TECHNICAL REPORT: THE EFFECTS OF FOREST DEGRADATION ON ECOSYSTEM SERVICES

The Changing Wealth of Nations 2021

Managing Assets for the Future

WORLD BANK GROUP

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The Effects of Forest Degradation on Ecosystem Services

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Executive summary

Forests are an important asset in many countries both for the market goods they provide, like timber, and for the non-market goods and services they provide, like regulating hydrological flows. These benefits, whose average value is estimated at USD111/ha/year, are lost or greatly diminished when forests are lost (Siikamäki and others, 2015). Less attention has been given to date on forest degradation—on changes in forest condition that fall short of outright deforestation. Forest degradation also affects the degree to which forests provide ecosystem services. This report reviews what is known about the impacts of forest degradation on ecosystem services.

Forest degradation can take many forms. A typology of forest degradation includes three major types (Table ES1), each with two sub-types: ***fragmentation of forests*** (including isolation of forest patches and fragmentation of large, contiguous patches); ***impoverishment of species composition*** (including low diversity tree plantations and decline of fauna species); and ***loss of biomass and structure*** (including gradual, difficult-to-detect biomass loss (for example, selective logging, pathogen outbreaks, excessive human use) and abrupt, temporary biomass loss (for example, rotational logging, excessive fire).

Tropical dry forests and temperate broadleaf and mixed forests are the most degraded, in terms of fragmentation and human pressures, of the major forest biomes. Boreal forests are the least degraded.

Table ES1: Major types and causes of forest degradation

Type of forest degradation	*Major causes*
Fragmentation of forests:	
Isolation of patches and exposed edge	Agricultural expansion
Fragmentation and barriers in contiguous patches	Transport and logging infrastructure
Impoverishment of species composition:	
Simplification of flora community	Plantations, invasive species, small patches
Loss of fauna species	Hunting, poaching, bushmeat, loss of connectivity, loss of key flora
Loss of biomass and structure:	
Abrupt, temporary biomass loss	Clear-cut logging, excessive fire
Gradual biomass loss	Selective logging, pathogen outbreaks, excessive human use

Fragmentation of forests into isolated patches affects water services by altering the hydrologic properties of forests at the edges of patches. Direct evidence of the effects of forest edges on water yield and water quality are not available. However, a number of empirical fragmentation

experiments suggest that forest within 500m of the edge of a patch of humid tropical forests, and 250m of the edge of a patch of temperate forest, display reduced soil moisture, canopy cover, and other attributes that affect water balances. In humid tropical forests, which are not adapted to fire, edges are particularly susceptible to human-caused fire. In addition, logging roads and other infrastructure that expose bare soil potentially lead to sedimentation in water bodies from soil erosion, although best management practices can alleviate the impact (medium certainty).

Barriers and fragmentation in large forest patches can impede the movement of large, charismatic animal species relevant for wildlife tourism and recreation. Over time, the inability of these animals to disperse affects the viability of the populations with potentially negative consequences for revenue and jobs from wildlife tourism (medium certainty).

Biotic impoverishment from tree plantations can alter water services depending on water availability and tree species. Establishing plantations in water-limited regions, especially when plantations replace grasslands and shrublands, can decrease water yields and lead to increased water scarcity. In less water-limited regions, plantations are generally poorer at water flow regulation than forests, and are associated with lower base/dry season flow, and elevated flood risk. Plantations in both water-limited and non-water limited regions have less value than natural forests due to increased sedimentation and reduced ability to regulate nutrients. For all the above functions, there is considerable variation among different types of forests and plantations (medium certainty). The magnitude of effect of plantations on water services is highly variable depending on tree species, biome type, and water availability. Studies indicate 45 percent lower water yield from plantations that replace grasslands and shrub lands. Where plantations replace natural forests, studies indicate 50 percent reduction in flood regulation and 50-96 percent lower base flow (low certainty).

Biomass loss can have a substantial impact on water balance and water quality. Studies of rotational clear-cutting in temperate and boreal forests show substantial impacts on water balance and water quality where a large proportion of a watershed is cleared. Studies of selective logging in tropical forests show a similar relation between percent biomass loss and water yield. Sediment and nitrate concentrations are highly variable but generally increase when percent of watershed cut/biomass loss reaches high values. Fires in catchment areas can increase flash flooding and sediment in downstream water supplies depending on fire severity, soils, and hydrological events. While effects are forest-type and context-specific, logging and other human activities that reduce forest biomass generally increase potential for downstream flooding (high certainty) and increase nitrate and sediment concentrations when loss is severe (medium certainty). Abrupt degradation events, such as clear-cut logging and fire, tend to have more effect on water services than slow degradation events such as selective logging or pathogen outbreaks (medium certainty).

There is considerable variability in the impact of forest fires and logging on water yield and nitrate and sediment concentrations. Nitrate and sediment concentrations increase only when nearly the entire watershed is logged, but turbidity, total nitrate, and mean flow increase linearly with increasing burn severity, particularly in forests that experience infrequent fires. Loss of

biomass from pathogen outbreaks can increase nitrate concentrations, but no studies indicated concentrations above WHO standard (low certainty).

Forest degradation conceivably affects the availability of bushmeat for hunting, although degraded forests can support high populations of small mammals and rodents. Degraded riparian forest potentially affects fish populations. An additional effect of forest degradation is the potential reduction in availability of non-wood forest products important for livelihoods of people living in and around forests. These effects on local populations are difficult to quantify at a national level to include in the CWON but are highly relevant for marginalized rural populations throughout the tropics (low certainty).

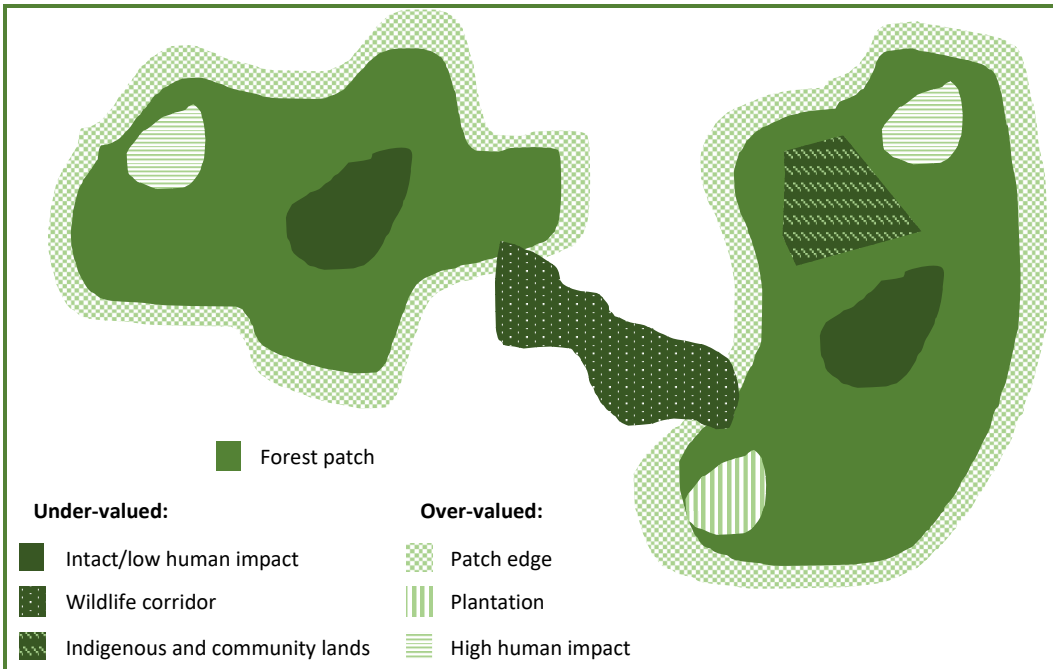
The robustness of the evidence to quantify the effects of forest degradation on the three **ecosystem services considered here varies across the types of degradation and the effects.** Table ES2 summarizes our assessment of the evidence to potentially quantify the effects.

Table ES2: Assessment of robustness of effects of forest degradation on ecosystem services

(--) Relatively strong evidence with high relevance, negative effect on ecosystem services
(-) Medium evidence
(?) Inconclusive evidence

	Water services		*Recreation, hunting, and fishing*	*Non-wood forest products*
Type of degradation	*Water balance*	*Water quality*		
Fragmentation:				
Isolation of patches and exposed edge	Edge effects (- -)			Habitat to maintain NWFP tree species (-)
Fragmentation and barriers in contiguous patches		Sediment (-)	Wildlife tourism (-)	
Impoverishment of species composition:				
Tree plantations	Increase runoff (-?)			Tree species for NWFP (?)
Loss of fauna species			Bushmeat availability (?)	
Biomass loss:				
Abrupt biomass loss (Clear-cut logging, fires)	Increase runoff (- -)	Nitrate and sediment runoff with high biomass loss (-)	Fish availability (?)	Availability of NWFPs for indigenous and local communities (-)
Gradual biomass loss (selective logging, pathogen outbreaks, unsustainable use)	Increase runoff only if high biomass loss (-)	Nitrate and sediment runoff with high biomass loss (?)		Availability of NWFPs for indigenous and local communities (?)

Estimates of the value of forests that do not take into consideration the condition of the forest may be incorrect. Although it isn't possible at present to arrive at estimates of forest that incorporate the effects of degradation, in some cases it is possible to say whether estimates that do not consider forest condition are likely to be under- or over-estimated. These effects are summarized in Figure ES1.



Schematic of approach to downscale (light green) or upscale (dark green) value of forest to account for degradation effects on the ecosystem services considered in this report.

Figure ES1: Possible biases in valuation that doesn't consider forest condition

Acknowledgments

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The cover photo is by Ruth DeFries.5.5

1. Introduction

Forests provide many ecosystem services, including provisioning services (for example, timber and fuel, fodder, and non-timber forest products to support local livelihoods), regulating services (for example, protection of watershed protection, climate regulation, and habitats for pollinators), and cultural services (for example, recreation and spiritual value).

As forest condition deteriorates, the amount of these services they are able to provide is likely to diminish overall. Attention so far has focused primarily on the effects of deforestation on ecosystem services, particularly the effects of deforestation on climate regulation through loss of carbon storage and water cycling. However, deterioration of forest condition that falls short of complete forest loss can also compromise the ability of forests to deliver ecosystem services. This report reviews the available evidence on how forest degradation affects the provision of key ecosystem services.

2. Principal forms of forest degradation

Definition of forest degradation. A clear definition of forest degradation has been elusive. Varying definitions relate to the ability of forests to regenerate (Ghazoul et al., 2015), to the potential to recover from disturbance (Trumbore et al., 2015), and to multiple criteria of productivity, biodiversity, unusual disturbances, protective functions (for example, soil erosion), and carbon storage (Thompson et al., 2013). The most straightforward indicators of degradation are fragmentation and disturbances such as fire, while other aspects of degradation such as invasive species are more difficult to measure (FAO and UNEP, 2020).

For the purpose of this report, we use the definition of forest degradation from the Food and Agriculture Organization (Food and Agriculture Organization, 2011): "reduction of the capacity of a forest to provide goods and services." Because a historical baseline is often not possible, we suggest that degradation be measured relative to a sustainably managed forest in the same ecological and socioeconomic conditions, which may or may not reflect a historical baseline.

Extent of forest degradation. Estimates of forest degradation in 1990 vary from 532 million hectares (ha), or 29 percent of total tropical forest area, to 850 million ha, or 60 percent of total tropical forest area, of which 350 million ha are so severely damaged that forests won't grow back spontaneously (ITTO, 2002). Degraded forests occupy larger areas than primary forests across over 80 percent of all tropical nations (Malhi et al., 2014).

Forms of forest degradation. Forest degradation can take many forms. Among these, the most important forms that affect a substantial portion of forests globally; are caused by direct and local human activities rather than long-term ecological processes (for example, succession) or global-scale impacts (for example, climate change and nitrogen deposition); and are linked to the ability of forests to deliver ecosystem services are:

- **Fragmentation of forests** — Fragmentation occurs as forest patches become surrounded by an alternative land use or as logging roads and other infrastructure bisect contiguous patches of forest, leaving isolated fragments that differ markedly in biodiversity and function from contiguous forest.
- **Impoverishment of species composition** — Forests degraded by anthropogenic activity often experience biodiversity declines and shifts in species composition. This impoverishment, in turn, poses a pervasive threat to floral and faunal diversity that may be of a comparable magnitude to that posed by deforestation.
- **Loss of biomass and structure** — Extractive use of forests thins tree cover and reduces biomass without complete removal of the forest. Changes in structure, for example from even-aged stands and removal of understory, can also affect ecosystem services such as carbon storage and nutrient and water cycling.

Box 2.1: Tree plantations

Tree plantations pose a particular set of considerations. Plantations are difficult to distinguish from natural forests with standard methods in remote sensing for mapping tree cover. Plantations vary widely and can include monocultures of non-native species for commercial harvesting to mixed species. Ecosystem services from plantations can differ from natural forests (see section 2.3.3). Consequently, replacing a natural forest with a plantation might constitute degradation in some circumstances. Likewise, plantations on lands that were not previously forest can constitute a forest that is 'degraded' compared to natural forest. Conversely, plantations on degraded land in forest biomes could represent an improvement, albeit one that falls short of that which would have been provided by regenerated natural forests.

Causes of forest degradation. Many human activities affect the condition of forests, which in turn alter the ability of forests to provide ecosystem services (Lewis et al., 2015). In particular, rapidly expanding transport infrastructure, logging and associated infrastructure, human-ignited fires, collection of biomass for fuelwood and charcoal, and hunting and poaching are degrading forests throughout the world.

Objective. The aim of this report is to assess how these processes are altering the ability of forests to deliver ecosystem services. We focus on the ecosystem services considered in the World Bank's *Changing Wealth of Nations* report (Lange et al., 2018): watershed protection; recreation, hunting, and fishing; and non-wood forest products.

3. Measuring the extent and rate of forest degradation

Unlike deforestation, which is the conversion of forest to a non-forest land use and is straightforward to identify through satellite data, forest degradation is more difficult to quantify for several reasons. First, drivers of forest degradation vary across regions, socio-economic settings, and forest types. Degradation drivers can include fragmentation in humid tropical forests, intensified fires regimes in boreal or dry tropical forests, pathogen outbreaks, local over-harvesting of biomass where livelihoods rely on non-timber forest products, and depletion of fauna where people hunt bush meat. Each of these processes leaves a distinct pattern. A single approach to identify degradation cannot be applied uniformly across all forests. Second, the fine spatial scale of degradation indicators such as logging decks and exposed soil make identification with satellite data problematic. Fine-scale drivers such as understory grazing, fuelwood and charcoal extraction, and hunting defy detection with standard methods in remote sensing (Mitchell et al., 2017). Third, forests are dynamic as their composition, structure, and biomass fluctuate in response to climate variability and successional processes. Historical baselines do not generally provide a fixed, static benchmark against which to assess current condition.

Three main sources of information provide the basis for assessing rates and extent of the three types of forest degradation: remote sensing, threat indices, and self-reporting by countries.

3.1 Remote sensing and other spatially-explicit products

Remote sensing with satellite data has become routinely used over the last few decades to map the extent and trends of many variables relevant for assessing degradation, including global tree cover and change since 2000 (Hansen et al., 2013), pan-tropical above-ground biomass (Avitabile et al., 2016), and change from 2003-2014 (Baccini et al., 2017), active fire (Giglio et al., 2016), and burn scars (Chuvieco et al., 2019). Many of these data sets are available on public platforms, including Global Forest Watch and UN Biodiversity Lab. Remote sensing of fine-scale, gradual forms of degradation such as fuelwood extraction and changes in species composition has been limited. Compilations of multiple sources (Harris et al., 2018) and hyperspectral sensors (Fagan et al., 2015; Fagan et al., 2018) are improving the ability to distinguish plantations and tree crops from natural forest. Emerging remote sensing products from new sensors on-board satellites, including LiDAR, radar, and hyperspectral sensors, will improve future abilities in the coming years to observe degradation processes and changes in forest structure that have been problematic for standard optical sensors to map over large areas.

Other spatially-explicit, global data sets provide proxy information about the rates and extent of forest degradation, including road density (Meijer et al., 2018), gridded human population density (Doxsey-Whitfield et al., 2015), and gridded livestock density (Robinson et al., 2014). Additional data products and platforms are becoming available, such as proximity to roads and other infrastructure. At more local scales, patterns from logging decks and skid rows indicate the presence of logging (Grecchi et al., 2017; Souza Jr et al., 2013). As forest degradation results from a combination of all of these drivers, other data sets combine these individual attributes into composite indices.

Spatially-explicit data sets on indigenous and community lands indicate where forests support local livelihoods such as non-wood forest products (Garnett et al., 2018). An overlay with forest area would indicate the forest areas used by these communities.

3.2 Composite indices of threats as proxy of forest condition

If degradation cannot be measured directly, spatial data sets on threats to forests can serve as indirect proxies for forest condition. These data sets, particularly when combined with satellite-derived direct measures of forest condition, are potentially useful for consistently comparing rates and extent of pressures on forest across countries and through time. These composite data sets include:

Human footprint and other threat maps. The human footprint is a composite, global index of the terrestrial environment at 1km^2 spatial resolution from 1993 to 2009. It maps the cumulative pressure from eight human variables: extent of built environments, crop land, pasture land, human population density, night-time lights, railways, roads, and navigable waterways. The average human footprint score across the world's non-Antarctic land area was 6.15 out of a maximum of 50, with considerable geographic variation. The average value increased by 9 percent from 1993 (Venter et al., 2016).

Similar, more recently derived composites to indicate human pressure include the map of Low Impact Areas, which combines livestock density, forest change, land cover, and nighttime lights at 1 km^2 resolution to indicate locations which have low human impact (Jacobson et al., 2019), and Human Modification Index, which combines human settlement, agriculture, transportation, mining, energy production, and electrical infrastructure to indicate the proportion of a landscape modified by humans at 1 km^2 resolution (Kennedy et al., 2019).

Forest Health Index. The index scores the health of forests ranging from 0 (lowest) to 10 (highest) globally in 2019 at 300m resolution. The index combines spatially explicit datasets on: forest extent, direct pressure from high impact, localized activities (infrastructure, agriculture, and recent deforestation); indirect pressure associated with edge effects and other activities such as hunting and selective logging; and anthropogenic changes in forest connectivity. The analysis finds that 40.5 percent of forests globally can be considered to be in high health. Healthy forests are mostly found in Canada, Russia, the Amazon, Central Africa, and New Guinea (Grantham et al., 2020).

Intact forest landscapes. This global dataset maps seamless mosaics of forest and naturally treeless ecosystems with no remotely-sensed detected signs of human activity and a minimum area of 500 km^2. These large remaining patches of forest provide critical conservation value for wide-ranging species and resilience to disturbance. Intact forest landscapes comprise 20 percent of tropical forest area and declined by 7.2 percent between 2000 and 2013 (Potapov et al., 2017).

Biodiversity intactness index. The index estimates the average proportion of natural biodiversity remaining in terrestrial ecosystems globally at 1 km^2 resolution. Modeled estimates are based on a database of the occurrence or abundance of 39,123 species predicted from land use, land use intensity, human population density, and distance to roads. The index is the average abundance

of present species across a broad range of species relative to abundance in undisturbed habitat (Newbold et al., 2016).

Forest integrity indices. These indices of forest integrity cover humid tropical forests. The Forest Structural Condition Index combines remotely-sensed estimates of canopy height, tree cover, and time since disturbance to distinguish short, open-canopy, or recently disturbed stands from tall, closed canopy stands typical of primary or late secondary forest. The Forest Integrity Index overlays the human footprint index on the structural condition index to identify structurally complex forests with low human pressure. These locations are the most valuable for biodiversity and ecosystem functions and represent the "best of the last" (Hansen et al., in press).

Defaunation index. The index estimates mammal defaunation across tropical forests based on models of observed abundances and socioeconomic drivers of hunting pressure (distance to hunters; access points, accessibility of urban markets, human population density, poverty levels, and access to domestic meat). The index ranges from 0 (intact mammal assemblage) to 1 (fully defaunated mammal assemblage) (Benítez-López et al., 2019).

These data sets provide a useful global perspective but do not capture many regionally or locally important differences. For example, fire detected in humid tropical forests where fire does not naturally occur indicates a human threat and potential degradation, while fires in dry forests occur naturally and do not necessarily indicate a threat. The indirect proxies also do not represent varying effectiveness of governance in different places and across time, such as enforcement against illegal logging.

Table A1.2 in Appendix 1 summarizes the spatial data sets that could be useful for valuation of the impact of forest degradation on ecosystem services, either as proxy indicators or direct measures of degradation. As an example of the utility of these data sets, Appendix 1.2 applies the human modification index (Kennedy et al., 2019) to obtain the average value for the forest remaining within forest ecoregions (Olson et al., 2001), with remaining forest identified from a land cover classification (Bontemps et al., 2013). Boreal forest contains 48 percent of the world's remaining forest with an average human modification index of 0.03 (on a scale of 0 to 1 for low to high modification), compared with temperate broadleaf and mixed forest containing 14 percent of remaining forest with an average index of 0.24 and humid tropical forest containing 26 percent of remaining forest with average index of 0.13.

3.3 Self-reporting by countries

Remotely-sensed and spatially-explicit data provide globally consistent and repeatable information on forest condition. However, these sources have several drawbacks: they do not provide information on degradation that is unobservable from satellites, and they do not indicate the drivers of degradation. The biennial Food and Agriculture Organisation (FAO) report includes self-reported information (based on varying sources including country-level analyses of satellite data and field observations) from countries to provide some of this information at an aggregated country level. Countries report area under planted forest, wood fuel removals, and area affected by woody invasive species which are difficult to observe with remote sensing (Food and Agriculture Organization, 2016a, b).

Country readiness plans for Reducing Emissions from Deforestation and Degradation (REDD+) include reports of drivers of degradation such as livestock grazing in forest, uncontrolled fires, fuelwood/charcoal collection, and timber/logging. CIFOR country reports also assess drivers of degradation. A compilation of these country reports across 46 countries indicates that timber extraction and logging are the main drivers for most of the degradation with variations across continents, while conversion for agriculture is the main driver for deforestation (Hosonuma et al., 2012).

Country reports have the limitation of inconsistent reporting protocols across countries. Moreover, the single assessment for an entire country precludes the reality that different drivers and levels of degradation are likely to occur in different forest types and regions in a country, particularly for large countries.

4. Extent, rates, and drivers of forest degradation

Table 4.1 summarizes the major drivers for each of the three types of degradation. The extent, rates, and drivers of degradation are heterogeneous across the world's forests. For example, fuelwood collection and forest grazing are major drivers of degradation in tropical dry forests with high human densities of forest-dependent populations, while logging is more likely to be the main driver in humid tropical forests with lower densities of human populations and more valuable tree species.

Table 4.1: Major types and causes of forest degradation

Type of forest degradation	*Major causes*	*Possible metrics*
Fragmentation of forests:		
Isolation of patches and exposed edge	Agricultural expansion	% of forest in different size patches; % of forest within distance of patch edge
Fragmentation and barriers in contiguous patches	Transport and logging infrastructure	Intact forest index, road density, human footprint index
Impoverishment of species composition:		
Simplification of flora community	Plantations, invasive species, small patches	Planted forests, forest integrity index, biodiversity intactness index
Loss of fauna species	Hunting, poaching, bushmeat, loss of connectivity, loss of key flora	Human footprint index, defaunation index
Loss of biomass and structure:		
Abrupt, temporary biomass loss	Clear-cut logging, excessive fire	Human footprint index, forest integrity index, logging roads, fire and burn scars, biomass change
Gradual biomass loss	Selective logging, pathogen outbreaks, excessive human use	Human footprint index, logging roads, biomass change

4.1 Fragmentation of forests

Effects of fragmentation. The size, shape, and spatial configuration of forest fragments in a landscape influence many aspects of a forest, including the assemblages of flora and fauna, persistence of wildlife that require large areas susceptibility to fire, and microclimates. Fragmentation occurs as patches become surrounded by an alternative land use or as logging roads and other infrastructure bisect contiguous patches of forest (Figure 4.1).

Fragmentation from agricultural expansion leaves isolated forest fragments that differ markedly in biodiversity and function from contiguous forest. Isolated patches support less biodiversity and different assemblages of flora and fauna than larger patches. Edges are exposed to invasive

species, fires, and drier microclimates, which cause further degradation (Haddad et al., 2015). For large-ranging species, loss of connectivity between forest patches can lead to local extinctions and loss of genetic viability of the population (Resasco, 2019).

A particularly relevant impact of forest fragmentation is the risk of fire. The combination of desiccation from decreased moisture and exposure to escaped fires initiated outside forests makes edges vulnerable. Increased fire frequency in forest edges relative to core forest is a greater concern in humid tropical forests than in temperate forests. Unlike drier temperate or boreal forests, natural fires are rare in humid tropical forests and some wet temperate forests where trees are ill-adapted to burning. Trees in the humid tropics generally are less resistant to fire damage than dry forest temperate trees due to thin bark, higher mortality, and inability of seeds to regenerate (Cochrane, 2003; Staver et al., 2019).

Drivers and patterns of fragmentation. Within contiguous forest, fragmentation occurs with expansion of road networks, power lines, logging skid rows, and other linear infrastructure. Infrastructure expansion opens up the forest to cascading human impacts including fire and deforestation. Highest road densities are in Northwest Europe and parts of South and East Asia and far less dense in northern parts of Canada and the Russian Federation and large parts of the Amazon forest (Meijer et al., 2018). The rapid expansion of the number of roads and other infrastructure is creating a major wave of fragmentation throughout Asia, Africa, and Latin America (Laurance et al., 2014).

Approximately 10 percent of the Brazilian Amazon is within 500m of an edge and 74 percent of forest in England is within 100m of forest edge. The largest contiguous patches of forest are in the humid tropical regions of the Amazon and Congo River Basins, while globally more than 70 percent of the world's forest are within 1 km of a forest edge and 20 percent within 100m (Haddad et al., 2015; Riutta et al., 2014). Boreal forest has the highest proportion of forest area in intact forest and the least loss in proportion of intact forest between 2000 and 2013 (Potapov et al., 2017). Patch size distributions indicate that tropical dry forests and temperate broadleaf forests are the most highly fragmented forest biomes with the lowest average distance to patch edge (1.6 km and 1.4 km respectively compared to 5.6 in boreal forests) (Jacobson et al., 2019).

Atlantic Forest Brazil

Fragmentation of contiguous forest with logging roads in the Peruvian Amazon

Credits: (L) https://desdemonadespair.net/2011/09/brazils-atlantic-forest-stores-less.html
(R) https://infoamazonia.org/en/2015/12/proliferation-of-logging-roads-in-the-peruvian-amazon/#!/story=post-14247

Figure 4.1: Examples of forest patches from fragmentation

Infrastructure associated with industrial timber extraction was the primary cause of loss of intact forests globally, with clear cuts causing forest alteration in temperate and boreal regions of North America and Eurasia. Based on country reports, timber extraction and logging account for more than 70 percent of total degradation in Latin America and Asia (Hosonuma et al., 2012).

4.2 Impoverishment of species composition

Effects of species impoverishment. Forests degraded by anthropogenic activity often experience biodiversity declines and shifts in species composition, particularly in cases where biodiversity rich natural forests are degraded through conversion to relatively biodiversity poor or monoculture tree plantations (Barnes et al., 2017; Gibson et al., 2011). Forest degradation poses a pervasive threat to floral and faunal diversity that may be of a comparable magnitude to that posed by deforestation (Barlow et al., 2016) (Figure 4.2).

Species loss and changes in the composition of animal communities have feedbacks that affect floristic composition, with further cascading impacts on forest ecosystem functions and services (Dirzo et al., 2014). In tropical forests, for example, large frugivores and herbivores such as hornbills, primates, elephants, and tapirs strongly influence tree seed dispersal and regeneration patterns (Berzaghi et al., 2019). Local extirpations or declines in abundances of these large animals due to hunting and forest fragmentation can reduce regeneration of forest tree species (Gardner et al., 2019b), exacerbating forest degradation and resulting in losses of key ecosystem services such as carbon storage over time (Berzaghi et al., 2019). Simulation models suggest that such carbon losses would be greater (up to 12 percent) in forests composed mainly of animal-dispersed species (for example, tropical forests of Africa, the Americas and South Asia) than in forests composed of tree species having other mechanisms for seed dispersal (Osuri et al., 2016).

Impacts on ecosystem services can also extend beyond forest boundaries, by affecting crop pollination and pest control in surrounding agricultural landscapes. The majority of food crops globally depend not just on honeybees, but also on wild bees and other wild insects for

pollination (Garibaldi et al., 2013; Rader et al., 2016). A study in a coffee-growing landscape of Costa Rica suggests that wild bees originating from forest fragments can enhance crop yields and quality in adjacent surrounding areas to the order of USD50 per ha per year (Ricketts et al., 2004). However, reductions in fragment area and resultant declines in pollinator diversity and abundance could result in the loss of such pollination ecosystem services (Krishnan et al., 2012). Similarly, while insectivorous birds in agricultural areas could benefit crop production by controlling pests—estimated at USD310 per ha per year in West Indian coffee farms (Johnson et al., 2010)—insectivores have been shown to be among the more sensitive of bird groups to disturbances such as forest fragmentation (Şekercioğlu et al., 2002).

Drivers and patterns of species impoverishment. Degraded natural forests—including those that have been subject to selective logging, and secondary forests growing in former agricultural or pasture lands—have largely replaced structurally complex natural forests in most regions. Perhaps the most extreme form of species impoverishment from forest degradation is conversion of high biodiversity forests to species-poor or monoculture plantations. Between 1990 and 2015, the extent of planted forests (tree crops and timber plantations and trees planted for restoration) increased from 168 million ha to 278 million ha (+65 percent), even as global forest extent (natural forests plus plantations) fell by 290 million ha over the same period (Payn et al., 2015). In some regions (for example, India), plantation activity has led to overall increases in estimated forest cover, in spite of declines in the extent of natural forests due to deforestation (Puyravaud et al., 2010). Elsewhere, plantations are increasing forest cover simply through expansion into non-forested natural ecosystems such as grasslands and savannas (Bond et al., 2019). The increase in plantations is driven by global demand for timber, pulp, and tree crops such as rubber, and the growing popularity of tree plantations as a tool for afforestation and terrestrial carbon sequestration within national and international climate policies (Lewis et al., 2019). Advances in satellite technology have improved the potential for tracking changes in plantation extent, and for distinguishing plantations from high biodiversity natural forests using satellite imagery (Harris et al., 2018). Failure to do so in regional or national assessment of forest cover can significantly underestimate rates of natural forest loss and overestimate biodiversity and ecosystem services (Hua et al., 2016; Puyravaud et al., 2010).

Degradation of natural forests impacts species diversity to varying levels across different biodiversity groups. For example, an increase in removal of logs in tropical forests from 38 m^3/ha to 63 m^3/ha can drive 50 percent declines in overall mammal and amphibian diversity, whereas bird communities may be more resilient, with forest-dependent but not open-country species responding negatively to increased disturbance (Burivalova et al., 2014). Similarly, young secondary forests in the Amazon retain less than 40 percent of the bird and tree species occurring in primary forests, but the representation of primary forest species in secondary forests exceeds 80 percent for large mammals and certain groups of insects (Barlow et al., 2007). Among plants, degradation generally favors a few 'winners', which are typically light-loving non-forest species, while several shade-tolerant forest species decline in degraded sites (Clark and Covey, 2012; Tabarelli et al., 2012).

The biological diversity and species composition of degraded natural forests are also likely to vary with time since disturbance, with older forests (>20-30 years) more closely resembling primary forests than younger forests for a range of biodiversity indicators (Letcher and Chazdon, 2009).

However, the potential for such recovery may be constrained in heavily fragmented and human-dominated landscapes, where processes of natural recovery and succession may be limited by proximity to inhospitable habitat edges and chronic anthropogenic resource extraction (Tabarelli et al., 2008).

The direct harvest of wildlife (legal and illegal) for meat, wildlife products, and pets resulting in the depletion and degradation of faunal communities—often termed 'defaunation'—affects virtually all forests (Young et al., 2016). Wildlife trade targets around 18 percent of all mammal, bird, amphibian, and reptile species. While the trade affects both tropical and temperate species, hotspots of trade, in terms of the numbers of species targeted, invariably lie in the tropics (Scheffers et al., 2019).

Bushmeat hunting for subsistence use and commercial markets is considerably more prevalent in tropical than in temperate forests (Ripple et al., 2016). One reason for this is greater dependence on wildlife for meat in these forests, given that local communities in many tropical regions do not keep livestock (Fa et al., 2005; Golden, 2009). Bushmeat hunting generally has stronger negative impacts on large bodied mammal and bird species (Benítez-López et al., 2017; Benítez-López et al., 2019), although hunters are known to switch to targeting smaller bodied species once large game become scarce (Bennett et al., 2002). While estimating the extent and severity of hunting induced defaunation over large areas remains a challenge, recent studies have modeled and mapped defaunation based on correlates of hunting such as proximity to access points, human population density and socio-economic status, and the presence of livestock.

Photo by R. DeFries

Figure 4.2: Monoculture eucalyptus plantation

4.3 Loss of biomass and structure

Depleted biomass and reduced canopy cover potentially reduce water cycling to the atmosphere through roots and leaves and reduces groundwater recharge through infiltration into the soil, while compacted exposed soil increases runoff and downstream flooding. Bare soil also creates opportunities for invasive species. In addition, loss of biomass compromises the ability for local forest-dependent communities to obtain livelihood needs from the forests by reducing the availability of forest resources and/or increasing collection time. The value of non-timber wood products is difficult to quantify because these products often do not enter the commercial marketplace, although they are essential to generate income for millions of people living on forest fringes (Food and Agriculture Organization, 2016b).

Pathogen outbreaks. Tree pathogen outbreaks can be part of the natural disturbance dynamic in many forest ecosystems, but in recent decades pathogen outbreaks are increasing in frequency and in diversity of pathogens. This is primarily due to increased connectivity and movement of pathogens between continents, which also favors hybridization and creation of novel strains. Some cases of pathogen-related mortality may be intensified by increased stress put on trees by climate change. Pathogen-associated degradation most affects low-diversity forests such as temperate and boreal forests, where the density of key species is high enough to facilitate rapid pathogen spread and where the loss of a single abundant tree species can have a major impact on forest structure and ecosystem services.

Drivers and patterns of biomass loss. Forest degradation can result in a loss of biomass either through slow, gradual processes, such as selective logging and pathogens, or through abrupt losses, such as fires and clear-cut logging. Selective logging is the most widespread human disturbance in tropical forests and produces approximately one-eighth of global timber. Depending on the intensity, selective logging can reduce biomass and biodiversity (Martin et al., 2015). Where high population densities depend on forests for fuelwood, fodder, and other livelihood needs, such as in dry tropical forests of South Asia, unsustainable harvesting can alter forest structure and reduce overall biomass (Agarwala et al., 2016). Urban demand for charcoal also drives biomass loss, particularly in forests that supply charcoal of cities in Sub-Saharan Africa (Bervoets et al., 2017).

Indicators. The impact of slow and small-scale degradation on biomass is unlikely to be accurately quantified in remotely-sensed global estimates of biomass, for example, (Baccini et al., 2017), due to the low signal (true biomass loss) to noise (range in biomass estimates) ratio. We suggest that proxy variables, such as human population densities and dependence on forests for fuelwood, non-timber forest products, and other livelihood needs, might be used to assess the extent of this type of degradation. These dependencies are likely to be particularly strong where indigenous and local communities rely on forest products. According to country reports, fuelwood and charcoal extraction is the leading cause of degradation in Africa but is less important in the Americas. Timber and logging are leading causes of degradation in Asia and the Americas, with forest grazing reported as having a lesser role in all three continents (Hosonuma et al., 2012). The Human Footprint and other composite threat indicators may serve as a reasonable indicator of gradual, small-scale forest degradation.

Fires. On a more rapid time scale, excessive fires lead to loss of biomass and can inhibit regeneration if too frequent. Fire is a very commonly-used tool for land clearing and agricultural management of crop residue and pests, particularly in the developing world. When fires escape into surrounding forests, they can spread in the understory and damage the forest. Fires occur very infrequently in humid tropics through lightning strikes because moisture inhibits ignition. Essentially all forest fires in the humid tropics are human-caused, either directly or indirectly through escaped fires. In other forest types, such as Mediterranean, temperate, boreal and tropical dry forests, lightning naturally causes fires. In such forests, the ecosystem is adapted to fire through traits such as fire-resistant bark and ability to re-sprout after fire, and tree species even depend on fire for regeneration of seeds. In non-fire adapted systems such as humid tropical forests, the incursion of even mild fires can cause large-scale tree mortality and forest degradation (Barlow and Peres, 2008; Brando et al., 2014). In fire-adapted forest systems, frequent, small fires reduce fuel load to effectively tamper the intensity and duration of fires when they do occur. However, excessive human-driven increases in fire intensity or frequency in fire-adapted systems are a major form of degradation. They lead to reduced biomass, destruction of seed banks for regeneration, and exposed soil and erosion from burn scars, along with downwind effects on air quality from smoke emissions.

Degraded dry tropical forest in central India

Burn scar in Mato Grosso, Brazil

Photos by S. Khanwilkar (left) and R. DeFries (right)

Figure 4.3: Loss of biomass and structure

Global trends indicate that on net, fires and burn scars reduced by one quarter between 1998 and 2015. However, the reduction has mainly occurred in savannahs and grasslands, while closed-canopy forests have experienced an increasing trend (Andela et al., 2017). Climate change increases the risk of fires, as seen in boreal forests (Astrup et al., 2018), because of increased frequency and intensity of heatwaves and drought and because long-term global warming is leading to increases in atmospheric water vapor deficit, enhancing vegetation water stress and fuel flammability year-on-year (Rifai et al., 2019). Structural degradation through logging also changes forest microclimate and increases flammability and the risk of fire propagation.

4.4 Interactions between degradation types

The three major types of degradation are linked through both their causes and their impacts on ecosystem services. Human presence, as indicated through roads and human and livestock population densities, is the overarching driver of all of the types of degradation. We suggest that an indicator of human pressure sufficiently captures the likelihood of all three types of degradation. Added metrics to measure a few additional attributes—for example, patch distributions to assess fragmentation, identification of "the best of the last" and forests valuable for their intactness, excessive fires and burn scars, and biodiversity intactness provide a basis for valuation of forest condition. Despite the large number of possible data sets and metrics, a handful of metrics will likely suffice to characterize the multiple types of degradation.

The types of degradation overlap in complex ways. Isolation of forest patches makes forests vulnerable to fires and invasive species, which contributes to impoverishment of species composition and loss of biomass. Hunting and roads open frontier forests to further degradation. All the types of degradation compromise the delivery of protective ecosystem services (for example, watershed protection and resistance to excessive fires and invasive species), the ability of forests to provide non-wood timber products in the future, and possibly recreation value.

5. Effects of forest degradation on ecosystem service provision

Forest degradation can inhibit the ability of forests to deliver services while healthy, intact forests can enhance services. This report develops, as far as possible based on available information, relationships between forest condition and three main ecosystem services: water services; recreation, hunting, and fishing; and non-wood forest products (referred to as response functions). These are the three ecosystem services considered in the World Bank's *Changing Wealth of Nations* report (Lange and others, 2018). In cases for which quantitative relationships are not available, we provide qualitative assessments and examples. Our aim is to provide background on the biophysical relationships between forest condition and ecosystem services. The next step would be to incorporate these relationships in the valuation of forests, but we do not attempt to provide that valuation in this report.

Although we restrict this report to three main ecosystem services considered in the *Changing Wealth of Nations* report, many other services are also highly relevant to society's benefits from forests, for example, biodiversity, carbon sequestration, microclimate, pollination, and coastal protection.

Table 5.1: Primary effects of forest degradation on provision of ecosystem services

Type of degradation	*Watershed services*		*Recreation, hunting, and fishing*	*Non-wood forest products*
	Water balance	*Water quality*		
Fragmentation	Isolation of patches and exposed edge	Soil erosion from roads and infrastructure	Connectivity effects on wildlife tourism	
Species impoverishment	Increased runoff from tree plantations	Increased erosion and reduced nutrient regulation (with exceptions) in plantations	Loss of bushmeat protein source from defaunation	Loss of tree species from plantations
Loss of biomass	Increased runoff and peak flow from logging, fires, and pathogen outbreaks	Nitrate and sediment concentrations from logging, fires, and pathogen outbreaks	Loss of fish protein source from degraded riparian forest	Local loss of access in logged areas and burn scars, key species lost to pathogens or over-harvesting

Table 5.1 summarizes our assessment of the primary effects of forest degradation on the provision of the three main ecosystem services. For some of these effects, the literature enables us to develop relationships between a measurable type of degradation (for example, patch size or extent of tree plantation), and delivery of the ecosystem service (for example, water yield or sediment load). For other effects, data limitations allow us only to provide examples or qualitative relationships. To the extent possible, we identify the varying effects of forest

degradation on ecosystem services across biophysical and cultural contexts. For example, the effects of tree plantations on water yield are most relevant in water-scarce locations and the effects of forest degradation on non-wood forest products are most relevant where local populations depend on forest resources for their livelihoods.

The following sections address each of the effects in Table 2.1, followed by possible methods to include these effects in valuation efforts such as that of the World Bank's *Changing Wealth of Nations* report.

5.1 Impacts of forest degradation on water services

Overview of water services from forests

Forest degradation potentially alters both water quantity and water quality for downstream water users in a watershed. Reduced canopy cover, fragmentation, and logging rotations can alter many aspects of the water balance, including evapotranspiration, runoff, and infiltration. Reduced infiltration diverts a smaller fraction of precipitation into sub-surface flow, so that more water runs off and consequently can increase downstream flooding and reduce levels of streamflow during dry periods. Degradation can also alter microclimates and make forests susceptible to fires that might not otherwise occur in a healthy forest.

The effect of forests on streamflow depend on the net balance between infiltration and evapotranspiration (Figure 5.1). Streamflow is more sensitive to changes in forest cover in drier than in wetter regions suggesting that the effects of forest degradation more strongly influence water services where water is limited (Figure 5.2) (Zhang et al., 2017).

Less degraded	More degraded

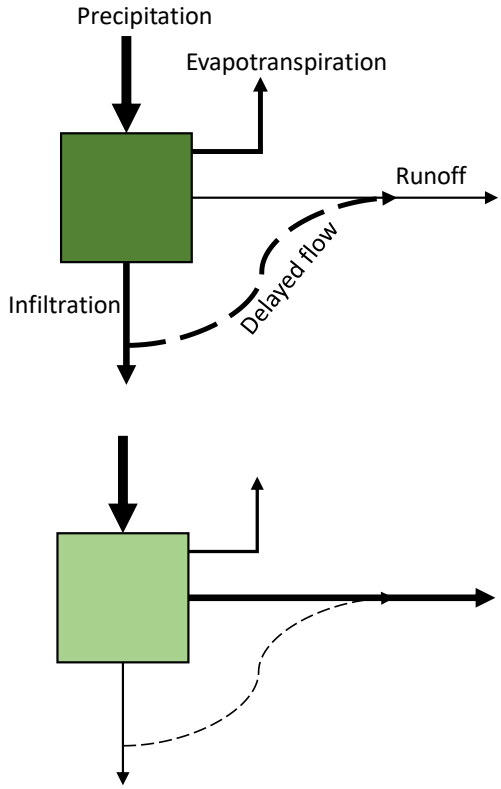


Figure 5.1: Components of water balance

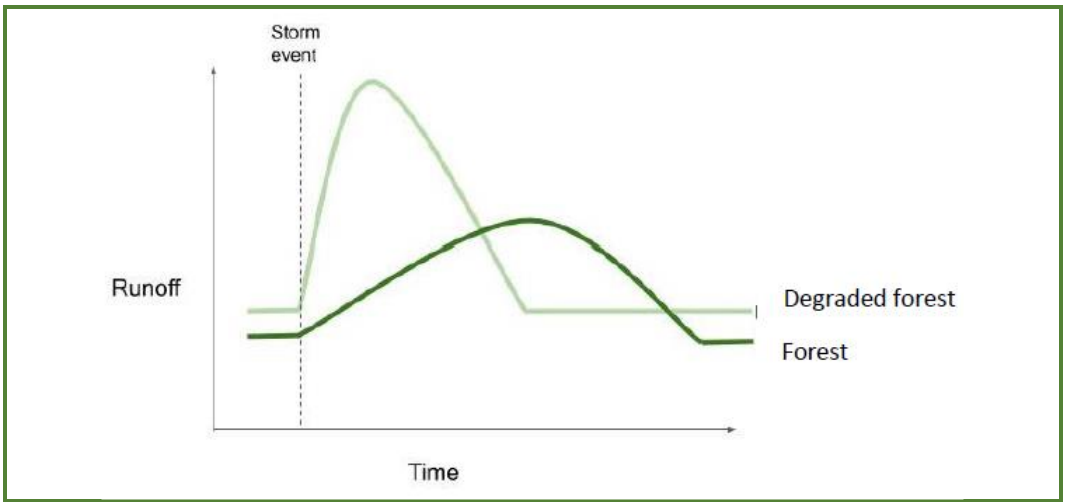


Figure 5.2: Theoretical effects of degraded forest on flood protection and runoff

Forests also provide an important service by reducing sediment loads and filtering contaminants, nutrients (nitrogen and phosphorus), and pathogens as water infiltrates slowly through the soil and into streams. Many cities that rely on surface water depend on filtration services of upstream forests to reduce costs of water treatment, most famously New York City but also hundreds of large cities throughout the world (McDonald et al., 2016).

The temporal spread of overall water yields over months or seasons is a critical component of hydrological ecosystem services. Forests and other ecosystems partition incoming precipitation into runoff at the surface, and infiltration into soil. Water entering the soil is further partitioned into sub-surface flow that ultimately contributes to downstream runoff (that is often delayed compared to surface flow), and evapotranspiration by forest vegetation that cycles water back to the atmosphere (Figure 5.1). Water flow regulation by forests and other ecosystems can thus play an important part in controlling floods and maintaining water supply during dry seasons or summer months.

Different types of forest degradation (fragmentation, biotic impoverishment, and biomass loss) have varying effects on water balances and water quality as described in the following sections.

Impacts of forest fragmentation on water services

Fragmentation of forests into isolated patches exposes forest edges and alters many components of hydrology. The general response in exposed edges is decreased canopy cover, altered microclimate, and reduced soil moisture relative to core forest. These effects can alter interception by the canopy, runoff, and infiltration.

A number of field experiments have quantified the impacts of fragmentation by measuring effects as a function of distance from edge into core forest. The main focus of most of these experiments is biodiversity and the occurrence of species (Table 5.1). Some fragmentation

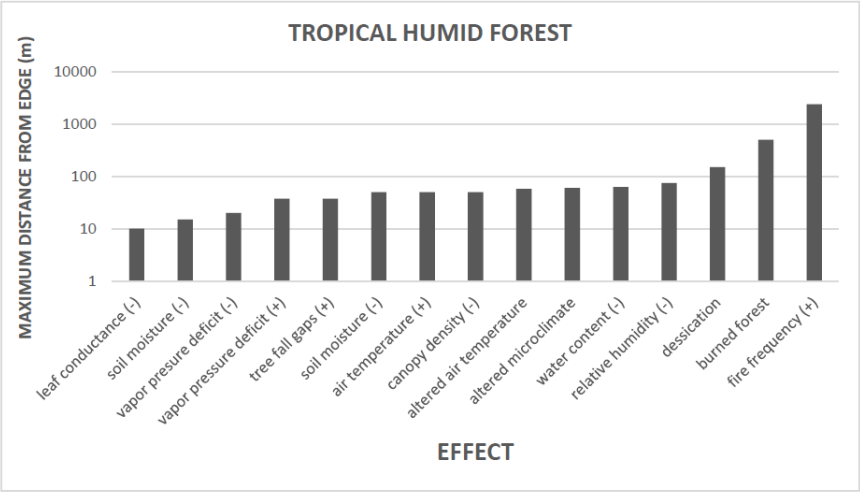
experiments have quantified hydrologic parameters, but no standard set of measurements exists across fragmentation experiments. These measurements identify the distance from edge that the effect penetrates into the forest. Data is sparse, but the available evidence suggests that the impact of fragmentation on increased fire frequency (and consequently on increased runoff and sedimentation) in tropical humid forests can extend over 2 kilometers into the forest (Appendix 2, Table A2.1).

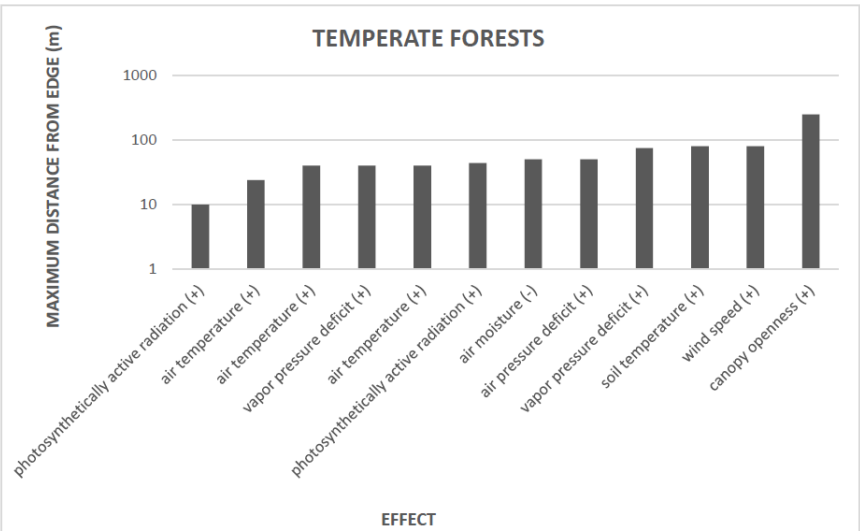
We compiled published measurements on the distance from edge that the effects of forest fragmentation penetrate into a forest. These measurements do not provide a direct measure of proportion of forest in edge and water yield. Rather they provide an indication of the extent to which fragmentation alters the water balance in edges of isolated forest patches. Although data are limited, **we conclude that a distance of 500m from forest edge in humid tropical forests and 250m in temperate forests would reasonably capture the maximum distance in which fragmentation could alter the water balance** (Table 5.2).

Table 5.2: Major large-scale forest fragmentation experiments

Experiment	*Forest type*	*Location*	*Dates*	*Effects of interest*	*Key references*
Biological Dynamics of Forest Fragments Project (BDFPP)	Humid tropical forest	Brazil	1980-present	Patch size and edges on species richness and microclimate	Laurance and others, 2018
Stability of Altered Ecosystems (SAFE)	Humid tropical forest	Malaysia	2009-present	Land use gradient on biodiversity and hydrology	Ewers and others, 2011
The Wog Wog Habitat Fragmentation Experiment	Temperate forest (eucalyptus)	Australia	1985-present	Fragmentation and patch size on species richness	Margules, 1992
The Savannah River Site Corridor Experiment (SRS)	Temperate forest (pine)	USA	1994-present	Corridors between patches on species composition	Damschen and others, 2006; Orrock and others, 2011
Kansas Fragmentation Experiment	Temperate forest (regrowth)	USA	1984-present	Regrowth on species composition	Cook and others, 2005
The Woodland Creation and Ecological Networks Project (WrEN)	Temperate woodland	UK	2013-present	Restoration on species	Watts and others, 2016
Calling Lake Fragmentation Experiment	Boreal mixed woodland	Canada	1993-1998	Fragmentation on avian populations	Schmiegelow and Hannon, 1999

Notes: Reviewed in (Debinski and Holt, 2000) and (Resasco et al., 2017).





Note log scale for Y axes

Figure 5.3: Data from fragmentation experiments on variables that affect water balance

Because (1) susceptibility to fire is a major impact of fragmentation, and (2) burn scars affect hydrologic properties, the difference in return interval of fires between edge and core can serve as a rough proxy for the magnitude of fragmentation on water services. In the tropics, fire return intervals within 1km of edge increased from 5-16 years to 6-19 years beyond 2 km in one Amazon site. In another site, the increase was from 5-22 to 29-109 years (Cochrane, 2001). Based on these two sites, a midpoint value of tropical edge effects might be 0.51 (the average ratios of fire return

intervals between edge and core). In temperate forests, evidence to assess the magnitude of effects is limited. One site indicates a reduction in relative humidity from approximately 80 percent at 250m from edge to 60 percent at the edge (0.8 ratio) (Gehlhausen et al., 2000). Boreal forests, which are naturally patchy in many places, do not experience fragmentation into isolated patches in the same way as tropical and temperate forests (see below on logging impacts in boreal forests).

Logging roads, mines, powerlines, and other infrastructure can affect water quality for downstream users through increased sediment loads from erosion of exposed bare ground. Best management practices provide soil erosion reduction strategies through guidelines on riparian buffer widths and slope steepness above which activities cannot occur (Wenger et al., 2018). Because the extent to which entities follow best management practices is not available across countries, and because erosion varies by soil type, slope, rainfall intensity and other factors, a specific response function between area used for infrastructure and sediment load is difficult to justify. However, rapid infrastructure expansion across developing countries highlights the potential for this type of degradation to affect water quality. The data set on road density could provide a means to include fragmentation into valuation of non-timber forest resources, with higher road density indicating more degradation. Spatial models that account for slope, rainfall, and other determinants of sediment yields could provide additional insight into the impacts of infrastructure expansion in specific locations (Hamel et al., 2015).

Impacts of plantations and biotic impoverishment on water services

The replacement of natural ecosystems by low-diversity tree plantations is the main pathway through which biotic impoverishment alters water services. Tree plantations can strongly modify hydrological processes and affect the provisioning of water services through altered water balance and water quality (Jackson et al., 2005).

The replacement of natural forests by tree plantations affects water services through changes in vegetation structure, species composition, and soil characteristics. In this section, we review, and where possible attempt to quantify, how natural forests and tree plantations differ with respect to three key hydrological functions, namely (1) overall water yield—also referred to as stream flow or runoff; (2) water flow regulation, which underlies societal benefits such as flood control, and irrigation during the dry season and (3) water quality, including measures of sediment and nutrient load in water. The conversion of forests to plantations is widely associated with reductions in water flow regulation, elevating flooding risks and reducing water availability during dry seasons (Guzha et al., 2018; Locatelli and Vignola, 2009; Wenjie et al., 2011). This is because plantations generally sustain lower rates of soil infiltration than forests (Roa-García et al., 2011). The comparatively lower rates of infiltration in plantations are attributable to soil compaction resulting from management-oriented activities such as road construction and timber harvest (Waterloo et al., 2007; Wenjie et al., 2011). Another contributing factor could be that forests are more structurally complex habitats than plantations, which can delay streamflow (Baillie and Davies, 2002) and mitigate flooding.

Water yield. Water yield (defined as the average amount of runoff in an unregulated watershed (that is, without dams or other impoundments) depends strongly on plantation age. Water yields

are known to decrease when plantations replace treeless or sparsely tree-covered habitats such as grasslands and shrublands (Farley et al., 2005). Previous research has shown that replacing grasslands and shrublands with plantations can decrease overall water yields by 45 percent, thereby intensifying water shortages in water stressed regions (Farley et al., 2005; Jobbágy et al., 2011; Sikka et al., 2003). In contrast, the impacts of converting forested ecosystems to tree plantations on overall water yields are less clear. Clear-felling natural forests for plantations is associated with an initial increase in water yields (Komatsu et al., 2008), as young plantations intercept less incoming precipitation, and use and transpire less water than older plantations. The long-term impacts of converting natural forest to plantations, however, are affected by the structure and species composition of forests and plantations, resulting in wide variation (Cao et al., 2008; Wei et al., 2005), and a lack of consistent differences in overall water yields, between natural forests and mature plantations. Evapotranspiration, or the release of water to the atmosphere during photosynthesis, is a key factor underlying variation in water yields across different forest and plantation types. Previous studies have shown that certain commercial plantation species such as *Eucalyptus spp.* and conifers are more water demanding, have lower water use efficiency, and have higher rates of evapotranspiration, than native forest species (Kagawa et al., 2009; Licata et al., 2008). Plantations of such water demanding species are therefore known to decrease overall water yields relative to natural forests (Locatelli and Vignola, 2009). In other cases, water yields by plantations may exceed those of forests for a number of reasons including greater water use efficiency of plantation species, lower biomass volume (and hence lower net evapotranspiration) of plantations (Krishnaswamy et al., 2012), and reduced soil water infiltration resulting in greater surface run-off in plantations (Krishnaswamy et al., 2013).

In the absence of consistent overarching patterns, modeling overall water yield responses to conversion from natural forests to plantations would need to consider both plantation age and plantation/forest type. Young plantations are associated with higher water yield than mature natural forests; water yield then decreases with age, and might attain lower or higher water yield than forests over three or more decades, depending on species (Figure 5.4). However, a major constraint is that many types of plantations and natural forests are not well studied in this regard (Locatelli and Vignola, 2009).

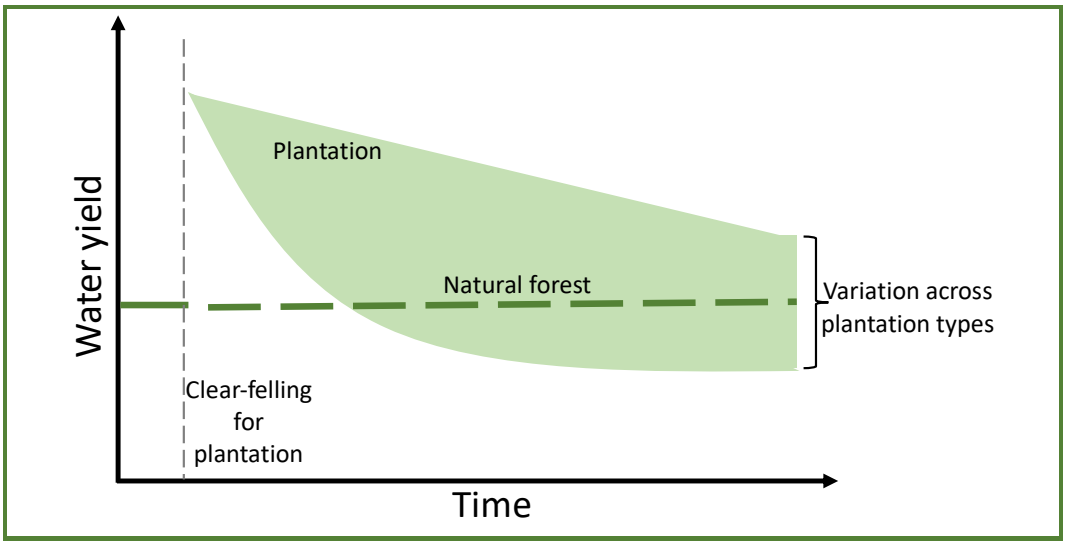


Figure 5.4: Effect of replacing forest with plantation

Flow regulation. Multiple lines of empirical evidence highlight the reduced capacity of plantations compared to forests for water flow regulation. For example, Krishnaswamy and others (2013) reported 30-40 percent lower rates of groundwater recharge via infiltration in *Acacia auriculiformis* plantations compared to tropical forests in India. In China, Wenjie and others (2011) show that water passing through catchments under rubber cultivation has nearly 45 percent lower residence time than water passing through tropical rainforest catchments. This results in an reduction in flood regulation by plantations compared to forests of more than 50 percent (Wenjie et al., 2011). Another study from China reported a six-fold increase in post-rainfall peak flow in China-fir plantations compared to native hardwood forests (Tsai et al., 2009). Lower base-flows in plantations than forests are widely reported (Jobbágy et al., 2011; Locatelli and Vignola, 2009)—for example, China-fir plantations sustain 96 percent lower base-flow than native hardwood forests in China (Tsai et al., 2009), and summer flow rates are 50 percent lower in Douglas fir plantations than natural temperate forests in the USA (Perry and Jones, 2017).

Water quality. Vegetation structure and composition can affect water quality by controlling erosion and decreasing sediment loads (concentrations of suspended solids) and by regulating levels of vital nutrients such as nitrogen and phosphorous in catchments. Erosion control and nutrient regulation by forests and plantations closely track differences in water yields and regulation between these contrasting types of vegetation.

Decreases in vegetation cover are typically associated with increased erosion resulting from greater raindrop impact and wind, and such erosion may be exacerbated further by surface runoff through paths and tracks (Croke and Hairsine, 2006). Thus, sediment erosion is likely to increase by several orders of magnitude when young tree plantations are established by clearing mature plantations or forests (Carlson et al., 2014). While sediment erosion would likely track changes in overall water yields and thereby decrease with plantation age, it appears unlikely that

erosion rates in mature plantations would return to levels as low as those of mature forests. For example, a study found suspended solid concentrations following storm events to be over 10-times higher in a 24-year old China-fir plantation than in natural hardwood forest in Taiwan (Tsai et al., 2009). In another example, a study in Indonesia found sediment yields to be 200 times higher in mature oil palm plantations than mature rainforests, and suspended solid concentrations to be 140-times higher (Carlson et al., 2014). Aside from differences in habitat structure and species composition, management interventions that disturb soil structure, such as roads and skid trails, are an important reason for elevated rates of erosion in plantations compared to natural forests (Sidle et al., 2004).

Regulation of water flow by natural or modified ecosystems can control leaching of nutrients such as nitrogen and phosphorous into waterways, thereby mitigating the ecological and public health risk of eutrophication in downstream areas. In the previous sub-section, evidence indicated that natural forests have a greater capacity for regulating water flow through catchments than do tree plantations, as a result of higher infiltration rates, more structurally complex habitats, and differences in biomass and species composition. Similarly, studies have also found evidence for lower nutrient concentrations in natural forests than in plantations. For example, a study in Chile reported total nitrogen concentrations in streams flowing through *Eucalyptus* plantations to be nearly 80 percent higher than concentrations in native forests (Little Cárdenas et al., 2015). In a New Zealand study, nutrient exports from catchments comprising pine plantations were considerably higher than from catchments under native forest cover (Quinn and Ritter, 2003). However, nutrient regulation characteristics can vary markedly by species composition and soil type. For example, native forests harboring N-fixing trees and bacteria can account for greater exports of dissolved nitrogen than plantations of non N-fixing species such as conifers (Hamilton et al., 2017). In another example, nutrient leaching may be relatively low in plantations of species that show high levels of nutrient uptake, such as oil palm (Comte et al., 2015).

Impacts of biomass loss on water services

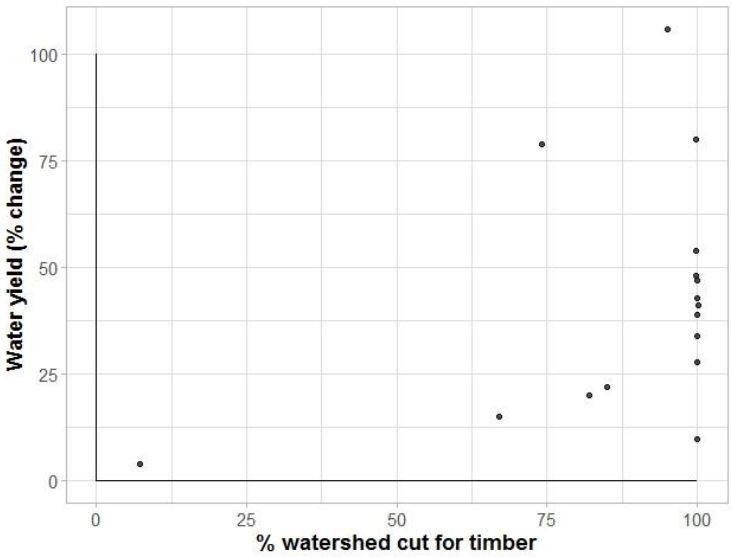
Several mechanisms lead to biomass loss in forests, either as abrupt, temporary loss, or as gradual loss. We consider several human-caused drivers that lead to abrupt biomass loss, including logging (rotational clearcutting in temperate and boreal forests) and excessive fires (which occur naturally in seasonally dry regions but rarely in humid tropical forests). Gradual loss can occur from unsustainable use of forests for livelihoods needs, such as fuelwood and fodder; tree pathogen outbreaks which are exacerbated by international trade and movement coupled with climate change and forest degradation; and selective logging. Each of these types of biomass loss can alter water balances and water quality. The effects on water services likely vary depending on the underlying cause of biomass loss—clear-cut, burn scar, pathogen outbreaks, or unsustainable use. Most evidence is available from studies of the effects of clear-cutting but similar relationships can be expected for other types of biomass loss.

Impact of logged forests on water services

Water balance. Changes in water yield following rotational clear cutting in forests are strongly dependent on time since clearing, the extent of the watershed cleared at any time in the

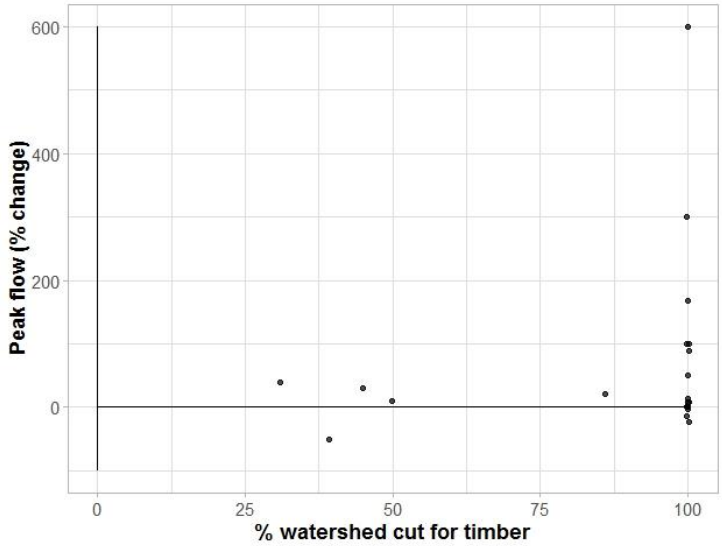
rotational cycle, and the location of clearing within the watershed. Clear-cutting is most prominent in temperate and boreal forests. Initial increases in water yield and peak flow occur after clearing, with the magnitude roughly proportional to percent biomass loss. The effect diminishes rapidly, usually within 3 to 10 years, but can be prolonged if natural regrowth does not occur (Hornbeck et al., 1993). A compilation of studies (Appendix 2.2) suggests that water yield (Figure 5.5) and peak flow (Figure 5.6) alter substantially only when close to 100 percent of the watershed is cleared for rotational harvesting. There is substantial variability in impacts on water yield and peak flow even with 100 percent of clearing in the watershed.

The relationships for temperate and boreal clear-cut forests between water yield/peak flow and biomass loss are broadly consistent with tropical forests, considering proportional biomass loss through selective logging. Modest increases in streamflow occur with modest loss of biomass and abrupt but variable increase at high biomass loss (Bruijnzeel, 2004).



Notes: The y-axis is comprised of a mix of measurements on water quantity: water yield as percent of annual precipitation and percent changes in annual runoff, baseflow, and mean annual streamflow measured between 1 and 4 years after clearcut.
Source: Data are compiled from 14 studies (see Appendix 2.2).

Figure 5.5: Effect of logging on water yield



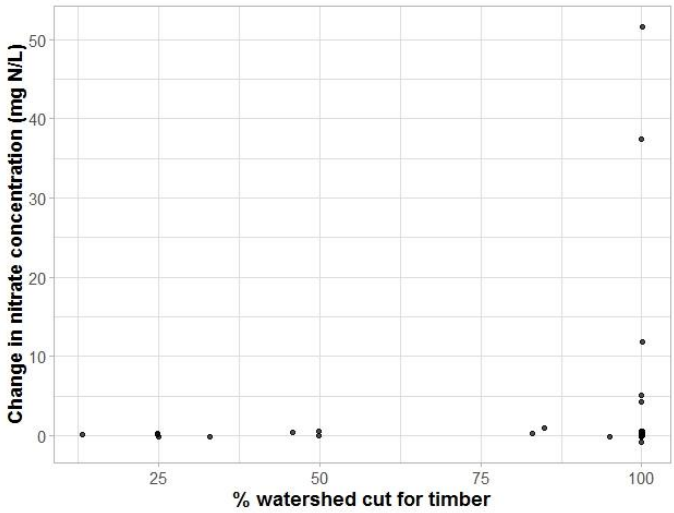
Source: Data are compiled from 17 studies (APPENDIX 2.2).

Figure 5.6: Effect of logging on peak flow

Water quality. Nitrate concentration is a commonly-used metric of water quality, with high levels being associated with eutrophication of lakes and streams and with methemoglobinemia in infants from drinking water. World Health Organization (WHO) standards indicate poor water quality above 10 mg nitrate per liter (World Health Organization, 2003). A compilation of 24 studies (Appendix 2.2) showed little evidence of deterioration of water quality until nearly complete clearance of the watershed during rotational harvest (Figure 5.7). Even at complete clearance, studies show high variance in nitrate concentrations, with the degree of water quality deterioration dependent on the extent to which remaining ground layer vegetation is able to buffer water flow and runoff. Generally, sites with lower nitrate concentrations in control (uncut) sites showed more modest increases than sites with high concentrations in control sites. In some systems, although freshwater nitrate concentration did not increase, increased water yields caused a higher nitrate export when the forest was clear cut (Brown et al., 1973). A thin forest litter layer due to high rates of decomposition in a wet and mild climate and rapid regrowth of the forest understory appears to limit stream nitrate increases after forest harvest (Fredriksen et al., 1975).

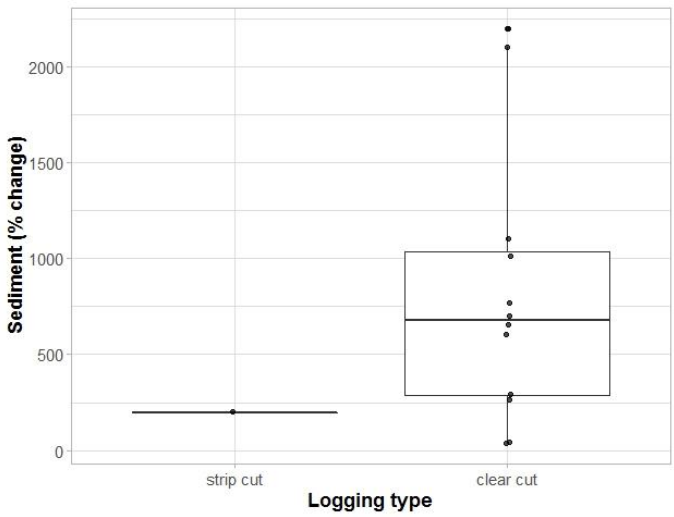
In addition to nitrate concentrations, higher streamflow and velocities with clear cuts result in additional transport of solid and dissolved materials that can adversely affect water quality for human use and damage aquatic habitat (Robichaud et al., 2000). A compilation of studies (Appendix 2.2) shows substantial but highly context-dependent increase in sediment content in clear cuts (Figure 5.8).

Based on data compiled from multiple studies (Appendix 2.2), the relationship between sediment content and degradation showed substantial but highly context-dependent increases indicating that logging can adversely affect water quality for human use and damage aquatic habitat.



Source: Data compiled from studies listed in Appendix 2.2.

Figure 5.7: Effect of logging on stream or groundwater nitrate concentration



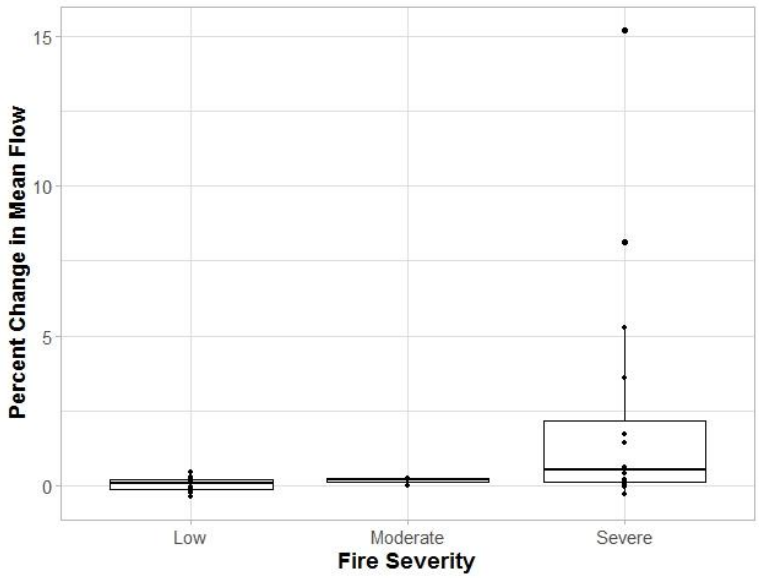
Source: Data compiled from studies listed in Appendix 2.2.

Figure 5.8: Effect of logging on sediment concentration

Impacts of fires on water services

Water balance. Wildfires generally increase water yields, peak flows, and the potential for flash floods, depending on soils and on the intensity and duration of rainfall events (Hallema et al., 2018; Neary and Tecle, 2015; Robichaud et al., 2000). With less than 10 percent of ground surface covered with plants and litter, surface runoff can increase over 70 percent and erosion can increase by three orders of magnitude, depending on burn severity and hydrologic events (Robichaud et al., 2000). Increased sediment loads following fires have negatively affected water supplies in cities in the southwestern United States and Australia over the last two decades (Martin, 2016).

The degradation effects of a wildfire depend on both the fire's intensity (for example, litter layer or grass fires vs crown fires) and evolutionary adaptation of the trees to fires (for example, fire-adapted Mediterranean, temperate, and boreal forests vs non-fire adapted humid tropical forests). Water yield and water flow tend to only be affected by severe fires (Fig 5.9), which result in extensive tree mortality and loss of understory vegetation and litter later.

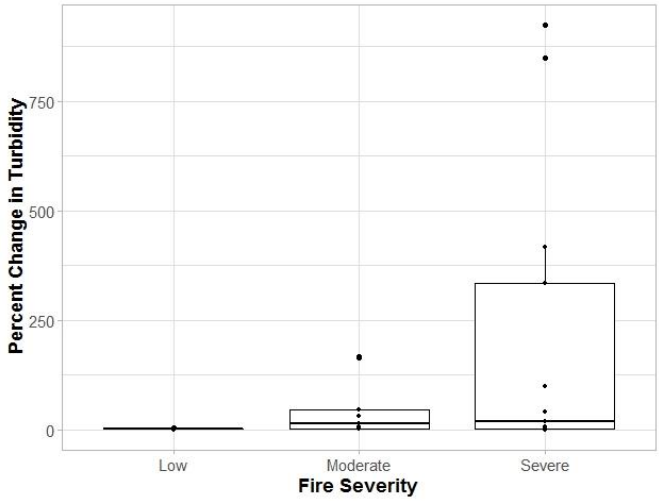


Notes: Data points were compiled from studies listed in Appendix 2.2. Fire severity was determined from reports in each study and the percent of watershed affected by fire. Measurements were taken 2 to 12 years following the fire.

Figure 5.9: Effect of forest fire severity on mean stream flow

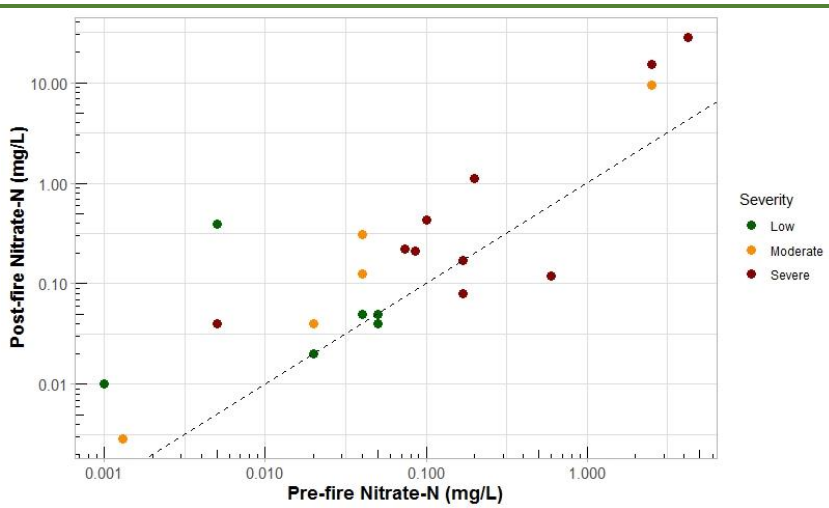
Water quality. Fire severity particularly affects stream sediment load/turbidity (Figure 5.10) and in some cases stream nutrient content (Figure 5.11). In general, the effect is small although in some studies the change in turbidity increased many hundred-fold relative to pre-fire conditions. Nutrient content also increased many-fold, but only a few studies indicate nitrogen levels above WHO drinking water standards (10 mg/L of nitrate-N). Nearly all of the studies were undertaken

in temperate forests which are more resilient to fire than humid tropical forests. For the studies of turbidity, measurements were taken between 5 months and 5 years after the fire, while in studies of nitrogen measurements were taken between 3.5 and 12 years after the fire.



Notes: Fire severity was estimated based on reports in the studies and the percent of watershed affected by fire. Data are from a compilation of studies listed in Appendix 2.2).

Figure 5.10: Effect of forest fire severity on turbidity



Notes: Fire was estimated based on reports in the studies and the percent of watershed affected by fire. Data points are compiled from studies listed in Appendix 2.2. Dashed line is one-to-one line. Note log-log scale.

Figure 5.11. Effect of forest fire severity on nitrate-nitrogen

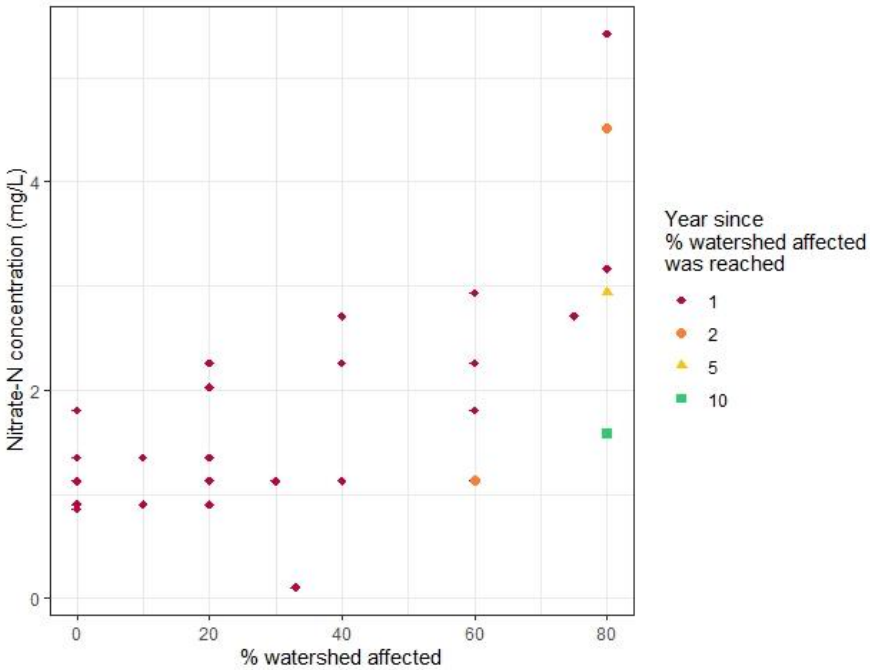
Impacts of pathogen outbreaks on water services

Pathogen outbreaks tend to have less effect on water quality than logging or fire outbreaks, mainly because the understorey vegetation and soil structure tend to remain intact and mortality tends to be a slow, drawn-out process, whereas in fire or logging disturbances biomass is either burned or removed over much shorter time periods (Edburg et al., 2012). Tree pathogen outbreaks are a natural cause of forest degradation and are exacerbated by introduction through movement of people and good that carry pathogens and by climate change. Studies that monitor forest ecosystem services following insect outbreaks capture the effect of a particular type of biomass loss, one that gradually occurs over long time periods. Outbreaks can last several years and gradually impact forest functioning before physical biomass loss occurs (it takes years to decades for trees stems to fall following an attack).

Overall, the literature suggests that tree pathogen outbreaks lead to moderate increases in nitrate levels (for example, see Beudert and others, 2015, and Zimmermann and others, 2000, for the effects of bark beetle; and Swank and others, 1981, for the effects of fall cankerworm. The few studies with a steep increase in nitrate levels were characterized by a very rapid and extensive spread of the disturbance (Beudert et al., 2015). Importantly, peak nitrate levels even in those instances of extensive spread stayed below the WHO limit for water-drinking safety. In terms of recovery, the effect in these watersheds was dampened within 5 years and fully disappeared within 10 years.

The amount of ground vegetation cover could mitigate nitrate losses resulting from pathogen outbreaks, as is hypothesized in the case of logging. The more ground vegetation there is to soak up the extra nutrients, the milder the impact of forest dieback on nitrate loading in the water (Huber, 2005). This was also documented in an experimental setting, where continuous weeding was used to prevent regrowth after felling (Vitousek et al., 1979). More generally, this study shows that vegetation recovery is critical for conferring resilience to ecosystems and inhibiting nitrate losses after a disturbance. Figure 5.12 shows an example for stream nitrate quality. A high peak level of nitrate concentration (top red dot in 80 percent affected column) is reached in the only watershed where the outbreak spread is rapid and extensive. However, even this peak is below the WHO limit for water-drinking safety. The effect in this watershed is dampened within 5 years and falls within the range of values observed in un-disturbed watersheds within 10 years. Ground vegetation coverage ameliorates the magnitude of the nutrient effect.

Studies that assess the impacts of unsustainable use of forests for fuelwood, fodder, and other livelihood needs on water quality are scarce. One can expect that the response to this type of gradual biomass loss is similar to that of pathogen outbreaks.



Notes: Data are compiled from Beudert and others, 2015; Zimmermann and others, 2000; and Swank and others, 1981); though it should be noted that the first two studies study the same watersheds over different but overlapping time periods. Nitrate concentration is reported as mg/L of nitrate-N; a conversion factor of 0.2258 is used when the original data was reported as mg/L of nitrate. Percent watershed affected is either reported as the proportion of trees killed by bark beetle (Beudert et al., 2015; Zimmermann et al., 2000), or as the proportion of leaf mass consumed by the fall cankerworm (Swank et al., 1981). In all studies, the watersheds under study are nearly totally forested. Note that all values are below the WHO standard of 10 mg/l.

Figure 5.12: Effect of pathogen-induced dieback on nitrate concentration

5.2 Impacts of forest degradation on recreation, hunting, and fishing

Impacts on connectivity for wildlife tourism

Tourism is a multi-trillion dollar industry and generates hundreds of thousands of jobs. Wildlife tourism is a growing component with potential advantages for much-needed job creation around protected areas and other remote locations (Twining-Ward et al., 2018). Wildlife tourism often depends on abundant populations of charismatic fauna, which in turn depend on habitat to maintain populations. Large, charismatic species such as tigers, bears, and jaguars with ranges extending beyond protected area boundaries require wildlife corridors to move between them (Gilbert-Norton et al., 2010). Persistence of these species depends on the quality of habitat in wildlife corridors (Hodgson et al., 2011).

Wildlife corridors are a well-established approach in conservation landscapes. Major existing wildlife corridors that connect forest habitats across large areas include the Great Eastern Ranges corridor for native plants and animals in Australia (Fitzsimons et al., 2013), the Yellowstone to

Yukon corridor for grizzly bears in North America (Chester, 2015), the Terai Arc Landscape for tigers in India and Nepal (Wikramanayake et al., 2010), and the Mesoamerican Biological Corridor for jaguars in Central America (Holland, 2012). Other migration routes for large animals that range between protected areas or other isolated forest patches could be relevant for recreation and tourism (Benz et al., 2016) (Box 5.1).

Box 5.1: Jaguars, connectivity, and tourism in the Brazilian Pantanal

The Brazilian Pantanal and its burgeoning jaguar tourism industry provide an example of the link between forest degradation and wildlife tourism. Jaguar populations have remained high in the Pantanal alongside a substantial human presence; 95 percent of the region is privately owned and has been used to ranch cattle for more than 200 years (de Souza et al., 2018). The yearly flood pulse that submerges the region in the wet season has discouraged intensive ranching operations and development. Consequently, the Pantanal's natural land cover has been largely preserved, allowing for broad persistence of jaguar across the landscape and the maintenance of natural corridors. The Pantanal is second only to the Amazon in terms of corridor potential (Silveira et al., 2014). Genetic analyses indicate that the region hosts high genetic variability. The combination of intact forest patches within the region and connectivity throughout the Pantanal has allowed for this robust population of jaguar to persist, in sharp contrast to populations in the nearby Atlantic Forest, which are highly fragmented and less genetically diverse (Valdez et al., 2015). This large, relatively visible population has become the foundation for a young tourism industry built over the last 20 years. One 2015 estimate across a representative portion of the Pantanal placed the annual land use revenue of jaguar tourism at approximately USD84.3 per ha per year (Tortato et al., 2017). As the industry expands, this value is approaching earlier estimates of revenue from land converted to cattle pasture at USD119–216 per ha per year (Seidl et al., 2001). Most jaguar tourism is directly associated with cattle ranches, a seemingly contradictory situation where jaguar represent economic gains from tourism and losses from cattle depredation (Tortato and Izzo, 2017). Furthermore, ranching has recently begun to intensify in the Pantanal and is now driving an increase in habitat fragmentation across the region (Silveira et al., 2014). As this cloudy relationship between ranching and jaguar tourism takes shape a firm valuation of forest corridors and intact forest patches could contribute stability to a region on the brink of economic and environmental transformation.

Many forested landscapes with value for wildlife tourism conservation are poised to lose connectivity with rapid infrastructure expansion throughout many developing countries (Laurance, 2018). Although we did not find studies that explicitly link connectivity, wildlife populations, and tourism revenue, we expect that valuation of natural capital would place high value on forest fragments that increase connectivity between protected areas or other suitable habitat for species relevant for tourism and recreation.

Impacts of forest degradation on food security from hunting and fishing

Forest degradation through biotic impoverishment of wildlife (defaunation) can contribute to food insecurity if substitutes for protein are not available.

Importance of bushmeat for local diets. Bushmeat, defined as "meat derived from any wild terrestrial mammal, bird, reptile, or amphibian harvested for subsistence or trade, most often illegally," (Cawthorn and Hoffman, 2015) contributes positively to food security for millions of

people throughout Africa, Latin America, and Asia. Bushmeat provides a principal dietary source of protein, micro-nutrients such as iron, fats, and medicine, as well as opportunities for income. People rely on bushmeat predominantly in places where domestic protein sources are unavailable or unaffordable. Bushmeat contributes 80 to 90 percent of animal protein in some rural regions of West and Central Africa and over 20 percent in the Amazon for some indigenous groups (Cawthorn and Hoffman, 2015). Many site-specific studies conclude that bushmeat provides critical food security and nutrition for forest-dwelling, poor rural populations, for example, (Borgerson et al., 2019; Sarti et al., 2015) (also see references in section 5.3). In addition to local use, in some areas there is considerable demand for commercially-harvested bushmeat, which is primarily consumed by wealthy populations in urban areas and is valued at several billion dollars annually (Brashares et al., 2011). Paradoxically, bushmeat also presents risks through unsafe handling practices. Exposure to blood and other fluids from butchering and preparing wild animals, particularly primates, can transmit zoonotic diseases such as HIV, Ebola, and COVID-19 (Van Vliet et al., 2017).

Causes of defaunation. Unsustainable bushmeat hunting is the leading cause of defaunation, particularly from increasing demand in urban areas that drives commercial exploitation of wildlife resources (Ripple et al., 2016). The direct effect of forest degradation through fragmentation, fire, and logging on availability of bushmeat is not straightforward. While fragmented forests support fewer large mammals than intact forests, degraded forest patches are likely to harbor higher abundances of small mammals, rodents, and ungulates (Torres et al., 2018). Indirectly, degradation from logging and other extractive industries in forests contribute to defaunation through roads that enable commercial bushmeat hunting with modern weapons. Settlements associated with labor to work in the industries provide a market for bushmeat (Cawthorn and Hoffman, 2015).

Effects of defaunation. Forests depleted of wildlife display an “empty forest syndrome”, where the vegetation may appear intact but the absence of medium and large fauna leads to cascading impacts into long-term changes in tree composition by reducing primary seed dispersers, notably large mammals and birds. Lower abundance of large-seeded trees, aggregation around parent trees, and reduced species diversity potentially alter wildlife habitat, carbon storage and other ecosystems services (Gardner et al., 2019a; Krause and Nielsen, 2019; Kurten, 2013). In sum, the effects of forest degradation on the availability of bushmeat are intertwined and context specific. Beyond the value of intact forests to maintain populations of large mammals, a straightforward approach to quantify the effects of forest degradation on food security and income from bushmeat at a national level is not obvious although of considerable consequence to poor, rural populations.

Effects on fishing. The effect of forest degradation on fishing is similarly potentially relevant, but indirect. Globally, fish provide 15 percent of protein along with essential micro-nutrients (Fisher et al., 2017). Poor and undernourished populations in developing countries are particularly reliant on inland fish compared with aquaculture and marine sources (McIntyre et al., 2016). Abundance and diversity of fish species depend on good water quality, which in turn reflects conditions in the watershed (Allan, 2004). Riparian vegetation filters sediment and excess nutrients from entering streams, reduces water temperature and fluctuations in temperature

through shade, stabilizes streambanks, and provides food for insects from overhanging leaves and woody debris to keep streams conducive to healthy fish populations (Tabacchi et al., 2000).

Riparian buffers free from logging and other types of degradation are well-established best management practices to reduce sediment loads into lakes, streams, and rivers. In temperate forests, best management practices from forestry agencies, for example in the USA, call for riparian buffers of 10 to 30 meters (Broadmeadow and Nisbet, 2004). Less consensus exists for widths of riparian vegetation in tropical forests. Guidelines from the Roundtable of Sustainable Palm Oil specify minimum riparian reserve widths ranging from 5 to 100 meters for river widths from 1 to more than 50 meters in oil palm plantations and call for wider buffers in small streams on steep slopes (Barclay et al., 2017). The Brazilian Forest Code specifies a minimum buffer depending on the size of the waterway (Soares-Filho et al., 2014). Best management practices for riparian areas, if practiced, are designed to maintain water quality that would maintain aquatic populations.

Effects on coastal areas. Logging in forests that drain into coastal areas can also increase sediment load into coral reefs that provide essential habitat for breeding and spawning fish (Hamilton et al., 2017). As is the case of bushmeat, effects of forest degradation on fishing can be critically important for coastal and inland poor populations. However, the fine-scale nature of riparian buffers and indirect relationships between forest degradation, water quality, fish populations, and food security are not easily captured in national-scale measures of natural capital.

5.3 Impacts of forest degradation on non-wood forest products

Impacts of forest degradation on availability of non-wood forest products. Forest degradation has significant negative consequences for the provision of non-wood (NWFPs,also referred to as non-timber forest products or NTFPs), often impacting indigenous and local communities whose livelihoods depend on the extraction of these forest products. In several places, from the Amazon tropical forest to rural parts of India, marginalized communities living on the edges of forests not only collect NWFPs for their own consumption, but also for commercial sale. This activity often represents the main source of income for these communities, making the extraction of NWFPs an important contributor to local economies in remote rural areas worldwide (Banerjee and Madhurima, 2013; Shanley and Luz, 2003). The loss of NWFPs due to forest degradation can have an even greater impact on indigenous communities who are inextricably linked to these ecosystems and completely rely on forests for their survival (Banerjee and Madhurima, 2013).

Among the many types of anthropogenic disturbances, studies indicate that logging, including selective timber harvesting, and fragmentation are the main threats for the provision of NWFPs. A review that collated examples across tropical regions showed that selective logging and associated fragmentation have pervasive impacts on a myriad of NWFPs of livelihood importance (Rist et al., 2012). Native fruits and nuts, oil, bushmeat, and medicinal plants were among the most affected NWFPs. Forest degradation from selective logging resulted in significant declines of fruit, nut, and game availability to local communities in eastern Amazon (Guariguata et al., 2009; Menton, 2003). Declines up to 75 percent of annual household consumption of game/bush meat and native fruits were recorded following logging and associated fire in the eastern Amazon

(Shanley et al., 2002). Selective logging and forest fragmentation due to the creation of roads for logging vehicles also led to a decline of 86 percent in tree oil harvest, as well as in the availability of bush meat, in Eastern Cameroon (Rist et al., 2012; Schneemann, 1995).

Impacts of forest degradation on medicinal plants. Forest degradation has also been implicated as a major threat to the availability to medicinal plants and traditional medicine practices of local communities living on the edges of forested ecosystems. More than 80 percent of the developing world still relies on traditional medicine for primary health care, as access to modern drugs is limited or often inexistent (Shanley and Luz, 2003). Furthermore, in places such as the Brazilian Amazon, the commercialization of medicinal plants is an important source of income to households in rural areas and the loss of access to these products can have significant consequences in already underprivileged and underdeveloped communities. Logging, fragmentation and over-harvesting in the Brazilian Amazon has greatly diminished the availability of widely used medicinal plants (Shanley and Luz, 2003). The diversity of medicinal plants was also significantly lower in heavily logged forests in northwest Pakistan (Adnan and Hölscher, 2011). This decline has major consequences for local communities in forested areas.

Indirect impacts of forest degradation. Finally, the loss of regulating ecosystem services due to forest degradation can also indirectly impact the provision of non-wood forest products. Studies have shown that forest fragmentation can change the composition of pollinator communities and result in decreases of pollinator diversity and abundance. These effects have been recorded for multiple types of forested ecosystems from temperate forests in North America to tropical forests in Costa Rica (Brosi et al., 2008; Taki et al., 2007). A case study from the eastern Amazon illustrates how the loss of pollinator diversity can affect the provision of non-wood forest products. Anthropogenic disturbances, including fragmentation and intensive agricultural practices, impact pollination services to açaí palm in the region, one of the main sources of income for local communities (Campbell et al., 2018). The açaí palm is highly dependent on biotic pollination and the loss of insect pollinators can not only reduce fertilization rates, but also decrease the yield of açaí berry in managed agroforests, strongly affecting local communities that rely on the berry as their main economic activity.

5.4 Impacts of forest degradation on other ecosystem services

The three ecosystem services considered above (water services; recreation, hunting and fishing; and non-wood forest products) are a subset of those affected by forest degradation.

Several additional ecosystem services are relevant at a regional scale. For example, degraded forests in coastal areas, including mangrove forests, are potentially relevant for estuarine and coastal protection to attenuate the strength of waves and storm surges (Barbier, 2015). Degraded forests with high human impact are also susceptible to fire. Emissions from forest fires are a serious public health hazard with negative effects on air quality for downwind populations, such as in fire-prone regions of North America (McClure and Jaffe, 2018) and human-caused fires that cause haze throughout Southeast Asia in the dry season (Koplitz et al., 2016). An additional ecosystem service affected by forest degradation is regional climate. To the extent that degraded forests alter hydrology and reduce evapotranspiration, regional precipitation patterns critical for agriculture can be affected (Coe et al., 2017; Salati et al., 1983).

At a global scale, forests play a major role in regulating climate through carbon sequestration and storage. Forest degradation that reduces biomass compromises this service (Chaplin-Kramer et al., 2015). Finally, habitat that supports biodiversity is a highly relevant service provided by forests although difficult to quantify in economic metrics.

A comprehensive methodology to incorporate forest condition in the valuation of natural capital from non-wood forest assets would incorporate these regional and global-scale ecosystem services. Due to transboundary flow of regional and global ecosystem services, a considerable challenge will be to link services from forests in one country to benefits or costs in another.

6. Conclusions

The available evidence shows that forest degradation can have important effects on ecosystem services. This has significant implications for any effort to estimate the value of forests, such as that prepared for the World Bank's *The Changing Wealth of Nations* (CWON) report. Such efforts attempt to estimate the value of forests by quantifying and valuing the flows of ecosystem services they provide, and generally consider a particular area to be either under forest or under another, non-forest use—no distinction is made for the condition of the forest.

Fully accounting for forest condition in estimates of forest value is difficult if not impossible at present due to the lack of data measuring the various aspects of degradation, except locally, and to still limited understanding of the impacts of degradation. In this chapter, we begin by assessing the robustness of the evidence on these effects. We then examine how forest degradation would affect efforts to estimate the value of forests.

6.1 Robustness of evidence on the effects of degradation

The robustness of the evidence to quantify the effects of forest degradation on the three ecosystem services considered here varies across the types of degradation and the effects. In this section, we list the effects in approximate order of most to least robust, based on the discussion in the previous sections. Table 6.1 summarizes our assessment of the evidence to potentially quantify the effects.

Edge effects from increased forest patchiness on water balance (--). Forest edges tend to have lower soil moisture, canopy cover, and infiltration.[1] Fragmentation experiments indicate that these effects penetrate approximately 500 into the forest from the edge of forest patches in tropical forests, and 250m in temperate forests. There is little direct evidence on the effect on water yields through measurements of stream flow. The observed hydrological effects suggest higher runoff and peak flows downstream of forests with an increase in proportion of forest within the edge. In the humid tropics, where trees are less adapted to fire than those in drier temperate forests, susceptibility of edge forest to human-caused fire is a particular concern.

Clear cut logging, fire, and pathogen outbreaks on services (--). Biomass loss from logging, fires, and pathogen outbreaks has cascading consequences for water services. Studies of rotational clear-cut logging in temperate and boreal forests indicate that temporary removal of biomass increases runoff and peak flow when a large proportion of the watershed is cleared. Nitrate and sediment concentrations increase non-linearly with a high percentages of biomass loss in watersheds. Effects are highly context-specific depending on soils, hydrological events, and vegetation cover. Fires in catchment areas can negatively impact downstream water quality through sediment loads.

Soil erosion from roads and infrastructure on water quality (-). Roads, mines, and other infrastructure lead to exposed ground and the potential for soil erosion to increase sediment load. Best management practices aim to reduce erosion. The extent to which these practices are

[1] The proportion of forest in edges could be calculated at a national level as in Chaplin-Kramer and others, 2015.

applied are not known, but data on road density and human impact can provide a proxy for the potential for downstream impacts.

Tree plantations on water services (-). In young tree plantations, water runoff increases relative to natural forests. Increased runoff can lead to reduced dry season flows and increased flood risk. In both water-scarce and water-abundant regions, lower canopy cover and exposed soil can increase sediment loads.

Table 6.1: Assessment of robustness of effects of forest degradation on ecosystem services

(--) Relatively strong evidence with high relevance, negative effect on ecosystem services
(-) Medium evidence
(?) Inconclusive evidence

	Water services		*Recreation, hunting, and fishing*	*Non-wood forest products*
Type of degradation	*Water balance*	*Water quality*		
Fragmentation:				
Isolation of patches and exposed edge	Edge effects (- -)			Habitat to maintain NWFP tree species (-)
Fragmentation and barriers in contiguous patches		Sediment (-)	Wildlife tourism (-)	
Impoverishment of species composition:				
Tree plantations	Increase runoff (-?)			Tree species for NWFP (?)
Loss of fauna species			Bushmeat availability (?)	
Biomass loss:				
Abrupt biomass loss (Clear-cut logging, fires)	Increase runoff (- -)	Nitrate and sediment runoff with high biomass loss (-)	Fish availability (?)	Availability of NWFPs for indigenous and local communities (-)
Gradual biomass loss (selective logging, pathogen outbreaks, unsustainable use)	Increase runoff only if high biomass loss (-)	Nitrate and sediment runoff with high biomass loss (?)		Availability of NWFPs for indigenous and local communities (?)

Loss of landscape connectivity on wildlife tourism (-). Wildlife tourism, which generates revenue and job opportunities, requires healthy populations of charismatic, large-ranging species which in turn require large areas without barriers to movement. Rapid infrastructure expansion creates such barriers and fragmentation restricts population movement. An overlap of corridor locations

on road density (and plans for future infrastructure if possible) could quantify this effect although a readily available global dataset of relevant corridors does not exist to our knowledge.[2]

Forest degradation effects on availability of NWFP for indigenous and local communities (-). Local case studies indicate that forest fragmentation and logging negatively affect the availability of NWFPs, such as medicines, fruit, nuts, and oils, which provide income and utility for many indigenous and local communities living near forests throughout the tropics.[3] The direct effects of forest degradation reduce the abundance of tree species relevant for NWFPs, while indirect effects alter habitat for pollinators leading to reduced yields of fruits such as the açaí berry.

Fragmentation and defaunation effect on bushmeat availability for hunting (?). Bushmeat provides an important source of protein and other nutrients for poor, rural population, particularly in Central and West Africa. However, evidence that forest degradation affects the ability of people to access bushmeat is sparse and conflicting, particularly since fragmented, degraded forest patches can harbor high abundances of rodents and small mammals.

Degradation of riparian vegetation on fish availability (?). Like bushmeat, fish from inland and coastal are an important protein source. Riparian forests are highly relevant for water quality and healthy aquatic populations. Direct evidence linking degraded riparian forests with loss of fish resources is sparse. The effect is difficult to capture at a national level due to the high resolution required to map riparian buffers, suggesting that this effect is problematic to capture in CWON.

6.2 Effects of forest degradation on estimates of forest value

Estimates of the value of forests that do not take into consideration the condition of the forest may be incorrect. Although it isn't possible at present to arrive at estimates of forest that incorporate the effects of degradation, in some cases it is possible to say whether estimates that do not consider forest condition are likely to be under- or over-estimated. These effects are summarized in Figure 6.1.

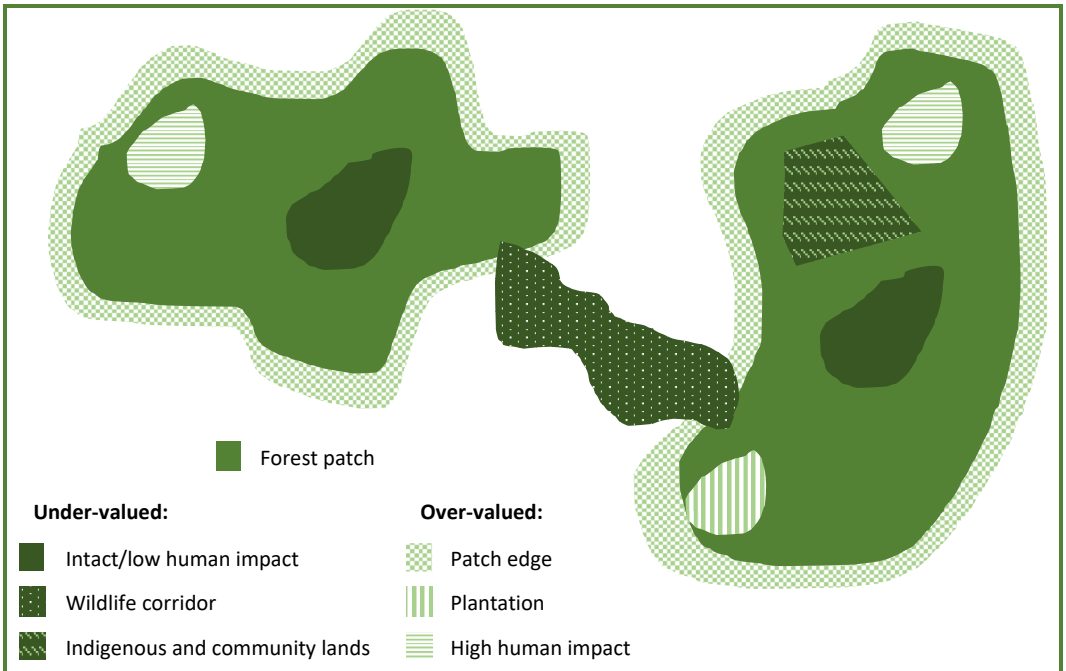
Valuation that does not consider forest condition is likely to overestimate the value of forests in the following locations:

- The **edges of forest patches** (500m for humid tropical forests, 250m for temperate forests) due to their impact of edges of forest patches on water yield and susceptibility to fire;
- **Forests fragmented** by roads and other infrastructure, due to their impact of soil erosion on water quality and on landscape connectivity;
- **Monoculture tree plantations** (particularly in water-scarce regions) due to their impact on water balance and loss of non-wood forest products;
- **Heavily logged forests**, due to the impact on water yield and water quality; and

[2] As a proxy, datasets of intact landscapes, biodiversity intactness, low impact areas combined with defaunation index could indicate where forests have particularly high potential value for connectivity.

[3] While mapping forests where people rely on NWFPs is not straightforward, data on locations of indigenous and local community lands in forests can serve as a proxy, for example, see Garnett and others (2018).

- **Riparian forests** that buffer streams, due to their impact on water quality and possibly on fishing.



Schematic of approach to downscale (light green) or upscale (dark green) value of forest to account for degradation effects on the ecosystem services considered in this report.

Figure 6.1: Possible biases in valuation that doesn't consider forest condition

Conversely, valuation that does not consider forest condition is likely to underestimate the value of forests in:

- **Intact forest** (which likely overlaps with species-rich forest, and forest with low human impact);
- **Forests that provide landscape connectivity** to maintain value of charismatic species for recreation and wildlife tourism.
- **Forested indigenous and community lands** that provide NWFPs to support livelihoods.

The extent to which forest values that do not consider degradation may be over- or under-estimated is difficult to assess with accuracy based on existing information. Moreover, the effects of degradation are likely to vary with biophysical conditions such as soil and climate. Table 6.2 below provides some very crude estimates of the relative magnitudes of these effects of forest degradation on estimates of forest value. It should be stressed that these estimates are very uncertain.

Table 6.2: Relative magnitudes of effects of forest degradation on ecosystem services in over- and under-valued forests

	Water services	*Recreation, hunting, and fishing*	*Non-wood forest products*
Enhance value:			
Intact/low human impact/high biomass	++		
Wildlife corridors		+	
Indigenous and community lands			+ as function of livelihood dependence of local population
Reduce value:			
Forest edge	- - for tropical humid forests - for temperate forests		
Plantation	- - for plantations in previously un-forested locations - for plantations replacing forest		
High human impact/low biomass	-		

Notes: Assessment is based on the literature review in Chapter 5 using the value with the greatest impact.
+ + and - - indicate relatively higher or lower effect on forest value than + and -.

Appendix 1. Data sets on forest degradation

The World Bank's *Changing Wealth of Nations* report includes assets if at least 100 countries can be included, data are publicly available, and data are regularly updated to provide a time series. In the data sets that are available to apply varying values of forest to incorporate degradation (or intact forest), global data sets are generally available for many of the attributes. An exception is spatial data on tree plantations available for a subset of countries but non-spatial, country-level data can be obtained for countries without spatial data (Table A1.1).

Table A1.1: Potential data sets to assess the extent of forest degradation

Forest attribute	*Potential data sets*	*Covers >100 countries*	*Publicly available*	*Periodically updated*
Enhance value:				
Intact/low human impact	Intact landscapes	Yes	Yes	No
	Defaunation index	No	Yes	No
	Forest integrity index	No	Yes	No
	Biodiversity intactness	Yes	Yes	No
	Biomass maps	Yes	Yes	No
Wildlife corridor	Maps of corridors?	?	?	NO
Indigenous and community lands	Indigenous and community territories	Yes	On request from author	No
Reduce value:				
Forest edge	Forest cover	Yes	Yes	Yes
Plantation	Plantations	No	Yes	No
High human impact	Human modification index	Yes	Yes	No
	Road density	Yes	Yes	No

The available data sets are produced by the scientific and NGO community rather than national or international institutions. They consequently are not necessarily produced consistently at periodic intervals. However, time series of some data sets, for example, human impact, road density, forest cover, have been made publicly available. Ideally, institutions such as the United Nations Food and Agriculture Organization would produce these data sets after the scientific community has developed a repeatable and sound methodology. With new sensors and publicly available data streams, additional methods and data sets to monitor forest degradation are likely to become available in coming years.

A next step is to overlay relevant data sets using a spatial, GIS approach to assess which are most appropriate. Some data sets that provide similar information. For example, human footprint and

road density, need to be examined to determine how best to use them in combination. National-level statistics can be derived from the spatial data sets.

Some types of forest degradation are not possible to map globally or at a resolution needed to develop country-level assessments. Maps of wildlife corridors, for example, are available for well-known corridors but data sets will not include all relevant corridors. In addition, data sets to assess local impacts on food security and incomes from bushmeat and fish as well as local access to non-wood forest products are not globally available. These impacts are most likely to affect vulnerable, rural populations who depend on healthy forests for livelihoods. Although difficult to include in national assessments, they are nevertheless critical components of efforts to incorporate forest degradation in valuing forest assets. An approach could value forests in locations with indigenous and local populations who rely on forests for livelihoods, based on data such as that compiled by (Garnett et al., 2018).

Table A1.2 summarizes the spatial data sets that could be useful for valuation of the impact of forest degradation on ecosystem services, either as proxy indicators or direct measures of degradation.

Table A1.2: Remote sensing and other spatial data products for mapping forest degradation				
Data product	*Spatial extent*	*Temporal coverage*	*Source*	*Reference*
Fragmentation indicators:				
Human footprint	Global, 1 km^2	2009, 1993	Sedac.ciesin.columbia.edu/data/set/wildareas-v3-2009-human-footprint	(Venter et al., 2016)
Human modification index	Global, 1 km^2	2016	Figshare.com/articles/Global_Human_Modification/7283087	(Kennedy et al., 2019)
Low impact areas	Global, 1 km^2	?	Doi.org/10.5061/dryad.z612jm67g (broken link)	(Jacobson et al., 2019)
Road density	Global, 8 km^2	?	Www.globio.info/download-grip-dataset	(Meijer et al., 2018)
Intact landscapes	Global	2000, 2013, 2016	Http://www.intactforests.org/data.ifl.html	(Potapov et al., 2017)
Tree cover (for patch size distribution)	Global, 30m	2001 to present biannually	Https://www.globalforestwatch.org	(Hansen et al., 2013)
Species impoverishment indicators:				
Forest integrity index	Humid tropics	Circa 2010	www.unbiodiversitylab.org/	(Hansen et al., in press)
Plantations	82 countries	2015	www.globalforestwatch.org	(Harris et al., 2018)
Biodiversity intactness	Global, 1 km^2	Current?	https://data.nhm.ac.uk/dataset/global-map-of-the-biodiversity-intactness-index-from-newbold-et-al-2016-science	(Newbold et al., 2016)
Defaunation index	Tropics, 1 km^2	Input data sets span from 1980 to 2017	figshare.com/projects/Intact_but_emtpy_forests_Patterns_of_hunting-induced_mammal_defaunation_in_the_tropics/31118	(Benítez-López et al., 2019)
Loss of biomass indicators:				
Active fires	Global	Daily since 2000	Earthdata.nasa.gov/earth-observation-data/near-real-time/firms/active-fire-data	(Giglio et al., 2016)

Table A1.2: Remote sensing and other spatial data products for mapping forest degradation				
Data product	*Spatial extent*	*Temporal coverage*	*Source*	*Reference*
Burn scars (Global fire emissions data)	Global, 0.25°	Monthly from 1997 to present	Www.globalfiredata.org/data.html	(Giglio et al., 2013)
Above ground biomass	Tropics and global, 1 km²	Circa 2016 and change 2003-2014	Https://www.unbiodiversitylab.org/	(Avitabile et al., 2016) (Baccini et al., 2017)
Other:				
Indigenous and community lands	Some countries	Current	http://www.landmarkmap.org/	(Garnett et al., 2018) (on request)
Forest Health Index	Global	2019	Data will be available for download when paper is published	(Grantham et al., 2020)

Notes: Some of these data sets are available at the UN Biodiversity Lab platform at www.unbiodiversitylab.org in addition to the source provided.

Appendix 2: Data sources for tables and figures

Table A2.1: Summary of measurements of distance from edge affected by fragmentation for hydrological parameters relative to core areas of forest patches

Forest type, Location	*Effect*	*Distance from edge (m)*	*Reference*
Humid tropical			
Amazon	Decreased relative humidity	50-100	(Laurance et al., 2018)
Amazon	Decreased soil moisture	25-75	(Laurance et al., 2018)
Amazon	Altered air temperature	40-75	(Laurance et al., 2018)
Amazon	Increased air temperature	50	(Laurance et al., 2018)
Amazon	Increased vapor pressure deficit	25-50	(Laurance et al., 2018)
Amazon	Leaf conductance	10	(Laurance et al., 2018)
Amazon	Leaf relative water content	10-15	(Laurance et al., 2018)
Amazon	Decreased canopy foliage density	50	(Laurance et al., 2018)
Amazon	Increased treefall gaps	25-50	(Laurance et al., 2018)
Amazon	Decreased vapor pressure deficit	20	(Kapos, 1989)
Amazon	Lower soil moisture	10-20	(Kapos, 1989)
Amazon	Microclimate alterations	60	(Kapos, 1989)
Amazon	Desiccation	100-200	(Malcolm, 1998); (Didham and Lawton, 1999)
Amazon	Burned forest	500	(Cochrane, 2001)
Amazon	Increased fire frequency	2400	(Cochrane, 2001)
Temperate			
New Zealand	Increased air temperature	30-50	(Young and Mitchell, 1994)
New Zealand	Increased vapor pressure deficit	50-100	(Young and Mitchell, 1994)
New Zealand	Increased photosynthetically active radiation	10	(Young and Mitchell, 1994)
New Zealand	Increased soil temperature	80	(Davies-Colley et al., 2000)
New Zealand	Increased vapor pressure deficit	40	(Davies-Colley et al., 2000)
New Zealand	Increased air temperature	40	(Davies-Colley et al., 2000)
New Zealand	Increased wind speed	80	(Davies-Colley et al., 2000)
Illinois	Increased canopy openness and air temperature, decreased soil moisture and relative humidity	250	(Gehlhausen et al., 2000)
Pennsylvania and Delaware	Decreased air moisture	50	(Murcia, 1995)

Table A2.1: Summary of measurements of distance from edge affected by fragmentation for hydrological parameters relative to core areas of forest patches			
Forest type, Location	*Effect*	*Distance from edge (m)*	*Reference*
Pennsylvania and Delaware	Increased air pressure deficit	50	(Murcia, 1995)
Pennsylvania and Delaware	Increased air temperature	24	(Murcia, 1995)
Pennsylvania and Delaware	Increased photosynthetically active radiation	44	(Murcia, 1995)

Table A2.2: References for figures

References for Figure 5.5: Effect of logging on water yield

Bren & Papworth (1991); Harr (1976); Harris (1973); Hewlett & Doss, (1984); Hornbeck and others (1987); Keppeler & Ziemer (1990); Lal, (1997); Likens and others (1970); Miller and others (1988); Patric & Reinhart (1971); Pearce and others (1980); Pererira (1962); (1964); Rothacher (1970); Swift & Swank (1981); Swindel and others (1982).

References for Figure 5.6: Effect of logging on peak flow

Bren & Papworth (1991); Brown and others (1974); Goodell (1958); Harr and others (1979); Hewlett & Doss (1984); Hewlett & Helvey (1970); Kochenderfer and others (1997); Miller and others (1988); Neary (1995); Pierce and others (1970); Richardson and others (2002); Swindel and others (1983); Thomas & (1998); Verry (1972).

References for Figure 5.7: Effect of logging on stream or groundwater nitrate concentration

Adams & Stack (1989); Aubertin & Patric (1974); Bateridge (1974); Binkley & Brown (1993); Brown and others (1973); Corbett and others (1975); Dissmeyer (2000); Douglass & Swank (1975); Fredriksen (1971); Fredriksen and others (1975); Hewlett & Doss (1984); Johnston (1984); Kubin & Krecek (2009); Likens and others (1970); Pierce and others (1970); Pierce and others (1972); Richardson and others (2002); Rosén and others (1996); Snyder and others (1975); Stottlemyer & Michigan (1987); Stuart (1976); Tremblay and others (2009); Verry (1972).

References for Figure 5.8: Effect of logging on sediment concentration

Beasley (1979); Heede (1987); Hornbeck and others (1987); Leaf (1966); Nainar and others (2017); O'Loughlin & Pearce (1976); O'Loughlin and others (1980); Richardson and others (2002); Riekerk and others (1980); Swanson & Dyrness (1975); Swanson and others (1986); Van Lear and others (1985).

References for Figure 5.9: Effect of forest fire severity on mean stream flow

Carignan and others (2000); Dahm and others (2015); Neary & Tecle (2015); Richter and others (1982); Rust and others (2018); Schindler and others (1980); Smith and others (2011).

References for Figure 5.10: Effect of forest fire severity on turbidity

Alexander and others (2004); Brown (1972); Campbell and others (1977); Carignan and others (2000); Chessman (1986); Dahm and others (2015); Douglass & Van Lear (1983); Gallaher and others (2002); Gerla & Galloway (1998); Helvey (1980); Helvey and others (1985); Landsberg & Tiedemann (2000); Lane and others (2008); Leak and others (2003); Malmon and others (2007); Mast & Clow (2008); Neary (1995); Rust and others (2018); Sheridan and others (2007); Smith and others (2011); Ursic (1970); Van Lear and others (1985); Wells (1986); White and others (2015); Wilkinson and others (2006); Wright and others (1976); Wright and others (1982).

References for Figure 5.11. Effect of forest fire severity on nitrate-nitrogen

Anderson and others (1976); Campbell and others (1977); Carignan and others (2000); Chorover and others (1994); Douglass & Van Lear (1983); Fredriksen (1971); Gottfried & DeBano (1990); Hauer & Spencer (1998); Helvey (1972); Landsberg & Tiedemann (2000); Richter and others (1982); Riggan and others (1994); Rust and others (2018); Schindler and others (1980); Tarapchak & Wright (1977); Taylor and others (1993); Tiedemann and others (1978); Tiedemann (1973); Van Lear and others (1985).

References for Figure 5.12: Effect of pathogen-induced dieback on nitrate concentration

Bearup and others (2014); Beudert and others (2015); Chamier and others (2012); Clow and others (2011); Edburg and others (2012); Huber (2005); Lovett and others (2002); McCormick and others (2009); Mikkelson and others (2013); Zimmermann and others (2000).

References

Adnan, M., Hölscher, D. (2011) Medicinal plants in old-growth, degraded and re-growth forests of NW Pakistan. Forest Ecology and Management 261, 2105-2114.

Agarwala, M., DeFries, R.S., Qureshi, Q., Jhala, Y.V. (2016) Changes in the dry tropical forests in Central India with human use. Regional Environmental Change 16, 5-15.

Allan, J.D. (2004) Landscapes and riverscapes: the influence of land use on stream ecosystems. Annu. Rev. Ecol. Evol. Syst. 35, 257-284.

Andela, N., Morton, D., Giglio, L., Chen, Y., Van Der Werf, G., Kasibhatla, P., DeFries, R., Collatz, G., Hantson, S., Kloster, S. (2017) A human-driven decline in global burned area. Science 356, 1356-1362.

Astrup, R., Bernier, P.Y., Genet, H., Lutz, D.A., Bright, R.M. (2018) A sensible climate solution for the boreal forest. Nature Climate Change 8, 11.

Avitabile, V., Herold, M., Heuvelink, G.B., Lewis, S.L., Phillips, O.L., Asner, G.P., Armston, J., Ashton, P.S., Banin, L., Bayol, N. (2016) An integrated pan-tropical biomass map using multiple reference datasets. Global change biology 22, 1406-1420.

Baccini, A., Walker, W., Carvalho, L., Farina, M., Sulla-Menashe, D., Houghton, R. (2017) Tropical forests are a net carbon source based on aboveground measurements of gain and loss. Science 358, 230-234.

Baillie, B.R., Davies, T.R. (2002) Influence of large woody debris on channel morphology in native forest and pine plantation streams in the Nelson region, New Zealand. New Zealand Journal of Marine and Freshwater Research 36, 763-774.

Banerjee, A., Madhurima, C. (2013) Forest degradation and livelihood of local communities in India: A human rights approach. Journal of Horticulture and Forestry 5, 122-129.

Barbier, E.B. (2015) Valuing the storm protection service of estuarine and coastal ecosystems. Ecosystem Services 11, 32-38.

Barclay, H., Gray, C.L., Luke, S.H., Turner, E. (2017) RSPO Manual on Best Management Practices (BMPs) for the Management and Rehabilitation of Riparian Reserves. Unpublished. https://doi. org/10.13140/rg 2.

Barlow, J., Gardner, T.A., Araujo, I.S., Ávila-Pires, T.C., Bonaldo, A.B., Costa, J.E., Esposito, M.C., Ferreira, L.V., Hawes, J., Hernandez, M.I. (2007) Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. Proceedings of the National Academy of Sciences 104, 18555-18560.

Barlow, J., Lennox, G.D., Ferreira, J., Berenguer, E., Lees, A.C., Mac Nally, R., Thomson, J.R., de Barros Ferraz, S.F., Louzada, J., Oliveira, V.H.F. (2016) Anthropogenic disturbance in tropical forests can double biodiversity loss from deforestation. Nature 535, 144.

Barlow, J., Peres, C.A. (2008) Fire-mediated dieback and compositional cascade in an Amazonian forest. Philosophical Transactions of the Royal Society B: Biological Sciences 363, 1787-1794.

Barnes, A.D., Allen, K., Kreft, H., Corre, M.D., Jochum, M., Veldkamp, E., Clough, Y., Daniel, R., Darras, K., Denmead, L.H. (2017) Direct and cascading impacts of tropical land-use change on multi-trophic biodiversity. Nature ecology & evolution 1, 1511.

Benítez-López, A., Alkemade, R., Schipper, A., Ingram, D., Verweij, P., Eikelboom, J., Huijbregts, M. (2017) The impact of hunting on tropical mammal and bird populations. Science 356, 180-183.

Benítez-López, A., Santini, L., Schipper, A.M., Busana, M., Huijbregts, M.A. (2019) Intact but empty forests? Patterns of hunting-induced mammal defaunation in the tropics. PLoS biology 17, e3000247.

Bennett, E., Eves, H., Robinson, J., Wilkie, D. (2002) Why is eating bushmeat a biodiversity crisis. Conserv. Biol. Pract 3, 28-29.

Benz, R.A., Boyce, M.S., Thurfjell, H., Paton, D.G., Musiani, M., Dormann, C.F., Ciuti, S. (2016) Dispersal ecology informs design of large-scale wildlife corridors. PloS one 11, e0162989.

Bervoets, J., Boerstler, F., Borelli, S., Duma-Johansen, M., Thulstrup, A., Xia, Z. (2017) Achieving food and wood security in the context of climate change: The role of urban forests and agroforestry in the Nationally Determined Contributions in sub-Saharan Africa. Nature & Faune 39, 22.

Berzaghi, F., Longo, M., Ciais, P., Blake, S., Bretagnolle, F., Vieira, S., Scaranello, M., Scarascia-Mugnozza, G., Doughty, C.E. (2019) Carbon stocks in central African forests enhanced by elephant disturbance. Nature Geoscience, 1-5.

Beudert, B., Bässler, C., Thorn, S., Noss, R., Schröder, B., Dieffenbach-Fries, H., Foullois, N., Müller, J. (2015) Bark beetles increase biodiversity while maintaining drinking water quality. Conservation Letters 8, 272-281.

Bond, W.J., Stevens, N., Midgley, G.F., Lehmann, C.E. (2019) The trouble with trees: Afforestation plans for Africa. Trends in ecology & evolution 34, 963-965.

Bontemps, S., Defourny, P., Radoux, J., Van Bogaert, E., Lamarche, C., Achard, F., Mayaux, P., Boettcher, M., Brockmann, C., Kirches, G., (2013) Consistent global land cover maps for climate modelling communities: Current achievements of the ESA's land cover CCI, Proceedings of the ESA Living Planet Symposium, Edimburgh, pp. 9-13

Borgerson, C., Razafindrapaoly, B., Rajaona, D., Rasolofoniaina, B.J.R., Golden, C.D. (2019) Food Insecurity and the Unsustainable Hunting of Wildlife in a UNESCO World Heritage Site. Frontiers in Sustainable Food Systems 3, 99.

Brando, P.M., Balch, J.K., Nepstad, D.C., Morton, D.C., Putz, F.E., Coe, M.T., Silvério, D., Macedo, M.N., Davidson, E.A., Nóbrega, C.C. (2014) Abrupt increases in Amazonian tree mortality due to drought–fire interactions. Proceedings of the National Academy of Sciences 111, 6347-6352.

Brashares, J.S., Golden, C.D., Weinbaum, K.Z., Barrett, C.B., Okello, G.V. (2011) Economic and geographic drivers of wildlife consumption in rural Africa. Proceedings of the National Academy of Sciences 108, 13931-13936.

Broadmeadow, S., Nisbet, T. (2004) The effects of riparian forest management on the freshwater environment: a literature review of best management practice.

Brosi, B.J., Daily, G.C., Shih, T.M., Oviedo, F., Durán, G. (2008) The effects of forest fragmentation on bee communities in tropical countryside. Journal of Applied Ecology 45, 773-783.

Brown, G.W., Gahler, A.R., Marston, R.B. (1973) Nutrient losses after clear-cut logging and slash burning in the Oregon Coast Range. Water Resources Research 9, 1450-1453.

Bruijnzeel, L.A. (2004) Hydrological functions of tropical forests: not seeing the soil for the trees? Agriculture, ecosystems & environment 104, 185-228.

Burivalova, Z., Şekercioğlu, Ç.H., Koh, L.P. (2014) Thresholds of logging intensity to maintain tropical forest biodiversity. Current Biology 24, 1893-1898.

Campbell, A.J., Carvalheiro, L.G., Maués, M.M., Jaffé, R., Giannini, T.C., Freitas, M.A.B., Coelho, B.W.T., Menezes, C. (2018) Anthropogenic disturbance of tropical forests threatens pollination services to açaí palm in the Amazon river delta. Journal of Applied Ecology 55, 1725-1736.

Cao, Y., Ouyang, Z., Zheng, H., Huang, Z., Wang, X., Miao, H. (2008) Effects of forest plantations on rainfall redistribution and erosion in the red soil region of southern China. Land degradation & development 19, 321-330.

Carlson, K.M., Curran, L.M., Ponette-González, A.G., Ratnasari, D., Lisnawati, N., Purwanto, Y., Brauman, K.A., Raymond, P.A. (2014) Influence of watershed-climate interactions on stream temperature, sediment yield, and metabolism along a land use intensity gradient in Indonesian Borneo. Journal of Geophysical Research: Biogeosciences 119, 1110-1128.

Cawthorn, D.-M., Hoffman, L.C. (2015) The bushmeat and food security nexus: a global account of the contributions, conundrums and ethical collisions. Food Research International 76, 906-925.

Chaplin-Kramer, R., Ramler, I., Sharp, R., Haddad, N.M., Gerber, J.S., West, P.C., Mandle, L., Engstrom, P., Baccini, A., Sim, S. (2015) Degradation in carbon stocks near tropical forest edges. Nature Communications 6, 10158.

Chester, C.C. (2015) Yellowstone to Yukon: Transborder conservation across a vast international landscape. Environmental science & policy 49, 75-84.

Chuvieco, E., Mouillot, F., van der Werf, G.R., San Miguel, J., Tanasse, M., Koutsias, N., García, M., Yebra, M., Padilla, M., Gitas, I. (2019) Historical background and current developments for mapping burned area from satellite Earth observation. Remote Sensing of Environment 225, 45-64.

Clark, J.A., Covey, K.R. (2012) Tree species richness and the logging of natural forests: A meta-analysis. Forest Ecology and Management 276, 146-153.

Cochrane, M.A. (2001) Synergistic interactions between habitat fragmentation and fire in evergreen tropical forests. Conservation biology 15, 1515-1521.

Cochrane, M.A. (2003) Fire science for rainforests. Nature 421, 913.

Coe, M.T., Brando, P.M., Deegan, L.A., Macedo, M.N., Neill, C., Silverio, D.V. (2017) The forests of the Amazon and Cerrado moderate regional climate and are the key to the future. Tropical Conservation Science 10, 1940082917720671.

Comte, I., Colin, F., Grünberger, O., Whalen, J.K., Widodo, R.H., Caliman, J.-p. (2015) Watershed-scale assessment of oil palm cultivation impact on water quality and nutrient fluxes: a case study in Sumatra (Indonesia). Environmental Science and Pollution Research 22, 7676-7695.

Croke, J., Hairsine, P. (2006) Sediment delivery in managed forests: a review. Environmental Reviews 14, 59-87.

Davies-Colley, R.J., Payne, G., Van Elswijk, M. (2000) Microclimate gradients across a forest edge. New Zealand Journal of Ecology, 111-121.

de Souza, J.C., da Silva, R.M., Gonçalves, M.P.R., Jardim, R.J.D., Markwith, S.H. (2018) Habitat use, ranching, and human-wildlife conflict within a fragmented landscape in the Pantanal, Brazil. Biological Conservation 217, 349-357.

Debinski, D.M., Holt, R.D. (2000) A survey and overview of habitat fragmentation experiments. Conservation biology 14, 342-355.

Didham, R.K., Lawton, J.H. (1999) Edge structure determines the magnitude of changes in microclimate and vegetation structure in tropical forest fragments 1. Biotropica 31, 17-30.

Dirzo, R., Young, H.S., Galetti, M., Ceballos, G., Isaac, N.J., Collen, B. (2014) Defaunation in the Anthropocene. Science 345, 401-406.

Doxsey-Whitfield, E., MacManus, K., Adamo, S.B., Pistolesi, L., Squires, J., Borkovska, O., Baptista, S.R. (2015) Taking advantage of the improved availability of census data: a first look at the gridded population of the world, version 4. Papers in Applied Geography 1, 226-234.

Edburg, S.L., Hicke, J.A., Brooks, P.D., Pendall, E.G., Ewers, B.E., Norton, U., Gochis, D., Gutmann, E.D., Meddens, A.J. (2012) Cascading impacts of bark beetle-caused tree mortality on coupled biogeophysical and biogeochemical processes. Frontiers in Ecology and the Environment 10, 416-424.

Fa, J.E., Ryan, S.F., Bell, D.J. (2005) Hunting vulnerability, ecological characteristics and harvest rates of bushmeat species in afrotropical forests. Biological Conservation 121, 167-176.

Fagan, M., DeFries, R., Sesnie, S., Arroyo-Mora, J., Soto, C., Singh, A., Townsend, P., Chazdon, R. (2015) Mapping species composition of forests and tree plantations in Northeastern Costa Rica with an integration of hyperspectral and multitemporal Landsat imagery. Remote Sensing 7, 5660-5696.

Fagan, M., Morton, D., Cook, B., Masek, J., Zhao, F., Nelson, R., Huang, C. (2018) Mapping pine plantations in the southeastern US using structural, spectral, and temporal remote sensing data. Remote Sensing of Environment 216, 415-426.

FAO, UNEP, (2020) The state of the world's forests: Forests, biodiversity and people, Rome, Italy

Farley, K.A., Jobbágy, E.G., Jackson, R.B. (2005) Effects of afforestation on water yield: a global synthesis with implications for policy. Global change biology 11, 1565-1576.

Fisher, B., Naidoo, R., Guernier, J., Johnson, K., Mullins, D., Robinson, D., Allison, E.H. (2017) Integrating fisheries and agricultural programs for food security. Agriculture & Food Security 6, 1.

Fitzsimons, J., Pulsford, I., Wescott, G. (2013) Lessons from large-scale conservation networks in Australia. Parks 19, 115-125.

Food and Agriculture Organization, (2011) Assessing forest degradation: Towards the development of globally applicable guidelines, Forest Resource Assessment Working Paper 177, Rome

Food and Agriculture Organization, (2016a) Global forest resource assessment 2015: desk reference, Rome

Food and Agriculture Organization, (2016b) Global forests resource assessment 2015: How are the world's forests changing?, in: edition, S. (Ed.), Rome

Fredriksen, R., Moore, D., Norris, L. (1975) The impact of timber harvest, fertilization, and herbicide treatment on streamwater quality in western Oregon and Washington.

Gardner, C.J., Bicknell, J.E., Baldwin-Cantello, W., Struebig, M.J., Davies, Z.G. (2019a) Quantifying the impacts of defaunation on natural forest regeneration in a global meta-analysis. Nature Communications 10, 1-7.

Gardner, C.J., Bicknell, J.E., Balwin-Cantello, W., Struebig, M.J., Davies, Z.G. (2019b) Quantifying the impacts of defaunation on natural forest 3 regeneration worldwide. Nature Communications.

Garibaldi, L.A., Steffan-Dewenter, I., Winfree, R., Aizen, M.A., Bommarco, R., Cunningham, S.A., Kremen, C., Carvalheiro, L.G., Harder, L.D., Afik, O. (2013) Wild pollinators enhance fruit set of crops regardless of honey bee abundance. Science 339, 1608-1611.

Garnett, S.T., Burgess, N.D., Fa, J.E., Fernández-Llamazares, Á., Molnár, Z., Robinson, C.J., Watson, J.E., Zander, K.K., Austin, B., Brondizio, E.S. (2018) A spatial overview of the global importance of Indigenous lands for conservation. Nature Sustainability 1, 369.

Gehlhausen, S.M., Schwartz, M.W., Augspurger, C.K. (2000) Vegetation and microclimatic edge effects in two mixed-mesophytic forest fragments. Plant Ecology 147, 21-35.

Ghazoul, J., Burivalova, Z., Garcia-Ulloa, J., King, L.A. (2015) Conceptualizing forest degradation. Trends in ecology & evolution 30, 622-632.

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

Giglio, L., Randerson, J.T., van der Werf, G.R. (2013) Analysis of daily, monthly, and annual burned area using the fourth-generation global fire emissions database (GFED4). Journal of Geophysical Research: Biogeosciences 118, 317-328.

Giglio, L., Schroeder, W., Justice, C.O. (2016) The collection 6 MODIS active fire detection algorithm and fire products. Remote Sensing of Environment 178, 31-41.

Gilbert-Norton, L., Wilson, R., Stevens, J.R., Beard, K.H. (2010) A meta-analytic review of corridor effectiveness. Conservation biology 24, 660-668.

Golden, C.D. (2009) Bushmeat hunting and use in the Makira Forest, north-eastern Madagascar: a conservation and livelihoods issue. Oryx 43, 386-392.

Grantham, H.S., Duncan, A., Evans, T.D., Jones, K.R., Beyer, H., Schuster, R., Walston, J., Ray, J., Robinson, J., Callow, M. (2020) Only 40% of the world's forests are in good health. bioRxiv.

Grecchi, R.C., Beuchle, R., Shimabukuro, Y.E., Aragão, L.E., Arai, E., Simonetti, D., Achard, F. (2017) An integrated remote sensing and GIS approach for monitoring areas affected by selective logging: A case study in northern Mato Grosso, Brazilian Amazon. International journal of applied earth observation and geoinformation 61, 70-80.

Guariguata, M.R., Licona, J.C., Mostacedo, B., Cronkleton, P. (2009) Damage to Brazil nut trees (Bertholletia excelsa) during selective timber harvesting in Northern Bolivia. Forest Ecology and Management 258, 788-793.

Guzha, A., Rufino, M.C., Okoth, S., Jacobs, S., Nóbrega, R. (2018) Impacts of land use and land cover change on surface runoff, discharge and low flows: Evidence from East Africa. Journal of Hydrology: Regional Studies 15, 49-67.

Haddad, N.M., Brudvig, L.A., Clobert, J., Davies, K.F., Gonzalez, A., Holt, R.D., Lovejoy, T.E., Sexton, J.O., Austin, M.P., Collins, C.D. (2015) Habitat fragmentation and its lasting impact on Earth's ecosystems. Science Advances 1, e1500052.

Hallema, D.W., Sun, G., Caldwell, P.V., Norman, S.P., Cohen, E.C., Liu, Y., Bladon, K.D., McNulty, S.G. (2018) Burned forests impact water supplies. Nature Communications 9, 1307.

Hamel, P., Chaplin-Kramer, R., Sim, S., Mueller, C. (2015) A new approach to modeling the sediment retention service (InVEST 3.0): Case study of the Cape Fear catchment, North Carolina, USA. Science of the Total Environment 524, 166-177.

Hamilton, R.J., Almany, G.R., Brown, C.J., Pita, J., Peterson, N.A., Choat, J.H. (2017) Logging degrades nursery habitat for an iconic coral reef fish. Biological Conservation 210, 273-280.

Hansen, A., Barnett, K., Jantz, P., Phillips, L., Goetz, S., Hansen, M., Venter, O., Watson, J., Burns, P., Atkinson, S., S., R.-B., Ervin, J., Virnig, A., Supples, C., De Camargo, R. (in press) Global humid tropics forest structural condition and forest structural integrity maps. Scientific Data.

Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S., Tyukavina, A., Thau, D., Stehman, S., Goetz, S., Loveland, T.R. (2013) High-resolution global maps of 21st-century forest cover change. Science 342, 850-853.

Harris, N., Goldman, E., Gibbes, S., (2018) Spatial database of planted trees (SPDT) version 1.0. World Resources Institute, Washington, DC

Hodgson, J.A., Moilanen, A., Wintle, B.A., Thomas, C.D. (2011) Habitat area, quality and connectivity: striking the balance for efficient conservation. Journal of Applied Ecology 48, 148-152.

Holland, M.B., (2012) Mesoamerican biological corridor, Climate and Conservation. Springer, pp. 56-66.

Hornbeck, J., Adams, M., Corbett, E., Verry, E., Lynch, J. (1993) Long-term impacts of forest treatments on water yield: a summary for northeastern USA. Journal of Hydrology 150, 323-344.

Hosonuma, N., Herold, M., De Sy, V., De Fries, R.S., Brockhaus, M., Verchot, L., Angelsen, A., Romijn, E. (2012) An assessment of deforestation and forest degradation drivers in developing countries. Environmental Research Letters 7, 044009.

Hua, F., Wang, X., Zheng, X., Fisher, B., Wang, L., Zhu, J., Tang, Y., Douglas, W.Y., Wilcove, D.S. (2016) Opportunities for biodiversity gains under the world's largest reforestation programme. Nature Communications 7, 12717.

Huber, C. (2005) Long lasting nitrate leaching after bark beetle attack in the highlands of the Bavarian Forest National Park. Journal of environmental quality 34, 1772-1779.

ITTO, (2002) ITTO guidelines for the restoration, management and rehabilitation of degraded and secondary tropical forests, ITTO Policy Development Series No 13. International Tropical Timber Organization

Jackson, R.B., Jobbágy, E.G., Avissar, R., Roy, S.B., Barrett, D.J., Cook, C.W., Farley, K.A., Le Maitre, D.C., McCarl, B.A., Murray, B.C. (2005) Trading water for carbon with biological carbon sequestration. Science 310, 1944-1947.

Jacobson, A.P., Riggio, J., Tait, A.M., Baillie, J.E. (2019) Global areas of low human impact ('Low Impact Areas') and fragmentation of the natural world. Scientific Reports 9, 1-13.

Jobbágy, E.G., Baldi, G., Nosetto, M.D., (2011) Tree plantation in South America and the water cycle: impacts and emergent opportunities, Forests in Development: A vital balance. Springer, pp. 53-63.

Johnson, M., Kellermann, J., Stercho, A. (2010) Pest reduction services by birds in shade and sun coffee in Jamaica. Animal conservation 13, 140-147.

Kagawa, A., Sack, L., Duarte, K.e., James, S. (2009) Hawaiian native forest conserves water relative to timber plantation: species and stand traits influence water use. Ecological Applications 19, 1429-1443.

Kapos, V. (1989) Effects of isolation on the water status of forest patches in the Brazilian Amazon. Journal of tropical ecology 5, 173-185.

Kennedy, C.M., Oakleaf, J.R., Theobald, D.M., Baruch-Mordo, S., Kiesecker, J. (2019) Managing the middle: A shift in conservation priorities based on the global human modification gradient. Global Change Biology 25, 811-826.

Komatsu, H., Kume, T., Otsuki, K. (2008) The effect of converting a native broad-leaved forest to a coniferous plantation forest on annual water yield: a paired-catchment study in northern Japan. Forest Ecology and Management 255, 880-886.

Koplitz, S.N., Mickley, L.J., Marlier, M.E., Buonocore, J.J., Kim, P.S., Liu, T., Sulprizio, M.P., DeFries, R.S., Jacob, D.J., Schwartz, J. (2016) Public health impacts of the severe haze in Equatorial Asia in September–October 2015: demonstration of a new framework for informing fire management strategies to reduce downwind smoke exposure. Environmental Research Letters 11, 094023.

Krause, T., Nielsen, M.R. (2019) Not Seeing the Forest for the Trees: The Oversight of Defaunation in REDD+ and Global Forest Governance. Forests 10, 344.

Krishnan, S., Kushalappa, C.G., Shaanker, R.U., Ghazoul, J. (2012) Status of pollinators and their efficiency in coffee fruit set in a fragmented landscape mosaic in South India. Basic and Applied Ecology 13, 277-285.

Krishnaswamy, J., Bonell, M., Venkatesh, B., Purandara, B.K., Lele, S., Kiran, M., Reddy, V., Badiger, S., Rakesh, K. (2012) The rain–runoff response of tropical humid forest ecosystems to use and reforestation in the Western Ghats of India. Journal of Hydrology 472, 216-237.

Krishnaswamy, J., Bonell, M., Venkatesh, B., Purandara, B.K., Rakesh, K., Lele, S., Kiran, M., Reddy, V., Badiger, S. (2013) The groundwater recharge response and hydrologic services of tropical humid forest ecosystems to use and reforestation: Support for the "infiltration-evapotranspiration trade-off hypothesis". Journal of Hydrology 498, 191-209.

Kurten, E.L. (2013) Cascading effects of contemporaneous defaunation on tropical forest communities. Biological Conservation 163, 22-32.

Lange, G.-M., Wodon, Q., Carey, K., (2018) The Changing Wealth of Nations: Building a Sustainable Future. International Bank for Reconstruction and Development/The World Bank, Washington, D.C.

Laurance, W.F. (2018) Conservation and the global infrastructure tsunami: Disclose, debate, delay! Trends in ecology & evolution 33, 568-571.

Laurance, W.F., Camargo, J.L., Fearnside, P.M., Lovejoy, T.E., Williamson, G.B., Mesquita, R.C., Meyer, C.F., Bobrowiec, P.E., Laurance, S.G. (2018) An Amazonian rainforest and its fragments as a laboratory of global change. Biological Reviews 93, 223-247.

Laurance, W.F., Clements, G.R., Sloan, S., O'connell, C.S., Mueller, N.D., Goosem, M., Venter, O., Edwards, D.P., Phalan, B., Balmford, A. (2014) A global strategy for road building. Nature 513, 229.

Letcher, S.G., Chazdon, R.L. (2009) Rapid recovery of biomass, species richness, and species composition in a forest chronosequence in northeastern Costa Rica. Biotropica 41, 608-617.

Lewis, S.L., Edwards, D.P., Galbraith, D. (2015) Increasing human dominance of tropical forests. Science 349, 827-832.

Lewis, S.L., Wheeler, C.E., Mitchard, E.T., Koch, A., (2019) Restoring natural forests is the best way to remove atmospheric carbon. Nature Publishing Group

Licata, J.A., Gyenge, J.E., Fernández, M.E., Schlichter, T.M., Bond, B.J. (2008) Increased water use by ponderosa pine plantations in northwestern Patagonia, Argentina compared with native forest vegetation. Forest Ecology and Management 255, 753-764.

Little Cárdenas, C.L., Cuevas, J.G., Lara Aguilar, A., Pino, M., Schoenholtz, S.H. (2015) Buffer effects of streamside native forests on water provision in watersheds dominated by exotic forest plantations.

Locatelli, B., Vignola, R. (2009) Managing watershed services of tropical forests and plantations: can meta-analyses help? Forest Ecology and Management 258, 1864-1870.

Malcolm, J.R. (1998) A model of conductive heat flow in forest edges and fragmented landscapes. Climatic Change 39, 487-502.

Malhi, Y., Gardner, T.A., Goldsmith, G.R., Silman, M.R., Zelazowski, P. (2014) Tropical forests in the Anthropocene. Annual Review of Environment and Resources 39.

Martin, D.A. (2016) At the nexus of fire, water and society. Philosophical Transactions of the Royal Society B: Biological Sciences 371, 20150172.

Martin, P.A., Newton, A.C., Pfeifer, M., Khoo, M., Bullock, J.M. (2015) Impacts of tropical selective logging on carbon storage and tree species richness: A meta-analysis. Forest Ecology and Management 356, 224-233.

McClure, C.D., Jaffe, D.A. (2018) US particulate matter air quality improves except in wildfire-prone areas. Proceedings of the National Academy of Sciences 115, 7901-7906.

McDonald, R.I., Weber, K.F., Padowski, J., Boucher, T., Shemie, D. (2016) Estimating watershed degradation over the last century and its impact on water-treatment costs for the world's large cities. Proceedings of the National Academy of Sciences 113, 9117-9122.

McIntyre, P.B., Liermann, C.A.R., Revenga, C. (2016) Linking freshwater fishery management to global food security and biodiversity conservation. Proceedings of the National Academy of Sciences 113, 12880-12885.

Meijer, J.R., Huijbregts, M.A., Schotten, K.C., Schipper, A.M. (2018) Global patterns of current and future road infrastructure. Environmental Research Letters 13, 064006.

Menton, M. (2003) Effects of logging on non-timber forest product extraction in the Brazilian Amazon: community perceptions of change. International Forestry Review 5, 97-105.

Mitchell, A.L., Rosenqvist, A., Mora, B. (2017) Current remote sensing approaches to monitoring forest degradation in support of countries measurement, reporting and verification (MRV) systems for REDD+. Carbon balance and management 12, 9.

Murcia, C. (1995) Edge effects in fragmented forests: implications for conservation. Trends in ecology & evolution 10, 58-62.

Neary, D., Tecle, A. (2015) Water quality impacts of forest fires. J. Pollut. Eff. Control 3.

Newbold, T., Hudson, L.N., Arnell, A.P., Contu, S., De Palma, A., Ferrier, S., Hill, S.L., Hoskins, A.J., Lysenko, I., Phillips, H.R. (2016) Has land use pushed terrestrial biodiversity beyond the planetary boundary? A global assessment. Science 353, 288-291.

Olson, D.M., Dinerstein, E., Wikramanayake, E.D., Burgess, N.D., Powell, G.V., Underwood, E.C., D'amico, J.A., Itoua, I., Strand, H.E., Morrison, J.C. (2001) Terrestrial Ecoregions of the World: A New Map of Life on EarthA new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity. BioScience 51, 933-938.

Osuri, A.M., Ratnam, J., Varma, V., Alvarez-Loayza, P., Astaiza, J.H., Bradford, M., Fletcher, C., Ndoundou-Hockemba, M., Jansen, P.A., Kenfack, D. (2016) Contrasting effects of defaunation on aboveground carbon storage across the global tropics. Nature Communications 7, 11351.

Payn, T., Carnus, J.-M., Freer-Smith, P., Kimberley, M., Kollert, W., Liu, S., Orazio, C., Rodriguez, L., Silva, L.N., Wingfield, M.J. (2015) Changes in planted forests and future global implications. Forest Ecology and Management 352, 57-67.

Perry, T.D., Jones, J.A. (2017) Summer streamflow deficits from regenerating Douglas-fir forest in the Pacific Northwest, USA. Ecohydrology 10, e1790.

Potapov, P., Hansen, M.C., Laestadius, L., Turubanova, S., Yaroshenko, A., Thies, C., Smith, W., Zhuravleva, I., Komarova, A., Minnemeyer, S. (2017) The last frontiers of wilderness: Tracking loss of intact forest landscapes from 2000 to 2013. Science Advances 3, e1600821.

Puyravaud, J.P., Davidar, P., Laurance, W.F. (2010) Cryptic destruction of India's native forests. Conservation Letters 3, 390-394.

Quinn, J., Ritter, E. (2003) Effects of land use and pine forest logging on stream nutrients at Purukohukohu, Central North Island. Proceedings and Report. Rotorua Lakes, 149-157.

Rader, R., Bartomeus, I., Garibaldi, L.A., Garratt, M.P., Howlett, B.G., Winfree, R., Cunningham, S.A., Mayfield, M.M., Arthur, A.D., Andersson, G.K. (2016) Non-bee insects are important contributors to global crop pollination. Proceedings of the National Academy of Sciences 113, 146-151.

Resasco, J. (2019) Meta-analysis on a Decade of Testing Corridor Efficacy: What New Have we Learned? Current Landscape Ecology Reports, 1-9.

Resasco, J., Bruna, E.M., Haddad, N.M., Banks-Leite, C., Margules, C.R. (2017) The contribution of theory and experiments to conservation in fragmented landscapes. Ecography 40, 109-118.

Ricketts, T.H., Daily, G.C., Ehrlich, P.R., Michener, C.D. (2004) Economic value of tropical forest to coffee production. Proceedings of the National Academy of Sciences 101, 12579-12582.

Rifai, S.W., Li, S., Malhi, Y. (2019) Coupling of El Niño events and long-term warming leads to pervasive climate extremes in the terrestrial tropics. Environmental Research Letters 14, 105002.

Ripple, W.J., Abernethy, K., Betts, M.G., Chapron, G., Dirzo, R., Galetti, M., Levi, T., Lindsey, P.A., Macdonald, D.W., Machovina, B. (2016) Bushmeat hunting and extinction risk to the world's mammals. Royal Society open science 3, 160498.

Rist, L., Shanley, P., Sunderland, T., Sheil, D., Ndoye, O., Liswanti, N., Tieguhong, J. (2012) The impacts of selective logging on non-timber forest products of livelihood importance. Forest Ecology and Management 268, 57-69.

Riutta, T., Slade, E.M., Morecroft, M.D., Bebber, D.P., Malhi, Y. (2014) Living on the edge: quantifying the structure of a fragmented forest landscape in England. Landscape Ecology 29, 949-961.

Roa-García, M., Brown, S., Schreier, H., Lavkulich, L. (2011) The role of land use and soils in regulating water flow in small headwater catchments of the Andes. Water Resources Research 47.

Robichaud, P.R., Beyers, J.L., Neary, D.G. (2000) Evaluating the effectiveness of postfire rehabilitation treatments. Gen. Tech. Rep. RMRS-GTR-63. Fort Collins: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. 85 p. 63.

Robinson, T.P., Wint, G.W., Conchedda, G., Van Boeckel, T.P., Ercoli, V., Palamara, E., Cinardi, G., D'Aietti, L., Hay, S.I., Gilbert, M. (2014) Mapping the global distribution of livestock. PloS one 9, e96084.

Salati, E., Lovejoy, T., Vose, P. (1983) Precipitation and water recycling in tropical rain forests with special reference to the amazon basim. Environmentalist 3, 67-72.

Sarti, F., Adams, C., Morsello, C., Van Vliet, N., Schor, T., Yagüe, B., Tellez, L., Quiceno-Mesa, M.P., Cruz, D. (2015) Beyond protein intake: bushmeat as source of micronutrients in the Amazon. Ecology and Society 20.

Scheffers, B.R., Oliveira, B.F., Lamb, I., Edwards, D.P. (2019) Global wildlife trade across the tree of life. Science 366, 71-76.

Schneemann, J. (1995) Exploitation of Moabi in the humid dense forests of Cameroon: harmonization and improvement of two conflicting ways of exploitation of the same forest resource. BOS Nieuwsletter 31, 2.

Seidl, A.F., de Silva, J.d.S.V., Moraes, A.S. (2001) Cattle ranching and deforestation in the Brazilian Pantanal. Ecological Economics 36, 413-425.

Şekercioğlu, Ç.H., Ehrlich, P.R., Daily, G.C., Aygen, D., Goehring, D., Sandí, R.F. (2002) Disappearance of insectivorous birds from tropical forest fragments. Proceedings of the National Academy of Sciences 99, 263-267.

Shanley, P., Luz, L. (2003) The impacts of forest degradation on medicinal plant use and implications for health care in eastern Amazonia. BioScience 53, 573-584.

Shanley, P., Luz, L., Cymerys, M. (2002) Subsistence issues: The interface of timber and non-timber resources: declining resources for subsistence livelihoods (Brazil). Tapping the green market: certification and management of non-timber forest products. Earthscan, London, UK, 313-321.

Sidle, R.C., Sasaki, S., Otsuki, M., Noguchi, S., Rahim Nik, A. (2004) Sediment pathways in a tropical forest: effects of logging roads and skid trails. Hydrological processes 18, 703-720.

Sikka, A., Samra, J., Sharda, V., Samraj, P., Lakshmanan, V. (2003) Low flow and high flow responses to converting natural grassland into bluegum (Eucalyptus globulus) in Nilgiris watersheds of South India. Journal of Hydrology 270, 12-26.

Silveira, L., Sollmann, R., Jácomo, A.T., Diniz Filho, J.A., Tôrres, N.M. (2014) The potential for large-scale wildlife corridors between protected areas in Brazil using the jaguar as a model species. Landscape Ecology 29, 1213-1223.

Soares-Filho, B., Rajão, R., Macedo, M., Carneiro, A., Costa, W., Coe, M., Rodrigues, H., Alencar, A. (2014) Cracking Brazil's forest code. Science 344, 363-364.

Souza Jr, C., Siqueira, J., Sales, M., Fonseca, A., Ribeiro, J., Numata, I., Cochrane, M., Barber, C., Roberts, D., Barlow, J. (2013) Ten-year Landsat classification of deforestation and forest degradation in the Brazilian Amazon. Remote Sensing 5, 5493-5513.

Staver, A.C., Brando, P.M., Barlow, J., Morton, D.C., Paine, C.T., Malhi, Y., Murakami, A.A., del Aguila Pasquel, J. (2019) Thinner bark increases sensitivity of wetter Amazonian tropical forests to fire. Ecology letters.

Swank, W., Waide, J., Crossley, D., Todd, R. (1981) Insect defoliation enhances nitrate export from forest ecosystems. Oecologia 51, 297-299.

Tabacchi, E., Lambs, L., Guilloy, H., Planty-Tabacchi, A.M., Muller, E., Decamps, H. (2000) Impacts of riparian vegetation on hydrological processes. Hydrological processes 14, 2959-2976.

Tabarelli, M., Lopes, A.V., Peres, C.A. (2008) Edge-effects drive tropical forest fragments towards an early-successional system. Biotropica 40, 657-661.

Tabarelli, M., Peres, C.A., Melo, F.P. (2012) The 'few winners and many losers' paradigm revisited: emerging prospects for tropical forest biodiversity. Biological Conservation 155, 136-140.

Taki, H., Kevan, P.G., Ascher, J.S. (2007) Landscape effects of forest loss in a pollination system. Landscape Ecology 22, 1575-1587.

Thompson, I.D., Guariguata, M.R., Okabe, K., Bahamondez, C., Nasi, R., Heymell, V., Sabogal, C. (2013) An operational framework for defining and monitoring forest degradation. Ecology & Society 18.

Torres, P.C., Morsello, C., Parry, L., Barlow, J., Ferreira, J., Gardner, T., Pardini, R. (2018) Landscape correlates of bushmeat consumption and hunting in a post-frontier Amazonian region. Environmental Conservation 45, 315-323.

Tortato, F.R., Izzo, T.J. (2017) Advances and barriers to the development of jaguar-tourism in the Brazilian Pantanal. Perspectives in Ecology and Conservation 15, 61-63.

Tortato, F.R., Izzo, T.J., Hoogesteijn, R., Peres, C.A. (2017) The numbers of the beast: Valuation of jaguar (Panthera onca) tourism and cattle depredation in the Brazilian Pantanal. Global ecology and conservation 11, 106-114.

Trumbore, S., Brando, P., Hartmann, H. (2015) Forest health and global change. Science 349, 814-818.

Tsai, C.-J., Lin, T.-C., Hwong, J.-L., Lin, N.-H., Wang, C.-P., Hamburg, S. (2009) Typhoon impacts on stream water chemistry in a plantation and an adjacent natural forest in central Taiwan. Journal of Hydrology 378, 290-298.

Twining-Ward, L., Li, W., Bhammar, H., Wright, E., (2018) Supporting sustainable livelihoods through wildlife tourism. World Bank

Valdez, F.P., Haag, T., Azevedo, F.C., Silveira, L., Cavalcanti, S.M., Salzano, F.M., Eizirik, E. (2015) Population genetics of jaguars (Panthera onca) in the Brazilian Pantanal: molecular evidence for demographic connectivity on a regional scale. Journal of Heredity 106, 503-511.

Van Vliet, N., Moreno Calderón, J.L., Gomez, J., Zhou, W., Fa, J.E., Golden, C., Nobrega Alves, R.R., Nasi, R. (2017) Bushmeat and human health: assessing the evidence in tropical and sub-tropical forests.

Venter, O., Sanderson, E.W., Magrach, A., Allan, J.R., Beher, J., Jones, K.R., Possingham, H.P., Laurance, W.F., Wood, P., Fekete, B.M. (2016) Sixteen years of change in the global terrestrial human footprint and implications for biodiversity conservation. Nature Communications 7, 12558.

Vitousek, P.M., Gosz, J.R., Grier, C.C., Melillo, J.M., Reiners, W.A., Todd, R.L. (1979) Nitrate losses from disturbed ecosystems. Science 204, 469-474.

Waterloo, M., Schellekens, J., Bruijnzeel, L., Rawaqa, T. (2007) Changes in catchment runoff after harvesting and burning of a Pinus caribaea plantation in Viti Levu, Fiji. Forest Ecology and Management 251, 31-44.

Wei, X., Liu, S., Zhou, G., Wang, C. (2005) Hydrological processes in major types of Chinese forest. Hydrological Processes: An International Journal 19, 63-75.

Wenger, A.S., Atkinson, S., Santini, T., Falinski, K., Hutley, N., Albert, S., Horning, N., Watson, J.E., Mumby, P.J., Jupiter, S.D. (2018) Predicting the impact of logging activities on soil erosion and water quality in steep, forested tropical islands. Environmental Research Letters 13, 044035.

Wenjie, L., Wenyao, L., Hongjian, L., Wenping, D., Hongmei, L. (2011) Runoff generation in small catchments under a native rain forest and a rubber plantation in Xishuangbanna, southwestern China. Water and Environment Journal 25, 138-147.

Wikramanayake, E., Manandhar, A., Bajimaya, S., Nepal, S., Thapa, G., Thapa, K., (2010) The Terai Arc Landscape: A tiger conservation success story in a human-dominated landscape, Tigers of the World. Elsevier, pp. 163-173.

World Health Organization, (2003) Nitrate and nitrite in drinking-water: Background document for development of WHO Guidelines for Drinking-water Quality. World Health Organization

Young, A., Mitchell, N. (1994) Microclimate and vegetation edge effects in a fragmented podocarp-broadleaf forest in New Zealand. Biological Conservation 67, 63-72.

Young, H.S., McCauley, D.J., Galetti, M., Dirzo, R. (2016) Patterns, causes, and consequences of anthropocene defaunation. Annual Review of Ecology, Evolution, and Systematics 47, 333-358.

Zhang, M., Liu, N., Harper, R., Li, Q., Liu, K., Wei, X., Ning, D., Hou, Y., Liu, S. (2017) A global review on hydrological responses to forest change across multiple spatial scales: Importance of scale, climate, forest type and hydrological regime. Journal of Hydrology 546, 44-59.

Zimmermann, L., Moritz, K., Kennel, M., Bittersohl, J. (2000) Influence of bark beetle infestation on water quantity and quality in the Grosse Ohe catchment(Bavarian Forest National Park). Silva Gabreta 4, 51-62.