OCCASIONAL PAPER

Futures of tropical production forests

Francis E Putz
Claudia Romero

OCCASIONAL PAPER 143

Futures of tropical production forests

Francis E Putz
University of Florida and CIFOR

Claudia Romero
University of Florida and CIFOR

Center for International Forestry Research (CIFOR)

Occasional Paper 143

ISBN 978-602-387-016-5
DOI: 10.17528/cifor/005766

Putz FE and Romero C. 2015. *Futures of tropical production forests.* Occasional Paper 143. Bogor, Indonesia: CIFOR.

Photo by Tri Saputro/CIFOR
A general view of a forest in Lambakara village, Southeast Sulawesi, Indonesia.

CIFOR
Jl. CIFOR, Situ Gede
Bogor Barat 16115
Indonesia

T +62 (251) 8622-622
F +62 (251) 8622-100
E cifor@cgiar.org

cifor.org

We would like to thank all funding partners who supported this research through their contributions to the CGIAR Fund. For a full list of the 'CGIAR Fund' funding partners please see: http://www.cgiar.org/who-we-are/cgiar-fund/fund-donors-2/

Contents

List of figures

Figures

Abbreviations

CL	Conventional logging
DBH	Diameter at breast height
FMU	Forest management unit
FSC	Forest Stewardship Council
IBIF	Instituto Boliviano de Investigación Forestal
LTSRPs	Long-Term Silvicultural Research Program
RIL	Reduced-impact logging
RUIL	Reduced undesirable impact logging
SBK	Sari Bumi Kusuma
SFM	Sustainable forest management
SILIN	Sistem Tebang Pilih Tanam Intensif Indonesia (Indonesian Cutting and Planting System)
STY	Sustained timber yields
TPTI	Tebang Pilih dan Tanam Indonesia (Indonesian Selective Logging and Planting System)

Acknowledgments

This research was conducted with the financial support of the UKAID-funded KnowFOR (Forestry Knowledge) program and the CGIAR research program on Forests, Trees and Agroforestry (FTA).
We are grateful to Plinio Sist and two anonymous reviewers who provided helpful comments on an earlier draft of this document. We also acknowledge the many insights gained from discussions with Ruslandi, Bryan Finegan, Tang Hon Tat, Marielos Peña-Claros, Pieter Zuidema, Paolo Cerutti, Daisuke Naito, Dennis Dykstra, Johan Zweede, Anand Roopsind, Timothy Synnott, John Palmer, John Healey, Jack Ewel, Bronson Griscom and Peter Ellis.

Executive summary

There are many silvicultural options for tropical forests that are spared from conversion for agriculture but not from commercial use for timber production. These options are summarized in three general trajectories that describe increasing silvicultural intensities and consequent increasing forest domestication. The silvicultural approaches discussed include: changes in timber harvesting regimes from polycyclic (e.g. single tree selection) to monocyclic (e.g. shelterwoods); treatments to increase growth of future crop trees (e.g. liberation thinning); treatments to promote natural regeneration (e.g. seed tree retention and soil scarification); and artificial regeneration (e.g. enrichment planting). The environmental, social and economic benefits of natural forest management will be maximized where these and other silvicultural treatments are applied in appropriate landscapes, as determined through functional zoning based on land-use capabilities and other factors. Unfortunately, research on tropical silviculture is currently insufficient to provide clear insights about the many tradeoffs associated with each possible intervention (e.g. timber production versus biodiversity retention, profits from timber sales versus maintenance of hydrological functions). In most of the tropics, large-scale, long-term silvicultural research efforts that are designed to inform decisions about forest fates are still lacking. Funds for that research should be forthcoming from forest-based efforts to mitigate climate change through increased carbon sequestration and from efforts to restore degraded forests. In the absence of solid interdisciplinary landscape-scale research, the value of forests spared from outright conversion will decline as a result of ill-informed management decisions.

Caveats and disclaimers

Although the focus of this paper is on the future of tropical production forests and emphasizes timber, this should not be interpreted as deprecating the importance of other land uses, products and services. Protected areas, agroforests, trees-on-farms and non-timber forest products all play critical roles in conservation and development but are only marginally addressed in this document. Similarly, in full recognition of the overwhelming importance of the socioeconomic and political contexts of production forests, the focus here is on silvicultural approaches to natural forest management. While we recognize the importance of good governance (e.g. participation, transparency, accountability and predictability) to sound forestry, the emphasis here is on forest management practices. Finally, although we describe shortcomings of forestry in the tropics, many of the same deficiencies apply equally well in temperate and boreal areas.

Lest we disappoint readers or cause them to feel that their interests or contributions were unjustifiably disregarded, we want to make clear what this paper is and is not. First, we focus on the future and do not pretend to review the extensive literature on tropical forestry – to which we refer readers to Wyatt-Smith (1963), Baur (1964), Lamprecht (1993), Wadsworth (1997), Louman et al. (2001) and Gunter et al. (2011). Second, although we make use of plot-based research results, our focus is on what is happening and likely to happen outside of experimental areas and at landscape scales. Third, we emphasize forest management practices without particular regard for the worldviews of the agents who employ these practices, except when the likelihood of practice adoption varies among agents in predictable manners.

1 Introduction

The fates of natural production forests in the tropics will continue to be determined by market forces, labor availability, governmental policies, qualities of governance and institutional frameworks (e.g. tenure and rights allocation), and cultural values, which all interact with the diverse impacts of climate change and the many effects of globalization (Putz and Romero 2014). Of these factors, financial issues, as affected by governnmental policies, their enforcement and institutions, will most often determine whether forests are degraded or managed (e.g. see Roda et al. 2015). These financial considerations are mediated by security of property rights (Agrawal et al. 2008), which in the case of forests includes concession granting (Karsenty et al. 2008), economic policies (Chomitz et al. 2007; Rautner et al. 2013) and cultural proclivities (Coomes et al. 2008; Feintrenie et al. 2010; Meijaard et al. 2013). Overall, if there is to be widespread replacement of exploitative timber harvesting by responsible forest management, and if this change is to help maintain semi-natural forest cover, radical changes are needed in the culture and practice of forestry. Furthermore, responsible natural forest management needs to be better understood and more widely accepted where it is an environmentally, socially and economically viable approach to conservation and development. It also needs to be kept in mind that decisions affecting forested lands lie at the intersection of socioeconomics and politics, as framed by diverse and often contested visions for the territory where forests are located.

Three different theoretical trajectories for tropical production forests constitute the core of this Occasional Paper (see Figure 1). The first represents the business-as-usual scenario in which forests are repeatedly but sporadically logged until they are converted to some non-forest land use (e.g. cattle ranches, agricultural fields or plantations

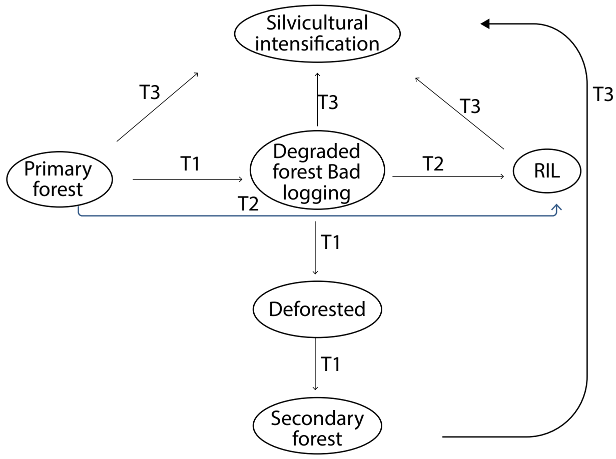

Figure 1. Representation of three likely trajectories (T1–T3) for natural production forests in the tropics.

of rubber, oil palm or wood fiber) or set aside as protected areas. Along the second trajectory, the forest is still repeatedly logged but the harvests are less frequent and reduced-impact logging (RIL) techniques (Dykstra and Heinrich 1996) are employed. The third trajectory involves the same RIL practices combined with other silvicultural interventions of various intensities. Before discussing the three trajectories, the differences between forest degradation and forest management are reviewed, reference conditions (i.e. comparators) for managed forests are considered, the general geographical contexts of tropical production forests are described and the economics of production forest fates are outlined. The overall goal of this paper is to inform discussions and decisions about forest management and conservation options available to a diversity of stakeholders across the full spectrum of tropical landscapes. The simplifications inherent in consideration of only three trajectories are acknowledged, as are the limitations in our perspectives, but we hope that what we present proves useful.

1.1 Forest management versus forest degradation

Trained crews carrying out logging as part of a silvicultural system designed to sustain timber yields while maintaining the forest's capacity to provide other goods and services, should be considered management and not degradation (Thompson et al. 2013). This conclusion stands even though timber harvests and other silvicultural treatments – no matter how carefully performed – decrease forest carbon stocks at least temporarily while modifying forest structure and composition. In fact, modification of forest structure and composition is the goal of silvicultural interventions. It is therefore unavoidable that what is management to one person is degradation to another. Justifiable differences of opinion about what constitutes degradation (Morales-Barquero et al. 2014; Ghazoul et al. 2015) are distressing insofar as policies designed to avoid or mitigate degradation will need to be based on explicit definitions. Given that managed temperate and boreal forests are not considered degraded, we cannot condone the inherent bias in reserving this derogatory designation for managed tropical forests.

From a carbon emissions perspective, whether or not a forest is degraded depends on where the measurements stop. To clarify, it is becoming increasingly clear that managed forests may sequester more carbon than those that are unmanaged, at least if harvested stands recover their carbon quickly and if a portion of the harvested timber is used in place of concrete, steel or other carbon-intensive materials, or as a substitute for fossil fuels (van Kooten et al. 2014). Although forest-based climate change mitigation projects in the tropics are not yet required to carry out full lifecycle analyses of the fates of forest carbon, that requirement looms large (see Section 1.6.1).

Tropical forestry suffers a terrible reputation for environmental and socio-economic abuses that is too often well deserved (e.g. Zimmerman and Kormos 2012). With increases in the urbanization of human populations and consequent decreases in direct exposure of the populace to natural resource management activities, this demonization is unlikely to diminish. In an essay entitled *Are you an environmentalist or do you work for a living? Work and nature*, White (1995) explores this tendency in general terms. However, the status of loggers is even worse than that of most resource managers (Putz 2004). The assumption that tropical forests are clear-cut for timber and the more nuanced but often fallacious assumption that logging somehow causes deforestation are both widespread. Even environmental scientists who recognize the importance of logged and managed forests for the retention of biodiversity focus on ways to keep logging from happening rather than on improving the practices utilized in logged stands (e.g. Edwards et al. 2014). More to the point, their focus is often on demarcation of protected areas within logging concessions rather than on improvements in silvicultural practices. The low regard for timber-based forestry is also evident in the staffing of international forestry research organizations such as the Center for International Forestry Research (CIFOR). During its 20 years of existence, social scientists have always abounded at CIFOR and seldom was there a shortage of other sorts of experts, but scientists with expertise in silviculture or forest engineering have often been scarce. Many international donors share this apparent low regard for the core traditional disciplines of tropical forestry; their opinion perhaps justified by the many millions of dollars wasted on failed attempts to reform the tropical forestry sector.

Fundamentally, if production forestry in the tropics is to continue, there needs to be cultural shifts from timber exploitation to forest management, from short-term profit-taking toward sustainable forest management (SFM), from a focus on single commodity production to multiple objective forest management, and from stand-level to landscape-level perspectives (e.g. Sayer et al. 2013). To a limited extent, some of these transitions have been made in some tropical forest management units (FMUs; e.g. concessions, community forests and private land holdings), often in the absence of governmental support (e.g. Katila et al. 2014). We need to understand why, where and when these changes have happened, and by whose instigation (e.g. Ongolo and Karsenty 2015). Miner et al. (2014) claim that "[t]he demand for wood keeps land in forest, provides incentives for expanding forests and improving forest productivity, and supports investments in sustainable forest management," but it is not clear if and where this statement applies in the tropics.

Environmentally, socioeconomically and politically appropriate natural forest management in the tropics is impeded by forest and other policies that do not recognize differences in land-use capabilities, differences in the expectations of forest owners and other stakeholders, the vast differences among species in their silvicultural requirements, and the wide variety of available silvicultural alternatives. Landscape scale sustainability, including biodiversity retention and realization of production goals, will best be served by functional zoning based on the full range of relevant biophysical, social, historical and other factors (e.g. Schöngart 2008; Messier et al. 2009; Grau et al. 2013; Wu 2013; von Wehrden et al. 2014; Law et al. 2015). Perhaps the systematic review of tropical silviculture underway by Petrokofsky et al. (2015) will reveal much of the information needed to guide functional zoning decisions at the forested end of the land-use planning process. However, based on other reviews of this topic (e.g. Günter et al. 2011), this seems doubtful. Instead it appears that even what seem like the most straightforward questions about tropical silviculture, such as whether reduced-impact logging is more or less profitable than conventional logging, do not yet have definitive answers (see Section 1.6). Even less is known about the biodiversity and other environmental effects of silvicultural treatments other than logging, or the financial cost-effectiveness of these interventions.

If responsible production forestry in the tropics is to be successful, the inherent tradeoffs need to be recognized, their costs made explicit and the benefits made visible. To a large extent it is not difficult to recognize responsible practices. For example, workers should employ safe practices, streams should not be blocked or polluted, environmentally and silviculturally undesirable stand damage should be avoided, and social conflicts should be appropriately addressed. But when it comes to nuanced or even drastic but intentional changes in stand structure and composition, the issue of the appropriate reference state needs to be addressed (see Section 1.2). In other words, when forestry activities are assessed in terms of their impacts, it is critical that undesired impacts (i.e. degradation) are differentiated from intentional impacts. Admittedly, most silvicultural interventions result in at least temporary reductions in biomass and changes in species composition, but such changes are transient and short timescales are not appropriate for evaluation of forest management (de Avila et al. 2015). Again, the fundamental difference between degradation and management relates to whether the observed changes were intentional and part of an approach to silviculture or incidental impacts of timber exploitation. Finally, if responsible forest management is to be promoted across the diversity of forest conditions and forest stakeholders, these concepts of management and degradation need to be appropriately negotiated and tailored (Goldstein 2014). At the least, anyone involved in a discussion of forest degradation should insist on clarity about what is meant by the term.

1.2 With what should managed forests be compared?

The use of primary forest as the baseline or reference condition against which to measure managed and degraded areas, which we recently recommended (Putz and Romero 2014), is challenging and perhaps wrong for both analytical and normative reasons. For one thing, it is becoming increasingly difficult to find primary forests with which to compare. Then there is the growing sentiment that the very concept of primary forest is outmoded given the increased recognition of the near ubiquity of human influences on nature (e.g. Ellis et al. 2010). Some researchers (e.g. Hobbs et al. 2013) go so far as to suggest that society accept as the "new normal" ecosystems that are novel in terms of structure, composition and dynamics.

Many traits of primary forests are inimical to sustained timber yields. For example, the clusters of huge, slow-growing, large buttressed trees that characterize many primary forests (Figure 2) are not silviculturally suitable where timber yields are to be sustained over financially viable cutting cycles. Timber yields can be sustained and these characteristics retained only if the harvest intensities are very low, cutting cycles are long and logging is carried out with exceptional care (Sist et al. 2003a, 2003b) but waiting more than 60 or 100 years for the next harvest is economically viable under only the most peculiar and unrealistic conditions (e.g. zero or negative discount rates and no other viable land uses). If yields are to be sustained under a polycyclic (i.e. uneven aged) management regime with more acceptable or at

Figure 2. Primary tropical rainforests with clusters of large trees with big buttresses might not be a suitable reference condition against which to evaluate the impacts of silvicultural treatments on the structure (e.g. standing stocks of timber) and composition of forests managed for profitable yields of timber (Photo by FE Putz)**.**

least typical cutting cycle durations (e.g. 25–30 years), then at least some of the traits of primary forests will unavoidably and intentionally be lost under the required silvicultural regime (e.g. Dauber et al. 2005; Peña-Claros et al. 2008). The required interventions will cause even greater departures from the primary forest reference state if the main commercial timber species are light-demanding (Fredericksen and Putz 2004). These interventions (e.g. liberation thinning, soil scarification and enrichment planting) are implemented for the expressed purpose of changing forest composition to favor trees of commercial species (de Avila et al. 2015). Unfortunately, at least from a timber yield perspective, these silvicultural treatments are seldom applied outside of the experimental plots where they were shown to be effective.

This lack of spontaneous commercial adoption of recommended silvicultural techniques is testimony to the failure of researchers to appropriately design their studies, their failure to communicate their results effectively and/or because these interventions are cost prohibitive.

A fundamental issue is that when a stand is managed principally for a few species or often just one life form (e.g. trees or palms), it is by definition managed against many others. For example, forest management for well-formed trees of commercial species is, to varying extents, management against trees with poorly shaped or hollow boles, lianas, hemiepiphytes (e.g. many *Ficus* spp.), giant understory herbs (e.g. *Heliconia* spp. and *Musa* spp.), shrubs (e.g. *Acanthus pubescens*) and palms (e.g. *Eugeissona tristis*). Changes in stand structure and composition of course vary with intensity of silvicultural intervention (e.g. cutting of all lianas or only those on future crop trees), as well as with the success of those interventions at achieving their silvicultural goals. Where applied effectively and intensively, the biodiversity and associated impacts can be substantial and natural forest takes on many of the characteristics of plantations.

In contrast to forest management, log mining (i.e. high-grading, creaming or exploitative timber harvests) changes stand composition through depletion of commercial tree species, increases in the relative abundances of non-commercial trees (e.g. Imai et al. 2014), and proliferation of lianas, shrubs, giant herbs, ferns and other non-arboreal life forms. The impacts of this exploitative trajectory on stand composition could be differentiated from management impacts on the basis of changes in the relative abundances of different species and life forms. However, the assignment of value to those different life forms requires participatory and transparent processes to elucidate the normative values of the appropriate stakeholders. For example, post-logging increases in the abundance of non-commercial, shade-tolerant, understory trees that produce fleshy fruits would be a positive change for wildlife (Costa and Magnusson 2003), a negative one for timber production and probably not affect hydrological services. Similarly, liana elimination might be a boon for timber production but have severe biodiversity impacts (Putz et al. 2001b). Which of these perspectives takes precedence will need to be negotiated on almost a stand-by-stand basis.

1.3 Sustained timber yields, a basic tenet of responsible natural forest management

Compared to the easy-to-invoke but hard-to-capture concept of SFM, the concept of sustained timber yields (STY) seems relatively straightforward. STY means that timber yields do not diminish over time. However, diminish relative to what starting volume? Often primary forest is taken as the baseline or reference state but is it reasonable to expect a managed forest to recover in a financially viable cutting cycle the volumes of timber that might have taken centuries to accumulate before the first cut? When researchers calculate the time needed for timber volumes to return to this primary forest reference level in polycyclic management regimes (i.e. multiple-age, selective logging), their estimates are generally much longer than the minimum cutting cycles established by governments, which are usually 25–35 years. But what if timber volumes in tropical production forests need only to return to 50% or 75% of primary forest levels? This seemingly radical departure from convention is apparently accepted in forests of the Pacific Northwest of the USA, why not in the tropics as well? Given that global production of tropical timber from natural forest has already passed its peak (Shearman et al. 2012) and many stands are being commercially logged for the second and third times, reconsideration of exactly what constitutes STY seems warranted.

Claims of STY deserve careful scrutiny, but if sustaining forests is the goal, perhaps flexibility is justified. Commonly, the way timber stocks are calculated evolves over time to include more species and trees of smaller stature and lower quality. Some critics are dubious about these changes in definition. But they do reflect changes in markets and the smaller profit margins when forests are harvested for the second and third time without silvicultural interventions to increase stocking and growth of commercially valuable species.

These considerations about STY remain based on comparisons with primary forest, which become less relevant and less possible over time. Recent enacted forestry regulations in Indonesia avoid this problem by sidestepping the fundamental idea of a minimum cycle in forests managed under its selective harvest (i.e. Tebang Pilih dan Tanam Indonesia, TPTI) guidelines. Rather than specify a minimum time between harvests, stands can be re-entered when they have >40 m^3/ha of commercial timber, as determined by decadal inventories. The financial attractiveness of this approach to loggers is obvious; if not too much time has elapsed, re-harvesting costs are generally low due to the availability of previously constructed roads, log landings and main skid trails. Instead of criticizing any reductions in minimum felling cycles (e.g. Arroyo-Mora et al. 2014), perhaps tropical silviculturalists need to accept that multi-decadal inter-harvest delays are simply not financially viable and develop ways to make alternatives more environmentally sound (Schöngart 2008). That said, the proportion of harvestable volumes for the second and subsequent cuts that represents recruitment and growth should be differentiated from wood from trees that were passed over until changes in species acceptability or decreased quality standards rendered them marketable; wood from trees of the latter sort should not be used to bolster claims of STY.

After application of the mono-cyclic Malayan Uniform System to lowland forests in Peninsular Malaysia, the government set the cutting cycle at 60 years, which was considered to be sufficient for timber stocks to recover (Okuda et al. 2003) but was apparently too long for loggers to wait. The few lowland dipterocarp forests that escaped conversion into oil palm plantations were re-harvested long before the end of the designated cutting cycle, generally with governmental approval. The message here is that the long cutting cycles often advocated by researchers concerned about sustainability are unlikely to be adhered to, even if nominally required by governmental regulations. Clearly the time value of money is simply too high for loggers to wait, and government officials generally accommodate their requests and permit premature re-entry logging. People concerned about environmental sustainability, future timber yields and intergenerational equity should push for acceptance of long cutting cycles. However, with forest conversion such a financially attractive option, arguing for the mitigation of the deleterious impacts of intensification of all aspects of management might be more efficacious.

1.4 Geographies of production forest fates

Forest fates are often correlated with their accessibility, which substantially influences the profitability of different land uses. This relationship was captured in a graphical model that was first proposed by von Thünen (1966), later adapted by modern economists and geographers (e.g. Chomitz et al. 2007; Southworth et al. 2011; Angelsen and Rudel 2013), and then further modified to emphasize the high cost of securing property rights near forest frontiers (Hyde 2012) (see Figure 3). The basic idea is that improved access increases potential land rents while spurring immigration and promoting good governance. Support for these trends derives only in part from the fact that many areas that are and will remain remote are otherwise unsuitable for intensive land uses due to steep slopes, high elevations, particularly infertile soils or social unrest.

A von Thünen approach will be used in this paper but one caveat first needs to be registered. This model, at least as generally used, is based on an assumption that distance to markets (or travel time or transport costs) determines the profitability of different land uses but is not specific about the nature of these markets (e.g. whether they are formal or informal). Where the determiners of forest fates are tightly integrated into international, national or at least regional markets, it makes sense to assume that the market of interest is located near the closest city or port to which products are delivered. What is questioned here is the assumption of a single market and hence a single metric for accessibility. Where the focus is on plantations and agricultural crops, this assumption seems reasonable. In contrast, accessibility as measured for these sorts of land uses does not pertain if the forest fate deciders are subsistence farmers or actual forest dwellers. Although few farmers are entirely subsistence oriented, many are only marginally integrated into even regional markets. For those people, distance or travel time to cities, mills or transportation hubs matters little to their land-use decisions. And while road construction facilitates their access to land, the quality of those roads is not so important. For example, roads constructed by loggers facilitate extensive forest clearing by swidden farmers in Indonesia even after those roads become impassable by vehicles (see Figure 4). That said, distance along those roads to formal markets seems to have little influence on the forest clearing practices of these farmers.

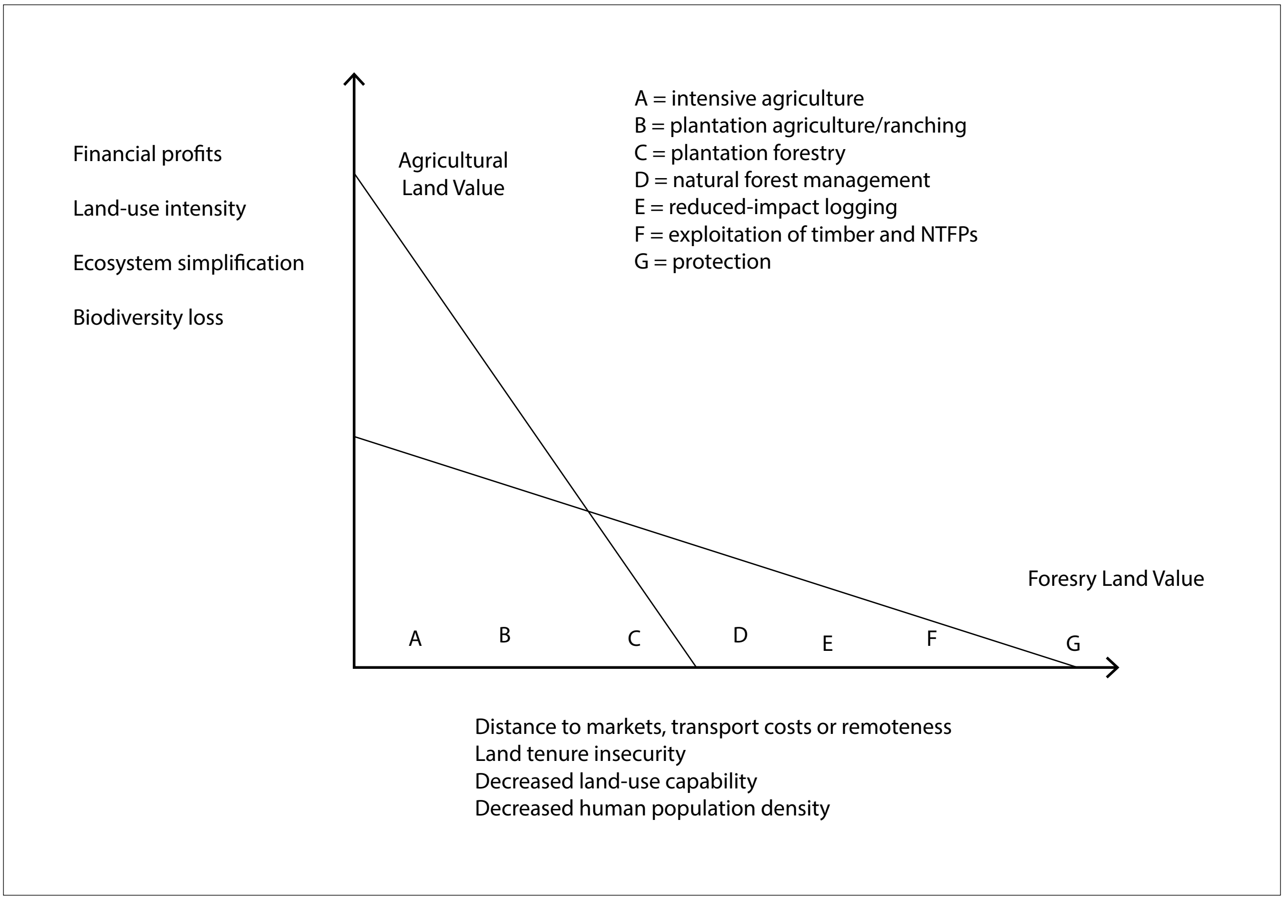

Figure 3. A modified von Thünen diagram depicting the relationship between accessibility and its correlates with land-use intensity and profitability.

Figure 4. ***Imperata cylindrica (alang-alang)*** **grassland in an abandoned forestry concession within a government-designated production forest in West Kalimantan, Indonesia.** After abandonment by a parastatal concession, the area was rapidly cleared by swidden farmers, then invaded by pyrogenic grass and subjected to frequent fires. The Ministry of Forestry still claims the area as part of its forest estate (Photo by Ruslandi).

Some of the present deforestation threats in tropical production forests can be traced back to when governments first gazetted lands without regard to the seemingly legitimate land claims of local people (e.g. Sears and Pinedo-Vásquez 2011; Hecht et al. 2014). While the criminalization of traditional land uses is unethical, clear conflicts between production forestry and traditional agriculture are often yet to be addressed.
For example, forest concessions in Kalimantan (50,000–150,000 ha), Indonesia, typically host 5000–10,000 people in 5–15 villages (Ruslandi et al. 2015). Concessionaires are expected to negotiate with these people and do so in various ways with various outcomes. Unfortunately, cultural, linguistic, religious and power differences characterize the negotiations, which are plagued with misunderstandings. Given the acceleration of tropical forest concession granting and the apparent deceleration of tenure granting to rural people (Corriveau-Bourque et al. 2014), avoiding such misunderstandings is a paramount concern. Despite the deficiencies in company-community relations, the widespread and rapid clearance of forests by swidden farmers when concessions are abandoned suggests that the negotiated agreements between concessionaires and villagers are at least somewhat effective in slowing deforestation. Where forestry operations serve as a magnet for forest colonists, the challenges are somewhat different. In one famous case in Africa, for example, the Congolaise Industrielle des Bois (CIB) concession now hosts thousands of people, most of who reside in a town that emerged around a timber processing facility (personal communication from J Poulson, March 2015).

Another geographical characteristic of forests that influences their fate is their spatial extent. Forest fragments isolated by deforested land or other sorts of matrix vegetation unsuitable for forest species (e.g. well-manicured plantations) are susceptible to edge effects as well as to the longer-term consequences of isolation of small populations. Small patches of forest, especially if previously high-graded of their valuable species, yield only small profits when harvested again, which often precludes silvicultural interventions with substantial set-up costs. For example, due to the costs of transporting bulldozers, excavators and other large machines, industrial scale logging is only profitable in forests >100–500 ha. Where forest properties are smaller, owners can band together to reduce operational costs and thereby attract loggers. This sort of collaboration among property owners is also financially favorable if each owner can thereby avoid the often substantial costs of writing and registering management plans and then administering timber harvests (e.g. Sist et al. 2014a).

1.5 Tenure regimes in tropical production forests

To predict the fate of an area of land, it helps to know who owns it and what constitutes ownership – from informal tenure and rights to fully recognized rights and responsibilities (e.g. Corriveau-Bourque et al. 2014; Putzel et al. 2015). Tenure is a bundle of different rights held with different strengths for different durations. Owners can include governments, communities (indigenous and otherwise), and private individuals and firms. Forestland owned (or at least claimed) by governments can be designated for protection or conversion, parceled out in concessions, or remain unclassified, and all of these designations are subject to change (e.g. Phelps et al. 2010; Mascia and Pallier 2011). For example, concessions can be of various durations and include a variety of rights and responsibilities ranging from short-term logging contracts to long-term and renewable concessions of >30 years.

The concept of community ownership of land is more complicated, with rights ranging from non-commercial usufruct to the full set of rights that accompany full ownership (i.e. exclusion, due process and compensation, and unlimited duration). Along that complex continuum of tenure rights, communities can parcel out portions of their lands to individuals or hold and manage the land communally. Lands in all tenure categories range in size, suitability for different uses and pertinent regulatory regimes. At the same time, landowners vary in their utility functions (i.e. their preferences or what they want to maximize or optimize), planning horizons, access to financial capital and labor, individual capabilities, and personal predilections. Finally, many of these designations are questioned, such as where governments have granted concessions for lands claimed by local people. Recognition of these competing land claims is increasing, particularly in Latin America and potentially in Indonesia (Kelly and Peluso 2015), but contested land claims continue to characterize much of the production forests in the tropics.

Here we are concerned with the ways land tenure might influence the fates of tropical production forests, particularly in regards to their silviculture. While we support the return of large areas of tropical forest to the people from whom they were wrested by governments and others, we do not believe that this act alone will spontaneously lead to large-scale forest protection or sustainable management (e.g. Gould 2006). In contrast, smaller scale and higher intensity management of natural forests and plantations is more likely where local people have secure tenure, especially if they have seasonal labor surpluses (Batra and Pirard 2015).

1.6 Economics and the fates of production forests

From a global environmental perspective, leaving tropical forests unlogged, gold and coal un-mined, and petroleum reserves un-tapped would be ideal. While the leave-it-alone option might seldom be exercised for any of these resources, there are places where forests should be left unexploited for reasons of low profit margins, or high environmental or cultural costs. For whatever reason, where society values unlogged forests for their biodiversity, ecosystem services, recreational potential or aesthetic values more than for the profits from resource exploitation, protection is possible even if the opportunity costs of that option are substantial. There may also be places where, after the harvest of scattered precious timbers, the opportunity costs of forest retention are low enough to render protection feasible (Rice et al. 1997). Given the infrastructure required for even exploitative logging and the facilitation of land-use intensification by even the most rudimentary roads, there are likely few places where the log-and-preserve option is viable.

Production forest fates are often determined by economic factors. However, assessment of those factors is fraught with challenges mostly related to lack of quality data, failure to account for spatial and temporal variability in profit margins, unjustified faith in market incentives based on the capture of environmental and social externalities, and the need to consider the perspectives of a variety of stakeholders not all of whom share the same preferences (Applegate et al. 2004). At its most simple, any conclusions about the economic competitiveness of production forestry as a land use need to be informed by reliable financial data about forest management costs and benefits from the FMU's perspective. Unfortunately, to avoid taxation (*senus latu*, which includes both official and unofficial taxes), it is generally to a firm's financial advantage to exaggerate costs and to minimize profits. If FMUs are directly linked with exporters or processing industries, then data distortion can be an integral component of "transfer financing" schemes. Then there is the problem of using out-of-date data (e.g. Fisher et al. 2011; Ruslandi et al. 2011), which is often understandable given the rapidly changing world of tropical forestry. Finally, there is the challenge of accurately capturing the costs of governmental corruption, which can be substantial (Cerutti et al. 2013).

In a highly regulated industry such as forestry in which there are large numbers of administrative requirements, opportunities for corruption abound and the cost of legality can exceed the likely profits from forest management. The proliferation of unofficial 'administrative' costs is fostered by the remoteness of many forest operations. The lack of transparency in forest industries, which might provide some tax advantages, also renders firms prone to the depredations of corrupt government officials. Every time an annual cutting permit needs to be approved, every harvesting operation inspection, each log that needs to be scaled and graded, every logging truck that passes a checking station, and every load of logs that needs to be consigned for shipping provides a ready-made opportunity for graft. To facilitate each of these administrative steps, forestry firm representatives often need to make unofficial payments to government officials. Such payments are often accepted as part of the cost of doing business and are counted as administrative fees. Lest readers think that this hard-to-substantiate but generally accepted conclusion pertains only to tropical forestry, note that avoidance of corruption is one reason why employees of the US Forest Service involved in timber sales are regularly required to change districts and even states.

A good example of the difficulties in assessing the financial aspects of tropical forestry operations are provided by published attempts to address the question of: “Is RIL cheaper?” The first challenge is to unpack that question into its components: cheaper to whom, per cubic meter or hectare, at what discount rate, over what time period and with the capture of which externalities (e.g. carbon, biodiversity or hydrological services). In an effort to answer the main question, Medjibe and Putz (2012) and more recently Holmes (2015) assembled data from the thirteen published studies that compared the harvesting costs of RIL and conventional logging (CL). Even with this seemingly simple contrast, (i) small sample sizes, (ii) missing information, and (iii) variation in research approaches, frames-of-reference and data reporting protocols rendered the results ambiguous and otherwise questionable. For example, the studies varied in whether they reported cost data per cubic meter harvested or per hectare. With the per hectare data, it was not clear how to compare the results from a study that used unreplicated 100 ha blocks on level terrain that were subjected to either RIL or CL (Holmes et al. 2002) with those from a study in which a large portion of one of the four replicated 40–60 ha RIL plots was steep and not logged but most of its four similar-sized CL plots were completely logged despite the abundance of steep areas (Healey et al. 2000). The former study, which according to Google Scholar (accessed 29 September 2015) was cited more than twice as often as the latter (205 versus 87 citations, respectively), concluded that RIL is substantially more profitable whereas the latter concluded the opposite. Clearly, the Holmes et al. (2002) result is strongly preferred by researchers but we suspect that the truth lies somewhere between the two studies. The more recent review by Holmes (2015) concluded that RIL is cost-effective in the neotropics but less so in Southeast Asia. We wonder about the extent to which his conclusion rests on topographical differences in the particular forests studied, gentle in the former and steep in the latter. There does seem to be general agreement on the long-term and non-timber yield comparative benefits of RIL adoption.

A free, downloadable software package[1] called RILSIM will hopefully be increasingly used to improve the thoroughness of studies of logging costs. With a user-friendly interface, this package helps identify, partition and track the various costs of logging, and then helps with cost calculations. If the protocols described in this package were more widely utilized, the problems with comparing financial analyses of logging would diminish substantially. It might be useful to expand RILSIM (or a model like it) beyond industrial timber management to incorporate other forest values with financial costs and benefits that deserve tracking.

Another tool was recently released that has some of the same functions as RILSIM but for communities and smallholders who manage forests. The Green Value Tool[2] has helped elicit discussions around the financial benefits of timber exploitation. Although its application to date has mostly been in Latin America, it is hoped that its applicability can be mainstreamed to other geographies where small-scale forestry contributes to local livelihoods.

Despite the insufficiencies in data and analyses, there seems to be general consensus that the opportunity costs of forest retention are high (e.g. Fisher et al. 2011). Such a conclusion is rife with assumptions, many of which are violated precisely where production forestry has a future. The fundamental questions about this conclusion are: compared to what, where and from whose perspective. If the comparison is to natural forest management for timber and oil palm plantations on level terrain, where labor is abundant and

1 http://www.blueoxforestry.com/rilsim/

2 http://earthinnovation.org/our-work/case-studies/green-value/

cheap, titles are clear, travel times to an extraction facility are less than a few hours, and from the unique perspective of the plantation's CEO, then an order-of-magnitude difference in net present values is believable. In contrast, if the comparison pertains to an area far from a processing facility that suffers from adverse terrain, contested ownership and limited labor, a different conclusion might be reached. Between these two extremes there is a gradient of conditions that influence which of the range of possible outcomes are most likely. Furthermore, if a prospective plantation owner values work schedule flexibility or some other utility more than profit maximization (e.g. Dove 2011), then the very demanding business of oil palm plantation management might not be very attractive. More generally, the high capital costs of oil palm cultivation coupled with the need to be in close proximity to a processing facility often precludes participation by many landowners.

With any positive discount rate, the time required for timber trees to reach their commercial potential often renders natural forest management financially unattractive at least when profit is reckoned per unit land area. Unfortunately, if recommendations for lower logging intensities and longer cutting cycles are followed (e.g. Sist et al. 1998), the net present values of standing forests are even lower. That said, some forest owners prefer large but occasional (i.e. 'lump sum') payments from timber sales to steady but small payments from other land uses. Sist et al. (2014c) explore this situation in the Brazilian Amazon where associations of smallholders contract logging companies to harvest timber from the 80 ha of legally required forest reserves on each of their 100 ha land allotments. The logger bears the considerable costs of management plan preparation and approval as well as the capital costs of logging equipment, and then pays each smallholder USD 16/m^3 for the timber harvested from their allotment (13–16 m^3/ha). Although the annualized incomes from logging were only about 10% of those from agriculture, farmers benefitted from the lump-sum payments of USD 5000–30,000 from the selective logging of forest that they could not legally clear for agriculture anyway.

Our discussion of forest fates focuses on volumetric yields of timber but it needs to be kept in mind that timber sale prices differ by an order-of-magnitude. For example, the harvest of 5 m^3/ha of cabinet-grade woods in the Amazon might be much more profitable than the harvest of 50 m^3/ha of commodity-grade timber in Borneo. The end uses of this material also need to be considered in management decisions insofar they affect the degree to which timber from different species can be substituted, as well as the life cycle of the sequestered carbon. The substitutability issue is particularly pertinent at the lower end of the timber price spectrum where plantation wood can provide lower-cost substitutes for wood harvested from natural forests. Although sale prices of tropical timber have not increased greatly over the past decades, there are important differences among species and over time on different markets; comparisons of prices are somewhat challenging because they can justifiably be given as domestic and export prices, freight-on-board or at the millgate, roundwood (i.e. logs), saw timber, or veneer sliced or peeled to different thicknesses. Given the wide range in conversion efficiencies (i.e. roundwood to these semi-finished products) among mills, countries and products, comparisons are rendered even more difficult if log prices are not available.

Market-based incentives for responsible forest management have and will continue to help motivate the transition from timber exploitation to responsible forest management. Although credible empirical assessments of the impacts of these incentives are only now underway (e.g. Romero et al. 2013), there is some evidence and widespread belief in their effectiveness.

1.6.1 Payments for environmental services

Although efforts to commoditize the environmental and social benefits provided by tropical forests are not yet operational beyond the project scale, future capture of these externalities in financial comparisons of different land uses seems likely (Wunder 2013; 2015). For example, it makes fundamental sense to include consideration of the hydrological benefits of SFM (e.g. control of erosion from logging road construction on steep slopes). Similarly, given that willingness-to-pay studies suggest that many people value biodiversity, it seems reasonable to use any forthcoming funds for biodiversity retention or recovery to reform tropical forestry. Also, given that tropical forests can contribute substantially and cost-effectively to mitigation and adaptation to climate change, it makes sense to include improved forest management in REDD+ programs and subsequent climate change interventions. Finally, decision makers should recognize the recreational potential of many managed forests as well as their yields of many non-timber forest products. Unfortunately, despite the demonstrated or potential capacity of these benefits to contribute to financial profitability of SFM, they remain external to the financial calculations that determine the fates of most tropical forests. While efforts to capture these externalities (e.g. forest certification and REDD+ interventions) need support, their inclusion at present serves to obfuscate the drivers of forest degradation and the currently relevant financial arguments for conversion (e.g. Pokorny and Pachecho 2014).

The cause of production forestry will be served by well-designed and comprehensive economic evaluations. For these evaluations to be credible as well as accurate, they should be broadly participatory. In other words, input is needed from stakeholders along the entire pathway from the forest to the end users of forest products. Failure to adequately consider the perspectives of forest industries, for example, may help explain why the acronym RIL is still interpreted by some as standing for reduced-income logging (Applegate et al. 2004). It might also explain the disregard of the research results that demonstrate the financial benefits of modified excavators (e.g. Log Fishers or yoaders) for timber yarding – some companies in Malaysia that purchased these machines reportedly leave them parked and have returned to using crawler tractors to skid logs to roadsides.

We believe that soon, contrary to the well-accepted assumption on which international climate change mitigation policies are currently based, it will be more widely recognized that prevention of logging-induced forest degradation, here defined as loss of carbon from forests that remain forests, may not reduce net carbon emissions to the atmosphere. Instead, to maximize emission reductions, sometimes more timber should be harvested, not less (e.g. Oliver et al. 2013; van Kooten et al. 2014; Lippke et al. 2015). This argument rests on a combination of benefits from replacement of fossil fuels with biomass fuels and replacement of carbon-costly steel, aluminum and cement with wood-based products, as well as on the assumption that forest carbon stocks in the harvested stands recover quickly. Implementers of a REDD+ intervention that converts a tropical forest slated for logging into a protected area might justifiably claim to reduce net carbon emissions at the site level (i.e. *in situ* carbon). If in response to the intervention the loggers simply go elsewhere, current policies require that the accounts reflect this leakage of carbon benefits. What we argue here is that the emissions benefits of such an intervention also need to be adjusted downwards if materials with large carbon footprints or fossil fuels replace the non-harvested wood. The extent to which carbon life-cycle analyses will diminish the climate change mitigation benefits of REDD+ interventions that limit timber harvests remains to be determined, but nevertheless needs to be considered. Clearly, more life cycle analyses of tropical forest carbon are needed lest climate change mitigation efforts actually exacerbate the global problem. What are not in question are the carbon and conservation benefits of efforts to limit the unnecessary emissions from tropical timber harvests without reductions in volumetric yields (i.e. RIL).

1.6.2 Plantations and natural forests

Whether forestry plantations reduce pressure on natural forests (e.g. Sedjo and Botkin 1997), depends in part on the substitutability of plantation and forest timbers as well as on how plantations and forests are distinguished (Putz and Romero 2014). With increased intensity of natural forest management and increased diversity of plantation owners and objectives (Batra and Pirard 2015), the distinction will become even more blurred and communication about the environmental, economic, and social tradeoffs associated with management intensification will continue to be impeded. That said, it is and will remain easy to distinguish forests from large-scale industrial 'fast-wood' forestry plantations managed as monocultures of exotic species from which biomass, pulp and other low-value commodities are harvested with clearcuts at very short intervals (e.g. 5–15 years). It is unlikely that this sort of plantation will do much to relieve pressure on natural forests. In contrast, where rotations are longer and saw timber and veneer logs from native species are among the product mix, the forest-plantation distinction becomes less clear. Currently, only teak (*Tectona grandis*) plantations are a substantial source of cabinet-grade hardwoods (known in some circles as "noble" timbers), but this situation seems to be changing. In contrast, utility-grade timbers from natural forests and plantations already compete for market share and the latter often win due to lower production costs. For example, poplar (*Populus* sp.) from plantations in China and rubberwood (*Hevea brasiliensis*) from plantations in Malaysia are suitable substitutes for much of the Dipterocarpaceae timber harvested from natural forests in Indonesia and sold as veneer for utility-grade plywood.

With increasing intensities of natural forest management (i.e. Trajectory 3 described in Section 4), it will become increasingly difficult to differentiate forests from plantations. One pertinent example is where enrichment planting operations are carried out successfully. It is important to note that in his classic book Plantation Forestry in the Tropics, Julian Evans (1982) refers to enrichment planting as a forest conversion technique. The extent of that conversion is often not evident until after the first harvest (see Section 4.2) when huge volumes of timber from broad-crowned trees are extracted. It should be recognized that with such harvests, multi-aged stands formerly managed by selective logging under a polycyclic regime are thereby transformed into even-aged stands with monocyclic management. With increasingly frequent and strident calls for restoration of degraded tropical forests (e.g. Suding et al. 2015), plantation establishment through enrichment planting is likely to become much more of an issue in the near future.

It unfortunately appears that in most instances when the availability of inexpensive plantation wood reduces the profitability of harvesting timber from tropical forests, harvesting of the latter does not stop but instead shifts to lower capitalized harvesters (e.g. Mejia and Pacheco 2014). Sadly, the abandonment of industrial concessions after the first cut when profitability of management is reduced further by the availability of plantation timber, many forests are rapidly destroyed by a sequence of even more predatory loggers and then colonist farmers (e.g. Piu and Menton 2013). Logger-built roads facilitate this unplanned conversion but it appears that even the presence of commercial loggers who do not harvest timber in a sustainable manner can slow the rate of deforestation (e.g. Gaveau et al. 2013).

While we hope that industrial fast-wood plantations continue to be differentiated from forests, there are intermediate cases that do not fully deserve to be called either. Certainly from biodiversity, productivity and other perspectives, intensively managed stands of naturally regenerated, native, fast-growing commercial species can be quite plantation like. A good example is where naturally regenerated stands of *Euterpe oleraceae* (acai palms) are intensively managed as monocultures for fruit in the estuaries of the Amazon (e.g. Freitas et al. 2015). As in that instance, such conditions are more likely to occur where labor is abundantly available and markets are good for forest products. Another example is in Amazonian Peru where many smallholders manage portions of their agricultural fallows for construction-grade wood from a naturally regenerating native pioneer tree species, *Guazuma crinita* (bolaina; Putzel et al. 2013). Sawtimber from these stands, which is harvestable after only 11–12 years, feeds booming domestic markets for low-cost building material. Unfortunately for the farmers, because they do not plant the trees, the stands are subjected to the full set of very cumbersome regulations governing natural forests, which compels many toward informal markets.

1.6.3 Cost of being legal

Compliance with governmental forest regulations is costly, which reduces the profitability of legal operations. In addition to the direct costs of annual permits, land-area based fees and royalties that might vary with species, the indirect costs need to be captured (e.g. taxes on fuel and lubricants, and duties on imported equipment and spare parts) as well as the staff time required to set up and administer the permitting processes. Even more difficult to calculate are the costs accrued when government permits are not forthcoming in a timely manner. For example, if forest workers are still paid when logging is halted for administrative reasons, if mills cannot work at full capacity due to lack of logs, or if export quotas cannot be filled and business deals consequently suffer, profits to forest industries decline. One consequence of this condition is that the commercial value of standing forests decline, which renders other land uses more financially attractive.

Although the regulatory constraints on what seem like reasonable silvicultural practices (e.g. patch cuts in the Yucatan of Mexico and fallow forest harvests in Peru; Putzel et al. 2013) are often based on reasonable premises, they nevertheless can have perverse and long-term consequences when the rules are slow to change. On the other hand, it needs to be recognized that when regulations change, especially when they are relaxed or simplified, abuses are commonplace (e.g. Finer et al. 2014).

The high cost of compliance with governmental forestry regulations often limits participation in the sector to only the best-capitalized and most experienced firms. In Cameroon, for example, Eba'a Atyi et al. (2013) estimated that the cost of preparation of a government-mandated management plan averages USD 294,855 for a 58,971 ha concession. Similarly, in Brazil, costs of required inventories and management plans for a community-held concession exceed USD 100,000 (Drigo et al. 2013). Some companies overcome these cost barriers, especially where legality verification is mandatory, but FMU abandonment is also likely (Papp and Vidal, in press).

Governmental attempts to capture rents from the forestry sector are often justified. However, it now appears that the rents and associated costs are sometimes too onerous for the viability of forest industries in general, and the participation of the less affluent and powerful in particular (e.g. Hirakuri 2003). Several governments have attempted to reduce the administrative burden for harvesters of small volumes of timber from privately-owned and community lands. Unfortunately, most efforts to reduce the forest planning and reporting requirements of smallholders, however well intentioned, often just provided another avenue for laundering illegally-harvested timber and served to keep the price of timber low (e.g. Finer et al. 2014). Governments that want to maintain viable forest industries that are linked to sustained sources of timber need to carefully examine their regulations and, when necessary, change them. For example, it has been recognized for decades that log export bans or any other restrictions on sales to the highest bidder might serve to strengthen national industries. However, if it keeps log prices low to the point that forest management is not profitable, then the benefits down the market chain are only short lived (e.g. Vincent 1992). Perhaps such bans serve a purpose during some stages in the development of a country's forest industries, after which they have deleterious impacts.

2 Trajectory I: Continued forest degradation by poor logging

The business-as-usual scenario of continued forest degradation by loggers is likely under a variety of tenure regimes and under a wide range of regulatory, socioeconomic and biophysical conditions. Anywhere that the fates of forests are principally determined by profit-maximizing loggers, be they rural community members or employees of industrial timber concession, this trajectory is favored. Degradation is likely to lead to deforestation in accessible areas where non-forestland uses are financially remunerative, at least over the short term (e.g. Asner et al. 2006). Basically, this trajectory is likely where governments and other landowners fail to maintain the ecological integrity and productivity of their forests; if forest industries are allowed to whither and forestry expertise diminishes, forest management options will be curtailed. In contrast, where due to shortages of capital or labor, or where better land for cultivation is available, timber-depleted forests will remain standing in a degraded state.

Forests degraded by multiple-entry logging as well as secondary forests recovering after clearing for agriculture characterizes many of the private non-industrial forests in the USA, particularly east of the Mississippi River. It is also likely to characterize many smallholder forests in the tropics. In the USA, there are marked differences in land-use practices between private and public (i.e. government owned) lands, as well as between private non-industrial forestland and large corporate-owned forests. High intensity management practices are much more likely to be encountered in corporate-owned forests than on public lands to which concession rights have been granted. While much industrial forestry follows an extremely productive agronomic model with short rotations and massive inputs, private non-industrial forests are more likely logged when sufficient timber has recovered or when markets open for new species or lower quality logs. Close-to-nature silviculture that mimics natural disturbance regimes, where practiced, is more likely on public than on private land.

For privately owned non-industrial forests in the USA of 10–100 ha, owners are typically approached every decade or so by free-lance or industry-linked foresters who offer cash for their timber. Some landowners resist the financial temptation and protect their forests but many go along with the deal without making many stipulations about minimizing environmental impacts or assuring sufficient regeneration. The activities of loggers are controlled somewhat by their voluntary compliance with best management practices developed in each state, but other than providing some protection for water bodies, best management practices are weak and even full compliance falls far short of SFM. In any case, most private non-industrial forestland in the USA is logged without the benefit of detailed harvest plans. In a few states (e.g. Oregon), replanting is required to maintain land tax exemptions but usually the logged-over forest is left to recover without management interventions. As a future for smallholder forests in the tropics, this scenario may seem dim but many of those mistreated forests in the USA are locally cherished and yield much more than timber.

2.1 Deforestation and forest degradation within logged production forests

Recent remote sensing studies in Peru (Oliviera et al. 2007; Miranda et al. 2014), Sumatra (Gaveau et al. 2011) and Kalimantan (Gaveau et al. 2013) reported that deforestation rates within legally gazetted production forests were lower than in nearby counterfactual forests. Similar research in the Republic of Congo reached the opposite conclusion (Brandt et al. 2014) but the legitimacy of that result has been questioned on the basis of problems with the research design and interpretation of the available remote sensing images (personal communication from P Cerutti, 2015). Whatever the results, explanations of the fates of production forests need to be informed by insights about social and political processes that are not detectable by satellites (e.g. Gaveau and Pirard 2015). In other words, without an appropriate theory-of-change as well as on-the-ground validation with appropriate socioeconomic information, false attribution and other forms of misinterpretation are almost unavoidable.

Oliviera et al.'s (2007) satellite-based estimates of rates of deforestation and forest degradation in forest concessions in Amazonian Peru might be precise but the study nonetheless demonstrates how failure to establish proper counterfactuals as well as lack of social, political and economic ground truthing can lead to spurious conclusions. The authors reported that between 1999 and 2005, deforestation and forest degradation rates within legally allocated forest concessions were much lower than in surrounding forests. We do not question their interpretation of the satellite images but believe that the most likely explanation for this surprising finding is more insidious.

Amazonian Peru is rife with illegal logging and disregard of forest policies is the norm (e.g. Smith et al. 2006), which together allows some timber industries and governmental officials to profit substantially (Finer et al. 2014). Recent efforts to enforce harvest regulations within concessions and thereby reduce the markets for illegally harvested timber from elsewhere are commendable, but certainly during the period covered by Oliviera et al.'s (2007) study, illegal timber was abundantly available to forest enterprises. That there was less logging within the concessions more likely indicates that the concessionaires used their logging permits to launder illegal logs (e.g. Urrunaga et al. 2012). It was certainly commonplace at that time to observe logs being floated down the Amazon, Napo and Ucayali Rivers that accumulated stamps, tags and other bogus indications of legality as they approached the government offices in Iquitos and Pucalpa (FE Putz, personal observation).

The usefulness of remote sensing studies of deforestation and forest degradation within production forests increases if the primary and secondary impacts of forest management activities are differentiated (Putz et al. 2001a) and the parties responsible for both are identified. Culpability is challenging to assign because the attendant processes are complex. Loggers build roads, which are widely recognized as promoting forest colonization by people, forest degradation (e.g. defaunation and increased incidence of wildfires) and outright deforestation (e.g. Laurance et al. 2014).

Ultimately, the builders of roads into formerly roadless forests, be they loggers, gold-miners, oil companies or government-sponsored development projects, are directly responsible for some deforestation and forest degradation but are also responsible in more complicated ways for the secondary effects on the same (see Section 2.2). The von Thünen model, introduced in Section 1.4, supports this. If forests remained roadless, their susceptibility to environmental abuse would remain low. That said, forestry firms are in the business of harvesting timber and are granted concessions by governments for that purpose. Furthermore, many of the roads built by loggers fit into the infrastructure development plans of governmental officials and others. Finally, it should be remembered that forest industries provide jobs and contribute revenues directly into governmental coffers in the form of concession fees, royalties, etc. They also contribute in less transparent ways to corrupt governmental officials. These economic contributions notwithstanding, the challenge remains how to foster resource-based development while minimizing environmental and social damage. For example, more needs to be learned about how to most effectively close logging roads so as to mitigate their direct (e.g. erosion) and indirect (e.g. increased forest access) environmental impacts. Certainly from an environmental standpoint, calls for restrictions on road building into roadless areas deserve strong support (Laurance et al. 2014).

2.2 Secondary impacts of forestry on degradation and deforestation

Historically, when governments declared areas to be permanent production forests, many of those areas were occupied by people who had no influence on the decision (e.g. Obidzinski and Kusters 2015). Those occupants typically held no formal titles to the lands they traditionally occupied, which was one reason why their *de* *facto* rights could be disregarded. When officially designated production forest in which people already dwell is allocated to a concessionaire, it is typically the responsibility of the concessionaire to negotiate land-use rights and other relationships with local residents. The responsibilities are a bit different in the case of colonists who arrived after concessions were granted, but many of the same issues remain salient. These issues change little when the settlement and land-use patterns of local residents shift to take advantage of the road access created by the forestry firm.

For a variety of reasons that include concerns about fairness and a desire to avoid conflicts, forest industries often negotiate agreements with villagers in their concessions. These negotiations and subsequent enforcement of any agreed upon rules are rendered challenging by differences between forest industry representatives and local people in language, religion, ethnicity and culture. Utility functions of the two parties in these negotiations are often vastly different, and misunderstandings are almost unavoidable. Forest industry representatives in these negotiations are backed by more money, have better information and are overwhelmingly more powerful than the locals. This lack of balance in power often renders negotiated outcomes inequitable.

Many questions need to be addressed if culpability for deforestation within FMUs is to be correctly assigned. For example, if local residents in government-granted forestry concessions deforest as part of their land-use strategy, should this deforestation be counted against the concessionaire? Does it matter whether the parties responsible for the deforestation are long-term residents or newcomers? Should local residents blame the forest industry for deforestation if the government does not support restrictions on that activity?

If village boundaries within concessions are agreed upon, it does not seem fair to blame deforestation within those areas on the forest industry. Unfortunately, understanding of the meaning of negotiated boundaries often differs between villagers and concessionaires. These misunderstandings often become apparent when newly constructed roads facilitate farmer access to areas beyond the boundaries they did not recognize. Villagers may also disregard previously agreed upon restrictions on farming beyond the negotiated boundaries of their village area out of ignorance of the locations of those boundaries or out of a sense of entitlement. Claims to ever-larger portions of concessions may emerge from local population growth (intrinsic or augmented by new colonists), increased capacity to clear forest due to acquisition of chainsaws, intra-community schisms and increased political power.

The issues broached above likewise pertain to illegal logging and market hunting within concessions. Other than where the concessionaire provides an outlet for these ill-gotten (or at least 'criminalized') goods, the likelihood of extensive forest degradation will vary with the costs of transport to willing buyers. Where concessionaires are hurt financially by these activities, such as where they jeopardize certification, vigilance can help. Market hunting by local people is generally harder to suppress than market-oriented illegal logging because of the higher value per weight of meat, pelts, etc. (e.g. casques from hornbills or gall bladders from bears), but can be controlled to a substantial degree.

Conscientious or practical concessionaires who invest in the wellbeing of the long-term residents inside their concession boundaries risk doing too little or too much. As beneficiaries of governmental failures to recognize traditional territorial rights, they often suffer condemnation. If concession-based forest industries are to survive – which some might not applaud (e.g. Kelly and Peluso 2015) – concessionaires need to find ways to assure compliance with agreed-upon guidelines or legal restrictions and, hopefully at the same time, contribute to social welfare while not creating a magnet for additional colonists.

3 Trajectory II: Adoption of reduced-impact logging

The first big step that still needs to be taken toward SFM in much of the tropics is the minimization of the deleterious environmental impacts of selective timber harvests through the application of RIL techniques (Figure 5). Regardless of the silvicultural approach applied, protection of soil from compaction, streams from excessive sediment loads, future crop trees (if any) from damage and workers from unnecessary risks are all important (see Putz et al. 2008). As expected, use of RIL results in substantial increases in retention of carbon (e.g. Griscom et al. 2014) and biodiversity (e.g. Bicknell et al. 2014). RIL guidelines vary somewhat in the practices that they recommend and in their inclusiveness, but other than for forest managed by single tree selection with a polycyclic (i.e. uneven-aged) approach, they fall short of fully describing a silvicultural treatment.

Figure 5. A felling gap filled with natural regeneration of commercial tree species. Nine months before the harvest, the inventory crew cut the lianas on the trees to be felled, which reduced incidental harvest damage and limited post-logging liana proliferation (Photo by V Medjibe).

3.1 What is RIL and what should it become?

While not questioning the fundamental goals of RIL, given the variety of silvicultural approaches needed for the wide range of tropical forest species and conditions, it would have been clearer if the acronym was RUIL, for 'reduced *undesirable* impacts of logging.' This modification would clarify that in some cases, substantial impacts of logging are silviculturally desirable. For example, where the objective of a silvicultural intervention is regeneration of light-demanding tree species, slavish attention to traditional RIL renders the intervention no better than high-grading (Fredericksen and Putz 2003). Instead of trying to minimize stand damage, special efforts might be required to achieve the desired amount of canopy opening and skidder drivers might be encouraged to scarify the soil surface in logging gaps to expose mineral soil. The root of the problem is that most RIL guidelines were devised with single tree selection in mind whereas that approach is inappropriate for many stands in the tropics.

In most RIL research and promotional campaigns, emphasis has been on appropriate use of ground-based extraction devices, generally either bulldozers (= crawler tractors) or skidders with rubber tires. These machines will undoubtedly continue to be important in tropical forestry but other yarding devices should be more utilized in the future for environmental as well as social and economic reasons. Motivation for this change comes from the increasingly adverse terrain where logging happens in the tropics, as well as from the increasing importance of loggers who lack the capital needed to purchase and operate large machines.

A small amount of forest clearing is unavoidable where timber is harvested commercially, but that portion can often be reduced with good planning and strict adherence to environmental guidelines (e.g. Pinard et al. 2000). To harvest timber, roads are needed, but the lengths and widths of those roads can often be reduced. Road lengths can be minimized through proper engineering but capturing that benefit requires accurate topographic maps and qualified engineers who go to the forest to ground truth their planned road alignments. Until topographic maps are available that are based on crown-penetrating light-detection and ranging radar (e.g. LiDAR) imagery, which should be a priority, reliance on maps constructed with aerial photographs or digital elevation models generated using passive remote sensing (e.g. Landsat imagery) will not allow optimization of road alignments (see Section 3.2).

In addition to minimizing road lengths through proper planning, deforestation for logging roads can be reduced by clearing less forest along their paths. However, these reductions often come at a financial cost because the working surfaces of roads bordered by wide forest clearings dry faster than those where adjacent forest is left standing (see Figure 6). Trafficability is reduced when road surfaces are wet, and financial and environmental costs escalate if wet roads are used due to the subsequent need for costly repairs. Down the product chain, inability to deliver logs to mills due to impassable roads has ripple effects that can be quite severe. To achieve the same trafficability with narrow road corridors, improvements are needed in road engineering, construction and maintenance. Roads with domed and graveled working surfaces, for example, dry quickly and become passable faster than where those practices are not implemented. However, the direct financial costs of construction and maintenance of such roads are high even if the longer term and more inclusive economics of those investments are favorable. For example, hauling gravel substantial distances is costly, but those costs might be recovered if road use is thereby increased.

Deforestation for log landings (i.e. cleared areas along main roads where logs are temporarily stored until they are hauled to mills, log ponds or central log yards) – another primary impact of forestry activities on forest cover – can often be reduced through good planning and changes in harvesting practices. If roads remain trafficable and hauling and yarding activities are coordinated, it is often possible to substitute small roadside clearings for extensive log landings (e.g. Pinard et al. 1995). Forest clearing for other infrastructure such as logging camps, workshops and playing fields is also a primary effect of logging that is attributable to forest industries. If forest owners realize that due mostly to soil damage, these intensively used areas will not likely produce timber in the foreseeable future, perhaps they will be motivated to improve planning to limit their extent.

To minimize undesirable impacts of logging on residual stands, soils and hydrological functions, vehicular traffic should be minimized. This objective only increases in importance as logging is relegated to sites with thin soils, steep slopes and wetlands. Improved road layouts and better engineering are perhaps the most important changes needed, but where the focus is on timber yarding, the use of cable winching needs to increase. To a large extent, yarding damage can be reduced if rather than driving up to each stump, yarder operators maximized the use of the winch with which their

A B

Figure 6. The contrast between a responsibly constructed logging road (A) and an overly wide road of the same order (B). Due to lack of direct sun on the narrow road, it was engineered and constructed to increase drying and maintain trafficability (e.g. domed and graveled) (Photo by G Wilkinson).

machines are equipped. Winch pulls of >20 m need to become the rule rather than the exception. The switch from bulldozers to tracked skidders with more centrally mounted winches (e.g. Caterpillar 527 Track Skidder) will serve to decrease unwanted damage. Because CAT 527 Track Skidders are substantially lighter than the D-7s they commonly replace, soil compaction is also reduced and they are also less expensive to purchase and operate. Furthermore, winching distances with tracked skidders can easily average >20 m.

Where slopes exceed 20% or where soil trafficability is poor and erosion proneness is high, yarding devices with much longer haulback capacities than bulldozers or skidders need to be utilized. In areas with very high intensity harvests or clearcuts and yarding distances of <300 m, small-scale skyline yarders are appropriate and cost-effective. With skyline systems, a carriage moves along a mainline cable that runs from some sort of tower mounted on a moveable platform (e.g. a modified excavator) to an anchor point, often the base of a large tree. Such devices are often known in North America as "yoaders," and vary in tower heights and yarding capacities. For more selective harvests, the need to frequently move and anchor the mainline cable reduces the efficiency of these skyline systems and high-lead approaches become more attractive.

Among the environmentally concerned, high-lead logging has a terrible reputation derived from the use of huge and mostly stationary towers, and high intensities of harvesting. The sort of device being recommended here is much smaller and more mobile. The Log Fishers and Rimbaka machines sometimes used in Malaysia are good examples of this technology (e.g. Norizah et al. 2012). Basically they are excavators with rear mounted winches from which 100–200 m of cable runs over a pulley at the top of the bucket arm. With only a single drum, the cable needs to be pulled out manually for each log yarded, but long haulbacks are possible. To shorten the pulls and to minimize soil damage from the logs that are dragged along the ground, the excavator-yarder moves frequently along a ridgeline road or skid trail. Attaching a skidding pan or cone to the front of logs to be winched can also reduce hang-ups while reducing the likelihood of cable breakage. Efficiency is reduced marginally if the excavator-yarder needs to be guy-wired for stability each time it stops to yard a log, but this operation is generally not necessary.

In the absence of the financial capital needed to own and operate large yarding equipment, small, sled-mounted cable yarders can be used. Such devices, employed by many illegal loggers in Kalimantan, consist of a diesel engine (18-25 HP), a truck transmission and a winch pulley with 100 m of cable mounted on a 1.5 x 2 m welded-steel sled. Steam powered versions of this sort of device were used in Europe and North America as much as 150 years ago. The models being made in Indonesia cost about USD 6000 to construct and use very little fuel but are capable of yarding only about 10–20 m^3 per day. To equal the productivity of a D-7 crew of four workers (chainsaw operator and assistant plus the dozer driver and assistant) requires six monocable winches with six workers each. Where labor is not limited and having that many people in the forest is not a concern (e.g. safety issues and potential poaching problems), monocable winches should be considered.

For RIL to contribute the most it can to SFM, well-trained workers are essential. Unfortunately, due to the transience of many in the forestry work force, training needs are continuous. If working conditions and salaries were improved, this transience might be reduced. In any event, investment in forest worker training pays off in terms of post-logging forest conditions and rates of stand recovery as well as worker safety.

3.2 Logging on steep slopes

With increases in global human populations and *per capita* consumption, demands for food, fuel and fiber increase and more natural forests are cleared. While every effort should be made to satisfy these demands through enhanced productivity per unit land area, improved crop storage and handling, and more equitable distribution, the on-going expansion of cropland is likely to continue. Where arable lands on level ground are used for crop production, production forestry will be increasingly relegated to poorly drained areas and onto steeper slopes. Here we focus on the latter and address some of the consequences of different limits on slope angles and lengths. The issues related to steep slope logging are pertinent to varying but generally increasing degrees throughout the tropics as natural forest management is relegated to sites with more and more adverse conditions.

Slopes affect many physical processes due to gravitation (e.g. rates of surface water flow) and trigonometric factors (e.g. per unit ground surface area rates of incoming fluxes of precipitation and solar radiation). While attention often focuses on slope angles, slope lengths also influence many biophysical processes, as captured in the "universal soil loss equation" (e.g. Sidle et al. 2006). With increased slope angles and lengths, soil erosion and mass wasting events (i.e. slumps, landslips and landslides) become an increased concern, especially after trees are cut and soil-binding roots decay. With increased slopes, soil depths typically decrease, rock outcrops become more common, forests are more dynamic (e.g. treefall rates increase) and vegetation communities change in both structure and composition (e.g. Ferry et al. 2010).

In addition to concerns about increased environmental damage with increased terrain slope, the costs of ground-based timber yarding increase with slope, which reduces the productivity and profitability of logging. For example, loaded log trucks and tracked skidders (i.e. bulldozers) often have trouble climbing grades of >5% and 15%, respectively. To access steeper terrain, switchbacks are needed (see Figure 7), which increases operational costs as well as environmental damage. Alternatively, timber harvesters can switch from ground-based yarding with bulldozers or skidders to long-line cable systems that keep heavy equipment off steep slopes.

In recognition of the effects of slopes on the impacts of land-use activities, many environmental guidelines set by governmental and non-governmental bodies try to address this. Unfortunately, many fail to indicate the slope lengths or the areas over which slopes should be measured, which renders them ineffective because average slopes decrease with the area over which they are averaged. This means that slopes measured with digital elevation models based on remote sensing always decline with increasing size of the pixels used. Disregard of this pixel problem is evident in numerous publications and policies, of which we highlight only one. In an otherwise excellent paper by Austin et al. (2015), the authors conclude that oil palm plantation operational costs in Kalimantan are not influenced by slope probably because their digital elevation model was based on 250 m pixels. Another operational problem is that slopes measured by field crews or with LiDAR are steeper than those based on passive remote sensing images (e.g. Landsat or ASTER) at least partially because the crown surface is less topographically dissected than the ground surface.

Figure 7. Switchbacks cut with a D-7 bulldozer up a steep hillside in Sabah to access timber (Photo by M Pinard).

In Europe and North America, logging is carried out regardless of slope angle but with strict controls on erosion. Loggers employ a variety of log yarding methods to limit erosion and to thereby avoid government sanction. Admittedly, some of the techniques they use (e.g. skyline yarding systems) are financially viable only where harvest intensities are high, and work better in clearcuts than in selectively logged forests. In contrast, long-line (150–250 m) cable yarding systems with mobile towers have been shown to be effective in the tropics (Norizah et al. 2012).

3.3 Logging and other forest management activities in swamps

A great deal of attention has been paid of late to the forested wetlands of Indonesia and Malaysia, particularly peat swamps. This attention is warranted given the rapid rate at which they are being cleared and degraded, which leads to massive carbon emissions and smoke plumes from soil fires that greatly affect life in the entire region (e.g. Miettinen et al. 2012; Marlier et al. 2015). While clearing and draining peat swamps deserves attention, as does their restoration where degraded (e.g. Page et al. 2009), research on their silvicultural management has dwindled while logging has not (Miettinen and Liew 2010). Although Malaysia in particular has a long history of peat swamp silvicultural research (e.g. Wyatt-Smith 1963; Bruenig 1996), little use seems to be made of that research, which deserves to be extended and modernized. It is still debated, for example, how swamp timber yields might be sustained and logs extracted without major hydrological disruption.

Floodplain logging in the Amazon Basin, which is often carried out by small operators, is fairly well understood – policy challenges notwithstanding (e.g. Schöngart 2008; Fortini et al. 2015). Research on topics beyond just logging is needed to assure both sustainability and that other values (e.g. fish stocks) are not unnecessarily compromised where timber production is the principal goal of management.

4 Trajectory III: Silvicultural intensification in natural forests

Despite the diversity of tropical silvicultural systems described in the literature (e.g. Gunter et al. 2011), the results of the well-publicized experiments on shelterwoods (e.g. Hutchinson 1988), strip clearcuts (e.g. Hartshorn 1989), liberation thinning (e.g. Hutchinson 1981; Wadsworth and Zweede 2006) and other silvicultural treatments are seldom applied at operational scales by forest industries. Instead, when enough trees of commercial species reach harvestable size, they are selectively felled, with or without governmental approval. To glorify this mining of timber as “silviculture” disgraces the term. In the few examples where post-logging silvicultural treatments have been applied beyond the confines of experimental plots, the costs were generally borne by external agencies including the United States Agency for International Development (USAID; e.g. Villegas et al. 2009), the Food and Agriculture Organization of the United Nations (FAO; e.g. Hutchinson 1981), New England Power (e.g. Pinard and Putz 1996) and the Forests Absorbing Carbon dioxide Emissions (FACE) Foundation (e.g. Pinso and Moura-Costa 1993). When the subsidies stopped, so did the silviculture. One exception to this finding was in Queensland, Australia, where industrial logging firms eventually complied with fairly rigid RIL guidelines. However, that ended when the country’s tropical forests were taken out of production. The emphasis on RIL by some Forest Stewardship Council (FSC) auditors and government agencies should also be acknowledged. In this section we describe a trajectory towards silvicultural interventions beyond RIL that stops short of natural forest conversion into plantations of exotic species.

4.1 Motivation for silvicultural intensification

Under some conditions, declining revenues from timber exploitation after the first harvest can motivate forestry firms to intensify management. Especially where there are alternative land uses or income sources, which is mostly everywhere, for-profit firms are unlikely to reduce harvest intensities and allow longer periods for forests to recover before the next planned harvest. Such restraint would be to the long-term best interest of society and especially of future generations, but is seldom evident. If management intensification is the exercised option, there are many alternatives but tree planting is usually preferred (e.g. Sasaki et al. 2011). This preference is more an expression of the dominance of an agronomic approach to forest management than a response to lack of natural regeneration that might be cost-effectively encouraged. All-too-often this abundant natural regeneration is not recognized because the responsible parties spend too little time in the field or are not adequately trained in dendrology.

4.2 Silvicultural intensification through enrichment planting

In the field of industrial-scale silviculture in natural tropical forests, the work of the concessionaire Sari Bumi Kusuma (SBK) in Central Kalimantan is somewhat unique (Putz and Ruslandi 2015). SBK applies intensive silvicultural treatments but is not subsidized by an outside party. In response to governmental policies and recognition of short-falls in natural regeneration, SBK employs RIL techniques but then follows the second episode of selective logging with enrichment planting of seedlings of commercial timber species along cleared lines through the harvested forest. This approach to forest management is generally referred to as SILIN (Sistem Tebang Pilih Tanam Intensif Indonesia) or TPTJ (Sistem Tebang Pilih Tanam Jalur).

Methodological details of SILIN have evolved over time (e.g. planting line clearance width and rules, inter-plant and inter-line spacing, and species selection) but as of late 2014, SBK had line-planted 49,000 ha (Figure 8). The company's intention is to plant an additional 27,000 ha over the next few years. Another large portion of the concession (70,000 ha), which is mostly classified as Limited Production Forest, is allocated for TPTI, which is basically RIL without line planting. Note that although its name includes planting (*tanam*), this component of TPTI has been abandoned. Large areas with difficult access and steep slopes (>25%) were allocated to TPTI by SBK with the remainder allocated for nature conservation (12,000 ha) and local community use (13,000 ha).

Given the great variation in what passes for RIL, it is important to provide some details about SBK's approach. The company carries out 100% stock mapping of trees >40 cm DBH and produces its own topographic and stock maps based on data collected by its inventory crews. Based on these maps, roads and skid trails are pre-planned.

Figure 8. Enrichment planting along cleared lines through twice logged forest. The planted trees (*Shorea leprosula*) were 15 years old and mostly 25–35 cm diameter at breast height (DBH) when this photo was taken (Photo by Ruslandi).

Lianas are not common in unlogged forest in SBK but if they are in the crowns of trees to be harvested, inventory crews reportedly cut them. For the last several years, log yarding has been principally with tracked skidders (CAT 527).

One important recent change in Indonesian forest policy is abandonment of the idea of a minimum cutting cycle. As mentioned earlier, in many other countries where logging is legal, logging companies seek governmental permission to re-enter stands before termination of the government-mandated minimum cutting cycle. Elsewhere, pre-mature re-entry logging occurs widely both with and without governmental approval. The difference is that in Indonesia, pre-mature re-entry is now codified by a regulation that calls for 100% timber stock inventories at 10-year intervals. Based on inventory results, any stands with >40 m^3/ha of commercial timber can legally be re-entered. Re-entry logging is attractive not due to post-logging volume increments but mostly due to changes in markets that render it profitable to harvest smaller trees (down from 50 cm to 40 cm DBH in Indonesia), trees with stem qualities not acceptable at the time of the previous harvest and acceptance of new species. Also, re-entry is generally cheaper because logging roads have already been constructed. In relation to this topic, research is needed on the impacts of re-use of previously opened skid trails when logged stands are harvested for the second and third times.

Application of SILIN requires a great deal of money as well as a dedicated and motivated staff. SBK has so far invested about USD 30 million in SILIN (SBK 2014). Each year, 650 SBK staff members clear 3 m-wide planting lines spaced 20 m apart, dig 30 x 30 x 30 cm holes at 5 m intervals along those lines, and plant nursery-grown seedlings in compost inoculated with ectomycorrhizae. Planted seedlings are then tended annually for 3 years so as to keep them liana free and to reduce shade from encroaching plants. SILIN guidelines call for a pre-commercial thinning of planted trees at 5–10 years but this treatment has not yet been implemented.

The medium-term silvicultural results of SILIN, as applied in SBK, are impressive, with seedling survival rates of about 70% and rates of stem diameter increment consistently averaging about 2 cm per annum, based on 10–14 years of post-planting data from permanent sample plots (e.g. Pamoengkas et al. 2014; Inada et al. 2015). Although Kusuma et al. (2014) predicted that at the end of the planned 25 year rotation, harvestable timber will amount to 235 m^3/ha, recent field studies indicate that yields will be about half of that amount (Nitikusuma et al. 2015). Given that the first cut in SBK's forests typically yielded 60–70 m^3/ha and the second cut, just prior to planting, yielded only 40 m^3/ha, the projected amount of timber to be harvested at the end of the 25-year rotation is a bit unsettling. At these rates, standing stocks of commercial timber at the end of the planned rotation will be much greater than before the first harvest. In other words, yields could be much more than sustained, which might be profitable but is nevertheless quite worrisome when potential logging impacts are considered.

Concerns about the likely impacts of a harvest of all trees >40 cm DBH in stands subjected to SILIN – which is permitted under current regulations – derive from studies of RIL. In particular, based on a study in East Kalimantan, Sist et al. (1998) reported that the benefits of RIL relative to CL disappear if harvests exceed 8 trees or 60–80 m^3/ha. With harvestable tree densities and volumes much higher than these limits, if all eligible trees are harvested the results will resemble clearcuts. If that were to happen where enrichment planting is successful, the huge profits would be at the expense of biodiversity and other environmental benefits of retention of natural forest between the planted strips. It appears that forest managers and forestry officials in Indonesia, as well as forest auditors for certification programs, are not fully cognizant of the consequences of the shift from polycyclic to monocyclic stand management (Ruslandi et al. 2014a).

Recent studies conducted in SILIN areas in SBK reveal some reasons to question estimates of available commercial volumes of timber when the planted stands reach the minimum rotation age of 25 years and shed light on potential harvesting challenges. Given the global push for restoration of degraded forests (e.g. Suding et al. 2015) and reawakening interest in enrichment planting (e.g. Ghana Forestry Commission 2013), it seems worthwhile to review these findings here. In one study that compared the crown architectures of line-planted and naturally regenerated *Shorea leprosula* trees, Hardiyansyah et al. (2015) found that the clear boles of 15-year-old planted trees 20–40 cm DBH averaged 13% shorter than naturally regenerated but substantially older trees of the same size growing in the bands of logged-over forest between the planting lines. If low branches are maintained until the harvest, commercial timber yields will be reduced by approximately that amount. But even if those large branches are shed before the harvest, the value of the upper logs will be reduced due to the presence of large knots. In a related study in the same stands, Nitijusuma et al. (2015) explored the frequently discussed option of restricting the harvest to planted trees and felling them down the planting line to reduce or at least concentrate stand damage. Based on stem and crown measurement complemented by the opinions of directional felling experts, they concluded that 91% of the planted trees could be felled in the direction of the planting line. This option seems less attractive in light of the results of a simple growth projection model (i.e. linear projection of average growth over the first 15 years after planting to the end of the 25 year rotation) that revealed that only half of the planted trees will have reached the minimum cutting diameter (40 cm DBH). If permission to harvest trees <40 cm DBH could be obtained from the government, then the lines could be clearcut. But given that the planted trees at 30–40 cm DBH grew >1 cm DBH/year for 25 years, and would presumably grow even faster after their larger neighbors are removed, a selective harvest of just the larger planted trees would seem advisable. Exactly how that harvest might be carried out is not clear, but the authors recommended cable-yarding instead of a ground-based timber harvest.

For industrial-scale enrichment planting, the initial challenge to develop an effective planting and tending system was addressed adequately by John Wyatt-Smith in 1963 for Malaysia and then more thoroughly in 1966 by Colyear Dawkins for Ghana (see Dawkins and Philips 1998). Now, with ample evidence that enrichment planting along cleared lines with dipterocarps can work in terms of timber volume increments, it is time to consider design modifications that enhance long-term sustainability and biodiversity maintenance. In particular, stand establishment practices should be designed with future harvests in mind. For example, if the intention is to selectively log enriched stands and not clear fell them (i.e. to retain a polycyclic approach and not switch to a monocyclic one), then perhaps the density and spatial patterns of planting should be adjusted accordingly. Instead of planting 80–100 trees/ha of one species or a few species that grow at similar rates, mixtures of species could be planted that will mature at different times and yield timber of different qualities and values. For instance, fast-growing species that produce timber used mostly for utility-grade plywood might be mixed with slower-growing species with more highly valued timber used for cabinets, flooring and naturally rot-resistant patio furniture. The spatial patterns of these mixtures should reflect topography, accessibility, protected buffer zones and edaphic factors as well as planned harvesting technologies. To promote wildlife in enrichment planted stands, a proportion of the planted trees might be species that produce fleshy fruits, some of which also produce commercial timber. This particular approach to biodiversity impact mitigation was utilized in the Forests Absorbing Carbon Emissions (FACE) Foundation of the Netherlands-funded enrichment planting project in Sabah (Moura-Costa et al. 1996).

Where swidden farmers threaten to clear production forests, enrichment planting may reduce the threat of land colonization and forest clearance for cultural reasons. At least among some indigenous groups in Kalimantan, the planting of trees confers ownership on the planter (Dove 2011). The extent to which this tradition applies to tree planters in industrial forestry concessions remains to be determined but seems like a reasonable possibility. And if it is the case, forest managers will have another reason to concentrate their intensive management activities within the proximity of roads so as to reduce management costs while simultaneously reducing the risk of forest conversion.

There are good reasons to question the economics of high intensity enrichment planting but it is useful that there are some industrial scale examples that can be evaluated. In the areas managed by SILIN in the SBK concession, for example, will the investment of USD 700/ha required for the first 3 years of treatment (i.e. line clearing, planting and tending) pay off when the stands are harvested 25 years later? Might it be more profitable if instead of fast-growing dipterocarps (e.g. *Shorea leprosula, S. parvifolia, and S. johorensis*), SBK planted slower-growing species with higher-valued timber, like bangkirai (*S. laevis*)? What about investing in the liberation of natural regeneration rather than planting nursery stock at set intervals? Training forest workers in seedling identification would be required, as would a reward system structured so that compensation is provided for locating and liberating natural regeneration. However, these costs seem small given the higher likelihood of survival of naturally regenerated trees and the biodiversity benefits they provide.

While enrichment planting may be justified in some areas in logged-over or otherwise degraded forests (e.g. Omeja et al. 2011), given the costs and major environmental impacts of this silvicultural intervention, it should not be carried out everywhere (Sasaki and Putz 2009; Sasaki et al. 2011). In recognition of the need for some level of site capability differentiation, SBK logs but does not apply SILIN in areas that are generally steep. In areas designated for SILIN, in contrast, current regulations exempt only riparian areas from line clearing and planting. On the basis of the higher management costs (i.e. planting, tending and harvesting) on steep slopes >50 m long as well as the exacerbated environmental damage, the professional foresters in the concession are considering exemption of these areas from planting as well.

The well-intentioned requirement of the Indonesian Ministry of Forestry that logged stands be enrichment planted provides another example of how legislated silviculture that focuses on a single commodity might lead to avoidable environmental damage and economic inefficiencies (Messier et al. 2014). What restrictions on harvesting should be applied at the end of the rotation if there are 50 or more harvestable trees per hectare? Taking them all at once would constitute a clearcut, but which trees should be harvested and how? Given that the planted trees start reproducing when 15–20 years old, there should be plenty of regeneration on the ground when the stands are harvested at 25 years. What about applying a shelterwood harvest to avoid excessive stand damage and the cost of replanting (e.g. Ashton et al. 2011)? Might the timber be harvested using long-line cable techniques rather than ground-based skidders? Should existing skid trails be re-used, replanted with extremely fast-growing species that will be available for harvest at the time of the next entry (e.g. *Anthocephalus chinensis*), or abandoned?

The principal barrier to intensification along the lines of SILIN is financial. For reasons that are not clear, governments provide little financial support for forest management but instead often enact regulations that provide further impediments. For example, high import duties on logging machinery (e.g. cable yarders) render their purchase unnecessarily difficult. Long-line cable yarders can be fashioned from modified excavators or from constructed towers mounted on tractor bodies but there are still costs involved in the change in technology. Some of those costs might be recouped if the yarding proves more efficient than traditional approaches (e.g. Norizah et al. 2012) but there are still start-up costs. Private banks are reportedly also reluctant to lend money to forest industries. Although all concessions in Indonesia contribute USD 16/m^3 harvested to the "reforestation fund" (*Danai Reforestasi*), none of that money is returned to them for reforestation or implementation of SILIN. And despite the widely held hopes of funding for improved forest management from REDD+ and other payments for ecosystem services, it seems unlikely that those funds will be forthcoming given the common association of tropical forest industries with forest degradation and deforestation (e.g. Didham 2011; Zimmerman and Kormos 2012). Finally, governmental failures to protect concessions from corruption add substantially to the costs-of-doing business in the forestry sector.

In some areas the survival of tropical production forests and forest industries will require silvicultural interventions starting but not ending with RIL. But whatever landscape mixture of silviculture approaches prove most financially and environmentally sound, economic efficiency is critical. For example, that manufactured flooring made by timber industries in Java uses red oak imported from North America as facing veneer is no reason to abandon managed natural forests in Indonesia as a source of raw materials.

5 Conclusions: Toward management and away from exploitation

Whether tropical production forests start to be responsibly managed or continue to be exploited until they are replaced by some other land use will continue to be influenced by economic development and associated improvements in governance as well as with the identities of the responsible and affected parties. At least over the short-term and where population densities are low, poverty, political instability, social conflict, non-democratic regimes, smuggling of drugs and other contraband, and poor infrastructure will limit forestry activities to exploitation of the most valuable species but will also serve to protect forests from large-scale conversion because they increase financial risks and cause capital constraints (e.g. Price 2003; Larjavaara 2012; but see McSweeney et al. 2014). Under higher population pressure or on dynamic demographic frontiers, in contrast, these same factors can promote large-scale forest degradation and deforestation from a multitude of small-scale events.

What is clear is that land-use decisions need to be informed by knowledge about the tradeoffs between the financial benefits of forest management intensification and the associated costs in biodiversity and other natural and social forest values (Rudel and Meyfroidt 2014). Devolution of control over forestlands to rural communities in the tropics seems likely to accelerate (e.g. Agrawal et al. 2008; Pokorny and Johnson 2008; Bowler et al. 2011) but it is not clear whether this power shift will change the fates of many production forests. Rates of large-scale conversion may decline, at least as long as these communities remain impoverished, poorly organized, and beset with land tenure problems and governance failures (e.g. Börner et al. 2010). Under these conditions, logging – legal or otherwise – is the most likely land-use. Under some conditions, payments for environmental services, including carbon sequestration and certification of products from well-managed forests, could tip the balance toward forest conservation if their implementation effectively thwarts governance failures and avoids land grabbing by outsiders to capture these novel rents (e.g. Zoomers 2010; Cuffaro and Hallam 2011). When forest-controlling rural communities accrue financial and institutional capital, such payments could steer communities away from forest exploitation and conversion and towards responsible forest management. The likelihood of this trend will depend on the value members of these communities place on forests and will be subject to change in response to economic opportunities and environmental education (Coomes et al. 2008; Pfund et al. 2011; Meijaard et al. 2013). It will help if one explicit goal of company-community-government partnerships is production without destruction.

To inform decisions about the landscape level distribution of silvicultural prescriptions where environmental and economic concerns loom large, much more needs to be learned about the consequences of interventions other than just ground-based selective logging with RIL. It is already well established that the deleterious impacts of logging increase with harvest intensities (e.g. Burivalova et al. 2014) and decrease where RIL techniques are employed (e.g. Bicknell et al. 2014), but what if instead of skidding the logs out along the ground with big tractors, they were cable yarded? To what extent would that change in harvesting practices mitigate the deleterious environmental impacts of logging, at what financial cost (or benefit) to the loggers, and at what costs or benefits to forest workers (e.g. safety risks)? When silvicultural options other than just logging are considered and those options are distributed in different patterns and intensities across forested landscapes, the number of issues in need of research increases rapidly. For example, what are the financial and environmental costs and benefits of liberation of future crop trees, of soil scarification in gaps to promote regeneration and of shelterwood harvests or group selection rather than single tree selection? Equally important, what combinations of penalties and rewards will most effectively promote adoption of these practices?

While research on tropical ecology has flourished, tropical forestry research has remained in a backwater. Plenty of funding is available to address the question of why there are so many tree species in the tropics but comparatively little on how for-profit management interventions might be designed to retain that diversity. Too many tropical foresters remain focused entirely on the biophysical impacts of logging and too much of their research is carried in plots that are too small to assess the financial aspects of different interventions. The network of Long-Term Silvicultural Research Program (LTSRP) plots maintained by the Instituto Boliviano de Investigación Forestal (IBIF)[3] is certainly not perfect, but might serve as a model for similar efforts elsewhere in the tropics. Those plots, which were established only 15 years ago, have already yielded more than a dozen research publications in reviewed journals, one of the most recent of which was authored by Corrià-Ainslie et al. (2015). At their La Chonta site, IBIF maintains three blocks of four 27 ha plots each, with three plots per block receiving a different silvicultural treatment and one reserved as a control. Despite the scientific productivity of this international effort, IBIF's funding remains a small fraction of what is spent annually on even one 50 ha plot in unlogged forest in the Center for Tropical Forest Science - Forest Global Earth Observatories (CTFS-ForestGEO) network.[4] Reducing this disparity in funding and promoting collaborative research in LTSRP-like plots around the tropics will require leadership of an organization like the Center for International Forestry Research to overcome donor resistance. The Tropical managed Forest Observatory (TmFO; Sist et al. 2014b) represents a laudable effort to retrieve and utilize data from established sample plots. Unfortunately, many of the plots are small and were designed with only timber in mind, which is no longer the only factor of concern.

3 http://www.ibifbolivia.org.bo

4 http://www.forestgeo.si.edu

The tradeoffs between timber yields and biodiversity, carbon sequestration, hydrological functions and other valued ecosystem properties/services, and conditions for forest workers are unlikely to be simple and consistent among different geographies. The tradeoffs are also likely to vary from the perspectives of the wide variety of stakeholders with aspirations and entitlements or just concerns. It is this complexity, as it plays out over time, with which forest researchers need to grapple. To address the complexity of natural forest management in the tropics (see Section 5.4), researchers need to embrace the full diversity of goals, constraints and agents. As Francis Crome wrote nearly 20 years ago about over-simplified analytical methods, we need to wean ourselves from approaches suitable only for "tame, toy problems" if we mean to address "wicked real" ones (Crome 1997, 490).

5.1 Governmental disincentives, inadvertent and otherwise

Complicated government bureaucracies and consequently large administrative burdens thwart many efforts in responsible forest management. The profit margins of large forestry corporations are lowered by these encumbrances but community-based forest management efforts are often completely stymied by the same (e.g. Hirakuri 2003). Streamlining of administrative requirements for forestry, as through the Small or Low Intensity Managed Forest (SLIMF) program of the FSC, might help a great deal. Unfortunately, many well-intended attempts at reducing regulatory burdens end up promoting illegal harvesting by providing opportunities to launder illegally harvested logs. Learning from these experiences is critical lest the same mistakes be made in new efforts to streamline regulations to make forest management a more attractive land-use option.

The administrative requirements for timber harvest and sale are often much more cumbersome than they are for agriculture. In addition to the direct costs of compliance, each regulation provides an additional opportunity for graft. For small rural forestry firms, private or community-based, one big direct cost of legality is often the requirement to make repeated trips to governmental offices to satisfy administrative requirements. For any sort of firm, the illegal payments to governmental officials at every step along the way toward fully legal operations can substantially increase the cost of doing business. Failure to pay the expected amounts to the right people and in the proper way can lead to costly delays.

Whether the determiners of forest fates are rural people, government functionaries or board members of multinational corporations, management intensities will likely continue to vary with market demands, availability of financial capital and labor, security of property rights, site capabilities, accessibility and the associated costs of management, and cultural preferences (e.g. Rudel et al. 2002). Alternatives to environmentally destructive management intensification on lands spared from agricultural conversion will only become likely if there is recognition of the variety of possible interventions in areas classified as forested. It will also help if the local, regional and global benefits of natural forests are taken into account when decisions are made about land-use intensification (e.g. Oliveira et al. 2013).

5.2 Silviculture at appropriate temporal and spatial scales

Among the many challenges facing tropical forest managers, tree diversity looms large. Phenomenally high numbers of species that differ in growth requirements, growth rates, marketability, ecological roles and other relevant traits means that simple silvicultural guidelines are unlikely to be satisfactory (e.g. Schöngart 2008). For example, setting a single minimum cutting diameter or forest-wide cutting cycles unavoidably selects for some species over others and thereby can reduce potential profits from management. The application of a single silvicultural system to a diversity of stand types over a range of terrains is likewise inefficient and ineffective. Rectifying this problem will require the insights of people experienced with forest policies with insights about how to avoid the misuse of even the best intentioned of rule changes.

When attention shifts from hectare or stand-level phenomena to landscapes, results that were reasonable at small scales often lose their relevance. Unfortunately, this shift in focus is woefully slow in coming even in the scientific community, but policy making falls even further behind. Continued faith in single tree selection, with natural gap-phase regeneration is a case-in-point. While no one would argue with the benefits of RUIL (e.g. protection of soils and hydrological functions), prohibition of any but the gentlest of silvicultural interventions is inimical to the regeneration of many commercially valuable species. A 0.5 ha patch clearcut at least initially looks terrible at the 1 ha scale but if regeneration of prime timbers is thereby enhanced, such a treatment might be entirely justified at the landscape level. Even more intensive forms of forest domestication, such as enrichment planting, might be justified for accessible areas with suitable soils and terrain. Such justifications for intensive silvicultural interventions are predicated on the existence of lucrative alternative land uses (e.g. cattle pastures or oil palm plantations) and therefore large opportunity costs for forest retention. Unfortunately, well-intentioned national laws or certification rules that emphasize retention of alpha-level (i.e. within stands) biodiversity (e.g. Mexico) often prohibit intensification. Furthermore, to remote sensors, patch clearings to promote regeneration of light-demanding species will look like deforestation and therefore be condemned. Where the alternative is forest degradation by depletion of commercial timber stocks and therefore increased likelihood of conversion of the value-depleted forests, those intensive interventions seem quite acceptable from an environmental perspective.

5.3 Management of tropical forests as complex adaptive systems

Given the complexity of tropical forests and forestry in our rapidly changing world, government-mandated, one-size-fits-all regulatory approaches focused on single commodities (e.g. timber) will remain difficult to implement, economically inefficient, and socially and environmentally unsatisfactory.

That said, calls for more flexible and adaptable multi-objective approaches to forest management with less dependence on top-down, command-and-control approaches (e.g. Putz and Romero 2012; Filotas et al. 2014; Messier et al. 2014) need to be coupled with step-by-step guidelines for the transition from the current status of forestry in much of the tropics. These guidelines need to be based on theories-of-change that are tailored for different forest owners and operators who work under different socioecological, political and economic conditions. In some cases, self-policing might work but this is not the case for the vast majority of the remaining tropical forests of the world that are subjected to exploitation, management or destruction.

It is important to recognize that the goals of multiple-objective management need not be met in each and every stand (e.g. Ashton et al. 2011; Putz 2013). For example, there are conditions under which it would not be economically viable nor environmentally strategic to push for a switch from intensively managed, short-rotation monocultures for wood fiber production toward mixed species and multiple-objective management. With rotations of <10 years, the diversity-provides-resilience argument is not very convincing; switches in species, provenances or genotypes after each harvest is often a more attractive way to deal with the threats than stand diversification. In contrast, it is always appropriate to insist on protection of riparian areas and application of other RUIL techniques. At the other end of the forest management intensity continuum, in a single semi-natural forest there might be stands where the principal objectives might be biodiversity protection (e.g. near salt-licks), maintenance of ecosystem functions (e.g. riparian buffer zones), timber production, and mixed-management for timber and non-timber forest products. In the stands where timber production is the principal objective and management is by a well-capitalized firm with trained staff, single tree selection with long-line cable yarding might be prescribed for steep slopes, shelterwood management for more gentle terrain where the light-demanding commercial tree species are well represented in the overstory, and intensive harvesting followed by enrichment planting on similarly gentle terrain in accessible areas where natural regeneration is not already present or easily secured. Such prescriptions would clearly not be appropriate for a biophysically similar forest parceled out to multiple owners with little forestry experience and less capital. Nor would they be appropriate for a forest under the control of a biodiversity-maximizing non-governmental organization. At a larger scale, these prescriptions might need to be modified if the forest is an important watershed, extremely far from timber markets or unique in the broader landscape.

Motivation seems to be growing for landscape-scale forest management that benefits from the full range of possible silvicultural options. One indication of this growth in recognition of the importance of managed forests is that over the past decade, the historical antagonism between environmentalists and foresters seems to have waned. Rather than solely focusing on protected areas for biodiversity maintenance, selectively logged forests have repeatedly been shown to retain large numbers of species and continue to deliver many desired ecosystem services (e.g. Putz et al. 2012). But now with the enlarged palette of land use and silvicultural options, research needs to be expanded substantially to include the full range of treatments applied at different intensities and with different distributions across landscapes.

Successful implementation of adaptive forest management depends entirely on the availability of a cadre of well-trained and motivated foresters supported by enlightened decision-makers and endorsed by an informed and involved society. Given that few foresters are being trained in this type of silviculture, the viability of this sort of flexible approach is in serious doubt. For it to work, something needs to be done to prepare a new generation of broadly trained, physically fit and un-corruptible field-based foresters.

A core component of the management of complex adaptive systems is experimentation (Filotas et al. 2014). Although most research has traditionally dealt with the impacts of particular silvicultural regimes on forest attributes (e.g. stand recovery, biodiversity), the challenges faced by tropical forestry requires that experimentation duly recognizes financial and social matters while it informs policy. At broader governance levels, the existence of different mechanisms to support responsible resource management (e.g. sustainability certification and legality verification) would benefit from an experimental approach that could reveal potential synergies and often overlapping and conflicting situations (Lambin et al. 2014). Testing a variety of silvicultural treatments at different scales with different stakeholders and with various mixtures of incentives and disincentives will help illuminate the windows of opportunity for policy design and adjustment (e.g. Ndjondo et al. 2014; Fortini et al. 2015). Such exercises can help create a learning community of researchers and practitioners engaged in management, foster dialogue and spaces for negotiation, and overall, promote a transformative shift toward responsible forest management.

Tropical forestry has changed greatly of late and needs to change even more. For one thing, researchers need to go beyond simple logged versus unlogged comparisons – especially those based on weak experimental designs and small plots – to interdisciplinary experimental studies at landscape scales. After nearly a century of isolated efforts and dust gathering on critical data sets from managed forests (Ruslandi et al. 2014b), collaboration among forestry researchers and even data sharing are becoming more common (e.g. Rutishauser et al. 2015). As managed tropical forests become increasingly recognized as critical for both conservation and development, the ranks of tropical forestry have also opened to a wide diversity of contributors. While innovative ideas and novel approaches to management need to be welcomed, new and old ones need to be evaluated with the most robust methods (e.g. Baylis et al. 2015) so that there is evidence to use in making decisions about their acceptance, adjustment or rejection. All this newness and methodological sophistication should be embraced, but important roles will remain for foresters who spend the required years in the field gaining first-hand experience with species, ecosystems, management, and exposure to the social and political dilemmas of valuable resources. Meta-analyses and remote sensing studies can help us learn about tropical forests and their management, but there is no substitute for field savvy.

6 References

Agrawal A, Chhatre A and Hardin R. 2008. Changing governance of the world's forests. *Science* 320:1460–62.

Angelsen A and Rudel TK. 2013. Designing and implementing effective REDD+ policies: A forest transition approach. *Review of Environmental and Economic Policy* 7:91–113.

Applegate G, Putz FE and Snook LK. 2004. *Who Pays For and Who Benefits from Improved Timber Harvesting Practices in the Tropics? Lessons Learned and Information Gaps*. Bogor, Indonesia: Center for International Forestry Research.

Arroyo-Mora JP, Svob S, Kalacska M and Chazdon RL. 2014. Historical patterns of natural forest management in Costa Rica: The good, the bad and the ugly. *Forests* 5: 1777–97.

Ashton MS, Singhakumara BMP, Gunateilleke N and Gunateilleke S. 2011. Sustainable forest management for mixed-dipterocarp forests: A case study in southwest Sri Lanka. *In* Gunter S, Weber M, Stimm B and Mosandl R, eds. *Silviculture in the Tropics*. Berlin, Germany: Springer-Verlag. 193–213.

Asner GP, Broadbent EN, Oliveira PJC, Keller M, Knapp DE and Silva JNM. 2006. Condition and fate of logged forests in the Brazilian Amazon. *Proceedings of the National Academy of Sciences* 103:12947–50.

Austin KG, Kasibhatla PS, Urban DL, Stolle F and Vincent J. 2015. Reconciling oil palm expansion and climate change mitigation in Kalimantan, Indonesia. *PLoS ONE* 10(5):e0127963.

Batra P and Pirard R. 2015. *Is a typology for planted forests feasible, or even relevant?* Info Brief No. 121. Bogor, Indonesia: Center for International Forestry Research. doi:10.17528/cifor/005608

Baur G. 1964. *The Ecological Basis of Rainforest Management*. New South Wales, Australia: Forestry Commission, Department of Conservation.

Baylis K, Honey-Rosés J, Börner J, Corbera E, Ezzine-de Blass D, Ferraro PJ, Lapeyre R, Persson M, Pfaff A and Wunder S. 2015. Mainstreaming impact evaluation in nature conservation. *Conservation Letters* 00(0):1–7. doi:10.1111/conl.12180

Bicknell JE, Struebig MJ, Edwards DP and Davies ZG. 2014. Improved timber harvest techniques maintain biodiversity in tropical forests. *Current Biology* 24(23):1119–20.

Börner J, Wunder S, Wertz-Kanounnikoff S, Rügnitz-Tito M, Pereira L and Nascimento N. *2010.* Direct conservation payments in the Brazilian Amazon: Scope and equity implications. *Ecological Economics* 69: 1272–82.

Bowler DE, Buyung-Ali LM, Healey J, Jones JPG, Knight TM and Pullin AS. 2011. Does community forest management provide global environmental benefits and improve local welfare? *Frontiers in Ecology and the Environment* 10:29–36.

Brandt JS, Nolte C, Steinberg J and Agrawal A. 2014. Foreign capital, forest change and regulatory compliance in Congo Basin forests. *Environmental Research Letters* 9(4):1–9. doi:10.1088/1748-9326/9/4/044007

Bruenig EF. 1996. Conservation and Management of Tropical Rainforests: An Integrated Approach to Sustainability. Wallingford, UK: CAB International.

Burivalova Z, Şekercioğlu CH and Koh LP. 2014. Thresholds of logging intensity to maintain tropical forest biodiversity. *Current Biology* 24:1893–8. doi:10.1016/j.cub.2014.06.065

Cerutti PO, Tacconi L, Lescuyer G and Nasi R. 2013. Cameroon's hidden harvest: Commercial chainsaw logging, corruption, and livelihoods. *Society and Natural Resources* 26(5):539–53.

Chomitz KM, Buys P, De Luca G, Thomas TS and Wertz-Kanounnikoff S. 2007. *At loggerheads? Agricultural expansion, poverty reduction, and*

environment in tropical forests. Washington, DC: World Bank.
Coomes OT, Gimard F, Potvin C and Sima P. 2008. The fate of the tropical forest: Carbon or cattle? *Ecological Economics* 65:207–12.
Corrià-Ainslie R, Camarero JJ and Toledo M. 2015. Environmental heterogeneity and dispersal processes influence post-logging seedling establishment in a Chiquitano dry tropical forest. *Forest Ecology and Management* 349:122–33.
Corriveau-Bourque A, Springer J, White A and Ogden DB. 2014. Roots of recognition and contested claims: Opportunities and challenges for pro-community forest tenure reform since 2002. *In* Nikolakis W and Innes J, eds. *Forests and Globalization: Challenges and Opportunities for Sustainable Development*. New York: Routledge. 87–105.
Costa FRC and Magnusson WE. 2003. Effects of selective logging on the diversity and abundance of flowering and fruiting understory plants in a central Amazonian forest. *Biotropica* 35:103–14.
Crome FHJ. 1997. Researching tropical forest fragmentation: Shall we keep on doing what we are doing? *In Laurance WF and Bierregaard RO*, eds. *Tropical Forest Remnants.* Chicago: The University of Chicago Press. 485–501.
Cuffaro N and Hallam D. 2011. "Land grabbing" in developing countries: Foreign investors, regulation, and codes of conduct. *Social Science and Research Network*:1–14. doi:10.2139/ssrn.1744204
Dauber E, Fredericksen TS, Peña-Claros M. 2005. Sustainability of timber harvesting in Bolivian tropical forests. *Forest Ecology and Management* 214:294–304.
Dawkins HC and Philips MS. 1998. *Tropical Moist Forest Silviculture and Management: A History of Success and Failure*. Wallingford, UK: CAB International.
de Avila AL, Ruschel AR, de Carvalho JOP, Mazzei L, Silva JNM, do Carmo Lopes J, Araujo MM, Dormann CF 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.
Didham RK. 2011. Life after logging: Strategic withdrawal from the Garden of Eden or tactical error for wilderness conservation? *Biotropica* 43:393–95.
Dove MR. 2011. *The Banana Tree at the Gate: A History of Marginal Peoples and Global Markets in Borneo*. New Haven: Yale University Press.
Drigo L, Piketty M-G, Peña D and Sist P. 2013. Cash income from community-based forest management: Lessons from two case studies in the Brazilian Amazon. *Bois et Forêts du Tropiques* 315:39–49.
Dykstra D and Heinrich R. 1996. *FAO Model Code of Forest Harvesting Practice*. Rome: FAO.
Eba'a Atyi RS, Assembe-Mvondo S, Lescuyer G and Cerutti P. 2013. Impacts of international timber procurement policies on Central Africa's forestry sector: The case of Cameroon. *Forest Policy and Economics* 32:40–48.
Edwards DP, Gilroy JJ, Woodcock P, Edwards FA, Larsen TH, Andrews DJR, Derhe ME, Dochery TDS, Hsy WW, Mitchell SL, et al. 2014. Land-sharing versus land-sparing logging: Reconciling timber extraction with biodiversity conservation. *Global Change Biology* 20:183–91.
Ellis EC, Goldewijk KK, Siebert S, Lightman D and Ramankutty N. 2010. Anthropogenic transformation of the biomes, 1700 to 2000. *Global Ecology and Biogeography* 19:589–606.
Evans J. 1982. *Plantation Forestry in the Tropics.* Oxford, UK: Oxford University Press.
Feintrenie L, Schwarze S and Levang P. 2010. Are local people conservationists? Analysis of transition dynamics from agroforests to monoculture plantations in Indonesia. *Ecology and Society* 15(4):37.
Ferry B, Morneau F, Bontemps J-D, Blanc L and Freycon V. 2010. Higher treefall rates on slopes and waterlogged soils result in lower stand biomass and productivity in a tropical rain forest. *Journal of Ecology* 98:106–16.
Filotas E, Parrott L, Burton PJ, Chazdon RL, Coates KD, Coll L, Haeussler S, Martin K, Nocentini S, Puettmann KJ, et al. 2014. Viewing forests through the lens of complex systems science. *Ecosphere* 5:1–23.
Finer M, Jenkins CN, Blue Sky MA and Pine J. 2014. Logging concessions enable illegal logging crisis in the Peruvian Amazon. *Scientific Reports* 4:4719. doi:10.1038/srep04719
Fisher B, Edwards DP, Giam X and Wilcove DS. 2011. The high costs of conserving Southeast Asia's lowland rainforests. *Frontiers in Ecology and Environment* 9:329–34.

Fortini LB, Cropper Jr. WP and Zarin DJ. 2015. Modeling the complex impacts of timber harvests to find optimal management regimes for Amazon tidal floodplain forests. *PLoS ONE* 10(8):e0136740. doi:10.1371/journal.pone.0136740.

Fredericksen TS and Putz FE. 2003. Silvicultural intensification for tropical forest conservation. *Biodiversity and Conservation* 12:1445–53.

Freitas MAB, Viera IMG, Albanez ALKM, Magalhães JLL and Lees AC. 2015. Floristic impoverishment of Amazonian floodplain forests managed for açaí fruit production. *Forest Ecology and Management* 351(1):20–27.

Gaveau DLA, Curran LM, Paoli GD, Carlson KM, Wells P, Besse-Rimba A, Ratnasari D and Leader-Williams N. 2011. Examining protected area effectiveness in Sumatra: Importance of regulations governing unprotected lands. *Conservation Letters* 5:142–48.

Gaveau DLA, Kshatriya M, Sheil D, Sloan S, Molidena E, Wijaya A, Wich S, Ancrenaz M, Hansen M, Broich M, et al. 2013. Reconciling forest conservation and logging in Indonesian Borneo. PLoS One 8(8):e69887. doi:10.1371/journal.pone.0069887.

Gaveau D and Pirard R. 2015. *Satellites can mislead: Policy makers beware*. CIFOR Blog 22 April 2015. Bogor, Indonesia: Center for International Forestry Research. Accessed 1 September 2015. http://blog.cifor.org/28010/satellites-can-mislead-policy-makers-beware#.VZLmvvlVhHz

Ghana Forestry Commission. 2013. *Draft Ghana Forest Plantation Strategy: 2015-2040*. Ghana Forestry Commission. Accessed June 2015. http://www.fcghana.org/userfiles/files/Plantation%20Annual%20Report/Ghana%20Forest%20Plantation%20Strategy%20 2015-%202040.pdf

Ghazoul J, Burivalova Z, John Garcia-Ulloa J and King LA. 2015. Conceptualizing forest degradation. *Trends in Ecology and Evolution* 30:622–32.

Goldstein JE. 2014. The afterlives of degraded tropical forests: New value for conservation and development. *Environment and Society: Advances in Research* 5:124–40.

Gould KA. 2006. Land regularization on agricultural frontiers: The case of northwestern Peten, Guatemala. *Land Use Policy* 23: 395–407.

Grau R, Kuemmerle T and Macchi L. 2013. Beyond 'land sparing versus land sharing': Environmental heterogeneity, globalization and the balance between agricultural production and nature conservation. *Current Opinion in Environmental Sustainability* 5(5):477–83.

Griscom B, Ellis P and Putz FE. 2014. Carbon emissions performance in commercial logging concessions of East Kalimantan, Indonesia. *Global Change Biology* 20:923–37.

Gunter S, Weber M, Stimm B,and Mosandl R, eds. 2011. *Silviculture in the Tropics*. Berlin, Germany: Springer-Verlag.

Hardiyansyah G, Ruslandi, Hidayat D, Nugraha A, Mahardi M and Putz FE. 2015. Clear bole lengths of line-planted and naturally regenerated *Shorea leprosula* trees in Kalimantan, Indonesia. *Journal of Tropical Forest Science* 27(3):420–26.

Hartshorn GS. 1989. Application of gap theory to tropical forest management: Natural regeneration on strip-clearcuts in the Peruvian Amazon. *Ecology* 70:567–69.

Healey JR, Price C and Tay J. 2000. The cost of carbon retention by reduced impact logging. *Forest Ecology and Management* 139:237–55.

Hecht SB, Morrison KD and Padoch C, eds. 2014. *The Social Lives of Forests: Past, Present, and Future of Woodland Resurgence*. Chicago: The University of Chicago Press.

Hirakuri SR. 2003. Can law save the forest? Lessons from Finland and Brazil. *Forest Law Enforcement Governance and Trade*. Bogor, Indonesia: Center for International Forestry Research.

Hobbs RJ, Higgs ES and Hall C. eds. 2013. *Novel Ecosystems: Intervening in the New Ecological World Order*. Oxford, UK: Wiley-Blackwell.

Holmes TP. 2015. Financial and economic analysis of reduced impact logging. *In* Kohl M and Pancel L, eds. *Tropical Forestry Handbook*. Berlin, Germany: Springer-Verlag. 1–15.

Holmes TP, Blate GM, Zweede JC, Pereira Jr. R, Barreto P, Boltz F and Barch R. 2002. Financial and ecological indicators of RIL logging performance in the eastern Amazon. *Forest Ecology and Management* 163:93–110.

Hutchinson ID. 1988. Points of departure for silviculture in humid tropical forests. *Commonwealth Forestry Review* 67:223–34.

Hutchinson ID. 1981. *Sarawak liberation thinning: Part A, background and initial analysis of*

performance. Kuching, Sarawak, Malaysia: United Nations Development Programme; Food and Agriculture Organization of the United Nations.
Hyde WF. 2012. *The Global Economics of Forestry*. New York: Resources for the Future Press.
Imai N, Tanaka A, Samejima H, Sugau JB, Pereira JT, Titin J, Kurniawan Y and Kitayama K. 2014. Tree community composition as an indicator in biodiversity monitoring of REDD+. *Forest Ecology and Management* 313:169–79.
Inada T, Kitajima K, Kanzaki M, Ano W, Hardiwitono S, Sadono R, Setyanto PE and Saminto. 2015. Neighboring tree effect on the survival and growth of *Shorea johorensis* under a line planting system in a Bornean dipterocarp forest. *Tropics* 24:23–31.
Karsenty A, Drigo IG, Piketty M-G and Singer B. 2008. Regulating industrial forest concessions in Central Africa and South America. *Forest Ecology and Management* 256:1498–508.
Katila P, Galloway G, de Jong W, Pacheco P and Mery G. eds. 2014. *Forests under pressure — Local responses to global issues*. IUFRO World Series 32. International Union of Forest Research Organizations.
Kelly AB and Peluso NL. 2015. Frontiers of commodification: State lands and their formalization. *Nature and Society* 28:473–95.
Kusuma AP, Damayati A, Amijaya AM, Hardiansyah G, Chozin M, Suparna N and Saribanon N. 2014. *Road Map: Implemantasi silvikultur intensif pada hutan alam producksi Indonesia 2045*. Pontianak, Indonesia: Fakultas Kehutanana Universitas Tangungpura Pontianak Press.
Lambin EF, Meyfroidt P,Rueda X, Blackman A, Börner J, Cerutti PO, Dietsch T, Jungmann L, Lamarque P, Lister J, et al. 2014. Effectiveness and synergies of policy instruments for land use governance in tropical regions. *Global Environmental Change* 28:129–40.
Lamprecht H. 1993. Silviculture in the natural tropical forests. *In* Pancel L, ed. Tropical Forestry Handbook. Berlin, Germany: Springer-Verlag. 728–810.
Larjavaara M. 2012. Democratic less-developed countries cause global deforestation. *International Forestry Review* 14:299–313.
Laurance WF, Clements GR, Sloan S, O'Connell CS, Mueller ND, Goosem M, Venter O, Edwards DP, Phalan B, Balmford A, et al. 2014. A global strategy for road building. *Nature* 513: 229–34.
Law EA, Meijaard E, Bryan BA, Mallawaarachchi T, Koh LP and Wilson KA. 2015. Better land-use allocation outperforms land sparing and land sharing approaches to conservation in Central Kalimantan, Indonesia. *Biological Conservation* 186:276–86.
Lippke B, Oneil E, Harrison R, Skog K, Gustavsson L and Sathre R. 2015. Life cycle impacts of forest management and wood utilization on carbon mitigation: Knowns and unknowns. *Carbon Management* 2(3):303–33.
Louman B, Quirós D and Nilsson M, eds. 2001. *Silvicultura de Bosques Latifoliados Húmedos con Énfasis en América Central*. Turrialba, Costa Rica: CATIE.
Marlier ME, DeFries RS, Kim PS, Koplitz SN, Jacob DL, Mickley LJ and Myers SS. 2015. Fire emissions and regional air quality impacts from fires in oil palm, timber, and logging concessions in Indonesia. *Environmental Research Letters* 10(8):1–9.
Mascia MB and Pailler S. 2011. Protected area downgrading, downsizing, and degazettement (PADDD) and its conservation implications. *Conservation Letters* 4:9–20.
McSweeney K, Nielsen EA, Taylor MJ, Wrathall DJ, Pearson Z, Wang O and Plumb ST. 2014. Drug policy as conservation policy: Narco-deforestation. *Science* 343:489–90.
Medjibe VP and Putz FE. 2012. Cost comparisons of reduced-impact and conventional logging in the tropics. *Journal of Forest Economics* 18:242–56.
Meijaard E, Abram NK, Wells JA,. Pellier A-S, Ancrenaz M, Gaveau DLA, Runting RK and Mengersen K. 2013. People's perceptions about the importance of forests on Borneo. PLoS ONE 8:e73008. doi:10.1371/journal.pone.0073008.
Mejia E and Pacheco P, eds. 2014. *Forest use and timber markets in the Ecuadorian Amazon*. CIFOR Occasional Paper 111. Bogor, Indonesia: Center for International Forestry Research.
Messier C, Puettmann K, Chazdon R, Andersson KP, Angers VA, Brotons L, Filotas E, Tittler R, Parrott L and Levin SA. 2014. From management to stewardship: Viewing forests as complex adaptive systems in an uncertain world. *Conservation Letters*:1–10. doi:10.1111/conl.12156

Messier C, Tittler R, Kneeshaw DD, Gélinas N, Paquette A, Berninger K, Rhealt H, Mekk P and Beaulieu N. 2009. TRIAD zoning in Quebec: Experiences and results after 5 years. *The Forestry Chronicle* 85:885–96.

Miettinen J and Liew SC. 2010. Degradation and development of peatlands in Peninsular Malaysia and in the islands of Sumatra and Borneo since 1990. *Land Degradation and Development* 21:285–96.

Miettinen J, Shi C and Liew SC. 2012. Two decades of destruction in Southeast Asia's peat swamp forests. *Frontiers in Ecology and the Environment* 10:124–8.

Miner RA, Abt RC, Boweyer JL, Buford MA, Malmsheimer RW, O'Laughlin J, Oneil EE, Sedjo RA and Skog KE. 2014. Forest carbon accounting considerations in US bioenergy policy. *Journal of Forestry* 112:591–606.

Miranda JJ, Corral L, Blackman A, Asner G and Lima E. 2014. *Effects of protected areas on forest cover change and local communities: Evidence from the Peruvian Amazon*. Discusssion Paper 14-14. Washington, DC; Resources for the Future. 1–34.

Morales-Barquero L, Skutsch M, Jardel-Peláez EJ, Ghilardi A, Kleinn C and Healey JR. 2014. Operationalizing the Definition of Forest Degradation for REDD+, with Application to Mexico. *Forests* 5:1653–81.

Moura-Costa PH, Yap SW, Ong CL, Ganing A, Nussbaum R and Mojiun T. 1996. Large scale enrichment planting with dipterocarps as an alternative for carbon offset-methods and preliminary results. *In* Appanah S and Khoo KC, eds. *Proceedings of the 5th Round Table Conference on Dipterocarps*: *Chiang Mai, Thailand 1994*. Kepong, Malaysia: FRIM. 386–96.

Ndjondo M, Gourlet-Fleury S, Manlay R, Engone Obiang N, Ngomanda A, Romero C, Claeys FL and Picard N. 2014. Opportunity costs of carbon sequestration in a forest concession in central Africa. *Carbon Balance and Management* 9:4. http://www.cbmjournal.com/content/9/1/4

Nitikusuma R, Yusro F, Ruslandi and Putz FE. 2015. Constraints on the harvest of line-planted trees in twice logged forest in Kalimantan, Indonesia. *Journal of Tropical Forest Science* 27(3):433–38.

Norizah K, Mohd Hasmadi I, Kamaruzaman J and Alias MS. 2012. Operational efficiency of Rimbaka timber harvester in hilly tropical forest. *Journal of Tropical Forest Science* 24:368–78.

Obidzinski K and Kusters K. 2015. Formalizing the logging sector in Indonesia: Historical dynamics and lessons for current policy initiatives. *Society and Natural Resources* 28:530–42.

Okuda T, Suzuki M, Adachi N, Quah ES, Hussein NA and Manokaran N. 2003. Effect of selective logging on canopy and stand structure and tree species composition in a lowland dipterocarp forest in Peninsular Malaysia. *Forest Ecology and Management* 175:297–20.

Oliver CD, Nassar NT, Lippke BR and McCarter JB. 2013. Carbon, fossil fuel, and biodiversity mitigation with wood and forests. *Journal of Sustainable Forestry* 33: 248–75.

Oliveira LJC, Costa MH, Soares-Filho BS and Coe MT. 2013. Large-scale expansion of agriculture in Amazonia may be a no-win scenario. *Environmental Research Letters* 8(2):1–10. doi:10.1088/17489326/8/2/024021

Oliviera PJC, Asner GP, Knapp DE, Almeyda A, Galvan-Gildemeister R, Keene S, Raybin RF and Smith RC. 2007. Land-use allocation protects the Peruvian Amazon. *Science Express* 317:1233–36. doi:10.1126/science.1146324.

Omeja PA, Chapman CA, Obua J, Lwanga JS, Jacob AL, Wanyama F and Mugenyi R. 2011. Intensive tree planting facilitates tropical forest biodiversity and biomass accumulation in Kibale National Park, Uganda. *Forest Ecology and Management* 261:703–9.

Ongolo S and Karsenty A. 2015. The politics of forestland use in a cunning government: Lessons for contemporary forest governance reforms. *International Forestry Review* 17:195–209.

Page S, Hosciło A, Wösten H, Jauhiainen J, Silvius M, Rieley J, Ritzema H, Tansey K, Graham L, Vasander H and Limin S. 2009. Restoration ecology of lowland tropical peatlands in Southeast Asia: Current knowledge and future research directions. *Ecosystems* 12:888–905.

Pamoengkas P, Gandaseca S, Hardiansyah G, Priyanto, and Jamaludin MR. 2014. Tree diameters and planting distance as the most important factors for the liberation of tree competitors in silvicultural systems of TPTJ. *Agriculture, Forestry and Fisheries* 3:392–96.

Papp L and Vidal E. In press. Typology of the timber sector and dynamics along the natural forest certification continuum. In *The Context of Natural Forest Management and FSC Certification in Brazil.* Bogor, Indonesia: Center for International Forestry Research.

Peña-Claros ML, Fredericksen TS, Alarcon A, Blate GM, Choque U, Leaño C, Mostacedo B, Pariona W, Villegas Z and Putz FE. 2008. Beyond reduced-impact logging: Silvicultural treatments to increase growth rates of tropical trees. *Forest Ecology and Management* 256:1458–67.

Petrokofsky G, Sist P, Blanc L, Doucet J-L, Finegan B, Gourlet-Fleury S, Healey J, Livoreil B, Nasi R, Palmer J, et al. 2015. Comparative effectiveness of silvicultural interventions for increasing timber production and sustaining conservation values in natural tropical production forests. A systematic review protocol. *Environmental Evidence* 4(8):1–7. doi:10.1186/s13750-015-0034-7

Pfund J-L, Watts JD, Boissiére M, Boucard A, Bullock RM, Ekadinata A, Dewi S, Feintrenie L, Levang P, Rantala S, et al. 2011. Understanding and integrating local perceptions of trees and forests into incentives for sustainable landscape management. *Environmental Management* 48:334–49.

Phelps J, Webb EL and Agrawal A. 2010. Does REDD+ threaten to recentralize forest governance? *Science* 328:312–13.

Pinard MA, Barker MG and Tay J. 2000. Soil disturbance and post-logging forest recovery on bulldozer paths in Sabah, Malaysia. *Forest Ecology and Management* 130:213–25.

Pinard MA and Putz FE. 1996. Retaining forest biomass by reducing logging damage. *Biotropica* 28:278–95.

Pinard MA, Putz FE, Tay J and Sullivan TE. 1995. Creating timber harvest guidelines for a reduced-impact logging project in Malaysia. *Journal of Forestry* 93:41–45.

Pinso C and Moura-Costa P. 1993. Greenhouse gas offset funding for enrichment planting a case study from Sabah, Malaysia. *Commonwealth Forestry Review* 72:343–49.

Piu HC and Menton M. 2013. *Contexto de REDD+ en Perú: Motores, actores e instituciones*. Occasional Paper 90. Bogor, Indonesia: Center for International Forestry Research.

Pokorny B and Johnson J. 2008. *Community forestry in the Amazon: The unsolved challenge of forests and the poor*. Natural Resource Perspectives 112. London: Overseas Development Institute.

Pokorny B and Pachecho P. 2014. Money from and forest forests: A critical reflection on the feasibility of market approaches for the conservation of Amazonian forests. *Journal of Rural Studies* 36:441–52. doi:10.1016/j.rurstud.2014.09.004

Price SV, ed. 2003. *War and Tropical Forests: Conservation in Areas of Armed Conflict*. New York: Haworth Press.

Putz FE. 2013. Complexity confronting tropical silviculturalists. *In* Messier C, Puettman KJ and Coates KD, eds. *Managing Forests as Complex Adaptive Sytems: Building Resilience to the Challenge of Global Change*. New York: Earthscan. 165–86.

Putz FE. 2004. Are you a logging advocate or a conservationist? *In* Zarin D, Putz FE, Alavalapati J and Schmink M, eds. *Working Forests in the Tropics*. New York: Columbia University Press. 15–30

Putz FE, Blate GM, Redford KH, Fimbel R and Robinson JG. 2001a. Biodiversity conservation in the context of tropical forest management. *Conservation Biology* 15:7–20.

Putz FE and Romero C. 2014. Futures of tropical forests (*sensu lato*). *Biotropica* 46:495–505.

Putz FE and Romero C. 2012. Helping curb tropical forest degradation by linking REDD+ with other conservation interventions: A view from the forest. *Current Opinion in Environmental Sustainability* 4:670–77.

Putz FE and Ruslandi. 2015. Intensification of tropical silviculture. *Journal of Tropical Forest Science* 27:285–88.

Putz FE, Sirot LK and Pinard MA. 2001b. Tropical forest management and wildlife: Silvicultural effects on forest structure, fruit production, and locomotion of non-volant arboreal animals. *In* Fimbel R, Grajal A and Robinson J, eds. *The Cutting Edge: Conserving Wildlife in Managed Tropical Forests*. New York: Columbia University Press. 11–34.

Putz FE, Sist P, Fredericksen TS and Dykstra D. 2008. Reduced-impact logging: Challenges and opportunities. *Forest Ecology and Management* 256:1427–33.

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 Letters* 5:296–303.

Putzel L, Cronkleton P, Larson A, Pinedo-Vasquez M, Salazar O and Sears R. 2013. *Peruvian smallholder production and marketing of bolaina (Guazuma crinita), a fast-growing Amazonian timber species*. CIFOR Brief 23. Bogor, Indonesia: Center for International Forestry Research.

Putzel L, Kelly AB, Cerutti PO and Artati Y. 2015. Formalization as development in land and natural resource policy. *Society and Natural Resources* 28:453–72.Rice RE, Gullison RE and Reid JW. 1997. Can sustainable management save tropical forests? *Scientific American* 276:44–49.

Roda J-M, Kamaruddin N and Tobias RP. 2015. Deciphering corporate governance and environmental commitments among Southeast Asian Transnationals: Uptake of sustainability certification. *Forests* 6:1454–75.

Romero C, Putz FE, Guariguata MR, Sills EO, Cerutti PO and Lescuyer G. 2013. *An overview of current knowledge about the impacts of forest management certification: A proposed framework for its evaluation*. Occasional Paper 91. Bogor, Indonesia: Center for International Forestry Research.

Rudel TK, Bates D and Machinguiashi R. 2002. A tropical forest transition? Agricultural change, out-migration, and secondary forests in the Ecuadorian Amazon. *Annals of the Association of American Geographers* 92:87–102.

Rudel TK and Meyfroid P. 2014. Organizing anarchy: The food security–biodiversity–climate crisis and the genesis of rural land use planning in the developing world. *Land-Use Policy* 36:239–47.

Ruslandi, Jatmoko, Japrianto, Purnomosidi, Budiarto S, Hardiansyah G, Inada T, Purnomo S and Putz FE. 2015. Buttress characteristics in relation to topography and crown eccentricity in planted and naturally regenerated *Shorea leprosula* trees. *Journal of Tropical Forest Science* 27(3):439–45.

Ruslandi, Klassen A, Putz FE and Romero C. 2014a. Forest Stewardship Council certification of natural forest management in Indonesia: Required improvements, costs, incentives, and barriers. *In* Katila P, Galloway G, de Jong W, Pacheco P and Mery G, eds. *Forests Under Pressure – Local Responses to Global Issues. IUFRO World Series 32*. 255–73.

Ruslandi, Roopsind A, Sist P, Peña-Claros M, Thomas R and Putz FE. 2014b. Beyond equitable data sharing to improve tropical forest management. *International Forestry Review* 16:497–503.

Ruslandi, Venter O and Putz FE. 2011. Over-estimating the costs of conservation in Southeast Asia. *Frontiers in Ecology and the Environment* 9:542–44.

Rutishauser E, Hérault B, Baraloto C, Blanc L, Descroix L, Sotta ED, Ferreira J, Kanashiro M, Mazzei L, d'Oliveira MVN, et al. 2015. Rapid tree carbon stock recovery in managed Amazonian forests. *Current Biology* 25:1–3.

[SBK] Sari Bumi Kusuma. 2014. *SBK provides a globally significant model for tropical silviculture*. Kalimantan, Indonesia: SBK. Accessed 29 September 2015. http://saribumikusuma.net/index.php?option=com_content&view=article&id=102:sbk-provides-a-globally-significant-model-for-tropical-silviculture&catid=58:berita-utama&Itemid=1

Sasaki N, Asner GP, Knorr W, Durst PD, Priyadi H and Putz FE. 2011. Approaches to classifying and restoring degraded tropical forests for the anticipated REDD+ climate change mitigation mechanism. *iForest* 4:1–6. doi:10.3832/ifor0556-004

Sasaki N and Putz FE. 2009. Critical need for new definitions of "forest" and "forest degradation" in global climate change agreements. *Conservation Letters* 2:226–32.

Sayer J, Sunderland T, Ghazoul J, Pfund J-L, Sheil D, Meijaard E, Venter M, Boedhihartono A, Day M, Garcia C, et al. 2013. Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. *Proceedings of the National Academy of Sciences* 110:8349–56.

Schöngart J von. 2008. Growth-Oriented Logging (GOL): A new concept towards sustainable forest management in Central Amazonian várzea floodplains. *Forest Ecology and Management* 256:46–58.

Sears RR and Pinedo-Vasquez M. 2011. Forest policy reform and the organization of logging in Peruvian Amazonia. *Development and Change* 42:609–31.

Sedjo RA and Botkin D. 1997. Using forest plantations to spare natural forests. *Environment* 39:14–30.

Shearman P, Bryan J and Laurance WF. 2012. Are we approaching 'peak timber' in the tropics? *Biological Conservation* 151:17–21.

Sidle RC, Ziegler AD, Negishi JN, Nik AR, Siew R and Turkelboom F. 2006. Erosion processes in steep terrain: Truths, myths, and uncertainties related to forest management in Southeast Asia. *Forest Ecology and Management* 224: 199–225.

Sist P, Fimbel R, Nasi R, Sheil D and Chevallier M-H. 2003a. Towards sustainable management of mixed dipterocarp forests of South East Asia: Moving beyond minimum diameter cutting limits. *Environmental Conservation* 30:364–74.

Sist P, Nolan T, Bertault J-G and Dykstra DP. 1998. Harvesting intensity versus sustainability in Indonesia. *Forest Ecology and Management* 108:251–60.

Sist P, Pacheco P, Nasi R and Blaser J. 2014a. Management of natural tropical forests in the past and present and projections for the future. *In* Katila P, Galloway G, de Jong W, Pacheco P and Mery G, eds. *Forests Under Pressure – Local Responses to Global Issues. IUFRO World Series 32*. 497–512.

Sist P, Picard N and Gourlet-Fleury S. 2003b. Sustainable rotation length and yields in a lowland mixed dipterocarp forest of Borneo. *Annals of Forest Science* 60:803–14.

Sist PE, Rutishauser E, Peña-Claros M, Shenkin A, Hérault B, Blanc L, Baraloto C, Baya F, Benedet F, Emidio da Silva K, et al. 2014b. The Tropical managed Forests Observatory: A research network addressing the future of tropical logged forests. *Applied Vegetation Science* 18:171–74.

Sist P, Sablayrolles P, Barthelon S, Sousa-Ota L, Kibler J-F, Ruschel A, Santos-Melo M and Ezzine-de-Blas D. 2014c. The contribution of multiple use forest management to small farmers' annual incomes in the eastern Amazon. *Forests* 5:1508–31.

Smith J, Colan V, Sabogal C and Snook L. 2006. Why policy reforms fail to improve logging practices: The role of governance and norms in Peru. *Forest Policy and Economics* 8:458–69.

Southworth J, Marsik M, Qiu Y, Perz S, Cumming G, Setvens F, Rocha K, Duchelle A and Barnes G. 2011. Roads as drivers of change: Trajectories across the tri-national frontier in MAP, the Southwestern Amazon. *Remote Sensing* 3:1047–66.

Suding K, Higgs E, Palmer M, Callicott JB, Anderson CB, Baker M, Gutrich JJ, Hondula KL, LaFevor MC, Larson BMH, et al. 2015. Committing to ecological restoration. *Science* 348:638–40.

Thompson ID, Guariguata MR, Okabe K, Bahamondez C, Nasi R, Heymell V and Sabogal C. 2013. An operational framework for defining and monitoring forest degradation. *Ecology and Society* 18:20.

Urrunaga AJ, Orbegozo ID and Mulligan F. 2012. *The laundering machine: How fraud and corruption in Peru's concession system are destroying the future of its forests*. Environmental Investigation Agency.

van Kooten GC, Bogle TN and de Vries FP. 2014. Forest carbon offsets revisited: Shedding light on darkwoods. *Forest Science* 61(2):370–80.

Villegas Z, Peña-Claros M, Mostacedo B, Alarcón A, Licona JC, Leaño JC, Pariona W and Choque U. 2009. Silvicultural treatments enhance growth rates of future crop trees in a tropical dry forest. *Forest Ecology and Management* 258:971–77.

Vincent JR. 1992. The tropical timber trade and sustainable development. *Science* 256: 1651–55.

von Thünen J. 1966. *Der Idolierte Staat Bezeihung auf Landwirtschaft und Nationalokonomie* [The Isolated State]. Translated by CM Waternberg. New York: Pergamon Press. (Original work published 1826).

von Wehrden H, Abson DJ, Beckmann M, Cord AF,Klotz S and Seppelt R. 2014. Realigning the land-sharing/land-sparing debate to match conservation needs: Considering diversity scales and land-use history. *Landscape Ecology* 29:941–48.

Wadsworth FH. 1997. *Forest production for tropical America*. USDA Forest service Agriculture Handbook 710. Washington, DC: United States Department of Agriculture.

White R. 1995. Are you an environmentalist or do you work for a living? Work and nature. *In* Cronon W, ed. *Uncommon Ground: Toward Reinventing Nature*. New York: W.W. Norton. 171–85.

Wu J. 2013. Landscape sustainability science: Ecosystem services and human well-being in changing landscapes. *Landscape Ecology* 28:999–1023.

Wunder S. 2013. When payments for environmental services will work for conservation. *Conservation Letters* 6:230–37.

Wunder S. 2015. Revisiting the concept of payments for environmental services. *Ecological Economics* 117:234–43.

Wyatt-Smith, J. 1963. *Manual of Malayan Silviculture for Inland Forests*: *Malayan Forest Record.* No. 23. Kuala Lumpur, Malaysia.

Zimmerman BL and Kormos CF. 2012. Prospects for sustainable logging in tropical forests. *BioScience* 62:479–87.

ISBN 978-602-387-016-5
DOI: 10.17528/cifor/005766

CIFOR Occasional Papers contain research results that are significant to tropical forest issues. This content has been peer reviewed internally and externally.

Forests are landscape-embedded complex systems with fates determined by multitudes of changing and interacting factors that are sectoral and extra-sectoral, biophysical and political, predictable and chaotic. The diversity of forest states (e.g. secondary, degraded, fragmented, invaded and managed) and the fact that none of these states is permanent gives reason for hope; even deforestation need not be permanent. With so many forest values recognized to different degrees by different people, the future of tropical production forests is likely to represent an ever-changing mosaic of a gradient of forested-type landscapes. To assure that this future is as environmentally, socioeconomically and politically sound as possible, researchers need to synthesize and evaluate what is known and then build on that knowledge while they continue learning. There is a critical need for interdisciplinary research at appropriate scales with the best designs possible to capture the impacts of relevant silvicultural treatments on the full range of response variables

RESEARCH PROGRAM ON
Forests, Trees and Agroforestry

This research was carried out by CIFOR as part of the CGIAR Research Program on Forests, Trees and Agroforestry (CRP-FTA). This collaborative program aims to enhance the management and use of forests, agroforestry and tree genetic resources across the landscape from forests to farms. CIFOR leads CRP-FTA in partnership with Bioversity International, CATIE, CIRAD, the International Center for Tropical Agriculture and the World Agroforestry Centre.

cifor.org

blog.cifor.org

Fund

UKaid
from the British people

Center for International Forestry Research (CIFOR)
CIFOR advances human well-being, environmental conservation and equity by conducting research to help shape policies and practices that affect forests in developing countries. CIFOR is a member of the CGIAR Consortium. Our headquarters are in Bogor, Indonesia, with offices in Asia, Africa and Latin America.