of the Central Vietnamese province Thừa Thiên Huế (TTHP). Using data derived from careful analyses of satellite images, we investigated the detailed patterns of land cover changes over a period of more than five decades, i.e. a period which encompasses an initial phase of net deforestation (1960's–1980's) followed by a phase (since the 1990's) of intensive tree planting and forest protection policies, and associated economic restructuring. Taking inspiration from critical approaches by Walters (2017, 2022), the aim of the present study was essentially twofold, namely 1.) to investigate the intrinsic socio-environmental complexities of forested land cover changes in Vietnam, specifically in TTHP; and 2.) to make visible such complexities to stimulate more grounded discussions of 'FTs' and socio-environmental sustainability.

2. Study site

Thừa Thiên Huế Province in North-Central Vietnam (5033 km^2; Fig. 2) includes coastal lagoons and plains (lowlands), lower-elevation undulating hills (midlands), and forested mountains and valleys (uplands)[6]. The climate is wet-tropical with a strong monsoonal influence

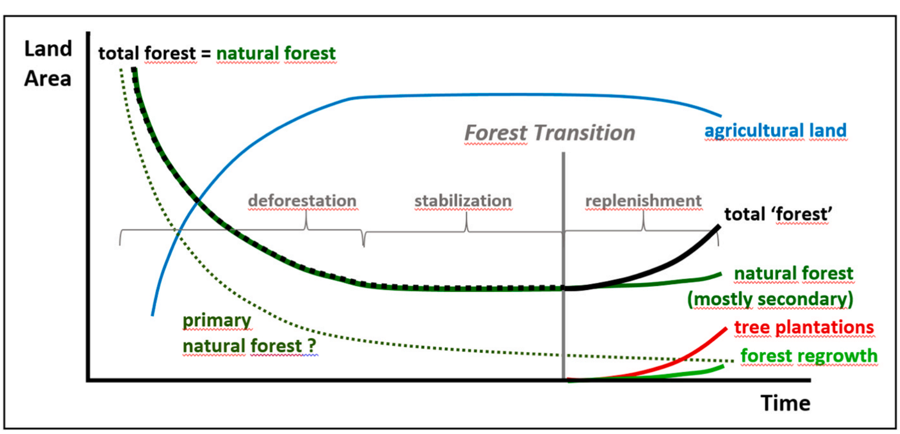
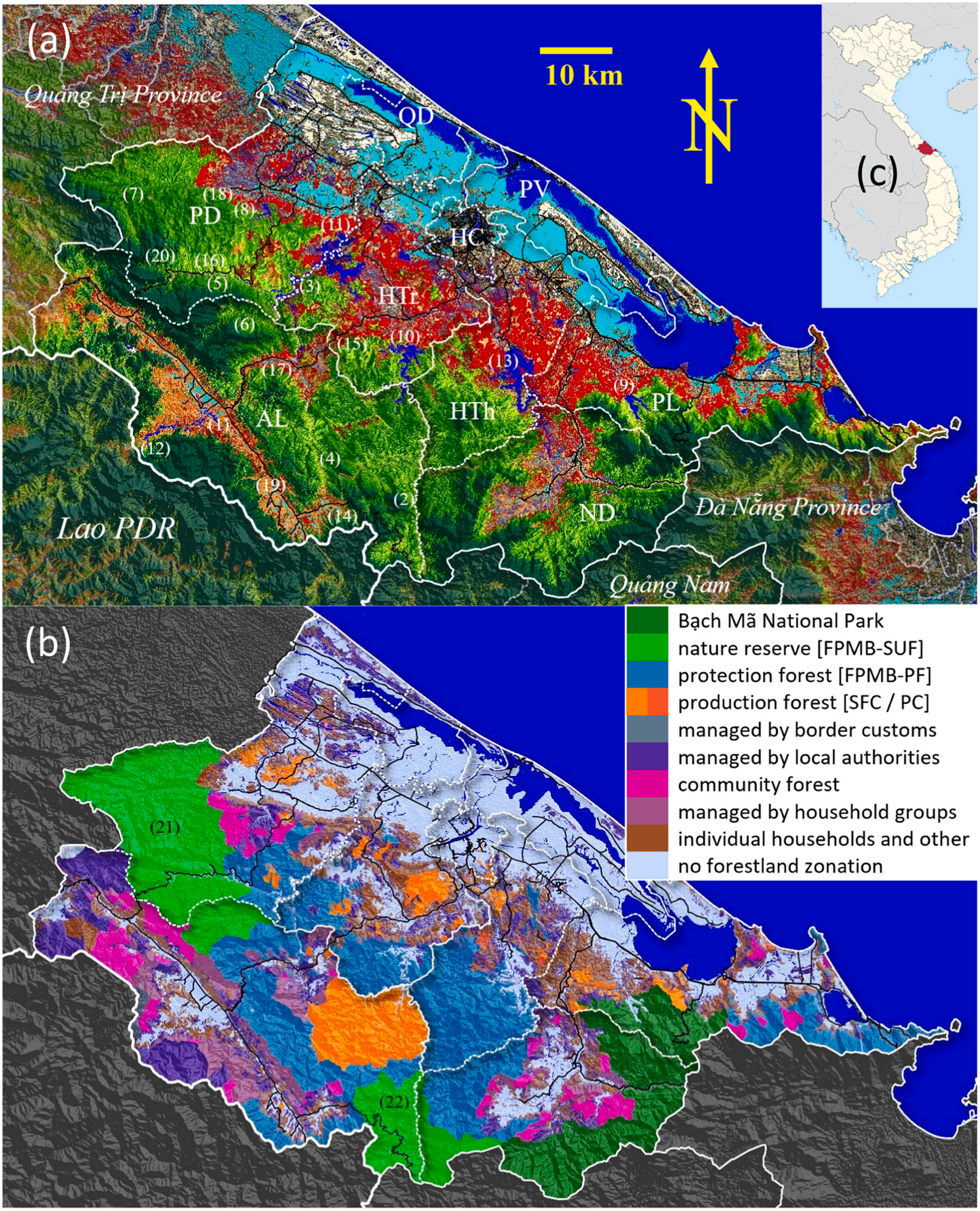


Fig. 1. Theorized schematic representation of a U-shaped forest transition (FT) curve, showing a differentiation of integral 'forest' types over time. In the replenishment phase, natural forest cover may partly increase due to spontaneous tree regrowth, but an increase of overall tree cover is often brought about by active planting of trees. In Northern temperate contexts tree planting with native species may be largely commensurate with 'forest restoration'. In many tropical contexts, however, expanding tree plantations represent an entirely new vegetation type which essentially replaces (and perhaps displaces via direct encroachment) original forest. In addition, observation of a FT does not necessarily imply that degradation of natural forest is effectively halted and reversed. For example, in the tropics the cover of primary rainforest may still be declining alongside increases of secondary natural forests which emerge in diverse degraded/simplified forms.
Figure adopted from Barbier et al. (2010), with modifications.



(caption on next page)

Fig. 2. Study site maps. **(a)** Land cover and **(b)** administrative forestland zones in 2019 of **(c)** Thừa Thiên Huế Province in Central Vietnam. White lines are country boundaries (thick line), provinces (medium), and districts (hashed); black lines are roads. For the color code of land cover types in Fig. 2a see Table 1. Districts are: AL - A Lưới, ND - Nam Đông, PL - Phú Lộc, HTh - Hương Thủy, HTr - Hương Trà, PD - Phong Điền, QD - Quảng Điền, PV - Phú Vang, HC - Huế City. Numbers show locations mentioned in the text: (1) A Lưới Valley; (2) Hữu Trạch River, (3) Bồ River, (4) Rào Nái River, (5) Rào Trăng River, (6) Rào Lo River, (7) Mỹ Chánh River, (8) Ô Lâu River; (9) Truồi dam, (10) Bình Điền dam, (11) Hương Điền dam, (12) A Lưới dam, (13) Tả Trạch dam; (14) Hồ Chí Minh Highway (QL 14), (15) Road QL 49, (16) Road 71; (17) Hồng Hạ Commune, (18) Phong Mỹ Commune; (19) former A Shau army camp, (20) former FSB Ripcord; (21) Phong Điền Nature Reserve, (22) Saola NR. State forestlands are managed by forest protection management boards (FPMB) for protection (PF) or special-use (SUF) forests, and state forest (SFC) or private (PC) companies.
Data source for map (a): this study. Map (b) source: Ministry of Agriculture and Rural Development. Map (c) source: Wikipedia.

(Huế City mean annual temperature is 25 °C; mean precipitation 320 cm, with however ~70% of annual rainfall during September-December; Tong et al., 2011). Inland soils are red-yellow clays/loams; in coastal areas sandy soils predominate (Thanh and de Smedt, 2014).

Settlement of the lowlands goes back many centuries to the times before the Champa Kingdom (Biggs, 2018a; Kiernan, 2019). The lowlanders (nowadays mostly Vietnamese-speaking Kinh people) relied on paddy cultivation in the coastal-alluvial plains. By contrast, the densely forested landscapes in the Trường Sơn mountains were inhabited by diverse Katuic-speaking peoples (ethnic minorities, in 2019 accounting for 5.1% of the population in TTHP)[7] who subsisted on swidden farming in rainforest glades and on forest-derived products (Biggs, 2018a, Århem, 2014; Van et al., 2016). In 2019, 1'129'500 people lived in TTHP (223 people km^{-2}), thereof 49% in urban centres (GSO 2021). Between 1960 and 2019 the population increased about threefold (WB Data, 2022)[8].

TTHP encapsulates many of Central Vietnam's historical and present development trajectories. TTHP is the location of Vietnam's former imperial capital Huế (Kiernan, 2019)[9]. It was an embattled territory during the First (1946–1954) and the Second (1955–1975) Indochina War[10]. During 1975–1986 TTHP was submitted to stringent communist rule, with land collectivisation, resettlement and migration programs, and state-led forestry as dominant drivers of land use/change. The Đổi Mới reforms in 1986 led to a period of constant economic growth, with Vietnam's annual per-capita gross domestic product (GDP) increasing from 95 US$ in 1989 to 547 US$ in 2004, and 2715 US$ in 2019 (WB Data, 2022). In TTHP, agricultural and forest land privatisation, and extensive 'reforestation' and forest 'protection' programs vis-à-vis fast infrastructural development have reshaped the landscapes since the 1990's

3. Study Approach and Rationale

3.1. Summary of the study approach

We produced a series of land cover maps dating from 1966 to 2019 (based on satellite imagery and historic maps). Using these maps we trace the historic land cover changes in TTHP. We first compare/contrast the empirically assembled curves of several 'forest' cover types (and other land covers; Fig. 3) to the schematic representation of FT shown in Fig. 1. We then engage in a detailed narrative-style discussion that draws on much background information from secondary literature. We do this in order to connect the dots between the fine-grained map-based results and inferred factors and processes which – within a variable spatial and temporal context – most pertinently explain the observed land cover changes *alias* 'FT' (Figs. 4–9). Alongside this stepwise 'explicative reconstruction' we also pay attention to land cover transformative changes (and associated processes) that are relevant to discussions linking forest change with sustainable development and environmentally sound land uses. Our approach is founded in the below-described rationales.

3.2. Recognizing bio-physical and spatial texture and dynamic change within a 'forest transition'

In any study on land cover changes there are evidently limits to the precision of vegetation description, classification, and mapping. The question whether a FT leads to more sustainable socio-environmental outcomes can however not be addressed in earnest if the diversity of biotic and structural forms of tropical vegetation (which may or may not qualify as 'forest') is essentially ignored. Tree stands need to be at least distinguished in a functional sense as tree plantations (for commercial uses and emerging from economic metabolism) and natural forests (as biodiverse habitat for native species). In the present study we aimed at further textural differentiation because 1.) different types of tree plantations are characterised by fairly distinct ecologies, socio-economic metabolisms and associated land cover dynamics, and as 2.) primary rainforests (~high biomass volume) can differ greatly in terms of socio-ecological qualities from regenerating and/or degraded secondary natural forests (~lower standing biomass). Using such refined data a FT can then be assessed in different ways, e.g. considering all types of tree cover or focussing on specific 'forests' (in particular highly biodiverse natural forests; cf. Fig. 1). In addition, detailed land cover maps allow for observing and describing spatial patterns of land cover and dynamic shifts over time, i.e. potentially important sustainability-relevant processes which occur 'within-territory' and which thus remain essentially hidden under the synthesis of an U-shaped FT-curve (Fig. 1).

3.3. Seeing 'forest transition' as more than just one possible conceptual construct

In the current study we consider FT not as one simple concept. The classic FT (*sensu* Mather, 1992) is in essence a 'forest area transition' (FTa). Assessment of a FTa for a specified territory can provide insights into land use changes in terms of tree-covered versus tree-less lands. FTa may however cache significant changes in terms of forest transformations (cf. Perz, 2007, Sloan et al., 2019). One may thus also consider other FTs such as a 'forest biomass transition' (FTb) where a decrease in woody biomass within a specific territory is followed by a resurgence in biomass (mostly determined by large trees). This can provide insights into large-tree versus small-tree (intensive use; short harvest rotation) valuations and dynamics of land uses. In addition, one could theoretically think of other more sustainability-focused transitions (FTc, etc.) towards forest biodiversity conservation and ecosystemic stability, tree species resurgence and enrichment, and perhaps even a general improvement and/or restitution of original natural forest ecosystems and associated ecosystem functions and services. While considerations of such FTs may remain largely implied rather than concrete (without robust data and associated analytical frameworks), the bottom line is that 'tree cover' (or 'biomass') alone is hardly sufficiently informative to address sustainability questions. This particularly applies to tropical regions where natural forest ecosystems are characterised by a huge biotic complexity, with associated challenges and limits in understandings.

3.4. Exploring the nature of 'change formation' within a 'forest transition'

FT theory has largely been developed around explanatory developmental models such as the 'economic development pathway' and the 'forest scarcity pathway' (Rudel et al., 2005)[11]. We recognize and partly refer to this theoretical basis and associated literature; yet, we also consider such 'pathways' somewhat critically as we deem that these can potentially invoke specific conclusions through an *ex ante* political-economic line of interpretation (cf. Walters, 2017, 2022). For our qualitative interpretation of spatially explicit (map-based) land cover changes we took some inspiration from economic analyses of FTs by Barbier et al., (2017, 2010); land use valuations, competing land use framework. We tried however to approach interpretations in a rather unprepossessed way, equally allowing for appreciation of path-dependencies of land cover changes, attention to unforeseen factors and effects (chance) and associated momentum, and the recognition of uneven leverages of power exerted by different actors.

For any specific time we see the causalities of further land cover change primarily contingent upon 1.) existing land cover configurations, 2.) valuation of different land cover, and 3.) socio-political leverages. In accordance, when studying the 'anatomy of a FT' we thus set our focus on intermeshed and interplaying factors or processes which – within a specific delimited landscape – may be observed within three general 'ambits of change formation'. The first ambit may be labelled the 'ambit of forestland-space-related factors and processes' (AFS)[12]. Here the analytical focus is set primarily on the manifest spatial configuration of land cover at any specific time, and associated (anthropogenic and/or natural) processes and potentials. Does this configuration (including specific land potentials) *allow (or dis-allow) to make room* for particular types of tree cover? The second ambit may be labelled the 'ambit of tree-valuation-related factors and processes' (ATV)[13]. This ambit is often somewhat contingent on the first ambit, but here the analytical focus is on the valuation (by humans; implicit or explicit) of land cover at any specific time (cf. Barbier et al., 2010, Satake and Rudel, 2007, Barbier, 2011). Do specific factors or processes (economic, political, social) *render higher (or lesser) values* on particular types of tree cover, possibly directing and/or incentivising specific pro-forest (or otherwise) investments such as tree planting and/or forest protection? Finally, the third ambit may be labelled the 'ambit of capital-political-related factors and processes' (ACP). Again, this ambit is in interplay with the other ambits, but here the focus is set on configurations of socio-political power and various other 'capital' which may or may not facilitate and/or allow (or dis-allow) specific changes (cf. Barbier and Tesfaw, 2015, Angelsen, 2010, von Benda-Beckmann et al., 2006). Do specific factors or processes *enable (or fail to enable) specific investments and/or actions* towards specific tree-related social, economic and/or political aims? Qualities associated with specific 'forests' (or competing land cover) may be valued differentially by various actors (ATV). These actors, in turn, may have different leverages and means to achieve specific visions (ACP) either directly (e.g. through land ownership/control and financial and other capital investments) or indirectly (e.g. through legal frameworks and/or economic networks which facilitate or inhibit specific land uses).

4. Materials and methods

4.1. Production of 1973–2019 land cover maps from Sentinel and Landsat satellite images

To reconstruct historic landscape changes in TTHP we produced seven satellite-based land cover maps, for the years 2019, 2016, 2008, 1998, 1988, 1979 and 1973. These maps showed the extent of up to twenty land cover types (seventeen main types; cf. Table 1; cf. detailed descriptions below). We started with producing high-quality maps for 2016 and 2019, using ten Sentinel-2 satellite images per assessed year, available in the Google Earth Engine (GEE) platform (acquired from the Copernicus program of the European Space Agency; Tables SB1 and SB2, Supplementary Materials B). On the same GEE platform we set up high resolution Planet Dove images (Images © 2016 and 2019 Planet Labs PBC), and Google Earth base maps (pixel resolution in ranges of 0.5–3 m). We used the high resolution images and ground truth information (from field visits) to identify numerous polygons[14] of land areas for which the land cover type was known with a very high certainty (i.e. knowing the vegetation from field visits and/or visually identifying vegetation from the texture of the high resolution images and/or any other best available images and inferred sources). Each image was individually processed to derive a per-image classification. To do so, the set of polygons was used to produce overall random 300k training samples of pixels (maximally 25k per class, or all pixels if <25k, with a minimum of 1k). Using the selected data a new Random Forest classifier per image was trained, and subsequently used to classify the complete area. Using the stack of 10 classified images (taken at different seasons throughout each assessed year; cf. Table SB2, Supplementary Materials B), a robust classification was then produced taking the mode at each pixel. We applied several map calculation iterations, polygon-adjustments, and other trials, additions and/or corrections[15] until we found that the land cover maps were of a very high quality.

For earlier land cover maps we proceeded in a similar way. We used ten images/year acquired with Landsat-5 for 2008, 1998 and 1988, one Landsat-3 image for 1979, and five Landsat-1 images for 1973 (source USGS/NASA, available on GEE; Tables SB1 and SB2, Supplementary Materials B)[16]. We did not usually have any high-resolution images for those earlier dates, and we therefore mostly had to rely on visual interpretations of the satellite images per se (rendered in true and false colour). To facilitate such interpretations we proceeded in a chronological back-tracing fashion, starting with the 2008 stacked satellite image where we could identify polygon areas which – with a high likelihood – still contained the same land cover types as in 2016 and 2019. We then continued with the 1998 image (comparing this to the 2008 land cover map), then 1988, and so forth, up to the earliest image 1973.

We thus arrived at a 'baseline map' for each of the seven examined years. These maps were already of a fairly high quality, yet with an increasing reliability[17] over time. To increase the reliability of all the maps, we proceeded with a final 'cross-filter' step, assuming that 1.) more recent land cover maps were generally more reliable than older maps, and 2.) land areas which had the same type of land cover on a newer as well as an older map were more likely to be correctly classified than those that did not have the same type of land cover. We thus calculated 'cross-filter maps' (showing areas with identical land cover) starting with the images of 2019→2016 (then proceeding with 2016→2008, 2008→1998, etc.). We used these 'filter-maps' to randomly re-select more than 300k (450k for Sentinel-2) training pixels in preceding years and image, and thus produce even more accurate maps for 2019 and 2016 (and then 2008, 1998, etc.). The 1973, 1979 and 2008 maps still contained a few classification errors (induced by haze/clouds in mountain areas) which were corrected manually (converting to 'forest HB', cf. Table 1). The 'final maps' (and ~1966 map, cf. below) were then used for GIS-analyses in the study, whereby specific land cover change statistics were calculated in the GEE platform.

4.2. Production of the 'pre-war' land cover map (ca. 1965–1970)

A series of topographic maps (scale 1:50'000) produced by the U.S. Army in 1965–1970 (accessible through the Perry-Castañeda Library Map Collection, 2021) provided a fairly adequate and detailed resource showing mostly pre-war forest cover (additional notes see Supplementary Materials B). Using a drawing tablet we georeferenced and digitized the land cover shown on these maps (using ArcMap 10.8.1, Adobe Photoshop 2018 and MATLAB programs). We refer to this map as the '~1966 map'.

4.3. Description of the land cover types shown on the maps

In our study[18] we distinguished among different types of natural forest and non-forest cover (mostly in terms of biomass) and different types of tree plantation cover (in terms of economically used tree species). Land cover types and categories of 'forest cover change', as shown on maps (Figs. 2, 6–9), are summarized in Tables 1 and 2 (including relevant references; see also Supplementary Materials C).

4.3.1. Types of natural forests and woods

We distinguished three types of natural evergreen rainforest cover. The 'natural forest HB' (land cover type 1a; HB ~relatively high biomass)[19] may be considered as the most 'pristine' type of forest cover. It is characterised by tall trees of various lush-evergreen species. The 'natural forest MB' (type 2; MB ~medium biomass) is characterised by variably tall trees of mostly evergreen species, but possibly including light-demanding and deciduous species. These forests likely represent secondary forests which had previously been selectively logged. The 'natural forest HMB' (type 1b) represents mostly 'intact forest' which combines types 1a and 2 on the ~1966 map (Fig. 6a)[20]. The 'natural forest LB' (type 3; LB ~low biomass) represents a fairly 'degraded' secondary type of forest. It is characterised by small trees, often interspersed with bushes and thickets, likely representing regrowth on previously deforested sites or heavily logged sites[21]. Some inland tree stands adjoining river courses were classified as 'riparian forest' (type 5)[22]. In contrast to inland forests on mineral soils, 'coastal woodlands' (type 6; including forests, woodlands or bushlands) growing on sandy soils, along rims of saltflats, and/or along roads in coastal urban areas are often characterised by deciduous and/or sclerophyllous tree species[23].

4.3.2. Non-tree land cover types

The type 4 land cover ('vine-bamboo thickets') represents areas covered by a dense layer of small (2–3 m tall) bamboo and/or bamboo-like grasses, nowadays often overgrown by vines[24]. It established on cleared forestlands, e.g. in abandoned swidden areas or on war-impacted sites[25]. Type 11a ('brushwood/regrowth'), by contrast, may represent various types of patchy secondary bushlands or scrublands (i.e. woody plants interspersed with grasses), or – in more recent times – early 'bush-like' tree regrowth on rotational acacia plantations (or other types of newly planted plantations). Type 11b combines types 4 and 11a on the ~1966 map (Fig. 6a)[26]. Areas with even lower plant biomass (e.g. newly harvested acacia plots, open cassava fields, ruderal 'wastelands', etc.) are denoted as 'sparsely vegetated ground' (type 12). Furthermore, we identified completely 'bare lands' (type 13; gravel pits, dry river beds, hardened salt flats), 'sandy surface' (type 14; beaches, dunes, salt flats), and 'built areas' (type 17; road asphalt, buildings, similar 'hard surface'). Some treeless lands were perhaps used for agricultural activities[27]. The vast majority of type 15a land cover represents 'rice fields', in addition to some other vegetated swamps and shallow waters (e.g. creeks). Due to a long drought period during February-June 2019, we distinguished between wet (type 15a) and dry (type 15b) 'rice fields' for classifying land cover in the 2019 map (Fig. 2). Areas covered by 'open water' (type 16) included coastal lagoons, rivers, and lakes.

Table 1
Color key of the land cover types depicted in the maps. These include six types of natural forests or 'woods', four types of plantation 'forests' or tree groves, and seven types of non-tree land cover. Note that for 1966 types 2 and 4 could not be identified (cf. types 1b and 10b), and for the unusually dry year 2019 type 15 was classified as two sub-types, i.e. 'normal' (15a) and dry or drought-affected (15b). The names in the edged brackets are the vegetation types indicated on the 1966 + topographic maps. All land cover types are described and illustrated in Supplementary Materials C.

Map color RGB	No	Map Category / Type of Vegetation
Natural forests, bushlands and woody thickets		
0/80/50	1a	natural forests HB (medium-high biomass/upland rainforest)
20/100/25	1b	natural forests HMB (HB or MB, 1966 ['dense forest or jungle'])
40/120/0	2	natural forests MB (medium biomass/midland rainforest)
100/180/0	3	natural forests LB (low-biomass or dense bushland ['clear forest'])
200/250/50	4	vine-bamboo thickets (not distinguished in 1966, cf. 11b)
80/150/130	5	riparian forests or thickets
120/120/80	6	coastal woodlands (trees or bushes on sand, salt flats, roadside)
Planted 'forests' and tree groves		
70/30/5	7	village tree groves (roadside trees, tree orchards near villages)
90/50/130	8	pine tree plantations
100/100/200	9	rubber tree plantations
230/0/0	10a	acacia tree plantations (medium-size, rotational acacia stands)
150/0/0	10b	acacias with natural woody undergrowth (often old-growth acacia)
Diverse non-tree land cover		
230/140/50	11a	brushwood or plantation regrowth (interspersed with grass)
200/160/50	11b	vine-bamboo thickets or brushwood (in 1966 ['brushwood'])
220/200/120	12	sparsely vegetated ground (cut plantation, grass, open land)
255/255/255	13	bare ground (gravel pit, dry river bed, consolidated salt flat)
250/250/200	14	sandy surface (beach, sand dune, salt flat)
0/250/250	15a	rice fields (incl. any vegetated wetlands, swamps, shallow water)
150/200/200	15b	rice fields (dry harvested paddy fields or dry swamps in 2019)
0/0/250	16	open water (river, lake, sea)
0/0/0	17	built area (concrete, roads, buildings, may include hard salt flat)

Relevant references: Type 1a: Nguyen et al., 2016a, Ha 2015, Stas etal. 2020; T2: Van & Cochard, 2017, Cochard et al., 2018, Ha, 2015, Stas et al., 2020; T3: Do et al., 2010, Nicolic et al., 2008, Ngo & Webb, 2016, Tran et al., 2009; T4: Le et al., 2012; T5: Rundel, 1999, Moggridge & Higgitt, 2014; T7: Sam et al., 2004, Trinh et al., 2003, Wetterwald et al., 2004; T6: Tuan et al., 2022, Nehren et al., 2017, Veetil et al., 2021; T8: Imannudin et al., 2020; T9: To & Tran 2014, Ahrends et al., 2015; T10a: Cochard et al., 2021, Nambiar et al., 2015, Amat et al., 2010; T10b: Tran, 2014, McNamara et al., 2006, Wang et al., 2011 ; T11a: Nicolic et al., 2008, McElwee, 2008, 2009

Table 2
Color key of the natural forest land cover changes depicted in the 'forest change' maps. The numbers refer to 'no forest' (0), and to natural forest HB (1), MB (2) and LB (3) (cf. Table 1).

Map color RGB	<No>	Change in Natural Forest Cover
40/40/40	=1=	stable natural forest HB (no change)
100/0/100	1>2	natural forest HB to MB (~degradation)
160/0/180	1>3	natural forest HB to LB (~degradation)
250/0/0	1>0	natural forest HB to non-forest (deforestation)
0/150/0	1<0	non-forest to natural forest HB (reforestation)
0/120/120	1<3	natural forest LB to HB (~regrowth)
0/90/90	1<2	natural forest MB to HB (~regrowth)
70/70/70	=2=	stable natural forest MB (no change)
180/100/140	2>3	natural forest MB to LB (~degradation)
250/80/0	2>0	natural forest MB to non-forest (deforestation)
0/200/0	2<0	non-forest to natural forest MB (reforestation)
0/180/180	2<3	natural forest LB to MB (~regrowth)
100/100/100	=3=	stable natural forest LB (no change)
250/160/0	3>0	natural forest LB to non-forest (deforestation)
100/250/0	3<0	non-forest to natural forest LB (reforestation)
200/220/255	=0=	stable non-forest (no change)

4.3.3. Types of planted 'novel forests' or tree groves

Parklands, tree alleys and orchards in and around cities and villages were classified as 'village tree groves' (type 7)[28]. 'Pine tree plantations' (type 8; presumably mostly native *Pinus latteri*)[29] were set up as parklands in peri-urban areas around Huế City[30]. In rural locations pine plantations were used for timber or pulpwood and/or resin production. Since the 1980's, plantations of eucalypts[31] and later acacias (*Acacia auriculiformis*, *A. mangium*, nowadays mostly *A. mangium*×*auriculiformis* hybrids) were increasingly set up for the production of woodchips (for pulp and paper) and timber. 'Acacia tree plantations' (type 10a)[32] therefore mostly represent economically-managed short- to medium-rotation single-species acacia mono-cultures with a closed tree canopy, in contrast to type 11a which may represent freshly replanted acacias. In contrast to type 10a, the type 10b represents acacia 'forests' (often older-growth acacias established for watershed protection and/or timber production) where a natural woody undergrowth (trees/shrubs) has usually established, perhaps as a successional stage with a potential to eventually revert to rainforest. 'Rubber tree plantations' (type 9; introduced *Hevea brasiliensis*) usually represent mono-species plantations where any naturally establishing undergrowth is continually cleared during plantation management.

5. Results and discussion

5.1. A broad overview of the 'forest transition' 1966–2019 in Thừa Thien Huế Province

Whether or not one may say that a FT has taken place in TTHP between ~1966 and 2019 depends on specific interpretations. The development of an undifferentiated 'tree-dominated land cover' (Fig. 3; i.e. combining natural forest cover with the cover of all types of tree plantations *alias* 'planted forests') indeed somewhat follows a classic U-shaped FT curve (cf. Fig. 1), with the tree cover first declining from ~1966 (3289 km^2) to a low in 1979 (2915 km^2), and then steadily increasing again to 1998 (3236 km^2), reaching a recent maximum in 2019 (3513 km^2). It is clear, however, that most of this increase is due to a steady increase in economically important mono-species plantations from 37 km^2 in ~1966 to 994 km^2 in 2019 (Fig. 3)[33].

Taking original natural forest cover as the criterion for a FT, a somewhat different image emerges (Fig. 3; cf. Fig. 1). The overall natural forest cover did not substantially re-bounce after a steep decline from ~1966 (3122 km^2) until 1979 (2512 km^2). Rather, the cover continued to slightly decline until 2019 (2329 km^2). In particular the relatively intact 'natural forest HMB' steeply declined from ~1966 (2849 km^2) until 1988 (1546 km^2) and then mostly stabilized (with minor trends of recovery 1988–1998 and 2008–2019, and decline 1998–2008), covering 1664 km^2 in 2019 (Fig. 3). Officially reported data (GSO, 2022) are broadly within a similar range, but suggest a natural forest cover increase[34] from 1604 km^2 in 1993 to 2114 km^2 in 2019.

Various potentially relevant (e.g. for improved sustainable land management) spatial and temporal patterns and dynamics remain unseen in conventional FT-curves (Figs. 1 and 3). A closer look at the data, for example, reveals that most of the forest changes took place at lower elevations, whereas at > 600 m above sea level the forest cover has remained comparatively stable (Fig. 4a). Similarly, in relatively steeper terrain shifts in tree cover were less prominent than on more even

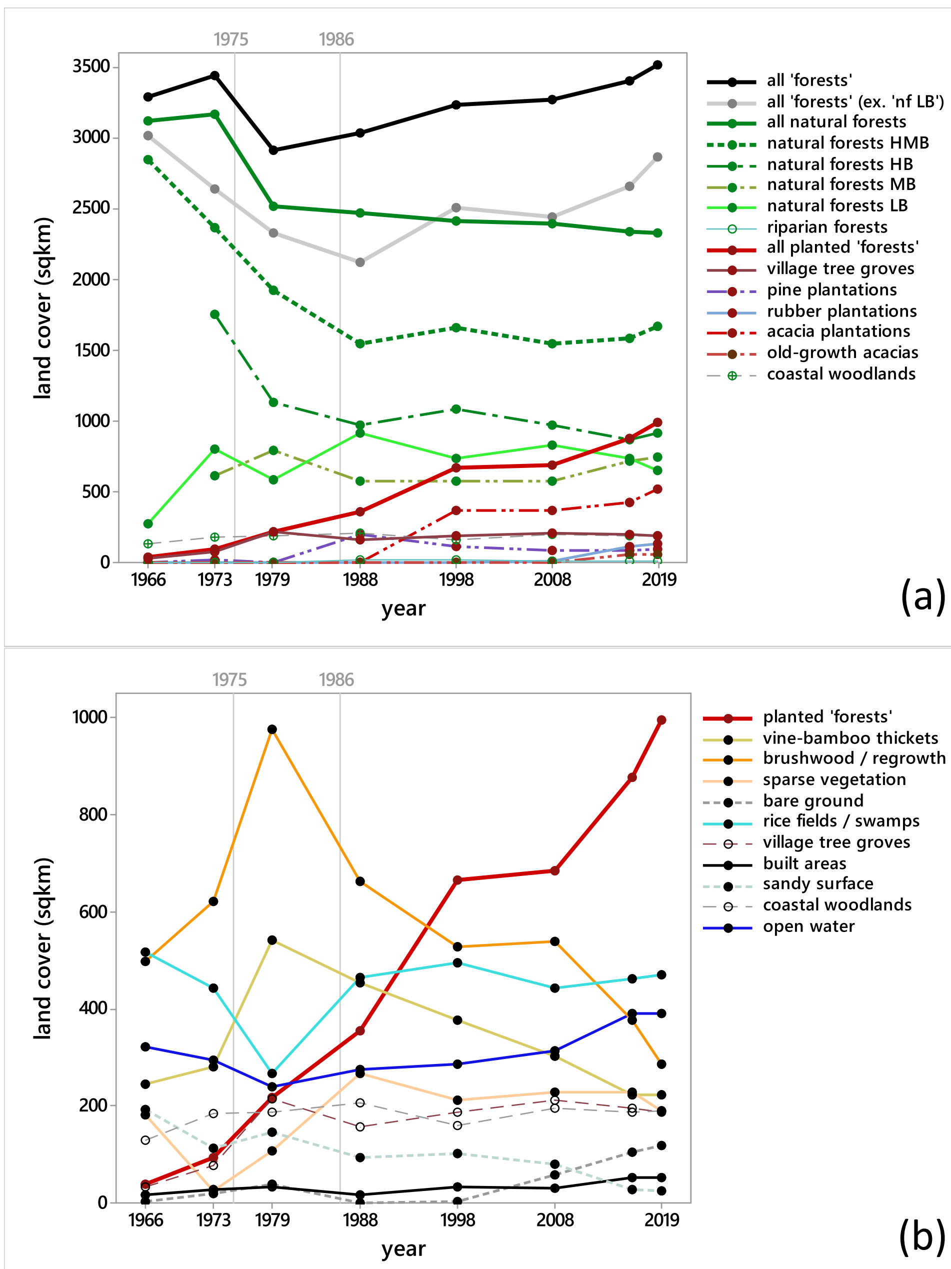


Fig. 3. Summary graphs of land cover changes from ~1966 until 2019, showing 'forest' cover (including natural forests and tree plantations) in the upper panel, and (mostly) non-tree land cover in the lower panel. The category 'all forests (ex. nf LB)' combines all tree cover except 'natural forest LB' which may be partly transitional from a structural 'forest' to 'thicket' or 'bushland'. Note that 'brushwood' and 'sparse vegetation' may represent cultivations, including harvested/regrowing acacia plantations (cf. description of vegetation types in the Methods).

grounds (Fig. 4b). This corresponds to studies at smaller scales of assessment which highlighted that it is mostly the rainforest biodiversity in well-accessible lowland areas which is at threat of being lost (Van and Cochard, 2017; Ha, 2015; cf. Wikramanayake et al., 2002). Figs. 3 and 4 however only show 'net outcomes'. In reality, land cover changes tend to be dynamic, and deforestation in one location within TTHP may be largely compensated by reforestation in another location (Table 3). A FT may therefore also be assessed in terms of the variable absolute shifts in different types of land cover, as visualized in Fig. 5. Such shifts can be relevant in terms of various types of intrinsic forest changes. Regrown forests likely differ from pristine forests in biomass, species composition (often simplified biodiversity), and forest structure (Chazdon, 2014).

There are different ways to assess the map data, and diverse possible viewpoints on whether specific observed changes may or may not be desirable and/or sustainable. In the following, we discuss the findings by taking a historic-grounded socio-environmental focus, combined with an interpretive focus on vegetation ecology. We may start with an imagined map which would show a dense unbroken forest cover in TTHP many thousand years ago, and discuss some 'transitions' until the 1960's (Fig. 6a), thus setting the scene for further events and developments during the specific FT-period.

5.2. *The natural forest cover of Thừa Thien Huế Province from 'forest-formative times' until the early 1960s*

Having evolved over millennia within a relatively stable monsoon-tropical climate (Woodruff, 2010; Meijaard and Groves, 2006), and located at a biogeographic intersection[35], the natural forest ecosystems

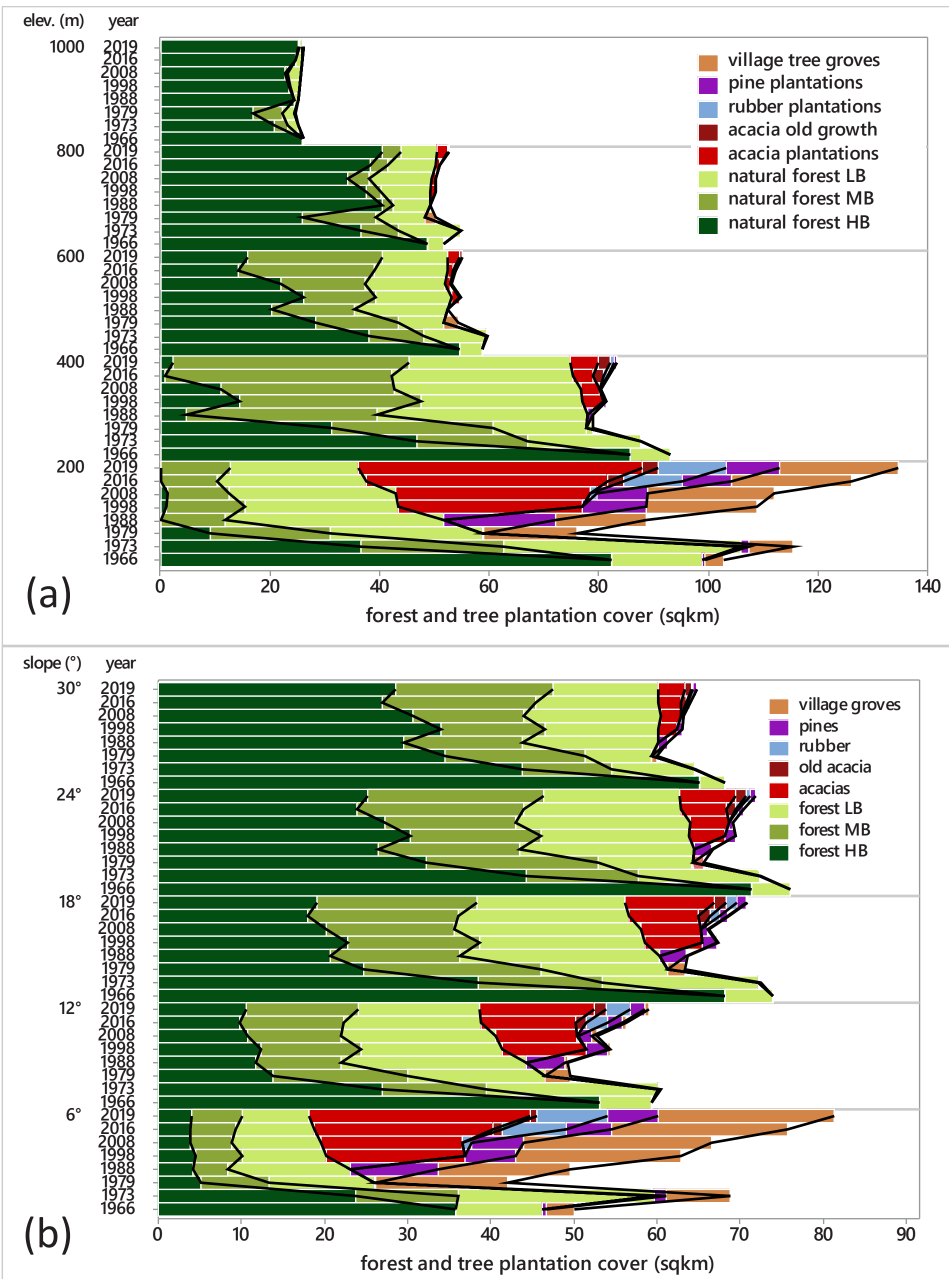


Fig. 4. Forest and tree plantation cover changes from ~1966 until 2019, as assessed by (a) elevation (bins are 200 m steps in elevation, from 0 to 200 m to 800–1000 m a.s.l.), and (b) terrain slope (bins are 6° steps, from 0°–6° to 24°–30° inclination). Note that the forest cover in 1966 could not be differentiated as forest MB or HB, but was probably mostly HB (here shown as forest HB). The figures are based on data from the land cover maps and a digital elevation model (DEM) from the NASA Shuttle Radar Topography Mission (SRTM; Farr et al., 2007).

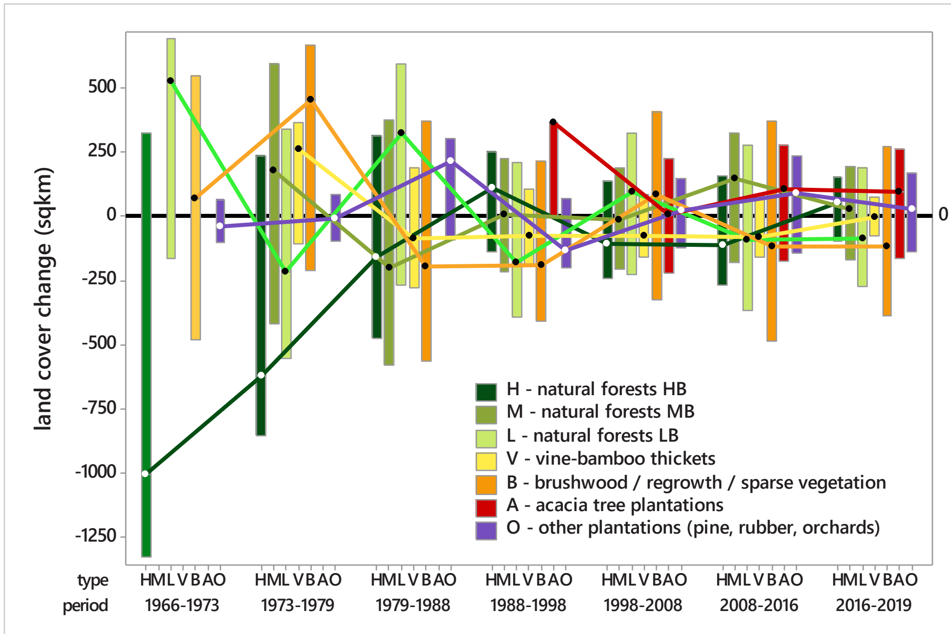


Fig. 5. Shifts in land cover in each of the assessed time periods. Bars represent the area (km^2) of a land cover added (>0 change) and lost (<0) during each period. The dots indicate the net change during that period (i.e. increases minus decreases in cover). For 1966–1973 forest cover was not differentiated as forest MB or HB (but was probably mostly HB), and 'vine-bamboo thickets' were not differentiated from 'brushwoods' (combined in one category).

of TTHP are exceptionally biodiverse and rich in endemic tree species (Van and Cochard, 2017; Morley, 2018; Hughes, 2017; Averyanov et al., 2003)[36]. This biodiversity implies that most tree species are rare and thinly dispersed, persisting within narrowly defined ecological niches (Cochard et al., 2018; Nguyen et al., 2016; Ashton, 2014; Wright, 2002). Accordingly, the forest ecosystems are endowed with a high resilience against plant pathogens, insect pests, and other inherently ecological risks (Sakschewski et al., 2016; Ennos, 2015; Coley and Barone, 1996)[37]. In these rainforests, intermittent creation of minor forest gaps occurs naturally – a process that fostered biodiversity over evolutionary times (Denslow, 1987)[38].

Larger-scale gap creation in rainforests represents a more recent, anthropogenic phenomenon[39]. In contrast to many temperate forests, regeneration of tropical rainforests after logging – partial or complete – is rarely straightforward, but may follow diverse successional pathways back towards a rainforest climax community[40], or towards an alternate vegetation state (Chazdon, 2014; Do et al., 2019, 2010; Nicolic et al., 2008; Fukushima et al., 2008). Increasing influences of human populations (through settlement, cultural advances and population growth) have thus produced new vegetation types and landscape configurations. The ~1966 map (Fig. 6a) shows the land cover of TTHP as it mostly looked[41] before the incisive impacts of the Second Indochina War. Rice fields and open woodlands or bushlands dominated the coastal lowlands already in the 19th century, and remaining near-coastal forests further declined, especially during critical times (e.g. conflicts, disease outbreaks, famines) (Biggs, 2018a, 2018b). Under the French colonial regime timber logging impacted some accessible forests in the midland hills[42], but most of the inland forests in TTHP probably remained out of reach.

Whereas the space between the forests and the coast was the realm of the rice-producing Chăm and Kinh (Việt) people[43], the forested uplands of TTHP largely remained a 'world apart'[44]. Despite age-old upland-lowland trade relations, the history of the Katuic upland peoples in TTHP is mostly off the written record (Hardy, 2009; Hickey, 1993)[45]. As described by Århem (2014)[46] the Cơ Tu peoples (and other related Katuic peoples) in upland TTHP lived in swidden areas immersed in a landscape which was dominated by natural forests and – in the sublime animistic experience of these peoples – many powerful forest-associated spirits[47].

Outsiders have often dismissed the 'slash-and-burn' (swidden) agricultural practices by the upland peoples as a wasteful and detrimental use of forests (McElwee, 2021, 2016; Cairns, 2017). In TTHP and nearby mountain regions, swidden fields were however not usually derived from pristine forest[48], but were cleared within old traditionally used swidden areas largely composed of variable matrices of swidden fallows (Århem, 2014, cf. also Robichaud et al., 2009)[49]. In the ~1966 map (Fig. 6a) most of the gaps within the forest layer may be assumed[50] to represent such traditional swidden areas[51].

The ~1966 map is not as fine-textured and adequately classified as the newer satellite-derived maps. Detailed examination of old aerial images[52] would likely reveal more variations in vegetation patterns. As described by Århem (2014), the most pristine forests (with tall emergent trees) were usually found on/around hills[53] and in higher mountain areas. More human-influenced parts of the landscape were characterised by secondary forests of diverse formations[54]. Yet, as shown in Fig. 6a, the landscape was still dominated by a contiguous layer of forest. Within this landscape wildlife was reportedly still quite abundant at least until before the mid-20th century (Århem, 2014)[55]. The forested landscape of TTHP essentially represented a rich bio-cultural landscape which was presumably characterised by a somewhat 'sustainable' socio-ecological balance. Regarding the FT-period after ~1966, decline and resurgence of 'forest' was mostly brought about by forces that were beyond the influence of the Katuic 'forest peoples'.

Anthropological accounts provide some glimpses into an 'archetypical world' of the Cơ Tu peoples at the end of the 1930's (Århem, 2014). Little is known about this upland world during the following two decades – a time which saw major strife starting in the 1940's[56]. During the First Indochina War (1946–1954) most of the rural areas in TTHP were under the control of the Việt Minh Front, which however mostly operated with popular support in the lowlands (Biggs, 2018a). Following the 1954 Geneva Accord, TTHP became part of the newly formed southern Republic of Vietnam (RVN), but TTHP was also located close to the northern Democratic Republic of Vietnam (DRV)[57]. The Saigon-based RVN government intermittently had some control of rural TTHP, establishing new civil outposts (e.g. the settlement of Nam Đông; Biggs, 2018a), and setting up airfields and army camps in the remote A Lưới

Valley[58]. In the 1950's unrest in the RVN[59] however led to new confrontations, the formation of the communist National Liberation Front (NLF) in 1960, with an increasing engagement of the DRV and – on different sides of the conflict – some foreign powers (most prominently the USA)[60] in an escalating and devastating war[61] during the 1960's and early 1970's

5.3. The impact of war, ca. 1960 until 1975

War represents suffering and sacrifice which fails any sensible description; such sorrowful events are interpreted and remembered (or put behind) in many different ways[62]. War not only leaves traces in memories or history; it also produces footprints in the bio-physical landscape which can persist in ways that may be relevant to post-war development and welfare, and associated aspects of environmental potentials and 'sustainability' (cf. Biggs, 2018a). 'Footprints of war' are produced as an outcome of a blunted logic, the 'logic of war'.

During the 1950's–1970's the forestlands of TTHP represented a place of covert movement and strategic retreat for the lesser technologically-equipped party of the war (i.e. the NLF/DRV)[63]. The more technologically-equipped party (USA/RVN), in contrast, sought to fight an enemy which remained largely elusive within a 'sea of trees'. The topographic maps (which we used as 'pre-war' data, Fig. 6a) were then produced in the context of the USA/RVN war effort which largely operated through airspace and which ultimately spent enormous resources to destructive effect[64]. To remove protective forest cover hundreds of thousands of gallons of tree defoliants (mainly Agent Orange)[65] were sprayed from airplanes during 1968–1971 along the main valleys and in hilly forestlands which were assumed to be conduits or strongholds of NLF/DRV combat units. Correspondingly, extensive aerial bombing missions targeted such areas (Biggs, 2018a, 2014)[66]. Despite some major operations and fierce battles in 1968/1969 involving ground troops in and around the A Lưới Valley[67], any US/RVN gains however remained transient[68].

Major reductions in forest density and cover between ~1966–1973 occurred in areas[69] which were heavily impacted by the establishment of army camps, the spraying of Agent Orange (and other defoliants), and by bombing missions (Fig. 6b; cf. figures of TTHP in Biggs, 2018a, 2014, Robert, 2011, Stellman et al., 2003). In contrast, forests on higher mountain areas and in the central Hữu Trạch River catchment remained comparatively intact (Fig. 6b)[70]. According to our data, closed-canopy 'forest HMB' was reduced by at least 17% (covering 2849 km^2 in ~1966 compared to 2361 km^2 in 1973; Fig. 3)[71], but forest destruction was probably higher: some forestlands in the plains of the A Lưới Valley were presumably already destroyed before the aerial images were produced for the topographic maps (Fig. 6a; cf. Supplementary Materials B)[72].

The first publicly available scientific descriptions provided a patchy image of direct environmental impacts of tree defoliants (Zierler, 2011). According to Westing (1971, 1976) virtually all forest trees were intermittently defoliated after one aerial spray of Agent Orange, but in upland forests usually 80–90% of the trees (i.e. lesser sensitive species)[73] survived one direct exposure. Rates of tree mortality however increased under repeated exposures (especially after brief intervals between sprays), with up to 85–100% tree mortality after four sprays. Impacts therefore ranged from complete destruction (mostly in relatively confined locations) to more transient effects (with large areas affected by one direct and/or drifting sprays), with highly variable consequences depending on specific local factors (Hay, 1983). From our map data it seems that some forest regeneration subsequently occurred between 1973 and 1979 in lesser affected areas (Fig. 6d)[74], possibly with shifts to more light-demanding faster-growing tree species (cf. Ashton, 1986). In contrast, some more severely affected forests further degraded (into 'forest LB' or thickets), mostly in interaction with increasing human uses and activities after the war (Figs. 6d, 7; cf. next section)[75]. Agent Orange targeted woody (dicot) species. Where tree canopies were permanently broken, monocot species were thus favored[76] (Westing, 1971; Do et al., 2019), and in some locations prolific growth of grasses and/or bamboo arrested forest regrowth (cf. van Kuijk, 2008, McNamara et al., 2006, Catterall, 2016). One may assume[77] that low-biomass forests (e.g. swidden fallow regrowth) converted to grasslands/bamboo under much lower herbicide exposure than old-growth forests. This may partly explain formation of grass/bamboo-dominated thickets also in areas where the surrounding 'forests HMB' were not much impacted[78]. Traces of dioxins (derived from Agent Orange) are still detectable in some forestland soils of TTHP (Banout et al., 2014). Knowledge about biological and environmental effects is still limited, but in most places (see however Le et al., 2019, My et al., 2021)[79] the chemicals probably no longer represent a determining factor for vegetation ecology. After herbicide exposures (or other war impacts) transformations of vegetation coincided with other consequential changes in soils (e.g. physical erosion and/or losses of organic matter and soil nutrients; cf. Westing, 1976, Sidle et al., 2006). Such combined ecological and pedological changes produced new vegetation formations and eco-systemic configurations (cf. Chazdon, 2014). Through vegetation regeneration and new land uses the 'footprint of war' eventually became more blurred; yet it is still widely traceable through vegetation biomass and associated indicators (cf. Dash and Curran, 2006). War-created spaces evidently exerted important influences on subsequent trajectories of change in the forestlands (in the past and up to the present; 'ambit of forestland-space-related factors and processes', ~AFS).

5.4. The post-war era 1975–1988: resettlement, salvaging wood, exploiting timber

End of April 1975 the guns were silent, yet war-torn re-unified Vietnam still found itself in adversity. The newly unified Socialist Republic of Vietnam (SRV) enacted new directions within a short period of time, and expressly so in the southern parts of the country[80], including TTHP. Following its 1976–1980 five-year plan the state organised farmworkers into cooperatives, terminated private property, trade and banking, and fixed prices for goods and services. The outcome was a further contraction of an already faint economy (Goscha, 2017). Within the collectivized systems many farmers (and labourers in other economic sectors) worked just enough land (or work units) sufficient for subsistence. Poverty increased and food crises started in 1978, lasting into the 1980's (Raymond, 2008; Goscha, 2017)[81]. Accordingly, our map data from TTHP show a decline of rice cultivation areas from 443 km^2 in 1973 to 268 km^2 in 1979 (Figs. 3b, 7a)[82].

Adding to this, Vietnam in this era was mostly isolated internationally[83]. The army remained mobilized in 1975 as new conflicts were on the horizon, which in December 1978 led to the Vietnamese occupation of Cambodia (lasting until 1989), and in February/March 1979 incurred a brief war with China (Goscha, 2017; Cesari, 1995)[84]. These conflicts no longer affected TTHP directly, but they came at significant costs for Vietnam[85] and – as with preceding conflicts – contributed to shape approaches to 'development' in the SRV and territories under its auspices[86].

Under such conditions, environmental concerns were overridden by other questions[87]. The SRV needed biofuel for energy uses, wood pulp for paper production[88], saw logs for reconstruction, and – with increasing importance – high-value timberwood for export and foreign exchange (McElwee, 2016; 'ambit of tree-valuation-related factors and processes', ~ATV). Moreover, the SRV envisioned people producing such resources in the socialist ways precast in the DRV[89] ('ambit of capital-political-related factors and processes', ~ACP). This implied significant human migrations and resettlements to and within forested regions. Upland ethnic minority people were to be relocated to specific state-defined spaces in valleys where they could form cooperatives and be introduced to 'advanced farming' (wet rice and other fixed-field crops to replace swiddening which was considered 'wasteful'; McElwee, 2021). Conversely, Kinh people from densely populated lowland areas[90]

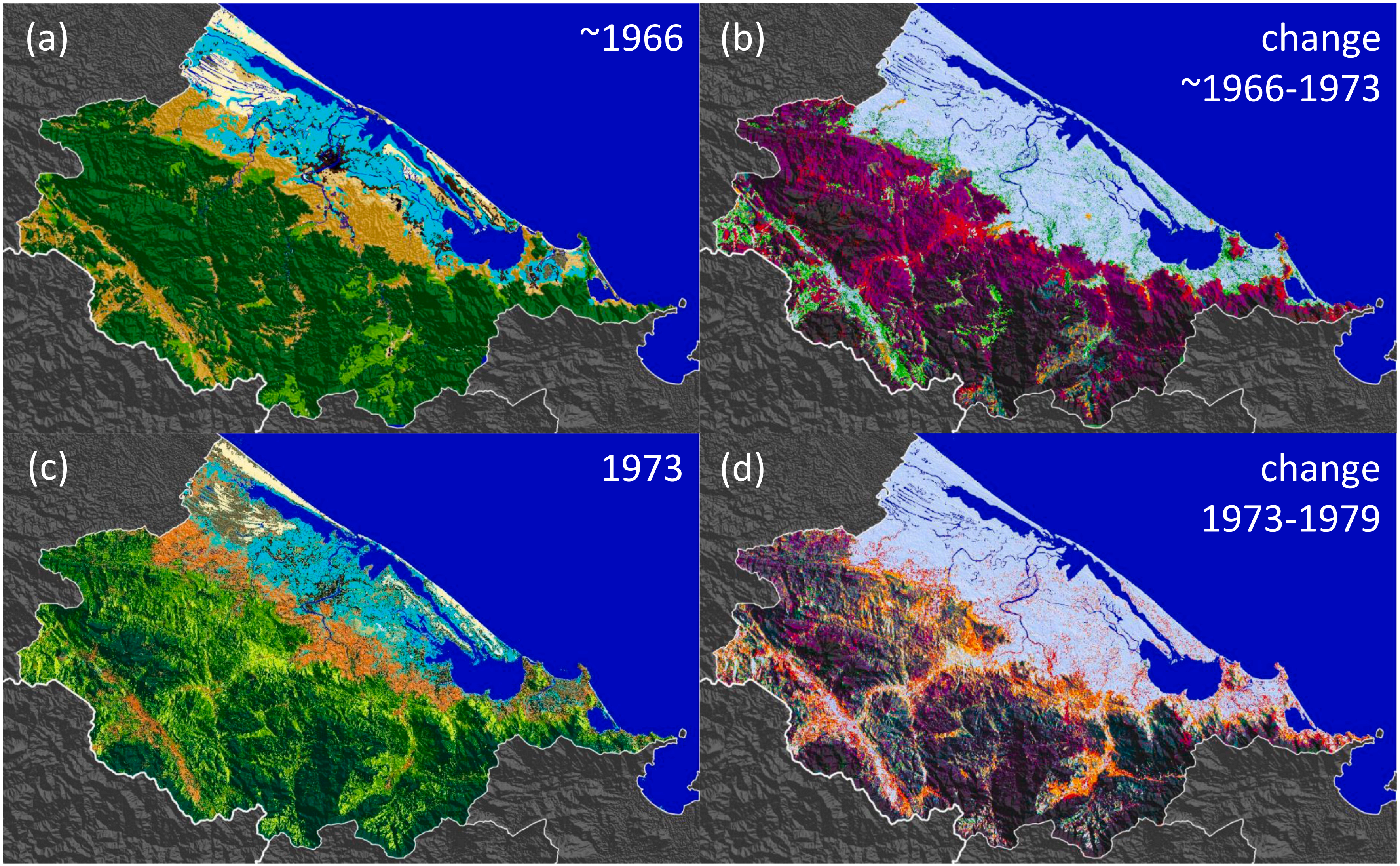


Fig. 6. Land cover ~1966 (a) and 1973 (b), and natural forest cover and transformations 1966–1979 (c, d). Color keys see Tables 1 and 2. The map of ~1966 does not distinguish between forest HB and MB (combined as 'forest HMB', cf. Table 1), but for calculating the change map ~1966–1973 we considered 'forest HMB' in ~1966 as 'forest HB'.

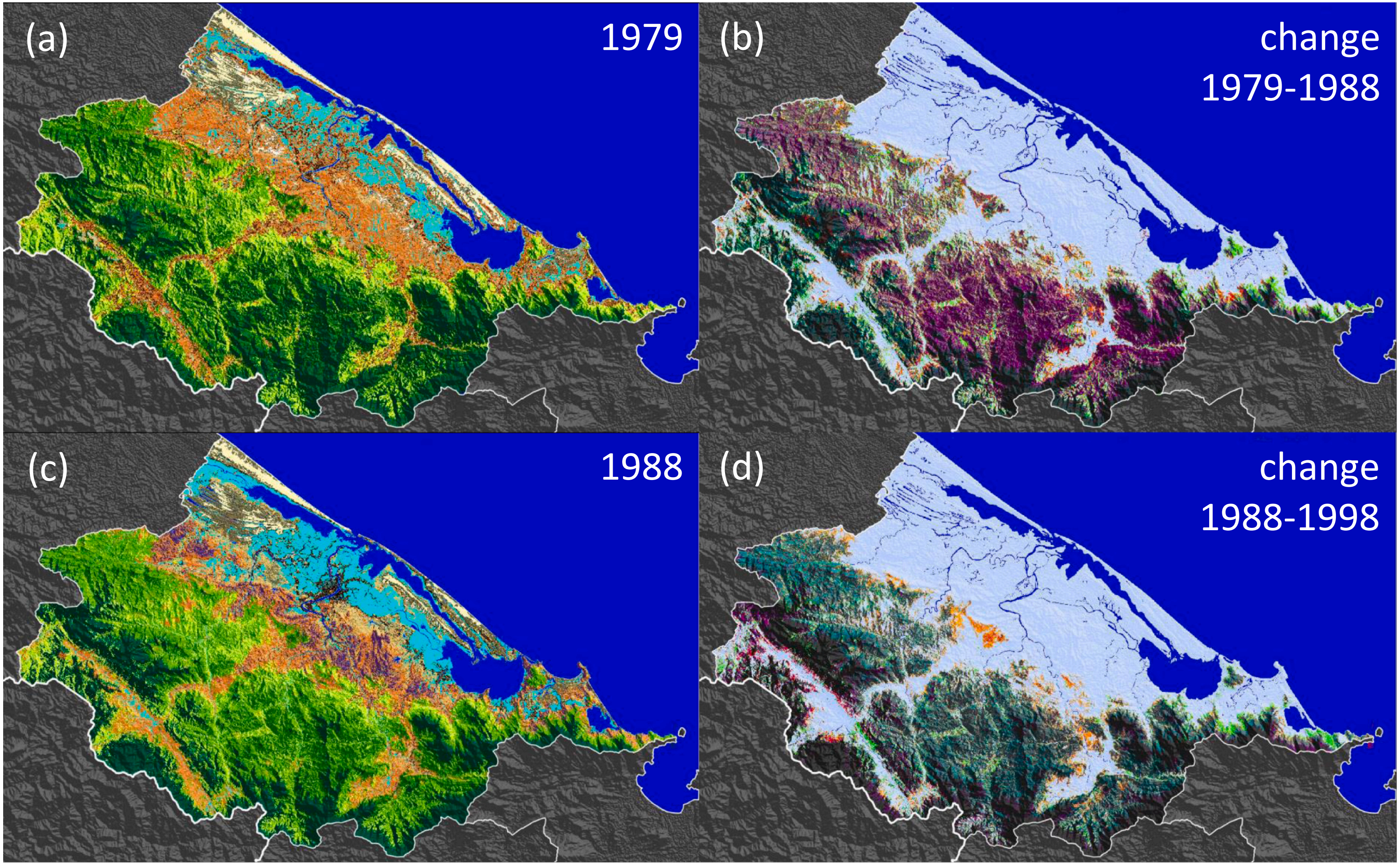


Fig. 7. Land cover 1979 and 1988, and natural forest cover transformations 1979–1998. Color keys see Tables 1 and 2.

were to be brought upland to help 'develop' the mountains via introducing new farming and forestry techniques. All types of forestlands (as the property of the state) were to be gainfully managed by state forest enterprises (SFEs)[91]. These SFEs were outfitted with technical equipment and mostly staffed with Kinh[92]. SFE logging targets were set by quota, and – as was usually the case – after exploitation many forest trees remained 'economically extinct' (McElwee, 2016).

Post-war changes (bio-physical and cultural) in the forested uplands differed widely across Vietnam, and were probably most far-reaching in the Central Highlands (Cochard et al., 2017; McElwee, 2016; Salemink, 2002). Within TTHP the changes were varied. Even though swiddening was officially banned since 1975 (Thiha et al., 2007)[93], some communities initially resumed traditional life in their old homelands[94], engaging in shifting cultivation (e.g. in the remote Hữu Trạch River catchment; Nguyen and Kull, 2022; Fig. 7)[95]. Many communities however had to resettle to other places, either because of war-time environmental destruction[96], and/or because – following the new policies – they were required to join the cooperatives (Århem, 2014; McElwee, 2016; Anh et al., 2016; Salemink, 2015). Newly founded settlements were usually at locations in the valleys or foothills where development of irrigated agriculture seemed feasible[97] (often in proximity of already existing settlements). Many of these locations were surrounded by degraded forests that were to be cleared for allowing new agricultural uses (e.g. pastures). Some clearing of (mostly degraded) forest between 1973 and 1979 (Fig. 6d) is therefore attributable to the resumption of human activities in and around former swidden areas (i.e. clearing of 'fallow forests') as well as to the expansion and establishment of settlements in newly opened spaces.

Newly established SFEs initially engaged in salvaging timber from trees killed from defoliation sprays[98], and then increasingly extracted timber in the more accessible lower-lying parts of the forests, and inland along the rivers (e.g. southern side of Rào Trăng River; Fig. 6d). Degraded forestlands were also a palpable source of firewood and charcoal for households and industries (McElwee, 2016; England and Kammen, 1993)[99]. Following wood exploitation, the degraded woodlands were often cleared via fire for pastures[100] or other agriculture (cf. Biggs, 2018b). Easily accessible residual forests in the lowland plains, the foothills, and in major valleys were the most likely forests to get cleared (Figs. 4, 6d, Table 3).

In contrast to the period 1973–1979, only relatively small areas were completely 'deforested' during 1979–1988; in remote locations (and higher elevations) the maps even show some forest regeneration on previously cleared areas (Figs. 4, 7b). In fact, in terms of an undifferentiated tree cover (i.e. including tree plantations), 'forests' were – according to our map data – already on the ascent, increasing from 2915 km^2 in 1979 to 3035 km^2 in 1988 (Fig. 3a). This incipient 'quasi-forest-area transition' (~FTa) can be interpreted as an outcome of processes relating primarily to spatial-material shifts in land use and productivity, and forest exploitation (allowing some room for spontaneous tree regrowth; ~AFS), and perhaps partly also to more stringent forest use restrictions and controls by SFEs and other state organs (processes related to ~ATV and ~ACP).

Some spatial shifts (~AFS) could perhaps be interpreted along rationales of Rudel et al.'s (2005) 'economic development pathway' which may however be better described as a 'political readjustment pathway'; this is because – at this incipient stage – the causal processes were mostly connected to a decisive policy shift rather than any new inherently 'economic' developments. During the food crises of the late 1970's, collectivization in rice agriculture had become untenable, and in 1981 – as a forerunner towards further political reforms (cf. next Section) – a contract system was introduced. This made work (and other investments) in the rice fields again rewarding (and attractive for labour), which lead to a dramatic increase in rice production. Already in 1989 Vietnam became a rice exporting country (Pingali and Xuan, 1992; Kompas et al., 2012), and in TTHP the rice cultivation area in 1988 (i.e. 464 km^2) was back to previous levels (cf. Figs. 3b, 7a)[101]. Similarly, during the 1980's some wood (mainly for fuel and paper production) was already produced in new tree plantations in the lowlands. This provided additional labour opportunities which probably also lead to some abandonment of comparatively unproductive marginal and/or degraded agricultural lands (cf. next Section).

During 1979–1988, the state (through SFEs) had consolidated its prime position as a 'forest manager' which enforced restrictions on traditional swiddening in forestlands (McElwee, 2016, 2021). Despite this, forest use restrictions and controls (~ATV, ~ACP) barely serve as an explanation for the observed increase in tree cover (~FTa). This is because the FTa during 1979–1988 in essence masks a massive deforestation at higher echelons of timber biomass. Intermittently (during the early to mid-1980's), there was no wood shortage (and labour opportunities in the wood industry), as large amounts of timberwood were extracted by SFEs via selective logging all throughout the lower-lying rainforests of TTHP (cf. Fig. 7b; cf. Lang, 2001, McElwee, 2016)[102]. In 1988 'forest HMB' reached its lowest extent (1546 km^2) which was mostly replaced by 'forest LB' (Fig. 3a, Table 3). Accordingly, 'forest HB' in 1988 became confined to higher elevations and/or steeper slopes in the mountains/hills; it has largely remained so since then (Fig. 4; Figures SE1-SE3, Supplementary Materials E).

5.5. *The Đổi Mới era 1986–2008: forestland zoning and allocations, 'illegal' logging, and tree planting campaigns*

The Đổi Mới reforms in 1986 heralded an era of 'new change'[103] that moved Vietnam's centrally planned economy to a socialist-oriented market economy. These reforms could be interpreted to have arisen at a critical time out of necessity[104] and/or out of incentivising changes taking place in other socialist countries[105] (Goscha, 2017; Hayton, 2010). Principal ideas for 'new change' had however been brewing up much earlier (Fjorde, 2007)[106].

Regarding the wood industry, the state certainly entertained an awareness about the dwindling timber stocks in rainforests, and some policy shifts were pre-tailored much before 1986. Trials of plantation forestry dating back to colonial and even pre-colonial times[107] had inspired ideas to 'reforest' extensive 'bare lands' (Biggs, 2018a, 2018b), and some tree planting campaigns, partly sponsored by the United Nations World Food Program[108], started during the 1970's (McElwee, 2016). In 1988 various tree plantations (pines, eucalypts[109], mixed tree orchards) already covered an area of around 354 km^2 in lowland TTHP (Figs. 3, 4, 7c; cf. also Disperati and Viridis, 2015)[110]. In the following decades the plantation areas however expanded considerably, reaching more than 994 km^2 in 2019 (Fig. 3)[111]. A specific variety of 'FT' took off along a 'pathway' of alien acacias and rubber trees. This post-1988 FT occurred not only in terms of tree-covered area (~FTa), but eventually also implied a net (mostly plantation-based) increase in wood biomass (~FTb).

The Đổi Mới policy change-over took place in a stepwise, and yet dynamic fashion. Along with this political 'transition', international resources were harnessed which facilitated the structural reforms. The first Land Law of 1988 allowed farmers control over fixed agricultural fields[112], further boosting rice production through investments and innovation[113] (Kompas et al., 2012). In the same year, laws were passed that allowed direct foreign investments in industries and increased decision-making powers of state-owned enterprises (SOEs) (Fjorde, 2007). In 1989 agricultural prices were allowed to fluctuate freely, and new fiscal and monetary measures were taken which brought inflation under control and facilitated exports. The banking system was reorganized for abetting capital savings and investments[114]. On the international arena, Vietnam pulled its troops out of Cambodia in 1989, thus saving much-needed resources. This also allowed it to normalize diplomatic relations with neighboring and Western countries[115], opening the doors for membership in regional and international political organizations (notably ASEAN in 1995), and in financial and trade organizations[116]; this – in turn – facilitated access to international aid,

Table 3
Summary of the land cover transitions (in hectares, as indicated from the maps) during different study periods. Some categories combine several land cover types, i.e. 'forests MB' (T2, T5), 'brushwood / sparse' (T11, T12, T13), and 'other tree plantations' (T7, T8, T9). In the periods starting with 1966 the 'forest HB' is 'forest HMB' and 'scrubwood / scarce' includes 'vine-bamboo thickets' (data in *italics*). Description of land cover types cf. Table 1. Data for types T6, T16 and T17 are not included (complete data see Tables S1-S13, Supplementary Materials D). Numbers in red color indicate likely natural forest degradation (or deforestation), in green color forest regeneration, in blue color other land cover changes.

Original land cover [percent cover of TTHP by year]	New land cover (or unchanged)	Land cover transitions over all assessed periods (data in hectares)							Pre-FT changes		Post-FT changes		Entire period	
		1966 -1973	1973 -1979	1979 -1988	1988 -1998	1998 -2008	2008 -2016	2016 -2019	1966 -1988	1973 -1988	1988 -2019	1998 -2019	1966 -2019	1973 -2019
▌forests HB	▌forests HB	*151759*	89700	65429	83018	83635	70585	76575	*89938*	74629	74792	76097	*85992*	70043
[1966: 54.1%]	› forests MB	*51867*	43088	33477	4920	13446	19731	5966	*56011*	39731	11345	22095	*71695*	47015
[1973: 33.3%]	› forests LB	*55202*	17119	12303	6929	9242	5404	2550	*77053*	34104	7192	7229	*53823*	24793
[1979: 21.5%]	› vine-bamboo thickets	*17657*	7713	1149	1177	240	181	123	*32262*	7520	1004	578	*16711*	5191
[1988: 18.4%]	› brushwood / sparse	*7215*	11618	494	613	768	994	756	*24621*	11556	1796	1186	*15566*	8934
[1998: 20.6%]	› acacia plantations	*6*	0	0	382	512	477	404	*1*	0	771	621	*27271*	9407
[2008: 18.5%]	› other tree plantations	*494*	484	141	7	53	60	13	*3286*	3538	13	26	*1911*	1946
[2019: 17.4%]	› rice fields	*432*	1606	96	6	52	57	7	*1024*	2408	41	44	*772*	2162
▌forests MB	› forests HB	*0*	17059	20137	16487	7844	6189	8247	*0*	10809	6349	5235	*0*	9648
[1966: -]	▌forests MB	*0*	19433	21359	37270	39311	40756	56267	*0*	14575	38257	38342	*0*	17252
[1973: 11.6%]	› forests LB	*0*	11407	31651	3498	9226	8643	6919	*0*	20276	9200	11186	*0*	12303
[1979: 15.0%]	› vine-bamboo thickets	*0*	5342	3493	108	84	131	39	*0*	5324	729	648	*0*	2975
[1988: 11.3%]	› brushwood / sparse	*0*	4999	1561	334	646	961	421	*0*	6369	1026	1075	*0*	4488
[1998: 11.2%]	› acacia plantations	*0*	0	0	818	2072	1141	918	*0*	0	2271	2016	*0*	8334
[2008: 11.1%]	› other tree plantations	*0*	456	614	368	94	202	249	*0*	1563	204	167	*0*	1125
[2019: 14.4%]	› rice fields	*0*	722	344	63	434	209	50	*0*	1387	108	215	*0*	1278
▌forests LB	› forests HB	6481	4812	7495	6571	4960	7864	4457	2295	8016	6303	7111	1690	7743
[1966: 5.2%]	› forests MB	2277	13946	2883	16157	4469	10091	11220	1830	3649	23129	12368	2365	9282
[1973: 15.2%]	▌forests LB	11024	24693	31943	52123	50570	46269	46443	7535	29705	34402	33533	5573	20214
[1979: 11.2%]	› vine-bamboo thickets	4494	21607	10765	6459	6015	6317	5004	5261	17748	8285	7859	2788	7864
[1988: 17.3%]	› brushwood / sparse	2705	10159	4703	3195	4352	5049	2372	8516	17632	5025	3789	5296	11097
[1998: 14.0%]	› acacia plantations	0	0	0	6477	2586	4811	2600	0	0	9655	5966	4910	14318
[2008: 15.8%]	› other tree plantations	157	83	843	313	150	1697	1677	1095	1781	493	236	962	1294
[2019: 12.4%]	› rice fields	206	388	146	22	43	108	27	470	840	176	108	672	1055
▌vine-bamboo thickets	› forests HB	*0*	405	2841	1511	473	261	131	*0*	1665	2928	1499	*0*	2085
[1966: -]	› forests MB	*0*	1105	363	136	77	206	119	*0*	197	1395	780	*0*	1001
[1973: 5.3%]	› forests LB	*0*	3652	11810	7633	6896	6527	4728	*0*	4890	10786	8557	*0*	6140
[1979: 10.3%]	▌vine-bamboo thickets	*0*	17343	26491	26957	21620	14230	14891	*0*	13142	11191	12039	*0*	5672
[1988: 8.6%]	› brushwood / sparse	*0*	3652	11529	6530	7225	4590	1246	*0*	6940	6658	5445	*0*	4161
[1998: 7.2%]	› acacia plantations	*0*	0	0	2415	1135	1907	710	*0*	0	7971	6341	*0*	5470
[2008: 5.7%]	› other tree plantations	*0*	57	593	15	93	1476	396	*0*	337	337	226	*0*	373
[2019: 4.2%]	› rice fields	*0*	191	222	95	122	199	36	*0*	432	289	210	*0*	488
▌brushwood / sparse	› forests HB	*12611*	914	671	361	254	1022	1950	*4495*	1649	1266	1190	*3697*	1865
[1966: 14.1%]*	› forests MB	*4434*	1416	617	638	307	654	820	*1028*	714	1278	660	*1794*	1235
[1973: 11.8%]	› forests LB	*13239*	1818	2319	2486	2652	4888	2708	*6681*	2156	3223	1976	*5833*	1583
[1979: 18.5%]	› vine-bamboo thickets	*5232*	1847	2289	2866	1509	1268	1253	*7667*	1402	874	636	*2668*	396
[1988: 12.6%]	▌brushwood / sparse	*39608*	45546	55754	52219	41615	33759	32275	*44467*	31778	25660	23459	*23305*	14160
[1998: 10.4%]	› acacia plantations	*0*	0	0	17739	11035	16028	17316	*0*	0	25403	19507	*23776*	16721
[2008: 10.2%]	› other tree plantations	*2620*	2256	18445	4080	5895	10646	6991	*17938*	13835	8270	5584	*10674*	8425
[2019: 5.4%]	› rice fields	*10050*	5409	21197	6043	3383	4970	2613	*5283*	10089	8073	4796	*5756*	10426
▌acacia plantations	› forests HB	0	0	0	0	265	369	251	0	0	0	525	0	0
[1966: 0%]	› forests MB	0	0	0	0	612	1601	1080	0	0	0	1404	0	0
[1973: 0%]	› forests LB	0	0	0	0	4187	1911	853	0	0	0	2634	0	0
[1979: 0%]	› vine-bamboo thickets	0	0	0	0	589	77	172	0	0	0	394	0	0
[1988: 0%]	› brushwood / sparse	0	0	0	0	13772	8123	10472	0	0	0	6547	0	0
[1998: 6.9%]	▌acacia plantations	0	0	0	0	14512	19897	31292	0	0	0	16942	0	0
[2008: 7.0%]	› other tree plantations	0	0	0	0	1364	2731	2474	0	0	0	2275	0	0
[2019: 10.9%]	› rice fields	0	0	0	0	457	663	261	0	0	0	395	0	0
▌other tree plantations	› forests HB	1127	54	31	31	64	38	23	0	51	31	24	0	54
[1966: 2.5%]	› forests MB	551	33	5	229	152	45	236	2	23	371	222	0	23
[1973: 1.8%]	› forests LB	241	29	20	490	144	129	1006	9	41	353	136	5	37
[1979: 3.6%]	› vine-bamboo thickets	158	40	20	53	13	28	672	25	23	44	16	1	15
[1988: 7.7%]	› brushwood / sparse	2913	5211	4133	5719	4193	6242	4811	3524	3284	9043	6486	2806	4176
[1998: 5.2%]	› acacia plantations	0	0	0	7446	3958	2008	2559	0	0	9652	4428	323	577
[2008: 5.7%]	▌other tree plantations	3053	10259	9905	20467	15287	15476	25179	1836	10763	12337	11593	2511	7006
[2019: 8.0%]	› rice fields	3928	1676	2959	1810	722	1330	1148	1774	3686	2019	837	1800	4108
▌rice fields	› forests HB	1895	85	73	21	12	26	46	0	95	27	12	1	104
[1966: 9.8%]	› forests MB	1417	106	95	452	104	94	123	0	45	121	31	2	50
[1973: 8.4%]	› forests LB	329	51	92	21	14	65	38	1	74	72	16	13	89
[1979: 5.1%]	› vine-bamboo thickets	384	95	76	27	41	32	27	21	70	33	16	9	33
[1988: 8.8%]	› brushwood / sparse	11036	23884	2230	1795	3741	2828	2241	9359	11698	2441	2591	3696	5179
[1998: 9.1%]	› acacia plantations	0	0	0	378	569	474	519	0	0	661	617	1047	2044
[2008: 8.4%]	› other tree plantations	2596	4590	2280	484	2146	1747	1424	3592	4728	2561	2568	3392	4723
[2019: 8.9%]	▌rice fields	25101	12051	18011	37326	36791	34348	39205	33530	22260	32108	33557	35376	22305

credit and investment, and opportunities for collaboration (Goscha, 2017).

As in other industries, the forestry sector underwent significant refitting. A malleable strategy aimed at 'timber restoration' via state-led spatial re-organization, mobilization of international funding, and the harnessing of rural labor. The Forest Protection and Development Law of 1991 stipulated the formation of three forestland management zones, namely 1.) production forestlands (designated for wood/timber production/exploitation), 2.) protection forestlands (for watershed/soil protection, and intermittent forest restoration), and 3.) special-use forestlands (mainly for nature conservation). In accordance, during the 1990's–2000's SFEs were dissolved and re-emerged as new state-owned forest organizations (SFOs), i.e. either state forest companies (SFCs) assigned to industrial forestry, or state forest management boards (SFMBs) for protection forests (SFMB-PFs) or for special-use forests (SFMB-SUFs) (Cochard et al., 2020).

SFMBs operated on narrow state funds. It was therefore rather opportune that the fallout of revenues and labor from logging occurred in an era when – after the Brundtland Report (WCED, 1987) – environmental concerns increasingly entered into international development discourses (McElwee, 2016). During the 1990's this combined with the (re-)discovery of the rich (yet endangered) fauna and flora of Vietnam, including – spectacularly – the description of large mammal species (e.g. the saola)[117] which had previously been unknown to the international scientific community (Sterling et al., 2006; Drollette, 2013). At a time when logging was rampant in neighboring Cambodia and Laos (with cross-border timber imports facilitating the Vietnamese FT; Meyfroidt and Lambin, 2009, Lang, 2001) the Vietnamese government pronounced several logging bans in natural forests (McElwee, 2016). Accordingly, the system of formally protected areas rapidly expanded with international support from 1435 km^2 in 1990 to 19'223 km^2 in 2002 (ICEM, 2003)[118]. Nature conservation is undoubtedly an essential issue in Vietnam, but logging bans and new conservation areas also served as useful implements to revive previous (or rationalize new) resettlement-for-development programs in the uplands. Similarly, the newly arising 'pro-forest' policy re-positioning served to problematize logging and other rainforest uses (especially swiddening) as primarily issues of local ('illegal') mis-management and poverty (~'backwardness', 'underdevelopment') rather than issues which in cause and effect were inextricably connected to the state-led wood industry and urban domestic and international demand for timber and other forest products (McElwee, 2004, 2006, 2016; Sowerwine, 2004). Within a context of growing economic 'metabolism' (domestically and linked to globalization) rainforest timber has become increasingly valuable (~ATV), incentivizing operations of 'illegal' logging during specific spatio-temporal windows of opportunity (Sikor and To, 2011; To et al., 2014; Kien and Harwood, 2016).

In contrast to SFMBs[119], SFCs were expected to become self-sustaining and to generate economic profits. With only a few SFEs still officially allowed to continue logging in natural forests[120] the transition from SFE to SFC was however rarely straightforward. Some SFEs were dissolved, but most SFEs – whilst initially often themselves engaged in 'illegal' logging[121] – were eventually kept afloat through internationally funded tree planting programs (~ACP; McElwee, 2016).

Program 327, called 'Greening the Barren Hills', was the first nationwide program under Đổi Mới, with funding of around 200 million US$ between 1992 and 1998[122]. This program was administered by SFOs which engaged households through a contract system[123]. Contractual conditions and payments for tree planting (initially usually eucalypts, later acacia; cf. below) were hardly attractive. A major reason why many households (mostly those with sufficient other income) nonetheless participated was an 'air of change' with a promise of newly arising income opportunities connected to trees (as a new agricultural crop; ~ATV) and – in particular – the associated permanent land tenure (~ACP) (McElwee, 2016, 2009; Cochard et al., 2021; cf. Sikor and Baggio, 2014).

A legal basis and mechanism for long-term forest land allocation (FLA)[124] to households was already set out by the second Land Law in 1993. In many regions (particularly in the uplands) implementation of FLA (via SFOs) however only started under Program 661 ('Five Million Hectare Reforestation Program') – the largest program with a total funding of > 1.5 billion US$ between 1998 and 2010[125]. Besides planned afforestations for industrial forestry (~three million hectares) and strategic landscape management (e.g. watershed protection, coastline stabilization; ~two million hectares) Program 661 integrated diverse (and partly conflicting) conventional development objectives[126]. The implementation of FLA further incentivized tree planting, but households connected to SFOs and/or active in previous tree planting campaigns were advantaged. In contrast, other households – usually the most poor and marginalized – saw communally used lands become enclosed as privately-owned new acacia 'forest' (McElwee, 2016; Cochard et al., 2021; Vu et al., 2023). This transition often exerted particularly negative livelihood effects on women (for whom non-timber forest products collected in open woodlands often provided an important income) and ethnic minority households which did not possess fixed farming lands (rice fields)[127] for their subsistence (Sikor and Nguyen, 2007; Bayrak et al., 2013; McElwee and Nghi, 2021; Pham et al., 2023).

With planting programs[128] starting in 1992 the 'planted' tree cover in TTHP grew rapidly from 354 km^2 in 1988 to 667 km^2 in 1998 (Fig. 3). Newly planted trees were mostly Australasian-originated species, initially often eucalypts, later almost exclusively acacias (*Acacia mangium* and *A. mangium*×*auriculiformis* hybrids)[129] (Cochard et al., 2021; McElwee, 2016, 2009). Most planting between 1988 and 1998 concentrated on hilly areas (especially west of Huế City) and on degraded slopes along the main valleys and in the foothills (e.g. along Road 49; Fig. 8a) – conforming with program aims of 'greening the barren hills' (~AFS, ~ACP). Yet, hardly any hills were completely 'barren'; the new 'forests' replaced brushwood, thickets, and – notably – some of the last rainforest patches close to Huế City (Table 3, Figs. 7d, 8a; cf. McElwee, 2009, 2016). Under Program 661, and ultimately further dynamized by FLA (~ACP) and an increasingly lucrative and booming acacia wood market (~ATV), plantations first expanded in other mid-/lowland areas (until 2008 especially closer to populated areas; Fig. 8c) and subsequently spread to the mountain valleys (Figs. 2, 4, 8–9a; cf. Cochard et al., 2021, 2020, 2017, Nguyen and Kull, 2022, Ngo and Webb, 2008). In addition to acacias, during the 2000's rubber plantations rapidly expanded around many villages[130] (especially in the foothills and in Nam Đông), covering around 134 km^2 in 2019 (Figs. 2, 3, 9a). This expansion was promoted by a government project of 'agricultural diversification' (~ACP) which itself was strongly incentivized by attractive international rubber prices that peaked at 6.2 US\$ kg^{-1} in February 2011 (~ATV) (Mai, 2016)[131].

During 1988–1998, some natural forests along the foothills were cleared for industrial tree plantations (~ATV). More inland, however, large expanses of logged-out rainforests apparently showed signs of recovery (Fig. 7d; cf. Fig. 5), with 165 km^2 of forest MB reverting to forest HB (compared to only 49 km^2 degrading from forest HB to MB), and 227 km^2 of forest LB reverting to forest HMB (compared to 104 km^2 HMB degrading to LB; Table 3). This most likely resulted from a shift of economic focus by SFOs[132]. First, exploitation of degraded rainforests was – during this time period – no longer economically lucrative (~ATV). Second, acacia-based 'reforestation' programs provided new incomes and opportunities for SFO workers (~ACP). Despite this, the 1988–1998 maps also indicate some continuing low-level selective logging in more accessible parts of forest HB (e.g. around A Lưới Valley), and after 1998 logging pressure in rainforest locally/temporally resurged (e.g. along newly constructed roads)[133]. While logging-induced losses in some locations were partly counterbalanced by some forest regeneration in other locations[134] (~AFS, Figs. 8, 9), the natural forests nonetheless somewhat continued to degrade (especially at lower elevations and in more even terrain, Fig. 4) with a HB:MB:LB ratio of 45:24:31 in 1998 vs. 41:24:35 in 2008 and 39:33:28 in 2019 (Table 3, Figs. 3, 4 and 5).

5.6. Socio-environmental transitions in TTHP, 1986–2008: additional considerations with relevance to 'sustainability'

In the initial years after Đổi Mới state organizations in TTHP set the ground (~ACP) for future political-economic land use zonings and structuring (production/protection forests, conservation areas, FLA) and development projects (in particular hydro-power lakes). For instance, Katuic communities which had still practiced traditional swiddening in remote areas (e.g. the Hữu Trạch River catchment; cf. Fig. 8)[135] were finally required to dislocate and 'sedentarize' in settlements in the main valleys and along roads (Nguyen and Kull, 2022; Boissière et al., 2009)[136]. In the resettlement areas land for irrigated rice production was often very limited and surrounding rain-fed lands were of poor quality for swiddening (which in any case was officially prohibited). In the first years after dislocation many resettled communities experienced increased hardship and food shortages – despite investments in agricultural 'development' through various programs (Århem, 2014; Beckman, 2011; Gomiero et al., 2000; cf. Krahn, 2005, Anh et al., 2016, Mai, 2016). During this period (and to some degree until today) many former swidden farmers tried to get by with the collection/selling of NTFPs and war-time scrap metals, and/or offering their labour to SFOs for tree planting work or sometimes 'illegal' logging in rainforests which was often sanctioned (perhaps even fostered) by SFOs (Boissière et al., 2011, 2009; Bayrak et al., 2013).

Poverty and marginalization, and the demise of traditional systems (and associated taboos) of forest land resource management often coalesced with 1.) increasingly attractive prices for wild forest products (timber, bushmeat, pets, NTFPs) spurred by demands of a growing and increasingly prospering urbanized population (~ATV), and 2.) newly emerging networks of illicit extraction and trade in wild products linked to new systems and controls of forest land access (~ACP) (Polesny et al., 2014; Wetterwald et al., 2004; Aldrich and Neale, 2020; Sandalj et al., 2016; Bullough et al., 2021). This socio-cultural 'transition' connects to a persistent 'within-forest' biodiversity crisis in TTHP (in particular defaunation; Tilker et al., 2019, Gray et al., 2018)[137]. Within this context, state-led establishment of protected areas alone – Bạch Mã National Park in 1991, Phong Điền Nature Reserve in 2003, and Saola Nature Reserve in 2013 – did not instil forms of nature conservation[138]

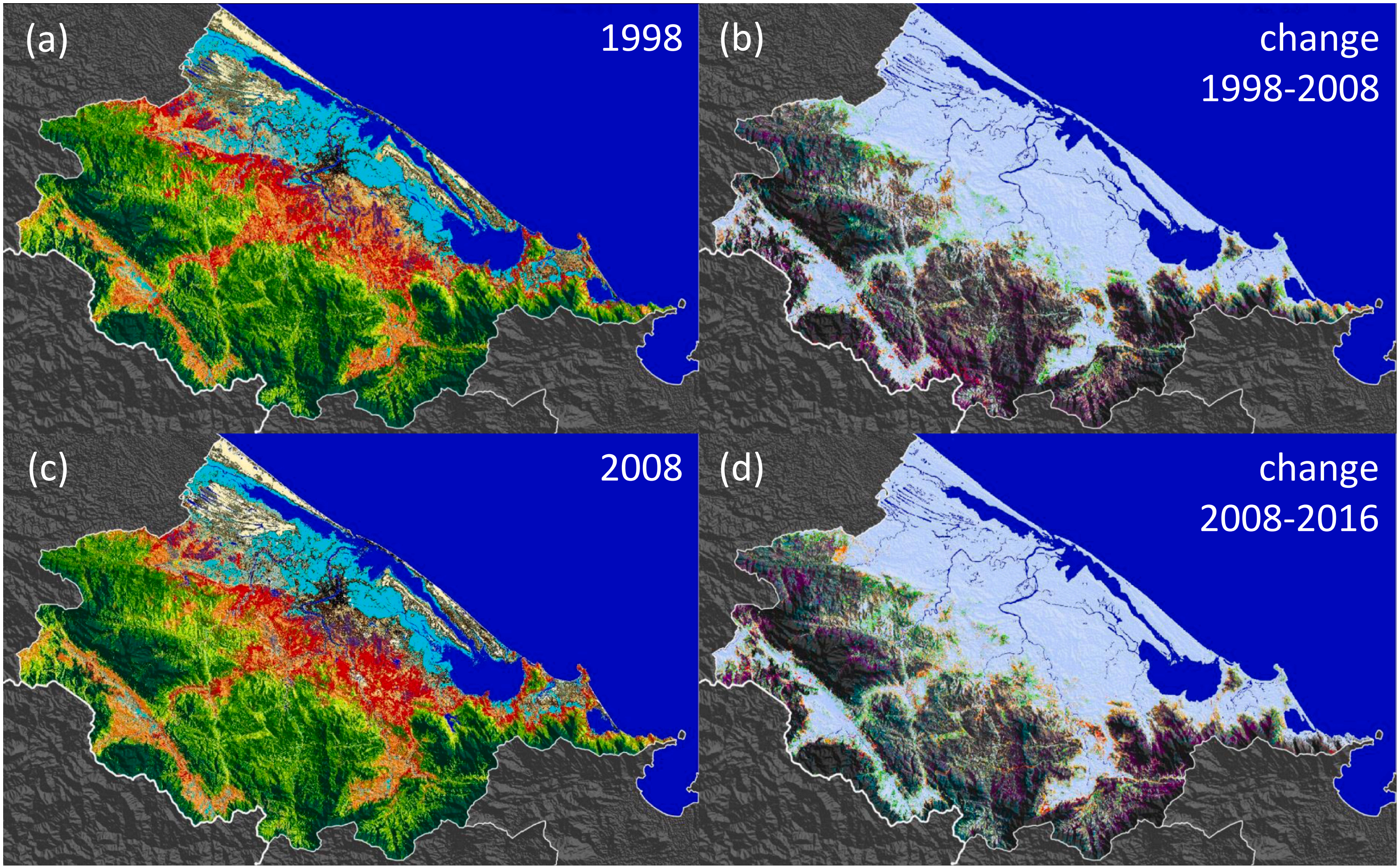


Fig. 8. Land cover 1998 and 2008, and natural forest cover transformations 1998–2016. Color keys see Tables 1 and 2.

that were 'naturally' countenanced by the original 'forest people'; yet, new forms of protected area co-management may eventually take shape (cf. McElwee, 2006, Boissière et al., 2009, Tuan et al., 2017, Huynh et al., 2016).

FLA was hardly a cure-all for negative effects incurred by resettlement and exclusion from forestlands set aside by the state for protection. In TTHP, FLA was administered by SFOs in a rather business-like way which often paid little attention to desires of local farming communities, especially with regard to formerly communally-used swidden areas (Gomiero et al., 2000; Bayrak et al., 2013; Nguyen and Kull, 2022)[139]. In a first phase (starting in the 1990's)[140], so-called 'degraded forestlands' (usually bushlands or swidden fallows) were allocated (DFLA; ~AFS, ~ACP) to households which were willing and able to plant eucalypt/acacia trees (Dung & Webb 2008). The first recipients were often relatively well-off households (mostly Kinh) with connections to SFOs and/or previous experience in plantation management (Cochard et al., 2021; McElwee, 2016; Sikor and Baggio, 2014). Communities of ethnic minorities initially had little interest in DFLA, because 1.) they traditionally had managed swidden areas communally rather than on privatized/fixed plots, 2.) they did not (yet) see much value in planting exotic trees (~ATV), and 3.) they lacked resources to start plantations on their own initiative (thus making them dependent on foreign capital; ~ACP) (Nguyen and Kull, 2022; Thulstrup, 2015). In Nam Đông District many Cơ Tu former swidden farmers reportedly sold (or rented out) their allocated 'degraded forestlands'[141], which further marginalized them in terms of land resources and work relations (Bayrak et al., 2015; Gomiero et al., 2000). Another move to allocate 'natural forestland' (NFLA) to local communities for shared forest uses and management (e.g. NTFP collection) was delayed by SFOs which still extracted timber from forests scheduled to become protected areas (Dung & Webb 2008)[142]. Within such politically and environmentally constrained contexts, the socio-economic value of acacia trees (in terms of possible livelihood profits and as a 'modern' means for reclaiming land) has since lead to a boom of acacia planting (~ATV), and associated land claims/struggles (~ACP) (Nguyen and Kull, 2022; Thulstrup et al., 2013; Sikor and Baggio, 2014; Pham et al., 2023; Vu et al., 2023). This largely explains the further spread of acacia plantations in upland regions of TTHP after 2008, partly with marginal encroachment on natural forestlands (Figs. 3, 4, 8c-d, 9a-b; Table 3).

5.7. The modern era, 2008 onwards: hydro-power, 'green' policy schemes, industrial forestry, pursuit of profit

Whilst manifesting an accentuated pro-environment rhetoric since Đổi Mới, the state has been primarily focused on fast-paced conventional infrastructure development (McElwee, 2016; Zingerli, 2005). In the uplands of TTHP, large reservoirs for irrigation, flood control, and hydro-electricity production have mushroomed (Figs. 2, 9a), until 2019 combining a hydropower capacity of more than 316 MW[143], with additional dams still under construction[144]. This contributed to economic growth and spurred a dynamic 'energy transition' (mostly in industrialising centres, and non-poor households) away from coal and biofuels towards 'cleaner' hydro-powered energy (Nguyen et al., 2019; Baltruszewicz et al., 2021)[145]; despite some challenging complexities it most likely also contributed to improved irrigation and flood risk management (Nguyen-Tien et al., 2018). Yet, hydropower development also came at a price in terms of externalities and new, unanticipated issues (e.g. river bank erosion) and risks (e.g. local earthquakes)[146]. By 2019 dammed lakes had inundated around 1170 km^2 of cultivated lands (tree plantations, rice fields, other croplands) and degraded forestlands (mostly forest LB and thickets; Table SD11, Supplementary Materials D), flooding riparian lands and obliterating river ecosystems many kilometres inland[147]. In total > 5000 people were resettled (Nguyen et al., 2017, 2016b)[148].

Within this context, Vietnam has been forging new policy schemes for 'Payments for Forest Environmental Services' (PFES). These schemes primarily served as mechanisms to generate new funds for SFOs and/or communities as forestland managers in watersheds upstream of hydropower plants (~ACP) (Cochard et al., 2020; To & Dressler, 2019). PFES pilot projects initially attracted valuable funding from international donors, including NGOs engaged in nature conservation. As a national policy set up in 2011, PFES however mainly served as a new type of value-added tax levied on the consumers of hydro-electric power (McElwee, 2016; McElwee et al., 2014)[149]. The distribution of PFES money to rural communities was managed largely through state organs, in particular SFOs, and has met with similar issues and challenges (elite capture and control, exclusion or non-participation of poor households, etc.) as previously FLA (Haas et al., 2019; Duong and de Groot, 2018; Nguyen et al., 2022; To et al., 2012).[150] Furthermore, there have hardly been any tangible assessments with regard to PFES policy influences on changing forest ecosystems and associated 'services' (To & Dressler 2019; McElwee et al., 2019).

Another infrastructure development was the expansion and upgrading of the road network, notably the Hồ Chí Minh Highway in 2006, and tracks and roads constructed during 2013–2016 that cut through prime natural forest in the Hữu Trạch and the Rào Trăng river catchments (Figs. 2, 8, 9)[151]. In the main upland valleys improved roads have facilitated the movement and trade of agricultural products, notably the transport of logs from the booming smallholder acacia plantations (~ATV).

Persistently attractive market prices for wood products have been driving the expansion of acacia plantations. Market demands for woodchips (domestic and for export)[152] seem almost bottomless, and even the poorest smallholders can now usually receive rewarding incomes for acacia wood sold to pulp-and-paper factories – provided they own (or have access to) some land (~ATV, ~ACP) (Tham et al., 2020; Nguyen and Kull, 2022; Vu et al., 2023).[153] Because of the equally booming Vietnamese furniture industries[154] disproportionately higher revenues could yet be gained from acacia sawlogs (Tham et al., 2021). Longer-term rotational acacia plantations however require markedly higher investments in terms of suitable land, labour, time (opportunity costs), risk taking[155], and technical means (Zhunusova et al., 2019). Consequently, SFOs and better-off tree farmers have again been in an advantaged position to produce wood for this high-profit market – with further shifting socio-environmental constellations that disfavor the already marginalized social groups (~ACP) (McElwee and Nghi, 2021). In this context, newly arising international timber certification policy and marketing schemes (such as by the Forest Stewardship Council)[156] have been opening access to potent 'green' markets – but again mostly tree farmers with large landholdings are involved and will profit (Cochard et al., 2021; Vu et al., 2023). Whether (and to what degree) such new types of 'green policy schemes' will effectively improve socio-environmental sustainability remains to be seen. It will largely depend on whether and how any such schemes will open up for types of participation considered socially and economically beneficial by rural households in the uplands. Alas, it seems that short-term profits from timber still drive the rationales of many SFOs – rather than any far-sighted landscape management for socially-grounded sustainable development. If not it would be difficult to explain why many old-growth acacia forests planted under Program 661 in 'protection forestlands' (some in steep terrain; Fig. 2) were in 2020 clear-cut by SFOs for saw-logs[157].

6. Summary and conclusions

As visualized in Fig. 8 (maps c, d), the time around 1988 divided a period of fast deforestation and forest degradation from more recent trends of increased forest stability and partial regeneration. Forest cover changes in TTHP did however not closely follow the conventional model and theoretical script of FT – if such changes are indeed to be considered to constitute a FT.

First, our study shows that substantial forest decline was hardly an

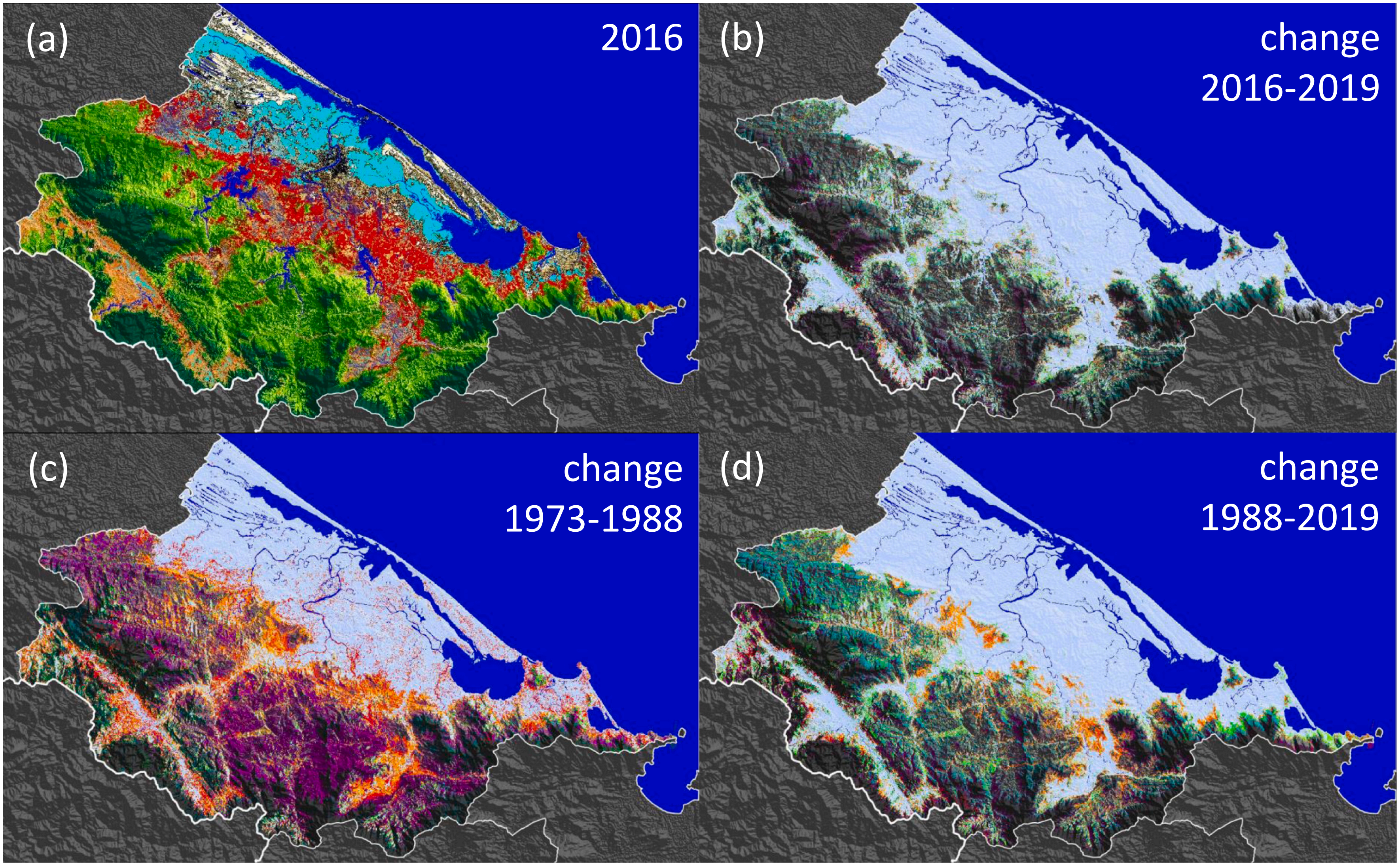


Fig. 9. Land cover 2016 and natural forest cover changes 2016–2019 (upper maps), and summary overviews of 'pre-transition' (1973–1988) and 'post-transition' (1988–2019) natural forest cover changes. Color keys see Tables 1 and 2.

outcome of 'poor' forest management by 'archaic' people who had subsisted by swiddening in confined, forest-surrounded spaces. Destruction of (or in) natural forests since ~1966 was mostly an outcome of powerful modern machineries, i.e. the violent impact of military devices (notably herbicides) during the war, and subsequently the extensive application of chainsaws.

Second, widespread logging of timber (mostly by SFEs/SFOs) did not usually directly lead to complete deforestation. In fact, our study revealed that a form of incipient forest-area transition (FTa) coincided with the period of most intense and widespread rainforest timber exploitation (1979–1988). This demonstrates that certain false assumptions can potentially arise from simplistic interpretations based solely on the U-shaped FT-curve (cf. Fig. 1).

Third, processes of deforestation occurred in steps and eventually coincided with widely transformative 'reforestation'. Repeated degradation of natural forests (often through multiple impacts) produced some types of lower-biomass forests, thickets or 'bush' which in more accessible lower-lying areas could later – step by step – be replaced with acacia monocultures which (as 'production forests') re-instated the specific 'ecosystem service' of wood (FTb). The FT has thus been characterized (and largely determined) by a bifurcation into spaces of newly planted tree cover (alien-species monocultures) vs remaining natural tree cover (mostly degrading or regenerating secondary rainforests). This implied significant changes in landscape qualities.

Fourth, transformative changes in tree cover were largely governed by ecological capacities and qualities of specific 'forest systems', and associated human valuations of 'forest'. At its roots, the 'bifurcation' has, on the one side, been driven by the persistently high value of wood-based products (planted ~ATV, ~economic demand for timber and woodchips) in interaction with the realization (from the industry perspective) that 'timber restoration' (~FTb) could be rapidly and conveniently achieved through industrially-managed acacia tree monocultures. On the other side, recognition (nationally and internationally) of the important role of rainforests for other types of 'ecosystem services' (hydrological functions, soil protection, biodiversity) has maintained a counter-balancing valuation of remaining natural forest cover (natural ~ATV, ~political demand for natural forest conservation)[158]. As a socio-environmental process, the 'bifurcation' has found its political expression in new state-defined forestland spaces for 'production' and 'protection' (including 'special-use'). Politically-defined forestland spaces were however malleable, and were dynamically adapted in interaction with effectively (bio-physically) available spaces and associated land use potentials during specific time periods (~AFS×ATV×ACP) – at regional (Cochard et al., 2020) as well as at more local scales (Nguyen and Kull, 2022; Cochard et al., 2021).

Fifth, capital investments (facilitated by political changes starting in the 1980's) were key to widespread rapid tree planting. This particular FT would hardly have taken place within such a short time period, were it not for the substantial capital input (from donors and credit institutions) which enabled the extensive national tree planting programs during the 1990's–2000's (~ACP). These programs also sustained the wood-based industry through a critical 'bottleneck' period, facilitated a socio-political restructuring, and provided incentives for tree planting which eventually went beyond any previous thresholds (Cochard et al., 2020; Vu et al. 2023).

What does the FT in TTHP imply in terms of a 'sustainability transition'? It can be questioned whether the FT in TTHP is in itself 'sustainable' (in the sense of being irreversible) and/or represents a genuine 'milestone' or 'beacon' of attaining socio-environmental 'sustainability'. Landscape transformative processes were largely driven by changeable political-economic visions of modernistic development as seen by key actors of the state. Within this context attention to important local aspects of 'sustainability', including community-based traditional environmental knowledge, understandings, and associated visions, has been marginal if not dismissive. As objects of changeable 'layer-upon-layer' state policies, local communities (particularly in the uplands) have experienced profound cultural alterations within the period of a lifetime. This evidently molded the communities' present-day perceptions and understandings of 'development', their subjectivities, and their relationship with the state (McElwee, 2016; Nguyen and Kull, 2022; Anh et al., 2016; To et al., 2015)[159]. Yet, the recent acacia boom has also opened new economic opportunities, allowing for improved incomes and livelihoods among many previously uprooted and poverty-stricken households. If such improvements can be harnessed to consolidate the agricultural basis, to strengthen social equity and cohesion, and to diversify rural enterprise and labor opportunities, this may allow to relieve some effects of past aberrations (Cochard et al., 2021).

According to our map data, the land area covered by trees in 2019 (3513 km^2) exceeded the forested land cover in ~1966 (3289 km^2). Yet, the 'bifurcation' and inherent changes in natural forests have evidently produced significantly different configurations of forest (or forest-like) ecosystems (Fig. 3), and associated alterations in ecosystem services. Provisioning services of wood and timber have shifted to tree plantations, yet continue at minor levels of extraction[160] in natural forests (mostly 'illegal' logging of high-value timbers). NTFPs, in contrast, are hardly found within 'novel forests'. NTFPs (as well as rare timbers) are closely linked to the immensely rich biodiversity of natural forests (Van and Cochard, 2017; Dao et al., 2016), and still provide an important income source for many of the poorest (often landless) households in the uplands (Wetterwald et al., 2004; Huynh et al., 2016; Polesny et al., 2014)[161]. Several natural forest products and resources have, furthermore, a high cultural and/or scientific significance (Cochard, 2017; Sterling et al., 2006). Natural forests – as an abode of spiritual beliefs and subjective experiences – are in any case far superior to planted forests in terms of cultural ecosystem services (Århem, 2014; McElwee et al., 2022). Whereas cultural aspects may not have been highly prized in the utilitarian thinking of recent times, water- and climate-related regulating services have mustered cogent arguments for forest protection. Such 'services' are non-trivial, and still relatively poorly understood in the case of Vietnam (McElwee, 2016; cf. Cochard, 2013, Ziegler et al., 2004, Sidle et al., 2006). Despite this, specific regulating functions of rainforests seem self-evident, and associated debates regularly surface after extreme events, e.g. following the record floods and multiple landslides in TTHP resulting from a sequence of typhoons in October 2020 (Luu et al., 2021; Tien et al., 2021). Such events also highlight the differential vulnerability and resilience of disparate tree-based ecosystems under naturally occurring impacts, ranging from storm winds and droughts to plant diseases (Beckman and Nguyen, 2016; Nambiar et al., 2018; Vu et al. 2023). The acacia plantation boom will eventually meet with social and ecological limits (Cochard et al., 2021). Visions for future socio-environmental development and welfare will have to balance out legitimate individual aspirations for improved livelihoods and economic opportunities, overall societal needs for biological conservation and environmental protection (within a context of increasing climate change), and the continued nurture of a rich bio-cultural heritage.

CRediT authorship contribution statement

Roland Cochard: Conceptualisation, Methodology, Validation, Investigation, Data curation, Visualisation, Writing – original draft, Writing – review & editing, Funding acquisition, Project administration. **Mathieu Gravey**: Conceptualisation, Methodology, Software, Validation, Formal analysis, Investigation, Resources, Data Curation, Visualisation, Writing – review & editing. **Luiz Gustavo Rasera**: Conceptualisation, Methodology, Validation, Software, Formal analysis, Investigation, Resources, Data Curation, Visualisation, Writing – review & editing. **Grégoire Mariethoz**: Writing – review & editing, Project administration. **Christian Kull**: Writing - review & editing, Funding acquisition, Project administration.

Declaration of Competing Interest

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

Data Availability

Data are available at: https://zenodo.org/record/8325706

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.landusepol.2023.106887.

References

Ahrends, A., Hollingsworth, P.M., Ziegler, A.D., Fox, J.M., Chen, H., Su, Y., Xu, J., 2015. Current trends of rubber plantation expansion may threaten biodiversity and livelihoods. Glob. Environ. Change 34, 48–58.
Aldrich, B.C., Neale, D., 2020. Pet macaques in Vietnam: an NGO's perspective. Animals 11, 60.
Amat, J.P., Phung, T.B., Robert, A., Tran, H.N., 2010. Can fast-growing species form high-quality forests in Vietnam? Examples in Thua Thien-Hue province. Bois For. Trop. 305, 67–76.
Angelsen, A., 2010. Policies for reduced deforestation and their impact on agricultural production. Proc. Natl. Acad. Sci. 107 (46), 19639–19644.
Anh, T.N.M., Kim, D.-C., Ubukata, F., 2016. Vietnam borderland – case study of an ethnic minority group in Central Vietnam. Belgeo 4, 1–25.
Århem, N., 2014. Forests, spirits and high modernist development. A study of cosmology and change among the Katuic peoples in the uplands of Laos and Vietnam. Upps. Stud. Cult. Anthropol. 55, 463.
Ashton, P., 2014. On the forests of tropical Asia. Lest the memory fade. Kew Publishing, Royal Botanic Gardens, Kew, UK.
Ashton, P.S., 1986. Regeneration in inland lowland forests in South Viet-Nam one decade after aerial spraying by Agent Orange as a defoliant. Bois For. Trop. 211 (1), 19–34.
Averyanov, L.V., Loc, P.K., Hiep, N.T., Harder, D.K., 2003. Phytogeographic review of Vietnam and adjacent areas of Eastern Indochina. Komarovia 3, 1–83.
Baltruszewicz, M., Steinberger, J.K., Ivanova, D., Brand-Correa, L.I., Paavola, J., Owen, A., 2021. Household final energy footprints in Nepal, Vietnam and Zambia: composition, inequality and links to well-being. Environ. Res. Lett. 16 (2), 025011.
Banout, J., Urban, O., Musil, V., Szakova, J., Balik, J., 2014. Agent Orange footprint still visible in rural areas of Central Vietnam. J. Environ. Public Health 2014 ID528965, 10.
Barbier, E.B., 2011. Capitalizing on nature. Ecosystems as natural assets. Cambridge University Press, UK.
Barbier, E.B., Tesfaw, A., 2015. Explaining forest transitions: the role of governance. Ecol. Econ. 119, 252–261.
Barbier, E.B., Burgess, J.C., Grainger, A., 2010. The forest transition: towards a more comprehensive theoretical framework. Land Use Policy 27, 98–107.
Barbier, E.B., Delacote, P., Wolfersberger, J., 2017. The economic analysis of the forest transition: a review. J. For. Econ. 27, 10–17.
Bayrak, M.M., Tu, T.N., Burgers, P., 2013. Restructuring space in the name of development: the sociocultural impact of the Forest Land Allocation Program on the indigenous Co Tu people in Central Vietnam. J. Political Ecol. 20, 37–52.
Bayrak, M.M., Tu, T.N., Burgers, P., 2015. Formal and indigenous forest management systems in Central Vietnam. Implications and challenges for REDD+. In: Cairns, M.F. (Ed.), Shifting cultivation and environmental change: Indigenous people, agriculture and forest conservation. Earthscan. Routledge, London, pp. 319–334.
Beckman, M., 2011. Converging and conflicting interests in adaptation to environmental change in central Vietnam. Clim. Dev. 3, 32–41.
Beckman, M., Nguyen, M.V.T., 2016. Upland development, climate-related risk and institutional conditions for adaptation in Vietnam. Clim. Dev. 8 (5), 413–422.
Biggs, D.A., 2014. Frame DS1050-1006DF129: March 20, 1969. Environ. Hist. 19 (2), 271–280.
Biggs, D.A., 2018a. Footprints of war: militarized landscapes in Vietnam. University of Washington Press, Seattle.
Biggs, D.A., 2018b. Clearing, "wasting", and regreening: an environmental history of bare hills in Central Vietnam. J. Asian Stud. 77 (4), 1037–1058.
Boissière, M., Sheil, D., Basuki, I., 2011. A booming trade? How collection of war residues affects livelihoods and forest in Vietnam. Int. For. Rev. 13, 404–415.
Boissière, M., Sheil, D., Basuki, I., Wan, M., Le, H., 2009. Can engaging local people's interests reduce forest degradation in Central Vietnam? Biodivers. Conserv. 18, 2743–2757.
Bullough, L.-A., Nguyen, N., Drury, R., Hinsley, A., 2021. Orchid obscurity: understanding domestic trade in wild-harvested orchids in Viet Nam. Front. Ecol. Evol. 9, 631795.
Cairns, M. (Ed.), 2017. Shifting cultivation policies: balancing environmental and social sustainability. CABI, Wallingford, UK.
Catterall, C.P., 2016. Roles of non-native species in large-scale regeneration of moist tropical forests on anthropogenic grassland. Biotropica 48 (6), 809–824.
Cesari, L., 1995. L'Indochine en guerres, 1945–1993. Belin, Paris.
Chazdon, R.L., 2014. Second growth. The promise of tropical forest regeneration in an age of deforestation. The University of Chicago Press, Chicago.
Chazdon, R.L., Brancalion, P.H.S., Laestadius, L., Bennet-Curry, A., Buckingham, K., Kumar, C., Moll-Rocek, J., Guimarães Vieria, I.C., Wilson, S.J., 2016. When is a forest a forest? Forest concepts and definitions in the era of forest and landscape restoration. Ambio 45 (5), 538–550.
Cochard, R., 2013. Natural hazards mitigation services of carbon-rich ecosystems. In: Lal, R., Lorenz, K., Hüttl, R.F., Schneider, B.U., von Braun, J. (Eds.), Ecosystem services and carbon sequestration in the biosphere. Springer, Heidelberg, pp. 221–293.
Cochard, R., 2017. Scaling the costs of natural ecosystem degradation and biodiversity losses in Aceh Province, Sumatra. In: Shivakoti, G., Pradhan, U., Helmi (Eds.), Sustainable natural resources management in dynamic Asia. Volume I: Redefining diversity and dynamics of natural resources management in Asia. Elsevier, Amsterdam, Netherlands, pp. 231–271.
Cochard, R., Van, Y.T., Ngo, D.T., 2018. Determinants and correlates of aboveground biomass in a secondary hillside rainforest in Central Vietnam. N. For. 49 (3), 429–455.
Cochard, R., Vu, B.T., Ngo, D.T., 2021. Acacia plantation development and the configuration of tree farmers' agricultural assets and land management – a survey in Central Vietnam. Land 10, 1304.
Cochard, R., Ngo, D.T., Waeber, P.O., Kull, C.A., 2017. Extent and causes of forest cover changes in Vietnam's provinces 1993-2013: a review and analysis of official data. Environ. Rev. 25 (2), 199–217.
Cochard, R., Nguyen, V.H.T., Ngo, D.T., Kull, C.A., 2020. Vietnam's forest cover changes 2005-2016: Veering from transition to (yet more) transaction? World Dev. 135, 105051.
Coley, P.D., Barone, J.A., 1996. Herbivory and plant defenses in tropical forests. Annu. Rev. Ecol. Syst. 27, 305–335.
Dao, T.H.H., Saborowski, J., Hölscher, D., 2016. Patterns of tree community differences in the core and buffer zones of a nature reserve in north-western Vietnam. Glob. Ecol. Conserv. 8, 220–229.
Dash, J., Curran, P.J., 2006. Relationship between herbicide concentration during the 1960s and 1970s and the contemporary MERIS Terrestrial Chlorophyll Index (MTCI) for southern Vietnam. Int. J. Geogr. Inf. Sci. 20 (8), 929–939.
De Jong, W., 2010. Forest rehabilitation and its implications for forest transition theory. Biotropica 42 (1), 3–9.
Denslow, J.S., 1987. Tropical rainforest gaps and tree species diversity. Annu. Rev. Ecol. Syst. 18, 431–451.
Disperati, L., Viridis, S.G.P., 2015. Assessment of land-use and land-cover changes from 1965 to 2014 in Tam Giang-Cau Hai Lagoon, central Vietnam. Appl. Geogr. 58, 48–64.
Do, H.T.T., Grant, J.C., Trinh, N.B., Zimmer, H.C., Tran, L.D., Nichols, J.D., 2019. Recovery of tropical moist deciduous dipterocarp forest in Southern Vietnam. For. Ecol. Manag. 433, 184–204.
Do, T.V., Osawa, A., Thang, N.T., 2010. Recovery process of a mountain forest after shifting cultivation in Northwestern Vietnam. For. Ecol. Manag. 259, 1650–1659.
Drollette, D., 2013. Gold rush in the jungle: the race to discover and defend the rarest animals of Vietnam's "lost world". Crown Publishers, New York.
Duong, N.T.B., de Groot, W.T., 2018. Distributional risk in PES: exploring the concept in the Payment for Environmental Forest Services program, Vietnam. For. Policy Econ. 92, 22–32.
England, S.B., Kammen, D.M., 1993. Energy resources and development in Vietnam. Annu. Rev. Energy Environ. 18, 137–167.
Ennos, R.A., 2015. Resilience of forests to pathogens: an evolutionary ecology perspective. Forestry 88, 41–52.
Farr, T.G., Rosen, P.A., Caro, E., Crippen, R., Duren, R., Hensley, S., Kobrick, M., Paller, M., Rodriguez, E., Roth, L., Seal, D., Shaffer, S., Shimada, J., Umland, J., Werner, M., Oskin, M., Burbank, D., Alsdorf, D., 2007. The shuttle radar topography mission. Rev. Geophys. 45, RG2004. https://doi.org/10.1029/2005RG000183.
Fjorde, A., 2007. Vietnamese state industry and the political economy of commercial renaissance. Dragon's tooth or Curate's egg? Chandos Publishing, Witney, Oxford, UK.
Fukushima, M., Kanzaki, M., Hara, M., Ohkubo, T., Preechapanya, P., Choocharoen, C., 2008. Secondary forest succession after cessation of swidden cultivation in the montane forest area in Northern Thailand. For. Ecol. Manag. 255, 1994–2006.
Gingrich, S., Lauk, C., Krausmann, F., Erb, K.-H., Le Noë, J., 2021. Changes in energy and livestock systems largely explain the forest transition in Austria (1830-1910). Land Use Policy 109, 105624.

Gingrich, S., Lauk, C., Niedertscheider, M., Pichler, M., Schaffartzik, A., Schmid, M., Magerl, A., Le Noë, J., Bhan, M., Erb, K.-H., 2019. Hidden emissions of forest transitions: a socio-ecological reading of forest change. Curr. Opin. Environ. Sustain. 38, 14–21.
Gomiero, T., Pettenella, D., Trieu, G.P., Paoletti, M.G., 2000. Vietnamese uplands: environmental and socio-economic perspective of forest land allocation and deforestation process. Environ. Dev. Sustain. 2, 119–142.
Goscha, C., 2017. The Penguin history of modern Vietnam. Penguin Books, London.
Gray, T.N.E., Hughes, A.C., Laurance, W.F., Long, B., Lynam, A.J., O'Kelly, H., Ripple, W. J., Seng, T., Scotson, L., Wilkinson, N.M., 2018. The wildlife snaring crisis: an insidious and pervasive threat to biodiversity in Southeast Asia. Biodivers. Conserv. 27, 1031–1037.
GSO, 2022. General Statistics Office of Vietnam. 〈https://www.gso.gov.vn/en/homepage/〉 (accessed 15. February 2022).
Ha, V.T., 2015. Forest fragmentation in Vietnam: Effects on tree diversity, populations and genetics. Utrecht University, Netherlands.
Haas, J.C., Loft, L., Pham, T.T., 2019. How fair can incentive-based conservation get? The interdependence of distributional and contextual equity in Vietnam's Payments for Forest Environmental Services Program. Ecol. Econ. 160, 205–214.
Hardy, A., 2009. Eaglewood and the economic history of Champa and Central Vietnam. In: Hardy, A., Cucarzi, M., Zolese, P. (Eds.), Champa and the Archeology of Mỹ Sơn (Vietnam). Elsevier, Amsterdam, pp. 107–126.
Hay, A., 1983. Defoliants in Vietnam: the long-term effects. Nature 302, 208–209.
Hayton, B., 2010. Vietnam. Rising dragon. Yale University Press, New Haven, USA.
Heilmayr, R., Echeverría, C., Fuentes, R., Lambin, E.F., 2016. A plantation-dominated forest transition in Chile. Appl. Geogr. 75, 71–82.
Hickey, G.C., 1993. Shattered world: adaptation and survival among Vietnam's highland peoples during the Vietnam War. University of Pennsylvania Press, Philadelphia.
Hughes, A.C., 2017. Mapping priorities for conservation in Southeast Asia. Biol. Conserv. 209, 395–405.
Huynh, H.T.N., de Bruyn, L.L., Prior, J., Kristiansen, P., 2016. Community participation and harvesting of non-timber forest products in benefit-sharing pilot scheme in Bach Ma National Park. Trop. Conserv. Sci. 9 (2), 877–902.
ICEM, 2003. Vietnam national report on protected areas and development. Review of protected areas and development in the Lower Mekong River Region. International Centre for Environmental Management (ICEM), Indooroopilly, Queensland, Australia.
Imanuddin, R., Hidayat, A., Rachmat, H.H., Turjaman, M., Pratiwi, Nurfatriani, F., Indrajaya, Y., Susilowati, A., 2020. Reforestation and sustainable management of *Pinus merkusii* forest plantation in Indonesia: a review. Forests 11, 1235.
Kien, N.D., Harwood, C., 2016. Timber demand and supply in Northwest Vietnam: the roles of natural forests and planted trees. Small-scale. Forestry 16, 65–82.
Kiernan, B., 2019. Viet Nam: a history from the earliest times to the present. Oxford University Press, Oxford, UK.
Kompas, T., Che, T.N., Nguyen, H.T.M., Nguyen, H.Q., 2012. Productivity, net returns, and efficiency: land and market reform in Vietnamese rice production. Land Econ. 88 (3), 478–495.
Krahn, J., 2005. The dynamics of dietary change of transitional food systems in tropical forest areas of Southeast Asia: The contemporary and traditional food system of the Katu in the Sekong Province, Lao, PDR. Rheinische Friedrich-Wilhelms University, Bonn, Germany.
Lang, C.R., 2001. Deforestation in Vietnam, Laos, and Cambodia. In: Vajpeyi, D.K. (Ed.), Deforestation, environment, and sustainable development: a comparative analysis. Greenwood Publishing Group, Westport, Connecticut, USA, pp. 111–137.
Le, B.T., Nguyen, T.L.T., Adkins, S., 2012. Damage caused by *Merremia eberhardtii* and *Merremia boisiana* to biodiversity of Da Nang City, Vietnam. Pak. J. Weed Sci. Res. 18, 895–905.
Le, L.T.H., Dat, N.D., Minh, N.H., Nguyen, K.-A., 2019. Characteristics of PCDD/Fs in soil and sediment samples collected from A So former airbase in Central Vietnam. Sci. Total Environ. 661, 27–34.
Lestrelin, G., Castella, J.C., Fox, J., 2013. Forest transitions in Southeast Asia: synergies and shortcomings in land change science and political ecology. In: Brannstrom, C., Vadjunec, J. (Eds.), Land change science, political ecology, and sustainability: synergies and divergences. Routledge, London, pp. 48–65.
Liu, J., Liang, M., Li, L., Long, H., de Jong, W., 2017. Comparative study of the forest transition pathways of nine Asia-Pacific countries. For. Policy Econ. 76, 25–34.
Luu, L.N., Scussolini, P., Kew, S., Philip, S., Hariadi, M.H., Vautard, R., Mai, K.V., Vu, T. V., Truong, K.B., Otto, F., van der Schrier, G., van Aalst, M.K., van Oldenborgh, G.J., 2021. Attribution of typhoon-induced torrential precipitation in Central Vietnam, October 2020. Clim. Change 169, 24.
Magerl, A., Matej, S., Kaufmann, L., Le Noë, J., Erb, K., Gingrich, S., 2022. Forest carbon sink in the U.S. (1870-2012) driven by substitution of forest ecosystem service flows. Resour. Conserv. Recycl. 176, 105927.
Mai, N.T.H., 2016. Forest and forestland use rights: An institutional and economic analysis of forest devolution in Upland Central Vietnam. Margraf Publishers, Weikersheim, Germany.
Mather, A.S., 1992. The forest transition. Area 24 (4), 367–379.
Mather, A.S., 2007. Recent Asian forest transitions in relation to forest transition theory. Int. For. Rev. 9 (1), 491–502.
Mather, A.S., Fairbairn, J., 2000. From floods to reforestation: the forest transition in Switzerland. Environ. Hist. 6, 399–421.
Mather, A.S., Needle, C.L., Coull, J.R., 1998. From resource crisis to sustainability: the forest transition in Denmark. Int. J. Sustain. Dev. World Ecol. 5 (3), 182–193.
McConnell, W., Viña, A., Kull, C.A., Batko, C., 2015. Forest transition in Madagascar's highlands: initial evidence and implications. Land 4 (4), 1155–1181.
McElwee, P., 2004. You say illegal, I say legal. The relationship between 'illegal' logging and land tenure, poverty, and forest use rights in Vietnam. J. Sustain. For. 19 (1–3), 97–135.
McElwee, P., 2006. Displacement and relocation redux: stories from Southeast Asia. Conserv. Soc. 4 (3), 396–403.
McElwee, P., 2008. Forest environmental income in Vietnam: household socio-economic factors influencing forest use. Environ. Conserv. 35 (2), 147–159.
McElwee, P., 2009. Reforesting "bare hills" in Vietnam: social and environmental consequences of the 5 million hectare reforestation program. Ambio 38 (6), 325–333.
McElwee, P., 2016. Forests are gold: trees, people, and environmental rule in Vietnam. University of Washington Press, Seattle.
McElwee, P., 2021. Shifting policies for shifting justifications: a history of anti-swidden interventions in Vietnam. J. Southeast Asian Stud. 53 (1–2), 153–182.
McElwee, P., Nghi, T.H., 2021. Assessing the social benefits of tree planting by smallholders in Vietnam: Lessons for large-scale reforestation programs. Ecol. Restor. 39 (1–2), 52–63.
McElwee, P., Huber, B., Nguyen, V.H.T., 2019. Hybrid outcomes of payments for ecosystem services policies in Vietnam : between theory and practice. Dev. Change 51 (1), 253–280. https://doi.org/10.1111/dech.12548.
McElwee, P., Vu, H., Vo, G., Le, D., 2022. Patriotism, place, and provisioning: assessing cultural ecosystem services through longitudinal and historical studies in Vietnam. Ecol. Soc. 27 (1), 3.
McElwee, P., Nghiem, T., Le, H., Vu, H., Tran, N., 2014. Payments for environmental services and contested neoliberalisation in developing countries: a case study from Vietnam. J. Rural Stud. 36, 423–440.
McNamara, S., Duong, V.T., Erskine, P.D., Lamb, D., Yates, D., Brown, S., 2006. Rehabilitating degraded forest land in central Vietnam with mixed native species plantings. For. Ecol. Manag. 233 (2–3), 358–365.
Meijaard, E., Groves, C.P., 2006. The geography of mammals and rivers in mainland Southeast Asia. In: Lehman, S.M., Fleagle, J.G. (Eds.), Primate biogeography. Developments in primatology: progress and prospects. Springer, Boston, MA, pp. 305–329.
Meyfroidt, P., Lambin, E.F., 2008. Forest transitions in Vietnam and its environmental impacts. Glob. Change Biol. 14, 1319–1339.
Meyfroidt, P., Lambin, E.F., 2008b. The causes of reforestation in Vietnam. Land Use Policy 25, 182–197.
Meyfroidt, P., Lambin, E.F., 2009. Forest transition in Vietnam and displacement of deforestation abroad. Proc. Natl. Acad. Sci. USA (PNAS) 106 (38), 16139–16144.
Meyfroidt, P., Lambin, E.F., 2011. Global forest transition: prospects for an end to deforestation. Annu. Rev. Environ. Resour. 36, 343–371.
Meyfroidt, P., Rudel, T.K., Lambin, E.F., 2010. Forest transitions, trade, and the global displacement of land use. Proc. Natl. Acad. Sci. 107 (49), 20917–20922.
Moggridge, H.L., Higgitt, D.L., 2014. Interactions between riparian vegetation and fluvial processes within tropical Southeast Asia: Synthesis and future directions for research. Prog. Phys. Geogr. 38 (6), 716–733.
Morley, R.J., 2018. Assembly and division of the South and South-East Asian flora in relation to tectonics and climate change. J. Trop. Ecol. 34, 209–234.
My, T.T.A., Dat, N.D., Van Langenhove, K., Denison, M.S., Long, H.T., Elskens, M., 2021. Evaluation of the dioxin-like toxicity in soil samples from Thua Thien Hue province using the AhR-CALUX bioassay – an update of Agent Orange contamination in Vietnam. Ecotoxicol. Environ. Saf. 212, 111971.
Nambiar, E.K.S., Harwood, C.E., Nguyen, D.K., 2015. *Acacia* plantations in Vietnam: research and knowledge application to secure a sustainable future. South. For. 77 (1), 1–10.
Nambiar, S.E.K., Harwood, C.E., Mendham, D.S., 2018. Paths to sustainable wood supply to the pulp and paper industry in Indonesia after diseases have forced a change of species from acacia to eucalypts. Aust. For. 81 (3), 148–161.
Nehren, U., Thai, H.H.D., Trung, N.D., Raedig, C., Alfonso, S., 2017. Sand dunes and mangroves for disaster risk reduction and climate change adaptation in the coastal zone of Quang Nam Province, Vietnam. In: Nauditt, A., Ribbe, L. (Eds.), Land use and climate change interactions in Central Vietnam. Springer, Singapore, pp. 201–222.
Ngo, D.T., Webb, E.L., 2008. Incentives of the forest land allocation process: implications for forest management in Nam Dong district, Central Vietnam. In: Webb, E.L., Shivakoti, G.P. (Eds.), Decentralization, forests and rural communities: Policy outcomes in South and Southeast Asia. Sage Publications, New Delhi, pp. 269–291.
Ngo, D.T., Webb, E.L., 2016. Reconciling science and indigenous knowledge in selecting indicator species for forest monitoring. In: Tran, T.N., Ngo, D.T., Hulse, D., Sharma, S., Shivakoti, G. (Eds.), Redefining diversity and dynamics of natural resources management in Asia, Volume 3. Natural resource dynamics and social ecological systems in Central Vietnam: development, resource changes and conservation issues. Elsevier, Amsterdam, pp. 125–138.
Nguyen, H.H., Uria-Diez, J., Wiegand, K., 2016a. Spatial distribution and association patterns in a tropical evergreen broad-leaved forest of North-Central Vietnam. J. Veg. Sci. 27, 318–327.
Nguyen, H.T., de Bruyn, L.L., Koech, R., 2016b. Impact of hydropower dam development on agriculturally based livelihoods of resettled communities: a case study of Duong Hoa Commune in central Vietnam. Int. J. Water Resour. Dev. 32 (6), 978–996.
Nguyen, T.T., Nguyen, T.T., Hoang, V.-N., Wilson, C., Managi, S., 2019. Energy transition, poverty and inequality in Vietnam. Energy Policy 132, 536–548.
Nguyen, V.H.T., Kull, C., 2022. Land acquisition through Bricolage? Politics of smallholder acacia plantation expansion in upland Central Vietnam. J. Peasant Stud. https://doi.org/10.1080/03066150.2022.2029849.
Nguyen, V.H.T., McElwee, P., Hue, L.T.V., Tuyen, N.P., Huong, V.T.D., 2022. The challenges of collective PES: insights from three community-based models in Vietnam. Ecosyst. Serv. 56, 101438.

Nguyen-Tien, V., Elliott, R.J., Strobl, E.A., 2018. Hydropower generation, flood control and dam cascades: a national assessment for Vietnam. J. Hydrol. 560, 109–126.
Ngyuen, H.T., Pham, T.H., de Bruyn, L.L., 2017. Impact of hydroelectric dam development and resettlement on the natural and social capital of rural livelihoods in Bo Hon village in Central Vietnam. Sustainability 9, 1422.
Nicolic, N., Schultze-Kraft, R., Nicolic, M., Böcker, R., Holz, I., 2008. Vegetation transitions and land degradation on barren hills: a case study in Northeast Vietnam. Environ. Manag. 42 (1), 19–36.
Pendrill, F., Persson, U.M., Godar, J., Kastner, T., 2019. Deforestation displaced: trade in forest-risk commodities and the prospects for a global forest transition. Environ. Res. Lett. 14, 055003.
Perry-Castañeda Library Map Collection, 2021. 1:50'000 - Vietnam, Series L7014 (Topographic Maps). U.S. Army Map Service, 1964–1971. ⟨https://maps.lib.utexas.edu:8080/maps/topo/vietnam/⟩ (accessed 15 May 2021).
Perz, S.G., 2007. Grand theory and context-specificity in the study of forest dynamics: forest transition theory and other directions. Prof. Geogr. 59, 105–114.
Pfaff, A., Walker, R., 2010. Regional interdependence and forest "transitions": substitute deforestation limits the relevance of local reversals. Land Use Policy 27 (2), 119–129.
Pham, T.P.T., Tran, N.T., Kull, C.A., Shackleton, R.T., Cochard, R., Nguyen, T.H.M., Ngo, T.D., Nguyen, T.H.V., Tran, Q.C., Vu, T.T.T., 2023. Factors influencing farmers' forestland-use changes over 15 years (2005-2020) in Thua Thien Hue province, Vietnam. Int. For. Rev. 25 (1), 71–91.
Pingali, P.L., Xuan, V.-T., 1992. Vietnam: Decollectivization and rice productivity growth. Econ. Dev. Cult. Change 40 (4), 697–718.
Polesny, Z., Verner, V., Vlkova, M., Banout, J., Lojka, B., Valicek, P., Mazancova, J., 2014. Non-timber forest products utilization in Phong Dien Nature Reserve, Vietnam: Who collects, who consumes, who sells? Bois For. Trop. 322, 39–49.
Putz, F.E., Redford, K.H., 2010. The importance of defining 'forest': tropical forest degradation, deforestation, long-term phase shifts, and further transitions. Biotropica 42 (1), 10–20.
Raymond, C., 2008. "No responsibility and no rice": the rise and fall of agricultural collectivization in Vietnam. Agric. Hist. 82 (1), 43–61.
Robert, A., 2011. Dynamiques paysagères et guerre dans la Province de Thua Thien Hue (Viet Nam Central). Entre défoliation, déforestation et reconquêtes végétales. PhD Thesis, Université Paris-Sorbonne.
Robichaud, W.G., Sinclair, A.R.E., Odarkor-Lanquaye, N., Klinkenberg, B., 2009. Stable forest cover under increasing populations of swidden cultivators in Central Laos: the roles of intrinsic culture and extrinsic wildlife trade. Ecol. Soc. 14 (1), 33.
Rudel, T.K., Coomes, O.T., Moran, E., Achard, F., Angelsen, A., Xu, J., Lambin, E., 2005. Forest transitions: towards a global understanding of land use change. Glob. Environ. Change 15 (1), 23–31.
Rundel, P.W., 1999. Conservation priorities in Indochina – WWF desk study. Forest habitats and flora in Lao PDR, Cambodia and Vietnam. World Wildlife Fund, Indochina Programme Office, Hanoi, Vietnam.
Sakschewski, B., von Bloh, W., Boit, A., Poorter, L., Peña-Claros, M., Heinke, J., Joshi, J., Thonicke, K., 2016. Resilience of Amazon forests emerges from plant trait diversity. Nat. Clim. Change 6, 1032–1036.
Salemink, O., 2002. The ethnography of Vietnam's Central Highlanders. A historical contextualization 1850-1990. Routledge, Milton Park, UK.
Salemink, O., 2015. Revolutionary and Christian ecumenes and desire for modernity in the Vietnamese Highlands. Asia Pac. J. Anthropol. 16 (4), 388–409.
Sam, H.V., Nanthavong, K., Kessler, P.J.A., 2004. Trees of Laos and Vietnam: A field guide to 100 economically or ecologically important species. Blumea 49, 201–349.
Sandalj, M., Treydte, A.C., Ziegler, S., 2016. Is wild meat luxury? Quantifying wild meat demand and availability in Hue, Vietnam. Biol. Conserv. 194, 105–112.
Satake, A., Rudel, T.K., 2007. Modelling the forest transition: forest scarcity and ecosystem service hypothesis. Ecol. Appl. 17 (7), 2024–2036.
Sidle, R.C., Ziegler, A.D., Negishi, J.N., Nik, A.R., Siew, R., Turkelboom, F., 2006. Erosion processes in steep terrain – truths, myths, and uncertainties related to forest management in Southeast Asia. For. Ecol. Manag. 224, 199–225.
Sikor, T., Nguyen, Q.T., 2007. Why may forest devolution not benefit the rural poor? Forest entitlements in Vietnam's Central Highlands. World Dev. 35 (11), 2010–2025.
Sikor, T., To, P.X., 2011. Illegal logging in Vietnam: lam tac (forest hijackers) in practice and talk. Soc. Nat. Resour. 24, 688–701.
Sikor, T., Baggio, J.A., 2014. Can smallholders engage in tree plantations? An entitlements analysis from Vietnam. World Dev. 64 (Supplement 1), S101–S112. https://doi.org/10.1016/j.worlddev.2014.03.010.
Sloan, S., Meyfroidt, P., Rudel, T.K., Bongers, F., Chazdon, R., 2019. The forest transformation: planted tree cover and regional dynamics of tree gains and losses. Glob. Environ. Change 59, 101988.
Sowerwine, J.C., 2004. Territorialisation and the politics of highland landscapes in Vietnam: negotiating property relations in policy, meaning and practice. Conserv. Soc. 2 (1), 97–136.
Stas, S.M., Le, T.C., Tran, H.D., Hoang, T.T.H., van Kuijk, M., Le, A.V., Ngo, D.T., van Oostrum, A., Phillips, O.L., Rutishauser, E., Spracklen, B.D., Tran, T.T.A., Le, T.T., Spracklen, D.V., 2020. Logging intensity drives variability in carbon stocks in lowland forests in Vietnam. For. Ecol. Manag. 460, 117863.
Stellman, J.M., Stellman, S.D., Christian, R., Weber, R., Tomasallo, C., 2003. The extent and patterns of usage of Agent Orange and other herbicides in Vietnam. Nature 422, 681–687.
Sterling, E.J., Hurley, M.M., Minh, L.D., 2006. Vietnam. A natural history. Yale University Press, New Heaven, USA.
Stibig, H.-J., Achard, F., Carboni, S., Raši, R., Miettinen, J., 2014. Change in tropical forest cover of Southeast Asia from 1990 to 2010. Biogeosciences 11, 247–258.
Tham, L.T., Darr, D., Pretzsch, J., 2020. Contribution of small-scale acacia hybrid timber production and commercialization for livelihood development in Central Vietnam. Forests 11, 1335.
Tham, L.T., Darr, D., Pretzsch, J., 2021. Analysis of Acacia hybrid timber value chains: a case study of woodchip and furniture production in central Vietnam. For. Policy Econ. 125, 102401.
Thanh, L.N., de Smedt, F., 2014. Slope stability analysis using a physically based model: a case study from A Luoi district in Thua Thien Hue Province, Vietnam. Landslides 11, 897–907.
Thiha, Webb, E.L., Honda, K., 2007. Biophysical and policy drivers of landscape change in a central Vietnamese district. Environ. Conserv. 34, 164–172.
Thulstrup, A.W., 2015. Livelihood resilience and adaptive capacity: tracing changes in household access to capital in Central Vietnam. World Dev. 74, 352–362.
Thulstrup, A.W., Casse, T., Nielsen, T.T., 2013. The push for plantations: drivers, rationales and social vulnerability in Quang Nam Province, Vietnam. In: Bruun, O., Casse, T. (Eds.), On the frontiers of climate and environmental change. Vulnerabilities and adaptations in Central Vietnam. Springer, Heidelberg, pp. 71–89.
Tien, P.V., Trinh, P.T., Luong, L.H., Nhat, L.M., Duc, L.M., Hieu, T.T., Cuong, T.Q., Nhan, T.T., 2021. The October 13, 2020, deadly rapid landslide triggered by heavy rainfall in Phong Dien, Thua Thien Hue, Vietnam. Landslides 18, 2329–2333.
Tilker, A., Abrams, J.F., Mohamed, A., Nguyen, A., Wong, S.T., Sollmann, R., Niedballa, J., Bhagwat, T., Gray, T.N.E., Rawson, B.M., Guegan, F., Kissing, J., Wegmann, M., Wilting, A., 2019. Habitat degradation and indiscriminate hunting differentially impact faunal communities in the Southeast Asian tropical biodiversity hotspot. Commun. Biol. 2, 396.
To, P., Dressler, W., 2019. The politics of payments for forest environmental services in Vietnam. Land Use Policy 81, 582–593.
To, P., Dressler, W., 2019. Rethinking 'Success': The politics of payment for forest ecosystem services in Vietnam. Land Use Policy 81, 582–593.
To, P.X., Tran, H.N., 2014. Rubber expansion and forest protection in Vietnam. Tropenbos International, Vietnam, and Forest Trends, Hue, Vietnam.
To, P.X., Mahanty, S., Dressler, W.H., 2015. 'A new landlord' (địa chủ mới)? community, land conflict and State Forest Companies (SFCs) in Vietnam. For. Policy Econ. 58, 21–28.
To, P.X., Dressler, W.H., Mahanty, S., Pham, T.T., Zingerli, C., 2012. The prosepct for Payments for Ecosystem Services (PES) in Vietnam: A look at three payment schemes. Hum. Ecol. 40, 237–249. https://doi.org/10.1007/s10745-012-9480-9.
To, S., Mahanty, P.X., Dressler, W., 2014. Social networks of corruption in the Vietnamese and Lao cross-border timber trade. Anthropol. Forum 24 (2), 154–174.
Tong, T.M.T., Shaw, R., Takeuchi, Y., 2011. Climate disaster resilience of the education sector in Thua Thien Hue Province, Central Vietnam. Nat. Hazards 63 (2), 685–709.
Tran, D.W., Rambo, A.T., Nguyen, T.L. (Eds.), 2009. Farming with fire and water: The human ecology of a composite swiddening community in Vietnam's Northern mountains. Kyoto University Press, Kyoto.
Tran, L.D., 2014. Using *Acacia* as a nurse crop for re-establishing native-tree species plantation on degraded lands in Vietnam. PhD thesis, University of Tasmania.
Trinh, L.N., Watson, J.W., Hue, N.N., De, N.N., Minh, N.V., Chu, P., Sthapit, B.R, Eyzaguirre, P.B., 2003. Agrobiodiversity conservation and development in Vietnamese home gardens. Agric. Ecosyst. Environ. 97, 317–344.
Tuan, H.H., Minh, N.V., Mai, N.T.H., Hang, T.T.T., Iwanaga, S., 2022. Determinants and challenges of community sand forest management in Vietnam. Forests 13, 561.
Tuan, N.D.A., Dung, N.T., Sharma, S., 2017. Comanagement approach for conflict management: a case study of the Phong Dien Nature Reserve. In: Tran, T.N., Ngo, D. T., Hulse, D., Sharma, S., Shivakoti, G. (Eds.), Redefining diversity and dynamics of natural resources management in Asia, Volume 3. Natural resource dynamics and social ecological systems in Central Vietnam: development, resource changes and conservation issues. Elsevier, Amsterdam, pp. 179–188.
Van, D.N., Son, C.T., Hung, L., 2016. Ethnic minorities in Vietnam. The Gioi Publishers, Hanoi, Vietnam.
Van, Y.T., Cochard, R., 2017. Structure, diversity and plant utilities of a lowland hillside rainforest fragment in Central Vietnam. For. Ecosyst. 4, 9.
Van Holt, T., Binford, M.W., Portier, K.M., Vergara, R., 2016. A stand of trees does not a forest make: Tree plantations and forest transitions. Land Use Policy 56, 147–157.
Van Kuijk, M., 2008. Forest regeneration and restoration in Vietnam. Utrecht University, Netherlands.
Von Benda-Beckmann, F., von Benda-Beckmann, K., Wiber, M.G. (Eds.), 2006. Changing properties of property. Berghahn Books, New York.
Vu, B.T., Cochard, R., Ngo, D.T., 2023. The acacia plantation boom in Thua Thien Hue Province, Central Vietnam: A survey of tree farmers' shifting livelihoods, environmental perceptions, and occupational perspectives. Trees For. People 13, 100405.
Walker, R., 2008. Forest transition: Without complexity, without scale. Prof. Geogr. 60 (1), 136–140.
Walters, B.B., 2017. Explaining rural land use change and reforestation: A causal-historical approach. Land Use Policy 67, 608–624.
Walters, B.B., 2022. Explaining land use and forest change: more theory or better methodology? Landsc. Ecol. 2022 https://doi.org/10.1007/s10980-021-01397-2.
Wang, H.-F., Lencinas, M.V., Friedman, C.R., Wang, X.-K., Qiu, J.-X., 2011. Understorey plant diversity assessment of *Eucalyptus* plantations over three vegetation types in Yunnan, China. N. For. 42, 101–116.
WB Data, 2022. World Bank Data Catalog. Vietnam data. ⟨https://datacommons.org/place/country/VNM#⟩ (accessed 10 March 2022).
WCED, 1987. Our common future. World Commission on Environment and Development (WCED). Oxford University Press, New York.
Westing, A.H., 1971. Forestry and the war in South Vietnam. J. For. 69 (11), 777–783.

Westing, A.H., 1976. Ecological consequences of the Second Indochina War. Stockholm International Peace Research Institute (SIPRI), Stockholm, Sweden.
Wetterwald, O., Zingerli, C., Sorg, J.-P., 2004. Non-timber forest products in Nam Dong District, Central Vietnam: ecological and economic prospects. Schweizerische Zeitschrift für Forstwesen 155, 45–52.
Wikramanayake, E., Dinerstein, E., Loucks, C., Olson, D., Morrison, J., Lamoreux, J., McKnight, M., Hedao, P. (Eds.), 2002. Terrestrial ecoregions of the Indo-Pacific: a conservation assessment. Island Press, Washington.
Woodruff, D.S., 2010. Biogeography and conservation in Southeast Asia: how 2.7 million years of repeated environmental fluctuations affect today's patterns and the future of the remaining refugial-phase biodiversity. Biodivers. Conserv. 19, 919–941.
Wright, S.J., 2002. Plant diversity in tropical forests: a review of mechanisms of species coexistence. Oecologia 130, 1–14.
Zhunusova, E., Sen, L.T.H., Schröder, J.-M., Ziegler, S., Dieter, M., Günter, S., 2019. Smallholder decision-making on sawlog production: the case of Acacia plantation owners in Central Vietnam. Forests 10 (11), 969. https://doi.org/10.3390/f10110969.
Ziegler, A.D., Gimabelluca, T.W., Tran, L.T., Vana, T.T., Nullet, M.A., Fox, J., Vien, T.D., Pinthong, J., Maxwell, J.F., Evett, S., 2004. Hydrological consequences of landscape fragmentation in mountainous northern Vietnam: evidence of accelerated overland flow generation. J. Hydrol. 287, 124–146.
Zierler, D., 2011. The invention of ecocide. Agent Orange, Vietnam, and the scientists who changed the way we think about the environment. University of Georgia Press, Athens, USA.
Zingerli, C., 2005. Colliding understandings of biodiversity conservation in Vietnam: global claims, national interests, and local struggles. Soc. Nat. Resour. 18, 733–747.