BURLEIGH DODDS SERIES IN AGRICULTURAL SCIENCE

Achieving sustainable management of tropical forests

Edited by Professor Jürgen Blaser, Bern University of Applied Sciences, Switzerland

Patrick D. Hardcastle, Forestry Development Specialist, UK

E-CHAPTER FROM THIS BOOK

burleigh dodds SCIENCE PUBLISHING

Defining sustainable forest management (SFM) in the tropics

Francis E. Putz, University of Florida-Gainesville, USA; and Ian D. Thompson, Thompson Forest Ltd.-Kelowna, Canada

1 Introduction

> 'Any claim of sustainable forest management should evoke the queries: What is sustained? What were the tradeoffs? Over what spatial and temporal scales?'

Sustainable forest management (SFM) is a conceptual codification of forest management practices that continues to evolve from its focus in the 1800s on sustained timber yields. Since the 1987 publication of 'Our Common Future' (also known as the Brundtland Report) by the World Commission on Environment and Development, the definition of SFM has expanded to include the much broader goals of sustaining the economic, social, and environmental benefits from forest. In the words of the United Nations, SFM is a 'dynamic and evolving concept [that] aims to maintain and enhance the economic, social, and environmental values of all types of forests, for the benefit of present and future generations' (FAO, 2018). This broadening of considerations is reflected in the definitions provided by the International Timber Trade Organization (ITTO):

http://dx.doi.org/10.19103/AS.2020.0074.19

Chapter taken from: Blaser, J. and Hardcastle, P. D. (ed.), *Achieving sustainable management of tropical forests*, Burleigh Dodds Science Publishing, Cambridge, UK, 2021, (ISBN: 978 1 78676 248 1; www.bdspublishing.com)

> [SFM is] the process of managing forest to achieve one or more clearly specified objectives of management with regard to the production of a continuous flow of desired forest products and services without undue reduction of its inherent values and future productivity and without undue undesirable effects on the physical and social environment. (ITTO, 2016)

In this chapter, we elaborate on these definitions of SFM in an effort to promote clarity about the avoidable and unavoidable trade-offs associated with all management decisions; management 'for' something is necessarily management 'against' something else. We also hope to promote measurement, monitoring, and verification of the various indicators of sustainability. It is written out of concern for the obvious deniability of many claims of SFM and the inclusion of vague terms in its definitions such as 'over the long term' without specification of time scales and 'without undue reduction' without clarification of 'undue'. In the chapter we also advocate for clarity about spatial scales and for expansion of the scale at which sustainability is considered from stands up to forested landscapes. Finally, we believe that graphical depictions of the components of SFM, like the one we propose, will help clarify these trade-offs and generally aid our understanding about the challenges of reaching the SFM goal.

Our approach follows the effort of Thompson et al. (2013) to clarify the complex condition of 'forest degradation' through its disaggregation into component biophysical parts. We hope that our un-clustering of the various dimensions of SFM will similarly help inform efforts to promote and evaluate forest management sustainability. We also expand the scope of SFM from individual stands, to which many definitions of SFM pertain, to the scale of forested landscapes, in keeping with other efforts towards the comprehensiveness of land use planning (e.g. Sayer et al., 2016). We hope that our efforts are of use in the development of principles, criteria, and indicators of sustainability for programs such as the Sustainable Landscape Production Certification program under development by the Landscape Standard Consortium (https://verra.org/project/landscape-standard/).

We proceed in this effort to clarify landscape-scale SFM by defining its principal components and then considering them at different spatial and temporal scales. We strive for measurability and precision, in recognition of the diversities of landscapes with forests, characteristics of managed forests and forest managers, forest management goals, and trade-offs associated with land-use interventions. We nevertheless recognize that any definition of SFM with wide applicability and acceptability must be somewhat vague and mutable. That said, whatever the definition of SFM that is adopted, clarity and measurability should be fundamental objectives.

2 Evolving concepts of sustainability

Since well before the Brundtland Report (1987), economists have grappled with what is meant by 'sustainable' and 'sustainability' (e.g. Solow, 1956). While the concerns of foresters about sustainability date back many centuries (reviewed by Wiersum, 1995), the focus was historically on sustaining timber yields, with non-diminishing yields being the goal of management. This focus, which remains relevant, is now referred to as 'strong' sustainability (reviewed by Luckert and Williamson, 2005). In contrast, 'weak' sustainability allows for the transfer of natural capital (e.g. timber stocks or biodiversity) for economic, built, social, and human capital as long as the overall sum of these five forms of capital does not decline. Recognition of the embeddedness of managed forests in landscapes of various other forest and non-forest land uses is more recent, and is reflected in what are known as landscape-level and jurisdictional approaches to sustainability (e.g. Sayer et al., 2016; Stickler et al., 2018; Runting et al., 2019; Griscom et al., 2019.

Expansions of SFM's scope were unavoidably accompanied by modification of the definition of 'sustainability' from one that requires non-diminishing supplies to one that is much more multi-dimensional and negotiable. One consequence of this broadening of the definition and the allowance for capital transfers is that it allows claims of 'sustainable development' and 'sustainable infrastructure'. The expansion of the concept of 'sustainability' to non-renewable resources, as exemplified by the *Journal of Sustainable Mining*, suggests that 'sustainable' is now just a synonym for 'responsible' or 'good' (Putz, 2018).

Here we consider a definition for SFM in the realm of tropical forests that accounts for multiple classes of managed, exploited, and unmanaged forests across landscapes that can include protected areas, selectively logged natural forests, logged forests subjected to additional silvicultural treatments to increase stocking and growth of commercial species, plantations, and forest restoration areas (Fig. 1). We also separate out for consideration forests under the control of rural, local, and/or indigenous communities in full recognition that their lands may host any of these sorts of management practices. Our approach to SFM differs from multiple-use forest management, which typically focuses on compromising goals in areas subjected to similar treatments in what has become recognized as 'land-sharing' (e.g. Phalan et al., 2011). Our approach also expands the 'triad' concept of Messier et al. (2009) in which the focus is on natural forest management, plantation forestry, and forest protection by additionally considering community-based forest management and forest restoration. We hope to shed light on the various benefits derived from different portions of landscapes with forests. We strongly recommend that this approach be expanded by the inclusion

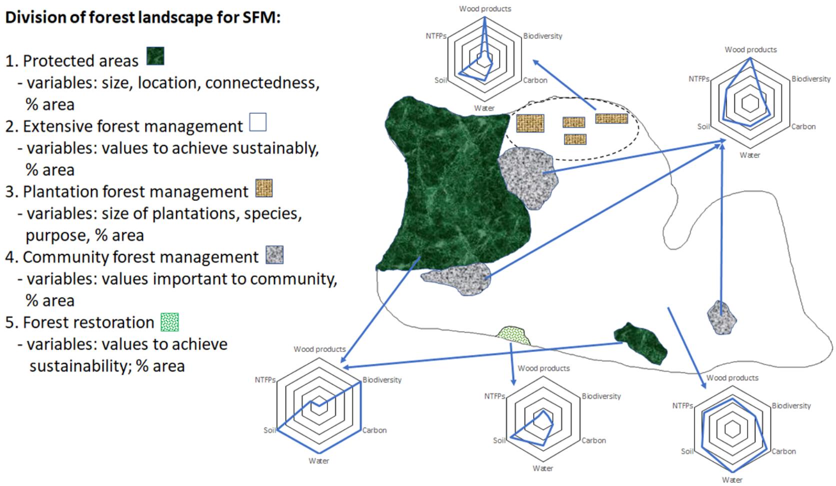

Figure 1 Partitioning a forest landscape for assessment of SFM (extensive and intensive natural forest management not differentiated, but the latter should occur in accessible areas such as near main roads). Each category of forest land-use is evaluated on the basis of the same six criteria illustrated by the biophysical resource hexagons. The overall sum of scores is a relative measure of SFM at the landscape scale for a particular year (The concentric lines inside the perimeter of the main polygons refer to % values of the indicators relative to a primary forest baseline; the blue lines are examples of monitored values from one year).

of more values that are social and economic, and consideration of other land-uses.

The criteria on which we focus are wood products, non-timber forest products, soils, water, carbon, and biodiversity. We assume that the aggregate measure of the extent to which these values are maintained is a measure of the degree of SFM. We recognize that this approach to the assessment of SFM is mostly restricted to biophysical attributes affected by intentional forest management and forest resource exploitation, but we do consider the sustainability of profits from the sale of timber and non-timber forest products (see below). Social, cultural, and other sorts of criteria for more complete assessments of SFM should be readily added to this basic system. Another possible modification that deserves consideration is the use of asymmetrical polygons that illustrate differences in emphases on the various values/criteria.

3 Appropriate scales for assessment of SFM

One difficulty with the concept of SFM is the uncertainty about the scale at which it should be assessed. While forestry practices are implemented at stand scales, given the many values of forests and the inherent trade-offs in any stand-level management regime, SFM might more logically be considered at landscape scales (e.g. Vincent and Binkley, 1993; Boscolo, 2000). In other words, stand-scale management cannot maintain everything, everywhere, all the time, nor should it aim to do so. Only at the landscape level can all forest values be sustained over time and space, if managed properly. With our approach, the sum of scores on all the objectives (i.e. axes in the trade-off polygons), weighted by the area of each land-use, represents a landscape-level measure of sustainability for all criteria (i.e. goods, services, water, carbon, biodiversity, recreation, etc.) at one point in time. We suggest that, with the exception of the most intensive short-rotation tree plantations, multiple-use management with multiple hoped-for benefits is likely to be the goal for most portions of managed forest landscapes. We also recognize that the constraints on achieving the goal of multiple forest management are basically the same, and equally as daunting, as those for SFM in the tropics (Sabogal et al., 2013).

Given the diversity of forest management options, each with its own inherent trade-offs, as well as the diversity of forest conditions, a landscape approach seems appropriate as a first step toward figuring out how the undesired outcomes of management can be minimized overall. Explicit recognition of trade-offs among land uses allows increased rationality in their assessment, as opposed to attempts to maintain all values everywhere all the time. Sizes of managed landscapes sufficient for SFM are likely to be dictated by existing natural constraints, negotiation, geography, and politics, but local values may bound all the others at the upper end of the spatial scale. In these cases, landscape size should reflect some value that would unavoidably be depleted or its maintenance rendered uneconomical as a result of managing at too small a scale. For example, rare species of trees or large mammals may require several thousand square kilometers for persistent populations (e.g. Schulze et al., 2005; Wikramanayake et al., 2011). In these cases, we do not suggest managing for the minimum viable population, but rather some upper value that accounts for temporal stochasticity as well as for controlled and uncontrolled exploitation. In other geographies, for SFM to be practicable, a sufficiently large area may be required to provide an adequate economic benefit from a valued resource (Nasi and Frost, 2009; Sabogal et al., 2013).

4 SFM trade-offs at different scales

Management, by definition, requires that when some species, conditions, processes, or values are managed for, some other species, conditions, processes, or values are managed against. In other words, trade-offs are as inherent to the act of management as they are to resource exploitation. In tropical forests, SFM requires consideration of sufficiently large landscapes for all values to persist, enabling sustainable wood production, sufficient ecosystem services for communities, and no losses of species. For this purpose, areas within the landscape used for different purposes are partitioned into use-categories, often with different values or benefits to society (Fig. 1). For example, intensively managed plantations have limited value for biodiversity but high value for commercial wood production, while protected areas are the opposite. These differences in value-maintenance among land-use categories are represented by the blue value lines inside the trade-off polygons; value lines that approach the outer perimeter of the polygon represent value maintenance relative to the primary forest baseline. Indicators for each of the six criteria are landscape-specific and depend on the specific circumstances and the selected objectives for each forest category. For example, a key indicator for protected areas might be elephant population persistence, while for a plantation, indicators might be a certain amount of wood, fuel, or rubber produced per year. We illustrate five forest categories (Fig. 1), but there are others, such as local community conservation areas and private lands, that may deserve consideration. Regardless of the number of forest categories, each can be evaluated on the basis of the same six criteria in full recognition of the different objectives for which different land-use categories are managed. For more complete assessments of SFM, criteria need to be added that capture the social, economic, and additional environmental values. Extension of this approach to non-forest land uses is possible, but will likely involve specification of new evaluation criteria.

5 Defining terms in SFM

To clarify how SFM might be attained and measured, we commence with definitions of forest (as opposed to plantation), management, and sustained yield. Again, we do not believe that our definitions are sacrosanct, but argue that agreed-upon definitions are needed lest discussions of SFM continue to be plagued by vagueness and ambiguity. A limitation of our approach is that the focus is on forest landscapes and excludes land cleared for agriculture, mining, impoundments or other non-forest land uses, even those with substantial tree cover (e.g., urban forests and some agroforestry systems). We also recognize that our focus is principally biophysical, but hope that our approach is sufficiently adaptable to accommodate social and economic considerations.

Forest versus Plantation: Tree-covered landscapes among which forests with natural regeneration are differentiated from plantations in which all future crop trees are planted (Sasaki and Putz, 2009; Putz and Redford, 2010) often with the intention to clear-cut at frequent intervals. This distinction is made in full recognition of intermediate states, such as selectively logged natural forests that are enriched by planting along cleared lines or in felling gaps. We also recognize that the deleterious environmental impacts of intensive plantation management can be mitigated in many ways such as by maintaining natural forest corridors in riparian areas, increasing structural and floristic diversity within stands, and extending rotations (e.g. Dudley, 2005).

Management: Intentional actions are taken with specified goals, to differentiate management from exploitation and its consequence, degradation. Management occurs at multiple scales and intensities, as fitting for different objectives and in recognition of different trade-offs, and includes protection as well as harvesting. For natural forest management, whether conducted by communities or industrial firms, we differentiate low intensity but extensive approaches based on reduced-impact logging, from higher intensity sorts of silvicultural interventions (e.g. liberation thinning and enrichment planting).

Sustained Yields: The topic of sustained yield forestry has received attention for centuries and may seem more straightforward a consideration than biodiversity, aesthetics, or other values, but we believe it is helpful to disaggregate claims of sustained yield for assessment purposes (Fig. 2). Although most of the data about the effects of sequential harvests are for timber, we believe the same situation applies to non-timber products, especially those for which individuals are harvested in their entirety (e.g. rattan palms).

In the more in-depth assessment of sustained yield proposed here, the degree to which volumetric yields are maintained from one harvest to the next is retained but only as one criterion, described by one axis in a trade-off pentagon (Fig. 2). Given the propensity for harvesters to 'high-grade' (i.e. to select the best individuals first), product quality typically declines with each successive harvest (e.g. increased prevalence of crooked, small, hollow, and heart-rotted trees); we capture this trend in another axis in the sustained yield pentagon. Included in this dimension of sustained yield would be changes in wood densities and working properties such as between old-growth timber and that of regenerating stands of fast-growing trees. The similar tendency to harvest the biggest trees first is represented by an axis that reflects the

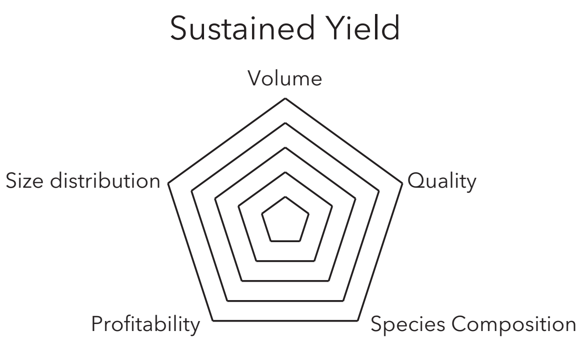

Figure 2 Sustained yield assessed by five criteria. The concentric lines refer to % values of the indicators relative to primary forest (outermost line).

size-class frequency distributions of stems in the post-harvest forest, at a point in time. Given the disproportionately large contributions of very large trees to forest structure, biodiversity maintenance, ecosystem processes, and both population and carbon dynamics (e.g. Lindenmayer et al., 2012; Slik et al., 2013; Sist et al., 2014; Thompson et al., 2014; Kohl et al., 2017), their retention in managed forests is of substantial environmental importance. This importance is reflected in the Brazilian forestry regulation that requires retention of at least 15% of all large trees or the three largest trees per 100 ha harvest block (CONAMA, Resolution no. 1 of 2015; Vidal et al., 2020). Also included in the yield component of SFM is an axis that reflects the sustainability of financial profits. As with all the trade-off polygons, a composite value for sustained yield is calculated as the sum of these five components. Other considerations might, of course, be included and the components might be differentially weighted, but the overall approach provides some clarity about the assessment of claims of sustained yields.

6 Land-use types in SFM

To secure the benefits of landscape-level assessments of SFM, landscapes need to be subdivided into different land-use categories. For a theoretical forested landscape in the tropics, we here consider the following six land-use types:

- 1 Protected Areas;
- 2A Natural Forest Management with Selective Harvests of Timber and Non-Timber Forest Products;
- 2B Natural Forest Management with Silvicultural Treatments After Selective Logging;
- 3 Tree Plantations;
- 4 Community Forests; and
- 5 Forest Restoration Areas.

6.1 Protected areas

Protected areas are designated to maintain ecosystem processes, protect biodiversity and especially low-density species, maintain high carbon stocks, and provide ecosystem services to surrounding landscapes and people. For an example of their importance in a landscape context, one of the few studies that found convergence of conditions in managed forests to those in primary forest noted the importance of associated large protected areas to species and ecosystem process recovery (Norden et al., 2009).

Intact ecosystems in protected primary forests also provide benchmarks against which to measure SFM. Individual protected areas are often not large enough on their own to protect wide-ranging or low-density species. At the landscape scale, proper management of surrounding buffer areas can provide the connectivity among protected areas that is required for persistence of some species (e.g. Hodgson et al., 2011). Like SFM in general, the effectiveness of tropical protected areas is very much a function of governance, stakeholder agreement, level of staff training and commitment, and sufficient funding (Bruner et al., 2001). Assessments of the extent to which protected areas deliver the expected or hoped-for values are made challenging by the tremendous variation in the degree to which protected areas are essentially abandoned or are actively protected with controls on access and resource exploitation.

Unmanaged or primary forests (i.e. those with no visible signs of human intrusion; FAO, 2018) are declining rapidly, especially those that are large. Potapov et al. (2017) reported that globally, the area of intact forest, defined as areas of >500 km^2 with no roads, declined by 7.2% between 2000 and 2013; such areas are already absent in many tropical countries. It is abundantly clear that the stocks of carbon and biodiversity in large primary forests exceed those in forested lands subjected to uses other than protection (e.g. Barlow et al., 2007; Luyssaert et al., 2008; Pan et al., 2011; Edwards et al., 2014; Watson et al., 2018). Many large-bodied and/or heavily exploited tropical animal species prefer intact forests, including the Asian elephant (*Elephas maximus*), African forest elephant (*Loxodonta cyclotis*), tiger (*Panthera tigris tigris*), and harpy eagle (*Harpia harpyja*) (Kinnaird et al., 2003; Barnes et al., 1991; Barlow et al., 2011; Birdlife International, 2016, but see Roopsind et al., 2017). Complicating the discussion of the conservation value of intact forest is the research demonstrating that up to 94% of the area in blocks of forest designated for selective logging remained intact due to the absence of commercial timber, adverse conditions, or poor planning and inadequate supervision (mean = 69%; Putz et al., 2019).

The vast majority of biodiversity exists outside protected areas and the ranges of many species protected partially or predominantly inside parks extend well beyond park boundaries. Hence, protected areas can rarely maintain viable populations of low-density species. Furthermore, in many areas of the world, protected areas either do not exist (Rodrigues et al., 2004) or are unmanaged and subject to illegal activities including poaching and logging (Loveridge et al., 2007; Wittemyer et al., 2008). Laurance et al. (2012) suggested that at least half of Earth's protected areas are failing to sustain their biodiversity. While the rate of loss of intact forests has generally been higher outside areas designated for protection, intact forest areas inside parks nevertheless often decline. For example, Virunga National Park in the Republic of Congo lost 3.3% of its forest cover in just over a decade (Potapov et al., 2017). Overall, the extensively managed forests that serve as buffers for protected areas are essential to sustain tropical biodiversity.

6.2 Natural forest management with selective harvests of timber and non-timber forest products

Natural forests managed for timber and non-timber forest products, which in the tropics typically involves selective harvests, maintain many values when managed properly, as detailed below (Fig. 3). Unfortunately, despite substantial expenditures of time, money, and effort, yields from the harvested species are seldom maintained even when governmental regulations are scrupulously followed (Putz et al., 2012; Vidal et al., 2020). Generally the most valuable timber species are harvested first, followed successively by each of the less valuable ones in subsequent harvests, often referred to as 'logging down the value chain' (e.g. Schaafsma et al., 2013). Furthermore, most government agencies and non-governmental certification bodies (e.g. the Forest Stewardship Council) lack the wherewithal to determine if yields from individual species or even entire forests are maintained (Romero and Putz, 2018).

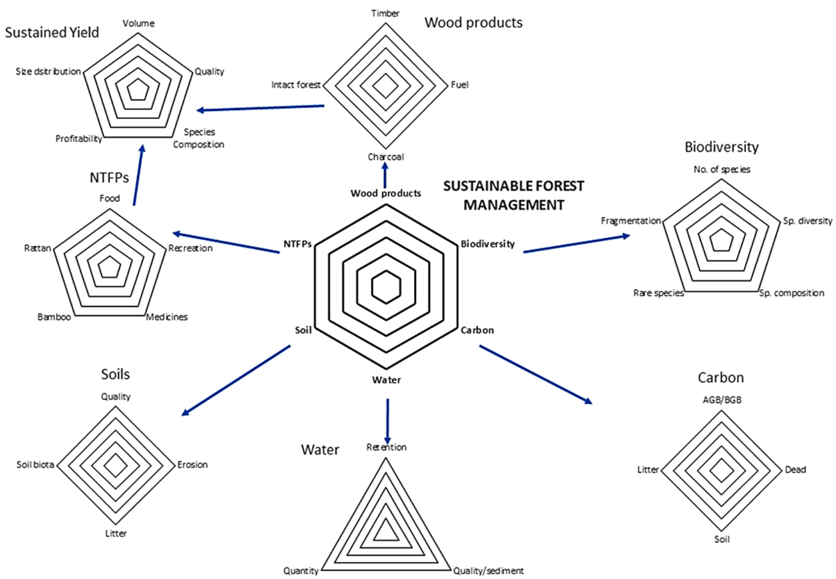

Figure 3 A suggested approach to disaggregation of SFM into its component values with emphasis on biophysical attributes. Note that the criteria and indicators for evaluation of sustained yield pertains to both timber and non-timber forest products. The concentric lines refer to % values of the indicators relative to a primary forest baseline.

Under current regulations in most tropical countries, timber stocks do not regain primary-forest volumes by the end of each officially designated minimum cutting cycle. After conventional timber harvests in Amazonian Brazil, for example, timber volumes take >60 years to recover, not the 25–30 years allowed by law (reviewed by Vidal et al., 2020). Some studies report eventual convergence of logged forests to primary forest conditions (e.g. Norden et al., 2009), while others suggest convergence will not occur (de Avila et al., 2015). A meta-analysis of studies on yield recovery based on >100 publications (Putz et al., 2012) revealed substantial variability, but concluded that timber yields declined by about 46% from the first harvest to the second harvest. That study also reported that, on average, 76% of carbon is retained in forests logged once, and that 85–100% of species of mammals, birds, invertebrates, and plants remain after logging, although long-term persistence is not assured. It is important to note, however, that such studies only report on a few taxa and do not consider all ecosystem functions, especially those delivered by complexes of co-evolved species. Furthermore, the studied forests were not selected at random and were likely representative of the best management underway when the studies were conducted. Finally, although many tropical forests are being logged for the second or third time, most of the reviewed studies focused on timber harvests from primary forest. One consistent message is that, despite the conservation potential of extensive selective logging, SFM is currently jeopardized in much of the tropics by poor logging practices (e.g. Ellis et al., 2019) and premature re-entry logging of previously harvested stands (Sasaki et al., 2016).

The value of extensive production forests for biodiversity and ecosystem services varies with logging intensity (e.g. Burivalova et al., 2014; Franca et al., 2017), logging practices (e.g. Pinard and Putz, 1996; Vidal et al., 2016), but particularly with post-harvest secondary effects including deforestation, poaching, and illegal logging (Michalski and Peres, 2013; Zimmerman and Kormos, 2012; Specht et al., 2015). It is clear that if logging and access are controlled, these secondary effects can be avoided and extensive areas of selectively logged forest will maintain considerable conservation value (Edwards et al., 2011; Putz et al., 2012; Edwards et al., 2014; Lewis et al., 2015; Roopsind et al., 2018, but see Laufer et al., 2013). While extensively managed forests can support much biodiversity, selective logging may nevertheless have substantial deleterious impacts on populations of high-value tree species and associated fauna (e.g. Fisher et al., 2011), partially due to losses of seed sources and dispersal agents. These populations can often be recovered only through carefully managed planting (see 2B).

The ITTO suggested that at least 500 million ha of tropical forest were degraded by 2002; we are not aware of any more recent global estimates of degradation except specifically for carbon (e.g. Baccini et al., 2017). While global attention has swung to reforestation of tree-free areas, restoration of these degraded forests should be a priority. Restoration of degraded forests implies increasing forest resilience, reducing the probability of successful invasions by exotic species, emulating natural processes in silvicultural regimes, and especially avoiding continued degradation. For instance, managing to ensure resilience means maintaining natural species composition and the capacity of forest composition to change under natural circumstances, with only gradual shifts in structure and function (Thompson et al., 2009). Accomplishment of this objective requires that forests are not degraded to a tipping point beyond which the ecosystem state changes radically to a novel and potentially stable condition (e.g. closed forest to open forest). Forest degradation is a difficult concept, however, owing to different perceptions for values derived from the forest, but generally refers to the loss of goods and services (CPF, 2010; Vásquez-Grandón et al., 2018). Degradation becomes easier to measure when considered at the landscape scale by using criteria and indicators (Thompson et al., 2013), like those proposed herein for SFM (Fig. 3) For example, many observers consider plantation to be highly degraded forests or not forests at all (Putz and Redford, 2010), to distinguish them from natural forest. Where plantations replace forests, the environmental losses cannot be recovered, but where plantations are established in already deforested areas, make up only a small proportion of the landscape, are societally accepted, and reduce pressure on natural forests as sources of wood products, they may be acceptable on both environmental and economic grounds. There are also many ways that the deleterious impacts of plantations can be mitigated (e.g. Dudley, 2005), but, based on our observations, few of the recommended practices are ever implemented at industrial scales.

The prerequisite conditions for SFM include proper policy support and legal frameworks, sufficient worker training, uncontested land tenure, sufficient financial incentives, and effective enforcement of regulations (e.g. Nasi et al., 2011; Sabogal et al., 2013; ITTO, 2015, 2016). Lack of financial remuneration for the many environmental services provided by natural tropical forests is one reason for the low financial competitiveness of forest management compared to other land uses such as agriculture and cattle-ranching. To the extent that reduced-impact logging is synonymous with reduced-income logging, it is not reasonable to expect loggers to adopt improved harvesting practices out of enlightened self-interest (Putz et al., 2000). Payments for ecosystem services (PES) seem like a viable mechanism to promote SFM, but successful uses of this tool are scarce and the benefits are ephemeral and funding dependent. It is noteworthy that the 'Socio Bosque' PES program in Amazonian Ecuador reportedly promoted reductions in both deforestation and forest degradation (Mohebalian and Aguilar, 2018). Aside from financial incentives, strong enforcement is also essential to secure the benefits from the vast quantities of carbon that could be sequestered by improved forest management (e.g. Pinard and Putz, 1996; Vidal et al., 2016; Ellis et al., 2019). Halting land grabs and poaching of both wildlife and timber is rendered especially difficult after logging roads improve access and thereby the profits from illegal activities, but the benefits from enforcement are substantial (e.g. Roopsind et al., 2018). Carbon crediting from improved management is sanctioned by the United Nations' Reduced Emissions from Deforestation and Forest Degradation (REDD+) program, but funds remain scarce for promoting the transition from forest exploitation to forest management.

6.3 Natural forest management with silvicultural treatments after selective logging

The principal intervention in tropical forests designated for timber production is selective logging. If properly conducted, selective logging can be considered as a silvicultural technique insofar as it can promote the regeneration and growth of commercial species (e.g. Vidal et al., 2016). Unfortunately, despite decades of promotion of reduced-impact logging (RIL) including millions of dollars spent on RIL policy development and training, most logging still more closely represents timber mining than timber stand management (Ellis et al., 2019). Foresters concerned about logging-induced reductions in timber yields and timber quality, as well as the sequential extirpation of commercial species with each harvest have long prescribed silvicultural interventions. Research firmly establishes the silvicultural benefits of these treatments, but apparently due to insufficient motivation, they are seldom applied outside of research plots.

The toolbox for tropical silviculture includes interventions that range from variations on felling regimes (e.g. strip clear-cuts and group selection harvests), pre-felling treatments such as the cutting of lianas on trees to be felled, as well as the planning of extraction pathways and the marking of trees for directional felling (i.e. RIL). Post-logging silvicultural treatments include liana cutting on future crop trees (FCTs), liberation of FCTs from arboreal competitors, mechanical scarification of felling gaps to promote regeneration, culling of non-commercial trees, and enrichment planting of commercial species along cleared lines or in felling gaps.

High harvest intensities in natural forests typically remove the valuable, mostly shade-tolerant hardwoods, while it damages young recruits, which leads to non-recovery of these species (Van Gardingen et al., 2003; Anitha et al., 2010). Often even low-intensity harvesting can deplete the valuable species (Peña-Claros et al., 2008; Sebben et al., 2008; Schulze et al., 2008a,b, Kukkonen and Hohnwald, 2009), hence the necessity of assisting natural regeneration of over-exploited species, such as with enrichment planting of mahoganies (*Swetenia* spp.), rosewoods (*Dalbergia* spp.), ipê (*Tabebuia* spp.), and cedar (*Cedrela* spp.). The silvicultural effectiveness of each of these treatments is supported by research, but even for these high-value species, few are applied outside of research areas (but see Navarro-Martínez et al., 2017). Historically, more broad-scale employment of silviculture occurred, such as the application of the Malayan Uniform System in Malaysia, but those treated stands were mostly converted to oil palm plantations and silvicultural treatments were discontinued in the forests that remained forest.

For reasons that are not completely clear but that include improved governance and increased recognition of current and pending shortfalls of timber supplies, there are now a few commercial-scale examples of silvicultural intensification of natural forest management (Puettmann et al., 2015). For one, liana cutting on future crop trees (i.e. trees smaller than the minimum cutting diameter that are expected to mature by the end of the cutting cycle) is reportedly more the rule than the exception in a major logging concession in Belize (Mills et al., 2019). Another example is in Indonesia where at least one logging concession carries out large-scale enrichment planting along cleared lines through twice-logged forest (Ruslandi et al., 2017a).

6.4 Tree plantations

While the area of primary forest in the tropics declines, the area of tree plantations (which we do not consider forest *sensu latu* because of the limited species composition, rapid turnover, and usually single objective for wood fiber; Putz and Redford, 2010) increased dramatically over the past two decades. Planted forests now cover >278 million hectares, increasing from 4% to 7% of the reported total tree-covered area between 2010 and 2015 (Payn et al., 2015). Here, we differentiate assisted natural regeneration of native species in extensively managed natural forests, involving directed post-harvest silvicultural treatments, from intensive plantation forestry. Among the plantations are those under short-term (i.e. fastwood) and longer-term cutting cycles, but most often involve a single species used for utility grade timber, chips and fibers, or fuel (Brockerhoff et al., 2008) and support limited biodiversity. Commonly planted are species of the genera *Acacia*, *Eucalyptus*, and *Pinus*. Our justification for this distinction is that forests with assisted natural regeneration also contain many naturally recruited trees and the planted species would not have recovered naturally without the intervention (Thompson et al., 2014; Ruslandi et al., 2017b). We note that although the majority of plantations we have observed in the tropics are appropriately considered 'green deserts', there is plenty of research demonstrating the benefits of biodiversity-enhancing design and management practices such as mixed species plantings and retention of natural forest along riparian corridors (e.g. Dudley, 2005; Paquette and Messier, 2010; Liu et al., 2018). We also note that the assumption that plantations take the pressure of natural forests (e.g. Sedjo and Botkin, 1997) seems supported under some conditions, but remains to be rigorously tested.

6.5 Community forests

We include community forests as a separate land-use category because, although they may be subjected to many different management practices, we assume that SFM is a principal goal. In some cases, however, community forests can also be fully protected, used for ecotourism purposes, or managed by commercial contractors. In any case, many tropical countries are trying to reduce the deleterious impacts of concession forestry and to redress prior local communities' rights violations by assigning management responsibilities to these constituencies. The ethical appropriateness of retuning land to traditional owners notwithstanding, the impacts of community forestry range from relative successes insofar as management improved to failures with rates of deforestation that do not differ from other forests (Bowler et al., 2012; Santika et al., 2017). These failures reportedly resulted from a combination of a lack of training, insufficient funding, disinterest by government in reviewing progress, lack of agreement and coherence of action among community members, land-grabbing, and various illegal/informal activities. In contrast, Porter-Bolland et al. (2012) found that 33 community forests generally had lower rates of deforestation than 40 protected areas, but the mechanisms responsible for this environmental benefit could not be specified due to lack of clear counterfactuals (i.e. what would have happened in the absence of community land tenure). Furthermore, if over time communities accumulate capital and increase in market integration, land-use practices may intensify especially if they are allowed to sell or lease their land.

6.6 Forest restoration areas

Given global attention to the potential benefits of forest restoration (e.g. Griscom et al., 2017), we include this land use, but with some misgivings. One cause of concern is that many of its proponents fail to distinguish between plantations and forests, so that the result of reforestation interventions can differ fundamentally. It is also often unclear whether forest products can be harvested from the reforested areas. Some projects do aim for full ecological restoration, which means recovering the species diversity and composition of primary forest, but it is not clear that this ambitious goal is attainable. Finally, differences in starting conditions affect the outcomes of restoration interventions. For example, the likely outcomes of forest restoration differ between areas that were deforested and then plowed, planted, fertilized, or overgrazed from those that suffered only a clear-cut. Spatial scale and landscape settings also matter, especially if propagules for regeneration need to be dispersed to great distances. In any case, restoration efforts are generally new and of limited overall consequence for the landscape, at least by the year 2020. Worst of all, when naturally tree-poor savannas and grasslands are afforested, the biodiversity consequences are grave (e.g. Veldman et al., 2015).

7 Challenges for SFM in the tropics

To various extents, tropical landscapes present a special case for the implementation of SFM that reinforces the need for disaggregated approaches to assessment, like the one proposed herein. First of all, many forested areas in the tropics are characterized by weak governance, contested land ownership, poverty, large numbers of forest-dependent people, rapid rates of exploitation and forest conversion, modest-to-high opportunity costs of forest retention, and/or political conflicts.

Considerations of sustainability are further complicated by the fact that far more wood is taken for fuel than for timber. For example, Sprecht et al. (2015) found that annual demand for fuelwood by 210 municipalities in Amazonian Brazil was about 300 thousand tons, which they noted would require the clearing of 1200–2100 ha of forest.

Efforts at SFM often face challenges related to the legacies of former interventions: many of the forests exploited for timber today were previously logged, either legally or illegally, but virtually always with little regard for the future. Even in forests with no recent history of exploitation, given high species diversity, tropical trees that produce commercial timber are generally scarce and patchily distributed, which can lead to their rapid commercial extirpation. Many such species, including rosewoods and mahoganies, are now listed by CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora). Growth and regeneration rates are such that, to maintain viable populations of many commercial timber species, large areas and low-harvest rates are required, at least unless silvicultural interventions, such as liberation thinning around future crop trees and enrichment planting, are applied. Complicating matters further is the fact that some of these species require interior forest conditions and are light, moisture, and thermally sensitive. Furthermore, many tree species depend on co-evolved relationships for pollination, seed dispersal, and nutrient acquisition (e.g. Lewis, 2009; Campos-Arceiz and Blake, 2011).

Given the widespread conversion of lowland forest on gentle terrain to more intensive land uses, natural forest management is increasingly relegated to lands less suitable for industrial agriculture or plantation forestry due to remoteness, nutrient impoverishment, steepness, or poor drainage (e.g. Putz et al., 2018). Remoteness generally increases the likelihood of governance failures while the adverse site conditions render forest lands more susceptible to soil damage and erosion. With the increased intensities of rainfall due to global climate change, soil compaction and erosion, including landslides, also increase, especially on steep slopes (Lele, 2009). The connections between these abuses in the hinterlands and downstream flooding need to be emphasized to spur improved enforcement of land-use regulations even in remote areas.

We recommend that to foster retention of the renewable natural resources and ecosystem services provided by tropical forests, the various values of forests should be disaggregated, considered individually, and then combined in an explicit manner to provide an overall evaluation of the sustainability of forest use at landscape scales. Increased transparency about the trade-offs associated with management decisions at stand up to landscape scales will at least inform debates. To increase the likelihood of political and behavioral changes that lead to improved fates of tropical forests, we advocate for the collaborative construction of detailed and place-specific theories-of-change in which the assumptions are enumerated, relevant actors are identified, their motivations and interactions are captured, and the contexts in which decisions are made are elucidated.

8 Ways forward

One major impediment to sustainable forest management at landscape scales is lack of appropriately trained foresters with the political wherewithal to have their voices heard. This deficiency increases as the number of forestry schools declines almost everywhere partially due to the demonization of tree cutting, despite the canonization of tree planting. Although people will always need wood and wood products, support for improved forest management by international organizations is likewise weak. While the abuses tropical forests suffer from timber mining operations are scrutinized by researchers, few and mostly naïve solutions are offered due to inattention to the relevant factors and constraints. Forest owners, be they governments or communities, also need to forgo some short-term profits so that the renewable natural resources in tropical forests have the chance to be renewed. Perhaps recognition that forest landscapes can be managed sustainably, without denying the many trade-offs, may help efforts to recruit motivated young people into the vibrant field of forestry.

9 References

Anitha, K., Joseph, S., Chandran, R. J., Ramasamy, E. V. and Prasad, S. N. 2010. Tree species diversity and community composition in a human-dominated tropical forest of western Ghats biodiversity hotspot India. *Ecological Complexity* 7(2), 217–24. doi:10.1016/j.ecocom.2010.02.005.

Baccini, A., Walker, W., Carvalho, L., Farina, M., Sulla-Menashe, D. and Houghton, R. A. 2017. Tropical forests are a net carbon source based on aboveground measurements of gain and loss. *Science* 358(6360), 230–4. doi:10.1126/science.aam5962.

Barlow, J., Gardner, T. A., Araujo, I. S., Avila-Pires, T. C., Bonaldo, A. B., Costa, J. E., Esposito, M. C., Ferreira, L. V., Hawes, J., Hernandez, M. I. M., Hoogmoed, M. S., Leite, R. N., Lo-Man-Hung, N. F., Malcolm, J. R., Martins, M. B., Mestre, L. A. M., Miranda-Santos, R., Nunes-Gutjahr, A. L., Overal, W. L., Parry, L., Peters, S. L., Ribeiro-Junior, M. A., da Silva, M. N. F., da Silva Motta, C. and Peres, C. A. 2007. Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. *Proceedings of the National Academy of Sciences of the United States of America* 104(47), 18555–60. doi:10.1073/pnas.0703333104.

Barlow, A. C. D., Smith, J. L. D., Ahmad, I. U., Hossain, A. N. M., Rahman, M. and Howlader, A. 2011. Female tiger Panthera tigris home range size in the Bangladesh Sundarbans: the value of this mangrove ecosystem for the species' conservation. *Oryx* 45(1), 125–8. doi:10.1017/S0030605310001456.

Barnes, R. F. W., Barnes, K. L., Alers, M. P. T. and Blom, A. 1991. Man determines the distribution of elephants in the rain forests of northeastern Gabon. *African Journal of Ecology* 29(1), 54–63. doi:10.1111/j.1365-2028.1991.tb00820.x.

BirdLife International. 2016. Harpia harpyja. The IUCN RED List of Threatened Species 2016 e.T22695998A93537912.

Boscolo, M. 2000. *Strategies for Multiple Use Management of Tropical Forests: An Assessment of Alternative Options.* CID Working Paper Series.

Bowler, D. E., Buyung-Ali, L. M., Healey, J. R., Jones, J. P. G., Knight, T. M. and Pullin, A. S. 2012. Does community forest management provide global environmental benefits and improve local welfare? *Frontiers in Ecology and the Environment* 10(1), 29–36. doi:10.1890/110040.

Brockerhoff, E. G., Jactel, H., Parrotta, J. A., Quine, C. P. and Sayer, J. 2008. Biodiversity and planted forests–oxymoron or opportunity? *Biodiversity and Conservation* 17(5), 925–51. doi:10.1007/s10531-008-9380-x.

Brundtland, G. H. 1987. *Our Common Future*. Report of the World Commission on Environment and Development. Oxford University Press, Oxford.

Bruner, A. G., Gullison, R. E., Rice, R. E. and Da Fonseca, G. A. 2001. Effectiveness of parks in protecting tropical biodiversity. *Science* 291(5501), 125–8. doi:10.1126/science.291.5501.125.

Burivalova, Z., Şekercioğlu, C. H. and Koh, L. P. 2014. Thresholds of logging intensity to maintain tropical forest biodiversity. *Current Biology* 24(16), 1893–8. doi:10.1016/j.cub.2014.06.065.

Campos-Arceiz, A. and Blake, S. 2011. Mega-gardeners of the forest–the role of elephants in seed dispersal. *Acta Oecologica* 37(6), 542–53. doi:10.1016/j.actao.2011.01.014.

CPF Collaborative Partnership on Forests. 2010. *Measuring Forest Degradation*. Available at: http://www.fao.org/3/i1802e/i1802e00.pdf.

de Avila, A. L., Ruschel, A. R., de Carvalho, J. O. P., Mazzei, L., Silva, J. N. M., do Carmo Lopes, M. M., Araujo, M. M., Dormann, C. F. and Bauhus, J. 2015. Medium-term dynamics of tree species composition in response to silvicultural intervention intensities in a tropical rain forest. *Biological Conservation* 191, 577–86. doi:10.1016/j.biocon.2015.08.004.

Dudley, N. 2005. Best practices for industrial plantations. In: Mansouran, S., Vallauri, D. and Dudley, N. (Eds), *Forest Restoration in Landscapes: Beyond Planting Trees*. Springer Science, New York, pp. 379–97.

Edwards, D. P., Larsen, T. H., Docherty, T. D. S., Ansell, F. A., Hsu, W. W., Derhé, M. A., Hamer, K. C. and Wilcove, D. S. 2011. Degraded lands worth protecting: the biological importance of Southeast Asia's repeatedly logged forests. *Proceedings of the Royal Society B* 278(1702), 82–90. doi:10.1098/rspb.2010.1062.

Edwards, D. P., Gilroy, J. J., Woodcock, P., Edwards, F. A., Larsen, T. H., Andrews, D. J. R., Derhé, M. A., Docherty, T. D. S., Hsu, W. W., Mitchell, S. L., Ota, T., Williams, L. J., Laurance, W. F., Hamer, K. C. and Wilcove, D. S. 2014. Land-sharing versus land-sparing logging: reconciling timber extraction with biodiversity conservation. *Global Change Biology* 20(1), 183–91. doi:10.1111/gcb.12353.

Ellis, P. W., Gopalakrishna, T., Goodman, R. C., Roopsind, A., Griscom, B., Umunay, P. M., Zalman, J., Ellis, E., Mo, K., Gregoire, T. G. and Putz, F. E. 2019. Climate-effective reduced-impact logging (RIL-C) can halve selective logging carbon emissions in tropical forests. *Forest Ecology and Management* 438, 255–66.

FAO. 2018. *Terms and Definitions FRA 2020*. Forest Resources Assessment Working Paper 188. Rome, Italy.

Fisher, B., Edwards, D. P., Larsen, T. H., Ansell, F. A., Hsu, W. W., Roberts, C. S. and Wilcove, D. S. 2011. Cost-effective conservation: calculating biodiversity and logging trade-offs in Southeast Asia. *Conservation Letters* 4(6), 443–50. doi:10.1111/j.1755-263X.2011.00198.x.

Franca, F. M., Frazão, F. S., Koraski, V., Louzada, J. and Barlow, J. 2017. Identifying thresholds of logging intensity on dung beetle communities to improve the sustainable management of Amazonian tropical forests. *Biological Conservation* 216, 115–22. doi:10.1016/j.biocon.2017.10.014.

Griscom, B., Adams, J., Ellis, P., Houghton, R. A., Lomax, G., Miteva, D. A., Schlesinger, W. H., Shoch, D., Woodbury, P., Zganjar, C., Blackman, A., Campari, J., Conant, R. T., Delgado, C., Elias, P., Hamsik, M., Kiesecker, J., Landis, E., Polasky, S., Putz, F. E., Sanderman, J., Siikamäki, J., Silvius, M., Wollenberg, L. and Fargione, J. 2017. Natural pathways to climate mitigation. *Proceedings of the National Academy of Sciences (USA)* 114, 11645–50.

Griscom, B. W., Burivalova, Z., Ellis, P. W., Halperin, J., Marthinus, D., Runting, R., Ruslandi, B., Wahyudi, B. and Putz, F. E. 2019. Reduced-impact logging in Borneo to minimize carbon emissions while preserving sensitive habitats and maintaining timber yields. *Forest Ecology and Management* 438, 176–85.

Hodgson, J. A., Moilanen, A., Wintle, B. A., Thomas, C. D. 2011. Habitat area, quality and connectivity: striking the balance for efficient conservation. *Journal of Applied Ecology* 48(1), 148–52. doi:10.1111/j.1365-2664.2010.01919.x.

ITTO. 2015. *Voluntary Guidelines for the Sustainable Management of Natural Tropical Forests.* ITTO Policy Development Series No. 20, Yokohama, Japan.

ITTO. 2016. *Criteria and Indicators for the Sustainable Management of Tropical Forests.* ITTO Policy Development Series No. 21, Yokohama, Japan.

Kinnaird, M. F., Sanderson, E. W., O'Brien, T. G., Wibisono, H. T. and Woolmer, G. 2003. Deforestation trends in a tropical landscape and implications for endangered large mammals. *Conservation Biology* 17(1), 245–57. doi:10.1046/j.1523-1739.2003.02040.x.

Kohl, M., Neupane, P. R.and Lotfiomran, N. 2017. The impact of tree age on biomass growth and carbon accumulation capacity: a retrospective analysis using tree ring data of three tropical tree species grown in natural forests of Suriname. *PLoS ONE* 12(8), e0181187. doi:10.1371/journal.pone.0181187.

Kukkonen, M. and Hohnwald, S. 2009. Comparing floristic composition in treefall gaps of certified conventionally managed and natural forests of northern Honduras. *Annals of Forest Science* 66(8), 809. doi:10.1051/forest/2009070.

Laufer, J., Michalski, F. and Peres, C. A. 2013. Assessing sampling biases in logging impact studies in tropical forests. *Tropical Conservation Science* 6, 16–34.

Laurance, W. F., Useche, D. C., Rendeiro, J., Kalka, M., Bradshaw, C. J., Sloan, S. P., Laurance, S. G., Campbell, M., Abernethy, K., Alvarez, P. and Arroyo-Rodriguez, V. 2012. Averting biodiversity collapse in tropical forest protected areas. *Nature* 489, 290–4.

Lele, S. 2009. Watershed services of tropical forests: from hydrology to economic valuation to integrated analysis. *Current Opinion in Environmental Sustainability* 1(2), 148–55. doi:10.1016/j.cosust.2009.10.007.

Lewis, O. T. 2009. Biodiversity change and ecosystem function in tropical forests. *Basic and Applied Ecology* 10(2), 97–102. doi:10.1016/j.baae.2008.08.010.

Lewis, S. L., Edwards, D. P. and Galbraith, D. 2015. Increasing human dominance of tropical forests. *Science* 349(6250), 827–32. doi:10.1126/science.aaa9932.

Lindenmayer, D. B., Laurance, W. F. and Franklin, J. F. 2012. Global decline in large old trees. *Science* 338(6112), 1305–6. doi:10.1126/science.1231070.

Liu, C. L. C., Kuchma, O. and Krutovsky, K. V. 2018. Mixed-species versus monocultures in plantation forestry: development, benefits, ecosystem services and perspectives for the future. *Global Ecology and Conservation* 15, e00419. doi:10.1016/j.gecco.2018.e00419.

Loveridge, A. J., Searle, A. W., Murindagomo, F. and Macdonald, D. W. 2007. The impact of sport-hunting on the population dynamics of an African lion population in a protected area. *Biological Conservation* 134(4), 548–58. doi:10.1016/j.biocon.2006.09.010.

Luckert, M. and Williamson, T. 2005. Should sustained yield be part of sustainable forest management? *Canadian Journal of Forest Research* 35(2), 356–64. doi:10.1139/x04-172.

Luyssaert, S., Schulze, E. D., Börner, A., Knohl, A., Hessenmöller, D., Law, B. E., Ciais, P. and Grace, J. 2008. Old growth forests as global carbon sinks. *Nature* 455(7210), 213–5. doi:10.1038/nature07276.

Messier, C., Tittler, R., Kneeshaw, D. D., Gélinas, N., Paquette, A., Berninger, K., Rheault, H., Meek, P. and Beaulieu, N. 2009. TRIAD zoning in Quebec: experiences and results after 5 years. *The Forestry Chronicle* 85(6), 885–96. doi:10.5558/tfc85885-6.

Michalski, F. and Peres, C. A. 2013. Biodiversity depends on logging recovery time. *Science* 339(6127), 1521–2. doi:10.1126/science.339.6127.1521-b.

Mills, D. J., Bohlman, S. A., Putz, F. E. and Andreu, M. G. 2019. Liberation of future crop trees from lianas in Belize: completeness, costs, and timber-yield benefits. *Forest Ecology and Management* 439, 97–104. doi:10.1016/j.foreco.2019.02.023.

Mohebalian, P. M. and Aguilar, F. X. 2018. Beneath the canopy: tropical forests enrolled in conservation payments reveal evidence of less degradation. *Ecological Economics* 143, 64–73. doi:10.1016/j.ecolecon.2017.06.038.

Nasi, R. and Frost, P. G. H. 2009. Sustainable forest management in the tropics: is everything in order but the patient still dying? *Ecology and Society* 14(2), 40. Available at: www.ecologyandsociety.org/vol14/iss2/art40.

Nasi, R., Putz, F. E., Pacheco, P., Wunder, S. and Anta, S. 2011. Sustainable forest management and carbon in tropical Latin America: the case for REDD+. *Forests* 2(1), 200–17. doi:10.3390/f2010200.

Navarro-Martínez, A., Palmas-Perez, A. S., Ellis, E. A., Blanco Reyes, P., Vargas Godínez, C., Iuit Jiménez, A. C., Hernández Gómez, I., Ellis, P., Álvarez Ugalde, A., Carrera Quirino, Y. G., Armenta Montero, S. and Putz, F. E. 2017. Remnant trees in enrichment planted gaps Quintana Roo, Mexico: reasons for retention and effects on planted seedling growth. *Forests* 8, 272. doi:10.3390/f8080272.

Norden, N., Chazdon, R. L., Chao, A., Jiang, Y. H. and Vílchez-Alvarado, B. 2009. Resilience of tropical rain forests: tree community reassembly in secondary forests. *Ecology Letters* 12(5), 385–94. doi:10.1111/j.1461-0248.2009.01292.x.

Pan, Y., Birdsey, R. A., Fang, J., Houghton, R., Kauppi, P. E., Kurz, W. A., Phillips, O. L., Shvidenko, A., Lewis, S. L., Canadell, J. G., Ciais, P., Jackson, R. B., Pacala, S. W., McGuire, A. D., Piao, S., Rautiainen, A., Sitch, S. and Hayes, D. 2011. A large and persistent carbon sink in the world's forests. *Science* 333(6045), 988–93. doi:10.1126/science.1201609.

Paquette, A. and Messier, C. 2010. The role of plantations in managing the world's forests in the Anthropocene. *Frontiers in Ecology and the Environment* 8(1), 27–34. doi:10.1890/080116.

Payn, T., Carnus, J. M., Freer-Smith, P., Kimberley, M., Kollert, W., Liu, S., Orazio, C., Rodriguez, L., Silva, L. N. and Wingfield, M. J. 2015. Changes in planted forests and future global implications. *Forest Ecology and Management* 352, 57–67. doi:10.1016/j.foreco.2015.06.021.

Peña-Claros, M., Fredericksen, T. S., Alarcón, A., Blate, G. M., Choque, U., Leaño, C., Licona, J. C., Mostacedo, B., Pariona, W., Villegas, Z. and Putz, F. E. 2008. Beyond reduced-impact logging: silvicultural treatments to increase growth rates of tropical trees. *Forest Ecology and Management* 256(7), 1458–67. doi:10.1016/j.foreco.2007.11.013.

Phalan, B., Onial, M., Balmford, A. and Green, R. E. 2011. Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science* 333(6047), 1289–91. doi:10.1126/science.1208742.

Pinard, M. A. and Putz, F. E. 1996. Retaining forest biomass by reducing logging damage. *Biotropica* 28(3), 278–95. doi:10.2307/2389193.

Piponiot, C., Rödig, E., Putz, F. E., Rutishauser, E., Sist, P., Ascarrunz, N., Blanc, L.,Derroire, G., Descroix, L., Laurent, C. G., Marcelino, H. C., Honorio Coronado, E., Huth, A., Kanashiro, M., Licona, J. C. and Mazzei, L. Neves d'Oliveira, M., Peña-Claros, M., Rodney, K., Shenkin, A., Rodrigues de Souza, C., Vidal, E., West, T., Wortel, V. and Hérault, B. 2019. Can timber provision from Amazonian production forests be sustainable? *Environmental Research Letters* 14(6), 064014. Available at: https://iopscience.iop.org/article/10.1088/1748-9326/ab195e.

Porter-Bolland, L., Ellis, E. A., Guariguata, M. R., Ruiz-Mallén, I., Negrete-Yankelevich, S. and Reyes-García, V. 2012. Community managed forests and forest protected areas: an assessment of their conservation effectiveness across the tropics. *Forest Ecology and Management* 268, 6–17. doi:10.1016/j.foreco.2011.05.034.

Potapov, P., Hansen, M. C., Laestadius, L., Turubanova, S., Yaroshenko, A., Thies, C., Smith, W., Zhuravleva, I., Komarova, A., Minnemeyer, S. and Esipova, E. 2017. The last frontiers of wilderness: tracking loss of intact forest landscapes from 2000 to 2013. *Science Advances* 3(1), e1600821. doi:10.1126/sciadv.1600821.

Puettmann, K. J., Wilson, S. M., Baker, S. C., Donoso, P. J., Droessler, L., Armente, G., Harvey, B. D., Knoke, T., Lu, Y., Nocentini, S., Putz, F. E., Yoshida, T. and Bauhus, J. 2015. Silvicultural alternatives to conventional even-aged management—what limits global adoption? *Forest Ecosystems* 2(1), 8. doi:10.1186/s40663-015-0031-x.

Putz, F. E. 2018. Sustainable = good, better, or responsible. *Journal of Tropical Forest Science* 30(1), 1–8. doi:10.26525/jtfs2018.30.1.18.

Putz, F. E. and Redford, K. H. 2010. Tropical forest definitions, degradation, phase shifts, and further transitions. *Biotropica* 42(1), 10–20. doi:10.1111/j.1744-7429.2009.00567.x.

Putz, F. E. and Romero, C. 2014. Futures of tropical forests (*sensu lato*). *Biotropica* 46(4), 495–505. doi:10.1111/btp.12124.

Putz, F. E., Dykstra, D. P. and Heinrich, R. 2000. Why poor logging practices persist in the tropics. *Conservation Biology* 14(4), 951–6. doi:10.1046/j.1523-1739.2000.99137.x.

Putz, F. E., Zuidema, P. A., Synnott, T., Peña-Claros, M., Pinard, M. A., Sheil, D., Vanclay, J. K., Sist, P., Gourlet-Fleury, S., Griscom, B., Palmer, J. and Zagt, R. 2012. Sustaining conservation values in selectively logged tropical forests: the attained and the attainable. *Conservation Letters* 5(4), 296–303. doi:10.1111/j.1755-263X.2012.00242.x.

Putz, F. E., Ruslandi, P., Ellis, P. W. and Griscom, B. 2018. Topographic restrictions on land-use practices: consequences of different pixel sizes and data sources for natural forest management in the tropics. *Forest Ecology and Management* 422, 108–13.

Putz, F. E., Baker, T., Griscom, B. W., Gopalakrishna, T., Roopsind, A., Umunay, P. M., Zalman, J., Ellis, E. A., Ellis, P. W. and Ellis, P. W. 2019. Intact forest in selective logging landscapes in the tropics. *Frontiers in Forests and Global Change* 2, 30. doi:10.3389/ffgc.2019.00030.

Rodrigues, A. S., Akcakaya, H. R., Andelman, S. J., Bakarr, M. I., Boitani, L., Brooks, T. M., Chanson, J. S., Fishpool, L. D., Da Fonseca, G. A., Gaston, K. J. and Hoffmann, M. 2004. Global gap analysis: priority regions for expanding the global protected-area network. *BioScience* 54, 1092–100.

Romero, C. and Putz, F. E. 2018. Theory-of-change development for evaluation of Forest Stewardship Council certification of sustained timber yields from natural forests in Indonesia. *Forests* 9(9), 547. doi:10.3390/f9090547.

Roopsind, A., Caughlin, T. T., Sambhu, H., Fragosa, J. M. V. and Putz, F. E. 2017. Logging and indigenous hunting impacts on persistence of large Neotropical animals. *Biotropica* 49(4), 565–75. doi:10.1111/btp.12446.

Roopsind, A., Caughlin, T. T., van der Hout, P., Arets, E. and Putz, F. E. 2018. Trade-offs between carbon stocks and timber recovery in tropical forests are mediated by logging intensity. *Global Change Biology* 24(7), 2862–74. doi:10.1111/gcb.14155.

Runting, R. K., Ruslandi, R., Griscom, B. W., Struebig, M. J., Satar, M., Meijaard, E., Burivalova, Z., Cheyne, S. M., Deere, N. J., Game, E. T., Putz, F. E., Wells, J. A., Wilting, A., Acrenaz, M., Ellis, P., Khan, F. A. A., Leavitt, S. M., Marshall, A. J., Possingham, H. P., Watson, J. E. M. and Venter, O. 2019. Larger gains from improved management over sparing–sharing for tropical forests. *Nature Sustainability* 2(1), 53–61. doi:10.1038/s41893-018-0203-0.

Ruslandi, W., Cropper, W. P. and Putz, F. E. 2017a. Effects of silvicultural intensification on timber yields, carbon dynamics, and tree species composition in a dipterocarp forest in Kalimantan, Indonesia: an individual-tree-based model simulation. *Forest Ecology and Management* 390, 104–18. doi:10.1016/j.foreco.2017.01.019.

Ruslandi, C., Romero, C. and Putz, F. E. 2017b. Financial viability and carbon payment potential of large-scale silvicultural intensification in logged dipterocarp forest in Indonesia. *Forest Policy and Economics* 85, 95–102. doi:10.1016/j.forpol.2017.09.005.

Sabogal, C., Guariguata, M. R., Broadhead, J., Lescuyer, G., Savilaakso, S., Essoungou, N. and Sist, P. 2013. *Multiple-Use Forest Management in the Humid Tropics: Opportunities and Challenges for Sustainable Forest Management*. FAO Forestry Paper No. 173. Food and Agriculture Organization of the United Nations, Rome, and Center for International Forestry Research, Bogor, Indonesia.

Santika, T., Meijaard, E., Budiharta, S., Law, E. A., Kusworo, A., Hutabarat, J. A., Indrawan, T. P., Struebig, M., Raharjo, S., Huda, I., Andini, S., Ekaputri, A. D., Trison, S., Stigner, M. and Wilson, K. A. 2017. Community forest management in Indonesia: avoided deforestation in the context of anthropogenic and climate complexities. *Global Environmental Change* 46, 60–71. doi:10.1016/j.gloenvcha.2017.08.002.

Sasaki, N. and Putz, F. E. 2009. Critical need for new definitions of "forest" and "forest degradation" in global climate change agreements. *Conservation Letters* 2(5), 226–32. doi:10.1111/j.1755-263X.2009.00067.x.

Sasaki, N., Asner, G. P., Pan, Y., Knorr, W., Durst, P. B., Ma, H. O., Abe, I., Lowe, A. J., Koh, L. P. and Putz, F. E. 2016. Sustainable management of tropical forests can reduce carbon emissions and stabilize timber production. *Frontiers in Environmental Science* 4. doi:10.3389/fenvs.2016.00050.

Sayer, J. A., Margules, C., Boedhihartono, A. K., Sunderland, T., Langston, J. D., Reed, J., Riggs, R., Buck, L. E., Campbell, B. M., Kusters, K., Elliott, C., Minang, P. A., Dale, A., Purnomo, H., Stevenson, J. R., Gunarso, P. and Purnomo, A. 2016. Measuring the effectiveness of landscape approaches to conservation and development. *Sustainability Science* 12(3), 465–76. doi:10.1007/s11625-016-0415-z.

Schaafsma, M., Burgess, N. D., Swetnam, R., Ngaga, Y., Ngowi, S., Turner, K. and Treue, T. 2013. Tanzanian timber markets provide early warnings of logging down the timber chain. In: *15th Annual BIOECON Conference, Conservation and Development: Exploring Conflicts and Challenges*, Cambridge, UK, pp. 18–20.

Schulze, M., Vidal, E., Grogan, J., Zweed, J. and Zarin, D. 2005. Madeiras nobres em perigo. *Revista Ciência Hoje* 36, 66–9.

Schulze, M., Grogan, J., Landis, R. M. and Vidal, E. 2008a. How rare is too rare to harvest? *Forest Ecology and Management* 256(7), 1443–57. doi:10.1016/j.foreco.2008.02.051.

Schulze, M., Grogan, J., Uhl, C., Lentini, M. and Vidal, E. 2008b. Evaluating ipê (*Tabebuia*, Bignoniaceae) logging in Amazonia: sustainable management or catalyst for forest degradation? *Biological Conservation* 141(8), 2071–85. doi:10.1016/j.biocon.2008.06.003.

Sebben, A. M., Degen, B., Azevedo, V. C. R., Silva, M. B., de Lacerda, A. E. B., Ciampi, A. Y., Kanashiro, M., Carneiro, F. S., Thompson, I. and Loveless, M. D. 2008. Modelling the long-term impacts of selective logging on genetic diversity and demographic structure of four tropical tree species in the Amazon forest. *Forest Ecology and Management* 254(2), 335–49. doi:10.1016/j.foreco.2007.08.009.

Sedjo, R. A. and Botkin, D. 1997. Using forest plantations to spare natural forests. Environment: Science and Policy for Sustainable Development 39: 14–30.

Slik, J. W. F., Paoli, G., McGuire, K., Amaral, I., Barroso, J., Bastian, M., Blanc, L., Bongers, F., Boundja, P., Clark, C., Collins, M., Dauby, G., Ding, Y., Doucet, J., Eler, E., Ferreira, L., Forshed, O., Fredriksson, G., Gillet, J., Harris, D., Leal, M., Laumonier, Y., Malhi, Y., Mansor, A., Martin, E., Miyamoto, K., Araujo-Murakami, A., Nagamasu, H., Nilus, R., Nurtjahya, E., Oliveira, Á, Onrizal, O., Parada-Gutierrez, A., Permana, A., Poorter, L., Poulsen, J., Ramirez-Angulo, H., Reitsma, J., Rovero, F., Rozak, A., Sheil, D., Silva-Espejo, J., Silveira, M., Spironelo, W., ter Steege, H., Stevart, T., Navarro-Aguilar, G. E., Sunderland, T., Suzuki, E., Tang, J., Theilade, I., van der Heijden, G., van Valkenburg, J., Van Do, T., Vilanova, E., Vos, V., Wich, S., Wöll, H., Yoneda, T., Zang, R., Zhang, M. and Zweifel, N. 2013. Large trees drive forest aboveground biomass variation in moist lowland forests across the tropics. *Global Ecology and Biogeography* 22(12), 1261–71. doi:10.1111/geb.12092.

Sist, P., Mazzei, L., Blanc, L. and Rutishauser, E. 2014. Large trees as key elements of carbon storage and dynamics after selective logging in the Eastern Amazon. *Forest Ecology and Management* 318, 103–9. doi:10.1016/j.foreco.2014.01.005.

Solow, R. M. 1956. A contribution to the theory of economic growth. *The Quarterly Journal of Economics* 70(1), 65–94. doi:10.2307/1884513.

Sprecht, M. J., Pinto, S. R. P., Albuquerque, U. P., Tabarelli, M. and Melo, F. P. L. 2015. Burning biodiversity: fuelwood harvesting causes forest degradation in human-dominated tropical landscapes. *Global Ecology and Conservation* 3, 200–9. doi:10.1016/j.gecco.2014.12.002.

Stickler, C., Duchelle, A. E., Nepstad, D. and Ardila, J. P. 2018. Subnational jurisdictional approaches policy innovation and partnerships for change. In: Angelsen, A., Martius, C., De Sy, V., Duchelle, A. E., Larson, A. M. and Pham, T. T. (Eds), *Transforming REDD+: Lessons and New Directions*. CIFOR, Bogor, Indonesia.

Thompson, I. D., Mackey, B., McNulty, S. and Mosseler, A. 2009. *Forest Resilience, Biodiversity, and Climate Change*. A synthesis of the biodiversity/resilience/stability relationship in forest ecosystems. Secretariat of the Convention on Biological Diversity, Montreal. Technical Series no. 43, 67pp.

Thompson, I. D., Okabe, K., Parrotta, J. A., Brockerhoff, E., Jactel, H., Forrester, D. I. and Taki, H. 2014. Biodiversity and ecosystem services: lessons from nature to improve management of planted forests for REDD-plus. *Biodiversity and Conservation* 23(10), 2613–35. doi:10.1007/s10531-014-0736-0.

Thompson, I. D., Guariguata, M. R., Okabe, K., Bahamondez, C., Nasi, R., Heymell, V. and Sabogal, C. 2013. An operational framework for defining and monitoring forest degradation. *Ecology and Society* 18(2), 20. doi:10.5751/ES-05443-180220.

Van Gardingen, P. R., McLeish, M. J., Phililips, P. D., Fadilah, D., Tyrie, G. and Yasman, I. 2003. Financial and ecological analysis of management options for logged-over dipterocarp forests in Indonesian Borneo. *Forest Ecology and Management* 183(1–3), 1–29. doi:10.1016/S0378-1127(03)00097-5.

Vásquez-Grandón, A., Donoso, P. and Gerding, V. 2018. Forest degradation: when is a forest degraded? *Forests* 9(11), 726. doi:10.3390/f9110726.

Veldman, J. W., Overbeck, G. E., Negreiros, D., Mahy, G., Le Stradic, S., Fernandes, G. W., Durigan, G., Buisson, E., Putz, F. E. and Bond, W. J. 2015. Where tree planting and forest expansion are bad for biodiversity and ecosystem services. *BioScience* 65(10), 1011–8. doi:10.1093/biosci/biv118.

Vidal, E., West, T. A. P. and Putz, F. E. 2016. Recovery of biomass and merchantable timber volumes twenty years after conventional and reduced-impact logging in Amazonian Brazil. *Forest Ecology and Management* 376, 1–8. doi:10.1016/j.foreco.2016.06.003.

Vidal, E., West, T. A. P., Lentini, M. W., de Souza, S. E. X. F., Klauberg, C. and Waldhoff, P. 2020. *Sustainable Forest Management in the Brazilian Amazon*.

Vincent, J. R. and Binkley, C. S. 1993. Efficient multiple-use forestry may require land-use specialization. *Land Economics* 69(4), 370. doi:10.2307/3146454.

Watson, J. E. M., Evans, T., Venter, O., Williams, B., Tulloch, A., Stewart, C., Thompson, I., Ray, J. C., Murray, K., Salazar, A., McAlpine, C., Potapov, P., Walston, J., Robinson, J. G., Painter, M., Wilkie, D., Filardi, C., Laurance, W. F., Houghton, R. A., Maxwell, S., Grantham, H., Samper, C., Wang, S., Laestadius, L., Runting, R. K., Silva-Chávez, G. A., Ervin, J. and Lindenmayer, D. 2018. The exceptional value of intact forest ecosystems. *Nature Ecology and Evolution* 2(4), 599–610. doi:10.1038/s41559-018-0490-x.

Wiersum, K. F. 1995. 200 years of sustainability in forestry: lessons from history. *Environmental Management* 19(3), 321–9. doi:10.1007/BF02471975.

Wikramanayake, E., Dinerstein, E., Seidensticker, J., Lumpkin, S., Pandav, B., Shrestha, M., Mishra, H., Ballou, J., Johnsingh, A. J. T., Chestin, I., Sunarto, S., Thinley, P., Thapa, K., Jiang, G., Elagupillay, S., Kafley, H., Pradhan, N. M. B., Jigme, K., Teak, S., Cutter, P., Aziz, M. A. and Than, U. 2011. A landscape-based conservation strategy to double the wild tiger population. *Conservation Letters* 4(3), 219–27. doi:10.1111/j.1755-263X.2010.00162.x.

Wittemyer, G., Elsen, P., Bean, W. T., Burton, A. C. O. and Brashares, J. S. 2008. Accelerated human population growth at protected area edges. *Science* 321(5885), 123–6. doi:10.1126/science.1158900.

Zimmerman, B. L. and Kormos, C. 2012. Prospects for sustainable logging in tropical forests. *BioScience* 62, 479–87.