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Using ecosystem integrity to maximize climate mitigation and minimize risk in international forest policy

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Several key international policy frameworks involve forests, including the Paris Agreement on Climate Change and the Convention on Biological Diversity (CBD). However, rules and guidelines that treat forest types equally regardless of their ecosystem integrity and risk profiles in terms of forest and carbon loss limit policy effectiveness and can facilitate forest degradation. Here we assess the potential for using a framework of ecosystem integrity to guide policy goals. We review the theory and present a conceptual framework, compare elements of integrity between primary and human-modified forests, and discuss the policy and management implications. We find that primary forests consistently have higher levels of ecosystem integrity and lower risk profiles than human-modified forests. This underscores the need to protect primary forests, develop consistent large-scale data products to identify high-integrity forests, and operationalize a framework of ecosystem integrity. Doing so will optimize long-term carbon storage and the provision of other ecosystem services, and can help guide evolving forest policy at the nexus of the biodiversity and climate crises.

KEYWORDS

Paris Agreement, primary forest, carbon, forest degradation, deforestation

Introduction

Forest ecosystems are central to international agreements and frameworks that support and set policy agendas, including the United Nations (UN) Framework Convention on Climate Change (UNFCCC), Convention on Biological Diversity (CBD), Sustainable Development Goals (SDGs), and Convention to Combat Desertification (UNCCD). Forests and their ecosystem services provide critical data to inform global environmental assessments such as the Global Forest Resource Assessments (FRAs) of the UN Food and Agriculture Organization (FAO), the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), the System of Environmental Economic Accounting–Ecosystem Accounting (SEEA-EA), and the World Bank's reports on the Changing Wealth of Nations (Lange et al., 2018). The mitigation significance of forests is recognized in Article 5 of the Paris Agreement. Given their mitigation value, updating forest management practices to reduce emissions and increase withdrawals from the atmosphere should be included in many countries' Nationally Determined Contributions (NDCs; Forsell et al., 2016; Grassi et al., 2017; Roe et al., 2019). Forestry practices have the potential to provide a majority fraction of the Agriculture, Forestry, and Other Land Use (AFOLU) sector's contributions to climate mitigation, which may represent up to one-third of net emission reductions needed to limit warming below 1.5–2°C above pre-industrial levels (Federici et al., 2017; Grassi et al., 2017; Griscom et al., 2017; Roe et al., 2019). The current emissions gap between NDCs and what is required to limit warming to 1.5 or 2°C (UNEP, 2019) means that the role of forests may be even greater; for example, forests are referenced heavily in the Intergovernmental Panel on Climate Change (IPCC) special report on 1.5°C in the context of negative emissions (Dooley et al., 2018; IPCC, 2018).

However, given the finite area of available land and the many ecosystem services they provide, there are often conflicting goals for the management of forests in national and international policy contexts, resulting in incoherent policies and policy objectives (Kalaba et al., 2014; Koff et al., 2016; Tegegne et al., 2018; Timko et al., 2018). For example, many of the UN SDGs focused on promoting economic development are at odds with conserving forests and biodiversity (Ibisch et al., 2016). Unclear and inconsistent definitions and accounting rules mean that forest mitigation measures can have a range of results from large-scale protection that preserves carbon storage, sequestration, and ecosystem services, to perverse outcomes with net carbon loss, degraded ecosystems, and negative impacts on other policy goals (Mackey et al., 2013). For example, bioenergy with carbon capture and storage (BECCS) is used in the majority of current global socioeconomic model scenarios to stay below 1.5–2°C of warming (Roe et al., 2019). At these scales, BECCS will require the conversion of vast quantities of native forests into tree plantations or short-rotation forests (Fuss et al., 2014; Creutzig et al., 2015; Smith et al., 2016; IPCC, 2018). Increased bioenergy use is currently resulting in forest degradation and deforestation that will generate net carbon emissions for decades or longer (Birdsey et al., 2018; Booth, 2018; Sterman et al., 2022). Part of the problem is that forest cover and types are largely seen as fungible within the UNFCCC guidelines (UNFCCC, 2002), with no criteria for forest condition or carbon longevity (Ajani et al., 2013; Hansen A. J. et al., 2020; Keith et al., 2021).

From a carbon perspective, "risk of loss" of the stock is of central importance. The risk of loss from disturbances means that some land-based carbon activities will not provide long-term protection of carbon from release into the atmosphere (e.g., Anderegg et al., 2020). This risk is a primary reason that forest-based solutions are often not considered as reliable ways to reduce net emissions and hence are not prioritized as mitigation activities (Grassi et al., 2017). Yet little consideration has been given to differentiating forest types and management schemes based on their "risk of loss" profiles. The Paris Agreement mentions criteria for mitigation that speak to risk, such as equity, sustainability, and integrity, but as of yet there is little guidance on implementation.

The concept of "ecosystem integrity," or related "ecological integrity," has a long history in theoretical and applied ecology (e.g., Kay, 1991; Tierney et al., 2009; Wurtzebach and Schultz, 2016) and is explicitly referenced [e.g., Paris Agreement, CBD post-2020 Global Biodiversity Framework (Convention on Biological Diversity [CBD], 2021), IPCC Working Group II (IPCC, 2022)] or implied in international agreements and national-level legislation and agency directives (e.g., Australian Government, 1999). By providing a holistic view of ecosystem structure, function, composition, and adaptive capacity, the objective of maximizing ecosystem integrity may have the potential to minimize risk of carbon loss and maximize the ecosystem services provided by forests, thereby facilitating greater policy coherence across sectors (Koff et al., 2016; Dooley et al., 2018; Barber et al., 2020). However, the concept is not prioritized in international policy nor operationalized in most national forest policies, thus falling well short of its potential. There are no specific actions or supporting mechanisms for ecosystem integrity in the Paris Agreement, and parties have not articulated how they will identify and protect high-integrity ecosystems. Instead of representing a guiding framework, ecosystem integrity is largely viewed as a potential co-benefit (Bryan et al., 2016; Funk et al., 2019). Particularly important is providing a definition and framework for ecosystem integrity that the CBD (though the Global Biodiversity Framework) and the UNFCCC (through the Global Stocktake) can utilize to achieve their biodiversity and climate mitigation objectives.

Here we review the potential for a framework of ecosystem integrity to minimize risk in forest-based mitigation policies and maximize ecosystem service co-benefits. We first discuss the theory of ecosystem integrity and provide a working conceptual framework. We then compare important elements of ecosystem integrity between primary and human-modified forests, with a focus on elements most relevant for carbon mitigation including risk profiles. Finally, we discuss the policy and management implications of this comparative analysis. By drawing on ecological theory and several sub-disciplines within ecology, we integrate knowledge into a coherent framework of ecosystem integrity (**Figure 1**) that can be used to guide both forest policy at the international level as well as implementation in the form of land use decisions, metrics, and priorities at the national and jurisdictional levels. Our review draws upon decades of evolving forest policy and published literature, including but not limited to peer-reviewed articles, as well as engagement with stakeholders, practitioners, policy makers, and forest ecologists.

Framework for forest ecosystem integrity

Definition

Many definitions of ecosystem integrity exist because ecosystem integrity is not a simple absolute physical property but rather a multidimensional and scale-dependent emergent phenomenon that encompasses important system components and their interactions. The concept has received considerable attention over the past several decades because of the human benefits derived from natural processes and ecosystem states. As noted by Muller et al. (2000), "ecosystem integrity turns out to be the ecological branch of sustainability."

Here we adopt and build upon the general framework originally provided by Kay (1991), whereby ecosystem integrity *integrates different characteristics of an ecosystem that collectively describe its ability to achieve and maintain its optimum operating state, given the prevailing environmental drivers and perturbations, and continue its processes of self-organization and regeneration (i.e., autopoiesis)*. One of the main theoretical divides about ecosystem integrity relates to differentiating compositional (e.g., species richness, genetic diversity, or presence of threatened species), structural (e.g., vegetation density, biomass, food chains, and trophic levels) or functional (e.g., productivity, energy flows, and nutrient cycling) aspects of integrity (De Leo and Levin, 1997; Pimentel et al., 2013; Roche and Campagne, 2017). We suggest these are largely inseparable given the fundamental importance of structural and compositional elements in supporting functional forest ecosystem integrity and the many interdependencies among composition, structure, and function. In practice, available data and resources will determine what can be measured at a particular spatial and temporal scale. Because ecosystem integrity includes the provision of ecosystem services for human benefit, its evaluation typically includes a human dimension (Kay, 1991; De Leo and Levin, 1997; Kay and Regier, 2000; Dorren et al., 2004; Roche and Campagne, 2017).

Components of ecosystem integrity

Based on decades of theoretical and applied studies, we provide a framework for understanding the components of forest ecosystem integrity, their drivers, and their inter-linkages (**Figure 1**). It is important to note that all elements of ecosystem integrity are affected by the prevailing environmental and site characteristics of a given forested location, which must be accounted for when comparing specific locations in space and/or time.

Foundational elements

Forest ecosystem integrity is based on physiological structures that efficiently use and dissipate energy (**Figure 1**). These dissipative structures, or "ecological orientors" (Muller et al., 2000), generate a gradient of energy degradation *via* metabolic reactions that create and maintain themselves (i.e., self-organization). Progressively accumulated exergy (i.e., available energy) becomes stored emergy (i.e., all the energy used to generate a product or service) (Campbell, 2000; Kay and Regier, 2000; Muller et al., 2000). Over the course of evolution, community assembly, and forest succession, this process generates optimized (generally high but not too high; Hengeveld, 1989; May, 2001) ecosystem complexity and distance from thermodynamic equilibrium (Odum, 1969; Kay, 1991; Holling, 1992; Campbell, 2000; Muller et al., 2000), with associated levels of structural complexity, functional diversity, and niche complementarity (Tilman, 1996; Tilman and Lehman, 2001; Thompson et al., 2009). Ecosystem processes that sustain and regulate this self-organizing system, such as productivity, evapotranspiration, reproduction cycles, and nutrient cycling and retention, are optimized in the process (Muller et al., 2000; Dorren et al., 2004; Migliavacca et al., 2021). The resulting forest is a non-linear, self-organizing, holarchic and open system, with reciprocal power relationships between levels (Kay and Regier, 2000).

A critical property of ecosystem integrity that is difficult to assess from structural or compositional elements alone is stability. Following Grimm and Wissel (1997), stability is comprised of resistance (or constancy), resilience, and persistence, which collectively represent an ecosystem's ability to resist or be resilient to change at both short and long time scales (Kay, 1991, 1993; Regier, 1993; Muller, 1998; Kay and Regier, 2000; Andreasen et al., 2001; Parrish et al., 2003). In the case of forest ecosystem integrity, primary drivers of change (exposure) include human land use and other human pressures, and climate change including extreme weather events and increasing disturbances. Resistance indicates a forest's ability to maintain stability *via* dynamic equilibrium within defined ecosystem

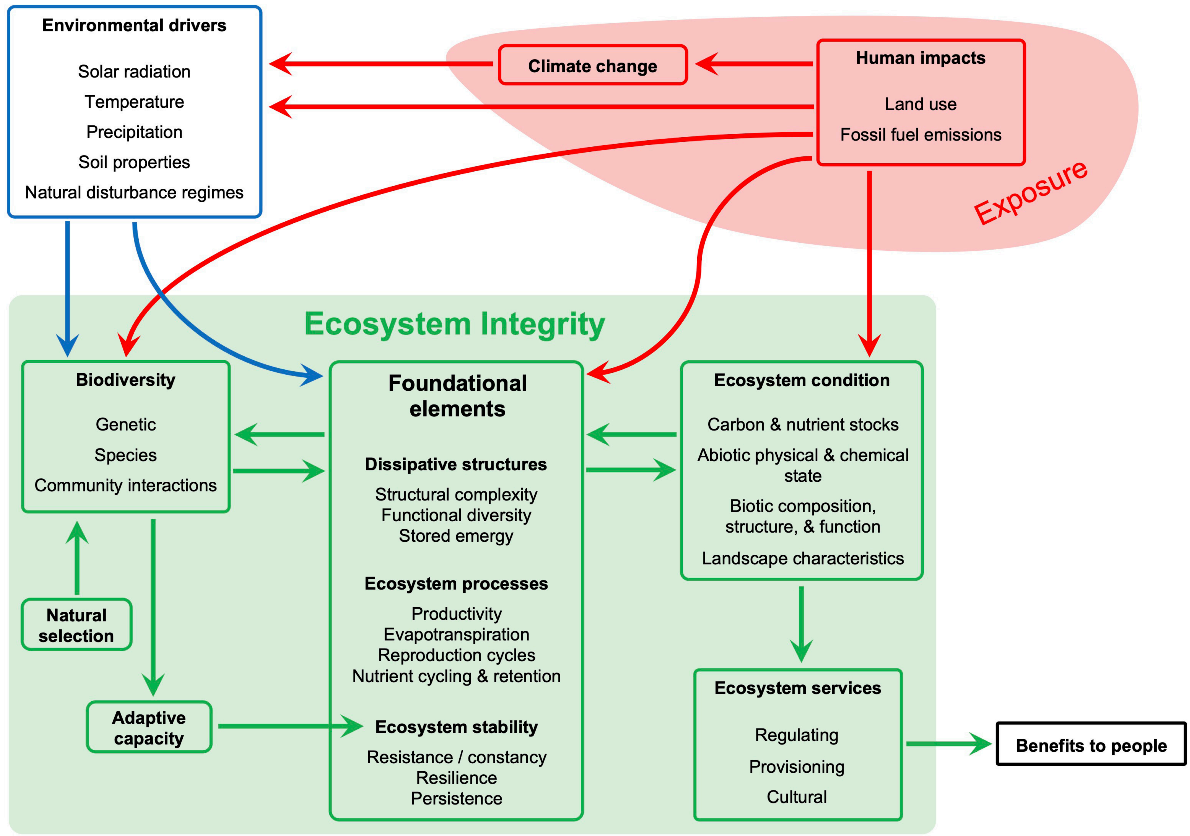

FIGURE 1
Conceptual framework of ecosystem integrity. Integrity is based on foundational elements including dissipative structures, ecosystem processes, and ecosystem stability. These are underpinned by biodiversity, natural selection, and adaptive capacity, and in turn generate a given ecosystem condition and benefits to people. Ecosystem integrity is impacted by environmental drivers and human impacts, including land use and climate change.

bounds (Hughes et al., 2002; Loreau et al., 2002) in response to these drivers. Forest resistance is conferred by negative feedbacks and buffers, for example stable microhabitats in forest interiors and functional redundancy across species. Resilience indicates the ability to return to optimal operating conditions after a state-altering perturbation (Holling, 1973; Kay, 1991; Kay and Regier, 2000; Muller et al., 2000; Thompson et al., 2009). The resulting ecosystem state can be somewhat altered (i.e., "ecological resilience" as opposed to "engineering resilience"), but when viewed over an appropriate time span, a resilient forest is able to maintain its "identity" in terms of taxonomic composition, structure, ecological functions, and process rates–and hence exhibit persistence (Thompson et al., 2009). Forest resilience is generally conferred by regenerative capacity *via* biological legacies (Franklin et al., 2000; Lindenmayer et al., 2019). These components of stability are supported by an ecosystem's adaptive capacity, or the capacity for adaptive change in response to new conditions (Angeler et al., 2019). For example, genetic diversity, species diversity, and phenotypic plasticity allow for varied and time-evolving expression of adaptive traits and species within an ecosystem in response to changing environmental conditions, disturbances, or other pressures (Savolainen et al., 2007; Reed et al., 2011; Rogers et al., 2017). Hence, adaptive capacity is supported by biodiversity (**Figure 1**).

Biodiversity

These foundational elements of integrity are derivatives of the underlying biodiversity of a forest ecosystem, including diversity at the genetic, species, and community levels (**Figure 1**). A wealth of literature provides evidence that biodiversity supports net primary productivity (Chapin et al., 1997; Diaz and Cabido, 2001; Hooper et al., 2005; Thompson et al., 2009; Tilman et al., 2014; Liang et al., 2016; Duffy et al., 2017; de Souza et al., 2019; Matos et al., 2020), adaptation (Steffen et al., 2015; King et al., 2019), resistance (Pimm, 1984; Walker, 1995; Ives et al., 1999; Lehman and Tilman, 2000; McCann, 2000; Loreau et al., 2002; Dorren et al., 2004; Hooper et al., 2005; Thompson et al., 2009; Hautier et al., 2015), resilience (Peterson et al., 1998; Loreau et al., 2001;

Hooper et al., 2005; Drever et al., 2006; Thompson et al., 2009; Ajani et al., 2013; Oliver et al., 2015; King et al., 2019), functional diversity (Cadotte et al., 2011; Levin, 2013; Karadimou et al., 2016), and overall ecosystem functioning (e.g., Lawton, 1997; Tilman, 1997; Hooper et al., 2005; Cardinale et al., 2012; Watson et al., 2018; King et al., 2019). These relationships exist because natural selection yields the characteristic biodiversity and phenotypic plasticity best suited to prevailing environmental conditions, including fluctuating resource inputs, extreme events, periods of stress, and natural disturbances. Specific mechanisms include biotic control of grazing, population density, and nutrient cycling; niche selection and complementarity; biotic and abiotic facilitation; and functional redundancy (i.e., the "insurance hypothesis") (e.g., Naeem et al., 1995; Tilman, 1996; Tilman et al., 1997; Yachi and Loreau, 1999; Loreau, 2000; Tilman and Lehman, 2001; Pretzsch, 2005; Scherer-Lorenzen and Schulze, 2005; Jactel and Brockerhoff, 2007; Thompson et al., 2009; Hantsch et al., 2014; Wright et al., 2017; Liu et al., 2018).

Ecosystem condition

The foundational elements of ecosystem integrity form the basis for assessing ecosystem condition (Keith et al., 2020), specifically in the context of the System of Environmental-Economic Accounting (Committee of Experts on Environmental-Economic Accounting, 2021). Ecosystem condition is defined as "the quality of an ecosystem that may reflect multiple values, measured in terms of its abiotic and biotic characteristics across a range of temporal and spatial scales" (Keith et al., 2020). Ecosystem condition is measured in terms of variables that reflect the state, processes, and changes in the ecosystem, including (i) carbon and nutrient stocks, (ii) abiotic physical and chemical states such as water quantity and quality; (iii) biotic composition, structure, and function; and (iv) landscape diversity and connectivity. Indicators of condition are derived when variables are transformed by assessment against a reference condition. For a given biome and prevailing environmental conditions, these state variables are optimized by the foundational elements of ecosystem integrity and biodiversity (Phillips et al., 1994; Thompson et al., 2009; Roche and Campagne, 2017; Di Marco et al., 2018; Liu et al., 2018).

Ecosystem services

Characteristics of ecosystem condition that relate to the supply of ecosystem services represent an instrumental anthropocentric dimension. Specific ecosystem services can be linked to characteristics of ecosystem condition, and condition indicators can be associated with multiple services (Keith et al., 2020). Ecosystem services can be broadly categorized as regulating, provisioning, and cultural services (Millennium Ecosystem Assessment, 2005; Kandziora et al., 2013; IPBES, 2019; Committee of Experts on Environmental-Economic Accounting, 2021). Regulating services include clean and regulated water flow, air quality, pest and pathogen containment, erosion control, nutrient regulation, resistance and resilience to natural hazards, waste regulation, carbon sequestration and storage, and climate regulation from local to global scales. Provisioning services include the animals, plants, and minerals used for food, medicine, energy, and infrastructure. Cultural services include customary values, ecotourism and nature-based recreation, scientific research, and education.

The concept of ecosystem integrity is useful because it integrates across many properties of forest ecosystems, and thereby optimizes values useful to humans and other organisms. In the words of Koff et al. (2016), "ecosystem integrity is a scientific paradigm that fits the political needs of the present global development agenda focused on complex human-environmental interactions." The concept is holistic and can be adapted to local, national, or international contexts. At jurisdictional levels, the related concepts of "ecological integrity" and "biological integrity" have been used operationally to provide benchmarks for natural resource management (Karr, 1996; Harwell et al., 1999; Campbell, 2000; Muller et al., 2000; Parrish et al., 2003; Tierney et al., 2009; Wurtzebach and Schultz, 2016; Roche and Campagne, 2017). However, as noted above, the international policy community has yet to implement these terms. This is important because ecosystem integrity may be directly linked to forest and carbon risk profiles that, if understood and prioritized, could greatly aid our ability to utilize forests for mitigation and adaptation.

Comparison of ecosystem integrity between forest types

Here, we compare components of ecosystem integrity most relevant for international policy across commonly recognized broad categories of forest types, focusing on primary forests and forests with significant levels of human modification and pressure. We focus on components of ecosystem integrity most pertinent to forest-based climate mitigation, including forest risk profiles as governed by exposure and stability as well as carbon stocks and fluxes. As noted previously, direct comparisons between forest types must account for environmental and site drivers, including the prevailing biome (e.g., tropical, temperate, or boreal) and heterogeneity within as determined by climate, soils, hydrology, and natural disturbance regimes.

Following Kormos et al. (2018), Food and Agriculture Organization of the United Nations [FAO] (2020), and IUCN (2020), primary forests are defined as: (i) largely undisturbed by industrial-scale land uses such as logging, mining, hydroelectric development, and road construction; (ii) established and regenerated by natural biological, ecological, and evolutionary processes; (iii) including the full range of successional stages at a landscape level from pioneer, secondary growth, and old-growth forest stands; and (iv) with the vegetation structure, community networks, and taxonomic composition principally reflecting natural processes including natural disturbance regimes. Primary forests can therefore be distinguished from naturally regenerating forests that are subject to conventional forestry management for commodity production (Puettmann et al., 2015), as well as planted forests, including plantations. For our purposes, primary forest therefore encompasses a range of commonly recognized forest descriptors including intact, virgin, ecologically mature, and old growth forests (Buchwald, 2005; Mackey et al., 2013; DellaSala et al., 2022b).

Foundational elements of ecosystem integrity

Comparison of dissipative structures

In this section we focus on structural complexity because of its importance for carbon stocks. Other components of dissipative structures (**Figure 1**) will be highlighted for their role in supporting ecosystem integrity in following sections (including functional diversity as it relates to biodiversity in the section "Biodiversity," and stored emergy as manifested in biomass and carbon stocks in section "Ecosystem condition"). High-integrity forests that have been allowed time to respond to their emergy signature develop a set of relatively complex ecosystem structures (Campbell, 2000). Canopy structure is particularly influential for other elements of ecosystem integrity such as microclimate, runoff, nutrient cycling, and biodiversity (Hobbie, 1992; Parker, 1995; Didham and Lawton, 1999; Siitonen, 2001; Asner et al., 2010; Goetz et al., 2010; Hansen et al., 2014). Primary tropical forests in particular develop tall, multi-story dense canopies with large variations in plant size and emergent canopy dominants (Kricher, 2011; Hansen A. J. et al., 2020). Temperate forests also develop complex forest canopies as they age, which is associated with high levels of biodiversity and carbon storage (DellaSala et al., 2022b).

Canopy height, in turn, is positively related to aboveground biomass and carbon storage. For example, in Brazil, Democratic Republic of the Congo, and Indonesia, primary forests were 38–59% taller and contained 70–148% more aboveground biomass than other dense tree cover types, including degraded forests, secondary regrowth, and tree plantations (Turubanova et al., 2018). When felling the largest trees or clear-cutting entire stands, logging decreases canopy height, homogenizes forest canopies, and reduces structural complexity (Pfeifer et al., 2016; Rappaport et al., 2018; Bourgoin et al., 2020), which can take centuries to recover. Structural complexity also relates to non-living forest structures, such as dead wood, that provide supporting functions including nutrient cycling, soil formation, and habitat for myriad species (Janisch and Harmon, 2002; Millennium Ecosystem Assessment, 2005; Gamfeldt et al., 2013). When directly compared, primary forests consistently contain a greater volume and diversity of dead wood than forests managed for commodity production (e.g., Guby and Dobbertin, 1996; Siitonen et al., 2000; Siitonen, 2001; Debeljak, 2006).

Comparison of ecosystem processes

Here we focus on ecosystem productivity given its importance for climate mitigation, but note that other ecosystem processes will be highlighted in following sections (evapotranspiration as it relates to drought risk in section "Comparison of risks from drought," reproduction cycles as they relate to regeneration in section "Comparison of regenerative capacity," and nutrient cycling and retention as it relates to nutrient stocks in section "Comparison of ecosystem condition"). Differences in ecosystem productivity and carbon fluxes among forest seral stages have been the subject of much debate. One viewpoint is that forests containing younger trees are more productive, with both higher net primary productivity (NPP, including photosynthesis and autotrophic respiration) and net ecosystem productivity (NEP, also including heterotrophic respiration) than ecologically mature forests (e.g., Ryan et al., 1997; Simard et al., 2007; Goulden et al., 2010). This view has often justified the conversion of primary forests into regrowth forests. While it is true that secondary forests often have higher rates of photosynthesis, this is not always the case, particularly when accounting for the impacts of higher species richness in older primary forests (Liu et al., 2018) and the entire age profile of timber rotations, including times with bare soil and young trees. A wealth of evidence clearly shows that old-growth forests continue to sequester carbon in significant quantities in aboveground biomass, dead wood, litter, and soil organic matter (Phillips et al., 1998; Zhao and Zhou, 2006; Luyssaert et al., 2008; Lewis et al., 2009; Thompson et al., 2013; Gatti et al., 2014; Grace et al., 2014; McGarvey et al., 2015; Schimel et al., 2015; Lacroix et al., 2016; Baccini et al., 2017; Phillips and Brienen, 2017; Qie et al., 2017; Lafleur et al., 2018; Mitchard, 2018). This is why Pugh et al. (2019) found that old-growth forests (defined in that study as > 140 years) cover roughly 39% of global forest area and contribute 40% of the current global forest carbon sink, which in turn represents roughly two-thirds of the terrestrial carbon sink (Friedlingstein et al., 2019).

More importantly, when comparing these CO_2 fluxes in the context of mitigation actions, the entire life cycle of management and disturbance must be taken into account. From a carbon balance perspective, converting primary forests into young forests logged for biomass energy, wood supply, or other uses does not offset the original conversion emissions for many decades to centuries (Cherubini et al., 2011; Holtsmark, 2012; Mitchell et al., 2012; Keith et al., 2015; Birdsey et al., 2018; Hudiburg et al., 2019; Malcolm et al., 2020), creating a large carbon debt on policy-relevant timescales (generally years to 1–3 decades). Hence the size, longevity, and stability of accumulated forest carbon stocks, including in the soils, are important mitigation metrics in addition to the rate of annual sequestration (Mackey et al., 2013; Keith et al., 2021).

Stability and risk profiles

Ecosystem stability is comprised of resistance, resilience, and longer-term persistence (**Figure 1**). Combined with exposure to external perturbations, properties of ecosystem stability provide critical information for risk assessments. Risk assessments are undertaken and utilized in a wide variety of scientific and operational contexts (Fussel and Klein, 2006; Glick et al., 2011; Oppenheimer et al., 2014; Rogers et al., 2017), and are critically important to ensure mitigation actions result in long-term carbon storage. Nevertheless, risk assessments are currently either not undertaken or done so in mostly rudimentary and incomplete ways for forest-based carbon mitigation (Mignone et al., 2009; Ajani et al., 2013; Anderegg et al., 2020). Here we focus on the risk of a forest ecosystem experiencing a state-altering disturbance that results in carbon loss to the atmosphere.

Comparison of risks from wildfire

Wildfires are major natural disturbances in temperate and boreal forest ecosystems, although historically rare in tropical wet forests unless caused by humans (Randerson et al., 2012; Archibald et al., 2013; Giglio et al., 2013; Andela et al., 2017). The area burned by wildfire has been increasing in high-canopy cover forests globally over the past 20 years (Andela et al., 2017), and human-caused fires are a major driver of the loss of intact forest landscapes (Potapov et al., 2017). Extreme fire weather conditions have increased in most forests globally over the last half-century (Jolly et al., 2015; Jain et al., 2017; Dowdy, 2018), and wildfires are projected to become more widespread and intense due to climate change (Ward et al., 2012; Flannigan et al., 2013; Abatzoglou et al., 2019; Dowdy et al., 2019; Rogers et al., 2020). Humans have increased forest fire risk by augmenting forest fuels through active management (DellaSala et al., 2022a) and by increasing the number and sources of ignition (Balch et al., 2017). The majority of documented megafires globally have been started by humans under extreme fire weather conditions (Ferreira-Leite et al., 2015; Bowman et al., 2017).

A large body of literature shows that forests managed for commodity production, degraded, or disturbed forests are generally more susceptible to fires because of drier microclimates and fuels, higher land surface temperatures that promote air movement between forests and neighboring open areas, and human ignitions due to access and proximity, particularly in the tropics (e.g., Uhl and Kauffman, 1990; Holdsworth and Uhl, 1997; Cochrane et al., 1999; Laurance and Williamson, 2001; Siegert et al., 2001; Donato et al., 2006; Lindenmayer et al., 2009, 2011; Brando et al., 2014; DellaSala et al., 2022a). Although fires are a natural disturbance agent throughout most boreal forests (Viereck, 1973; Payette, 1992; Gromtsev, 2002; Soja et al., 2007; Rogers et al., 2015), fire frequency in boreal forests increases in proximity to human land use due to fuel drying, human access, and forestry practices such as leaving slash on site, particularly in Siberia (Kovacs et al., 2004; Achard et al., 2008; Ponomarev, 2008; Laflamme, 2020; Terrail et al., 2020; Shvetsov et al., 2021).

In many forest systems, fires in previously logged or managed landscapes can be more intense/severe, emit more carbon to the atmosphere, and take longer to recover than fires in ecologically mature or primary forests due to increased fuel availability, lower fuel moisture, and dense secondary forests that carry crown fires and are susceptible to extensive tree mortality (Odion et al., 2004; Stone et al., 2004; Thompson et al., 2007; Lindenmayer et al., 2009, 2011; Price and Bradstock, 2012; Kukavskaya et al., 2013; Taylor et al., 2014; Bradley et al., 2016; Dieleman et al., 2020; De Faria et al., 2021; Landi et al., 2021). In general, larger and older trees have a greater chance of surviving fires due to thicker bark and lower relative scorch height (Laurance and Williamson, 2001; Lindenmayer et al., 2019). Increased fuel availability in secondary forests can also facilitate fire spread (Lindenmayer et al., 2011). Positive feedbacks between fires and secondary vegetation can lead to permanent forest loss, i.e. "landscape traps," at the warm / dry edge of forest ranges (Payette and Delwaide, 2003; Hirota et al., 2011; Lindenmayer et al., 2011; Staver et al., 2011; Brando et al., 2014; Kukavskaya et al., 2016; Lindenmayer and Sato, 2018). Primary forests are generally more resistant to fire because of higher humidity and fuel moisture, the presence of understory species such as ferns and mosses that limit light penetration to the forest floor and increase water retention, and much less human access (Ough, 2001; Lindenmayer et al., 2009; Taylor et al., 2014; Zylstra, 2018; Funk et al., 2019).

Comparison of risks from drought

Severe droughts represent 60–90% of climate extremes impacting gross primary productivity in the past 30 years (Zscheischler et al., 2014), are a major driver of tree mortality and forest die-off (Allen et al., 2010, 2015; Anderegg et al., 2013; McDowell and Allen, 2015; McDowell et al., 2016; Rogers et al., 2018), and are expected to increase with future climate change (Cook et al., 2014; Trenberth et al., 2014; Yi et al., 2014; Xu et al., 2019; Zhou et al., 2019; De Faria et al., 2021). A large body of literature indicates closed canopy forests are more resistant to drought, particularly in the tropics, due to shading, biophysical microclimate buffering, thicker litter layers, deeper roots, and increased water use efficiency as trees develop (e.g., Briant et al., 2010; von Arx et al., 2013; Frey et al., 2016; Brienen et al., 2017; Qie et al., 2017; Giardina et al., 2018; Caioni et al., 2020; Elias et al., 2020). For a given level of realized drought, some evidence points to larger older trees being more susceptible to drought impacts (Phillips et al., 2010; Girardin et al., 2012; Bennett et al., 2015; McDowell and Allen, 2015; McIntyre et al., 2015; Chen et al., 2016; Clark et al., 2016). Yet there is also contrasting evidence. For example, younger boreal forests can be more susceptible to drought compared to mature forests (Luo and Chen, 2013; Hember et al., 2017) due to competition for space and nutrients and less extensive and shallower root systems. Tree diversity, which is generally higher in primary compared to human-modified forests (see section "Biodiversity"), may increase resistance and resilience to drought *via* adaptive responses and functional redundancy (Jump et al., 2009; Sthultz et al., 2009; Dale et al., 2010; Harter et al., 2015), and intact forest canopies can be relatively resistant and resilient to short-term climate anomalies including drought (Williamson et al., 2000; Saleska et al., 2007). Evidence also suggests that mechanical "thinning," which is frequently proposed and implemented to combat drought, decreases stand-level water use in the short-term but actually increases individual tree water demand *via* higher leaf-to-sapwood ratios and hence drought vulnerability in the long-term (McDowell et al., 2006; Kolb et al., 2007; D'Amato et al., 2013; Clark et al., 2016).

Mature forests transpire large quantities of water from relatively deep in the soil profile, increasing regional cloud cover and precipitation. This acts to increase the proportion of "recycled" water within a given region and thereby decreases the prevalence of regional droughts (Foley et al., 2007; Spracklen et al., 2012; Ellison et al., 2017). For example, air passing over intact tropical forest landscapes can contain twice the moisture content as air over degraded forests or non-forest landscapes (Sheil and Murdiyarso, 2009). Degradation and the loss of intact forest landscapes increases dry and hot days, decreases daily rainfall intensity and levels, and exacerbates regional droughts (Deo et al., 2009; Alkama and Cescatti, 2016).

Comparison of risks from pests and pathogens

Pests and pathogens are an increasing threat to many forests globally, particularly as climate change alters life cycles, potential ranges, and host-pest interactions (Carnicer et al., 2011; Kautz et al., 2017; Seidl et al., 2017; Simler-Williamson et al., 2019). Mature boreal and temperate forests can be more susceptible to pests and pathogens compared to younger forests, in part due to decreases in the resin flow of defense compounds (Christiansen and Horntvedt, 1983; Hansen and Goheen, 2000; Baier et al., 2002; Dymond et al., 2010). Prominent examples include bark beetle and defoliator susceptibility (Kurz et al., 2008; Raffa et al., 2008; Taylor and MacLean, 2009; Krivets et al., 2015; Kautz et al., 2017). Nevertheless, ecologically mature forests tend to be resilient to biotic infestations, as these cyclical events initiate succession and lead to stand- and landscape-level heterogeneity (Holsten et al., 2008; Thompson et al., 2009). Moreover, tree diversity (measured in terms of genetic, species, and age) tends to limit pest and pathogen spread and damage because of resource dilution, host concealment, phenological mismatches, increased predators and parasitoids, alternative hosts, and metapopulation dynamics (Root, 1973; Karieva, 1983; Pimm, 1991; Watt, 1992; Zhang et al., 2001; Jactel et al., 2005; Pautasso et al., 2005; Scherer-Lorenzen and Schulze, 2005; Thompson et al., 2009; Guyot et al., 2016).

In terms of human influence, anthropogenic disturbances such as selective logging can introduce forest pests and diseases (Gilbert and Hubbell, 1996), including non-native, and evidence suggests forest edges and logged forests are more susceptible to beetle attacks due to increases in available host niches and altered moisture conditions (Sakai et al., 2001). Many pests, particularly in temperate and boreal forests, take advantage of weakened tree defenses during drought (Raffa et al., 2008; McDowell et al., 2011; Anderegg and Callaway, 2012; Hicke et al., 2012; Keith et al., 2012; Poyatos et al., 2013; Anderegg et al., 2015). Monocultures, or tree plantations, have been shown to be particularly vulnerable due to a lack of tree diversity, high tree density, and the associated host-pest interactions (Jactel et al., 2005; Macpherson et al., 2017; Lee, 2018).

Comparison of risks from windthrow

Windthrow events can lead to forest mortality and are expected to increase in some regions with climate change (Klaus et al., 2011; Saad et al., 2017). Although these events are somewhat stochastic, they are also influenced by soils, orography, regional climate regimes, and forest composition and structure. Similar to the risks of pests and pathogens, within a given stand there is evidence that older and taller trees are more susceptible to windthrow due to the physics of taller trees and root rot (Lohmander and Helles, 1987; Ruel, 1995). Nevertheless, fragmented or thinned forests experience elevated mortality and collapse of trees from windthrow because of increased exposure (Laurance and Curran, 2008; Reinhardt et al., 2008; Schwartz et al., 2017).

Comparison of risks from species range shifts

Climate regimes have strong influences on the potential and realized ranges of forest tree species, evidenced by the paleoecological record (Overpeck et al., 1991; DeHayes et al., 2000; Davis and Shaw, 2001) and current assemblages (e.g., Neilson, 1995; Foley et al., 2000), and considerable scientific effort is focused on projecting future responses to climate change (e.g., Sitch et al., 2003; Elith and Leathwick, 2009; Rogers et al., 2011, 2017; Ehrlen and Morris, 2015; Prasad et al., 2020). How trees and forest ecosystems will respond is uncertain due to complex interactions between the pace of climate change, physiological tolerances, dispersal and migration rates, phenotypic plasticity and adaptation, the presence of climate refugia, migration of associated species / symbionts, and forest fragmentation, among others (Davis and Shaw, 2001; Iverson et al., 2004; Jump and Penuelas, 2005; Mackey et al., 2008; Nicotra et al., 2010; Prasad, 2015; Rogers et al., 2017). In general, current and projected climate change is expected to degrade biodiversity due to species extinctions and the contraction of realized ranges (Miles et al., 2004; Campbell et al., 2009). Forest and landscape fragmentation in particular is known to hinder resilience and species migration because of the loss of suitable areas for dispersal and limitations on gene flow (Collingham and Huntley, 2000; Loreau et al., 2002; Scheller and Mladenoff, 2008; Thompson et al., 2009). Large areas of primary forests are expected to have higher adaptive capacity and stability compared to forests under human pressure because of their connectivity, biodiversity, and microclimate buffering (Mackey et al., 2015; Watson et al., 2018; Thom et al., 2019; see section "Biodiversity").

Comparison of risks from land use degradation

Human land use pressures on forests generally result in both direct environmental impacts as well as further, often unplanned, degradation or deforestation that accumulates spatially and temporally. This is exemplified by the fact that smaller fragments of primary forest have an elevated likelihood of loss (Hansen M. C. et al., 2020). New roads are the primary driver of further degradation as a result of their construction, use, and continued access (e.g., Trombulak and Frissell, 2000; Wilkie et al., 2000; Laurance et al., 2009; Laurance and Balmford, 2013; Ibisch et al., 2016; Alamgir et al., 2017; Venier et al., 2018; Maxwell et al., 2019). Roads render the surrounding forests much more susceptible to agricultural conversion (Asner et al., 2006; Boakes et al., 2010; Gibbs et al., 2010; Laurance et al., 2014; Kormos et al., 2018), logging (Laurance et al., 2009; Barber et al., 2014), and expanded networks of secondary and tertiary roads (Arima et al., 2008, 2016; Ahmed et al., 2014). Logging and transportation can also lead to severe erosion and nutrient runoff, impacting downstream water quality and quantity (Carignan et al., 2000; Hartanto et al., 2003; Foley et al., 2007), and damage the surrounding forest. For example, in the Amazon, it has been estimated that for every commercial tree removed *via* selective logging, roughly 40 m of roads are created, nearly 30 other trees greater than 10 cm in diameter are damaged, and between 600 and 8,000 m^2 of canopy is opened (Holloway, 1993; Asner et al., 2004). Furthermore, roads reduce animal habitat, are barriers to animal movement and lead to increased animal mortality, including from unregulated hunting, all of which decrease connectivity and genetic exchange (Dyer et al., 2002; Frair et al., 2008; Laurance et al., 2009; Taylor and Goldingay, 2010; Clements et al., 2014). One consequence is a decline in carbon-dense tree species due to overhunting of seed-dispersing animals (Osuri et al., 2016; Maxwell et al., 2019). It is important to note that roughly 95% of deforestation in the Amazon occurs within 5.5 km of a road (Barber et al., 2014), and that illegal logging represents 85–90% of all logging in the tropics (Lawson and MacFaul, 2010; Lawson, 2014; Hoare, 2015) and still roughly one-quarter of logging in Russia (Food and Agriculture Organization of the United Nations [FAO], 2012; Kabanets et al., 2013), which contains the largest areal forest coverage of any country (Food and Agriculture Organization of the United Nations [FAO], 2020). Overall, road building and industrial logging are the largest drivers of initial forest degradation and fragmentation (Hosonuma et al., 2012).

In addition to their direct impacts, roads and land use further degrade forests due to edge effects. Forests at or near an edge can have substantially drier microclimates, increased windshear and movement of dry air into forests, invasive species (dispersed *via* roads and more favorable microclimate conditions for competition), weeds and vines, sun exposure, soil erosion, and fuel loads due to drying and previous logging and fire (Laurance and Williamson, 2001; Mortensen et al., 2009; Brando et al., 2014). This leads to a variety of unfavorable impacts and further risks. Carbon densities tend to be significantly lower near forest edges. For example, biomass is reduced by roughly 50% within 100 m, 25% within 500 m, and 10% within 1.5 km of a forest edge (Laurance et al., 1997; Chaplin-Kramer et al., 2015; Maxwell et al., 2019). Aggregated across the tropics, edge effects are estimated to account for up to one quarter of all carbon loss from tropical deforestation (Putz et al., 2014). Primary productivity is also generally lower near forest edges, and fire susceptibility is higher due to elevated and drier fuel loads and increased human access (Laurance et al., 1998; Cochrane et al., 1999; Nepstad et al., 1999; Laurance and Williamson, 2001; Foley et al., 2007; Adeney et al., 2009; Brando et al., 2014). For example, roads are strong predictors of ignition and wildfire frequency in temperate forests (Hawbaker et al., 2013; Faivre et al., 2016; Parisien et al., 2016; Balch et al., 2017; Ricotta et al., 2018), and road expansion in Siberia has been shown to promote logging and human-caused forest fires (Kovacs et al., 2004). A variety of ecosystem services are degraded due to edge effects, including hydrologic regulation, water quality, modulation of regional climate, and amelioration of infectious diseases (Laurance and Williamson, 2001; Foley et al., 2007). Although the impacts are strongest at a forest edge, the effects can generally be detected up to 2 km from the edge, with higher tree mortality up to 1 km and wind disturbance up to 500 m (Broadbent et al., 2008). Globally, fragmentation is thought to be at a critical threshold, with roughly 70% of the world's forest within 1 km of a human-created forest edge (Haddad et al., 2015; Taubert et al., 2018).

Comparison of regenerative capacity

Ecosystem resilience is underpinned by the natural regenerative capacity of a forest ecosystem, and hence represents a major component of ecosystem stability and integrity (**Figure 1**). Regeneration from major disturbance events requires biological legacies, which are broadly defined as the remaining living and dead structures and organisms that can influence recovery (Franklin et al., 2000; Jogiste et al., 2017). These include living and dead trees, shrubs and other plants, seeds, spores, fungi, eggs, soil communities, and living animals (Franklin et al., 2000; Stahlheber et al., 2015; Lindenmayer et al., 2019). Compared to secondary or human-modified forests, primary forests tend to have the biological legacies (Catterall, 2016; Chazdon and Uriarte, 2016; Lu et al., 2016; Poorter et al., 2016; Lindenmayer et al., 2019) and favorable microclimates (von Arx et al., 2013) required for optimal regeneration. This is evidenced by the fact that secondary forest regeneration is aided by proximity to primary forests (Schwartz et al., 2015; Kukavskaya et al., 2016). Clearcut logging also generates low levels of biological legacies and higher regeneration failures after subsequent fires compared to forests not previously logged (Perrault-Hebert et al., 2017), which is exacerbated by post-fire "salvage" logging (Donato et al., 2006; Lindenmayer et al., 2019). Successive disturbances continue to decrease regenerative capacity, and can lead to permanent forest loss and emergence of non-forest ecosystems (Payette and Delwaide, 2003; Johnstone et al., 2016; Kukavskaya et al., 2016). Compared to degraded or human-modified forests, primary forests with large extents also host a much larger array of seed dispersers and pollinators (Muller-Landau, 2007; Wright et al., 2007; Abernethy et al., 2013; Harrison et al., 2013; Peres et al., 2016).

Comparison of biodiversity

Biodiversity underpins and is affected by the foundational elements of ecosystem integrity (**Figure 1**), but is also a metric of ecosystem condition and can be considered an ecosystem service in its own right. Globally, trees are among the most genetically diverse of all organisms, and forests collectively support the majority (roughly 80%) of terrestrial biodiversity (Hamrick and Godt, 1990; Barlow et al., 2007; Pimm et al., 2014; Federici et al., 2017). There is a substantial body of literature on the effects of disturbance and stand age on biodiversity, with some disagreement among studies depending on context (e.g., Paillet et al., 2010; Edwards et al., 2011; Moreno-Mateos et al., 2017; Kuuluvainen and Gauthier, 2018; Matos et al., 2020). Nevertheless, there are clear and definitive negative impacts of human disturbance and land use on biodiversity (Cairns and Meganck, 1994; Ellison et al., 2005; Barlow et al., 2007, 2016; Gibson et al., 2011; Alroy, 2017; Giam, 2017). Primary and ecologically mature forests typically harbor higher biodiversity than human-modified forests (Lesica et al., 1991; Herbeck and Larsen, 1999; Rey Benayas et al., 2009; Zlonis and Niemi, 2014; Miller et al., 2018; Watson et al., 2018; Lindenmayer et al., 2019; Thom et al., 2019), especially in the understory (e.g., Lafleur et al., 2018). Disturbance generally results in a change in species composition toward early pioneer species (e.g., Bawa and Seidler, 1998; Liebsch et al., 2008; Venier et al., 2014). The effect of human activities on the provision of ecosystem services is evident even if there is little change in the overall forest cover. Degradation in logged forests can be in the form of structural changes such as reduction in old age classes of trees that can cause loss in breeding habitat, particularly for birds (Rosenberg et al., 2019; Betts et al., 2022), and compositional changes such as shifts in tree species abundance that differ in foliar nutrient concentrations that support arboreal folivores (Au et al., 2019). Under less intensive agriculture management, agroforestry can maintain a significant fraction of biodiversity, but it is still considerably lower than in native forests (De Beenhouwer et al., 2013; Vallejo-Ramos et al., 2016).

Biodiversity analyses are also strongly dependent on spatial scale, whereby higher levels of management and disturbance homogenize forest composition and age structure across the landscape, and consequently the biota it supports (e.g., Devictor et al., 2008; de Castro Solar et al., 2015; Tomas Ibarra and Martin, 2015). What can be concluded is that (i) degraded and intensively managed forests tend to harbor lower biological and functional diversity compared to primary forests, which support many as yet unidentified species and act as repositories for species that cannot survive in secondary or degraded forests (Barlow et al., 2007; Gibson et al., 2011), and (ii) natural disturbances are effective at maintaining landscape heterogeneity and the species that depend on disturbed and young forests (Lindenmayer et al., 2019). Global biodiversity loss is currently orders of magnitude higher than background rates and is driven primarily by deforestation and forest degradation (Newbold et al., 2016; Giam, 2017). It is worth noting that although natural tree diversity in boreal forests is typically much lower than in temperate or tropical forests (Thompson et al., 2009; Hill et al., 2019), the biodiversity of other species groups such as bryophytes and lichens can be very high (DellaSala, 2011; Kuuluvainen and Gauthier, 2018), functional diversity in boreal forests is generally high (Esseen et al., 1997; Wirth, 2005), and the broad genetic variability and phenotypic plasticity of boreal trees allows them to tolerate a wide range of environmental conditions (Gordon, 1996; Howe et al., 2003).

Comparison of ecosystem condition

Given our focus on climate mitigation, the primary metric of concern for ecosystem condition is carbon stocks. Primary and ecologically older forests have been consistently found to have the highest carbon stocks compared to secondary, degraded, intensively managed, or plantation forests (e.g., Harmon et al., 1990; Cairns and Meganck, 1994; Nunery and Keeton, 2010; Burrascano et al., 2013; Mackey et al., 2013; Keith et al., 2015, 2017; Federici et al., 2017; Lafleur et al., 2018; Watson et al., 2018). For example, a recent meta-analysis shows that primary tropical forests store on average 35% more carbon than forests affected by conventional management for commodity production (Mackey et al., 2020). Across the tropics, intact forest landscapes cover approximately 20% of total area but store 40% of total aboveground biomass (Potapov et al., 2017; Maxwell et al., 2019). This is fundamentally a function of where carbon is stored in these forests. In wet tropical and some temperate primary forests, roughly half the biomass carbon is stored in the largest 1–3% diameter trees (Stephenson et al., 2014; Lutz et al., 2018; Mildrexler et al., 2020), which have long residence times (Koerner, 2017; van der Sande et al., 2017), and are typically the first to be felled (Cannon et al., 1998; Sist et al., 2014; Gatti et al., 2015; Rutishauser et al., 2016). Agricultural landscapes store comparatively less carbon, but the addition of trees *via* agroforestry has the potential to add up to 9 Pg C globally (Chapman et al., 2020). In boreal forests, especially those that are poorly drained, the majority of forest ecosystem carbon is stored in dead biomass, peat, and soil organic layers that accumulate over the course of forest succession, often protected by permafrost (Deluca and Boisvenue, 2012; Bradshaw and Warkentin, 2015; Lafleur et al., 2018; Walker X J et al., 2020). Boreal forests managed for timber are kept at younger ages, with soils that store significantly less carbon due to mechanical disturbance, tree species conversion, and impacts on litter composition, nutrient cycling, and bryophyte communities (Liski et al., 1998; Jiang et al., 2002; Seedre et al., 2014; Lafleur et al., 2018). Even outside the boreal zone, soil carbon can be a significant fraction of total ecosystem carbon (e.g., Keith et al., 2009), and logging activities generally deplete forest soil carbon due to soil compaction and disturbance, erosion, changes in microclimate that increase respiration rates, reduced leaf litter and root exudates, loss of micorrhizal network carbon, and post-logging "slash" burning (Rab, 2004; Zummo and Friedland, 2011; Buchholz et al., 2014; James and Harrison, 2016; Hume et al., 2018; Mayer et al., 2020). Globally, forests are thought to store only half of their potential carbon stock, with 42–47% of the reduction due to forest management and modification (the remainder being deforestation and land cover changes; Erb et al., 2018). Natural regeneration of forests could in turn restore 123 Pg C, or 27% of the total biomass carbon that has been lost (Erb et al., 2018).

Forest management, degradation, and conversion can also result in the loss of key nutrients such as nitrogen and phosphorous, among others, which are otherwise retained efficiently in undisturbed forests (Likens et al., 1970; Markewitz et al., 2004; Olander et al., 2005; Liu et al., 2019). Nutrients can be artificially added, but heavily managed systems require large inputs to maintain their state and productivity capacity (Noss, 1995; Merino et al., 2005; Pandey et al., 2007). Other elements of ecosystem condition are affected similarly and highlighted elsewhere (landscape connectivity / fragmentation in section "Comparison of risks from land use degradation," biodiversity in section "Comparison of biodiversity," and water quality and quantity in section "Comparison of ecosystem services").

Comparison of ecosystem services

A large body of literature indicates the higher number, quality, and value of ecosystem services provided by primary forests compared to human-modified forests and landscapes. These include regulating services such as water quality and quantity (DellaSala, 2011; Brandt et al., 2014; Keith et al., 2017; Kormos et al., 2018; Taylor et al., 2019; Vardon et al., 2019); carbon storage and sequestration as an ecosystem service of global climate regulation (United Nations [UN], 2021) [discussed above, but see Keith et al. (2019) and Uganda Bureau of Statistics [UBOS] (2020) for examples using Ecosystem Accounts]; local to regional biophysical cooling (Spracklen et al., 2012; Lawrence and Vandecar, 2015); regulation of runoff, sediment retention, erosion control, and flood mitigation (Hornbeck and Federer, 1975; Jayasuriya et al., 1993; Dudley and Stolton, 2003; Furniss et al., 2010; van Haaren et al., 2021); provisioning services such as abundance of game and fish (Gamfeldt et al., 2013; Brandt et al., 2014); cultural services such as landscape aesthetics, recreation, and tourism (Brandt et al., 2014; Brockerhoff et al., 2017); cultural practices and knowledge (Normyle et al., 2022); contributions to physical and psychological health (Stier-Jarmer et al., 2021); and general assessments across a suite of services (e.g., Myers, 1997; Harrison et al., 2014; Shimamoto et al., 2018; Maes et al., 2020).

For example, a detailed assessment of the differences between primary forests and post-logging regrowth forests in terms of their ecosystem condition, the physical supply of a suite of ecosystem services, and their monetary valuation showed the superior aggregated value of the primary forest (Keith et al., 2017). The impacts of mechanical disturbance due to logging, roading, and mining on soil properties reduce the ecosystem services of soil nutrient availability, water holding capacity and erosion prevention (Hamburg et al., 2019). A general assessment of the total economic value of ecosystem services provided by forest ecosystem types showed that primary forests had a higher median value (USD 139 ha^{-1} $year^{-1}$) compared with secondary forests (USD 128 ha^{-1} $year^{-1}$) (Taye et al., 2021). These aggregated values include only the market values for services when known and could not account for non-market values, for example that would be needed to assess biodiversity habitat or many cultural services. The highest reported values for specific ecosystem services were for airflow regulation, water cycle regulation and food for freshwater plants and animals. These services would all have their highest provision from natural ecosystems. In contrast, the value of timber and fiber products is significantly lower.

Lessons from comparative analysis

Taken as a whole and for a given set of environmental conditions, our comparative analysis shows that primary forests have the highest levels of ecosystem integrity compared to human-modified forests, including naturally regenerating forests managed for commodity production, plantations, and previously forested landscapes. One primary set of mechanisms are positive feedbacks whereby forest disturbance tends to beget more disturbance (e.g., Seidl et al., 2017), and degradation begets more degradation (e.g., Venier et al., 2018; Watson et al., 2018). In terms of variables most relevant for mitigation, adaptation, and other international forest policy goals, primary forests store the highest carbon stocks, present the lowest risks of forest and carbon loss reversal, have the highest biodiversity, and provide the largest stocks of ecosystem assets and highest quality flows of ecosystem services, including benefits to the global community, local communities (Vickerman and Kagan, 2014), and Indigenous peoples.

Based on our review, and because human-modified forests can encompass a wide range of management strategies and intensities, we provide further summaries of ecosystem integrity for five main categories of forest types: (A) primary forests; (B) secondary forests; (C) production forests; (D) agro-forests; and (E) plantations (**Figure 2** and **Table 1**). Primary forests have the most developed dissipative structures, the highest levels of ecosystem processes, greater stability and recovery, and thus greater resilience and the lowest risk of loss and damage. As defined here, secondary forests are in recovery from past human impacts especially logging. Although they can transition to primary forests over time, these forests lack some old growth characteristics, are more vulnerable to wildfire and other natural disturbances, and have missing elements of biodiversity. Production forests are a result of conventional forest management for commodity production, and tend to be kept at relatively young ages with associated reductions in dissipative structures, carbon stocks, and resilience. An example of commercial agro-forests is shade coffee where retaining some natural canopy tree cover provides some additional ecosystem service benefits. Subsistence agro-forests are common in many tropical development countries such as Vanuatu where these household and community gardens were, and in many cases still are, the main source of food. Commercial plantations include monocultures of trees species that are essentially tree farms for commodity production (wood, palm oil). Note that there are gradients of human modification, stand age, and ecosystem integrity within these broad categories. For example, mature forests recovering from past human disturbances may not have the full suite of structural, functional, and compositional benefits as primary forests, but they can gain these over time, and generally have higher ecosystem integrity than forests

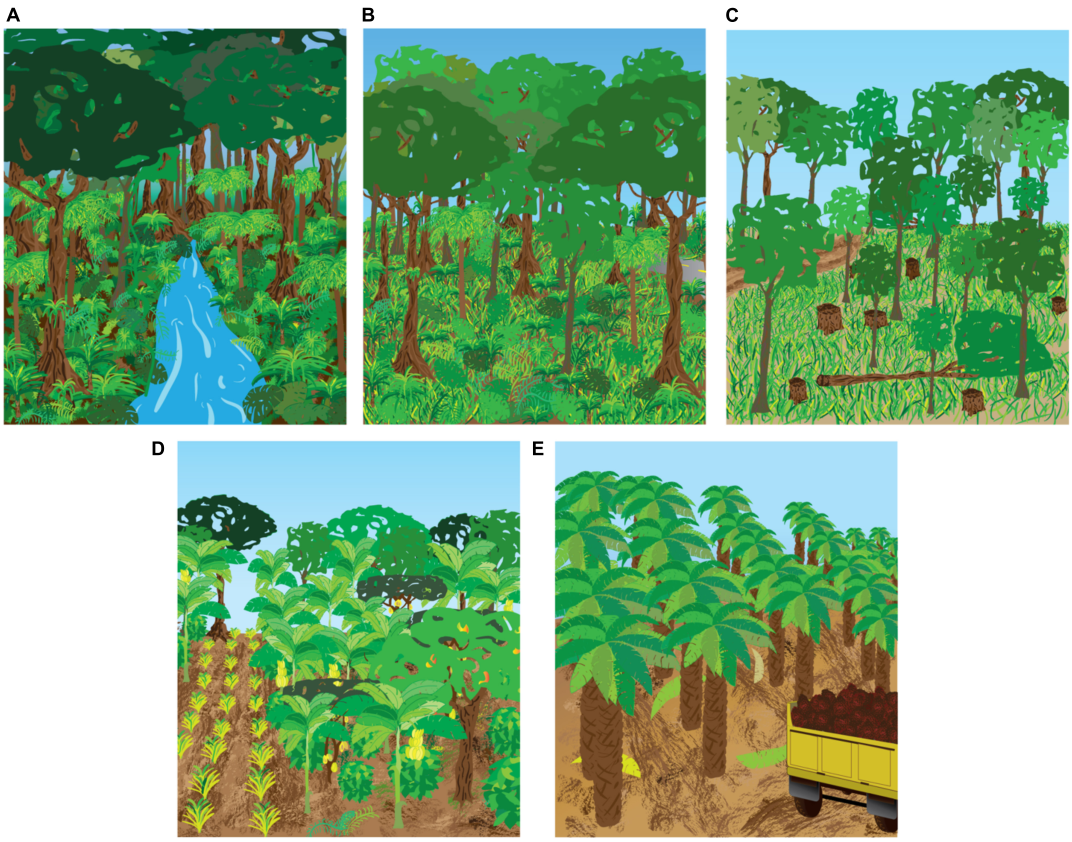

FIGURE 2
Graphical illustrations of five main forest types considered for ecosystem integrity comparisons, including **(A)** primary forests, **(B)** secondary forests, **(C)** production forests, **(D)** agro-forests, and **(E)** plantations. Note this illustration focuses on tropical forests, but the same general differences apply across forest biomes.

TABLE 1 Comparison of ecosystem integrity foundational elements between five main forest types.

Primary forest

• Naturally regenerated forest of native tree species, where there are no clearly visible indications of human activities and the ecological processes are not significantly disturbed
• Likely to have never been commercially logged or intensely managed
• At a landscape level, can comprise early successional (seral) stage following natural disturbances
• More likely to contain full complement of evolved natural biodiversity
• Often the customary territories of Indigenous Peoples

Dissipative structures	• *Ecosystem processes*	• *Stability and risk profiles*	• *Ecosystem integrity level*
• Canopy trees dominated by large, old trees • In wet tropics, closed canopies • Dense soil organic stocks • Typically significant quantities of dead biomass	• Fully self-generating (autopoiesis) • In temperate and boreal forests, includes seral stages following natural disturbances • Tight nutrient cycling with minimal leakage and/or erosion • Clean water supply	• Highly resistant and/or resilient to extreme weather events • In boreal and temperate biomes, fire-adapted plant species • Rich biodiversity provides functional and phenotypic adaptive capacity	• High levels for all three factors

Secondary forest

• Natural forests recovering from prior human land use impacts
• Canopies dominated by pioneer and secondary growth tree species
• If not subsequently disturbed by human land use, can continue to develop additional primary forest attributes over time

• *Dissipative structures*	• *Ecosystem processes*	• *Stability and risk profiles*	*Ecosystem integrity level*
• In wet tropics, canopy closure can occur within 1–2 decades • Aboveground living significantly less than primary forests • Some dead biomass may remain	• Fully self-regenerating so long as primary propagules/seed stock are available • Soil carbon and nutrients stocks can be depleted due to past erosion and biomass removal	• In temperate and boreal forests, increased exposure to wildfire and drought impacts due to more open canopy and drier forest interior • Reduced biodiversity impairs some key processes (e.g., pollination, top-down tropic control)	• Moderate depending on time since disturbance

Production forest

• The consequence of conventional forest management for commodity production (e.g., timber, pulp)
• Forest predominantly composed of trees established through natural regeneration, but management favors commercially valuable canopy tree species

• *Dissipative structures*	• *Ecosystem processes*	• *Stability and risk profiles*	*Ecosystem integrity level*
• Logging regimes maintain a predominantly even-aged, younger age structure (∼20–60 years) • Simplified vertical vegetation structure	• Canopy tree species natural regenerated but some level of assisted regeneration common • Ongoing soil loss	• More flammable forest conditions • Greater exposure to invasive species	• Low to moderate depending on intensity of logging regimes and biodiversity loss

Agro-forestry (commercial, subsistence)

• Some level of natural tree species is maintained with subsistence food or commercial crops grown (e.g., shade coffee).
• Swidden subsistence farming commonly used by traditional communities
• Utilizes a mix of natural and assisted regeneration

Dissipative structures	*Ecosystem processes*	*Stability and risk profiles*	*Ecosystem integrity level*
• A curated canopy of trees, often remnant from primary forest or planted from local stock • Little if any understory • Ground cover are food crops	• In tradition swidden system, closed nutrient cycle through use of natural regeneration • Canopy trees buffer food crops from extreme weather and help maintain soil moisture	• Intensive small-scale management and modest level of biodiversity provides assisted resilience and adaptive capacity	• Low to moderate given sufficient management inputs

(Continued)

TABLE 1 (Continued)

Commercial plantation			
• Forest predominantly composed of trees established through planting and/or seeding and intensely managed for commodity production (timber, pulp, plant oil)			
Dissipative structures	***Ecosystem processes***	***Stability and risk profiles***	***Ecosystem integrity level***
• Typically mono-cultures that are harvested at around a young age (~10–20 years)	• Soil water and nutrient retention • Can utilize natural pollinators from neighboring or remnant natural forests	• Exposed to extreme weather events, invasives, pests, and disease • Intensive large-scale management needed	• Low

recovering from more recent human disturbance (DellaSala et al., 2022b).

Implications for policy, management, and future research

Evaluating ecosystem integrity

We have shown that the risk of forest carbon loss can be minimized by prioritizing actions that maintain and enhance forest ecosystem integrity. Ecosystem integrity therefore has the potential to be used as an integrating framework for evaluating forest-based mitigation and adaptation actions. Because ecosystem integrity is an inherently complex concept, the scientific, management, and policy communities need approaches and tools to measure and interpret gradients of integrity consistently across forest types and jurisdictional boundaries (Karr, 1996; Grantham et al., 2020). The metrics and their interpretation should ideally account for the range of spatial and temporal scales involved: small patches of high-integrity forests are valuable, but landscape context is required; snapshots in time are useful, but longer-term dynamics are needed to fully understand integrity.

A complete and exhaustive global representation of forest ecosystem integrity may currently be beyond our reach. Nevertheless, several existing data products represent important elements of ecosystem integrity, each with their own advantages and limitations, and can be used to guide decision making. In the humid tropics, natural and hinterland forests (primary forests and mature secondary growth) have been mapped using multispectral satellite imagery (Turubanova et al., 2018) and spatial statistics (Tyukavina et al., 2016). Canopy structural integrity has recently been mapped using space-based lidar, multispectral imagery, and human pressure indices (Hansen et al., 2019; Hansen A. J. et al., 2020), representing an important step in delineating gradients of integrity. These mapping approaches are inherently more challenging outside the humid tropics where environmental gradients generate a range of potential forest cover and types. Global products therefore tend to rely more on metrics based on the relationships between forest loss/degradation and proximity to human activities, including roadless areas, forest fragmentation, loss of tree cover, and measures of the "human ecological footprint" (Hansen et al., 2013; Haddad et al., 2015; Ibisch et al., 2016; Venter et al., 2016b,a; Beyer et al., 2020; Grantham et al., 2020; Williams et al., 2020). Global Intact Forest Landscapes (Potapov et al., 2008, 2017) have been widely used, but these include patches of non-forest ecosystems and exclude areas of high-integrity forests in patches <50,000 ha. The Food and Agriculture Organization of the United Nations (FAO) has reported on primary forests since 2005 in their global forest assessment reports (Food and Agriculture Organization of the United Nations [FAO], 2020), but a lack of consistency in national-level reporting makes comparisons and trend detection difficult.

Similar to Grantham et al. (2020), we stress the importance of using local data and field observations to further identify and refine estimates of forest ecosystem integrity derived from coarser-scale global mapping products. These may include landscape-level metrics such as frequency distributions of stand age, biomass, coarse woody debris, biodiversity, forest patch sizes and shapes, and forest types and species composition. Individual countries have data archives, collection programs, and often agency directives that either include ecosystem integrity metrics or those with high relevance for integrity assessments (e.g., Muller et al., 2000; Tierney et al., 2009; Wurtzebach and Schultz, 2016). Applying the internationally endorsed SEEA-EA system should also enable a consistent framework for comparisons across spatial and temporal scales. The SEEA-EA standard provides guidance for classifications, definitions, spatially explicit analysis, and temporal consistency. Technical guidance on ecosystem integrity indicators was recently provided by Hansen et al. (2021). Although criteria were provided in the context of CBD's post-2020 Global Biodiversity Framework, many would apply outside this context, including a need for biome to global scale products with spatial resolution sufficient for management ($\leq$ 1 km), temporal re-assessment at intervals of 1–5 years, ability for indicators to be spatially aggregated without bias, credibility through validation and peer review, and accounting for reference states within a given climate, geomorphology, and ecology. Finally, we note the importance of understanding how any given metric of ecosystem integrity connects to the conceptual framework of ecosystem integrity (**Figure 1**).

Implementing ecosystem integrity

Protecting primary forests

Given the superior benefits of primary forests, follows that protecting them would significantly contribute to meeting international climate, biodiversity, and SDGs. Primary forests are disappearing at a rapid rate (e.g., Potapov et al., 2017; Food and Agriculture Organization of the United Nations [FAO], 2020; Hansen M. C. et al., 2020; Silva Junior et al., 2021) and urgently need higher levels of protection to ensure their conservation; only roughly one-fifth of remaining primary forests are found in the International Union for Conservation of Nature (IUCN) Protected Areas Categories I-VI (Mackey et al., 2015). Proven effective mechanisms to protect primary forests include enforcing existing and establishing new reserves and protected area networks, limiting new road construction, payments for ecosystem services, effective governance, and protecting the rights and livelihoods of indigenous peoples and local communities (Mackey et al., 2015; Kormos et al., 2018; Walker W. S et al., 2020). Complementary measures and enabling conditions include supporting legislation and enforcement of protection status, industry re-adjustment to source alternative fuel, food and wood products, and management of weeds, pests, feral animals, and livestock grazing (Mackey et al., 2020).

Protecting primary forests will also be facilitated by changes to current international forest and carbon accounting rules. Existing "net" forest cover accounting rules, such as the IPCC good practice guidelines for national greenhouse gas inventories and the land sector, are problematic because they report net changes and treat all forests equally, regardless of their integrity, thereby incentivizing the conversion of primary forests into commodity production (Mackey et al., 2013, 2015; Peterson and Varela, 2016; Moreno-Mateos et al., 2017; Funk et al., 2019; Skene, 2020). Such changes in forest management can have the perverse effect of accelerating emissions and degrading ecosystems. Similarly, flux-based carbon accounting effectively hides the emissions or lost sequestration potential from logging primary forests (e.g., Skene, 2020) and does not account for the risk profiles of different forest types. Reporting "gross" forest cover changes as well as adopting stock-based accounting (Ajani et al., 2013; Keith et al., 2019, 2021) could more fully leverage an ecosystem integrity framework, and ultimately ensure the maximum mitigation benefits and ecosystem services are secured from Earth's remaining forests.

Management of other forest types

Management of secondary forests for commodity production, along with tree plantations and agroforestry, can contribute to climate mitigation and other SDGs and reduce pressure on primary forests and other natural forests with high levels of ecosystem integrity (Watson et al., 2018; Roe et al., 2019; Chapman et al., 2020). However, the key is to direct these management activities to previously deforested or degraded lands and accompany them with systematic landscape planning and effective governance (Dooley et al., 2018; Kormos et al., 2018; Martin et al., 2020; Morgan et al., 2020). For example, much of the overall timber demand could be harvested from secondary forests, but these are often overlooked as resources by land owners, the timber industry, and governments (Bawa and Seidler, 1998). Globally, intensively managed tree plantations or planted forests supply over 50% of global wood supply (Warman, 2014) yet occupy only 7% of global forest cover (Food and Agriculture Organization of the United Nations [FAO], 2020). It is therefore feasible to meet global wood supply with existing plantations and additional ones established on previously cleared or degraded land. These land uses, however, are decidedly not beneficial for carbon budgets or ecosystem services when undertaken at the cost of clearing or degrading primary forests.

Governments and forest managers can aim to optimize the ecosystem integrity of secondary forests (for example in terms of yield, regenerative capacity, and biodiversity) within the confines of their intended uses (Thompson et al., 2009; Grantham et al., 2020). In tandem with alternative fibers, this will help alleviate pressures on primary forests. A similar argument exists for agricultural productivity (Laurance et al., 2001; Hawbaker et al., 2006; Sabatini et al., 2018). All of these activities can be done with appropriate landscape planning in ways that collectively increase economic yield and ecosystem services, and serve local communities (Bawa and Seidler, 1998; Burton et al., 2006; Mathey et al., 2008; Food and Agriculture Organization of the United Nations [FAO], 2012; Naumov et al., 2016).

Afforestation, forest restoration, and proforestation (i.e., allowing secondary forests to naturally regrow and restore their ecosystem carbon stocks) are also important components of forest-based mitigation and conservation activities (Giam et al., 2011; Griscom et al., 2017; Verdone and Seidl, 2017; Moomaw et al., 2019; Roe et al., 2019; Cook-Patton et al., 2020). Proforestation holds promise for near-term mitigation because the established trees are already on the steepest part of their growth curve (Moomaw et al., 2019; Mackey et al., 2020). However, none of these forest management activities can replace the carbon stocks and ecosystem services of high-integrity primary forests on decadal to century timeframes. It is also generally less expensive to protect primary forests than to reforest or restore forests (Possingham et al., 2015; Griscom et al., 2017). Furthermore, potential "overcrediting" for offset and restoration schemes can result in net harm and carbon emissions, whereas "overcrediting" for primary forest protection only reduces the benefits, but does not lead to net societal and climate damages (Anderegg et al., 2020). We therefore urge that forest restoration should be conducted in concert with protection of primary forests, and not instead.

Finally, we note that selective logging, or so called "reduced impact logging" in tropical forests has been shown many times to be unsustainable (Zimmerman and Kormos, 2012; Kormos et al., 2018), as it results in significant damage to the target forests as well as collateral damages to surrounding forests due to road building, transportation, and further clearing for land uses such as agriculture (Kormos and Zimmerman, 2014; Mackey et al., 2020). Generally, as timber extraction becomes less intensive, the per-tree collateral damages increase exponentially (Gullison and Hardner, 1993; Boot and Gullison, 1995; Bawa and Seidler, 1998; Umunay et al., 2019; Zalman et al., 2019). After the first cut, selective logging is much less economically viable compared to plantations and intensive forestry (Bawa and Seidler, 1998; Naumov et al., 2016). Even measures aimed at reducing emissions *via* collateral damages from selective logging may not generate benefits and merely serve to justify and subsidize the degradation of high-integrity primary forests (Macintosh, 2013; Watkins, 2014; Gatti et al., 2015). Overall, selective logging and its associated degradation may be as much or more harmful than outright deforestation for pan-tropical forests and their carbon stocks (Nepstad et al., 1999; Foley et al., 2007; Baccini et al., 2017; Erb et al., 2018; Bullock et al., 2020; Matricardi et al., 2020).

Relevance for international policy

There has been a recent uptick in the recognition of the importance of ecosystem integrity and primary forests for multiple climate, biodiversity, and SDGs. For example, the preamble to the Paris Agreement notes the importance of ensuring the integrity of all ecosystems, and recent international policy developments point to the importance of maintaining and restoring ecosystem integrity for achieving the goals of the Rio Conventions and all of the SDGs, but in particular SDG 15 (Life on Land). The importance of primary forests for achieving synergistic climate and biodiversity outcomes was also reflected in Working Group II (IPCC, 2022) and III (Nabuurs et al., 2022) of the IPCC's Sixth Assessment Report, as well as key decisions from the CBD 14th Conference of the Parties (14/5 and 14/30) (Convention on Biological Diversity [CBD], 2018).

We strongly recommend an increased focus on integrating climate and biodiversity action, which provides an opportunity to deliver multiple societal goals through ensuring the integrity of ecosystems (Barber et al., 2020). The importance of the nexus between effective action on climate change and biodiversity is reflected in the findings of the first ever joint workshop of the IPCCC and IPBES held in 2021 (Pörtner et al., 2021), which encouraged synergistic climate and biodiversity action and identified priorities for action, in particular the protection and restoration of carbon and species rich natural ecosystems such as forests.

The integrity of ecosystems is also being promoted by civil society as an important factor to consider in the UNFCCC Global Stocktake, a central pillar of the Paris Agreement against which its success or failure will be judged (Climate Action Network, 2022). We suggest that utilizing the UN SEEA-EA to benchmark protection and restoration actions would provide critical information on ecosystem integrity elements for the Global Stocktake to inform high-benefit / low-risk nature-based solutions in evolving NDCs. Successful implementation of the ecosystem provisions of the UNFCCC and the Paris Agreement, including decisions made at COP 25 (1.CP 25 para. 15) calling for integrated action to prevent biodiversity loss and climate change; and COP 26 (CMA/3 para. 21 and 1.CP/26 para. 38) emphasizing "...the importance of protecting, conserving and restoring nature and ecosystems, including forests ...," depends upon understanding the significance of ecosystem integrity for stable long term carbon storage and the overall health of the biosphere.

Other recent policies and guiding documents include the Glasgow Leaders' Declaration on Forests and Land Use (United Nations Climate Change, 2021), CBD post-2020 Global Biodiversity Framework (Convention on Biological Diversity [CBD], 2021), IUCN Policy Statement on Primary Forests Including Intact Forest Landscapes (IUCN, 2020), IPBES Global Assessment Report (IPBES, 2019), the New York Declaration on Forests 5-Year Assessment Report (NYDF Assessment Partners, 2019), the European Parliament resolution to protect and restore forests (European Parliament, 2020), and Indonesia's moratorium on converting primary forests and peatlands (Austin et al., 2019).

Nevertheless, there is still much work to be done at national and international levels, with the evolving Paris Rulebook and country NDC's arguably representing the largest opportunity. Translating all these international declarations into coherent national and jurisdictional policies will require an agreed-upon framework of ecosystem integrity, such as provided here, and applicable data products tools for implementation.

Future research directions

Because ecosystem integrity is such an integrative and multidisciplinary concept, research gaps are relatively extensive. We therefore do not offer an exhaustive list, but rather a prioritized assessment of future research directions to improve the understanding, valuation, and operationalization of ecosystem integrity. First and foremost, operationalizing forest ecosystem integrity at scales relevant to policy and planning that span from landscape planning (Morgan et al., 2022) to national strategies (Center for Biological Diversity [CBD], 2022) and international agreements (United Nations [UN], 2021) requires

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accurate and updated maps of ecosystem integrity and its components. Existing products (described in section "Evaluating ecosystem integrity") touch on aspects of canopy structural integrity, can be used to identify areas of remaining natural forests, and, using time series data, can locate where they have been lost (**Figure 3**). However, their ability to differentiate levels of integrity between forests is limited, and they do not account for the longer-term ecosystem dynamics that comprise functional integrity. It will therefore be helpful to leverage the time series of now decades-long satellite records such as Landsat and the Moderate Resolution Imaging Spectroradiometer (MODIS) to incorporate metrics of stability / resistance, and to capture smaller patches of high-integrity forests, such as in Shestakova et al. (2022). In boreal and temperate forests with naturally occurring stand-replacing disturbances, for example wildfire, it will be critical to accurately separate these from human disturbances, for example by using spatial pattern recognition techniques (e.g., Curtis et al., 2018).

For the purpose of primary forest protection, accurate maps of regularly updated primary forests are needed at sufficient spatial scales and accuracy to support both country-level assessments as well as local decision making. Spatial assessments of forest ecosystem integrity and components, as opposed to categorical maps of forest/no-forest or broad forest types, are particularly needed. In addition to developing countries, this information is needed in the United States, Europe, and other developed countries with little remaining primary forests. In these cases, the most ecologically mature forests for a given ecosystem type (e.g., DellaSala et al., 2022b) likely represent the highest integrity levels rather than primary forests per se (**Table 1** and **Figure 2**) and similarly require both field and remote sensing analysis to be defined and identified (e.g., Federal Register, 2022). Aside from mapping methodologies and data products, we stress the need for continued and new field monitoring programs that evaluate and track ecosystem integrity components as they are impacted by climate and human land use at various scales.

More focused scientific studies on the components of ecosystem integrity as described here (**Figure 1**) are needed to better define, quantify, and monitor integrity in different ecoregions. For example, we know relatively little about how biodiversity and ecosystem composition in many forested regions globally is responding to the combined impacts of climate change, landscape fragmentation, and land use, nor how these will continue to evolve in the future. Such understanding would facilitate management decisions to increase ecosystem integrity or limit its decline, which is particularly important for managing future risks and vulnerability of carbon stocks in the context of carbon markets and offsets (Anderegg et al., 2020). Developing methods for comprehensive yet transferable ecosystem service valuations are particularly important for both scientific understanding as well as conservation mechanisms such as Payments for Ecosystem Services and the UN System of Environmental Ecosystem Accounting.

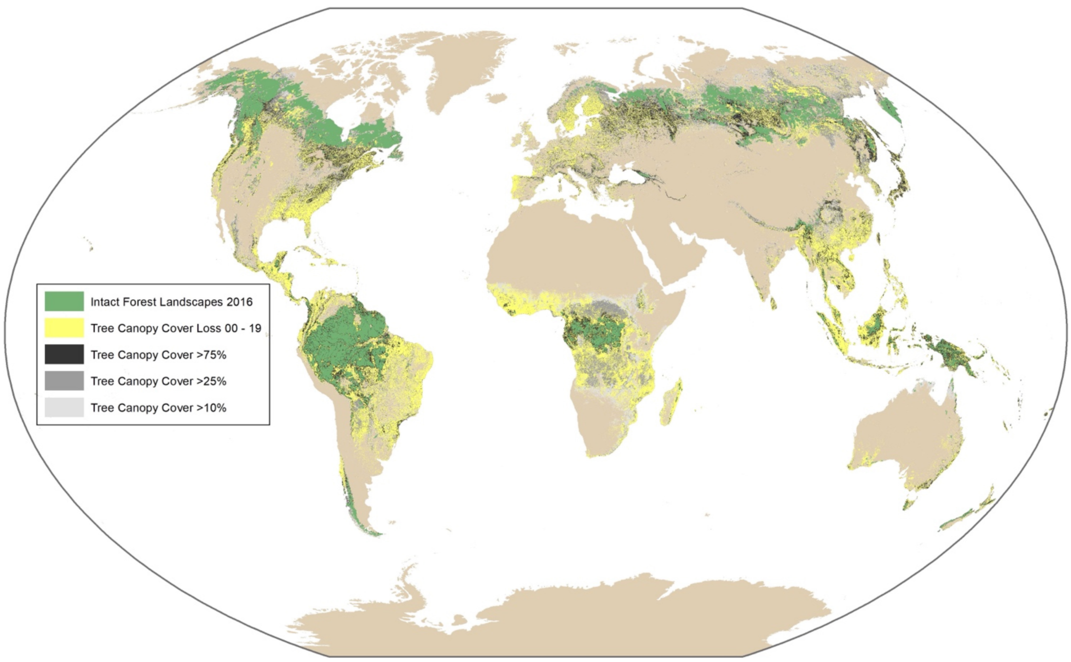

FIGURE 3
Global forest condition as indicated by metrics of Intact Forest Landscapes (IFLs), tree canopy cover, and tree canopy cover loss (from 2000 to 2019). IFLs for the year 2016 are taken from Potapov et al. (2017), and tree cover and tree cover loss outside of IFLs are from Hansen et al. (2013).

Finally, we suggest prioritizing research that optimizes the distribution of secondary forest management, including intensive plantations, to alleviate the pressure on primary and high integrity natural forests worldwide, as well as policy mechanisms needed for incentivization. Such research needs to account for regionally varying economic and equity issues in order to be effective.

Conclusion

In this paper we reviewed the components, importance, and potential for ecosystem integrity to help guide international forest policy and foster greater policy coherence across the climate, biodiversity, and sustainable development sectors. Our operating framework for forest ecosystem integrity encompasses biodiversity, dissipative structures, ecosystem processes, ecosystem stability, and the resulting ecosystem condition and services. A comparative analysis showed that, compared to forests with significant human modification, primary forests generally have higher ecosystem integrity and thus lower risk profiles for climate mitigation.

The scientific and management communities need better tools to accurately forecast the risks associated with different forest ecosystems, particularly those being managed for natural climate solutions and mitigation (Anderegg et al., 2020). Given these tools may be years or more away, we suggest focusing on ecosystem integrity is an optimal solution for categorizing forest-based risks and protecting ecosystem services. Doing so would (i) optimize investment in land carbon stocks and mitigation potential, (ii) identify stocks that provide the best insurance against risk of loss, and (iii) ensure the highest levels of benefits from ecosystem services, thereby optimizing compatibility and synergy between mitigation, adaptation, and SDGs. A number of large-scale data products exist to guide this focus. Nevertheless, there are substantial remaining gaps in terms of understanding, mapping, monitoring, and forecasting forest ecosystem integrity and its components in the midst of increasing human pressure and climate changes. Because primary forests have a higher level of ecosystem integrity than forests managed for commodity production, plantations, or degraded forests, we stress the continuing and increased need for their protection. An effective strategy is to create high carbon density strategic carbon and biodiversity reserves that include primary forests and recovering secondary forests that are quickly accumulating carbon (Law et al., 2022).

Author contributions

BR, BM, VY, and HK conceived the study. BR, BM, and HK led the writing, with contributions from CK, DD, GB, JD, RH, RB, TS, VY, and WM. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

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