Forest Ecology and Management 352 (2015) 134–145

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Review and synthesis

Forest Resources Assessment of 2015 shows positive global trends but forest loss and degradation persist in poor tropical countries ☆

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A R T I C L E I N F O

Article history:
Received 16 March 2015
Received in revised form 7 June 2015
Accepted 8 June 2015
Available online 7 September 2015

Keywords:
Global forest trends
Forest transitions
Sustainable forest management
Plantation forests
Planted forests
Forest pests and diseases
Canopy cover reduction

A B S T R A C T

The Global Forest Resources Assessment 2015 shows that deforestation has slowed and afforestation has increased globally during 1990–2015. Planted forests have increasingly provided goods and services hitherto derived from natural forests, and mosaic forests in agricultural landscapes are increasing. Forest gain is occurring at higher latitudes and in richer countries whilst forest loss continues in poor countries in the tropics. Some middle income tropical countries are now also transitioning to forest gain. These transition countries are characterised by reforms to forest management and improvements in agricultural practices but also by significant expansions of planted forest, which account for ~25–100% of gains. Forest-area estimates of the FRA align with satellite-derived estimates, with deviations of ⩽±7% globally and ⩽±17% for the tropics. Mosaics comprised of trees outside forests, remnant forest patches, and young regenerating forests constitute a modest proportion of the tropical forest estate and are seemingly well inventoried by the FRA. Extensive areas of forest experienced partial canopy cover reduction since 2000, particularly in the tropics where their area is ~6.5 times that deforested since 1990. The likelihood of the eventual loss of these forests and a decline in their capacity to provide goods and services is a matter of concern. Demand for industrial wood and fuelwood increased 35% in the tropics since 1990, principally in poorer countries, and growth in demand will accelerate into the future, particularly in the Asia-Pacific region. Notwithstanding significant increases in forests within protected areas since 1990 to 517 Mha (16.3%) globally and 379 Mha (26.6%) in the tropics, increasing demands for ecological services, forest products, and climate change mitigation is likely to be met from an expanding area of planted forests more than from the declining area of natural forests, particularly in Africa. The global rate of planted-forest expansion since 1990 is close to a target rate of 2.4% per annum necessary to replace wood supplied from natural forests in the medium term, though the expansion rate has declined to 1.5% since 2005. Multiple-use forests permitting both production and conservation account for 26% of the global forest area and 17% of the tropical forest area, and have increased by 81.8 Mha or 8.5% globally since 1990, with most gains in the tropics. Sustainable forest management in low-income and tropical countries remains modest, with only 37% low-income country forests covered by forest inventories. International support has proven effective at increasing this coverage since 2010.

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Contents

☆ This article is part of a special issue entitled "Changes in Global Forest Resources from 1990 to 2015".

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http://dx.doi.org/10.1016/j.foreco.2015.06.013
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1. Introduction

The Global Forest Resources Assessment (FRA) of the Food and Agricultural Organization of the United Nations (FAO) has undertaken global assessments of forest area, characteristics, and production every 5–10 years since 1948 (MacDicken, 2015). More so than previous FRAs, the FRA of 2015 paints a broadly positive picture of the state of the world's forests. Whilst there are many reasons to be optimistic about the future of forests, there remain major areas of concern as stark regional variations belie the apparent progress at the global scale.

The global rate of forest loss has decreased since 2010 to 3.3 million hectares (Mha) or 0.08% annually, being half the rate in the 1990s. Forests are stable or expanding in temperate and boreal regions, and the rate of deforestation in the tropics is slowing (Keenan et al., 2015). Similarly, rates of afforestation are steady or rising not just in temperate countries, where planted forests have long been integral elements of the forest estate, but also in the tropics where the extent of planted forests has nearly doubled since 1990 (Payn et al., 2015). Forest inventories and management plans now exist in more countries than ever and local stakeholders are increasingly engaged in managing and owning forests (MacDicken et al., 2015). International processes that seek to build a global consensus regarding multi-functional forest management are moving slowly towards agreement, as reflected in steady progress in national planning and commitments (MacDicken et al., 2015). The importance of maintaining forest area as part of the portfolio of measures to address climate change is also increasingly recognised (Federici et al., 2015), and we are inching towards commitments to fund measures to reduce deforestation and forest degradation (REDD+), with the United Nations REDD program now supporting forest monitoring initiatives in 58 countries.

Notwithstanding this progress, forest conversion for agriculture, especially for estate crops, remain significant in many, mainly poorer, tropical regions. Partial canopy cover reduction is extensive in many tropical countries and may lead to eventual forest loss (Van Lierop and Lindquist, 2015). Whilst forest diseases and pests are reported at significant scales only in richer high latitude countries (Van Lierop and Lindquist, 2015: Table 6), they are likely also a growing and under-appreciated threat to forests in poorer tropical countries. Even where forest areas are stable, as in Central Africa, forest wildlife is being lost at historically high rates (Butchart et al., 2010). In South America forest conversion is slowing, land use change is increasingly regulated, and protected-areas systems are expanding. Notwithstanding this, South America continues to experience the greatest losses of forest by far (Keenan et al., 2015). In many African countries forest management institutions remain weak (Romijn et al., 2015), leaving forests highly vulnerable to clearance and degradation (Keenan et al., 2015). In the Congo Basin low levels of forest conversion largely reflect ongoing conflict and a related lack of investment and infrastructure, rather than good management (de Wasseige et al., 2010). In South Asia stronger forest institutions are conserving the modest remaining areas of forest and encouraging the expansion of plantations, but total forest area is now critically low in many countries (Pandit et al., 2007; Sloan et al., 2014). In South East Asia forest conversion remains high as forest departments and corporate investors alike respond to global demand for estate crops such as oil palm, sugar, and wood fibre. Disparities in power over land resources is an issue in all regions as increasing influence is concentrated in the hands of the rich (Piketty and Goldhammer, 2014) and the interests of hopefully more conservation-minded local communities are marginalised.

Massive infrastructure investments are planned for many tropical regions and will soon open most of the world's remaining remote and pristine forests to commercial interests seeking land for estate crops, including industrial forest plantations (Weng et al., 2013; Edwards et al., 2014; Laurance et al., 2014a). The effects of such investment on forests are difficult to anticipate and have arguably not been fully accounted for in national economic and forest management strategies (Edwards et al., 2014). Agricultural expansion along new and improved roadways may concentrate populations and enable agricultural transformation and intensification; where this occurs in agriculturally favourable areas a depopulation of hinterlands may reduce pressures on forests (Angelsen and Rudel, 2013; Masters et al., 2013; Rudel, 2013). In countries with weak governance, new infrastructure may pave the way for opportunistic land development, with negative consequences for forests and the people dependent on them (Laurance et al., 2014b). Whereas infrastructure development would ideally be directed towards regions with high agricultural potential and little forest cover, the contrary is often the case when infrastructure expansion targets mineral resources or estate crops and, to a lesser degree, industrial timber plantations (Gutiérrez-Vélez et al., 2011; Durán et al., 2013; Weng et al., 2013; Gaveau et al., 2014). New mineral infrastructure poses significant threats to the major tropical forests in the Amazon and Congo Basins as well as the islands of Borneo and New Guinea – collectively accounting for most of the world's intact tropical forests. For the first time, the 2015 FRA gathered data on forest areas earmarked for conversion and official targets for total forest area by 2020 and 2030. Most of the countries surveyed and all climatic domains (tropical, subtropical, temperate, boreal, polar) anticipate *greater* forest areas[1] by 2030 than today. Such increases in forest area will have to be achieved in a world in which agriculture will pose a significant competing demand for land (Sayer and Cassman, 2013; Laurance et al., 2014b).

The dynamics of forest change have shifted over the 25-year period surveyed by the 2015 FRA and will continue shifting in the near term. In the tropics, where most forest change has occurred, deforestation due to smallholder agricultural expansion has given way to large-scale, enterprise-driven forest conversion (Hecht, 2005; Rudel, 2007; Asner et al., 2013). Rates of forest loss have declined, but the increasingly globalised pressures on forest lands pose significant new challenges to maintaining these declines (Lambin and Meyfroidt, 2011). In this context we integrate the analyses presented in this special issue of *Forest Ecology and Management* to assess the significance of the major trends in forest change reported by the 2015 FRA for 1990–2015. Special attention is given to the tropics, where the environmental stakes are highest and forests are still declining rapidly.

This article is structured as follows. The following section considers the FRA forest estimates in light of newly available satellite estimates and an increasing extent of dispersed mosaic forests. Section 3 summarises global patterns of forest loss and gain and

[1] 'Forest area' here adheres to the FRA definition encompassing both natural and planted forests.

highlights instances of national transitions from loss to gain. Section 4 profiles the growing demands for forest products and environmental services and discusses potential responses for large-scale forest management. Section 5 concludes by emphasising the growing challenge faced by large-scale forest inventories such as the FRA in capturing forest extent and services in heterogeneous forest landscapes.

2. The FRA and realities on the ground

The FRA 2015 gives a more precise and consistent picture of realities on the ground than previous assessments (Romijn et al., 2015). Estimates of forest area for 1990–2010 have been expressed in metrics that allow comparisons with those of 2015 (Keenan et al., 2015; MacDicken, 2015). Modern remote sensing has increased capacities to detect forest changes and 70% of countries are now deploying remote-sensing surveys as part of the FRA forest-inventory exercise. The FRA 2015 saw 17 additional tropical countries qualify as having 'good' or 'very good' capacities for forest monitoring and remote sensing compared to the FRA 2005, an increase of 30%, with a corresponding increase in the proportion of tropical forest area monitored to such a level from 69% to 83% (Romijn et al., 2015). Globally the number of countries with such capacities rose to 54, out of the 99 tropical countries surveyed by Romijn et al. (2015). Only 11% of global forest area for 2015 remains in the lowest of three tiers of data quality whilst 59% of the global forest area qualifies for the highest of the tiers (Keenan et al., 2015: Table 5). This uppermost tier is defined as forest-area data sourced from either repeated national forest inventories or from remote sensing or a national forest inventory completed $\leqslant$10 years ago and entailing ground truthing (FAO, 2015). Many poor countries still lack capacity for autonomous high-quality forest monitoring and reporting and international efforts to improve this situation should be pursued (MacDicken, 2015). The 12 countries with >5 Mha of forest of a Tier 1 grade accounted for only 9% of the global forest area but for 20% of global forest loss since 1990.

Successive FRAs have been criticised as imprecisely and inconsistently estimating forest area, particularly by analysts seeking to compare forest area between countries and over time (Grainger, 1996, 2008; Grainger, 2010; MacDicken, 2015). But the FRA is now assessing 120 different variables concerning forest resources and attributes for which data are provided by a range of national authorities. Methods, sampling intensity, and competence vary amongst countries. The term 'forest' describes everything from scattered trees in dry landscapes to dense, closed canopy old growth forests in high rainfall areas (Lund, 2006). The FRA has contributed to the adoption of globally comparable approaches to defining and categorising forests but total uniformity of national approaches is likely to be elusive. In Indonesia, for instance, much land officially designated as forest has been converted to agriculture, often illegally, and only some of these lands have been subsequently re-designated as non-forest land (Indrarto et al., 2012; Sloan, 2014). This discrepancy partially reflects tensions between the central forest administration seeking to retain influence over the forest estate and decentralised administrative bodies seeking greater freedom to allocate land for agricultural development. Indonesia is struggling to reconcile conflicting maps of land allocated to forests and agriculture and to adopt a single national map of forest and agricultural cover (Unit Keria Presiden4, 2012; Sloan, 2014). These national dynamics may affect Indonesia's report of forest-change to the FRA and complicate comparisons with otherwise similar countries, such as Malaysia.

The recent production of several independent global satellite-based estimates of forest area and forest change has renewed debate over the utility of the FRA for tracking global forest change (Friedl et al., 2010; Hansen et al., 2010, 2013; Tateishi et al., 2010; FAO and JRC, 2012, 2014; Sexton et al., 2013; Achard et al., 2014; Kim et al., 2015; Keenan et al., 2015). This is exemplified by assertions of large discrepancies between forest-change estimates of the FRA and satellite observations (Hansen et al., 2013; Kim et al., 2015). Estimates derived from different data, methods, and definitions of forest inevitably diverge, and this is also true of estimates derived from different satellite observations and even the same satellite sensor (Keenan et al., 2015), complicating assertions of superior accuracy.

Global forest-area estimates defined by five remotely-sensed studies for approximately 1990, 2000, 2005, and/or 2010 (Hansen et al., 2010, 2013; FAO and JRC, 2012, 2014; Gong et al., 2013) range from −21.4% to +2.2% of the estimates from the FRA 2015 for the same years (Keenan et al., 2015: Table 11). This range narrows considerably to −6.5% to +2.2% upon omitting Hansen et al. (2010) from consideration, and narrows further still to −6.5% to −2.6% upon considering only the two remotely-sensed estimates which are consistent with the FRA in defining forests as a land use[2] rather than as general tree cover (FAO and JRC, 2012, 2014). The reported accuracies of remotely-sensed estimates vary in the order of 80–95% plus a margin of error dependent on methodological factors including errors in classification, interpretation and sampling. No comparable accuracy assessment exists for FRA forest-area estimates.

In the tropical domain the range of forest-area estimates defined by six remote-sensing exercises for approximately 1990, 2000, 2005, and/or 2010 (FAO, 2001: Ch. 46; Hansen et al., 2010, 2013; FAO and JRC, 2012, 2014; Achard et al., 2014) is centred relatively evenly at −16.8% to +12% of the FRA 2015 estimate for the same years. The range for the tropics is no more narrow than the range for global estimates in spite of the fact that changes in forest use (as reported by the FRA) are believed to correspond most closely to changes in forest cover (as reported by satellite observations) in the tropics (Coulston et al., 2013). The range for the tropics similarly narrows to between −16% and −3.8% upon considering the three estimates which use forest definitions consistent with the FRA (FAO, 2001: Ch. 46; FAO and JRC, 2012, 2014), again suggesting that FRA estimates align well with reality at large scales. Comparisons of rates of forest loss within the tropical domain suggest more significant deviations between the FRA and remotely-sensed estimates (Keenan et al., 2015). However, different periods of observation and forest definitions preclude confident comparisons of rates of loss, especially since deforestation rates have varied over time (Keenan et al., 2015) and slowed considerably in some global sub-regions between successive FRA periods (Payn et al., 2015).

Forests which are naturally regenerating or subject to direct human influence are becoming increasingly important components of tropical landscapes (Lugo and Helmer, 2004; Chazdon et al., 2009; Meyfroidt and Lambin, 2011; Chazdon, 2014b,a; Schnell et al., 2015). Tree cover of >10% canopy cover increased by 13% (49 Mha) within nominally agricultural landscapes in South America over the 2000s according to satellite estimates (Zomer et al., 2014) whilst total forest area declined over this same period. In South America as well as South East Asia and Central America tree cover of >10% canopy occupies well over half of the total agricultural extent (Zomer et al., 2014), and may account

[2] The FRA defines forests as land *use*. This definition excludes lands predominantly dedicated to other land uses, such as agriculture, even where these have appreciable forest cover. Similarly it includes lands dedicated to forest production or conservation even where these are temporarily destocked. This definition is therefore distinct from one of forest as a land *cover*, that is, the simple presence of tree cover above a certain density.

for 16–21% of national above-ground tree biomass generally (Schnell et al., 2015: Table 3). It is therefore noteworthy that a proportion of the discrepancies between the FRA and satellite estimates reflect the fact that FRA forest-area estimates exclude tree cover in landscapes nominally managed for agricultural purposes even where canopy cover exceeds the 10% threshold applied elsewhere. Tree cover in agricultural and other non-forest landscapes was reported to the FRA by only 58% of countries for 2015 because many countries lack data (MacDicken pers. comm.). When such tree cover is reported it is aggregated with non-forest tree cover such as orange groves or oil palm and recorded as 'trees outside forests' (TOF) separately from total forests area (FAO, 2012a,c). The FRA 2015 reports 284 Mha of TOF globally, of which 214 Mha or 75% are in the tropics. In tropical Asia community forests and home-garden mosaics with dense tree cover occupy vast areas officially designated as agriculture (Collins et al., 1991; Zomer et al., 2014). The African context is similar, with extensive areas of orchard-bush in sub-humid areas. In Latin America there are large areas of spontaneous forest regrowth on semi-abandoned agro-pastoral lands (Hecht and Saatchi, 2007; Asner et al., 2009; Aide et al., 2013). According to Aide et al. (2013) some 22–36 Mha of net forest cover gain occurred on abandoned agricultural lands over 2000–2010 across Latin America. The correct categorisation of such tree cover is challenging where agricultural landscapes and tree cover therein are rapidly changing in their extent, form, and usage. The FAO provides countries with guidance on defining landscapes managed for agricultural purposes (FAO, 2012a,c) but the degree and manner to which regenerating and managed mosaic forest cover therein are recorded is variable and not well understood by data users.

The extent of forest mosaics and recent forest regeneration in nominally agricultural landscapes appears to be modest relative to the FRA estimates of tropical forest extent or reasonably well inventoried otherwise. No direct independent measures exist, however, and comparisons of FRA and other estimates of forest area and agricultural landscapes are complicated by differences in definitions, forest detection capacities and other methodological factors. For instance, the FRA 2015 tropical forest area having >10% canopy cover (1770 Mha) is 324 Mha less than the relatively inclusive tropical tree-cover estimates of Hansen et al. (2013) having >25% canopy covey (2094 Mha). This negative discrepancy suggests a first-order estimate of such mosaic and regenerating forests in the tropical domain. However, this estimate of 324 Mha overlooks the fact that the FRA reports some of the tree cover observed by Hansen et al. (2013) as 'other wooded land'[3] (OWL) rather than as forest. As with the FRA forest category, OWL excludes nominally agricultural landscapes. Unlike with the forest category, OWL includes an unknown and potentially large proportional area with canopy cover between 5% and 10%. Given 517 Mha of OWL in the tropics, the combined area of FRA tropical forest and OWL (2287 Mha) is actually 193 Mha greater than the Hansen et al. (2013) estimate, but this positive discrepancy would be reduced significantly and potentially become negative if the unknown proportion of OWL with <10% canopy cover was excluded. Therefore whilst no precise discrepancy can clearly indicate the extent to which the mosaic and regenerating forests in question are reported as TOF rather than as forest or OWL, its potential extent appears to be modest relative to the many hundreds of millions of hectares of forest cover of >10% canopy in agricultural landscapes pan-tropically (Zomer et al., 2014).

FRA estimates may be less robust for countries where the forest area is proportionally small, agricultural landscapes relatively dynamic, and extensive subsistence agricultural is widely practiced. El Salvador is illustrative, with only 14% forest area according to the FRA and ubiquitous agro-pastoral mosaics. Satellite observations indicate that El Salvador has experienced net gain in tree cover since the early 1990s following widespread regrowth on abandoned pastures (Hecht and Saatchi, 2007). In contrast, El Salvador reported steady deforestation over 1990–2015 to the FRA, mostly in areas of dense remnant forests (Hansen et al., 2013). El Salvador's so-called 'invisible forests' are therefore allocated largely to the FRA category 'trees outside of forest' (Hecht et al., 2006), where observed. This tendency has also been noted with FRA 2010 for Africa (Hansen et al., 2013), although there dry-forest contexts and extensive agro-pastoral systems introduce notable uncertainties in the remote detection and categorisation of forest area, particularly where forest-canopy thresholds are close to 10% (FAO and JRC, 2012: 23; Beuchle et al., 2015).

The current FRA converges with remotely-sensed studies more than previous FRAs. Many countries have improved forest monitoring and remote-sensing capacities relative to previous FRAs, most notably in Africa, and in so doing have enhanced the degree to which they capture recovering or disperse forest mosaics in managed landscapes. Countries that improved their monitoring capacities reported *net forest gains* for the FRA 2015, whereas previously they reported net forest losses (Romijn et al., 2015: Fig. 9). This trend is consistent with an enhanced detection and reporting of mosaic and recently regenerated forests as forest in the current FRA. There are also indications that improved capacity amongst forestry institutions correlates with better outcomes for forests (Keenan et al., 2015; MacDicken et al., 2015).

The time series FRA data has permitted numerous advances in the science and practice of forest management, including the documentation and understanding of recent declines in rates of tropical forest loss (e.g., Rudel, 1998). The FAO has invested heavily in negotiating forest definitions and norms with member countries for which definitions are politically sensitive and reflect the views of diverse stakeholders – there are more than 1500 definitions of the term "forest" (Lund, 2006). The current FRA definition of forest has been in use for 15 years, but it is inevitable that this definition does not reflect the understanding of the term forest as understood by all users of the FRA. The FRA definition of forest has nonetheless brought clarity to international debates on changes in forest resources. The FRA does reflect realities on the ground to the extent that these may be captured by aggregate national data, notwithstanding uncertainties concerning the categorisation of newly established forest cover in semi-abandoned agricultural landscapes. The FRA arguably remains the best dataset for tracking large-scale forest change trends since 1990 – a period for which there are very few large-scale, high-resolution remote-sensing estimates available – and significant changes in its methodology might actually prove *dis*advantageous because they would entail interruption of the unique FRA time series. To this end, in consideration of the fact that others will continue to produce global satellite-based estimates of tree cover, it is important that the FRA continue to provide long-term, authoritative series of estimates according to its existing standards. As illustrated below, such time series provide important insights into divergent trajectories of forest change.

3. Patterns of forest change, 1990–2015

Perhaps the most striking pattern revealed by the FRA 2015 is the dichotomy between forest gains and forest losses in different global sub-regions and climatic domains. FRA 2015 figures reveal

[3] Other wooded land is not considered as 'forest' by the FRA. Whilst it also spans 0.5 ha and reaches >5 m in height, it is defined by a tree canopy cover of only 5–10% *or* by a combined cover of shrubs, bushes, and trees of >10%, excluding lands predominantly under agricultural use.

Table 1
Change in the Area of Planted Forest, Natural Forest, and Total Forest, 1990–2015, by FRA Period and National Income Level.

Income per capita	Period	Change in Natural Forest Area (1000s of ha)	Change in Planted Forest Area (1000s of ha)	Change in Total Forest Area (1000s of ha)	Ratio of Change of Planted Forest to Natural Forest
High (>$12,746)	1990–2000	−8650	16,819	8171	−1.94
	2000–2005	−6773	8986	2212	−1.33
	2005–2010	316 6	9237	12,404	2.92
	2010–2015	−1138	4795	3656	−4.21
	1990–2015	**−13,396**	**39,836**	**26,443**	**−2.97**
Upper-Middle ($4125–$12,745)	1990–2000	−29,209	14,616	−14,593	−0.50
	2000–2005	−18,872	14,718	−4153	−0.78
	2005–2010	−13,214	9373	−3841	−0.71
	2010–2015	−7172	7954	782	−1.11
	1990–2015	**−68,467**	**46,661**	**−21,805**	**−0.68**
Lower-Middle ($1046–$4125)	1990–2000	−13,661	3473	−10,172	−0.25
	2000–2005	−5656	3352	−2308	−0.59
	2005–2010	−5762	3259	−2507	−0.57
	2010–2015	−6843	2820	−4022	−0.41
	1990–2015	**−31,922**	**12,904**	**−19,010**	**−0.40**
Low (<$1045)	1990–2000	−29,371	527	−28,835	−0.02
	2000–2005	−12,673	585	−12,096	−0.05
	2005–2010	−13,545	910	−12,635	−0.07
	2010–2015	−12,721	303	−12,002	−0.02
	1990–2015	**−68,310**	**2326**	**−65,568**	**−0.03**

Source: Data from the FRA of 2015. National income levels defined by World Bank (2013). See http://data.worldbank.org/about/country-and-lending-groups.
Notes: *N* = 126 countries. All countries were considered except those for which: (i) income level was not recorded (typically small island nations), (ii) FAO 'desktop' estimate were given in lieu of nationally-reported estimates, and (iii) valid values were not reported for the tabled variables for all FRA periods (notable countries include Australia, Cameroon, Great Britain, Indonesia, New Zealand, and Venezuela).

that forest loss is now occurring almost exclusively in the tropics, with either stable or expanding forest in other domains (Keenan et al., 2015). Half of the world's forest area is within sub-regions where forest extent has been declining since 1990 (Central and South America, South and South East Asia, and all three sub-regions of Africa), whilst the other half is within sub-regions where forest extent is stable or increasing.

National wealth is an apparent determinant of this dichotomy. Since 1990, richer countries have registered forest gains, poorer countries forest losses, and many middle-income countries have transitioned from forest loss to forest gain (Table 1) (Keenan et al., 2015). This is consistent with forest-transition theory (Rudel, 2005; Sloan, 2015) as well as environmental Kuznets curves (Mather et al., 1999; Culas, 2012). Both conceptualisations of forest change anticipate deforestation giving way to forest expansion as 'development' proceeds. Nevertheless, in the tropics at least, the gain-loss dichotomy is probably *driven* less by wealth than by the improved 'rational' allocation of land amongst forest and agricultural uses, often following economic shifts and forward-looking government interventions (cf. Whiteman et al., 2015). National wealth is not especially high amongst the 13 tropical countries identified as undergoing forest transitions since 1990 by the FRA 2015 (Keenan et al., 2015: Table 9). Only two of these countries have per capita GDP greater than the 75th percentile for all 142 tropical countries (Costa Rica, Puerto Rico), and only a further three have a per capita GDP greater than the 50th percentile (Cuba, Dominican Republic, India).

Since the 1990s most tropical countries apparently undergoing forest transitions have also undergone processes that impacted on the national culture and practice of forest management and land use, thus creating conditions wherein forest expansion could occur. Vietnam and Cuba underwent economic liberalisation entailing more efficient, flexible, and privately-owned agricultural enterprises resulting in the re-establishment of forests on lands hitherto dedicated to inefficient, often quasi-collective agriculture. Vietnam also invested heavily in industrial plantation forestry (Rosset and Benjamin, 1994; Rosset, 1998; Meyfoidt and Lambin, 2008; Payn et al., 2015). India instituted its Joint Forest Management programs and similarly encouraged private forestry, giving communities economic incentives for forest production and conservation (Foster and Rosenzweig, 2003) whilst also phasing out the exploitation of natural public forests (Jürgensen et al., 2014). Costa Rica pioneered the use of payments for environmental services to landholders in order to encourage reforestation and forest conservation (Pagiola, 2008), and recently introduced laws to protect regenerating forests from re-clearance (Fagan et al., 2013). Puerto Rico underwent rapid urbanisation as a result of its special economic and migratory relationship with the USA (Rudel et al., 2000). Such shifts in the culture and practice of forest and land management have only indirect relationships with wealth per se. Much of their effect on forest expansion was indirect and accidental, and arguably stemmed from changes to underlying economic conditions unamenable to forest conservation and regeneration. This raises the question as to whether the resultant expansion of forest area was simply a fortuitous 'one-off' phenomenon (Müller et al., 2014; Sloan, 2015). Ongoing forest expansion in northern nations and the parallels between temperate and tropical forest transitions gives some reason to believe forest expansion in tropical countries with growing economies will continue.

The literature on the socio-economic forces underlying shifts to forest expansion and conservation has neglected the role of planted forests and the economics of forest production (Meyfroidt and Lambin, 2011). Yet the FRA 2015 clearly illustrates that planted forests[4] account for much of the world's total forest-cover change, particularly in countries with positive or low rates of forest change. Indeed planted forest account for ~25–100% of the gains in forest area since 1990 in many of the tropical forest transition countries (Fig. 1), suggesting that planted-forest expansion is a significant and overlooked factor in forest transitions. Globally, countries in each of the four per capita income categories recognised by the FRA expanded their planted-forest extents over all four FRA periods since 1990, with expansion rates peaking in

[4] 'Planted forests' encompass a variety of forest types. The FRA 2015 defines them as "predominantly composed of trees established through planting and/or deliberate seeding" (www.fao.org/fra/fra2015). Planted forests are inclusive of all types of plantings, including semi-natural forests, that meet the FRA definition of forest (Jürgensen et al., 2014).

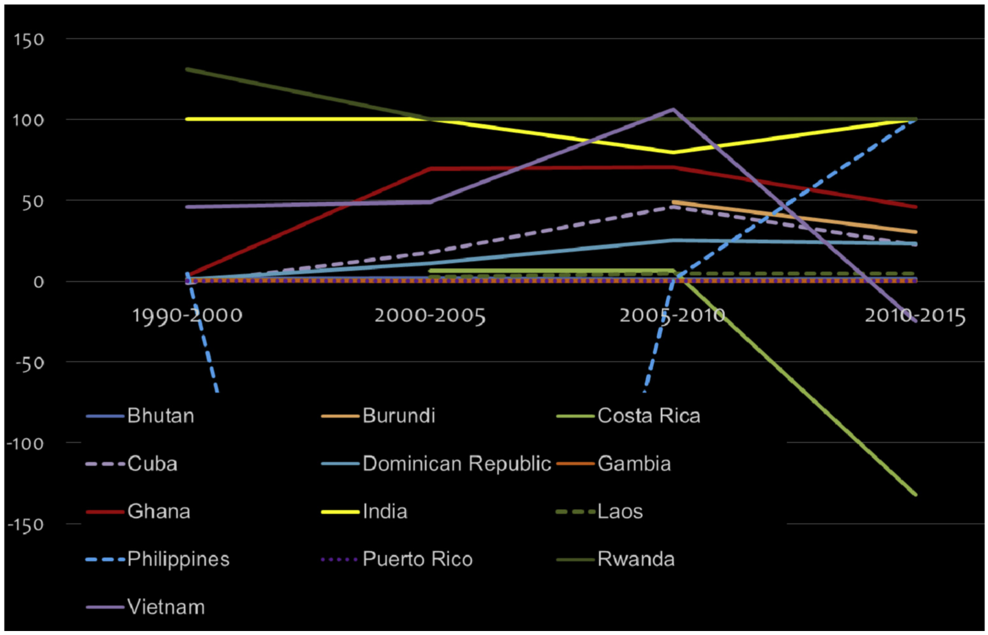

Fig. 1. Change in Planted Forest Area as a Percent of Change in Total Forest Area, for Tropical Forest-Transition Countries Identified by the FRA 2015, by FRA Period, 1990–2015.Source: Data from the FRA of 2015. Notes: Graphed data pertain only to countries and FRA periods for which total forest-area change was positive. Negative percentages indicate that a loss of planted forest area coincided with a gain of total forest area. Negative measures are not strictly percentage measures per se but are graphed here for consistency. *Philippines*: The unseen negative measure for 2000–2005 is −587%. *Vietnam:* The sharp downward trend after the 2005–2010 FRA period owes to a relatively small decline (−160,000 ha) in planted-forest area between the periods 2005–2010 and 2010–2015, coincident with a continued increase in total forest area. Total forest-area change was slightly negative for 2005–2010 (−0.89%) but the graphed measure was retained in to permit an unbroken trend line. *Puerto Rico*, *Gambia*, *Laos*, and *Costa Rica*: Have measures of ~0% for the time series (or for 2000–2005 and 2005–2010 for Costa Rica), reflecting largely natural forest gain over these periods.

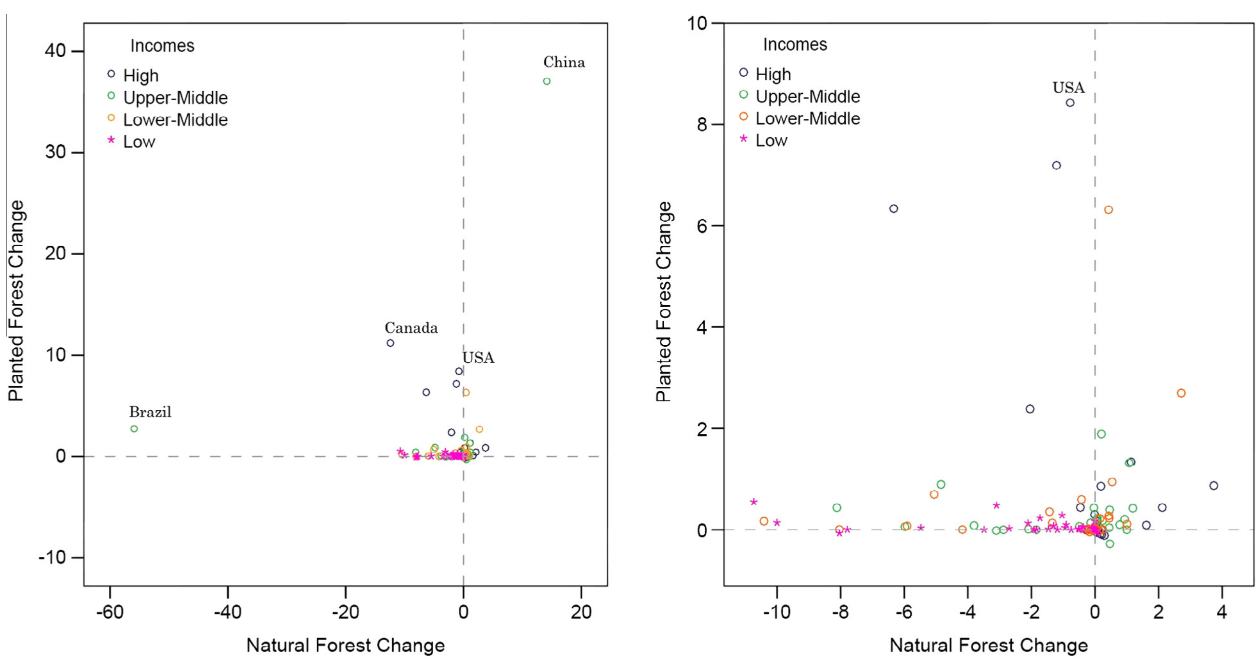

Fig. 2. Changes in the Area (Mha) of Natural Forest and Planted Forest, by Income Level and Country, 1990–2015.Source: Data from the FRA of 2015. Notes: Countries were selected in the same manner as described for Table 1.

the mid-2000s (Table 1). Only the 'high' and 'upper-middle' income categories experienced net forest gains, however, and only these countries consistently established planted forests at higher rates both absolutely and relative to rates of total forest change (Table 1). The increase in planted-forest area for high and upper-middle income countries was driven predominantly by China, Canada, the USA, Russia and Sweden, where gains in planted-forest area more than offset losses of natural forest (excepting China, where natural forests also increased (Liu et al., 2008)) (Fig. 2). In most other high and upper-middle income countries both natural-forest and planted-forests areas increased simultaneously, distinguishing such countries from lower-income countries (Fig. 2).

The historical situation whereby richer, temperate countries contained most planted forests is giving way to the reality that expanding populations and economies in the global South are

Table 2
Multiple-use Forest Area by Climate Domain, as Percent Total Forest Area, 1990 and 2015.

	Tropical		Subtropical		Temperate		Boreal		Polar	
	Mha	%	Mha	%	Mha	%	Mha	%	Mha	%
2015	287.4	17	98.1	39	159.6	25	503.6	41	0.0002	100
1990	254.9	16	19.9	35	154.1	26	538.1	44	0.0002	100

Source: Data from the FRA of 2015.
Notes: Percentages pertain to countries reporting multiple-use forests, numbering 183 for 2015 and 174 for 1990. The countries for 2015 account for 95% of global forest area.

rapidly establishing forests in response to market opportunities. Whilst the expansion of planted-forest area since 1990 was greatest in temperate countries (including China), planted-forest expansion in East Asia (led by China) and South and South-East Asia (led by India, Thailand, and Indonesia) was equal to or larger than expansion in Europe (including Russia) and North America (Payn et al., 2015). Globally planted forests are increasingly playing a role in balancing competing demands for forest conservation, amenity, and production.

4. Meeting the world's needs for forest goods and services

Much environmental discourse implicitly presupposes that maximising forest extent will provide the greatest global benefit (cf. Griscom, 2014; Planet Experts, 2014; Unger, 2014). Maximising forest area is a concern amongst conservationists (Edwards et al., 2011; Gibson et al., 2011) and a major issue in the climate change debate. Political and some corporate interests are committing to 'zero deforestation' (Butler, 2013; Lister and Dauvergne, 2014), and significant international financing is assisting tropical countries to develop and implement REDD+ programs and related forest monitoring schemes to reduce deforestation and increase afforestation (e.g., Satgas REDD+, 2012). The governments of some tropical countries may subscribe to this rhetoric, but many stakeholders, including the resource poor people who live at the forest margins, often have very different perspectives on what constitutes the optimal extent and type of forest cover (Poore and Sayer, 1991). The sufficiency of the global forest estate is better considered in terms of the provision of multiple services, e.g., carbon, timber and biodiversity protection, which the 2015 FRA has attempted to inventory. From this perspective there are indeed signs of an impending shortfalls of tropical forest area on multiple fronts as increasingly more services are demanded of a decreasing forest estate (Millennium Ecosystem Assessment, 2005). We are witnessing the transformation of forest management, conservation, and production so as to satisfy multiple goals simultaneously from the same landscape (Sayer et al., 2008, 2009).

An appreciable portion of the global forest estate is already designated for multiple use allowing both production and conservation without prioritising either. Globally 1.049 Mha or one quarter of the forest estate were designated as multiple-use forests in 2015, an increase of 81.8 Mha ha (8.5%) since 1990 (Köhl et al., 2015). In contrast, the area of protective forests assessed by the FRA – those dedicated to protecting soils, water, and certain ecological and cultural values – is at least as extensive as multiple-use forests but did not meaningfully increase since 1990, potentially because the protective-forest designation may preclude forest exploitation in some countries (Miura et al., 2015). Multiple-use forests registered the greatest gains in South America (87.5 Mha), Oceania (52.5 Mha), and East Asia (48.3 Mha), and registered the greatest declines in South and Southeast Asia (−56.5 Mha), Europe (−31.9 Mha), and North America (−19.6 Mha). Regional differences in the area of multiple-use forests relative to total forest area are great and reflect differences in forest production and management practices. In North America and Europe (including Russia), where many forests are under explicit 'nature-oriented' production regimes (Nabuurs et al., 2007, 2014), multiple-use forests account for 58% and 24% of the total forest estate, respectively, whereas in South America they account for only 13% despite having expanded 5.5-fold there since 1990. Parallel differences are also apparent amongst the climatic domains, with the tropics having a much smaller area of multiple-use forests as a proportion of total forest area (17%) compared to other domains (Table 2). Although the proportional area of multiple use forests is low in the tropics the absolute area of tropical multiple-use forests is still relatively large, at 287 Mha (Table 2).

Conservationists are concerned with the limited extent of primary and quality secondary forest habitat, particularly in humid and seasonally-dry tropical forests (Jepson et al., 2001; Miles et al., 2006; Ribeiro et al., 2009; Portillo-Quintero and Sánchez-Azofeifa, 2010) where most of the world's biodiversity resides. Agricultural expansion threatens these forests with conversion (Laurance et al., 2014b). Modelling by d'Annunzio et al. (2015) suggests that agricultural expansion to 2030 threatens a small fraction (2%) of the world's largest expanses of intact primary tropical forests, thereby implying that most threatened forests will be relatively fragmented, degraded, and proximate to human activity. This estimate by d'Annunzio et al. (2015) may be conservative given the 6% loss of total primary tropical forest area since 2000 (Morales-Hidalgo et al., 2015: Fig. 5) and the fact that modelling by d'Annunzio et al. (2015) does not account for the penetration of forests by new roads. Sayer and Cassman (2013) argue that future agricultural demand for land could be met without significant forest loss if agricultural innovations were deployed.

Even if tropical forest loss could be halted the current area and distribution of quality forest habitat is already insufficient for many forest species. The global biodiversity hotspots retain only 15% of intact natural vegetation (Sloan et al., 2014). One response to this situation has been a dramatic increase in protected area coverage since 1990. The proportion of forest area within protected areas has risen to 7.7–16.3% (308–517 Mha) globally and 10.8–26.6% (128.5–379 Mha) in the tropics[5] (Schmitt et al., 2009: Table 1; Morales-Hidalgo et al., 2015) (see also Watson et al., 2014). Yet, due to the tendency for protected areas (PAs) to be situated far from threats to forests and biodiversity (Joppa and Pfaff, 2009; Sloan et al., 2012), and given their inevitably limited extent, this expansion of protected-forests has been inadequate to safeguard biodiversity and ecological services generally (Millennium Ecosystem Assessment, 2005; Watson et al., 2014; Miura et al., 2015). Some 17% of threatened

[5] Upper estimates are from the 2015 FRA, and lower estimates derived from remotely-sensed data tabled in Schmitt et al. (2009). As with all comparisons of FRA and remotely-sensed data, differences in forest definitions must be considered. Schmitt et al. (2009) attempted to align their definition with the FAO's definitions and defined forest as >10% tree cover according to 2005 MODIS satellite imagery. However these definitions are still not totally comparable as FAO defines forest as a land use, not a land cover. Protected areas in Schmitt et al. (2009) have IUCN categories I-VI exclusively, and whereas the FRA's definition of protected areas is the same in this regard it may also encompass other areas (FAO, 2012a). Some countries, the USA for example, report forest reserves as protected areas whereas other countries do not. The extent of the 'tropics' in Schmitt et al. (2009: Table 1) is defined with respect to climate and geography whereas FAO allocates countries to a single climate domain as per FAO (2012b) even though many countries span several climatological regimes.

bird, mammal, and amphibian species (n = 4118) are not found in any PA (n > 162,906), and 85% are so poorly represented across PAs that they may not survive in the long-term (Venter et al., 2014), a situation that has deteriorated over the past decade (Rodrigues et al., 2004). Human pressures have increasingly isolated PAs and other intact forests (DeFries et al., 2005; Potapov et al., 2008). PAs are often remnants left after other land use needs have been met and are inadequate to offset a much more generalised decline in the integrity and extent of the larger forest estate.

Increasing demand for forest and agricultural products threatens to undermine efforts to arrest biodiversity decline and maintain the integrity of the forest estate (Laurance et al., 2014b). Demographic and economic growth have historically driven deforestation, forest exploitation, and agricultural demand (South, 1999; Wright and Muller-Landau, 2006; Cohen, 2014a,b), and such growth will continue rapidly in the developing world. Just as European consumption of forest products increased ∼50% with increasing prosperity in the latter half of the 20th Century[6], emergent economies are rapidly increasing their consumption today. Combined industrial and fuelwood removals in the tropics increased by 35% (3,928,650 m^3) over 1990–2015 or 1.4% per annum whilst either holding constant or declining slightly in the other climatic domains, so that tropical domain is currently the greatest source of removals globally. Similarly, industrial and fuelwood removals over 1990–2015 increased most rapidly in lower-middle and lower income countries (Köhl et al., 2015), where economic and demographic growth has been greatest (Wilson et al., 2010; United Nations, 2013). China increased its share of global log imports nearly three-fold to 37–50% over the last decade (Dieter, 2009; Cohen, 2014b), and demand for wood products in the Asia-Pacific region is projected to rise 80% by 2030 (FAO, 2008, 2009). Economic growth in the developing world is projected to double global consumption of forest products by 2030 (WWF and IIASA, 2012: Ch4. p. 9). Demand for industrial forest products in Asia-Pacific and Africa[7] may exceed forest production by 89 million m^3 by 2020, whereas Latin America may enjoy a modest surplus of 17 M m^3, albeit still accompanied by forest conversion to meet agricultural demand (FAO, 2008, 2009). Indeed, projections to 2030 suggest that 26% and 15% of current production forests in Latin America and Africa respectively are likely to be converted for agricultural (d'Annunzio et al., 2015: Table 5).

The apparent contradiction of continued globally-significant declines in forest biodiversity despite increases in protected areas highlights the shortcomings of a narrow focus on forest extent. The area of forest allocated for conservation, which is relatively easily measured, is important but equally important is the degree to which these forests are intact, include representative forest types, are inter-connected and subject to effective protection. For the first time the FRA 2015 has provided global figures on partial canopy cover reduction (PCCR), defined as the loss of >20% of tree cover between 2000 and 2012, presumably a result of forest degradation by fire, wood removals, small clearances, insect damage, and so on. Van Lierop and Lindquist (2015) report that the area subject to PCCR in the tropical domain is 6.5 times that deforested since 1990, and up to 15 times greater for South and Southeast Asia. Whilst the ratio of PCCR to deforestation is actually higher in boreal and subtropical domains than in the tropical domain, this is because deforestation in the former domains was negligible, not because PCCR was extensive.

The extensiveness of reduced-canopy forest cover in the tropics is of concern for various reasons. Degraded forests generally provide fewer environmental services such as biodiversity than intact natural forests (Gibson et al., 2011). More significantly the inclusion of reduced-canopy cover forests within the FRA forest category may encourage the mistaken impression that the natural forest extent is adequate, or at least that the forest conservation situation is better than it actually is, leading to complacency. As yet the PCCR estimates do not provide time-series trends for forest degradation so the long term fate of PCCR forests is not known. Of more immediate concern is the fact that several prominent environmental NGOs (e.g., Greenpeace) and the agri-businesses they have influenced (e.g., Golden Agri-Resources, Syme Derby, PT Smart) have begun assessing forests' conservation value and priority for conversion according to their carbon stock in efforts to direct conversion towards degraded forests (Dinerstein et al., 2014). The application of High Carbon Stock (HCS) assessments in particular is being promoted as a precondition for tropical forest conversion (Greenpeace SE Asia, 2014). The problem here lies in the fact that forests subject to relatively minor disturbances such as selective logging or long-cycle shifting agriculture may still retain high conservation value and will regain much of their former biodiversity value if allowed to recover (Meijaard et al., 2005; Edwards et al., 2011). Reduced-canopy cover tropical forests are already extensive and agricultural corporations now have a perverse incentive to degrade forests so as to then legitimately obtain them for conversion. Stakeholders should be cautious in considering the merits of conversion on the single criterion of reduced carbon stock.

Increasing demand for forest products and services as well as for forest land for agriculture is adding to the difficulty of mitigating global climate change via REDD+ initiatives. Analyses of FRA 2015 data (Federici et al., 2015; Köhl et al., 2015) and FRA 2010 data (Pan et al., 2011; Reich, 2011) indicate that, despite the loss of 10% (195 Mha) of the tropical forest estate since 1990, tropical forests are only *marginally* a net source of atmospheric carbon emissions. Carbon emissions from deforestation have been offset almost entirely by carbon sequestration due to forest regeneration as well as increases in forest carbon density as a result of 'carbon fertilisation' (Fang et al., 2014). Significant proportions of total national-level above-ground woody biomass occur as tree cover in non-forest lands such as nominally agricultural landscapes such that estimates of carbon flux may be sensitive to definitions of forest (Schnell et al., 2015). However it remains difficult to predict whether the reported gap between gross tropical carbon emissions and removals would broaden or narrow upon incorporating trees outside of forests. Tropical forests are now the only forests yielding net carbon emissions according to FRA data and could become a net carbon sink if sustainable forest management were applied only slightly more widely (Pan et al., 2011; Reich, 2011). The role of tropical forests in the global carbon budget will depend mainly on the rate of future conversion of forests to agriculture, the extent of which will be determined by delicate and unforeseeable compromises between competing forces (FAO, 2008, 2009; WWF and IIASA, 2012; Laurance et al., 2014b). The projected conversions of production forests to agriculture and a corresponding intensification of production forestry in Latin America and Africa would almost certainly entail net emissions, whereas achieving the national targets of net forest-area increases across Asia and Africa would entail the contrary (see Jürgensen et al., 2014 for China; d'Annunzio et al., 2015: Table 6). Tropical forests still have by far the smallest proportion of their total area under an intermediate or good level of sustainable forest management, at 23% (MacDicken et al., 2015: Fig. 2) High standards of sustainable management greatly increase the likelihood that a forest will serve as carbon sink (Putz and Pinard, 1993) so there is much scope for gain in this respect. Large areas of tropical forest are legally designated

[6] Whilst consumption increased 54% over 1964–2000, natural forest exploitation increased only 9%, and the total forest supply of timber increased, largely due significant increases in the efficiency of forest harvest, forest-product manufacture, recycling as well as forest protection (Nabuurs et al., 2007).

[7] Production and consumption in Africa is much lower than in the Asia-Pacific region. The African contribution to the 2020 production deficiency reported here is probably under-estimated because these figures do not consider fuelwood, which accounts for ∼88% of total wood removals in Africa (FAO, 2010: Table 5.11).

for sustainable permanent use but in practice remain vulnerable to unchecked exploitation (Blaser et al., 2011; MacDicken et al., 2015), and indeed virtually all tropical forest loss since 1990 has occurred in state lands (Whiteman et al., 2015: Table 2).

4.1. The growing role of planted forests

Growing demand for forest products makes a strong case for expanding planted forests. Planted forests may reduce pressure on natural forests (South, 1999), support biodiversity conservation (Sayer et al., 2004; Sayer and Elliot, 2005; Brockerhoff et al., 2008), and actively remove atmospheric carbon (Nilsson and Schopfhauser, 1995). There is significant scope to increase production from planted forests in several subregions according to the best available sub-regional data on timber production from planted forests and from all forests combined (natural and planted), which collectively account for 60% of global timber production from natural and planted forests. Planted forests contribute a low to moderate proportion of total production in Africa (10–30%, depending on the sub-region), Central America (34%), and South East Asia (49%) (see also Jürgensen et al., 2014; d'Annunzio et al., 2015: Table 1). Modelling based on FRA and FAOSTAT data indicates that if the global extent of planted forests were to increase at 2.4% per annum between 2010 and 2050 planted forests could replace natural forests as a source of timber and fibre (WWF and IIASA, 2012: Ch.4). This assumes that 80% of current planted forests are dedicated to production.[8] This target rate it is not dramatically different to the 2.05% per annum increase in planted forest expansion reported by the FRA for 1990–2015. However, the actual global rate of planted-forest establishment has fallen to 1.5% per annum since 2005. Target rates are higher in the Asia-Pacific region (excluding China) and Latin America, at 3.2% and 3% per annum respectively, as compared to their still appreciable actual rates of planted-forest expansion of 2.25% and 2.4% per annum for 1990–2015, underscoring the magnitude of the challenge of meeting wood demands exclusively from planted forests.

An expanded planted-forest estate is ultimately only one means of many necessary to achieve better natural forest conservation and management. There is a time lag between establishing planted forests and achieving significant wood production and carbon sequestration at a national to regional scale (Nilsson and Schopfhauser, 1995; Nabuurs et al., 2007, 2014). An extensive or rapidly expanding planted-forests estate must complement sustainable forest management more generally to yield functional, sustainable landscapes providing a full range of forest products and environmental services (Sayer et al., 2004). In South America, where planted forests meet 77–88% of total wood demand (Jürgensen et al., 2014; d'Annunzio et al., 2015: Table 1), deforestation is still greater than in other continents both absolutely and relatively in part because only 15% of forests there are subject to an intermediate level of sustainable forest management or better (MacDicken et al., 2015: Fig. 5). Modest areas of multiple-use forests and lower grades of sustainable forest management in many low-income and tropical countries suggest that improved management could allow production from natural forests to increase without undue threats to forest conservation, integrity, and biodiversity.

Low-income countries and the tropics have by far the greatest scope to improve sustainable forest management (SFM) (MacDicken et al., 2015). All forest areas are subject to national policies and legislation which promote SFM but low-income countries are in general failing to enact policy and legislation to the local scale, with potential implications for local conservation and production. National forest inventories (NFIs) and forest management plans (FMPs) appear critical for practical near-term improvement in SFM in such countries. Presently 309 Mha of permanent tropical forest is subject to policy and legislation but not covered by NFIs, a far greater area than in other climatic domains absolutely and relative to total forest area. Fortunately NFIs and FMPs are responsive to support from the international community and are efficacious means of advancing various aspects of SFM. The number of countries with NFIs more than doubled to 112 since 2010 thanks largely to support by UN-REDD+ and the FAO in poorer countries, although coverage remains low in Africa. Where implemented, FMPs have explicitly considered high-conservation value forest and community involvement in over 90% of their area, suggesting they may serve as portals to broad and effective SFM generally.

Care must be taken that any concentration of production in a planted-forest estate does not devalue natural forests and discourage investment in their sustainable management. In Borneo the designation of natural production forest has historically prevented forest conversion as effectively as protected forests (Curran et al., 2004; Gaveau et al., 2013). With the depletion of many natural production forests following decades of encroachment, fire, and logging, forest royalties declined correspondingly (ITS Global, 2011; Ministry of Forestry, 2012; Gaveau et al., 2014). The Indonesian government has since enacted policies to encourage the establishment of forest plantations, and in many cases these have replaced degraded natural production forests (Gaveau et al., 2014: Table S5). Agricultural expansion has similarly often occurred on poorly stocked or otherwise degraded natural forests (Abood et al., 2015). Should fibre production increasingly derive from planted forests investments in sustainable forest management in natural forests must arguably give increased priority to environmental, rather than economic, values, as by payments for environmental services.

FRA data on public expenditures on forests are mute in regards to the degree to which expenditure is directed towards natural or planted forests or towards conservation or production, as these targets are highly intertwined at the national scale. The case of China is illustrative. Expenditures in temperate countries including China increased significantly over 2000–2010 from $9.6 billion to $25 billion whilst public revenues from forests were much lower and increased relatively modestly from $1.1 billion to $2.7 billion (Whiteman et al., 2015). Superficially this dominance of expenditures over revenues, which is by far the greatest in the temperate domain, suggests significant investment in forest management and monitoring apart from production per se. China accounts for 62% of this growth in expenditure in the temperate domain and just less than half of its total expenditure in 2010. In China forest area increased since 2000 through state-sponsored initiatives such as the National Forest Conservation Program and 'Grains for Green' programs entailing natural forest restoration, payments for environmental services, reduced natural forest harvests and planted-forest establishment for environmental objectives (Zhang et al., 2000; Xu et al., 2007; Liu et al., 2008). The programs also entailed massive subsidised expansions of timber plantations for industrial production (Fig. 2). In China at least, the significant economic value invested in planted forests for production is inseparable from the significant non-economic values recently assigned to natural forests following their widespread degradation (Liu et al., 2008). In the tropical domain, where planted forests are less extensive, forest revenues exceeded expenditures and both increased far more modestly than in the temperate domain, at +$2.9 billion to $5.5 billion and at +$0.5 billion to $1.1 billion, respectively (Whiteman et al., 2015).

[8] Using earlier FAO data Sayer and Elliot (2005) report that 46% of the global planted-forest estate is explicitly dedicated to industrial production (e.g. timber, fibre), 26% is dedicated to nonindustrial uses (e.g., fuelwood, soil protection), and the remaining 26% has an unclear purpose, almost certainly being a mix of industrial and nonindustrial uses. Similarly, FAO (2010: 339) observes that 'productive plantations' accounted for ~79% of the global planted-forest estate in 2005, with the balance accounted for by protective planted forests.

5. Novel forests landscapes

In its earliest days the FRA focussed narrowly on the management of forests as sources of industrial timber but has since progressively adapted its scope to reflect a much broader range of forest attributes and values. This trend must continue as the demands that societies place upon forests continue to grow and vary. Forests are ever more scarce, contested, and valuable, and forest landscapes are ever more diverse and novel as they react to different societal needs and occur in changing environmental conditions (Lugo, 2009). The extent of natural and semi-natural forests is growing in many human-dominated, multi-functional landscapes (Aide et al., 2013; Zomer et al., 2014), and these forests are playing a growing role in sustaining production and environmental services (Brockerhoff et al., 2008; Sayer et al., 2013). Landscapes are emerging as an important organising framework for the provision of multiple forest services and are the object of significant international debate. Recently global landscape fora have taken place in association with the UNFCCC Conferences of the Parties and several international bodies have committed themselves to a Global Programme for Forest Landscape Restoration (Global Landscapes Forum, 2014). These novel forest landscapes will be difficult to accommodate within the current FRA reporting framework and will similarly present challenges for remotely-sensed estimates. Metrics for assessing the diverse conservation and production values of trees and forests in mosaic landscapes are currently inadequate and urgently needed.

Tensions exist between the discourse emphasising global forest conservation benefits and the need to alleviate the poverty of local forest dwellers (Kremen et al., 2000). These tensions signal 'wicked' problems where achieving agreement on even the nature of the problem is difficult and where solutions for one stakeholder create problems for another (Giller et al., 2008; Stewart et al., 2011). There has been a recent tendency to decentralize the locus of decision making on forests towards the local scale (MacDicken et al., 2015), typically with a good measure of improvisation as decision makers struggle with notions of 'balance' (Sayer et al., 2008). Countries will make their own decisions on how much forest to preserve how it should be managed as their needs evolve in the 21st Century. The FRA has provided the main inter-governmental framework for debate on forest needs and values and it should continue to do so as those values evolve over time. Forest goods and services in future are likely to derive from an increasingly wider range of land cover types and jurisdictions than presently. Private forests, community forests, planted forests, and trees outside forests will need to be distributed in multi-functional landscapes in ways that optimise the provision of the goods and services that societies will need. The challenge in the future will not be to quantify forest-area changes per se but rather to assess the ability of landscapes to meet a diversity of societal needs. Difficult choices will need to be made at all spatial scales, from local to global, and metrics that measure the ability of the broader landscape to provide a balance of goods and services will enable better decisions on these choices.

5.1. An expanding need for Forest Resources Assessments

The greatest challenge to improved global forest management is ultimately not the information provided by the FRA and similar sources but rather the capacity of institutions and civil society in low-income countries to exploit this information. The FRA is the apex of a hierarchy of management tools extending down to sub-national inventories and local consultations. Where forests continue to decline in poorer tropical countries there is often a weak link between national and local levels. Forest loss and degradation is mostly a result not of economic and demographic forces per se but of institutional failures to contain such forces. Information alone cannot resolve the tensions between values attributed by different stakeholders or the power differentials between local people and corporations. But the FRA continues to have an essential role in reconciling such tensions, both by stimulating discussion amongst stakeholders nationally during the FRA exercise (MacDicken, 2015) and by providing an over-arching framework under which global, national, and local management initiatives may flourish. The ultimate challenge is not to know how much forest exists but rather to what degree forests are meeting the varied needs of humanity.

Acknowledgements

Sean Sloan is supported by an ARC – Australia Laureate Fellowship awarded to Prof. William Laurance. We thank Ken MacDicken of the FAO and two anonymous reviewers for valuable comments on an earlier draft of this paper.

References

Abood, S.A., Huay Lee, J.S., Burivalova, Z., Garcia-Ulloa, J., Koh, L.P., 2015. Relative contributions of the logging, fibre, oil palm and mining industries to forest loss in Indonesia. Conserv. Lett. 8, 58–67.

Achard, F., Beuchle, R., Mayaux, P., Stibig, H.J., Bodart, C., Brink, A., Carboni, S., Desclée, B., Donnay, F., Eva, H.D., 2014. Determination of tropical deforestation rates and related carbon losses from 1990 to 2010. Glob. Change Biol. 20, 2540–2554.

Aide, T.M., Clark, M.L., Grau, H.R., López-Carr, D., Levy, M.A., Redo, D., Bonilla-Moheno, M., Riner, G., Andrade-Núñez, M.J., Muñiz, M., 2013. Deforestation and reforestation of Latin America and the Caribbean (2001–2010). Biotropica 45, 262–271.

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

Asner, G.P., Rudel, T.K., Aide, T.M., DeFries, R., Emerson, R., 2009. A contemporary assessment of global humid tropical forest change. Conserv. Biol. 23, 1386–1395.

Asner, G.P., Llactayo, W., Tupayachi, R., Luna, E.R., 2013. Elevated rates of gold mining in the Amazon revealed through high-resolution monitoring. Proc. Natl. Acad. Sci. USA 110, 18454–18459.

Beuchle, R., Grecchi, R.C., Shimabukuro, Y.E., Seliger, R., Eva, H.D., Sano, E., Achard, F., 2015. Land cover changes in the Brazilian Cerrado and Caatinga biomes from 1990 to 2010 based on a systematic remote sensing sampling approach. Appl. Geography 58, 116–127.

Blaser, J., Sarra, A., Poore, D., Johnson, S., 2011. Status of Tropical Forest Management 2011. ITTO Technical Series, No. 38. The ITTO. Yokohama, Japan.

Brockerhoff, E.G., Jactel, H., Parrotta, J.A., Quine, C.P., Sayer, J., 2008. Plantation forests and biodiversity: oxymoron or opportunity? Biodivers. Conserv. 17, 925–951.

Butchart, S.H., Walpole, M., Collen, B., van Strien, A., Scharlemann, J.P., Almond, R.E., Baillie, J.E., Bomhard, B., Brown, C., Bruno, J., 2010. Global biodiversity: indicators of recent declines. Science 328, 1164–1168.

Butler, R., 2013. The beginning of the end of deforestation in Indonesia? <http://news.mongabay.com/2013/0205-app-forest-policy.html> (accessed December 2014).

Chazdon, R.L., 2014a. PARTERS: Building a Socio-Ecological Understanding of Reforestation in the Tropics. <http://peoplefoodandnature.org/blog/partners-building-a-socio-ecological-understanding-of-reforestation-in-the-tropics/> and <http://partners-rcn.uconn.edu/page.php?4> (accessed 06.01.15).

Chazdon, R.L., 2014b. Second Growth: The Promise of Tropical Forest Regeneration in an Age of Deforestation. University of Chicago Press, Chicago.

Chazdon, R.L., Harvey, C.A., Komar, O., Griffith, D.M., Ferguson, B.G., Martínez-Ramos, M., Morales, H., Nigh, R., Soto-Pinto, L., Van Breugel, M., 2009. Beyond reserves: a research agenda for conserving biodiversity in human-modified tropical landscapes. Biotropica 41, 142–153.

Cohen, D., 2014a. Drivers of future forest use. Paper presented at The XXIV IUFRO World Congress. 'Sustaining Forets, Sustaining People: The Role of Research', Salt Lake City, USA, 7 October 2014.

Cohen, D., 2014b. Forces shaping the globe's forests and forest use. In: Nikolakis, W., Innes, J. (Eds.), Forests and Globalization: Challenges and Opportunities for Sustainable Development. Routledge, New York (Ch. 2).

Collins, N.M., Sayer, J.A., Whitmore, T.C., 1991. The Conservation Atlas of Tropical Forests: Asia and the Pacific. Macmillan Press Ltd.

Coulston, J.W., Reams, G.A., Wear, D.N., Brewer, C.K., 2013. An analysis of forest land use, forest land cover and change at policy-relevant scales. Forestry, cpt056.

Culas, R.J., 2012. REDD and forest transition: tunneling through the environmental Kuznets curve. Ecol. Econ. 79, 44–51.

Curran, L.M., Trigg, S.N., McDonald, A.K., Astiani, D., Hardiono, Y.M., Siregar, P., Caniago, I., Kasischke, E., 2004. Lowland forest loss in protected areas of Indonesian Borneo. Science 303, 1000–1003.

d'Annunzio, R., Sandker, M., Finegold, Y., Min, Z., 2015. Projecting global forest area towards 2030. For. Ecol. Manage. 352, 124–133.

de Wasseige, C., Devers, D., De Marcken, P., Eba'a Atyi, R., Nasi, R., Mayaux, P., 2010. The Forests of the Congo Basin: State of the Forest 2008. Publications Office of the European Union, Luxembourg.

DeFries, R., Hansen, A., Newton, A.C., Hansen, M.C., 2005. Increasing isolation of protected areas in tropical forests over the past twenty years. Ecol. Appl. 15, 19–26.

Dieter, M., 2009. Analysis of trade in illegally harvested timber: Accounting for trade via third party countries. For. Econ. 11, 600–607.

Dinerstein, E., Baccini, A., Anderson, M., Fiske, G., Wikramanayake, E., McLaughlin, D., Powell, G., Olson, D., Joshi, A., 2014. Guiding agricultural expansion to spare tropical forests. Conserv. Lett. http://dx.doi.org/10.1111/conl.12149.

Durán, A.P., Rauch, J., Gaston, K.J., 2013. Global spatial coincidence between protected areas and metal mining activities. Biol. Conserv. 160, 272–278.

Edwards, D.P., Larsen, T.H., Docherty, T.D.S., Ansell, F.A., Hsu, W.W., Derhé, M.A., Hamer, K.C., Wilcove, D.S., 2011. Degraded lands worth protecting: the biological importance of Southeast Asia's repeatedly logged forests. Proc. R. Soc. Lond., B 278, 82–90.

Edwards, D.P., Sloan, S., Weng, L., Sayer, J., Dirks, P., Laurance, W.F., 2014. Mining and the African environment. Conserv. Lett. 7, 302–311.

Fagan, M.E., DeFries, R.S., Sesnie, S.E., Arroyo, J.P., Walker, W., Soto, C., Chazdon, R.L., Sanchun, A., 2013. Land cover dynamics following a deforestation ban in northern Costa Rica. Environ. Res. Lett. 8, 034017.

Fang, J., Kato, T., Guo, Z., Yang, Y., Hu, H., Shen, H., Zhao, X., Kishimoto-Mo, A.W., Tang, Y., Houghton, R.A., 2014. Evidence for environmentally enhanced forest growth. Proc. Natl. Acad. Sci. USA 111, 9527–9532.

FAO, 2001. Global Forest Resource Assessment 2000. Forestry Paper No. 140, Food and Agricultural Organization of the United Nations. Rome.

FAO, 2008. Global Forest Product Projections, by R. Jonsson and A. Whiteman. The Food and Agricultural Organization of the United Nations (FAO). Rome, Italy.

FAO, 2009. State of the World's Forests 2009. The Food and Agricultural Organisation of the United Nations (FAO). Rome, Italy.

FAO, 2010. Global Forest Resources Assessment 2010: Main Report. FAO Forestry Paper, No. 163. The Food and Agricultural Organization of the United Nations (FAO). Rome.

FAO, 2012a. FRA 2015 Terms and Definitions. Forest Resources Assessment Working Paper 180. The Food and Agricultural Organization of the United Nations (FAO). Rome.

FAO, 2012b. Global Ecological Zones for FAO Forest Reporting: 2010 Update. Forest Resources Assessment Working Paper 179. Forest Resources Assessment Working Paper 179. The Food and Agricultural Organization of the United Nations (FAO). Rome.

FAO, 2012c. Guide for Country Reporting for FRA 2015. FRA 2015 Working Paper, the Food and Agricultural Organization of the United Nations (FAO). Rome.

FAO, 2015. Global Forest Resources Assessment 2015. FAO Forestry Paper No. 1. The Food and Agricultural Organization of the United Nations (FAO). Rome.

FAO, JRC, 2012. Global Forest Land-Use Change 1990–2005. FAO Forestry Paper No. 169. The Food and Agricultural Organization of the United Nations (FAO) with the E.U. Joint Research Centre (JRC). Rome.

FAO, JRC, 2014. Global Forest Land-Use Change 1990–2010. An update to FAO Forestry Paper No. 169. The Food and Agricultural Organization of the United Nations (FAO) with the E.U. Joint Research Centre (JRC). Rome.

Federici, S., Tubiello, F.N., Salvatore, M., Jacobs, H., 2015. New estimates of CO2 forest emissions and removals: 1990–2015. For. Ecol. Manage. 352, 89–98.

Foster, A., Rosenzweig, M.R., 2003. Economic growth and the rise of forests. Q. J. Econ. 118, 601–637.

Friedl, M.A., Sulla-Menashe, D., Tan, B., Schneider, A., Ramankutty, N., Sibley, A., Huang, X., 2010. MODIS collection 5 global land cover: algorithm refinements and characterization of new datasets. Remote Sens. Environ. 114, 168–182.

Gaveau, D.L.A., Kshatriya, M., Sheil, D., Sloan, S., Wijaya, A., Wich, S., Ancrenaz, M., Hansen, M., Broich, M., Molidena, E., Guariguata, M., Pacheco, P., Potapov, P., Turubanova, S., Meijaard, E., 2013. Reconciling forest conservation and logging in Indonesian Borneo. PLoS One 8, e69887.

Gaveau, D.L.A., Sloan, S., Molidena, M., Husnayanem, Wijaya, A., Ancrenaz, M., Nasi, R., Wielaard, N., Meijaard, E., 2014. Four decades of forest loss and degradation in Borneo. PLoS One 9, e101654.

Gibson, L., Lee, T.M., Koh, L.P., Brook, B.W., Gardner, T.A., Barlow, J., Peres, C.A., Bradshaw, C.J.A., Laurance, W.F., Lovejoy, T.E., Sodhi, N.S., 2011. Primary forests are irreplaceable for sustaining tropical biodiversity. Nature 478, 378–381.

Giller, K.E., Leeuwis, C., Andersson, J.A., Andriesse, W., Brouwer, A., Frost, P., Hebinck, P., Heitkönig, I., Van Ittersum, M.K., Koning, N., 2008. Competing claims on natural resources: what role for science? Ecol. Soc., 13 (Article 34)

Global Landscapes Forum, 2014. Global Landscapes Forum. Convened in parallel with the UNFCCC COP20 meeting. 2014 Global Landsdcapes Forum: Landscapes for a New Climate and Development Agenda, Lima, Peru, 6–7 December 2014.

Gong, P., Wang, J., Yu, L., Zhao, Y., Zhao, Y., Liang, L., Niu, Z., Huang, X., Fu, H., Liu, S., Li, C., Li, X., Fu, W., Liu, C., Xu, Y., Wang, X., Cheng, Q., Hu, L., Yao, W., Zhang, H., Zhu, P., Zhao, Z., Zhang, H., Zheng, Y., Ji, L., Zhang, Y., Chen, H., Yan, A., Guo, J., Yu, L., Wang, L., Liu, X., Shi, T., Zhu, M., Chen, Y., Yang, G., Tang, P., Xu, B., Giri, C., Clinton, N., Zhu, Z., Chen, J., Chen, J., 2013. Finer resolution observation and monitoring of global land cover: first mapping results with Landsat TM and ETM+ data. Int. J. Remote Sens. 34, 2607–2654.

Grainger, A., 1996. An evaluation of the FAO tropical resource assessment 1990. Geography J. 162, 73–79.

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

Grainger, A., 2010. Uncertainty in the construction of global knowledge of tropical forests. Prog. Phys. Geogr. 34, 811–844.

Greenpeace SE Asia, 2014. High Carbon Stock Approach – WWF, GP, RAN, FPP Joint Briefer. What is the HCS Approach? <http://www.greenpeace.org/seasia/Press-Centre/Press-Releases/HCS-Briefer/> (accessed April 2015).

Griscom, B., 2014. A Fully Loaded, Double-Barreled Forest Climate Solution, Accessed March 2015, http://blog.nature.org/science/2014/09/24/forest-nadine-unger-plant-trees-climate/.

Gutiérrez-Vélez, V.H., Ruth, D., Miguel, P.-V., María, U., Christine, P., Walter, B., Katia, F., Yili, L., 2011. High-yield oil palm expansion spares land at the expense of forests in the Peruvian Amazon. Environ. Res. Lett. 6, 044029.

Hansen, M., Stehman, S.V., Potapov, P.V., 2010. Quantification of global forest cover loss. Proc. Natl. Acad. Sci. USA 107, 8650–8655.

Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., Thau, D., Stehman, S.V., Goetz, S.J., Loveland, T.R., Kommareddy, A., Egorov, A., Chini, L., Justice, C.O., Townshend, J.R.G., 2013. High-resolution global maps of 21st-century forest cover change. Science 342, 850–853.

Hecht, S.B., 2005. Soybeans, development and conservation on the Amazon Frontier. Dev. Change 36, 375–404.

Hecht, S.B., Saatchi, S.S., 2007. Globalization and forest resurgence: changes in forest cover in El Salvador. Bioscience 57, 663–673.

Hecht, S.B., Kandel, S., Gomes, I., Cuellar, N., Rosea, H., 2006. Globalization, forest resurgence and environmental politics in El Salvador. World Dev. 34, 308–323.

Indrarto, G.B., Murharjanti, P., Khatarina, J., Pulungan, I., Ivalerina, F., Rahman, J., Prana, M.N., Resosudarmo, I.A.P., Muharrom, E., 2012. The context of REDD+ in Indonesia. Working Paper 92. Center for International Forestry Research (CIFOR). Bogor, Indonesia.

ITS Global, 2011. The Economic Contribution of Indonesia's Forest-Based Industries (Draft Report). ITS Global, Melbourne, p. 49. <http://www.itsglobal.net/sites/default/files/itsglobal/ITS_Indoforest_Economic_Report.pdf>.

Jepson, P., Jarvie, J.K., MacKinnon, K., Monk, K.A., 2001. The end for Indonesia's lowland forests? Science 292, 859–861.

Joppa, L.N., Pfaff, A., 2009. High and far: biases in the location of protected areas. PLoS One 4, e8273.

Jürgensen, C., Kollert, W., Lebedys, A., 2014. Assessment of Industrial Roundwood Production from Planted Forests. . Planted Forests and Trees Working Paper Series, No. FP/48/E. The Food and Agricultural Organization of the United Nations (FAO). Rome.

Keenan, R.J., Reams, G.A., Achard, F., de Freitas, J., Grainger, A., Lindquist, E., 2015. Dynamics of global forest area: results from the FAO Global Forest Resources Assessment 2015. For. Ecol. Manage. 352, 9–20.

Kim, D.-H., Sexton, J.O., Townshend, J.R., 2015. Accelerated deforestation in the humid tropics from the 1990s to the 2000s. Geophys. Res. Lett., 2014GL062777

Köhl, M., Lasco, R., Cifuentes, M., Jonsson, O., Korhonen, K., Mundhenk, P., de Jesus Navar, J., Stinson, G., 2015. Changes in forest production, biomass and carbon: results from the 2015 UN Global Forest Resources Assessment. For. Ecol. Manage. 352, 21–34.

Kremen, C., Niles, J.O., Dalton, M.G., Daily, G.C., Ehrlich, P.R., Fay, J.P., Grewal, D., Guillery, R.P., 2000. Economic incentives for rain forest conservation across scales. Science 288, 1828–1832.

Lambin, E.F., Meyfroidt, P., 2011. Global land use change, economic globalization, and the looming land scarcity. Proc. Natl. Acad. Sci. USA 108, 3465–3472.

Laurance, W.F., Clements, G.R., Sloan, S., O/'Connell, C.S., Mueller, N.D., Goosem, M., Venter, O., Edwards, D.P., Phalan, B., Balmford, A., Van Der Ree, R., Arrea, I.B., 2014a. A global strategy for road building. Nature 513, 229–232.

Laurance, W.F., Sayer, J., Cassman, K.G., 2014b. Agricultural expansion and its impacts on tropical nature. Trends Ecol. Evol. 29, 107–116.

Lister, J., Dauvergne, P., 2014. Voluntary zero net deforestation. In: Nikolakis, W., Innes, J. (Eds.), Forests and Globalization: Challenges and Opportunities for Sustainable Development. Routledge, New York (Ch. 2).

Liu, J., Li, S., Ouyang, Z., Tam, C., Chen, X., 2008. Ecological and socioeconomic effects of China's policies for ecosystem services. Proc. Natl. Acad. Sci. 105, 9477–9482.

Lugo, A.E., 2009. The emerging era of novel tropical forests. Biotropica 41, 589–591.

Lugo, A.E., Helmer, E., 2004. Emerging forests on abandoned land: Puerto Rico's new forests. For. Ecol. Manage. 190, 145–161.

Lund, H.G., 2006. Definitions of forest, deforestation, afforestation, and reforestation. For. Inf. Services.

MacDicken, K.G., Sola, P., Hall, J., Sabogal, C., Tadoum, M., de Wasseige, C., 2015. Global progress toward sustainable forest management. For. Ecol. Manage. 352, 47–56.

MacDicken, K.G., 2015. Global Forest Resources Assessment 2015: what, why and how? For. Ecol. Manage. 352, 3–8.

Masters, W.A., Djurfeldt, A.A., De Haan, C., Hazell, P., Jayne, T., Jirström, M., Reardon, T., 2013. Urbanization and farm size in Asia and Africa: implications for food security and agricultural research. Global Food Security 2, 156–165.

Mather, A.S., Needle, C.L., Fairbairn, J., 1999. Environmental Kuznets curves and forest trends. Geography 84, 55–65.

Meijaard, E., Sheil, D., Nasi, R., Augeri, D., Rosenbaum, B., Iskandar, D., Setyawati, T., Lammertink, A., Rachmatika, I., Wong, A., Soehartono, T., Stanley, S., O'Brien, T., 2005. Life after Logging: Reconciling Wildlife Conservation and Production Forestry in Indonesian Borneo. CIFOR, Bogor, Indonesia.

Meyfoidt, P., Lambin, E., 2008. Forest transitions in Vietnam and its environmental impacts. Glob. Change Biol. 14, 1–18.
Meyfroidt, P., Lambin, E.F., 2011. Global forest transition: prospects for an end to deforestation. Ann. Rev. Environ. Resour. 36, 9.1–9.29.
Miles, L., Newton, A.C., DeFries, R.S., Ravilious, C., May, I., Blyth, S., Kapos, V., Gordon, J.E., 2006. A global overview of the conservation status of tropical dry forests. J. Biogeogr. 33, 491–505.
Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-Being: Biodiversity Synthesis. United Nations Environmental Program and the World Resources Institute, Washington, D.C.
Ministry of Forestry, 2012. Statistik Kehutanan Indonesia 2011 (Forestry Statistics of Indonesia 2011). Kementerian Kehutanan Direktorat Jenderal Planologi Kehutanan (Ministry of Forestry Directorate General for Forestry Planning). Jakarta.
Miura, S., Amacher, M., Hofer, T., San-Miguel-Ayanz, J., Ernawati, Thackway, R., 2015. Protective functions and ecosystem services of global forests in the past quarter-century. For. Ecol. Manage. 352, 35–46.
Morales-Hidalgo, D., Oswalt, S.N., Somanathon, E., 2015. Conserving forests for biodiversity: status and trends from the Global Forest Resources Assessment 2015. For. Ecol. Manage. 352, 68–77.
Müller, D., Sun, Z., Vongvisouk, T., Pflugmacher, D., Xu, J., Mertz, O., 2014. Regime shifts limit the predictability of land-system change. Global Environ. Change 28, 75–83.
Nabuurs, G.-J., Pussinen, A., Van Brusselen, J., Schelhaas, M., 2007. Future harvesting pressure on European forests. Eur. J. Forest Res. 126, 391–400.
Nabuurs, G.-J., Schelhaas, M.-J., Orazio, C., Hengeveld, G., Tome, M., Farrell, E.P., 2014. European perspective on the development of planted forests, including projections to 2065. NZ J. Forest. Sci. 44, S8.
Nilsson, S., Schopfhauser, W., 1995. The carbon-sequestration potential of a global afforestation program. Climatic Change 30, 267–293.
Pagiola, S., 2008. Payments for environmental services in Costa Rica. Ecol. Econ. 65, 712–724.
Pan, Y., Birdsey, R.A., Fang, J., Houghton, R., Kauppi, P.E., Kurz, W.A., Phillips, O.L., Shvidenko, A., Lewis, S.L., Canadell, J.G., 2011. A large and persistent carbon sink in the world's forests. Science 333, 988–993.
Pandit, M., Sodhi, N.S., Koh, L.P., Bhaskar, A., Brook, B.W., 2007. Unreported yet massive deforestation driving loss of endemic biodiversity in Indian Himalaya. Biodivers. Conserv. 16, 153–163.
Payn, T., Kollert, W., Liu, S., Rodriguez, L., Neves Silva, L., Wingfield, M., Michel-Carnus, J., Orazio, C., Freer-Smith, P., 2015. Changes in planted forests and future global implications. For. Ecol. Manage. 352, 57–67.
Piketty, T., Goldhammer, A., 2014. Capital in the Twenty-First Century. Belknap Press.
Planet Experts, 2014. Climate Scientist Respond to Dr. Nadine Unger. <http://www.planetexperts.com/climate-scientists-respond-dr-nadine-unger/> (accessed March 2015).
Poore, D., Sayer, J., 1991. The Management of Tropical Moist Forest Lands: Ecological Guidelines. IUCN.
Portillo-Quintero, C.A., Sánchez-Azofeifa, G.A., 2010. Extent and conservation of tropical dry forests in the Americas. Biol. Conserv. 143, 144–155.
Potapov, P., Yaroshenko, A., Turubanova, S., Dubinin, M., Laestadius, L., Thies, C., Aksenov, D., Egorov, A., Yesipova, Y., Glushkov, I., Karpachevskiy, M., Kostikova, A., Manisha, A., Tsybikova, E., Zhuravleva, I., 2008. Mapping the world's intact forest landscapes by remote sensing. Ecol. Soc. 13, 51.
Putz, F.E., Pinard, M.A., 1993. Reduced-impact logging as a carbon-offset method. Conserv. Biol. 7, 755–757.
Reich, P.B., 2011. Taking stock of forest carbon. Nat. Climate Change 1, 346–347.
Ribeiro, M.C., Metzger, J.P., Martensen, A.C., Ponzoni, F.J., Hirota, M.M., 2009. The Brazilian Atlantic Forest: how much is left, and how is the remaining forest distributed? Implications for conservation. Biol. Conserv. 142, 1141–1153.
Rodrigues, A.S., Andelman, S.J., Bakarr, M.I., Boltani, L., et al., 2004. Effectiveness of the global protected area network in representing species diversity. Nature 428, 640–643.
Romijn, E., Lantican, C.B., Herold, M., Lindquist, E., Ochieng, R., Wijaya, A., Murdiyarso, D., Verchot, L., 2015. Assessing changes in national forest monitoring capacities of 99 tropical non-Annex I countries. For. Ecol. Manage. 352, 109–123.
Rosset, P.M., 1998. Alternative agriculture works: the case of Cuba. Mon. Rev. (July–August), 21–28
Rosset, P.M., Benjamin, M., 1994. The Greening of the Revolution: Cuba's Experiment with Organic Agriculture. Ocean Press, Hoboken, NJ.
Rudel, T.K., 1998. Is there a forest transition? Deforestation, reforestation and development. Rural Sociology 63, 533–552.
Rudel, T.K., 2005. Tropical Forests: Regional Paths of Destruction and Regeneration in the Late Twentieth Century. Columbia University Press, New York.
Rudel, T.K., 2007. Changing agents of deforestation: from state-initiated to enterprise driven processes, 1970–1990. Land Use Policy 24, 35–41.
Rudel, T.K., 2013. The national determinants of deforestation in sub-Saharan Africa. Philos. Trans. R. Soc. Lond., B, 368.
Rudel, T.K., Perz-Lugo, M., Zichal, H., 2000. When fields revert to forests: development and spontaneous reforestation in post-war Puerto Rico. Prof. Geographer 52, 386–397.
Satgas REDD+, 2012. REDD+ National Strategy. Indonesian REDD+ Task Force (Satgas REDD+). Jakarta.
Sayer, J., 2009. Reconciling conservation and development: are landscapes the answer? Biotropica 41, 649–652.
Sayer, J., Cassman, K.G., 2013. Agricultural innovation to protect the environment. Proc. Natl. Acad. Sci. 110, 8345–8348.
Sayer, J., Elliot, C., 2005. The role of commercial plantations in forest landscape restoration. In: Mansourian, S., Vallauri, D., Dudley, N. (Eds.), Forest Restoration in Landscapes: Beyond Planting Trees. Springer, New York, pp. 379–383, 54.
Sayer, J., Chokkalingam, U., Poulsen, J., 2004. The restoration of forest biodiversity and ecological values. For. Ecol. Manage. 201, 3–11.
Sayer, J., Bull, G., Elliott, C., 2008. Mediating forest transitions:'Grand design'or'Muddling through'. Conserv. Soc. 6, 320.
Sayer, J., Sunderland, T., Ghazoul, J., Pfund, J.-L., Sheil, D., Meijaard, E., Venter, M., Boedhihartono, A.K., Day, M., Garcia, C., 2013. Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. Proc. Natl. Acad. Sci. USA 110, 8349–8356.
Schmitt, C.B., Burgess, N.D., Coad, L., Belokurov, A., Besançon, C., Boisrobert, L., Campbell, A., Fish, L., Gliddon, D., Humphries, K., Kapos, V., Loucks, C., Lysenko, I., Miles, L., Mills, C., Minnemeyer, S., Pistorius, T., Ravilious, C., Steininger, M., Winkel, G., 2009. Global analysis of the protection status of the world's forests. Biol. Conserv. 142, 2122–2130.
Schnell, S., Altrell, D., Ståhl, G., Kleinn, C., 2015. The contribution of trees outside forests to national tree biomass and carbon stocks—a comparative study across three continents. Environ. Monit. Assess. 187, 1–18.
Sexton, J.O., Song, X.-P., Feng, M., Noojipady, P., Anand, A., Huang, C., Kim, D.-H., Collins, K.M., Channan, S., DiMiceli, C., Townshend, J.R., 2013. Global, 30-m resolution continuous fields of tree cover: landsat-based rescaling of MODIS vegetation continuous fields with lidar-based estimates of error. Int. J. Digital Earth 6, 427–448.
Sloan, S., 2014. Indonesia's moratorium on new forest licenses: an update. Land Use Policy 38, 37–40.
Sloan, S., 2015. Development-driven recovery of tropical forests: the tropical forest transition with implications for REDD+. Ecol. Econ. 116, 1–11.
Sloan, S., Edwards, D.P., Laurance, W.F., 2012. Does Indonesia's REDD+ moratorium on new concessions spare imminently-threatened forests? Conserv. Lett. 5, 222–231.
Sloan, S., Jenkins, C.N., Joppa, L.N., Gaveau, D.L.A., Laurance, W.F., 2014. Remaining natural vegetation in the global biodiversity hotspots. Biol. Conserv. 117, 12–24.
South, D.B., 1999. How can we feign sustainability with an increasing population? New Forest. 17, 193–212.
Stewart, R.E., Desai, A., Walters, L.C., 2011. Wicked Environmental Problems: Managing Uncertainty and Conflict. Island Press.
Tateishi, R., Uriyangqai, B., Al-Bilbisi, H., Ghar, M.A., Tsend-Ayush, J., Kobayashi, T., Kasimu, A., Hoan, N.T., Shalaby, A., Alsaaideh, B., Enkhzaya, T., Gegentana, Sato, H.P., 2010. Production of global land cover data – GLCNMO. Int. J. Digital Earth 4, 22–49.
Unger, N., 2014. To Save the Planet, Don't Plant Trees, The New York Times. Op Ed, September 19. <http://www.nytimes.com/2014/09/20/opinion/to-save-the-planet-dont-plant-trees.html>.
Unit Keria Presiden4, 2012. ONE MAP Indonesia. Paper presented at Presentation by Nirarta Samadhi, Chair of Satgas REDD+ Moratorium Working Group, at Mid-Way Moratorium Workshop, 2012, Borodubur Hotel, Jakarta, 4–5 September, 2012.
United Nations, 2013. Millennium Development Goals Report 2013. The United Nations. New York.
Van Lierop, P., Lindquist, E., 2015. Global forest area disturbance from fire, insect pests, diseases and severe weather events. For. Ecol. Manage. 352, 78–88.
Venter, O., Fuller, R.A., Segan, D.B., Carwardine, J., Brooks, T., Butchart, S.H.M., Di Marco, M., Iwamura, T., Joseph, L., O'Grady, D., Possingham, H.P., Rondinini, C., Smith, R.J., Venter, M., Watson, J.E.M., 2014. Targeting global protected area expansion for imperiled biodiversity. PLoS Biol. 12, e1001891.
Watson, J.E., Dudley, N., Segan, D.B., Hockings, M., 2014. The performance and potential of protected areas. Nature 515, 67–73.
Weng, L., Boedhihartono, A.K., Dirks, P.H., Dixon, J., Lubis, M.I., Sayer, J.A., 2013. Mineral industries, growth corridors and agricultural development in Africa. Global Food Security 2, 195–202.
Whiteman, A., Wickramasinghe, A., Pina, L., 2015. Global trends in forest ownership, public income and expenditure on forestry and forest employment. For. Ecol. Manage. 352, 99–108.
Wilson, D., Kelston, A., Ahmen, S., 2010. Is this the 'BRICs decade'? BRICs Mon. 10, 1–4.
World Bank, 2013. Atlas of Global Development: A Visual Guide to the World's Greatest Challenges, fourth ed. Collins, Glasgow.
Wright, J.S., Muller-Landau, H.C., 2006. The future of tropical forest species. Biotropica 38, 1–15.
WWF, IIASA, 2012. Living Forests Report. WWF and the IIASA. Gland, Switzerland.
Xu, J., Yang, Y., Fox, J., Yang, X., 2007. Forest transition, its causes and environmental consequences: an empirical evidence from Yunnan of Southwest China. Tropical Ecol. 48, 1–14.
Zhang, P., Shao, G., Zhao, G., Le Master, D.C., Parker Jr., G.R., Dunning, J.B., Li, Q., 2000. China's forest policy for the 21st century. Science 288, 2135–2136.
Zomer, R.J., Trabucco, A., Coe, R., Place, F., van Norordwijk, M., Xu, J.C., 2014. Trees on Farms: An Update and Reanalysis of Agroforestry's Global Extent and Socio-Ecological Characteristics. ICRAF Working Papers, No. 179. World Agroforestry Centre (ICRAF). Bogor, Indonesia.