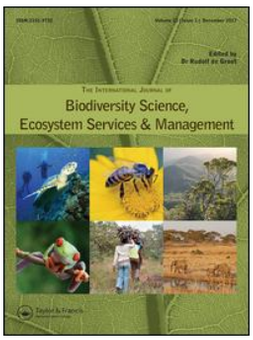

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Tropical forest-transition landscapes: a portfolio for studying people, tree crops and agro-ecological change in context

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ABSTRACT

Nudging the development trajectory of tropical landscapes towards sustainability requires a global commitment and policies that take diverse contexts and forest transitions into account. Out-scaling and upscaling landscape-level actions to achieve sustainable development goals globally need to be based on understanding of extrapolation domains and interconnectivity of products and services. We evaluated three portfolios of tropical landscape observatories and quantified extrapolation domains across ecological zones, stages of forest transition, human development index (HDI), population density and potential prominence of four dominant tropical tree crops (arabica coffee, cacao, rubber and oil palm). The ASB Partnership for Tropical Forest Margins portfolio was focussed on active humid forest margins and the Poverty and Environment Network on early stages of forest transition. The portfolio of sentinel landscapes of the Forests, Trees and Agroforestry (FTA) research programme provides a 5% sample of pantropical area, 8% of people, 9% of tree cover and 10–12% of potential tree crop presence, with quantified biases across zones, transition stages and HDI. In the 'water tower' configuration, relatively high population density coincides with biodiversity, coffee expansion and contested ecosystem services. The extrapolation domain of the FTA portfolio includes trade-off (tree loss) and synergy (restoration) phases of tropical forest transition.

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1. Introduction

Humans have a shared history with forests and trees (Williams 2003). This multifaceted relationship impacts several social, economic and environmental dimensions of our planetary and human systems. Humans converted and cleared forests to make space for agriculture, specific tree crops and development, often before they realized the value of what was being lost and the costs of restoring the ecosystem functions that society needs (Costanza et al. 2014).

Globally, there is no statistical relationship (Figure 1) between an aggregated Sustainable Development Goals (SDG) performance metric (Sachs et al. 2016) and forest cover, once the latter is adjusted for human population density on the basis of a global relationship (Köthke et al. 2013). On one hand, this indicates that forest conservation is not essential for achieving the SDG; on the other, that it isn't a hindrance either. Much depends, of course, on the operational definition of 'forest' in this context (see below). The tropics as a whole are about 13% lower in aggregate SDG performance than the non-tropics (as can be seen from the regression equations), although there is a substantial range in SDG score as well as forest cover in both climatic zones. How can the benefits of forests be used while the downsides are avoided? Rural poverty and food, water and energy security interact with income opportunities, climate change, biodiversity, governance and partnership agendas within the comprehensive SDG framework (Van Noordwijk et al. 2016b).

Interactions between people, ecosystems and tree crops at the local level are well covered in many case studies (Wiersum 1997; Meyfroidt et al. 2014; Wunder et al. 2014). However, the variability of these interactions across the tropics is much less addressed. Geographically targeted, evidence-based policies for managing sustainable development trade-offs are scarce (Minang et al. 2014b). To support the development of such policies, the characterization of variability and extrapolation domains is important for deepening our understanding of sustainability in the tele-connected context of rural–urban and local–global relationships. This can be seen in attempts to reduce (or avoid further) deforestation through interventions in the value chains – from producers to consumers – of tropical tree-crop commodities, which try to better couple the good intentions of companies and governments to the realities of producers, the environment and markets

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Supplemental data for this article can be accessed here.

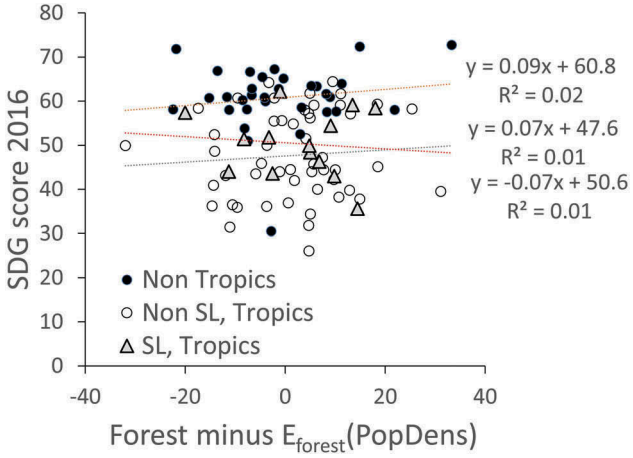

Figure 1. Relationship between the aggregated Sustainable Development Goal (SDG) score (Sachs et al. 2016) and the difference in forest cover between what is reported and expected on the basis of human population density (Köthke et al. 2013); SL indicates countries included in the sentinel landscape portfolio.

(Mithöfer et al. 2017): such efforts are best observed simultaneously at all levels – local, national and global – in order to understand the full range of influences and effects.

The path to sustainability is likely to include increased agricultural productivity with more efficient use of resources (a local issue), improved value chains for commodities produced across landscapes (regional), functioning mechanisms that reward provision of ecosystem services (global), inclusive land-use planning and policies (regional and national), strengthened capacities and partnerships with better enabling conditions (local), and comprehensive, local-to-global, 'green growth' strategies. To achieve sustainability, a thorough understanding is needed of what is most apt for particular agro-ecological zones and which interventions (or light-touch 'nudges'; Thaler and Sunstein 2008) can be effectively applied. Timely and effective responses to recognized and newly emerging sustainability issues are fundamental in achieving the SDG.

An example of an intervention rooted at the global level is 'certification', which can be seen as a response to a loss of trust between producers and consumers. With global commodity trade driving agricultural expansion into tropical forests, efforts to influence the drivers of land-use change require globally applicable standards, regulations and incentive mechanisms (Mithöfer et al. 2017). Major pathways of land-use change for tree crops have been described by Meyfroidt et al. (2014), where the land was either directly derived from forests or converted from pasture or crop land (that had historically been derived from forests). The production of commodities, such as tropical timber, cacao, coffee, palm oil and rubber, is often associated with negative impacts on biodiversity, ecosystem services and social well-being, especially when production stems from a direct conversion pathway (Meyfroidt et al. 2014). There have been abundant efforts across the tropics to improve local land governance, including using a 'landscape approach', that promotes change towards sustainable production and protection systems (Freeman et al. 2015; Minang et al. 2015). Whilst these efforts have been generating invaluable lessons that encourage replication in other areas with similar characterizations of human, ecosystem and tree-cropping patterns – which constitute extrapolation domains – systematic management of knowledge is not yet available.

The multiplicity of pathways towards solving the challenges facing sustainable development needs to be better understood if we are to secure a joint future for humanity and other life on the planet (Steffen et al. 2015). However, much current data and research are context-, system- and time-specific, highlighting the need for more global evidence on feedback and trends to inform the decisions of policymakers and others about land use and related issues. Such evidence also requires fine-tuning to the dynamics of any given system, where decision-makers operate often without the time or budget for detailed studies and data collection. For example, solutions that are presented to decision-makers as opposites from which to choose (such as 'land sparing versus land sharing': the segregation or integration of multiple land uses within a landscape) may share more commonalities once examined in a particular landscape (Lusiana et al. 2012), suggesting that the empirical basis for generic theory is weak. Within this context, we argue that trustworthy extrapolation domains for data need to be established, maintained and communicated. 'Case studies' are not able to provide wide-ranging conclusions, regardless of data analysis and meta-study compilation methods if the bias in their site selection cannot be quantified.

> Case researchers do not aspire to select cases that are directly representative of diverse populations and they usually do not and should not make claims that their findings are applicable to such populations except in contingent ways. (George and Bennett 2005)

Decision-makers aiming for immediate results may hastily try to replicate success stories from case studies without sufficient contextualization within local, regional and global domains. Extrapolation errors from 'case study' data can affect the three dimensions of usable knowledge (Clark, van Kerkhoff, et al. 2016; Clark, Tomich, et al. 2016): (1) salience of the underlying research (not being relevant to decisions in the actual context); (2) credibility (underlying data or system descriptions might appear to be unrelated to

the reality of the place) and (3) legitimacy (research that does not include local perspectives) (Clark, Tomich, et al. 2016; Leimona et al. 2015; Kowarsch et al. 2016; Posner et al. 2016).

The prevailing extrapolation mode is to describe individual case studies and identify general patterns through 'meta-studies' (Van Vliet et al. 2016). This method has several weaknesses. The geographic distribution of case studies quantifying ecosystem services in Africa, for example, probably tells more about the areas to which researchers have easy access rather than those where underlying issues are the most prominent and important (Kuyah et al. 2016). As methods tend to vary between studies as well as context, it is hard to judge whether differences in results reflect the bias and sampling and measurement errors of the researchers' methods or real variations in conditions and trends. In addition, further bias can be introduced through selective publication of 'significant' results (Forstmeier 2015; Kicinski et al. 2015).

A number of efforts have been made to harmonize methods in global comparative studies on the patterns and drivers of deforestation, and accompanying loss of biodiversity and ecosystem services, in relation to economic development and poverty alleviation. By far, the most extensive networked tropical studies have been carried out by the ASB Partnership for Tropical Forest Margins (ASB), Poverty Environment Network (PEN) and, more recently, the CGIAR Research Program on Forests, Trees and Agroforestry (FTA).

Since 1993, ASB has engaged with landscapes in Peru, Brazil, Cameroon, Indonesia and Thailand (Murdiyarso et al. 2002; Tomich et al. 2005, 2007; Clark, Tomich, et al. 2016; Gillison et al. 2013; Minang et al. 2014a). PEN had study sites in 24 countries (Angelsen et al. 2014; Jagger et al. 2014; Wunder et al. 2014). Other research efforts, such as the Landscape Mosaics project (Dewi et al. 2013) and the Diversitas Agrobiodiversity network (Jackson et al. 2012), partially overlapped with ASB and PEN in their choice of study sites. The breadth and value of the data gathered from such extended networks can be seen, for example, in statements from PEN, such as '28% of rural income is derived from forest' (Angelsen et al. 2014), which has associated confidence intervals based on variation within the data set (3 continents, 24 countries, 58 sites, 333 villages and 7978 households). This provides a first '*post hoc*' perspective on extrapolation possibilities within the domain of the data set.

However, the bias in extrapolating data beyond a research domain, area or geographic location to the level of the whole tropics is often difficult to assess because it depends on the 'external validity' of the portfolio of sites. As a consequence, the adoption of research findings and quantitative estimates in policies risks overstating or understating the evidence. But such problems can be avoided if the selection of research sites is part of a stratified sampling design within the desired extrapolation domain, using insights into (or hypotheses of) major dimensions of variability. The results from such a design can be used jointly with recommendations based more explicitly on theory to inform policies and interventions.

Against this background, we will here discuss the FTA research programme initiated in 2011. At the beginning of the research programme, it was deemed important to develop a portfolio of so-called sentinel landscapes that could be used to monitor patterns of change across the tropics and subtropics. The ambition was to integrate research across multiple scales – trees, farms, landscapes and governance domains – and provide international public goods, including 'change of theory' if existing 'theories of change' were found inadequate. The 'sentinel' terminology was derived from public health, where research to recognize generic patterns can lead to more effective prevention rather than only curing individuals (Shepherd et al. 2015). We will here focus on two questions:

(1) Can the 'forest transition' theory, in combination with climate and watershed data, be used for a rigorous typology of interactions between humans, ecosystems and tree cropping in which any set of sampling sites can be compared to the global frequency of similar conditions?
(2) What is the external validity of ASB, PEN and the FTA sentinel landscape portfolio for the pantropical extrapolation domain, according to such a typology? What are the consequences of the representativeness for the conclusions drawn from the three global comparative studies?

In the discussion, we will further focus on how useful the derived typology is for empirical data collection aimed at advising extrapolation domains from lessons learned locally and at drawing global, scientifically rigorous conclusions that can inform development and increase impact in FTA or similar research-in-development programmes. Before explaining the methods used in answering these questions, we provide further background on the forest-transition theory and biases in Theories of Induced Change (ToIC).

2. Forest-transition theory

Forest-transition theory (Figure 2) documents, describes and tries to explain non-linear changes in tree cover (loss of natural forests followed at some point by an increase in planted and managed trees) across the development trajectory of a country (Mather and Needle 1998). A spatial pattern of trees close to and far away from settlements was interpreted by Von Thünen (1842) on the basis of

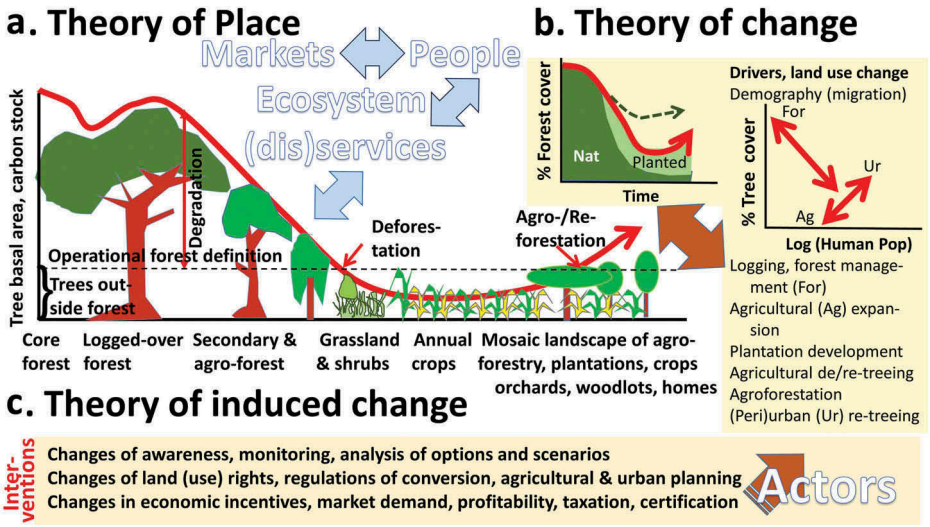

Figure 2. Forest-transition theory as ToP (A), ToC (B) or ToIC (C), connecting spatial patterns, non-linear temporal changes in natural and planted forests and intervention points for modifying trajectories.

distance-dependent costs of maintenance and protection versus value of the resource as a Theory of Place (ToP). Common versions use time, population density or distance to cities as the *X*-axis (Van Noordwijk et al. 1997; Angelsen and Rudel 2013; Köthke et al. 2013; Matthews et al. 2014; Minang et al. 2014a; Jouvet and Wolfersberger 2015). The theory provides a unified scheme for the multiplicity of changes (including in direction and rate of change) of tree cover that can take place at local and national scales. It offers a solid basis for a stratified sampling scheme at the forestry–agriculture interface because it logically connects direct, observable patterns (changes in forest and tree cover) with the drivers of change, the consequences and the response options (Van Noordwijk et al. 2011).

As detailed elsewhere (Van Noordwijk and Minang 2009), however, discrepancies exist between commonly used national and international definitions of 'forest', leading to confused time-series data sources (Grainger 2008). Common institutional concepts of forests are a complement of and incompatible with agriculture (Van Noordwijk et al. 2008). Meanwhile, tree-based concepts recognize the dichotomy as a continuum (Zomer et al. 2016). 'Agroforestry' and 'Community-Managed Forests' emerged institutionally as 'boundary object' (Van Noordwijk 2014). The lack of distinction between natural and planted forests (Sasaki and Putz 2009) has been a major obstacle to creating policies for sustainable forest, plantation and agroforest management. The confusion is substantial when the term forest is combined with the concept of 'restoration' (Chazdon et al. 2016), as either a return to the left or progression to the right side of the forest transition curve. In the midst of these terminological confusions, the term 'tree-cover transitions' has emerged as empirically observable relative to forest transitions (Ordonez et al. 2014; Van Noordwijk and Villamor 2014). However, given the common use of 'forest' transition, we will use this term throughout to refer to 'tree-cover' transitions.

Ultimately, most human land use falls outside of the various categories of 'forest' even though it begins with semi-permanent or permanent cultivation within a forest that leads to loss of tree cover and, finally, reclassification of land as 'non-forest'. This biophysical change interacts with institutional claims of forests as a domain controlled by the state. The 1217 Charter of the Forest in England, detailing royal concessions made in the Magna Carta, included 'disafforestation' as a return of royally claimed land to local control. At the earliest stage of the forest-transition curve, when population density is low, much of the forest is 'old growth' (but not necessarily 'pristine' because humanity has had a long and complex history of engagement with forests) and human occupancy is largely determined by biophysical suitability. At low population densities, shifting cultivation (also known as swidden or fallow rotation) has often been an effective forest-management practice (Van Noordwijk et al. 2015). Extraction of forest products, followed by conversion to other land uses, leads to further modification, degradation and deforestation, providing financial resources for more economic development. It is worth noting that large-scale logging of forests has, despite claims of sustainability and participatory forest management, rarely led to sustainable use beyond a first or second logging cycle, despite opportunities to do so (Putz et al. 2012; Bulkan and Palmer 2016). Usually before the last forest patch has disappeared, however, a transition point is reached where tree cover starts to increase (Köthke et al. 2014), either by a 'push' (people move to cities and abandon land which then regrows) or 'pull' (scarcity increases the value of the

products and services of the planted or remnant natural forests).

Historically, a wide range of nations have experienced this transition, following changes in population densities and percentages of tree cover that are often associated with the development of new, non-forest, non-agricultural sources of national income. The non-linear temporal pattern of forest and tree-cover change consists of trade-off and synergy stages between environmental and economic functionality (Tomich et al. 2005; Van Noordwijk et al. 2014). For example, replacement of natural forests by tree crops that feed global value chains may have a small net effect on tree cover but a large one on ecosystem services (Ordonez et al. 2014).

A number of hypotheses on patterns in tropical tree-cover transitions, as well as the consequences for local livelihoods of such changes in quantity and quality of tree cover, ecosystem services and biodiversity, were framed at the start of the FTA (Table 1), with the sentinel landscapes portfolio (see below) as the tool. However, the extrapolation domain of this portfolio has not yet been formally assessed.

3. Addressing biases in theory of induced changes

In the past, development agencies used 'log frames' as a planning tool, trying to manage activities towards achieving planned positive outcomes and impacts beyond a project's lifespan. The terminology of ToC has become common (Weiss 1997; Connell and Kubisch 1998), with emphasis on understanding how and why an initiative is expected to work. This may be more specifically described as a ToIC (Van Noordwijk et al. 2016a). Impacts are quantified as the difference in system performance between the trajectory of a system without intervention (but subject to external variability and change), described with a ToC, and a trajectory with planned interventions on top of external variability and change, described by a ToIC. The impact literature commonly recognizes the non-intervention counterfactuals as the weakest part of the logical chain, especially for natural resource management and policy research (Renkow and Byerlee 2010). As there is likely to be multiple feedback rather than a simple causal pathway, the use of information on context (ToP) can clarify our understanding of how systems work in a ToC (Minang et al. 2015). ToP starts from 'it matters where you are' or 'context matters' but continues to analyse how to describe and understand the 'where' beyond geographic coordinates or 'what is current context' and 'where else is it like this'. ToP potentially deals with a near-infinite dimensional space. A more pragmatic ToP seeks key dimensions of variation in social–ecological systems' structures, functions and dynamics and suggests that similarity along those dimensions can, to some extent, have predictive power in responses to new technologies, institutions or

Table 1. Hypotheses of interest in the FTA to be tested in the sentinel landscape portfolio.

Six hypotheses on patterns in tree-cover transition
Tree cover in landscapes changes in quality, quantity and pattern in non-linear fashion; depending on the operational definition of 'forest', tree-cover transitions at certain scales show a 'forest transition' graph of decline followed by recovery (*basic forest-transition hypothesis*)
Temporal tree-cover transitions can be understood as the result of time-variant processes, with increases in human population density (or rather the logarithm of it) linked to decreases in natural forest cover and increases in human development index (HDI) (or other economic indicators) linked to increases in tree cover (*population density and welfare hypothesis*)
The spatial pattern in quality and quantity of tree cover, from urban areas with (surrounding) trees to areas with few trees and open-field agriculture towards remaining natural forests, show more than coincidental resemblance with the temporal dynamics of hypothesis 2 because both patterns reflect benefits derived from tree cover relative to other land-cover types (*spatial forest-transition hypothesis*)
Institutional change from a 'forest' to an 'agrarian' regime of tenure and control is essential for the transition from decline towards increase of tree cover (*agroforestation or tenurial-reform hypothesis*)
What happens in one part of the tree-cover transition is linked at the level of drivers and/or actors to other parts of the landscape because (1) profitability of tree planting depends on access to tree and forest products elsewhere; (2) migration flows modify human population density in sink and source areas and (3) landscape-wide rules instigated to address specific issues in parts of the curve (e.g. 'illegal logging' control) affect actors elsewhere (*landscape-linkage hypothesis; the 'sparing' hypothesis that agricultural intensification saves forests is a special form of it*)
Drivers of tree-cover transition are space and time dependent and knowledge of past drivers in any particular landscape cannot be directly extrapolated to the future; yet, there may be predictability in the succession of drivers (*driver-change hypothesis*)
Six hypotheses on the consequences of tree-cover transitions
Land-use types that are part of the tree-cover transition differ in effectiveness of 'provisioning' and 'environmental' goods and services, labour absorption and profitability (*trade-off hypothesis*, ASB Matrix; Tomich et al. 2005)
Tree cover of all types and at all stages is positively associated with buffer functions in ecological, social and economic senses, with the spatial pattern and degree of integration linked to human resilience and adaptive capacity in the face of climate and market variability (*integration, buffer and resiliency hypothesis*)
Appreciation of tree cover and its associated ecosystem services varies with gender, wealth, cultural backgrounds, ecological knowledge and exposure to extreme events, leading to a diversity of opinions, preferences for the status quo and possible changes in tree cover (*'diversity of stakes' hypothesis; includes gender specificity*)
Feedback mechanisms from beneficiaries of (certain types of) tree cover to the drivers or agents can take multiple forms (rules, incentives, suasion, investment in value chains and technology) and need to be evaluated in the interaction between instruments rather than as specifically targeted approaches (*'no silver bullet' hypothesis*)
Dynamics of tree-cover changes can be influenced by multistakeholder negotiation-support processes that recognize multiple knowledge, perceptions, stakes, powers and influences (*negotiation-support hypothesis; includes gender specificity*)
Public discourse on aspects of tree-cover transition and the relevance of interventions follows a policy-issue cycle, with different opportunities for knowledge-based analyses to support and influence the emergence of transparent, effective, efficient and fair solutions, linking political will to actionable knowledge (*impact-pathway hypothesis*)

macroeconomic drivers. It can also help in seeking counterfactuals for quantitative impact studies, which is essential for finding evidence relevant to policymakers.

The forest transition theory (Figure 2) as used by the FTA is a good candidate for use as both a ToP and a ToC. It provides a logically appealing account of the diversity of situations (ToP) as well as plausible directions of historical change (ToC) and addresses the livelihoods, landscapes and governance dimensions of land-cover change. Its use as ToIC, however, remains unclear. FTA has confronted this theory of facts at scale by grounding extrapolation in the sentinel landscapes.

Bias in scientific evidence can stem from any one of several steps in a research cycle. Pantropical or global research for sustainable development that examines (1) scientific theories of endogenous change and (2) development-oriented ToIC typically involves a staged selection (moving downwards in Figure 3) of places for observation (within an explicit ToP), application of methods and collection of data sets.

Moving upwards in Figure 3 to derive conclusions to be used as 'evidence' requires a staged reintroduction of stratum weights and bias corrections. The first steps make use of the variability within the data set, with many statistical techniques at our disposal to assist in interpolation. The confidence intervals for conclusions derived from any stratified sampling depend on variability of data within a stratum plus uncertainty in estimates of stratum weights. Beyond the locations in which observations were made, however, there is an issue of 'external validity' and 'extrapolation domains'. These validity steps can be approached in two ways: by the (1) process (e.g. strict randomization with replication) used to derive the next level in the hierarchy or by (2) analysis of representativeness within global data sets. With existing global data sets on human population, forest and tree cover, elevation, climate, watersheds and soils plus national statistics on trade, economies and human development index (HDI), for example, the bias involved in a specific combination of case studies can be estimated and corrected for.

Here, we restrict our analysis to the first step – post-selection checks of representativeness – for the three portfolios of tropical landscapes, testing that means, variance and parameter distributions in the global data are conserved in the subsets, similar to what would be expected for a random sub-sampling of the pantropical domain.

4. Methods

4.1. The sentinel landscapes portfolio

The selection of landscapes for the portfolio (Table 2) was a complex process in which existing research networks, access to historical data, geographic balance, institutional agendas of the main FTA partners and insights of the researchers all played a role. Although the process of selection could not rely on any explicit and formal method of assessing pantropical representativeness, the researchers reflected on an analysis of representativeness in several dimensions of the PEN and ASB data sets. FTA partners were invited to propose landscapes and all were evaluated for their internal validity as well as potential contribution to the global set. Two landscapes that had not been selected in a first round were added subsequently (Nile–Congo and the Miombo) to ensure a better mix of African landscapes. Table 2 lists the names and locations of the 10 landscapes in the portfolio.

4.2. Spatial analysis

4.2.1. Sources of data

The extent of the study is the terrestrial areas of the tropics, between 23°26′13.8″N and 23°26′13.8″S. The

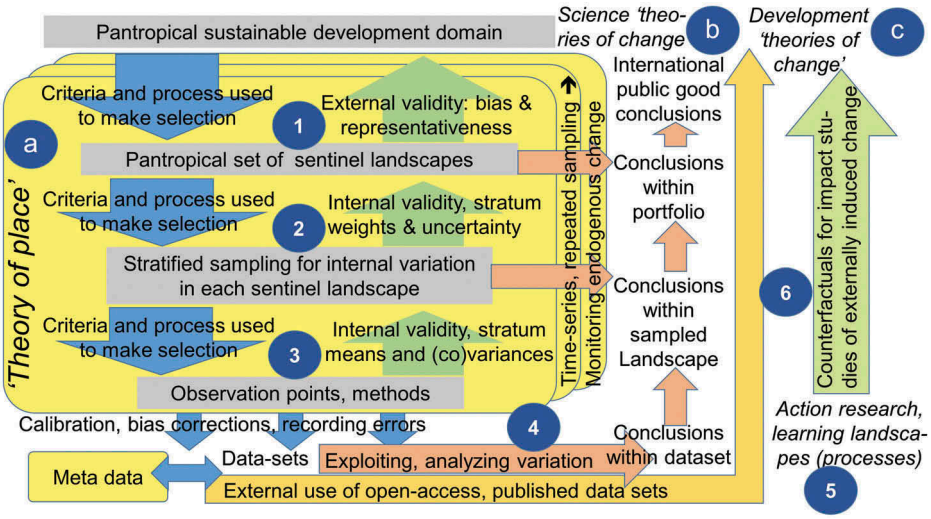

Figure 3. Schematic representation of a research cycle that collects data to support scientific understanding of geographic variations (ToP), spontaneous changes in socio-ecological systems (theory of endogenous change) or response to interventions.

Table 2. Ecological zones of the tropics and the sentinel landscapes.

Agro-ecological zone (indicative mean annual rainfall range)	Ecosystem services' issues related to dominant land uses; degradation (loss of services owing to loss of functions)	Recovery, restoration, agroforestation	Target geography (sentinel landscapes: most have multiple zones)	Human population density: (min)–mean–(max) km^{-2}; forest %
Dryland 18% of tropical area; 3% of tropical population (8.5 km^{-2}); 0% of sentinel landscapes' area	Few high-value tree crops; overharvesting of trees for fuelwood; annual fires; overgrazing by livestock; wind erosion; irrigated agricultural islands of functioning sub-humidity with risks of salinization	Specific attention to migratory circuits and routes for wildlife and pastoralists	Not a target for FTA research: dryland agroforestry and sparse tree cover are researched elsewhere	No sentinel landscapes
Semi-arid 17% of tropics; 17% of people (54.87 km^{-2}); 21% of sentinel landscapes' area **Dry sub-humid** 10% of area; 11% of people (62.3 km^{-2}); 20% of sentinel landscapes' area	Location-specific opportunities for tree-based participation in global markets; overharvesting of commercial timber and within urban reach for charcoal; land clearing for crop production; annual fires; local climate effects of tree-cover change	Controlled use of remaining forests, legalization of fuelwood trade as basis for investment; recognition of meso-climatic effects of tree cover in 'parklands'	**West African Savannah** (Ghana, Togo, Burkina Faso, Mali) **East African Miombo** (Zambia, Malawi, Mozambique)	(1)–35–(1758) 0% forest (at >50% cover) (0)–23–(4727) 14% forest
Water towers 11% of area; 15% of people (74.25 km^{-2}); 15% of sentinel landscapes' area	Expansion of highland crops and vegetables; coffee, tea, cacao; overharvesting of commercial timber; highly vulnerable biodiversity (endemic); changes in local climate; modified water flows; erosion/sedimentation; loss of soil C and nutrients	Recognition of specific hydrological functions, including attention to riparian zones as key to buffering, and ecological connectivity; incentives need to match downstream (including urban) interests	**Nile–Congo Water Towers** (Uganda, Kenya) **South Asian Water Tower** (Western Ghats in India)	(0)–152–(17,025) 14% (118)–213–(1360) 43%
Humid forest 19% of tropics; 22% of people (59.44 km^{-2}); 19% of SL area	Convertible to coffee, tea, cacao, rubber, bananas, oil palm, pasture. Overharvesting of commercial timber; connectivity loss ecological corridors; changes in local climate; soil compaction; erosion/sedimentation; loss of soil C and nutrients	Shift from monocultural tree crops to diversified agroforestry; restoration of degraded pastures; spatially explicit forest restoration for biological corridors; changes in land tenure may be needed	**Mekong** (China, Lao PDR, Viet Nam, Thailand) **Central America** (Nicaragua, Honduras)	(7)–69–(301) 67% (0)–74–(6929) 63%
(Per-) humid lowland forest 25% of tropics; 32% of people (69.07 km^{-2}); 25% of sentinel landscapes' area	Convertible to oil palm, rubber, pulp and paper plantations. Logging along rivers and major roads; overharvesting of commercial timber; high biodiversity loss; erosion/sedimentation; loss of soil C and nutrients; peatland issues	Opportunities for domestication and increased use of local resources may require changes of rules for market access; changes in land tenure may precede ecological recovery	**West Amazon** (Peru, Paraguay, Brazil) **Central African humid tropics** (Cameroon, Republic of the Congo, Democratic Republic of the Congo) **Insular Southeast Asia** (Sumatra and Borneo in Indonesia)	(0)–3–(229) 92% (1)–59–(5622) 82% (0)–43–(8705) 58%, 73%

tropical areas of Australia and the Arabian Peninsula were excluded from the analysis. Data sources are detailed in the supplementary information.

Georeferenced data of study sites were derived from

- ASB sites: http://www.asb.cgiar.org/map
- PEN villages: Poverty and Environment Networks: A comprehensive global analysis of tropical forests and poverty (CIFOR PEN database, see Wunder et al. 2014)
- FTA sentinel landscapes: ICRAF Sentinel Landscapes database (http://landscapeportal.org/)

Global statistical data at country level were derived from

- HDI at country level: UNDP, 2014. Muthayya et al. (2013) showed that the 'hidden hunger index' is tightly correlated with HDI; we did not include this index in addition to the HDI.

4.2.2. Preprocessing of raw data

All global spatial data were clipped to the tropics as defined above and reprojected into Mollweide projection to accommodate the semi-global nature of the data. We kept uniform cell size of all the raster data according to the highest resolution data layers, that is 300 × 300, through resampling procedures. Similarly, all the presence point data and the sites were transformed into the same projection system.

As a proxy of human well-being, we used the HDI 2014 at country level. The statistical data were converted into a raster map according to the political boundaries of countries. For PEN sites, 5 km surrounding the centre of villages was considered as the extent. For ASB sites, 35 km was used.

4.2.3. Analyses

We conducted three main analyses, which were mostly spatially explicit. Beyond that, extraction

from spatial data into tabular formats was conducted in a GIS environment and further analyses were conducted using statistical software.

4.2.3.1. Agro-ecological zonation. The first main factor we adopted in characterizing the agro-ecological zones was the aridity–humidity index, that is the ratio of precipitation and potential evapotranspiration at an annual time scale. The higher the index, the more humid the area [Zomer et al. (2016) renamed the index from 'aridity' to 'aridity–humidity']. We modified the CGIAR-CSI (2012) classification by adding a water-tower category (Table 3).

Considering the importance of spatial interconnectivity in transporting water from more elevated areas to the surroundings, the concept of a 'natural water tower' is crucial in landscape characterization. In identifying water towers, we needed to take into account variations in elevation within basins (the tower) in combination with humidity (the water). An area is considered a water tower if (1) it is either humid or (per-) humid, that is the aridity–humidity index is >0.65 and as such expected to generate river flow; and (2) the aridity–humidity index relative to elevation was >2.77, with relative elevation defined from the percentiles of elevation within each basin.

4.2.3.2. An operational forest-transition typology derived from global data sets. Expanding on existing analyses of forest-transition theory at national scale (Köthke et al. 2013, 2014), we changed the unit of analysis from country to sub-basin scale, which aligned with ecosystem services and local land governance within nested subnational, national and global systems. We conducted a cluster analysis of sub-basins based on two main factors: (1) land-use/cover composition and (2) forest configuration.

Land-use/cover composition was derived from global land-cover data of 2006, with simplified legends for closed forest, medium forest, open forest, shrub land, grassland, cropland, mixed cropland and trees, sparse vegetation, urban and bare land. The fraction of each type in the terrestrial part only was calculated for each sub-basin to quantify land use/cover composition with multiple variables. We adopted the forest-configuration approach coined by Chomitz et al. (2007) as an important proxy for anthropogenic activities with regards to forests. From the global land-cover map, we produced a forest map by reclassifying land-use/cover types into 'forest' and 'non-forest' classes. We then delineated pixels of forest as polygons of forest patches and calculated the areas and proximities. Based on this, we defined pixels into three types: (1) 'forest core', if the pixel belonged to a forest patch with an area of 20,000 ha or larger; (2) 'forest edge', if a forested pixel was located less than 10 km from the closest forest core area; and (3) 'forest mosaics', for the remaining forested pixels. For each sub-basin, we then calculated the proportions of the three types of forest configurations.

For each ecological zone, a *K*-means cluster analysis was conducted for sub-basins based on the information on land-use/cover composition and forest configuration. The results were then combined to group tropical sub-basins into six typologies to be used as a proxy of the stages of forest transitions.

Table 3. Ecological zones.

Rainfall/Potential evapotranspiration	Agro-ecological zone
<0.2	(Hyper-) arid
0.2–0.5	Semi-arid
0.5–0.65	Dry to sub-humid
0.65–1.0	Humid or water tower if AE > 2.77
>1.0	(Per-) humid or water tower if AE > 2.77

AE: product of aridity–humidity index and relative elevation within basin.
Source: Modified from the CGIAR-CSI (2012) classification of the aridity–humidity index (rainfall/potential evapotranspiration).

4.2.3.3. Similarity modelling of four commodities. Four pantropical commodities prominent in global trade (coffee, cacao, rubber and oil palm) are important elements in tree-cover transitions, both as drivers of change (deforestation) and as influences on re-emerging tree cover. We used a maximum entropy model to derive the areas most similar to where the crops were known to exist; this method is suitable for presence data modelling at the regional level (Phillips et al. 2006). MaxEnt software, the engine of the model, was run on a Q-GIS platform and integrated into Land-use Planning for Multiple Environmental Services software (Dewi et al. 2015) to reduce human error and increase efficiencies in data preprocessing and the iterative processes for producing the final model.

Our independent layers were (1) elevation, (2) agro-ecological zones and (3) climate. From the literature, we selected four variables within the WorldClim data set specific to the commodity species to be modelled based on the importance within the climatic envelopes of the four commodities, according to the literature (Ovalle Rivera et al. 2015; Paterson et al. 2015; Ray et al. 2015; Schroth et al., 2016) and personal communication with experts (Table Supp2).

The presence data points were collected from many sources, described in the supplementary information below. The most limited presence data we had was for oil palm, which mostly came from Southeast Asia. The modelling exercises were conducted separately for each region and each commodity species.

4.3. Tests of representativeness of ASB, PEN and sentinel landscape sites

Representativeness of the three data sets (ASB, PEN and the sentinel landscapes) was judged from correspondence in the fractions of areas and human

populations between pantropical data and the three sampling frames. Beyond these fractions, the relationships between ecological zones and/or forest-transition stages and properties, such as HDI, distance to protected areas and tree-crop distribution, were evaluated to estimate the bias in each.

5. Results

5.1. Forest-transition stages

The six forest-transition stages grouped sub-basins across the tropics according to forest fraction, forest configuration (core, edge and mosaic forest) and the logarithm of population density (4A). In stage 1, core forest covered around 80% of the total area of the sub-basins inhabited by human populations with densities below 1 km^{-2}. As population density increased, the fraction of core forest tended to decrease while the fraction of non-forest increased. Each stage was well separated from others with regards to human population density. Based on the fractions of core forest and non-forest areas, stage 1 was clearly separated from stages 2 and 3, and further with stages 4–6, while the last two groups were separated from each other but not within each other. Fractions of edge forest and mosaic forest did not discriminate the stages and did not seem to correlate with human populations at sub-basin level, indicating the two variables are merely intermediate states between core forest and non-forest.

The further stages were associated with increments in the logarithm of human population density and core forest and non-forest configurations but also with more specific characteristics of land cover (Figure 4(b)). The fraction of medium broadleaved forest and the fraction of tree cover decreased almost monotonically from stages 1 through 5 and increased again from stages 5 to 6, while the fraction of mixed tree cover followed exactly the opposite trend, suggesting an inflection point at stage 5. The fraction of cropland consistently increased and the fraction of shrub land was low at both ends and plateaued at stages 2–5. The initial increase in shrub land can imply logging as a driver of deforestation and/or non-permanent land uses that left degraded forest behind. The reduction of shrub land in the later stages implies that (planted) tree cover catches up with deforestation while further expansion of agriculture shifts to such land rather than remaining forest. Similar to the characterization of stages of forest configuration above, stage 1 was clearly separated from others with regards to fraction of tree and forest covers, while stages 2 and 3 were only barely separated based on the fraction of tree cover only. Stages 4–6 were quite well separated with regards to most variables.

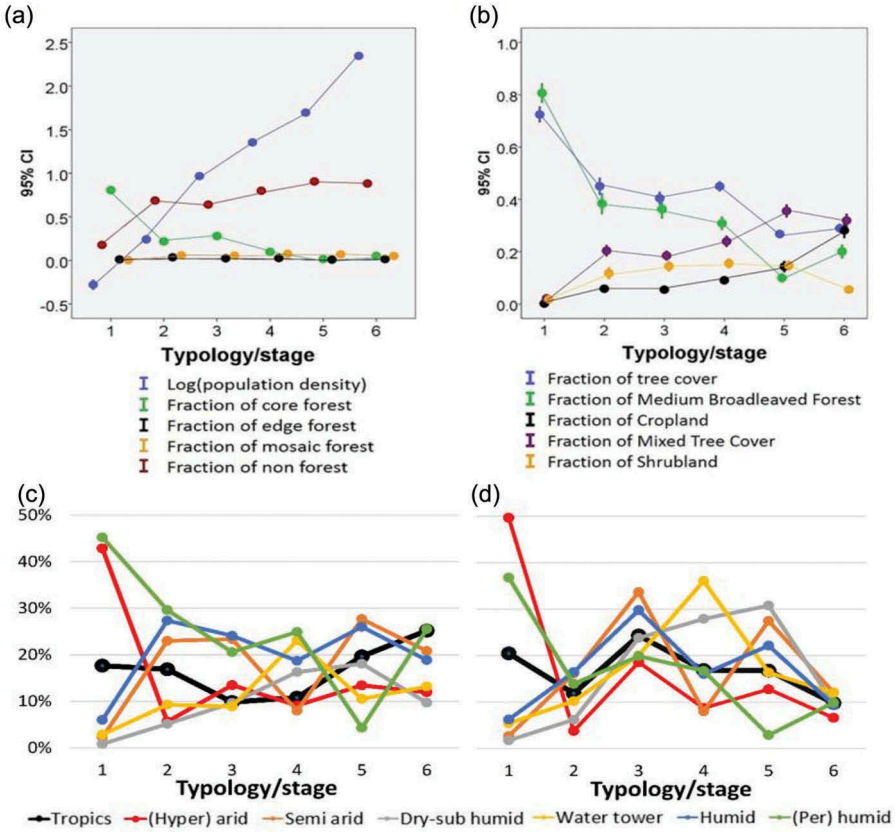

Figure 4. Fraction of forest and tree cover (a) and forest configuration (b) across typologies at subwatershed level; absolute (c) and relative (d) distribution of land across forest transition stages for the various ecological zones

5.2. Agro-ecological zones and human population density

Distribution of human populations across the agro-ecological zones is presented in Figure 5. Compared to the average for the tropics as a whole, human population densities were more evenly distributed in the semi-arid to humid zones and more concentrated in the (per-) humid zone (Figure 5). In the (hyper-) arid zone, this concentration was even stronger, with most people living in cities.

Scatterplots of (log) human population densities and tree cover at sub-basin level showed a clear downward trend but the highest population densities were associated with approximately 20% tree cover (Figure 6). This level of tree cover is still possible in (peri-) urban areas while primarily agricultural areas can have around 10% tree cover (Zomer et al. 2016). Agro-ecological zones clearly influenced the interactions between human population densities and tree cover (Figure 6 (c,d)). Tree cover in the (hyper-) arid and semi-arid zones was much lower compared to those in other zones with similar levels of human population densities.

5.3. Tree crops

The relationship between human population density and the potential distribution of the four major tree crops (Figure 7) reflected the agro-ecological zones in which they primarily occurred. Relatively high population densities were associated with coffee and oil palm areas, and lower ones with rubber and cacao, even though the climatic requirements of oil palm overlap with those of cacao and rubber but not with those of arabica coffee.

Coffee (*Coffea arabica*) was associated with water-tower zones with relatively high human population densities (Figure 7, 8) while oil palm (*Elaeis guineensis*) was effectively restricted to the (per-) humid zone, which it shares with rubber (*Hevea brasiliensis*) and cacao (*Theobroma cacao*). Similar areas suitable for the commodities were compared to the actual harvested areas, taken from FAO (2013) statistics for cacao, rubber and oil palm, and from Bunn et al. (2015) for coffee. The results were that conditions similar to where arabica is now grown feature in 3.56 million km^2 or about 57 times the current harvest area of 0.0625 million km^2 (Figure 9). For cacao, the potential area is 3.96 million km^2 or 40 times the current harvest area of 0.100 million km^2. Rubber could be grown on 5.47 million km^2 or 50 times the current harvest area of 0.110 million km^2. And oil palm could expand to 1.84 million km^2 or 19 times the current harvested area of 0.180 million km^2. Climate change will involve shifts in the geographic boundaries for all tree crops but future expansion is still most likely in areas with similar conditions.

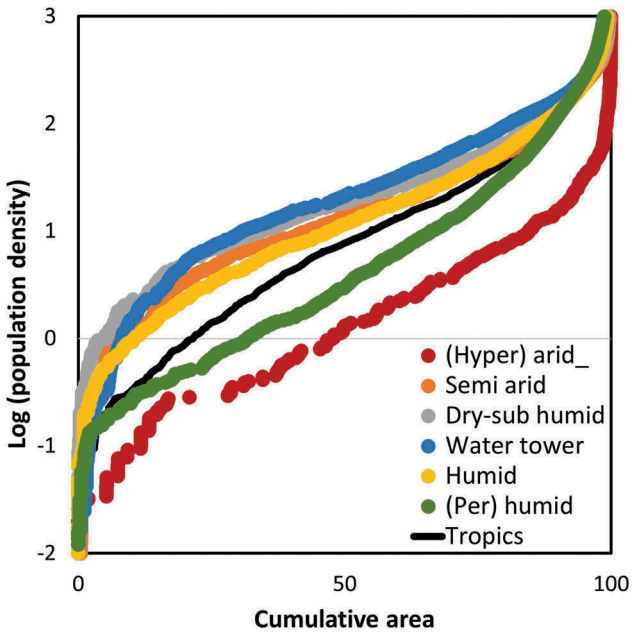

Figure 5. Cumulative frequency distribution (based on area) of human population densities (log form) for the ecological zones (A) and the areas with similarity to the four tree-crop domains (B).

5.4. Representativeness of the three sampling sets

The relative area in the various stages of forest transition differs between the agro-ecological zones, as the operational definition of these stages depends on the specific conditions of the zone (Figure 10). In the (hyper-) arid zone, the forest-transition classification does not apply as there is too little forest; the (per-) humid zone has a larger fraction in the forest-transition stage 1 than any other zone. In the water-tower zone, stage 4 dominates, which is a shift to the right compared with the humid and (per-) humid zones with similar rainfall.

The relative proportions of area in the various ecological zones are fairly well represented in the sentinel landscapes portfolio once the exclusion in the latter of the (hyper-) arid zone is taken into account while for both ASB and PEN, there are strong spatial biases, over representing the (per-) humid zone (Figure 11). In terms of human population, however, the sentinel landscapes underrepresent the (per-) humid zone and, by implication, the average human population density in the (per-) humid zone in the portfolio differs from that in the tropics as a whole. In this zone, human population density is more heterogeneous than in the other zones: there are both relatively large areas of low population density (stage 1) and high concentrations of people in stage 6.

In terms of forest-transition stages, the sentinel landscapes underrepresent the lowest tree cover, stage 6, but are again less biased than ASB and PEN

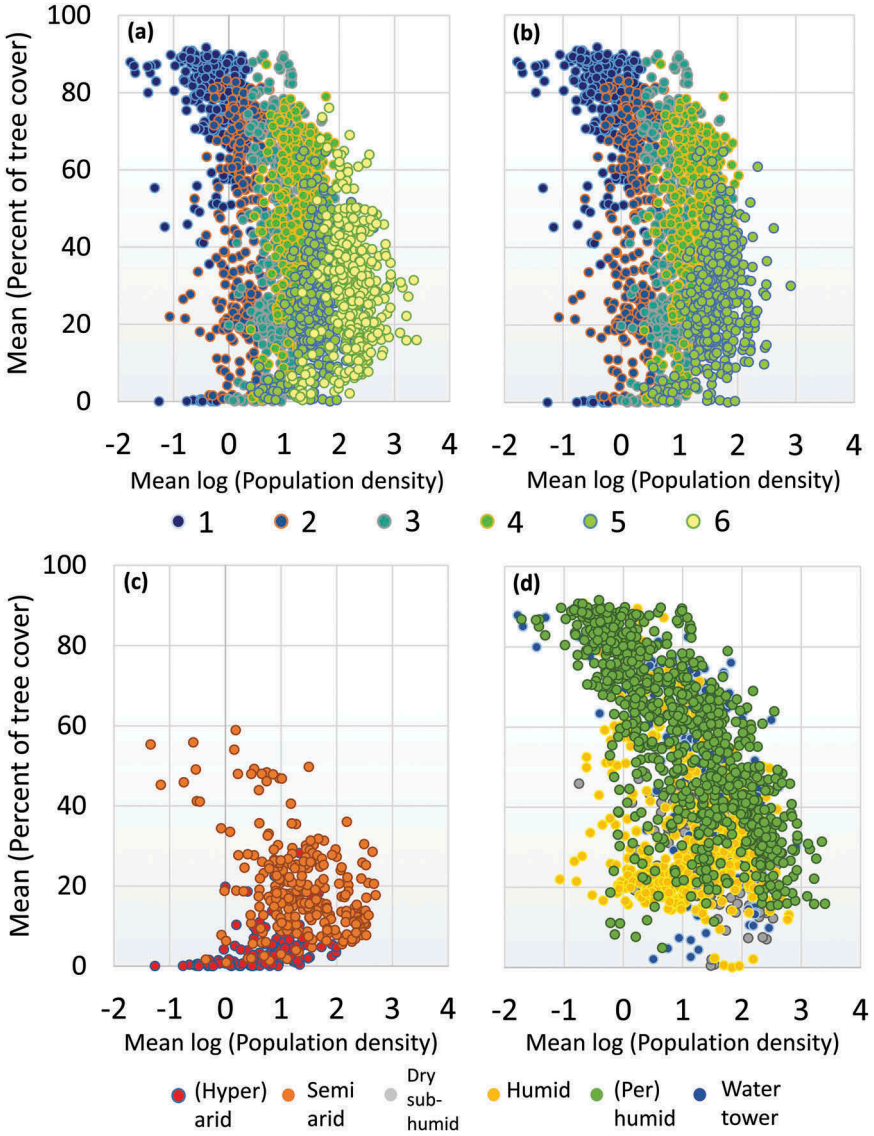

Figure 6. Scatterplots of human population densities (log) and tree cover at sub-watershed level.

Note: (a) for all sub-watersheds differentiated by forest-transition stages; (b) only for sub-watersheds of stages 1–5; (c) for sub-watersheds within (hyper-) arid and semi-arid zones; (d) for sub-watersheds in the remaining agro-ecological zones.

(Figure 12) while the representation of human populations is more biased than for area. Because human population densities tend to have a lognormal rather than normal distribution, we considered the logarithm of population density in this discrepancy.

Figure 11 shows that sub-watersheds in the sentinel landscapes span the full range of tree cover (0–100%) but might lack the extremes of both very high (cities) and very low (core forest) human population densities. The relationship between ecological zones, human population density, HDI and distance to protected areas within the sentinel landscapes corresponds well with those for the tropics as a whole (Figure 13) while conclusions based on ASB and PEN sites may be affected by considerable spatial bias. Similarly, conclusions with regards to tree crops derived from sentinel landscapes' data match those for the tropics as a whole much closer than is the case with ASB or PEN data (Fig. Supp1).

5.5. Summary of issues within agro-ecological zones and sentinel landscapes

Table 3 shows interactions between human population density and agro-ecological zones in shaping ES issues and options to address them, along with the priority areas of FTA.

6. Discussion

In response to the first question raised in the introduction, we conclude that the forest-transition concept and associated stages can, in combination with climate and watershed data, be used as a robust typology (ToP) of the range of human populations and agro-ecological zones and tree crops encountered in the tropics. Increases in human population densities up to about 30 km^{-2} are associated with reductions in tree cover down to about 50% (Figure 12), in line with the regression line

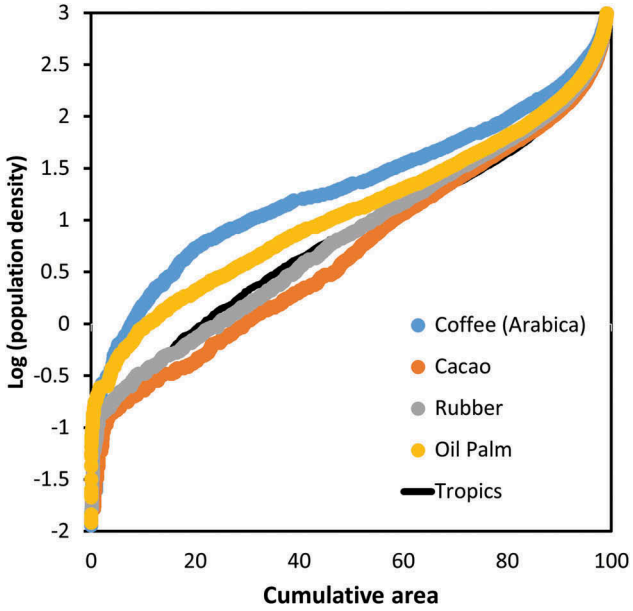

Figure 7. Cumulative frequency distribution (based on area) of human population density (log form) for the areas with similarity to the four tree-crop domains.

established at national scale by Köthke et al. (2013). The highest population densities can have a tree cover of around 20% while the lowest tree cover is found with lower population densities (probably in intensively used agricultural areas). Whether and how these spatial patterns align with qualitative tree-cover transitions (from diverse naturally established to selected trees planted) in rural–urban continuum perspective remains to be tested as part of a ToC and before ToIC can be constructed.

Within this typology, any set of sampling sites can be compared to the global frequency of similar conditions, as answer to question 2. In the comparison as described in Section 4.3, the ASB and PEN data sets were shown to have a rather skewed distribution relative to the tropics at large while the sentinel landscapes had a smaller, but not negligible, spatial bias that addition of more landscapes could reduce. The exclusion of the (hyper-) arid zone is easily acknowledged and justified on the basis of low forest and tree cover as well as low human population densities. Persistent poverty and low HDI, however, call for attention to options for sustainable development in the zone. Per unit of biomass, trees in this zone have great local value.

Based on this analysis, caution is called for as conclusions based on ASB and PEN data are primarily valid within their sample sites only. ASB is primarily concerned with stages 2 and 3 of the forest transition and focused on the humid and (per-) humid tropics. For PEN, the spatial bias to the early stages of the forest transition may colour conclusions about the roles of forest in the

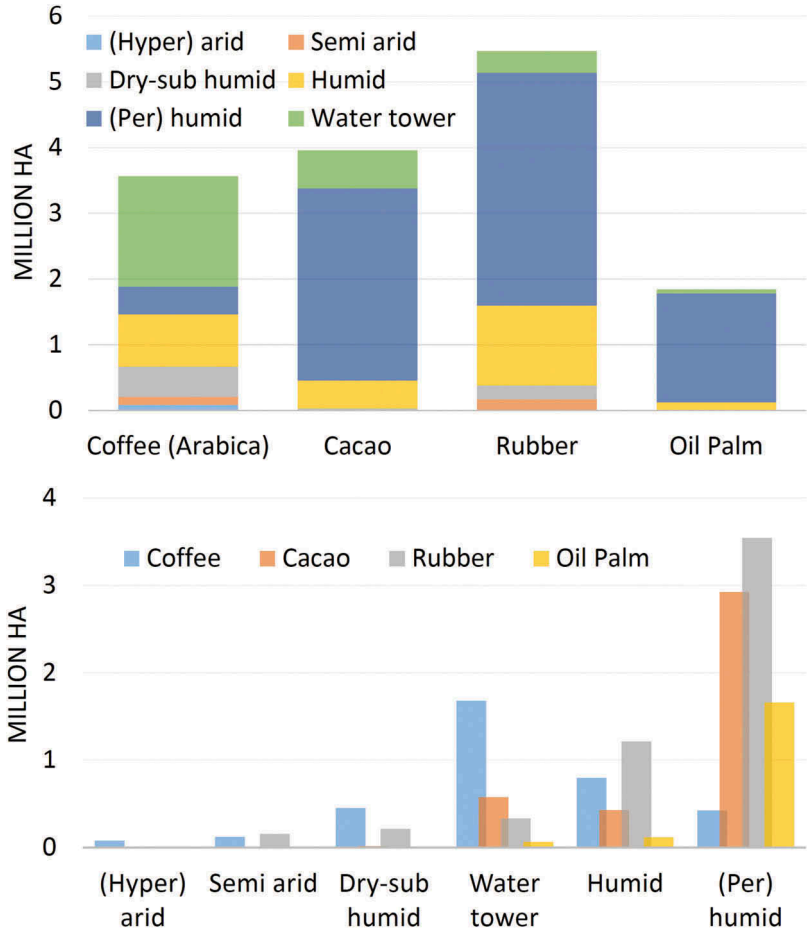

Figure 8. Distribution of four tree crops (area of high similarity with current spatial extent) across the ecological zones.

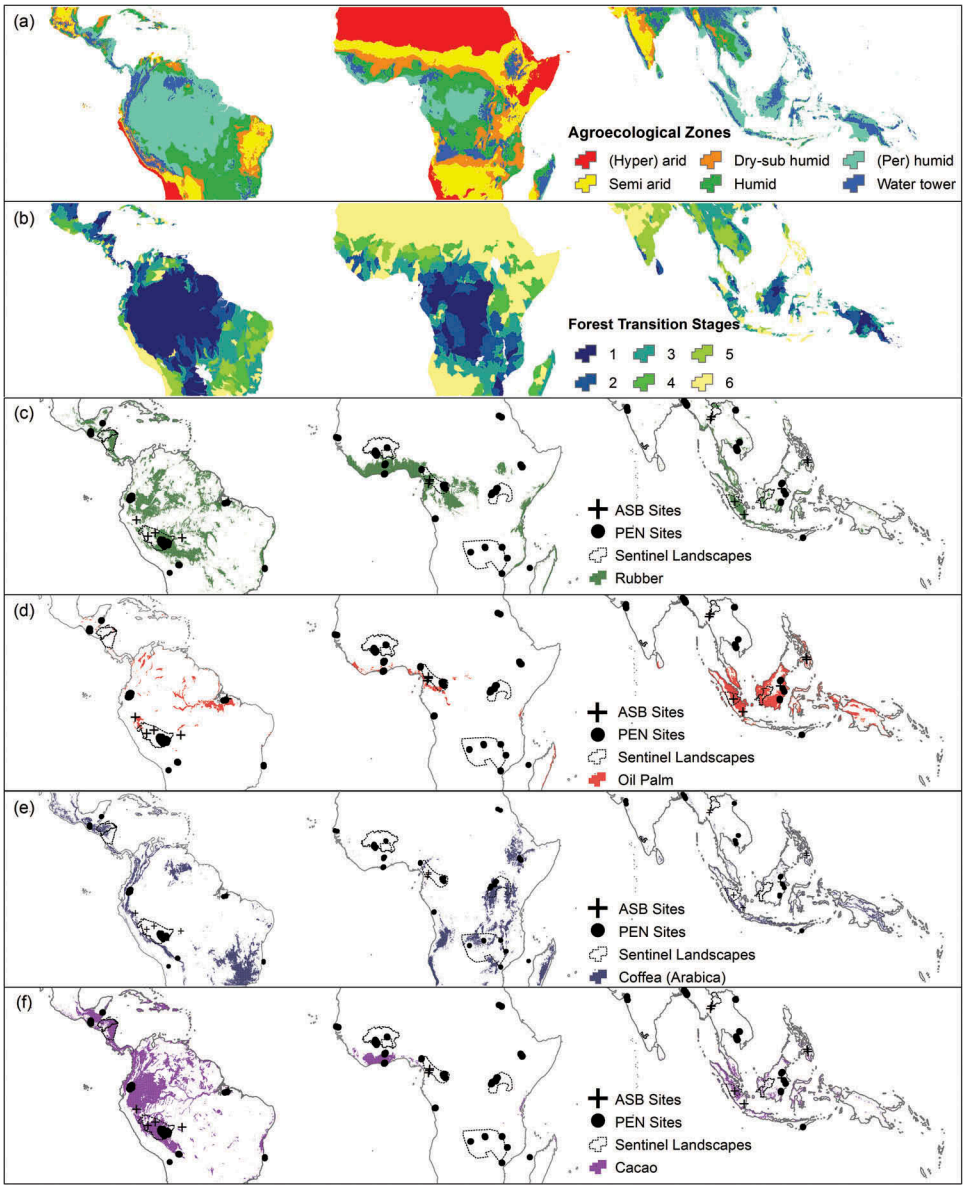

Figure 9. Areas of high similarity with current extent of four tropical tree crops superimposed on ecological zones, with sentinel landscapes, PEN and ASB sites indicated.

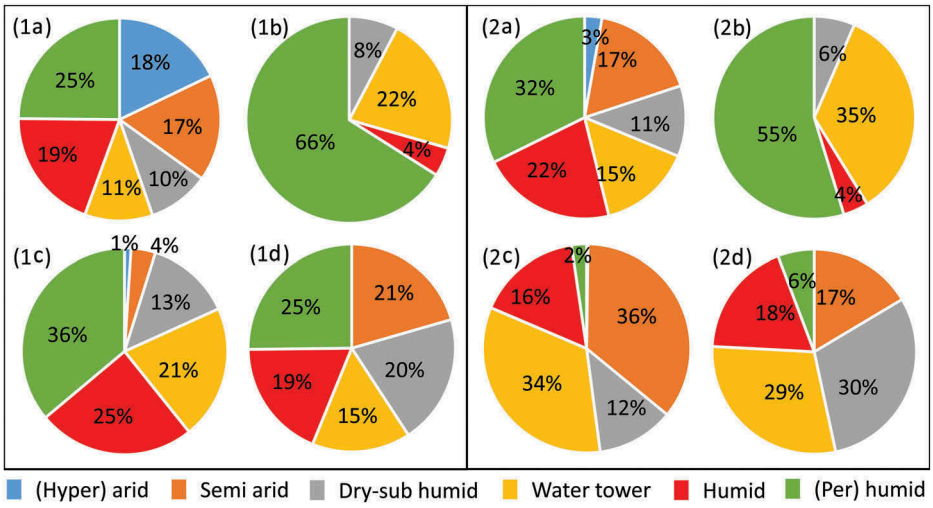

Figure 10. Area (left box) and human population (right box) by ecological zone for the tropics (A), ASB (B), PEN (C) and the sentinel landscapes (D).

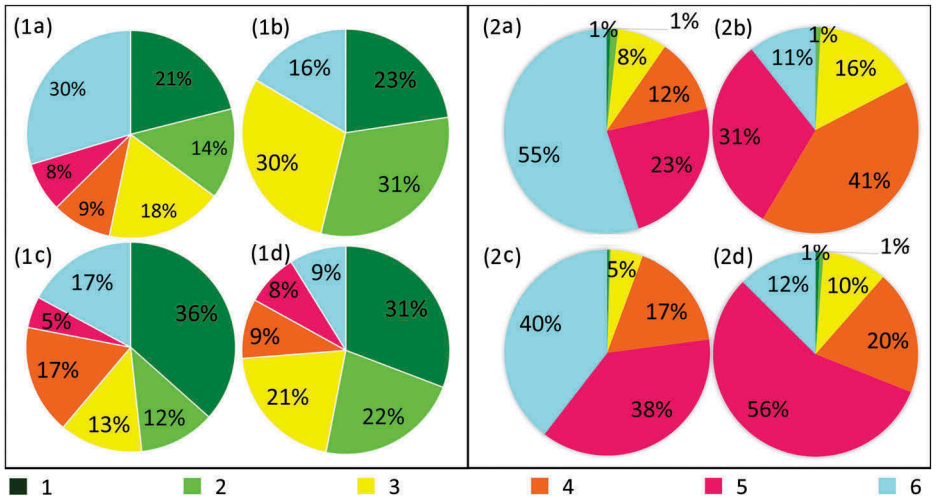

Figure 11. Area (left box) and human population (right box) by forest-transition stage for the tropics (A), ASB (B), PEN (C) and the sentinel landscapes (D).

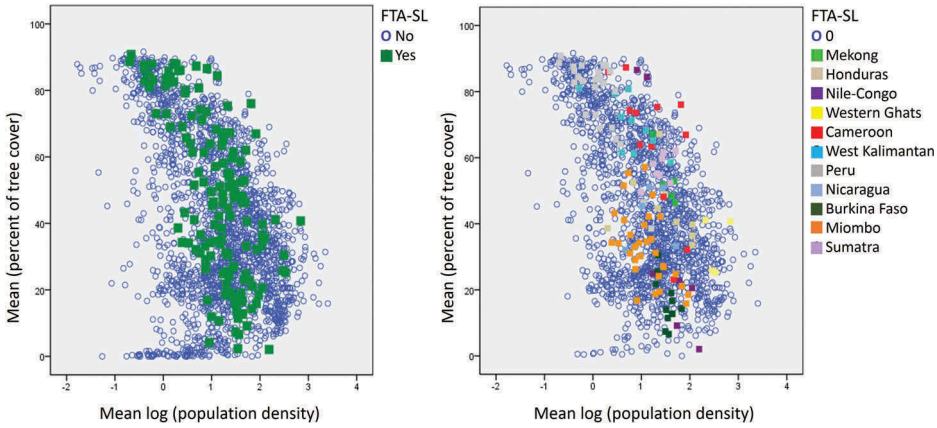

Figure 12. Scatterplots of human population density (log) and tree cover at sub-watershed level: (A) for all sub-watersheds, with labelled sentinel landscape sub-watersheds; (B) for all sub-watersheds with each sentinel landscape sub-watershed uniquely labelled.

relationship between poverty and environment. While the researchers are generally aware of this, the subsequent use of quantitative conclusions may lose track of the caveats and state presumed general relationships on the way poverty and forest relate to each other.

Our results indicate under-studied areas in several agro-ecological zones in the existing global comparative work, especially in drylands (which constitute 18% of the total area and 3% of the human populations in the tropics). Prioritizing future work in drylands will enrich the research. Further, the upward curve of the forest transition deserves further study. The sentinel landscape portfolio could consider including a better representation of the processes the curve represents. For instance, the review by Hecht et al. (2016) emphasizes the climate relevance, beyond carbon storage, by trees in urban and agricultural environments. Given the global movement to restore forests and landscapes, in the future, this forest-transition stage will deserve more effort, especially in delivering ecosystem services in highly populated areas. The recent surge in interest in a two-way relationship between vegetation and rainfall (Ellison et al. 2017) prompts further scrutiny of the sentinel landscape portfolio to include sources and sink areas for terrestrially recycled atmospheric moisture.

In the type of analysis presented here, we have been restricted to credible global data sets. A challenge is to link these data with country-level characterizations of important drivers for policy and legal frameworks. This is especially true for larger and very heterogeneous countries, such as India, Indonesia and Peru. An additional challenge is to account for the direct and indirect influences of international market forces and regulations and link these drivers to the data. Yet another challenge is to include counterfactuals for global drivers (especially if isolated from their specific temporal context), which could require different samplings strategies. Finally, most of the meta-studies of land-use change have

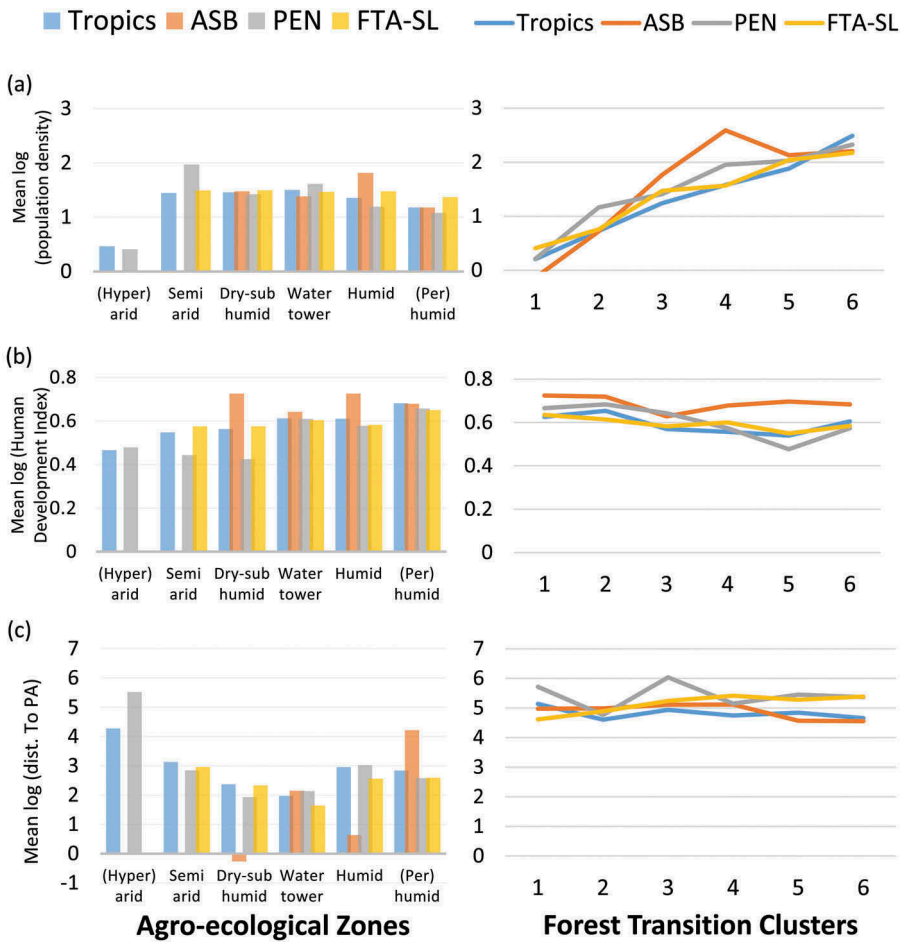

Figure 13. Comparison of key landscape characteristics (human population density, HDI and distance to nearest protected area) for ecological zones and forest-transition stages, as derived for the tropics as a whole and evaluated on the basis of ASB and PEN portfolios.

focused on the consequences for biodiversity and biogeochemical cycles of changes from forest to agriculture; meta-studies of the socioeconomic consequences of the further change to (peri-) urban settings are rare (Van Vliet et al. 2016).

Returning to the question how useful the derived typology is for empirical data collection aimed at usable knowledge, we have so far only addressed the external validity of data (Step 1 in Figure 3) as *ex ante* analysis within a global ToP for the tropics. Before the data collected in any of the sampling frames can be used for valid conclusions about hypotheses – such as those framed in Table 1 – the sampling processes (steps 2 and 3 in Figure 3) need to be assessed for additional biases. The noticeable continental-scale differences in relations between tree cover, climate and human population densities noted by Zomer et al. (2016) suggest major variations at that scale. At the next scale, the diversity of actors and their interactions is the key to understanding the social–ecological systems involved (Galudra et al. 2014; Minang et al. 2015). Such actors can, in a first approximation, be grouped into (1) large-scale companies, which directly respond to global markets; (2) medium-scale actors who generate income from other sectors and invest in land acquisition, either through purchasing from smallholders or paying local actors to obtain land in remote areas and provide labour and (3) smallholders – both indigenous and migrants – who mostly generate their livelihoods from on-farm activities. Whereas for analysis of extrapolation domains, we necessarily rely on global legends in classifying land cover and land use, research that aims to be locally relevant will have to use much richer locally determined typologies of land uses (Beaudoin et al. 2016) and the social and economic actors, and methods to analyse their interactions.

In various contributions to this special issue on eco-certification (Mithöfer et al. 2017), the network of sentinel landscapes across ecological zones was used to provide a balance in the evidence because previous studies tended to focus on areas where certification arose as a response to external concerns over local commodity production, often combining social and ecological issues. With studies on tropical timber, rubber, oil palm, coffee and cacao, the sentinel landscapes provide a sampling frame that includes places where several of the commodities interact outside their core areas, and in which any one commodity might be the dominant source of livelihoods in the interaction with large-scale operations. The synthesis

paper of this special issue will reflect on the difference between conclusions from the FTA-SL's and those from published case studies in this light.

The sentinel landscapes have the potential to enable a further delineations of ToP and articulation with ToC, linking the questions of (1) who? (people, migration histories, land rights, tenure security); (2) what and how? (land-use systems, their profitability, labour absorption and ways of meeting local requirements for food and energy) and (3) where? (spatial organization of landscape elements, for example, aligned with soil toposequences or distances to roads and markets) to the basic three elements of a ToC: (1) so what? (consequences for ecosystem services); (2) who cares? (stakeholders in status quo and those negatively affected) and (3) the opportunities for leverage on the drivers of change (Minang et al. 2015). The tools exist for an analysis of these questions across the knowledge systems of local communities, policymakers and researchers (Van Noordwijk et al. 2013) to allow comparison with baseline data collected from households and spatial sampling points (Vågen et al. 2015). A deeper understanding of how 'induced' change can effectively build on the spontaneous change in these complex socio-ecological systems is needed to make progress across the full spectrum of the SDG.

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References

Angelsen A, Jagger P, Babigumira R, Belcher B, Hogarth NJ, Bauch S, Börner J, Smith-Hall C, Wunder S. 2014. Environmental income and rural livelihoods: a global-comparative analysis. World Dev. 64:S12–S28.

Angelsen A, Rudel TK. 2013. Designing and implementing effective REDD+ policies: a forest transition approach. Rev Environ Econ Policy. 7:91–113.

Beaudoin G, Rafanoharana S, Boissière M, Wijaya A, Wardhana W. 2016. Completing the picture: importance of considering participatory mapping for REDD+ Measurement, Reporting and Verification (MRV). Plos One. 11:e0166592.

Bulkan J, Palmer J. 2016. Global ecological signpost, local reality: the Moraballi creek studies in Guyana and what happened afterwards. Forests. 7:317.

Bunn C, Läderach P, Ovalle Rivera O, Kirschke D. 2015. A bitter cup: climate change profile of global production of Arabica and Robusta coffee. Clim Change. 129:89–101.

[CGIAR-CSI] Consortium for Spatial Information. 2012. Global aridity and global PET methodology. http://www.cgiar-csi.org/wp-content/uploads/2012/11/Global-Aridity-and-Global-PET-Methodology.pdf

Chazdon RL, Brancalion PH, Laestadius L, Bennett-Curry A, Buckingham K, Kumar C, Moll-Rocek J, Vieira ICG, Wilson SJ. 2016. When is a forest a forest? Forest concepts and definitions in the era of forest and landscape restoration. Ambio. 45:538–550.

Chomitz KM, Buys P, De Luca G, Thomas TS, Wertz-Kanounnikoff S. 2007. At loggerheads? Agricultural expansion, poverty reduction, and environment in the tropical forests. Washington (DC): World Bank; p. 1–28.

Clark WC, Tomich TP, van Noordwijk M, Guston D, Catacutan D, Dickson NMMcNie E. 2016. Boundary work for sustainable development: natural resource management at the consultative group on international agricultural research (cgiar). Proc Natl Acad Sciences. 113(17):4615–4622.

Clark WC, van Kerkhoff L, Lebel L, Gallopin GC. 2016. Crafting usable knowledge for sustainable development. Proc Natl Acad Sciences. 113(17):4570–4578.

Connell JP, Kubisch AC. 1998. Applying a theory of change approach to the evaluation of comprehensive community initiatives: progress, prospects, and problems. In: Fulbright-Anderson K, Kubisch A, Connell J, eds. *New approaches to evaluating community initiatives* 2: theory, measurement, and analysis. Washington DC, USA: Aspen Institute; p. 15–44.

Costanza R, de Groot R, Sutton P, van der Ploeg S, Anderson SJ, Kubiszewski I, Farber S, Turner RK. 2014. Changes in the global value of ecosystem services. Global Environmental Change. 26:152–158.

Dewi S, Ekadinata A, Indiarto D, Nugraha A, Van Noordwijk M. 2015. Empowering local stakeholders for planning, Indonesia. In: Chavez-Tafur C, Zagt R, eds. Towards productive landscapes. Wageningen: ETFRN News 56, Tropenbos International; p. 51–57.

Dewi S, van Noordwijk M, Ekadinata A, Pfund JL. 2013. Protected areas in relation to landscape multifunctionality: squeezing out intermediate land use intensities in the tropics? Land Use Policy. 30:38–56.

Ellison D, Morris CE, Locatelli B, Sheil D, Cohen J, Murdiyarso D, Gutierrez V, van Noordwijk M, Creed IF, Pokorny J, et al. 2017. Trees, forests and water: cool insights for a hot world. Global Environmental Change. 43:51–61.

[FAO] Food and Agriculture Organization. 2013. Fao statistical yearbook. Rome: FAO.

Forstmeier W. 2015. Caution is needed when 90% of all possible estimates remain unpublished: a comment on Arct et al. Behav Ecol. 26:972–973.

Freeman O, Duguma L, Minang P. 2015. Operationalizing the integrated landscape approach in practice. Ecol Soc. 20(1). art24. https://doi.org/10.5751/ES-07175-200124

Galudra G, van Noordwijk M, Agung P, Suyanto S, Pradhan U. 2014. Migrants, land markets and carbon emissions in Jambi, Indonesia: land tenure change and the prospect of emission reduction. Mitig Adapt Strateg Glob Change. 19:715–732.

George AL, Bennett A. 2005. Case studies and theory development in the social sciences. Cambridge (Massachusets, USA): MIT Press.

Gillison AN, Bignell DE, Brewer KR, Fernandes EC, Jones DT, Sheil D, May PH, Watt AD, Constantino R, Couto EG, et al. 2013. Plant functional types and traits as biodiversity indicators for tropical forests: two biogeographically separated case studies including birds, mammals and termites. Biodivers Conserv. 22:1909–1930.

Grainger A. 2008. Difficulties in tracking the long-term global trend in tropical forest area. Proc Nat Acad Sci. 105:818–823.

Hecht SB, Pezzoli K, Saatchi S. 2016. Trees have already been invented: carbon in woodlands. Collabra. 2(24):1–34.

Jackson LE, Pulleman MM, Brussaard L, Bawa KS, Brown GG, Cardoso IM, De Ruiter PC, García-Barrios L, Hollander AD, Lavelle P, et al. 2012. Social-ecological and regional adaptation of agrobiodiversity management across a global set of research regions. Global Environmental Change. 22:623–639.

Jagger P, Luckert MMK, Duchelle AE, Lund JF, Sunderlin WD. 2014. Tenure and forest income: observations from a global study on forests and poverty. World Dev. 64: S43–S55.

Jouvet PA, Wolfersberger J 2015. *Growth, green capital and public policies* (No. 1504). http://www.webmeets.com/files/papers/eaere/2015/375/GGCPP.pdf

Kicinski M, Springate DA, Kontopantelis E. 2015. Publication bias in meta-analyses from the cochrane database of systematic reviews. Stat Med. 34:2781–2793.

Köthke M, Leischner B, Elsasser P. 2013. Uniform global deforestation patterns—an empirical analysis. For Policy Econ. 28:23–37.

Köthke M, Schröppel BElsasser P. 2014. National redd+ reference levels deduced from the global deforestation curve. Forest Policy and Econ. 43:18-28.

Kowarsch M, Garard J, Riousset JP, Lenzi D, Dorsch MJ, Knopf B, Harrs JA, Edenhofer O. 2016. Scientific assessments to facilitate deliberative policy learning. Palmgrave Communications. doi:10.1057/palcomms.2016.92

Kuyah S, Öborn I, Jonsson M, Dahlin AS, Barrios E, Muthuri C, Malmer A, Nyaga J, Magaju C, Namirembe S, et al. 2016. Trees in agricultural landscapes enhance provision of ecosystem services in sub-Saharan Africa. Int J Biodiversity Science, Ecosystem Serv Manag. 12:255–273.

Leimona B, Lusiana B, van Noordwijk M, Mulyoutami E, Ekadinata A, Amaruzaman S. 2015. Boundary work: knowledge co-production for negotiating payment for watershed services in Indonesia. Ecosystem Serv. 15:45–62.

Lusiana B, van Noordwijk M, Cadisch G. 2012. Land sparing or sharing? Exploring livestock fodder options in combination with land use zoning and consequences for livelihoods and net carbon stocks using the FALLOW model. Agric Ecosyst Environ. 159:145–160.

Mather AS, Needle CL. 1998. The forest transition: a theoretical basis. Area. 30:117–124.

Matthews RB, Van Noordwijk M, Lambin E, Meyfroidt P, Gupta J, Verchot L, Hergoualc'h K, Veldkamp E. 2014. Implementing REDD+ (Reducing Emissions from Deforestation and Degradation): evidence on governance, evaluation and impacts from the REDD-ALERT project. Mitigation and Adaptation Strategies for Global Change. 19:907–925.

Meyfroidt P, Carlson KM, Fagan ME, Gutiérrez-Vélez VH, Macedo MN, Curran LM, DeFries RS, Dyer GA, Gibbs HK, Lambin EF, et al. 2014. Multiple pathways of commodity crop expansion in tropical forest landscapes. Env Res Lett. 9:074012.

Minang PA, van Noordwijk M, Duguma LA, Alemagi D, Do TH, Bernard F, Agung P, Robiglio V, Catacutan D, Suyanto S, et al. 2014b. REDD+ Readiness progress across countries: time for reconsideration. Clim Policy. 14(6):685–708.

Minang PA, Van Noordwijk M, Freeman OE, Mbow C, De Leeuw J, Catacutan D. 2015. Climate-smart landscapes: multifunctionality in practice. Nairobi, Kenya: World Agroforestry Centre (ICRAF); p. 404 pp.

Minang PA, van Noordwijk M, Kahurani E. 2014a. Partnership in the tropical forest margins: a 20-year journey in search of alternatives to slash-and-burn. Nairobi, Kenya: World Agroforestry Centre (ICRAF), . 241 pp.

Mithöfer D, van Noordwijk M, Leimona B, Cerutti PO. 2017. Certify and shift blame, or resolve issues? Environmentally and socially responsible global trade and production of timber and tree crops. Int J Biodiversity Science, Ecosystem Serv Manag. 13:72–85.

Murdiyarso D, Van Noordwijk M, Wasrin UR, Tomich TP, Gillison AN. 2002. Environmental benefits and sustainable land-use options in the Jambi transect, Sumatra. J Vegetation Sci. 13:429–438.

Muthayya S, Rah JH, Sugimoto JD, Roos FF, Kraemer K, Black RE. 2013. The global hidden hunger indices and maps: an advocacy tool for action. PLoS One. 8(6):e67860.

Ordonez JC, Luedeling E, Kindt R, Tata HL, Harja D, Jamnadass R, van Noordwijk M. 2014. Constraints and opportunities for tree diversity management along the forest transition curve to achieve multifunctional agriculture. Curr Opin Environ Sustainability. 6:54–60.

Ovalle Rivera O, Läderach P, Bunn C, Obersteiner M, Schroth G. 2015. Replication data for: projected shifts in *Coffea arabica* suitability among major global producing regions due to climate change. Harvard Dataverse, V3. doi:10.7910/DVN/29634

Paterson RRM, Kumar L, Taylor S, Lima N. 2015. Future climate effects on suitability for growth of oil palms in Malaysia and Indonesia. Sci. Rep. 5:14457.

Phillips SJ, Anderson RP, Schapire RE. 2006. "Maximum entropy modeling of species geographic distributions". Ecol Modell. 190:231–259.

Posner SM, McKenzie E, Ricketts TH. 2016. Policy impacts of ecosystem services knowledge. Proc Natl Acad Sci. 113:1760–1765.

Putz FE, Zuidema PA, Synnott T, Peña-Claros M, Pinard MA, Sheil D, Vanclay JK, Sist P, Gourlet-Fleury S, Griscom B, et al. 2012. Sustaining conservation values in selectively logged tropical forests: the attained and the attainable. Conservation Lett. 5:296–303.

Ray D, Behera MD, Jacob J. 2015. Predicting the distribution of rubber trees (*Hevea brasiliensis*) through ecological niche modelling with climate, soil, topography and socioeconomic factors. Ecol Res. 31:75–91.

Renkow M, Byerlee D. 2010. The impacts of CGIAR research: A review of recent evidence. Food Policy. 35:391–402.

Sachs J, Schmidt-Traub G, Kroll C, Durand-Delacre D, Teksoz K. 2016. SDG Index and dashboards - global report. New York: Bertelsmann Stiftung and Sustainable Development Solutions Network (SDSN).

Sasaki N, Putz FE. 2009. Critical need for new definitions of "forest" and "forest degradation" in global climate change agreements. Cons Lett. 2(5):226–232.

Schroth G, Läderach P, Martinez-Valle A, Bunn C, Jassogne L. 2016. Vulnerability to climate change of cocoa in West Africa: patterns, opportunities and limits to adaptation. Harvard Dataverse, V1. doi:10.7910/DVN/5W7LGW

Shepherd KD, Shepherd G, Walsh MG. 2015. Land health surveillance and response: a framework for evidence-informed land management. Agric Syst. 132:93–106.

Steffen W, Richardson K, Rockström J, Cornell SE, Fetzer I, Bennett EM, Biggs R, Carpenter SR, De Vries W, De Wit CA, et al. 2015. Planetary boundaries: guiding human development on a changing planet. Science. 347(6223):1259855.

Thaler RH, Sunstein CR. 2008. Nudge: improving decisions about health, wealth, and happiness. New Haven: Yale University Press.

Tomich TP, Cattaneo A, Chater S, Geist HJ, Gockowski J, Kaimowitz D, Lambin EF, Lewis J, Ndoye O, Palm CA, et al. 2005. Balancing agricultural development and environmental objectives: assessing trade-offs in the humid tropics. In: Palm CA, Vosti SA, Sanchez PA, Ericksen PJ, Juo ASR, Eds. Slash and burn: the search for alternatives. New York (USA): Columbia University Press; p. 415–440.

Tomich TP, Timmer DW, Velarde SJ, Alegre J, Areskoug V, Cash DW, Cattaneo A, Ericksen P, Joshi L, Kasyoki J, et al. 2007. Integrative science in practice: process perspectives from ASB, the partnership for the tropical forest margins. Agric Ecosyst Environ. 121:269–286.

Vågen TG, Winowiecki LA, Tamene Desta L, Tondoh JE. 2015. The Land Degradation Surveillance Framework (LDSF)—field Guide v4_1. Nairobi, Kenya: World Agroforestry Centre.

Van Noordwijk M. 2014. Agroforestry as plant production system in a multifunctional landscape. Inaugural lecture upon taking up the special professorship in Agroforestry 16 October 2014. Wageningen, Netherlands: Wageningen University. ; p. 24 p.

Van Noordwijk M, Bizard V, Wangpakapattanawong P, Tata HL, Villamor GB, Leimona B. 2014. Tree cover transitions and food security in Southeast Asia. Global Food Security. 3:200–208.

Van Noordwijk M, Coe R, Sinclair F 2016a. Central hypotheses for the third agroforestry paradigm within a common definition. World Agroforestry Centre (ICRAF) Working paper 233, Bogor (Indonesia) doi:10.5716/WP16079.PDF

Van Noordwijk M, Hairiah K, Partoharjono S, Labios RV, Garrity DP. 1997. Sustainable food-crop based production systems, as alternative to *Imperata* grasslands? Agroforestry Syst. 36:55–82.

Van Noordwijk M, Kim Y-S, Leimona B, Hairiah K, Fisher LA. 2016b. Metrics of water security, adaptive capacity and agroforestry in Indonesia. Curr Opin Environ Sustainability. 21:1–8.

Van Noordwijk M, Lusiana B, Leimona B, Dewi S, Wulandari D, eds. 2013. Negotiation-support toolkit for learning landscapes. Bogor, Indonesia: World Agroforestry Centre (ICRAF) Southeast Asia Regional Program.

Van Noordwijk M, Lusiana B, Villamor GB, Purnomo H, Dewi S. 2011. Feedback loops added to four conceptual models linking land change with driving forces and actors. Ecol Soc. 16(1):r1. [online] URL. http://www.ecologyandsociety.org/vol16/iss1/resp1/

Van Noordwijk M, Minang PA 2009. If we cannot define it, we cannot save it. *ASB Policy brief, 15*. World Agroforestry Centre, Nairobi, Kenya.

Van Noordwijk M, Minang PA, Hairiah K. 2015. Swidden transitions, in an era of climate change. In: Cairns MF, Ed. Shifting cultivation and environmental change: indigenous people, agriculture and forest conservation. Oxon (UK: Earthscan; p. 261–280.

Van Noordwijk M, Suyamto DA, Lusiana B, Ekadinata A, Hairiah K. 2008. Facilitating agroforestation of landscapes for sustainable benefits: trade-offs between carbon stocks and local development benefits in Indonesia according to the FALLOW model. Agric Ecosyst Env. 126:98–112.

Van Noordwijk M, Villamor GB 2014. Tree cover transitions in tropical landscapes: hypotheses and cross-continental synthesis. GLP Newsletter Issue No. 10, http://www.globallandproject.org/newsletter/glp_news.php

Van Vliet J, Magliocca NR, Büchner B, Cook E, Benayas JMR, Ellis EC, Heinimann A, Keys E, Lee TM, Liu J, et al. 2016. Meta-studies in land use science: current coverage and prospects. Ambio. 45:15–28.

Von Thünen H. 1842. Der isolierte Staat in Beziehung auf Landwirtschaft und Nationalökonomie. 3rd ed. Berlin: Hermann Schumacher-Zarchlin.

Weiss CH. 1997. How can theory-based evaluation make greater headway? Eval Rev. 21:4.

Wiersum KF. 1997. From natural forest to tree crops, co-domestication of forests and tree species, an overview. NJAS Wageningen J Life Sci. 45:425–438.

Williams M. 2003. Deforesting the earth: from prehistory to global crisis. Chicago: University of Chicago Press.

Wunder S, Angelsen A, Belcher B. 2014. Forests, livelihoods, and conservation: broadening the empirical base. World Dev. 64:S1–S11.

Zomer RJ, Neufeldt H, Xu J, Ahrends A, Bossio DA, Trabucco A, van Noordwijk M, Wang M. 2016. Global tree cover and biomass carbon on agricultural land: the contribution of agroforestry to global and national carbon budgets. Sci Rep. 6:29987.