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The forgotten D: challenges of addressing forest degradation in complex mosaic landscapes under REDD+

Ole Mertz[a]*, Daniel Müller[b], Thomas Sikor[c], Cornelia Hett[d], Andreas Heinimann[d], Jean-Christophe Castella[e], Guillaume Lestrelin[e], Casey M. Ryan[f], David S. Reay[f], Dietrich Schmidt-Vogt[g], Finn Danielsen[h], Ida Theilade[i], Meine van Noordwijk[j], Louis V. Verchot[k], Neil D. Burgess[l], Nicholas J. Berry[f], Thu Thuy Pham[k], Peter Messerli[d], Jianchu Xu[m], Rasmus Fensholt[a], Patrick Hostert[n], Dirk Pflugmacher[n], Thilde Bech Bruun[o], Andreas de Neergaard[o], Klaus Dons[i], Sonya Dewi[j], Ervan Rutishauser[k] and Zhanli Sun[b]

[a]Department of Geography and Geology, University of Copenhagen, Øster Voldgade 10, 1350 Copenhagen K, Denmark; [b]Department Structural Change, Leibniz Institute of Agricultural Development in Central and Eastern Europe, Theodor-Lieser-Str. 2, 06120 Halle (Saale), Germany; [c]School of International Development, University of East Anglia, Norwich NR4 7TJ, United Kingdom; [d]Centre for Development and Environment (CDE) NCCR, University of Bern, Hallerstrasse 10, CH-3012 Bern, Switzerland; [e]Institut de Recherche pour le Développement, UMR GRED – IRD UPV Montpellier 3, PO Box 5992, Vientiane, Lao PDR; [f]School of Geosciences, University of Edinburgh, Grant Institute, The King's Buildings, West Mains Road, Edinburgh EH9 3JW, United Kingdom; [g]Kunming Institute of Botany, Chinese Academy of Sciences, 132# Lanhei Road, Heilongtan, Kunming 650204, Yunnan, China; [h]NORDECO, Skindergade 23, 1159 Copenhagen K, Denmark; [i]Forest and Landscape of Denmark, University of Copenhagen, Rolighedsvej 23, 1958 Frederiksberg C, Denmark; [j]World Agroforestry Centre (ICRAF), Bogor, Indonesia; [k]Center for International Forestry Research, PO Box 0113 BOCBD, 16000 Bogor, Indonesia; [l]Centre for Macroecology, Evolution and Climate, Department of Biology, University of Copenhagen, Denmark & WWF-US Conservation Science Program, Washington, DC, USA; [m]World Agroforestry Centre, Kunming 650201, China; [n]Geography Department, Humboldt-Universität zu Berlin, Unter den Linden 6, 10099 Berlin, Germany; [o]Department of Agriculture and Ecology, University of Copenhagen, Thorvaldsensvej 40, 1871 Frederiksberg, Denmark

International climate negotiations have stressed the importance of considering emissions from forest degradation under the planned REDD+ (Reducing Emissions from Deforestation and forest Degradation + enhancing forest carbon stocks) mechanism. However, most research, pilot-REDD+ projects and carbon certification agencies have focused on deforestation and there appears to be a gap in knowledge on complex mosaic landscapes containing degraded forests, smallholder agriculture, agroforestry and plantations. In this paper we therefore review current research on how avoided forest degradation may affect emissions of greenhouse gases (GHG) and expected co-benefits in terms of biodiversity and livelihoods. There are still high uncertainties in measuring and monitoring emissions of carbon and other GHG from mosaic landscapes with forest degradation since most research has focused on binary analyses of forest vs. deforested land. Studies on the impacts of forest degradation on biodiversity contain mixed results and there is little empirical evidence on the influence of REDD+ on local livelihoods and tenure security, partly due to the lack of actual payment schemes. Governance structures are also more complex in landscapes with degraded forests as there are often multiple owners and types of rights to land and trees. Recent technological advances in remote sensing have improved estimation of carbon stock changes but establishment of historic reference levels is still challenged by the availability of sensor systems and ground measurements during the reference period. The inclusion of forest degradation in REDD+ calls for a range of new research efforts to enhance our knowledge of how to assess the impacts of avoided forest degradation. A first step will be to ensure that complex mosaic landscapes can be recognised under REDD+ on their own merits.

Keywords: REDD+; forest degradation; deforestation; mosaic landscapes; forest carbon; greenhouse gases; livelihoods; biodiversity; governance; monitoring; remote sensing

Introduction

An international mechanism for Reducing Emissions from Deforestation and forest Degradation and enhancing forest carbon stocks (REDD+) has been negotiated at successive United Nations Framework Convention on Climate Change (UNFCCC) Conferences of Parties (COP) since 2005. The basic idea of REDD+ is simple: a developing country can negotiate financial compensation

*Corresponding author. Email: om@geo.ku.dk

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for reducing its greenhouse gas (GHG) emissions from deforestation and degradation of forests and also by increasing carbon (c) stocks within forests. In addition to the reduced emissions, it is envisaged that funds can be used to enhance livelihood conditions of forest-dependent communities and to improve biodiversity conservation.

Nonetheless, there are a number of obstacles to achieving international agreement on REDD+, including that potentially positive impacts in terms of reduced GHG emissions may be outweighed by negative impacts in other sectors (Ghazoul et al., 2010; Phelps et al., 2010; Putz & Redford, 2010). Particular concerns have been raised over defining reference levels and monitoring performance, the challenges of governance shortfalls, inadequate knowledge of carbon dynamics, the displacement of emissions to areas not included in a REDD+ project (leakage), complications in calculating additionality in terms of emissions that would have occurred in the absence of a project, and the permanence or long-term stability of carbon stocks (e.g. Campbell, 2009; Meyfroidt & Lambin, 2009). Moreover, there is consistent debate on whether the expected co-benefits of REDD+ such as biodiversity conservation (Gardner et al., 2012; Grainger et al., 2009; Putz & Redford, 2009) and improved local livelihoods will in fact be possible to realise (Blom et al., 2010; Mertz, 2009; Peskett et al., 2008).

While many of these policy issues can probably be dealt with at various levels in well-designed REDD+ programmes, the technical elements of REDD+ still need considerable innovative research inputs as fundamental issues remain unsolved. One important remaining obstacle is how to effectively integrate the second D in REDD+ – avoided forest degradation. Forest degradation was an addition to 'RED' at COP13 in Bali in 2007 as it was recognised that avoided forest degradation may be equally or more important than deforestation in terms of carbon losses, especially in developing countries where the population is heavily dependent on wood biomass for fuel and building materials (Asner et al., 2005; Blom et al., 2010; Wertz-Kanounnikoff & Angelsen, 2009). However, there is little consensus on how to address forest degradation in REDD+, partly because of the perception that carbon stocks and dynamics of degraded forests are less well known than those associated with undisturbed forests and deforestation, and so far forest degradation has been inappropriately grouped with deforestation in REDD+ (Herold & Skutsch, 2011).

In this paper we focus on forest degradation – or rather forests that may be considered degraded compared to old-growth forests. We specifically look at degraded forests that are part of complex mosaic landscapes dominated by shifting cultivation, agroforestry systems, plantations and different types of forested areas, which are subjected to different types of disturbances such as cultivation, fuelwood and forest product collection (Fox et al., 2011). These landscapes are dominated not by old-growth tropical forests but by a fine mesh of secondary forest patches of different ages, are characterised by high spatiotemporal dynamics (de Jong et al., 2001; Sirén & Brondizio, 2009), and remain inadequately classified (Hett, Castella et al., 2011; Padoch et al., 2007). We focus specifically on areas that are or used to be dominated by shifting cultivation rather than areas where forest degradation mainly takes place in standing forests that are rarely if ever used for agriculture (Ahrends et al., 2010).

We first review definitions of forest and forest degradation and assess the knowledge of the extent of complex mosaic landscapes. We then provide a status of current research in dealing with forest degradation under proposed REDD+ schemes and discuss the possibilities for degraded forests in complex mosaic landscapes to potentially qualify for REDD+. We focus specifically on the gaps in knowledge on measuring and monitoring

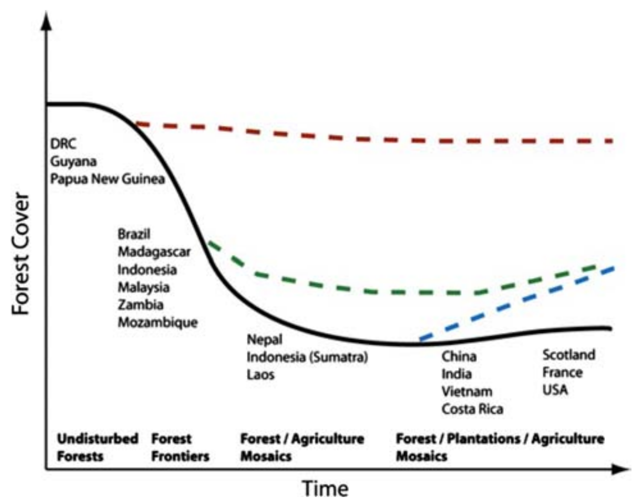

Figure 1. The forest transition and alternative paths offered by REDD+. Upper dashed line illustrates the REDD+ pathway for countries yet to significantly deforest or degrade forests, middle dashed line indicates reduced deforestation/degradation in countries currently deforesting and the lower dashed line the enhancement of forests in countries where forest cover – even if it is not natural – is stable. All pathways would be incentivised under the proposed UNFCCC REDD+ policy. Potential emission reductions from preventing logging or forest clearance, and sequestration from establishment of plantations, are well understood. Uncertainty remains, however, in the emissions associated with the management of mosaic landscapes. There is therefore a need to focus research efforts on understanding the emissions associated with the transition from forest to forest/agriculture mosaics, how to manage that transition to reduce emissions and enhance co-benefits, and how to increase natural forest cover or improve forest conditions in mosaic landscapes while continuing to provide sufficient food for growing populations.

carbon stocks and GHG emissions from forest degradation and the associated co-benefits expected from REDD++ in these landscapes. Moreover, we highlight some of the key challenges in governance linked to REDD+. Our hypothesis is that unless significant scientific advances are made on how to address forest degradation and encourage regeneration in landscapes with a mosaic of natural forest and agriculture, REDD+ may not apply to countries or regions that have already lost a large proportion of their natural forest cover (Figure 1).

The review was carried out by a literature search on ISI Thomson Reuters Web of Science. A few other non-peer reviewed sources were used, but these were not subjected to systematic searches. Searching was done by combining the words 'REDD' and 'forest degradation' with the words from the section titles in this article. Not all articles found were used due to the vast nature of the subject matter. As the topic is quite recent in the scientific literature, the review was conducted not as a meta-analysis aimed at quantifying the occurrence of values or subject matters in the articles reviewed, but as a state of the art approach based on a qualitative synthesis of available knowledge.

Defining forests and forest degradation

Defining forest degradation is not simple, nor is the definition of 'forest' itself. Defined by the Food and Agriculture Organization of the UN (FAO) as a land cover with 10% tree crown cover, forest definitions are linked to the long institutional history of treating 'forest' as a separate class of land cover and land use type. However, the ecological properties of tree cover, terrestrial carbon stocks and canopy–climate interactions show a continuum that no single forest definition can do justice to. This also has consequences for the definition of forest degradation.

The UNFCCC defines forest degradation as 'direct human-induced long-term loss (persisting for X years or more) of at least Y per cent of forest carbon stocks (and forest values) since time (T) and not qualifying as deforestation' (Penman et al., 2003, p. 16). Therefore, even an area that is temporarily devoid of trees may still qualify as degraded forest if those trees are likely to grow back, e.g. fields in shifting cultivation landscapes that are subsequently fallowed. However, there are more than 50 definitions of 'forest degradation' (Herold & Skutsch, 2011) and under the FAO forest definition, which is widely used, shifting cultivation is considered as deforestation because of the conversion of forest to agricultural uses. This makes the FAO definition inadequate for dealing with mosaic landscapes with shifting cultivation composed of both forest and non-forest patches. In addition, the climate biased focus on 'percent forest carbon stocks' ignores other important forest functions, such as biodiversity maintenance and livelihood provisioning services. There is thus no real agreement on how to define forest degradation and its outcomes (Herold & Skutsch, 2011; Sasaki & Putz, 2009; Schmidt-Vogt, 1998), and inclusion of forest degradation poses particular challenges to the design of effective, efficient and equitable governance arrangements (Sasaki & Putz, 2009; Skutsch et al., 2011).

Although the initial policy debate over deforestation, seen as a dichotomy of forest versus non-forest categories, did evolve into the more nuanced debate over forest degradation and enhancement of forest carbon stock in REDD+, the latter concept hinges on the exclusion of non-forest land from the quantification, with ensuing uncertainty over the emission consequences of interactions between forest and non-forest land categories. Operationalising REDD+ requires a choice of a non-ambiguous forest definition. If a stringent forest definition is used (e.g. focused on old-growth forest) the co-benefits of forest protection may be clear, but the scope for emission reduction is limited (van Noordwijk & Minang, 2009). If the forest definition is relaxed, with a logical extreme that all land can be considered forest regardless of tree cover, all emission consequences of land cover change can be handled, but hardly any statement about co-benefits can be made, unless other properties are closely correlated with carbon stocks. This approach has been suggested in the 'reducing emissions from all land uses' (REALU) approach (van Noordwijk et al., 2009), but REDD+ has received preferential treatment in the climate change negotiations.

Benefits and extent of complex mosaic landscapes

As mentioned, much degraded forest is found in complex mosaic landscapes that provide benefits in terms of multi-functional ecosystem services, such as forest products, agro-biodiversity (Rerkasem et al., 2009), watershed protection and diverse livelihoods (Cramb et al., 2009). They are also often characterised by a combination of multiple governance systems – such as community management of old-growth forests and household management of planted stands – within small areas and with significant variation between localities. However, since they are typically considered 'degraded' forest by most national forest authorities and also are defined as such according to the UNFCCC, they would only be eligible for REDD+ credits if further conversion to such landscape mosaics is avoided (the second D) or if natural or assisted regeneration of forests in these areas is implemented (the plus). However, shifting cultivation landscapes themselves could be eligible for REDD+ credits, if the credits would prevent intensification and shorter fallow periods that would have otherwise occurred.

The extent of mosaic landscapes on a global scale is not known, mainly because the 'degraded forest component' of these landscapes is not systematically classified – it can appear under classes such as secondary forest, degraded land, wasteland, idle land, etc., none of which gives a precise picture of what the land cover really is. This is mainly due to the relatively small extent of individual patches of the mosaic that are likely to be merged into wrong classes while processing coarse resolution satellite imagery. In Southeast Asia, for example, several reviews point to a severe lack of data on these landscapes (Padoch et al., 2007; Schmidt-Vogt et al., 2009) and mosaic landscapes have only been quantified for Lao PDR, where they make up about 29% of the land cover in the country (Messerli et al., 2009).

Land use intensification within these mosaic landscapes and transformation to other land uses are occurring rapidly. These factors lead to loss of old-growth forest if fallow length of cultivation cycles is reduced overall, and if other land uses such as large-scale rubber, oil palm, bamboo and pulp wood plantations take over (Schmidt-Vogt et al., 2009; van Noordwijk et al., 2008; Ziegler et al., 2009). Some of these plantations can technically qualify as 'forest', but there are strong arguments against this as they generally do not provide the same ecosystem services as forests (Xu, 2011) or even shifting cultivation systems. Only very short fallow shifting cultivation may have lower C-storage than plantations at landscape level and in most cases they will still contain higher biodiversity and provide greater watershed protection services (Fox et al., 2011). Indeed, it could also be argued that reforestation with monocultural plantation crops is a form of degraded forest relative to the alternative pathway of natural regeneration of old-growth forests (Xu, 2011).

C-stocks and greenhouse gas (GHG) emissions

Deforestation and forest degradation in tropical countries are estimated to emit 6–17% of anthropogenic carbon emissions (Nabuurs et al., 2007; van der Werf et al., 2009), and this large range indicates the current uncertainty in monitoring changes in forest carbon stocks (Denman et al., 2007), especially across land use gradients with a diversity of different forest and agricultural landscapes. Successful implementation of REDD+ therefore requires an improvement in current techniques for assessing carbon emissions and their changes through time (Gibbs et al., 2007).

Current methodologies for carbon stock assessment, detailed by IPCC (IPCC, 2006; Penman et al., 2003) and GOFC-GOLD (2009), provide the potential to produce accurate and precise carbon stock and emission estimates, but require comprehensive and expensive fieldwork (Angelsen et al., 2009). There are two approaches in the IPCC methods – the stock difference approach and the input output approach. For the stock difference approach, estimates are based on land use and land management categories and changes in carbon densities over a period of time. Carbon densities and emission factors show considerable spatial variation within countries (Angelsen et al., 2009; Skutsch et al., 2007) which calls for localised carbon stock and GHG flux information. The input output approach requires estimating losses from each C pool (e.g. harvesting, fire, mortality, etc.) and inputs (growth, transfer from one pool to another, etc.). These estimates also require expansion factors and emissions factors, which can vary between countries or regions.

For quantifying C-stocks in woody biomass the standard methods rely on large-scale forest inventories, the use of allometric models for above- and belowground biomass, and detailed monitoring of changes of forest extent and structure through time. This methodology is reliant on accurate allometric equations, but for tropical ecosystems with their high tree diversity these equations have wide confidence intervals as individual and relatively rare trees contribute a disproportionately high share of biomass (Chave et al., 2004, 2005; Neeff et al., 2005). Locally parameterised allometric models that are based on intensive sampling are hence essential (van Breugelet et al., 2011), but even then estimates of large tree biomass are highly uncertain (Chave et al., 2005). Existing estimates of carbon stock changes in aboveground vegetation during forest degradation and after deforestation are thus highly variable and data on carbon storage in belowground biomass of tree based systems rely on global 'defaults' that only vary by broad climatic zones (Houghton, 2005; Ramankutty et al., 2007). The belowground biomass is often not adequately accounted for as detailed knowledge of root biomass in different ecosystems has only emerged recently for a very limited number of regions and forest ecosystems.

With regard to soil carbon stocks, studies have indicated that they are relatively robust at intermediate levels of disturbance and degradation such as under shifting cultivation (Aumtung et al., 2009; de Neergaard et al., 2008). Conversion from old-growth forest to perennial crops results in an average depletion of soil organic carbon (SOC) by 30% that can be partially compensated in fallow systems (Don et al., 2011). However, most studies are restricted to few land use transitions or a single carbon compartment and rarely account for factors such as climate, soil type or land use intensity (Bruun et al., 2009). Overall, the understanding of interactions between soil type, land use and bioclimatic conditions on soil carbon stocks is generic at best.

Similarly, the magnitude of non-CO_2 greenhouse gas fluxes (e.g. Dalal & Allen, 2008) and direct radiative forcing effects, such as altered albedo (Betts, 2011), remain poorly understood in most forest ecosystems, and especially so in degraded forest areas (Mabuchi, 2011). Forest soils dominate the global soil sink for methane

(CH_4) of approximately 30 Tg yr^{-1} (Dunfield, 2007) and, at the local scale, forest degradation is likely to influence net CH_4 flux through changes in soil temperature, moisture and nitrogen (N) cycling (Reay et al., 2005; Sousa Neto et al., 2011). Of potentially much greater importance in terms of net GHG emissions is the impact of forest degradation on reactive N (Nr) inputs and associated carbon dioxide and nitrous oxide (N_2O) fluxes (Reay et al., 2008). Tropical rainforest soils represent the largest source of N_2O emissions after agriculture (Werner et al., 2007) and atmospheric Nr deposition rates have risen markedly in many tropical and sub-tropical regions in recent decades, with some forest areas receiving in excess of 10 kg Nr ha^{-1} yr^{-1} (Dentener et al., 2006).

Recent studies of tropical land-cover change have highlighted canopy-induced changes in fog and dry deposition of Nr, with forest canopies retaining a large proportion of deposited Nr and so reducing inputs to soil (e.g. Ponette-Gonzalez et al., 2010). As such, forest degradation has the potential to radically alter soil Nr availability and net N_2O fluxes, any substantial increase in Nr inputs to degraded forest soils being likely to enhance N_2O emissions. To robustly quantify total GHG emissions from degrading forests it is therefore important that non-CO_2 GHG fluxes and their interactions with key drivers, such as soil temperature, moisture and Nr input, are considered.

Common to all the mentioned inventories (above- and belowground biomass, soil C and GHG emissions), is that they are spatially and temporally highly variable and relatively costly to measure if plot sizes and replicates have to be adequate to avoid high coefficients of variation (Laumonier et al., 2010) and wide confidence intervals. This is particularly the case in mosaic landscapes, where the diversity of landscape elements is even higher than in old-growth forests. This offers an immediate challenge for estimating the reference levels and an ongoing challenge for monitoring and verification. Hence, development of cost-effective, accurate measurement techniques and approaches will remain a priority.

Biodiversity

An expected co-benefit of REDD+ is to safeguard biodiversity in tropical forests (Gardner et al., 2012; Grainger et al., 2009; Venter et al., 2009). However, there are concerns that governments and market forces will focus REDD+ activities into areas of threatened forest with low opportunity costs of land use (Ebeling & Yasue, 2008; Fisher et al., 2011), which may not necessarily contain important biodiversity values. Nonetheless, moderate congruence between biomass carbon and species richness has been demonstrated (Strassburg et al., 2010), and it has been suggested that the presence of species with very high conservation values, such as large carnivores, may perhaps tip the balance back in favour of carbon project activities in forest areas that support such species (Dickman et al., 2011). Overall, tropical moist forest biodiversity hotspots retain only about 10% of their original forest (Myers et al., 2000), have high human population growth (Cincotta et al., 2000), are poorly protected (Schmitt et al., 2009) and are experiencing continuing loss of forests (Scharlemann et al., 2010). The costs of reducing deforestation and forest degradation will be higher in these areas than in more sparsely populated areas such as the Amazon and Congo Basin, where at present 85% of the forest remains.

With regard to the role of degraded forests and mosaic landscapes for safeguarding biodiversity, recent studies suggest that the old-growth forests are the most important for global biodiversity conservation (Gibson et al., 2011; Phalan et al., 2011), especially for a significant proportion of rare, endemic and threatened species that are not very tolerant to disturbance of their native habitats. On the other hand, there is also evidence that the impact on biodiversity of conversion to complex mosaic landscapes dominated by agroforestry and shifting cultivation varies considerably (Finegan & Nasi, 2004; Scales & Marsden, 2008), and that these landscapes may in some cases maintain high levels of biodiversity (Berry et al., 2010; Rerkasem et al., 2009; Xu et al., 2009). Nonetheless, mosaic landscapes may have different – and from a conservation point of view less valuable – species compositions and forest animals observed in fallowed land may depend on the existence of nearby native habitats. The role of degraded forest land in conserving biodiversity is therefore dependent on the degree of degradation, connectivity to native forests and the intensity of management.

Another likely development is that the 'plus' in REDD+ gains importance. Then degraded forests in mosaic landscapes that are or could be potentially important for biodiversity may be stocked with timber, fast growing paper pulp species or other tree crop plantations that could be eligible for REDD+ credits. This will create forests with one or few dominant species that may store more carbon than the degraded forest or shrub land they replace but that will contain much less biodiversity (Brockerhoff et al., 2008). This is what has occurred with the forest transition in China (Xu et al., 2009) and to some extent in Vietnam (Meyfroidt & Lambin, 2008).

Local livelihoods

Another expected co-benefit of REDD+ is that local livelihoods can be improved by conserving forested land used for collection of forest products and by ensuring that parts of the REDD+ credits benefit local communities either through direct payments or through various development efforts such as support to agricultural intensification on non-forested land (Angelsen, 2010; D. Brown et al., 2008). However, the extent to which REDD+ projects will bring benefits in the long run especially with regard to ensuring livelihoods in remote areas dominated by shifting cultivation is uncertain (Fox et al., 2011; Mertz, 2009). The current rapid land use transitions in Southeast Asia (Mertz et al., 2009; Schmidt-Vogt et al., 2009), for example, have both negative and positive outcomes for local people depending on the local and national situations (Cramb et al., 2009; Xu et al., 2009) and on the share of local livelihoods derived from the use of degraded forests and mosaic landscapes. Secondary or degraded forest areas provide important services in terms of nutrient accumulation needed for subsequent cultivation and they often harbour biodiversity that provides more useful products for local people than old-growth forest (Christensen, 2002; Ebeling & Yasue, 2008; Pfund et al., 2011). Thus, REDD+ driven conversion of degraded forests to old-growth forests or to monocultural plantations may have negative impacts on local livelihoods and food security.

Overall, there is little empirical evidence on how REDD+ payments may improve the livelihoods and food security of rural communities and households (Jindal et al., 2008). Comparative studies of pre-REDD+ carbon projects are not fully conclusive on livelihood benefits (Li et al., 2011; Nelson & de Jong, 2003), or point out important flaws in monitoring methods that prevent the delivery of definitive answers in the absence of convincing counterfactual socioeconomic outcomes (Caplow et al., 2011). Moreover, most studies deal with hypothetical REDD+ situations as they lack sufficient historical background (e.g. Bellassen & Gitz, 2008).

If livelihood improvements through REDD+ are to be achieved, land users must be compensated for forest conservation, avoided forest degradation and C-stock enhancement above their opportunity costs of land use. Knowledge and mapping of place-based opportunity costs with realistic time scales and discount rates are thus crucial to calculate the potential costs of compensating land and forest users. At the national and sub-national levels, a generic approach has been developed integrating the real extent of different land use types with the estimated carbon stocks, economic profits and co-benefits (e.g. water provision, biodiversity) associated with each land use type (Pagiola & Bosquet, 2009; Swallow et al., 2007; World Bank, 2011). On this basis, projections of opportunity costs, carbon emission reductions and benefit distribution are made under different scenarios of land use change (e.g. 'business as usual', agrarian reform, shifting returns per hectare). In turn, these projections can provide guidance for managing trade-offs (e.g. food production vs. carbon sequestration vs. biodiversity) and targeting REDD+ initiatives at the sub-national level (Börner & Wunder, 2008; Börner et al., 2010). But counterbalancing opportunity costs is only one part of the complex process needed to bring about changes in practices (Gregersen et al, 2010; Wunder et al., 2008). As argued by some scholars (e.g. Ghazoul et al., 2010), approaches to the costs and benefits of REDD+ should be expanded to include a wider range of potential ecological and socio-political impacts.

In addition, protection of community forest rights and promotion of community participation in REDD+ are other important aspects of local livelihood security after REDD+ implementation (IFCA, 2007; Sunderlin et al., 2009; van Noordwijk, Suyamto et al., 2008), which may only be efficient if tenure security for local communities is increased (Lasco et al., 2010; Leimona et al., 2009). This is of particular importance in complex mosaic landscapes with contested areas such as land under fallow in shifting cultivation systems that are often not recognised as being part of the agricultural cycle with great importance for local livelihoods (Padoch et al., 2007). Under centralised conservation schemes, such areas are prone to be set aside for regeneration or reforestation without compensation of local land users. If REDD+ programmes are to achieve fair and equitable outcomes, it is therefore necessary that local communities are recognised as the owners and managers of mosaic landscapes and as legitimate recipients of the potentially emerging benefits from carbon payments.

Governance and benefit distribution mechanisms

Inclusion of forest degradation also carries direct implications for the governance of REDD+. Whereas the design of REDD+ faces steep challenges in general already (Corbera & Schroeder, 2011), these become particularly challenging with the inclusion of forest degradation. Governance arrangements suitable to reducing deforestation *and* forest degradation will have to reflect the particular processes underlying forest degradation, as those make it impossible to rely on relatively simple governance approaches such as protected areas or regulation of large corporations. Simple implementation of existing forest policies compatible with principles of good forest governance (Kanowski et al., 2011) will most likely not suffice, as existing forest policies have largely failed to respond to forest degradation processes in an adequate manner (Mertz et al., 2009). Inclusion of forest degradation also poses special governance challenges in respect of the difficulties encountered in assessing forest managers' performance and the generation of co-benefits if forest degradation is reduced, as indicated by the challenges of carbon stock assessments and realisation of biodiversity and livelihood co-benefits discussed above.

One of the most critical governance issues raised by forest degradation is the question about the basis on which carbon finance should be allocated to forest managers. If REDD+ takes a payment approach, what criteria should guide the allocation of payments (cf. Angelsen et al., 2009)? Similarly, if REDD+ provides other kinds of non-monetary benefits to forest managers, on what basis should the available benefits be distributed among forest managers? Three types of approaches have been discussed in REDD+ policy debates (Skutsch et al., 2011): (a) rewards for those who increase forest cover and enhance forest conditions (i.e. output-based allocation); (b) rewards for those who apply desirable forest management practices (i.e. input-based allocation); and (c) compensation for those who lose access to forest products and lands (i.e. opportunity cost-based allocation). Inclusion of forest degradation has direct implications for the effectiveness, efficiency and equity of the three approaches, as their outcomes will differ in complex mosaic landscapes from more homogeneous landscapes, and locations with multiple kinds and high numbers of stakeholders from socially uniform ones. For example, output-based allocation would cause high transaction costs in complex mosaic landscapes due to the required fine-grained carbon stock assessments (as opposed to landscapes experiencing deforestation). Similarly, opportunity cost-based allocation would suffer from high transaction costs, as the opportunities available to various kinds of stakeholders and relevant to different sorts of land uses are highly variable (in contrast to land use change driven by a single activity). The general argument speaking in favour of input-based allocation (Fry, 2011) may thus have particular purchase for efforts seeking to reduce forest degradation.

Another critical governance issue raised by forest degradation is the need to facilitate forest management that combines the reduction of degradation with existing forest uses. In many places, the monetary value of direct forest uses (e.g. fuelwood) may exceed the level of financial gains from carbon funds (Fisher et al., 2011; Karky & Skutsch, 2010). In consequence, REDD+ governance will be challenged to incorporate participatory decision-making processes over forest management that involve local forest managers and forest officers on an equal footing, something that has proven relatively elusive in forestry this far, even where local people have received tenure rights to forests (Fisher et al., 2011; Sikor & Tran, 2007). Moreover, once existing uses are recognised, local forest governance faces difficult decisions in the presence of competing claims on forests (Sikor & Nguyen, 2007). REDD+ governance will need to incorporate local decision-making processes that recognise the claims made by various stakeholders on forests, resolve competing claims in a legitimate manner, and reconcile direct uses with the reduction of deforestation and forest degradation. Similarly, local governance arrangements will have to – explicitly or implicitly – settle the allocation of carbon rights among competing stakeholders, with possibly grave consequences for disadvantaged groups (Brown et al., 2008; Lovera et al., 2008).

A third, equally significant governance question is about suitable constellations of functions of control over forest management (Agrawal et al., 2011). Important control functions include the above-mentioned powers to decide about forest management, the powers to exclude outsiders, the powers to handle financial transactions, and the mandate to monitor compliance. The distribution of these functions across various institutions is particularly critical in complex landscapes including multiple kinds and possibly large numbers of stakeholders. Bundling all control functions in the hands of a single institution, such as the administrative office of a protected area or a community management board, will not facilitate effective REDD+ governance. Communities, for example, would get into a significant conflict of interest if they were to manage forests, receive carbon rewards, and measure their own performance without outside involvement. Similarly, experience with top-down bureaucratic government programmes shows that local forest officers tend to report successful implementation regardless of actual outcomes if they are the ones who not only assess compliance but also handle financial transactions (Angelsen et al., 2009; UN-REDD et al., 2010). The challenge, therefore, is to distribute control functions among several institutions in a system of checks and balances. Such governance arrangements will most likely have to involve communities to counterbalance the dominant influence of government officials and other outsiders (Chhatre & Agrawal, 2009; Peskett et al., 2011).

Reference levels and monitoring

To assess the effectiveness of REDD+ in reducing GHG emissions and promoting associated co-benefits, improved long-term monitoring and clear definitions of reference levels (RL) are needed. In this section we will focus on carbon monitoring, which is the most immediate concern for REDD+. Even though considerable emission reductions and removal enhancements are achievable from sustainable forest management and reduced forest degradation (Stern, 2008), a lot of conceptual work to date has focused on defining 'business as usual' RL for deforestation only, be it at the national or sub-national level (S. Brown et al., 2007; Busch et al., 2009; Huettner et al., 2009). However, these scenarios often do not distinguish between managed and undisturbed forests, and are therefore insufficient for characterising carbon losses from forest degradation. According to Huettner et al. (2009), the only historical RL approach proposed to the UNFCCC that includes degradation is the Joint Research Centre (JRC) approach described by Achard et al. (2005) and Mollicone et al. (2007). The JRC approach includes an incentives mechanism for reduced forest degradation that suggests a spatial delineation of intact and non-intact forest land. The distinction is based on proxy variables that indicate potential forest disturbance, e.g. the construction of road networks and settlements. Degradation can thereafter be reported as a conversion from intact to non-intact forest land using either locally established or default biomass values. The vast majority of monitoring (or 'measurement' as used by UNFCCC), reporting and verification (MRV) systems also focus on deforestation rather than forest degradation (Angelsen et al., 2009), which is far from trivial to be monitored (Asner et al., 2005; Foley et al., 2007; Souza et al., 2005).

Methods to set forest degradation RL will differ considerably from methods setting deforestation RL, because degradation predictions need to include carbon stock changes within forest areas (Angelsen et al., 2009; DeFries et al., 2007). The extent and detectability of degradation depends on the type of degrading activities such as timber extraction, fuelwood collection, or intensification of shifting cultivation that reduces the average age of secondary forest in a mosaic landscape (Hett et al., 2011).

The extent of sub-canopy degradation is difficult to estimate over large areas due to the trade-off between pixel resolution and scene extent of satellite images and the current scarcity of multi-level remote sensing (RS) methodologies integrating the use of different sensors over various spatial scales. Furthermore there is a lack of knowledge for linking RS data and ground-based inventories for the periods needed which in developing countries often coincides with the absence of ground-based and remote sensing data consistent over time (Baker et al., 2010). The availability of remotely sensed optical imagery is also limited by cloud cover and high aerosol loads from biomass burning activities, particularly in moist tropical regions of high relevance to REDD+. Time series analysis of freely available high temporal resolution RS data with a medium spatial resolution (e.g. MODIS) can support uncovering forest degradation processes by analysing subtle changes in spectral signatures when carefully taking into consideration information on cloud cover and aerosol concentrations.

High resolution satellite data such as Landsat provide better opportunities for detailed monitoring, but coverage is very uneven around the globe. There are significant data gaps in developing countries, but this imbalance is most likely going to change in the near future as costs of satellite platforms have decreased, and national and international earth observation programmes have made significant commitments. For example, the Sentinel-2 Mission will be launched in 2013 as part of the European Global Monitoring for Environment and Security programme and has the objective to provide full and systematic coverage of most of the land surface with cloud-free products every 15–30 days (Martimort et al., 2007). Moreover, the US Geological Survey has recently released the Landsat archive, providing free-of-charge access to near-annual, cloud-free observations, though mainly for North America and Europe. These data policy changes have led to significant shifts in the paradigm of change-detection research and are likely going to facilitate future developments in operational land imaging. While past change detection studies were often limited to bi- and multi-temporal analyses of stand-replacing disturbances, easy access to dense image time series has led to developments of new algorithms that enable automated detection not only of abrupt changes (e.g. clearcuts) but also of subtle and long-term forest cover changes (Kennedy et al., 2010).

Other types of sensors have also been tested and promising estimates of forest degradation have been obtained from SAR (Synthetic Aperture Radar) data available since 2007 (Fagan & DeFries, 2009; Mitchard et al., 2009; Ryan et al., 2012). Plans are also underway for a space borne radar specifically designed to measure biomass (Le Toan et al., 2011). Finally, direct remotely sensed canopy degradation monitoring can also be achieved by using commercial very high resolution (VHR) imagery (e.g. Quickbird) or airborne techniques as LiDAR (light detection and ranging) from which individual tree crowns can be identified. However, limited spatial coverage and the high costs associated with these sensors limit their use for regional applications. Such data should ideally be used in a nested approach for large-scale inventories also including imagery with larger spatial coverage to benefit from the complementary information derived from high spatial and temporal resolution (Asner, 2009). The costs for implementing VHR data in forest degradation monitoring are expected to drop significantly during the coming years as an increasing number of operational VHR sensors are in place and the data archives grow larger.

The emerging techniques of object-based image analysis (OBIA) for land cover mapping offer additional opportunities for mapping (Blashcke et al., 2008). Rather than interpreting land use/cover at the pixel level based on spectral signature of the individual pixel, other sets of data, local knowledge and derived data from satellite imageries beyond individual pixels are used in forming a set of objects for further interpretation processes. Identification of drivers and activities from the local knowledge and translation of those into spatial patterns and relationships can enhance the analysis of satellite imageries beyond spectral-only data. For instance, a smooth texture within an area suggests plantation rather than old-growth forest and small patches relate to smallholder rather than large-scale activities. Multiple sensors and multiple resolutions can be integrated within OBIA. Identification of logged-over areas, agroforests, forest plantation and estates, which is crucial for monitoring forest degradation and mosaic landscapes, is possible and has been incorporated in Indonesia land use/cover maps (Ekadinata et al., 2011). But OBIA is still limited by the spatial and spectral resolution and thus faces the same constraints as pixel-by-pixel approaches for detecting small-scale land cover and land use changes. Another promising methodology for delineating forest degradation is based on Landsat and a landscape mosaic approach developed for Lao PDR and captures the extent of shifting cultivation in landscapes with different degrees of land use intensity (Hett et al., 2011; Messerli et al., 2009).

While RS-based approaches are essential for national and regional monitoring, they rely on skills that are often not locally available in developing countries (Sheil, 2001). Experiences suggest that RS is frequently perceived as too technically demanding by resource managers (Danielsen et al., 2005), and too costly for monitoring at the local level (Houghton & Goetz, 2008), and as a consequence may have limited bearing on forest management decisions in practice. It is therefore valuable to combine the RS methods with community-based approaches, where local people or local government staff are directly involved in data collection and interpretation, and where monitoring is linked to the decisions of local people (Danielsen et al., 2010), using methods that are simple, cheap and require few resources (Danielsen et al., 2009; Holck, 2008). Studies suggest that local people can accurately count trees, measure their girth, identify the species and cost-effectively collect large volumes of such data using IPCC guidelines in both old-growth and degraded forests (Danielsen et al., 2011; Skutsch et al., 2009).

Locally based monitoring can also build local capacity and cooperation between local people and the authorities, and can thereby stimulate local action and facilitate rapid forest management interventions (Danielsen et al., 2007, 2009). The locally based approaches to biomass and land use change monitoring have the potential to link meaningfully to national assessments and monitoring schemes and, at the same time, to be accurate, cost-effective, and capable of building local capacity. They can also generate ownership to REDD+ efforts, promote accountability and incorporate evidence-based assessments in decision-making at the local level. However, without rigorous validation studies, carbon traders, professional forest managers and national government staff could remain sceptical about the results and usefulness of linking community-based and national monitoring approaches.

Conclusions and perspectives

As outlined above, there are considerable gaps in our knowledge of and ability to deal with forest degradation in REDD+. The focus on deforestation is partly caused by the relative simplicity in dealing with large areas of forests and non-forested landscape elements, whereas adequately addressing forest degradation is more challenging. Carbon stocks and dynamics of mosaic landscapes, commonly bundled as 'degraded forest' or other non-specific terms, are not well described. RL setting and monitoring methods are not yet adequately developed and the benefits of degraded forests in terms of supporting local livelihoods and even conserving biodiversity are also not clear. As the inclusion of forest degradation has important implications for how REDD+ is implemented, a range of new research efforts to enhance our knowledge of degraded forest environments and to integrate these landscapes meaningfully in a REDD+ scheme is called for.

Improved allometric equations that include root biomass will yield more reliable carbon stock estimates, which combined with soil carbon assessment along chronosequences of different land use types – including forests in different stages of regrowth – in mosaic landscapes with differing mineralogy will yield better knowledge on ecosystem carbon storage and dynamics associated with forest degradation. To be able to monitor these carbon stocks associated with different land use and land degradation categories, recently developed algorithms that use dense image time series and multi-sensor data need to be tested. There is also an immediate need for developing approaches for integration of local estimates of land use and carbon stock changes from participatory approaches with the remote sensing based monitoring.

Significant implications for the expected livelihood co-benefits and governance of REDD+ are also evident with the inclusion of forest degradation. REDD+ payments may have to be made on a different basis (inputs instead of outputs or opportunity costs), management for reducing forest degradation and enhancing carbon stocks has to be carefully weighed with existing forest uses by multiple stakeholders, and involved control functions must be distributed among involved institutions through a suitable system of checks and balances. Research needs include how payment schemes are established and how payments are distributed. Whether these payments induce high enough and sufficiently long incentives to conserve and protect forests must also be thoroughly elucidated by research at local, sub-national and national levels. Moreover, innovative methods for *ex-ante* livelihood impact assessment such as participatory simulations of REDD+ scenarios are needed to ensure better prediction of impacts rather than only relying on *ex-post* evaluations.

Since degraded forests are rapidly increasing in many regions, it is very important that the REDD+ negotiations take a strong position on how to deal with these areas. Given the potential importance of complex landscapes for livelihoods and biodiversity, REDD+ could also integrate a mechanism for recognition of broader landscapes, which is also promoted in the REALU approach outlined in the introduction. Thus, avoided forest degradation under REDD+ should not be translated into complete prohibition of using these landscapes for agriculture, but rather taken to mean sustainable land use management that prevents further degradation of forests.

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