Synthesis

An Operational Framework for Defining and Monitoring Forest Degradation

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ABSTRACT. Forest degradation is broadly defined as a reduction in the capacity of a forest to produce ecosystem services such as carbon storage and wood products as a result of anthropogenic and environmental changes. The main causes of degradation include unsustainable logging, agriculture, invasive species, fire, fuelwood gathering, and livestock grazing. Forest degradation is widespread and has become an important consideration in global policy processes that deal with biodiversity, climate change, and forest management. There is, however, no generally recognized way to identify a degraded forest because perceptions of forest degradation vary depending on the cause, the particular goods or services of interest, and the temporal and spatial scales considered. Here, we suggest that there are types of forest degradation that produce a continuum of decline in provision of ecosystem services, from those in primary forests through various forms of managed forests to deforestation. Forest degradation must be measured against a desired baseline condition, and the types of degradation can be represented using five criteria that relate to the drivers of degradation, loss of ecosystem services and sustainable management, including: productivity, biodiversity, unusual disturbances, protective functions, and carbon storage. These criteria are not meant to be equivalent and some might be considered more important than others, depending on the local forest management objectives. We propose a minimum subset of seven indicators for the five criteria that should be assessed to determine forest degradation under a sustainable ecosystem management regime. The indicators can be remotely sensed (although improving calibration requires ground work) and aggregated from stand to management unit or landscape levels and ultimately to sub-national and national scales.

Key Words: *biodiversity; carbon; criteria and indicators; forest degradation; forest management; remote sensing*

INTRODUCTION

Forest degradation is a widespread global concern and an important contemporary issue for several United Nations (UN) organizations and conventions. These groups include the UN Convention on Biological Diversity (CBD), which set a global target for restoration of at least 15% of degraded ecosystems by 2020 (Convention on Biological Diversity 2010); the UN Forum on Forests that has an objective to reduce degradation; the UN Convention to Combat Desertification (UNCCD) that considers degradation on drylands; and the UN Framework Convention on Climate Change (UNFCCC) that proposes to recover degraded forests as carbon sinks. Recent climate negotiations have initiated the concept of reducing emissions from deforestation and forest degradation (REDD) to mitigate climate change through forest management, including the restoration of degraded forests (UNFCCC 2010). Along with deforestation, forest degradation has major consequences for human societies and biodiversity, and significantly contributes to greenhouse gas emissions (Secretariat of the Convention on Biological Diversity 2002, Parry et al. 2007, van der Werf et al. 2009, Mery et al. 2010). Deforestation is an obvious ecosystem change, but forest degradation is more difficult to discern and quantify (Sasaki and Putz 2009). The International Tropical Timber Organization (ITTO 2002) suggests that up to 850 million ha of tropical forest could already be degraded.

The Millennium Ecosystem Assessment (e.g., Diaz et al. 2005) states that many forest provisioning and regulating services are being degraded, but does not discuss quantification or monitoring. The causes of degradation vary globally, including unsustainable logging, poor agricultural practices, invasive species, fuelwood gathering, and wildfire, with synergistic effects (van Wilgen et al. 2001, Asner et al. 2006, 2008, Chazdon 2008, Murdiyarso et al. 2008, Kissinger et al. 2012).

Although there are numerous definitions of forest degradation (Simula 2009), many are reductionist (e.g., Parry et al. 2007), and none is adequate to inform decision making. There is a growing need to be able to identify a degraded forest from stand scales to landscape and national scales to inform decision making about where to invest in forest recovery (e.g., Holl and Aide 2011). The lack of an agreed common definition (and monitoring framework) hinders international recovery efforts (Sasaki and Putz 2009). The recently completed *Global Forest Resources Assessment* (FAO 2010) could not report an area of degraded forest for lack of a definition. In general, definitions of forest degradation (ITTO 2002, Secretariat of the Convention on Biological Diversity 2002, Norris 2012) suggest changes in forest structure, dynamics, and functions resulting mostly from human-induced causes relative to a preferred condition. In these latter definitions, the spatial scale

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is at the stand or site level, and the temporal scale is usually long term. The Collaborative Partnership on Forests (CPF, www.cpfweb.org) tries to reconcile the many definitions and proposes that forest degradation is the "reduction of the capacity of a forest to provide goods and services". Although this definition is also not operational, it provides a basis for assessing degradation through quantifiable indicators of ecosystem goods and services.

Perceptions about forest degradation are varied and depend on the causes, the goods or services of interest, and the temporal scale considered. Forest degradation has been described using variables such as changes in canopy and understory tree density, plant or animal species richness, and/ or carbon stocks as measured against a baseline from apparently undisturbed conditions (Lambin 1999, Devi and Behera 2003, Harrison 2011). Some authors consider tropical forests to be degraded once the forest has been logged (e.g., Sierra 2001, Foley et al. 2007), whereas others consider this only when the forest has been heavily burned and logged (Souza et al. 2005). Furthermore, societal choices largely determine what is considered degraded. For example, a manager who replaces a primary forest with a tree plantation to supply a sustained yield of wood is unlikely to perceive that forest as degraded. The same plantation, however, is less capable of providing other goods and services supplied in primary forests because of reduced biodiversity (e.g., Brockerhoff et al. 2008, Thompson et al. 2009). Slash-and-burn agricultural landscapes are degraded compared to well-managed forests, but not in the context of livelihood options (Schmidt-Vogt 1998). Similarly, unsustainable logging may degrade timber stocks and soils but enhance the yields of nontimber forest products (NTFPs) important to local communities (e.g., Salick et al. 1995). In terms of landscape sustainability, however, forest degradation should be considered with respect to multiple ecosystem goods and services that could be derived under sustainable ecosystem management. Consistent with the CPF definition, we suggest that forest degradation represents changes to ecosystem processes and a continuum of decline in services from levels in primary forests (i.e., unaltered by humans except for traditional forest uses), through various forms of human use and (mis-)management, to deforestation (see e.g., Chazdon 2008).

Here, we propose a suite of criteria and indicators that can be used by forest managers, and interpreted by policy makers, to quantify types of forest degradation according to specific circumstances and objectives. The indicators enable tests of hypotheses relative to expected conditions of ecological integrity. We have organized this paper as a series of sections that provide a discussion of the selected criteria and indicators, followed by a discussion of our approach to a minimum subset of indicators that could be immediately applied sub-nationally, nationally, and globally.

CRITERIA AND INDICATORS FOR DEFINING FOREST DEGRADATION

We selected our criteria based on the following considerations: relationship to the main causes of forest degradation (e.g., Kissinger et al. 2012); relationship to the provision of ecosystem services as defined in the Millennium Ecosystem Assessment (Diaz et al. 2005); relevance to variables reported in the *Global Forest Resources Assessment* (e.g., FAO 2010); potential for the application of remote sensing for monitoring; and relevance to reporting requirements for sustainable forest management (SFM), as required, for example, by the Montréal Process (2009), the Ministerial Conference on the Protection of Forests in Europe (Vienna, Liaison Unit 2003), and the Lapaterique Process (FAO 2002). SFM and forest degradation are related, but opposite, concepts, and so several SFM criteria can also be degradation criteria. Based on these considerations, we selected five criteria to determine degradation: productivity, biological diversity, unusual disturbances, carbon storage, and protective functions. We included carbon stock as a separate criterion, despite its close link to productivity, because of its high relevance to UNFCCC in the context of emissions reductions and for forest-carbon projects. These five criteria should not be viewed as necessarily equivalent; we expect that certain criteria might be considered more important than others, depending on local or national circumstances and on the objectives for a particular forest. However, under an ecosystem management regime, all criteria should be considered. For each criterion, we suggest using some existing SFM indicators (e.g., Siry et al. 2005), and we propose several others. Monitoring the indicators for each criterion enables the CPF definition of forest degradation to become operational. Although we do not provide guidance for monitoring, most of the suggested indicators have been assessed using remote sensing (Tables 1 and 2). We provide a review of the use of remote sensing to measure forest degradation.

Indicators and spatial scale

Degradation is ecosystem- and location-dependent in terms of parameters that might be measured, e.g., a particular species or certain forest goods. Stand-level assessments are required for local understanding, but the indicators must also be applicable for forest management units and at sub-national and national levels for international reporting purposes (e.g., to CBD; e.g., Sasaki and Putz 2009). Hence, the indicators must be unambiguous, readily repeatable, and provide quantitative data to assess trends over time (e.g., Gardner 2010) and also allow for scaling up. Management decisions about forest removals are often taken for large areas and not only for individual stands. Therefore, forest degradation will be most effectively reported for landscapes; these are usually locally defined areas > 5000 ha. Such landscapes will necessarily contain multiple stand types and conditions that often reflect forest recovery under natural or assisted processes, which may or may not be degraded.

Table 1. A suggested framework of criteria and indicators for defining and delineating areas of degraded forest. A suggested minimum set of seven indicators is indicated by an asterisk (*).

Criterion	Indicator(s)	Variable(s)	General methods
Production	Growing stock*	m³/ha of wood	Satellite imagery, LiDAR, ground plots
	Nontimber forest products	Monetary value, number/yr	Country reporting, questionnaires by management unit
Biodiversity	Ecosystem state*	Area of specific forest type	Satellite imagery
	Fragmentation*	Area fragmented	Satellite imagery, aerial photography
	Species	Presence/absence, population density, relative abundance, indicator of abundance	Aerial or ground surveys
Unusual disturbances	Invasive species*	Population density, area affected	Satellite imagery, aerial photography, ground surveys
	Fire*	Area affected	Satellite imagery, aerial photography
Protective function	Soil erosion*	Area affected	Satellite imagery, aerial photography
	Water volume or flow	Flow rate	River or stream flow meters
Carbon storage	Stored carbon*	Biomass/ha	Satellite imagery, ground plots
	High wood-density trees species	Tree density, relative abundance	Ground plots, aerial photography

Reference condition

Degradation is both a state (i.e., the forest is degraded) and a process as a continuum across time and/or space (i.e., the forest is being degraded; Chazdon 2008, Ahrends et al. 2010). With one exception (i.e., forest ecosystem state) our indicators address levels or amounts of degradation as a process. In all cases, reference conditions specific to a forest ecosystem type are needed against which to determine the occurrence and extent of degradation (e.g., Huettner et al. 2009). Forests change continuously through natural processes (e.g., climatic variation); although forests are normally resilient and resistant to such processes (e.g., Thompson et al. 2009), understanding the range of natural variability in the parameters of interest through space and time (Keane et al. 2009) is a prerequisite to measuring forest degradation (Fig. 1). When an indicator of forest condition is measured as being consistently below a specified level (i.e., from several measurements over time), the forest should be considered degraded. Therefore, repeated measurements are essential.

An appropriate reference state must be selected with care. Primary forests can provide a baseline against which to measure degradation, including in various successional stages. Natural forest, however, should not be the sole reference state considered because this implies that SFM degrades a forest, resulting in confusion about what is and is not degraded. Clearly, unplanned and poor logging practices are often the precursors to other human interventions leading to further degradation and ultimately to deforestation over time (Nepstad et al. 1999, Asner et al. 2006). Nevertheless, although a sustainably logged tropical forest stand may not necessarily contain a complete biodiversity complement (e.g., Asner et al. 2006, Lewis 2009, Klimes et al. 2012), SFM has dramatically

Fig. 1. Degradation must be measured against a desired baseline, and the natural variation of that baseline must be understood. The upper nonmonotonic line represents natural variation of an indicator over time (e.g., annual fluctuation) and enables the calculation of baseline variance. If the trajectory consistently falls into the red zone, as does the orange line, the forest should be considered degraded with respect to the particular indicator and criterion.
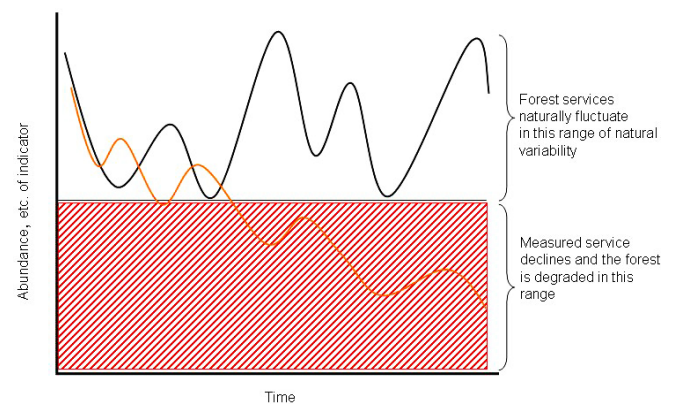
improved forest conservation (e.g., Foley et al. 2007, Peña-Claros et al. 2007, Putz et al. 2007, 2012, Edwards et al. 2011). Allowable removals can be determined for goods or services based on expected pre-defined management objectives; if these are properly established, they can serve as baseline values. Therefore, the appropriate baselines for indicators should be a combination of what would normally be expected

Table 2. Examples of studies that have used remote sensing to report on the minimum set of suggested indicators of forest degradation (from Table 1).

Indicator	Study	Location	Methods	Results/outputs
Growing stock	Hollaus et al. 2009	Austria	LiDAR and ground	LiDAR canopy correlated to growing stock
	Gallaun et al. 2010	Europe	Satellite and ground	MODIS model for aboveground biomass for Europe
	Souza et al. 2005	Brazil	Satellite and ground	Landsat TM model comparing biomass of primary and degraded forests
Ecosystem state	Strand et al. 2007	Summary report	Satellite	Multiple examples of Landsat, ASTER, SPOT HRV mapping of forest types at 15 to 250-m resolution
	Souza et al. 2005	Brazil	Satellite	Classification and mapping of forest types using Landsat TM and ETM
Fragmentation	Harper et al. 2007	Madagascar	Satellite	Landsat used to estimate decline in forest cover over time
	Achard et al. 2002	Global	Satellite	Rate of fragmentation and loss of tropical humid forests for 1990–1997
	Sánchez-Azofeifa et al. 2001	Costa Rica	Satellite	Fragmentation index derived showing change over time with Landsat TM
Alien invasive species	Theron et al. 2004	South Africa	Satellite and ground	Mapping invasion by *Acacia* using Landsat ETM
	Ferreira et al. 2005	Portugal	Aerial photography	Mapping invasion by *Pinus pinaster*
	Tsai et al. 2007	Taiwan	Satellite	Mapping invasion by *Leucaena leucocephala* using hyperspectral imagery
	Le Maître et al. 2002	South Africa	Aerial photography	Mapping invasion by *Acacia*
	Everitt and Deloach 1990	USA, Mexico	Aerial photography	Mapping invasion by *Tamarix chinensis*
	Pontius et al. 2008	USA	Satellite	Mapping area damaged by *Agrilus planipennis* using hyperspectral images
Fire	Souza et al. 2005	Brazil	Satellite	Area mapping using Landsat TM and ETM
	Dennis and Colfer 2006	Indonesia	Satellite	Area mapping using Landsat TM and ETM
	Matricardi et al. 2010	Brazil	Satellite	Area mapping using Landsat
Soil erosion	Prasannakumar et al. 2011	India	Satellite and ground	Quantifying erosion area
	Maya et al. 2004	Mexico	Satellite	Quantifying erosion area using SPOT
	Lu et al. 2004	Brazil	Satellite	Mapping of erosion and erosion potential using Landsat ETM
	Manchanda et al. 2002	India	Satellite	Mapping of erosion and erosion potential
Stored carbon	Baccini et al. 2012	Global tropical	Satellite	Carbon density mapping using satellite LiDAR
	Ryan et al. 2012	Mozambique	Satellite	Aboveground biomass mapping using synthetic aperture radar (0.06 ha resolution)
	Saatchi et al. 2011	Global tropical	Satellite	Carbon density mapping using satellite LiDAR
	Goetz et al. 2009	Africa	Satellite, LiDAR, ground	Comparison among methods
	DeFries et al. 2006	Global tropical	Satellite	Carbon density mapping using MODIS and Landsat

for that particular forest type, either as a primary forest or a sustainably managed forest with specified values for emphasized goods and services.

CRITERION 1: PRODUCTIVE FUNCTIONS
The long-term production of forest goods and services is a key objective of SFM, and the harvesting of forest goods is not considered degradation unless it leads to significant reductions in availability (Chazdon 2008). Forest degradation may result in lower future biomass accumulation in recovering forests owing to a shift in species composition (i.e., ecosystem state; see Criterion 2) and tree size class structure, loss of highly

functional species (Díaz and Cabido 2001, Russell et al. 2010), and/or a reduction at the landscape scale of mature forests (for example, Kirby and Potvin 2007, Lewis et al. 2009).

Production can be assessed by comparing the harvesting of forest goods to sustainable levels defined in a forest management plan. Examples of common indicators of timber production include the annual allowable cut, mean annual increment (MAI), and stocking density (e.g., Seidl et al. 2007). In cases of subsistence use of forest goods, evidence of resource degradation can be assessed by monitoring sequential reductions in the abundance, density, and/or size class distribution in a forest over time or over a given distance from

a defined point (e.g., Ahrends et al. 2010). Degradation of productive functions may be best estimated by assessing changes in growing stock and measures of the production of specific NTFPs. Carbon is a separate criterion (see Criterion 5), but clearly part of total forest biomass is estimated from growing stock.

Indicators of production

Growing stock
Growing stock is the standing volume of all trees (or selected commercial species) of a specified diameter and height in a forest area and can be measured as stocking density (m³/ha), basal area (m²/ha), or total volume (m³; e.g., FAO 2005). Growing stock is often estimated through field measurements using large-scale forest inventories (Bahamóndez et al. 2009). Minimum values for stocking density by forest type (and commercial tree species) and geographic zone must be established as reference values to determine if the actual growing stock can be considered adequate (undegraded; e.g., Phat et al. 2004); these equations already exist widely. Estimates of growing stock can be used to evaluate and monitor the potential of a forest for timber and fuelwood production. An annual allowable cut is the most common growing stock estimate, for a given time period, and can be used to monitor forest production and set limits for sustainable use (e.g., Seidl et al. 2007). Growing stock is usually reported as an aggregate figure for all commercial species, but can be reported by species, species groups, or stands. Information on growing stock is essential for understanding the ecological dynamics and productive capacity of forest stands, enabling managers to stay within the limits of sustainability. Degraded production refers to reduced growing stock, population declines of certain tree species, and diminished reproductive capacity of commercial species or targeted population levels (e.g., abandoned or exhausted coppices in certain forest types) relative to baseline values (Chazdon 2008). Although more difficult to assess in cases of subsistence management, growing stock degradation can be recognized through sequential reductions in abundance, density, and/or size class distribution in a forest, either over time or distance from a defined point (e.g., Ahrends et al. 2010). Loss or reduced populations of selectively harvested tree species are discussed under *Criterion 2: Biodiversity,* but also apply well as an indicator of productivity (e.g., Kirby and Potvin 2007).

For fuelwood and charcoal, national-level forest statistics on production and consumption tend to be gross estimates based on demographic data and estimated consumption per household (e.g., Girard 2002). Direct measurements of the resource and supply are more accurate and should be encouraged, particularly for dry tropical forests, where fuelwood and charcoal may be the principal uses of wood (Girard 2002). At a sub-national level, the supply of fuelwood can be determined by quantifying growing stock of species

used for charcoal and their expected and realized annual increments.

Various remote sensing techniques for assessing growing stock have been developed using aerial photography, satellite imagery, and more recently using LiDAR imaging from aircraft, based on correlations to ground data (Table 2). Several studies suggest that primary or well-managed forests vs. degraded forests can be detected readily via satellite remote sensing (e.g., Souza et al. 2005, Harper et al. 2007, Olander et al. 2008, Asner et al. 2009, Matricardi et al. 2010, Matsushita et al. 2010).

Nontimber forest products (NTFPs)
NTFPs contribute to local economies and include traditional medicines (e.g., Shanley and Luz 2003, Shackleton et al. 2007). The diversity of species, distribution patterns, or seasonality of plants providing NTFPs require specific sampling designs for quantification. Therefore, assessments of NTFPs have had to focus on areas where commercial collection is concentrated or that are known as excessively used for subsistence consumption. Countries report estimates of consumption of commercially important NTFPs (e.g., FAO 2010); however, the reliability of these estimates is questionable because the consumption is often local and is not recorded (Vantomme 2003, Shackleton et al. 2007), and some products originate from areas not classified as forest (FAO 2010). National-level data do not account for variation by geographical zones or among forest types.

We suggest that social and socioeconomic indicators can be used as proxies for NTFPs, with the caveat that these indicators must be adjusted for other factors that may influence NTFPs. Social indicators can help to validate information obtained through direct measurement. For example, Kleine et al. (2009) noted that local knowledge about the absence of desired forest products and services assisted in describing the level of forest degradation in India. Similarly, medicinal plant depletion was reported in an area of Brazil (Shanley and Luz 2003). Data on household consumption can be collected using rapid rural assessments such as those specified in the Poverty–Forest Linkages toolkit (PROFOR 2010). The methods of Ahrends et al. (2010) for assessing growing stock (see above) could be applied to production of NTFPs. More directly, age and size distributions of plant species used for NTFPs, at the stand or landscape levels, could be measured. For example, Peres et al. (2003a) determined the sustainability of Brazil nut harvesting across Bolivia, Brazil, and Peru by assessing use relative to tree population. Such data could be compiled over time and, assuming constant demand, reduced use of NTFPs may indicate a reduction in supply.

CRITERION 2: BIODIVERSITY
Biodiversity provides essential ecosystem functions in forests such as pollination, decomposition, seed dispersal, carbon

sequestration, pest and disease reduction, and emergent properties such as resilience (e.g., Balvanera et al. 2006). Reduced biodiversity often directly affects forest processes that produce goods or services (e.g., Díaz et al. 2003, Thompson et al. 2009), although species redundancy may retard visible effects on processes (Walker 1992). Nevertheless, reduced biodiversity is directly related to lower levels of many goods and services, particularly if the losses involve highly functional species (e.g., Díaz and Cabido 2001, Balvanera et al. 2006), key tree species used for wood products (Kirby and Potvin 2007), or food animals (Harrison 2011). For example, empty forests syndrome (e.g., Redford 1992, Harrison 2011) is a cryptic form of degradation in which animal species have been overexploited for food but there is no obvious change in forest structure or in the supply of other goods. Loss of animal species may eventually result in ecosystem degradation because of the loss of seed dispersal and changes in herbivory (e.g., Wright et al. 2007, Terborgh et al. 2008). Forest landscapes can become degraded from a biodiversity perspective through the loss of species, genes, forest types, forest fragmentation, or the large-scale conversion of forest ecosystems to other forest types. While biodiversity effects on ecosystem functioning occur at all scales (e.g., Balvanera et al. 2006), we suggest that species and landscape indicators are most appropriate to monitor degradation. We propose three indicators for biodiversity: ecosystem state, fragmentation, and selected species.

Indicators of biodiversity

Ecosystem state
Forest ecosystem state, or type, is the recognizable normal species composition and structure of the dominant trees for a given site, and a change in state refers to a change in species composition of the vegetative cover. Forests are normally resilient to natural disturbances within bounds and recover following major disturbances (Gunderson 2000, Folke et al. 2004, Walker et al. 2004), but a loss of biodiversity from excessive disturbances will often reduce that capacity, resulting in different vegetation (Folke et al. 2004, Thompson et al. 2009). Such changes in forest state may result in changes in processes, degrading the production of goods and services (Diaz et al. 2005, Pardini et al. 2010). Recovery of the original forest state is unlikely without management (e.g., Gunderson 2000, Macdonald 2004).

Classifications of forest ecosystems (dynamic complexes of plant, animal, and micro-organism communities and their nonliving environments) can be made at various scales (Allen and Hoekstra 1992), so the ecosystem state indicator can refer to individual stands or forest landscapes. At the landscape scale, classifications are broad forest types such as tropical dry forest or tropical rainforest (e.g., FAO 2001), and a change in state represents a major change in vegetation such as, for example, from dry forest to savannah. At finer scales,

classification of forest types is based on plant associations, elevation, moisture, and soil types (e.g., Swaine and Hall 1976, Letouzey 1985, Slik et al. 2009).

At the stand level, change in state refers to a different stand type such as an open canopy forest instead of closed canopy forest, or a monotypic stand instead of a mixed species stand. At the landscape scale, change in forest ecosystem state can be quantified as the area of existing forest types relative to the area of each forest type predicted or expected. Satellite imagery can be used to map forest or other land-cover types to assess changes in forest ecosystems at various scales (Table 1). Forest types that cannot be identified with accuracy using satellite images, usually at fine scales, require more expensive techniques such as aerial photography, LiDAR, or ground-truthing.

Forest fragmentation
Fragmented forests are degraded because species may be lost and ecosystem processes may be altered or severely disrupted (e.g., Pardini et al. 2010). Land-use change often leads not only to a reduction in forest area but also to the division of remaining forest into patches that may continue to decrease in size over time, creating excessive edges and isolation from continuous habitat (Collinge 1996, Fahrig 2003, Saura and Carballal 2004). At the landscape scale, there are system-specific and species-specific fragmentation thresholds that, once surpassed, cause significant loss of interior forest species (e.g., Andrén 1994, Santos et al. 2008, Swift and Hannon 2010, Pardini et al. 2010). Sufficient fragmentation may also cause cascading effects on a wide range of ecosystem functions and services (Wu et al. 2003, Diaz et al. 2005). For example, forest fragmentation can reduce carbon storage at the landscape scale (Groeneveld et al. 2009) and can alter hydrological cycles, causing changes in evapotranspiration, local climate, and run-off rates (Ziegler et al. 2007). Fragmentation has significant negative implications for many species, including effects from reduction in total available habitat, reduction in forest interior, increased edge effects, exploitation, and spatial and genetic isolation (Laurance et al. 2000, 2011, Fahrig 2003, Broadbent et al. 2008). Populations of large animals and rare species that prefer forest interiors generally require large areas of contiguous habitat for population persistence and so are especially vulnerable to fragmentation (e.g., Haskell et al. 2002, Cox et al. 2004, Vellend et al. 2006, da Silva and Mendes Pontes 2008, Issac et al. 2009, Thornton et al. 2011). Natural forests that have become increasingly fragmented on a global scale pose a substantial threat to biodiversity (see reviews by Fahrig 2003, Fischer and Lindenmayer 2007).

Fragmentation of forests can be determined from satellite imagery (Table 2) and aerial photography over time. Several metrics can be used to assess fragmentation, and these are available in software such as FRAGSTATS (McGarigal et al. 2012). The value of available fragmentation metrics depends

on the level of understanding of their relationship to the goods and services of interest. Moreover, to minimize confounding effects, indicators must be compatible with, and interpretable in the light of, information on change in forest area.

Species
Abundance of species, or groups of species, is used commonly as an indicator for monitoring the effects and effectiveness of forest management (e.g., Oliver and Beattie 1996, Noss 1999, Lewandowski et al. 2010). Many species are sensitive to forest degradation, and numerous examples are available of effects of forest change on species populations (e.g., Lindenmayer et al. 2002, Colles et al. 2009). The choice of species indicator must be made carefully relative to the scale of the effect and with the understanding that other processes may provide alternative hypotheses. Species may be site, stand, or landscape-level indicators depending primarily on their body size and area requirements (e.g., Gardner 2010).

The species approach to indicating forest change has been criticized because of the difficulty in obtaining precise annual population estimates and the high inter-year variability of populations (e.g., Landres et al. 1988, Rolstad et al. 2002). The data requirements may be limiting, both to demonstrate the correlation between species abundance and degradation and to detect changes in populations with statistically valid confidence (Lindenmayer et al. 2002). Therefore, the use of species as indicators requires an understanding of the limitations of the technique. We propose three types of species indicators, including tree species composition, selected focal species (i.e., flagship, umbrella, etc.), and functional species. In all cases, consistent decline in areas affected by humans compared to baseline levels are indicative of degraded forests and the habitats they once supported. For monitoring the multiple processes that together compose degradation, several species (or groups of species representing various trophic levels) indicators may be required.

Tree species composition: The criticisms leveled at species indicators do not apply to plants because they can be easily censused. Achieving a similar tree species composition over time in managed compared with unmanaged stands is an objective of SFM (e.g., Montréal Process 2009). Significant population decline of expected tree species could result in degradation of specific goods and services (e.g., Devi and Behara 2003, Luyssaert et al. 2008) and may indicate a loss of other obligate species associated with particular tree species, such as lichens (e.g., Linder and Östlund 1992). Tree species composition can change as a result of the over-harvesting of individual commercially valuable species, or excessive fire (e.g., Devi and Behera 2003, Asner et al. 2006, Foley et al. 2007). Unsustainable harvesting often occurs, particularly in tropical forests, where certain hardwood species are highly sought after, e.g., ipê (*Tabebuia* spp.; Schulze et al. 2008) and big-leaf mahogany (*Swietenia macrophylla;*

Kometter et al. 2004, Grogan et al. 2010). Monitoring the tree species composition of a stand over time provides information directly related to ecosystem change.

For humid tropical forest types, where there is high tree species richness at low individual densities, estimates of tree species composition may require that the baseline be defined at the landscape level or by concentrating monitoring on only a few valuable timber species (e.g., Kometter et al. 2004). For more rapid large-area comparisons, remote-sensing techniques are preferred, but only some tree species are identifiable from satellite images (e.g., Carleer and Wolfe 2004, Clark et al. 2005); most species require aerial photography, LiDAR, or ground surveys for identification.

Selected focal species: Many agencies and researchers have successfully used species indicators to monitor ecosystem change because changes in population size or distribution may suggest that habitat abundance or quality have been altered (for esample, McLaren et al. 1998, Festa-Bianchet et al. 2011). Other species are monitored because they are used as game (bushmeat) or are threatened (e.g., Nasi and Frost 2009, Harrison 2011). Hence, data may already be available to enable comparisons over time or against a baseline from a nearby protected area. Species-level monitoring in forests necessarily involves ground or possibly local aerial surveys.

Functional species or groups: Some species are more important than others in providing certain ecosystem functions (e.g., Walker 1992, Díaz et al. 2003), and their loss disproportionately reduces stand processes and the production of certain services (e.g., Díaz and Cabido 2001, Ellison et al. 2005). The loss of highly functional species, in the absence of redundancy (Walker 1992), has negative consequences for ecosystems to the point of ecosystem change or even collapse (Chapin et al. 1997, Ellison et al. 2005). For example, the loss of some or all pollinators would have severe consequences for reproduction by many plants. Some functional species are also keystone species that carry out roles in ecosystems that affect other species such that their loss results in cascading changes (e.g., Terborgh and Estes 2010) and so can indicate forest degradation.

Certain bird, butterfly, or ground beetle assemblages are often used as indicators because of data availability and ease of collection and because many have known functions such as pest reduction and pollination (Lawton et al. 1998). Woodpeckers have been suggested as a forest indicator because of their functional roles as cavity excavators (e.g., Mikuinski and Angelstam 1997, Drever and Martin 2010). Insectivorous birds can regulate insect herbivore populations (Bridgeland et al. 2010), act as seed dispersal agents, and some are pollinators. Many insectivorous birds respond negatively to selective logging and partial harvesting in tropical forests (e.g., Aleixo 1999) and to age composition changes at landscape scales (Yamaura et al. 2009) and so can be used as

indicators of forest degradation. Bird census methods are widely available but must be carried out in the field.

Pollinators are directly or indirectly related to ecosystem productivity (e.g., Ricketts et al. 2004, Klein et al. 2007), especially in tropical systems (e.g., Samejima et al. 2004). In southeast and east Asia, bees from the genera *Micrapis, Magapis,* and *Apis* are particularly useful indicators of forest degradation because they are pollinators of tree flowers and also nest in forests (Oldroyd and Wongsiri 2006, Hepburn and Radloff 2011). Most key pollinators (especially honeybees) require relatively large dead standing trees in which to nest (Seeley and Morse 1976), so the presence of these large snags could be correlated to the presence of bees.

Decomposers in forests help to maintain water and soil quality and promote nutrient cycling (Harris 2009). Microorganisms are important forest decomposers, but little qualitative or quantitative information is available about how they function (e.g., Harris 2003). The soil microbial community is dependent on the level of site disturbance, and therefore, may indicate effects from unsustainable management practices (Harris 2003, 2009). The suite of methods available for developing a monitoring program for decomposers is described by Ritz et al. (2009).

CRITERION 3: UNUSUAL DISTURBANCES

Extensive unusual disturbances affect the capacity of a forest landscape to supply ecosystem services and reduce resistance to biotic and abiotic stresses (e.g., van Wilgen et al. 2001, Balvanera et al. 2006, Aragão and Shimabukuro 2010). Forest ecosystems are continuously influenced by biotic and abiotic agents at all spatial scales (from individual trees to entire forest types), intensities of impact, and combinations of agents (for example, Attiwill 1994). As long as these disturbances do not exceed the natural variation of an ecosystem over time, they will not cause long-term forest degradation (e.g., Lugo and Scatena 1996, Folke et al. 2004, Thompson et al. 2009). However, some agents may exert severe negative effects to an extent that the original ecosystem changes and the delivery of services is impaired (e.g., van Wilgen et al. 2001, Balvanera et al. 2006). We propose two indicators of specific agents of change, i.e., alien invasive species and fire, both of which have become major causes of forest change in recent decades (e.g., Richardson 1998, Vié et al. 2009, Aragão and Shimabukuro 2010). Arguably, alien invasive species could be considered a biodiversity indicator, but we prefer to separate causes and effects: our biodiversity indicators illustrate effects of forest degradation, whereas invasive species can cause degradation. Other locally important indicators could include native invasive species, incidence of storms, and pest or pathogen outbreaks that are longer and more extensive than normal based on past monitoring.

Indicators of unusual disturbances

Alien invasive species
In a given forest type, an invasive alien species is a non-native species that has established a persistent population and is causing harm (Pimentel et al. 2005). Invasive alien species are an important global cause of ecosystem change (e.g., Pejchar and Mooney 2009, Vié et al. 2009), often resulting in a consequent reduction in local biodiversity and ecosystem services (Simberloff et al. 2002, Strayer et al. 2006, Pejchar and Mooney 2009) with economic consequences (Mack et al. 2000, van Wilgen et al. 2001). Globally, many forests have been degraded by invasive alien species (e.g., Chornesky et al. 2005) through mechanisms such as competition, herbivory, disease, or predation (e.g., Lucier et al. 2009). Some forests have been invaded by introduced tree species (Richardson 1998) that have caused ecosystem change. For example, invasive *Acacia* trees in South Africa are estimated to use up to 10% of the available water, substantially reducing flows in rivers (van Wilgen et al. 2001, Le Maître et al. 2002). Many invasive tree species occur in, or even dominate, forest canopies and so can been mapped remotely (Table 2). In Hawaiian montane rain forest, Asner and Vitousek (2005) used infrared imaging spectrometry to show that leaf nitrogen concentrations in native *Metrosideros polymorpha* forests were reduced in areas invaded by *Myrica faya.* Less obvious invasive tree species must be censused using ground surveys.

Alien invasive insect herbivores such as emerald ash borer (*Agrilus planipennis*) in eastern North America (http:// emeraldashborer.info/), pests such as pinewood nematode (*Bursaphelenchus xylophilus*) throughout eastern Asia (Liebhold et al. 1995, Zhao et al. 2008), and alien pathogens such as Dutch elm disease (*Ophiostoma ulmi* and *O. novo-ulmi*) in many areas (Hubbes 1999) have caused degradation over millions of hectares of forests. In most cases and depending on extent, once the cause of damage is known, the area of forest change can be mapped using aerial photographs or satellite imagery (e.g., Pontius et al. 2008). Other invasive species such as ants and earthworms have caused cascading effects over large areas as a result of competition or replacement of endemic species (Kenis et al. 2009, Straube et al. 2009). However, some of these changes may be subtle and difficult to monitor remotely and can only be observed using ground or aerial surveys.

Fire
Human-caused fire is a major cause of forest degradation (for example, Asner et al. 2006), possibly exacerbated by climate warming (e.g., Siegert et al. 2001). An unusual amount of fire can reduce the resilience of ecosystems, resulting in change or loss of ecosystem services (Thompson et al. 2009). Although fire is a natural element in many forest ecosystems, humans have altered fire regimes across 60% of global terrestrial habitats (Shlisky et al. 2009). Fires have spread in

extent and frequency in many tropical rainforests with the expansion of agriculture (Uhl and Buschbacher 1985), forest fragmentation, unsustainable shifting cultivation, and logging (Nepstad et al. 1999, Siegert et al. 2001, Alencar et al. 2006). Fire often follows unsustainable logging, resulting in poor-quality secondary forests (e.g., Asner et al. 2006). Trees in tropical humid forests are particularly susceptible to fire damage because fires are historically rare (Aragão and Shimabukuro 2010) and the regional flora and fauna are not adapted to such events (Uhl and Kaufmann 1990, Peres et al. 2003*b*). The loss of natural habitats and species due to excessive fires can cause changes in plant communities and trophic structures within those communities (Dobson et al. 2006, Wright et al. 2007). Fire is likely to increase in the future because of drought associated with global warming (e.g., Siegert et al. 2001). Fires can be quantified by frequency and area burned using satellites (Table 2), especially during or soon after fires because heat or blackened vegetation is readily detected (e.g., Cuomo et al. 2001).

CRITERION 4: PROTECTIVE FUNCTIONS

Protective functions refer to the intrinsic property of forest ecosystems to maintain soils, soil structure, quality, and moisture levels (e.g., Brandt 1988), which ultimately contribute to forest resilience. Soils play key roles in forest biogeochemical cycles; erosion causes degradation through siltation of watersheds, reduced soil stability, and reduced fertility (Islam et al. 2001, Hartanto et al. 2003, Cotler and Ortega-Larrocea 2006, Neary et al. 2009), and increased rates of rainfall run-off (Lal 1992). Roads cause a disproportionate amount of soil erosion in tropical forests (Dykstra and Heinrich 1992). Vegetation is key to controlling the type and intensity of erosion (e.g., Thornes 1990, Morgan 2005), and eroded soils are often difficult to re-forest (Morgan 2005). Forests play a role in landscape hydrology by regulating water flows through evapotranspiration and infiltration rates (Giambelluca 2002). Although our understanding of this function is not clear with respect to degradation, generally there are more predictable base flows and higher dry season flows from forested areas compared to areas where forest cover has been reduced (e.g., Bruijnzeel 2004, Locatelli and Vignola 2009). We propose two indicators of protective functions: soil erosion and water quantity.

Indicators of protective function

Soil erosion
Soil erosion occurs when wind and water translocate soil particles. Although some soil erosion is inevitable through normal precipitation and run-off, it is exacerbated by poor management practices such as inappropriate road placement or timber harvesting methods, especially in areas prone to soil movement such as steep slopes. Stand density plays a large role in avoiding soil erosion (Razafindrabe et al. 2010); for

example, forest cover loss in Ghana increased the erosion rate from 1 tonne ha^{-1} yr^{-1} to > 100 tonnes ha^{-1} yr^{-1} (Repetto 1990). Nutrient loss is a common result of soil erosion. For example, shifting cultivation causes 18–27% declines in soil carbon, and fallow periods of at least 35 years are required for soils to regain their original carbon content (Detwiler and Hall 1988). The area affected because of soil erosion is the reportable indicator based on remote sensing (Table 2).

Water quantity
In areas where forest cover is known to help regulate water flow, water levels in rivers and streams among and within years, compared to normal levels, may indicate if sufficient forest cover remains to regulate flows, especially in dry seasons (Bruijnzeel 2004), and in some cases, to prevent flooding (e.g., Douglas et al. 2005). We stress the need for more research on the value of this potential indicator. Flow rates would be taken from field data (Table 1).

CRITERION 5: CARBON STORAGE

We separated the carbon storage criterion from our production criterion because of current interest in the importance of forests for the global carbon cycle. Approximately 50% of global terrestrial carbon stocks reside in forest ecosystems as living and dead biomass and soil carbon (Parry et al. 2007, FAO 2010), and forests store an estimated 861 ± 66 Pg of carbon (Pan et al. 2011). Forest degradation from unsustainable logging, fire, shifting cultivation, and other human-related disturbances can result in substantial reductions in carbon stocks and in the capacity for carbon storage (e.g., Asner et al. 2006, 2009, Foley et al. 2007). Chazdon (2008) suggests a continuum of forest regeneration (in a sense, opposite to degradation) following human disturbances that influences the carbon dynamics in forests. Under this model (see Fig. 2 for an adaptation), the concept that natural and close-to-natural forests store more carbon than degraded forests has been reported for multiple forest types (e.g., Keller et al. 2004, Erskine et al. 2006, Foley et al. 2007, Asner et al. 2009, Chatterjee et al. 2009, Liao et al. 2010, Powers et al. 2011). A hierarchy of degradation effects (Fig. 2) is reinforced by studies showing that primary old-growth forests provide high-value carbon sinks and will continue to do so for centuries in all forest biomes unless they are disturbed (Phillips et al. 1998, Baker et al. 2004, Luyssaert et al. 2008, Lewis et al. 2009). Forest landscape heterogeneity and fragmentation also influence carbon storage in forests (Numata et al. 2011), suggesting that our fragmentation indicator may also correlate with carbon storage. Forest degradation reduces future biomass accumulation by causing a shift in species composition, tree size structure, tree species richness, and abundance of lianas (Asner et al. 2006, Foley et al. 2007, Russell et al. 2010). We propose two indicators: total carbon stored and populations of high wood-density trees.

Fig. 2. Degradation is a continuum of decline in ecosystem goods and services from primary forest to deforested lands. This continuum is illustrated for a generalized dry closed tropical forest. Biodiversity refers to species richness and ecosystem type richness relative to a baseline. Ecosystem services refers to services related directly to biodiversity such as pollination and biomass production. Sustainably managed forests are shown close to, but slightly below, primary forests based on multiple studies (e.g., Asner at al. 2006, Lewis 2009, Klimes et al. 2012; Putz et al. 2012).
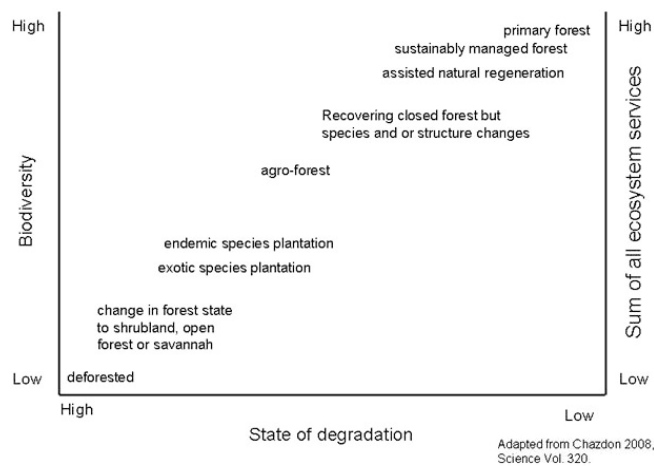
Indicators of carbon storage

Stored carbon
Methods and tools are available for estimating forest carbon stocks, with default values, including biomass functions and/ or conversion factors, available for different carbon pools and ecological zones (IPCC 2006, Herold et al. 2011*b*). Several studies have used satellites and other techniques to map carbon density (Table 2). Nevertheless, remote-sensing tools for biomass estimation still require ground corroboration in many cases to improve the accuracy and precision of carbon estimates (e.g., Herold et al. 2011*a*). Ground census work will improve estimates from models, especially for fallen dead wood, belowground carbon, and sub-canopy structure (e.g., Goetz et al. 2009, Bryan et al. 2010, Eckert et al. 2011).

High wood-density tree species
Tree growth rates and accumulated wood fiber biomass represent changes in the aboveground carbon sink. For the same size of tree, dense-wood species store more carbon than do trees with less-dense wood (e.g., Chave et al. 2006, Russell et al. 2010, Potvin et al. 2011). Although there is only an incomplete understanding of the relationship between carbon sequestration and tree species and among carbon sequestration, forest management, and disturbance, some tree

species require more carbon than others during growth, usually grow slowly, and ultimately store more carbon for a long period (e.g., Bunker et al. 2005, Vieira et al. 2005, Russell et al. 2010, Sharma et al. 2010). As a result, forests may be degraded in terms of the carbon stored if tree diversity is reduced, for example, through excessive selective logging of high wood-density tree species such as mahogany (Kirby and Potvin 2007). Hence, monitoring populations of high wood-density tree species over time relative to an expected population (by age class) can indicate forest ecosystem degradation (see stand-scale ecosystem diversity and biomass indicators, above) with respect to carbon storage over the long term (e.g., Vieira et al. 2005). Monitoring for individual trees would necessarily be at the stand level from aircraft or ground plots.

MINIMUM SET OF INDICATORS
As previously suggested, what constitutes forest degradation is often related to local perception and societal choice. Nevertheless, to understand fully how ecosystem services might have been reduced or lost, and especially under an ecosystem management regime, each of the five criteria should be reported to quantify types of degradation. To accomplish this, we suggest that a minimum subset of seven indicators should be monitored (Table 1). These seven indicators were selected based on the need for at least one indicator from each criterion, the current capability to measure the indicator using remote sensing, the existence of data to enable trend analysis in many parts of the world, and the direct linkage to regional and global reporting processes (e.g., CBD and SFM). The indicators can be compiled as a percentage deviation from the baseline and portrayed graphically (e.g., Fig. 3). This diagram maintains the strength of the individual indicators and also provides a graphical representation emulating a single composite index (e.g., Bahamóndez et al. 2007, 2010). Graphical representation easily enables managers to see the relative contribution by types degradation that are affecting their forests.

Measurement frequency
We suggest that 3 to 5 years should be a sufficient monitoring interval to allow enough time for change to occur for most indicators. It may be necessary, however, to monitor selected individual species more frequently because of potential rapid population change. In the cases of tree species richness, ecosystem diversity, and fragmentation, change is often much slower than for the other indicators; these indicators may only require monitoring every 5 years or more to observe a trend.

DISCUSSION
Forest degradation is a topic of global concern because of the loss of goods and services, the emission of greenhouse gases, and problematic restoration. Clearly, what is of greatest concern is the long-term, large-scale degradation of forests (for example, ITTO 2002), especially when the ecosystem has

changed states and is severely degraded. Although many definitions of forest degradation converge on the loss of ecosystem services (Simula 2009), no previous operational guidance has been available to help define degradation. We have proposed five types of forest degradation that represent a continuum of reduction in the supply of goods and services compared to pre-defined reference values. Recognizing that most ecosystem services can be measured provides a means to quantify the extent and progression of degradation (e.g., van Wilgen et al. 2001, TEEB 2009, Thompson et al. 2011).

Fig. 3. A schematic composite diagram proposed as a visual illustration of the multiple types of forest degradation. The interior solid red line represents scoring for each of the five criteria on scales for the indicators relevant to the forest type or region, as a percentage (dashed lines) of the desired or predicted baseline values. The diagram can be used for stand to landscape and sub-national level data and so is applicable for any scale of interest.
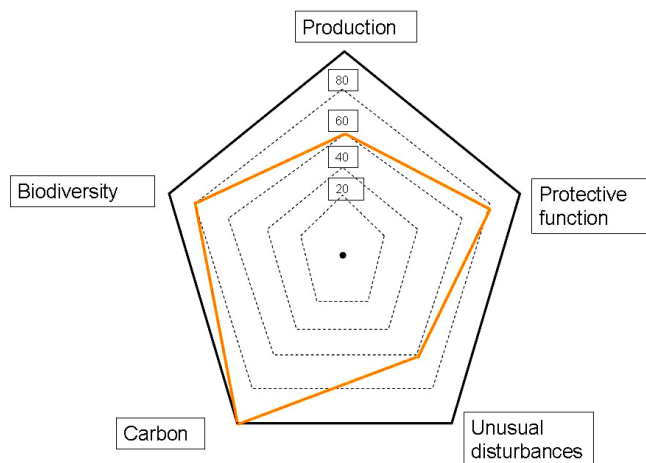
We suggest that criteria indicating changes in production, biodiversity, protective functions, unusual disturbances, and carbon storage will enable assessment of the amount and type of degradation (Fig. 3). The proposed criteria and indicators framework provides a means to distinguish among levels (for example, light to severe) of degradation and to reconcile among multiple perspectives. To use the criteria rigorously, agencies should decide a priori whether all or certain of the criteria are of interest. Most countries that currently report to the UN Food and Agriculture Organization on forests and to other intergovernmental forest-related organizations on forest processes (e.g., ITTO) already report on some of our proposed indicators. Nevertheless, all of the criteria should be applied to describe degradation where ecosystem sustainability is the forest objective.

Under the REDD mechanism of the UNFCCC, it will be necessary, in part, to identify a degraded forest to implement

some projects. Our approach provides a way to define forests that are degraded and can also enable specific focus, for recovery, on the key services that are most degraded. It is also important that reporting on degradation be understood in context with planned land management, where it exists. For example, much of the work on past selective logging in a land-use change context has been concerned with quantitative assessment using remote sensing techniques (e.g., Asner et al. 2006, Matricardi et al. 2010). Hence, such studies may not distinguish unsustainable logging from SFM guided by an approved plan (see, e.g., Foley et al. 2007 and commentary by Peña-Claros et al. 2007 and Putz et al. 2007). Distinguishing between forest degradation and well-managed regenerating forests is essential for reporting and project planning.

Challenges to implementation
Determining the range of natural variation for some of the indicators presents a major challenge. The task is easier, however, if unmanaged or sustainably managed forest is available as a benchmark, than for situations where general models must be used to establish baseline values (e.g., IPCC 2006). A further challenge for managers is to develop an effective monitoring program and maintain it over time to determine changes in the forests at large and/or local scales. Priority should be placed on improving skills and institutions, as these are ultimately the building blocks of effective forest management (Sheil et al. 2004, Nasi and Frost 2009).

It is important that the selected species or species groups do indeed indicate habitat quality and quantity (e.g., Niemi et al. 1997), and problems with obtaining statistically useful results are a concern (e.g., Rolstad et al. 2002). Indicator species analysis is a key way forward in the use of species indicators as analytical tools (e.g., Bakker 2008). Gardner et al. (2012) suggested a tiered approach to the use of indicators based on data quality and scale. This approach could be taken for monitoring degradation with progressive refinements to scale occurring over time as databases for indicators improve. Expert judgment remains the foundation of competent species selection, but improved data and expertise can inform this judgment.

Finally, once degraded forests have been delineated, recovering these areas to fully functioning forest ecosystems requires energy input and considerable planning (e.g., Mack et al. 2000), especially if the ecosystem has changed states. Forest managers and stakeholders may have to decide the level or amount of degradation that they are willing to accept across a landscape because trade-offs will often exist among management objectives (e.g., Lamb 1998, Thompson et al. 2011). Further, deciding how to allocate funds for forest recovery is not an easy task (Holl and Aide 2011). Once assessments have been accomplished to define degraded areas, their type, and level of degradation, considerable planning will be required to place areas into categories rated for relative ease and investment needed for recovery, with management

restoration plans devised and targets set against which to measure success in an adaptive manner (see, e.g., Lamb et al. 2005). Fortunately, comprehensive guidance for recovering or restoring degraded tropical forest areas can be found in ITTO (2002).

Responses to this article can be read online at: http://www.ecologyandsociety.org/issues/responses. php/5443

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