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When Active Management of high conservation value forests may erode biodiversity and damage ecosystems☆

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ABSTRACT

The increase in extent and severity of disturbances such as wildfires and insect outbreaks in forests globally has led to calls for greater levels of "Active Management" (AM), including in High Conservation Value Forests (HCVF) such as old growth stands. AM includes such activities as thinning, selective logging of large trees (that are sometimes fire resistant), post-disturbance (salvage) logging, recurrent prescribed burning, and road building; singularly or in combinations. We urge caution when implementing these aspects of AM, especially in HCVF such as old growth stands, intact areas, and complex early seral forests. This is because AM may have substantial impacts on ecosystem conditions and biodiversity, and could amplify subsequent natural disturbances. We illustrate potential impacts of AM in HCVF in case studies from western North America and south-eastern Australia. AM has overlooked or downplayed collateral ecosystem damages in HCVF, including: (1) habitat needs of at-risk species, (2) thinning effects on ecosystem function, carbon emissions and biodiversity, (3) the role of stand-replacing or partial stand-replacing natural disturbances (e.g. wildfire, insect outbreaks) that produce complex early seral habitats, and (4) extensive road networks with associated impacts. We argue the underlying science to support AM may be lacking in some cases and that more scrutiny is needed to ensure objectives are supported by rigorous science, including transparency in identifying collateral damages and ways to mitigate them. Large reference areas such as extensive old growth stands are needed to assess the cumulative impacts of AM, especially in in HCVF where its potential effects on biodiversity are greatest.

1. Introduction

There have been widespread calls for "Active Management" (AM) of natural forests (Bennett et al., 2024; Davis et al., 2024), including such activities as: thinning; post-disturbance (salvage) logging following natural disturbances; recurrent prescribed burning and pile burning; and road building, singularly or in various combinations (reviewed by Bernes et al., 2015; DellaSala et al., 2022). AM is driven, in part, by efforts to manage forests for presumed resilience (defined as the ability to return to pre-disturbance conditions (sensu Nimmo et al., 2015)) in the face of rapid climate change (Popkin, 2021; Prichard et al., 2021) as well as attempts to reduce wildfire intensity (Davis et al., 2024). The prevalence of intense wildfires is increasing in some forest types (Bousfield et al., 2023; Bowman et al., 2020; Halofsky et al., 2020). Indeed, the frequency of the most intense wildfires globally increased 2.2-fold from 2003 to 2023, with the six most intense events on record occurring since 2016 (Cunningham et al., 2024). AM practices such as large-scale tree and forest biomass removals in older stands notably, and the construction of extensive firebreaks and roads, are intended to reduce fire intensity, fire spread, and limit high-severity wildfires (Prichard et al., 2021). However, seldom are the consequences to forest biodiversity and carbon stores considered (DellaSala et al., 2022).

AM also been advocated for use in the Northern Hemisphere where there has been an increase in the extent of bark beetle outbreaks as a consequence of warming, overheating and drought (Bentz et al., 2009), as well as following windstorms (Bettega et al., 2024). These outbreaks

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have, in turn, driven greater efforts to control beetle populations and conduct post-outbreak (salvage) logging (e.g. Bettega et al., 2024; Mikusiński et al., 2018), including to reduce the perceived increase in the threat of subsequent wildfires (e.g., the US Active Forest Management, Wildfire Prevention and Community Protection Act (USDA Forest Service, 2024a, 2024b)).

In some jurisdictions, calls for AM also have been in response not only to natural disturbances, but also to a perception that some old-growth forests were "open and park-like" prior to industrial fire suppression and logging, including in Australia ~230 years ago (e.g. Gammage, 2011; Pascoe and Gammage, 2021) and in western North America more than a century ago (Hessburg et al., 2021).

AM practices often target high conservation value forests (HCVF). These are exceptional forest types where protection and management constraint is most necessary (Mikusiński et al., 2021). For the purposes of this study, we define HCVF (see Areendran et al., 2020) as: (1) old-growth forests (Watson et al., 2018) that support critical habitat for threatened species (e.g. Lee, 2018; Lindenmayer et al., 2019); (2) primary forests of all seral stages (Mackey et al., 2015), including complex early seral forests dominated by an abundance of standing dead trees (snags), downed logs, fire-following shrubs, and naturally-regenerating trees (DellaSala and Hanson, 2024; Swanson et al., 2011); (3) forests with high ecological integrity (DellaSala et al., 2025); (4) large intact areas (including those that are roadless) (Ibisch et al., 2016), and (5) forests classified as Endangered or Critically Endangered under the IUCN Red List Ecosystem approach (Keith et al., 2015). HCVF can also include localized biodiversity hotspots and/or areas that support many endemic taxa (Mittermeier et al., 1998). We note that HCVF often contain large, old trees that are increasingly rare on a global scale (Bettega et al., 2024; Lindenmayer and Laurance, 2017).

Here, we argue that caution is needed when implementing AM practices in HCVF that involve extensive biomass removal, for instance, by post-disturbance logging and commercial thinning, along with road building. This is because the underlying science to support some aspects of AM in HCVF may not be robust nor clearly understood. Moreover, AM has the potential to lead to significant degradation of the ecological integrity of HCVF (Table 1). We adopt the definition of degradation developed by DellaSala et al. (2025), and consider it as the loss of forest ecosystem integrity measured by comparing reference areas (unaltered conditions) to human-altered sites in terms of native-species composition, key ecological functions and processes, and keystone structures (such as large old trees).

We present case studies from south-eastern Australia and western North America to illustrate our concerns about the types of AM impacts that can lead to degradation of HCVF. Our focus is on tall, wet, old-growth and multi-aged forests of south-eastern Australia (Government of Victoria, 2024), old-growth forests and complex early seral forests in western North America (Swanson et al., 2011; USDA Forest Service, 2024a, 2024b), and rare ecosystems otherwise located in national parks such as Giant Sequoia (*Sequoiadendron giganteum*) groves that require high-intensity fire to regenerate effectively (Hanson et al., 2024a, 2024b). We also make limited reference to some examples of AM from forests in Europe such as those following beetle attacks and windstorms (e.g. Mikusiński et al., 2018; Popkin, 2021; Bettega et al., 2024). We illustrate how disturbance regimes and post-disturbance recovery processes may be disrupted by AM in HCVF, with subsequent impacts on their ecological integrity. Our findings may have policy and management implications necessary to avoid forest degradation from AM. This is especially important as many disturbance-adapted systems are dealing with unprecedented levels of compounded human interventions, that are likely amplifying climate impacts in a way that could further degrade ecosystems and reduce their ability to adapt (Paine et al., 1998; Lindenmayer et al., 2022a).

Table 1
Exemplary forms of degradation in High Conservation Value Forests (HCVF) that can result from Active Management (AM). A larger list of degradation indicators is presented in (DellaSala et al., 2025).

Degradation indicator	HCVF consequences	Source
Threatened, endangered, imperilled, range-restricted species	Loss of critical habitat, competition with invasive species.	(Bettega et al., 2024; DellaSala et al., 2025; Lee, 2018)
Cumulative effects	Ecosystems pushed beyond tipping points that trigger a state-shift.	(Lindenmayer et al., 2011)
Natural disturbance dynamics	Loss of complex early seral forests. Changes in fire behaviour and insect infestations.	(Swanson et al., 2011; Lesmeister et al., 2021; Taylor et al., 2020; Millikin et al., 2024)
Forest microclimate	Increased drying and higher wind speeds along edges.	(Ma et al., 2010; Trentini et al., 2017; Hao et al., 2024)
Tree mortality	Elevated tree loss beyond that from natural disturbance.	(Baker and Hanson, 2022)
Soils and below-ground processes	Soil nutrients and soil structure changes, soil compaction, mycorrhizal disturbances.	(Marchi et al., 2014; Picchio et al., 2012; Hwang et al., 2020; Bowd et al., 2019)
Carbon sequestration and stores	Increased emissions, impaired sequestration, and reduced carbon stores.	(Mackey et al., 2022; Harmon et al., 2022; Bartowitz et al., 2022)
Hydrology	Road stream intersections and high road densities alter peak flows, runoff, sedimentation, and water quality.	(Ibisch et al., 2016)
Forest fragmentation and landscape change	Landscape-scale logging and road building shifts seral stages to an abundance of young managed stands and small, isolated patches of old-growth forest.	(Laurance and Arrea, 2017; Taylor and Lindenmayer, 2020; Ma et al., 2023)
Large old trees	Selective logging, post-disturbance logging, and commercial thinning reduce populations of large trees.	(Russell et al., 2006; Thorn et al., 2018)

2. Some historical origins of "AM"

The term 'active management' (AM) is a superficially simple concept that conveys implied Western colonial perspectives on vegetation management. These should be made explicit in any discourse on the subject, as they fundamentally contrast with the perspectives of the Indigenous nations colonised in Australia and the Americas. The Canadian Cree, for example, maintain that human management of animals and the environment is not possible because humans do not dominate; instead, animals "control the hunt" (Bearskin et al., 1989). Similarly, many Indigenous Australian nations such as the Ngarrindjeri consider themselves to hold 'kinship' relationships with "all elements of the environment" (Ngarrindjeri Tendi, N.H.C., and Ngarrindjeri Native Title Management Committee, 2007). In contrast, AM centres human agency so that humans are placed above other species as managers. This reflects the Western cultural/Biblical maxim of human dominion, to rule over and "subdue the earth." As a perspective arising from the agricultural revolution of the Fertile Crescent, it is entirely foreign to the perspectives of colonised Indigenous nations.

AM further differentiates itself from Indigenous views by specifying that management cannot be passive. Western Australian Wadandi Pibulmun Yunungjarlu Elder Wayne Webb specified that a regime of low fire frequency was essential to maintain low wildfire hazard in Red Tingle (*Eucalyptus jacksonii*) forest (Webb, 2022). This claim is supported by extensive subsequent scientific analysis (Zylstra et al., 2022).

Such management does not qualify as 'hazard reduction' under one Australian legislative definition, as "appropriate fire regimes" must be actively delivered by "*controlled application*" (NSW Government, 1997). This example highlights a central conflict between AM and Indigenous perspectives. The natural processes of ecology are enshrined by many Indigenous Australian nations as part of the *Tjukurpa* (Bowdler, 2016) or Law, that defines the way we interact with the land. In contrast, AM is premised on the notion that such natural processes are insufficient to satisfy human concerns such as safety and must themselves be managed. Thus, the U.S. "Healthy Forests Restoration Act of 2003" mandated mechanical thinning to modify fire behaviour (U.S. Government, 2023), despite the fact that natural self-thinning is so well-documented (see Westoby, 1984) that some ecologists refer to it as "the only law in plant ecology" (Li et al., 2000). In 2014, Australia began to adopt this same approach (Ximenes et al., 2017), laying the groundwork for mechanical thinning of forests to replace natural (self) thinning as a core safety responsibility. As we outline in this article, these decisions did not arise because evidence showed any failure in the natural processes. Instead, such AM approaches arise from the belief that management or dominion is a moral imperative, that 'unmanaged' country is dangerous, and that the colonisers must subdue and profit from the land they are colonising. Indigenous nations modify specific places through actions such as hunting or burning using tightly defined guidelines that work in cooperation with Law and the ecology of the land (Prober et al., 2013), whereas AM assumes dominion over these processes.

3. Case study #1: tall, wet forests in south-eastern Australia

The tall, wet, old-growth and multi-aged forests of south-eastern Australia support a wide range of threatened species, including Endangered or Critically Endangered animals and plants (Lindenmayer et al., 2023). The extent of old growth tall, wet forest in Victoria, for instance, has declined by >77 % in the past 25 years from wildfire and logging (Lindenmayer and Taylor, 2020). HCVF dominated by stands of Mountain Ash (*Eucalyptus regnans*) have been assessed as both a Critically Endangered ecosystem (Burns et al., 2015) and a Threatened Ecological Community (Lindenmayer et al., 2023).

The natural fire regime in many tall wet forests is crown-scorching, stand-replacing wildfire that occurs on average every 75–150 years (McCarthy et al., 1999). When wildfire occurs in old growth stands, it produces a pulse of biological legacies that creates complex early seral or multi-aged forests for many taxa (e.g. the Critically Endangered Leadbeater's Possum, *Gymnobelideus leadbeateri*) (Lindenmayer et al., 2019). The severity of wildfires is lowest in old growth stands (Taylor et al., 2014) and in stands that have remained unlogged (Lindenmayer et al., 2022b). Nevertheless, there have been widespread calls for AM of HCVF (Bennett et al., 2024), including thinning of forests in National Parks and nature reserves (e.g. AFPA, 2020; Forestry Australia, 2024) and the construction of firebreaks (up to 1500 km in length and 40 m or more wide), in an attempt to limit the spread of wildfires, as well as to create points from which to ignite back burns as part of fire-fighting efforts (Government of Victoria, 2024).

3.1. *Degradation from extensive firebreaks*

An inherent problem with AM in the tall, wet forests is that whilst areas of old growth are increasingly uncommon (but critical for biodiversity conservation), firebreaks have been constructed through these stands and other HCVF, such as intact cool temperate rainforest (Government of Victoria, 2024). These firebreaks have led to the logging of hundreds of large old trees (some >3 m in diameter), which are keystone structures that are increasingly rare because of rapid rates of decay and very limited recruitment (Lindenmayer et al., 2024a). Large old trees are a critical habitat attribute for a suite of cavity-dependent vertebrates, including Endangered and Critically Endangered species, the declines of which are linked to loss of old trees (Lindenmayer et al., 2024a).

Additionally, some landscapes supporting HCVF in south-eastern Australia are characterized by high levels of fragmentation as a result of past logging (Taylor and Lindenmayer, 2020). Adding a dense network of firebreaks through large parts of the forest estate would further fragment these HCVF (Ibisch et al., 2016; Laurance and Arrea, 2017) and undermine their ecological integrity (sensu DellaSala et al., 2025). Importantly, as roads (and likely also firebreaks) are major point sources of human-caused ignitions in the forests of south-eastern Australia (Collins et al., 2015), expanded access to forests may increase rather than reduce the prevalence of fire. Finally, given the large distances over which wildfires can spot (and trigger further fires) (e.g., up to 13.9 km; see Storey et al., 2020), it is unclear how effective firebreaks actually are in reducing the spread of fire, even relatively wide ones (e.g. >60 m).

3.2. *Degradation from thinning operations*

Extensive thinning has been proposed as part of AM in the tall, wet HCVF of south-eastern Australia (Keenan, 2024). Thinning trials have removed approximately half of the stand basal area of some forests. Thinning in HCVF can degrade habitat suitability for threatened species. For example, the dense structure and understoreys of some tall, wet HCVF that are the target for thinning provides essential habitat for a large number of closed-canopy species, such as small-bodied bird taxa (Loyn, 1985) and arboreal marsupials (Lindenmayer et al., 1991).

Thinning operations in tall, wet HCVF generates large amounts of carbon emissions, typical of other kinds of timber harvesting in south-eastern Australia (see Keith et al., 2014b), a problem that is further magnified because most of the timber removed is used as firewood. There is also strong evidence that thinning either has limited impact on fire severity, or in some cases may elevate it (Taylor et al., 2020) – an issue well documented in forest management manuals in the Australian States of Victoria and Tasmania (Buckley and Cornish, 1991; Fagg, 2006; Sebire and Fagg, 1997).

Finally, thinning operations are conducted with heavy machinery and its widespread use would, in turn, demand an extensive road network to move that equipment around a forest landscape. Such thinning operations may compact soils, as has been documented not only in tall, wet forests (Rab, 1998) but in many forest types globally (e.g. Marchi et al., 2014; Picchio et al., 2012; Hwang et al., 2020). Thinning operations also may alter microclimates such as elevated surface temperatures and reduced moisture levels (e.g. Ma et al., 2010; Trentini et al., 2017; Hao et al., 2024).

3.3. *Degradation from inappropriate large-scale prescribed burning*

Widespread prescribed burning has been recommended as a key part of AM in extensive parts of the forest estate of south-eastern Australia (Keenan, 2024). Current evidence suggests that prescribed burning may have only limited benefits in reducing damage to human infrastructure under extreme fire weather conditions (Gibbons et al., 2012). HCVF that have been subject to frequent prescribed fire preceding subsequent wildfire, have exhibited significantly impaired post-wildfire recovery patterns relative to intact forests that burned naturally but were not subjected to previous prescribed fire (Driscoll, 2024). Some forest ecosystems, including tall, wet HCVF, are not well adapted to recurrent, frequent prescribed burning, which may trigger their collapse or lead to them becoming more (not less) flammable (Lindenmayer and Zylstra, 2024). Additionally, prescribed fires can increase the rates of loss of keystone forest structures such as large old trees by burning parts, or all of them (Holland et al., 2017).

4. Case study #2: threats to HCVF in western North America

The USDA Forest Service is a major federal landowner in the United

States and responsible for managing ~77.2 M ha across the national forest system. AM is focused primarily on commercial thinning and post-fire logging through timber sales to logging companies (USDA Forest Service, 2024a, 2024b) and particularly within HCVF, such as mature and old-growth forests, where rates of logging that target mature forests are projected to double between 2020 and 2070 for wildfire and insect concerns (USDA Forest Service, 2024a, 2024b) (Fig. S1). Moreover, post-fire salvage logging often targets HCVF consisting of complex early seral forests (DellaSala and Hanson, 2024). AM degradation effects include: (1) the cumulative effects of thinning on ecosystem function and carbon emissions (Campbell et al., 2012), (2) compromising the role of high-intensity fire as a key successional process in forests where high-intensity wildfire is part of the natural fire regime (i.e., 'circular succession' of old growth to complex early stages and back again (see DellaSala et al., 2025)), (3) the detrimental impacts of an expanded road network (Ibisch et al., 2016; Balch et al., 2017), and (4) undermining habitat suitability for, and the persistence of, threatened species (DellaSala et al., 2022).

4.1. Degradation of threatened species habitat in HCVF

AM in HCVF may have substantial impacts on threatened species. For example, the Spotted Owl (*Strix occidentalis*) has neutral or positive responses to mixed-intensity wildfires that occur in HCVF old-growth habitats, including very large wildfires (Lee, 2018). However, mechanical thinning is associated with a 43 % loss of spotted owl occupancy (Stephens et al., 2014), and post-fire logging of HCVF can result in an even greater reduction in site occupancy (Lee, 2018). The Pacific Fisher (*Pekania pennanti*), an old growth associated species, also actively forages in mixed-intensity fire areas, especially complex early seral forest HCVF (Hanson, 2015), but avoids old-growth HCVF subject to mechanical thinning (Garner, 2013). Some researchers have reported generally negative effects of high-intensity fire on these two imperilled species (Thompson et al., 2021; Jones et al., 2024; McGinn et al., 2025), but did not account for the adverse impacts of post-fire logging (Hanson, 2015; Hanson et al., 2021; Bond et al., 2022). When post-fire logging is disentangled from high-intensity fire, the evidence indicates a strong negative effect of post-fire logging and generally positive effects of mixed-intensity fire in general, and high-intensity fire in particular (Hanson, 2015; Lee, 2020; Hanson et al., 2021).

The Black-backed Woodpecker (*Picoides arcticus*) utilises complex early seral HCVF by preferentially selecting dense, older forests that have recently experienced high-intensity fire, but the species is largely extirpated by post-fire logging in the same habitat (Hanson and North, 2008). In the absence of fire, this woodpecker utilises dense old-growth HCVF with high dead standing tree (snag) densities (Tremblay et al., 2010) and is harmed by mechanical thinning, which removes many mature and old-growth legacies. This leaves far fewer dead standing trees than when such logged forests burn at higher intensities (Hutto, 2008).

4.2. Degradation from commercial thinning

Commercial thinning impacts are prominent even in such iconic HCVF as the Giant Sequoia (*Sequoiadendron giganteum*) groves within Sequoia National Park, California, USA. Thinning not only removes important wildlife habitat and stored carbon in these old-growth groves, but also disrupts key ecological processes. For example, it crushes and kills an estimated 83 % of the natural, post-fire Giant Sequoia seedlings that are most abundant in high-intensity fire patches (Hanson et al., 2024b) (Fig. 1).

The role of thinning in reducing fire intensity has mixed effects (e.g. Banerjee, 2020). A review (Davis et al., 2024) reported no statistically significant differences in subsequent wildfire intensity among thinning plus prescribed fire, thinning plus pile burning, and prescribed fire alone. Davis et al. (2024) nevertheless concluded that thinning plus prescribed fire had the "largest" effect on wildfire intensity, and promoted such management, particularly in dense, old HCVF even though fuel reduction efficacy was equivalent with burning alone. Importantly, Davis et al. (2024) did not consider the impact of widespread thinning-plus-burning in old-growth HCVF on threatened wildlife or on the level of tree mortality in, and carbon emissions from, such forms of AM in old-growth HCVF. This is a major omission because thinning conducted ostensibly to reduce tree mortality from fire or bark beetles can kill significantly more trees than it prevents from being killed (Fig. 2; Baker and Hanson, 2022). In addition, for a given unit of area, thinning can increase CO_2 emissions by three to five times relative to fire alone, largely because whole trees are removed from the forest, but only a minor portion of these trees ends up in a lumber product, while most is burned as fuel or decomposes on site (Campbell et al., 2012). This estimate may be conservative because even the biggest wildfires consume only ~1 % of total tree biomass, including all tree sizes and fire intensities (Harmon et al., 2022).

Fig. 1. Vigorous natural post-fire Giant Sequoia regeneration in a high-intensity fire patch, Nelder Grove (top), and removal of habitat and crushing and killing of the young giant sequoia forest (bottom) in the same high-intensity fire patch within Nelder Grove via mechanical thinning and burning (photos by Chad Hanson).

4.3. Degradation from road networks

The increase of logging in HCVF (see Fig. S1) would require thousands of km of logging roads (USDA, 2024a, 2024b), most of which the agency cannot feasibly maintain currently, and many of which are

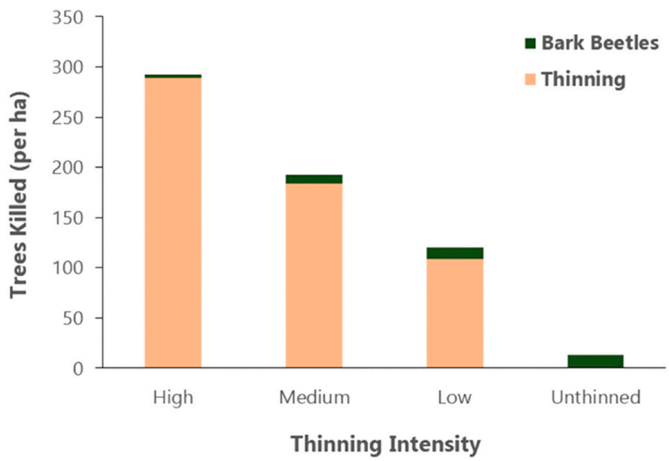

Fig. 2. Data from Fettig et al. (2012), demonstrating that thinning kills far more trees than it prevents from being killed by drought and native bark beetles (figure courtesy of B. Baker).

failing. Seldom do managers consider the duality of having such an extensive road network which, on the one hand, allows access to the public and firefighters, but conversely has substantial ecological impacts (Ibisch et al., 2016). For example, an expanded road network will likely create problems for wildlife (hunting, poaching, vehicle collisions), increase sources of ignitions (Balch et al., 2017), lead to chronic sedimentation in waterways (Ibisch et al., 2016), and increase pathways for invasion of exotic species (Gelbard and Harrison, 2003).

5. Case study #3 AM in post-bark beetle outbreak forests of the Western USA

Conifer-dominated HCVF of western North America have been shaped by, and co-evolved with, tree-killing bark beetles (Scolytinae) for millennia (Brunelle et al., 2008). Like wildfires, bark beetles are natural disturbances in these forests and play keystone roles in driving forest structure and composition. Their actions perpetuate within-stand and between-stand heterogeneity, support continuity in ecological processes, and generate biological legacies often associated with high biodiversity (Buonanduci et al., 2023; Przepiora et al., 2020; Winter et al., 2015). Their effects on forests are similar to those that influence the development of complex early HCVF generated by stand-replacing fire. Despite this, bark beetles are often seen as destructive, engendering AM to stop or reduce their effects before, during, and after outbreaks (Black et al., 2013). Additionally, a warming climate has driven an increase in the extent of bark beetle outbreaks (Bentz et al., 2009) as well as that of wildfires (Halofsky et al., 2020), creating an even stronger impetus for their control and to conduct post-outbreak logging to reduce a perceived increase in the threat of catastrophic fire.

Bark beetles are eruptive (not cyclic), developing sporadic outbreaks when climatic conditions support greater beetle survival and increased tree stress (Bentz et al., 2009). This accounts for the regional distribution of outbreaks rather than a distributed occurrence in patches in 'unhealthy forests' as is often implied. As long as favourable environmental conditions exist, an outbreak will proceed regardless of management efforts (reviewed in Six et al., 2014). Despite this, a widespread belief commonly held by managers and policy makers is that logging, either through commercial thinning or "sanitation cuts," is an effective way to control bark beetle populations. Likewise, logging is seen as an effective way to reduce the threat of severe wildfire following beetle outbreaks. This perspective is reflected in forest plans and legislative bills such as the Active Forest Management, Wildfire Prevention and Community Protection Act introduced in 2021 (U.S. Congress, 2021), and the Fix Our Forests Act (U.S Congress, 2024). These and other bills circumvent the National Environmental Protection Act to allow emergency exemptions without more comprehensive environmental impact statements. As the extensive outbreaks of the 1990s–2010s have now collapsed, most attention has shifted from outbreak suppression (itself of questionable efficacy, Six et al., 2014) to 'emergency' fuel reduction treatments in post-outbreak forests as part of the national 'wildfire crisis strategy' (e.g., many forest plans now include removal of beetle kill (U. S. Forest Service, 2024a, 2024b)). Given that beetles target mature trees, these plans often target old growth and mature HCVF.

5.1. Fire in post-bark beetle-outbreak forests

The co-occurrence of large wildfires and bark beetle outbreaks has led to the perception that post-outbreak forests fuel catastrophic fire. However, while some models have predicted more intense fires in beetle-affected stands (Derose and Long, 2009; Page and Jenkins, 2007; Schoennagel et al., 2012), others (and most empirical studies) have found small effects, no effect, or even dampening effects on fire intensity, severity, and spread, with effects varying depending upon time since outbreak and weather conditions (Jenkins et al., 2008; Romualdi et al., 2023; Sieg et al., 2017; Simard et al., 2011). Importantly, post-outbreak stands do not have a greater likelihood of high-intensity crown fires that pose the greatest threat to human communities (Simard et al., 2011). Crown fires depend on canopy fuel connectivity and bulk. In early-stage post-outbreak forests, the amount of canopy fuels remains unaffected. While flammability can be higher in dead than live needles (which themselves are highly flammable), dead needles are retained for only 1–3 years post-tree death (Romualdi et al., 2023). After needle drop, needle-free snags create gaps in the canopy, slowing or even stopping the spread of crown fire (Collins et al., 2012; Romualdi et al., 2023). Increases in surface fuels from dropped needles and twigs are ephemeral due to rapid decay (Simard et al., 2011) and larger fuels that accumulate as dead trees fall can support hotter ground fires but contribute little to fire spread (Donato et al., 2013).

In general, AM in post-outbreak forests to reduce the threat of fire is unwarranted (Leverkus et al., 2021). Forests affected by bark beetles have not burned more extensively or severely than forests not affected by beetles or those that were salvage logged (Bigler et al., 2005; Kulakowski and Veblen, 2015). In cases where severe fires have occurred, fire behaviour has been driven predominantly by fire weather (Hart and Preston, 2020).

5.2. Degradation from AM in post-bark beetle-outbreak HCVF

AM in post-outbreak HCVF can degrade ecological integrity in many ways (Fig. 3). For example, post-disturbance "salvage" logging removes most or all beetle-killed trees and, in many cases, also removes trees that survived the outbreak (Collins et al., 2012; Sullivan et al., 2010). It also has short- and long-term impacts on HCVF by altering successional trajectories and reducing overall forest resilience (Thorn et al., 2018).

Resilience in post-outbreak forests is typically high due to the presence of residual living trees of multiple diameter classes, and legacy understory plant and soil microbial communities (Amaroso et al., 2013; Kayes and Tinker, 2012). Because of this, succession proceeds in unmanaged stands more rapidly than if they are logged (Donato et al., 2013; Griffin et al., 2013; Rhoades et al., 2020; Steinke et al., 2020). Unlogged HCVF regenerating after outbreaks retain functional biotic communities and heterogeneous stand structures. They typically undergo succession as complex early seral forests interspersed within a matrix of older forest, returning to a similar state over time while remaining HCVF throughout succession (i.e. resilience).

In contrast, removal of beetle-killed trees alters successional pathways (Fig. 3) and the distribution and structure of dead wood legacies that are crucial for maintaining biodiversity and the soil conditions most supportive of recovery (Thorn et al., 2016). When surviving trees are also cut, not only is recovery slowed for years, if not decades, but

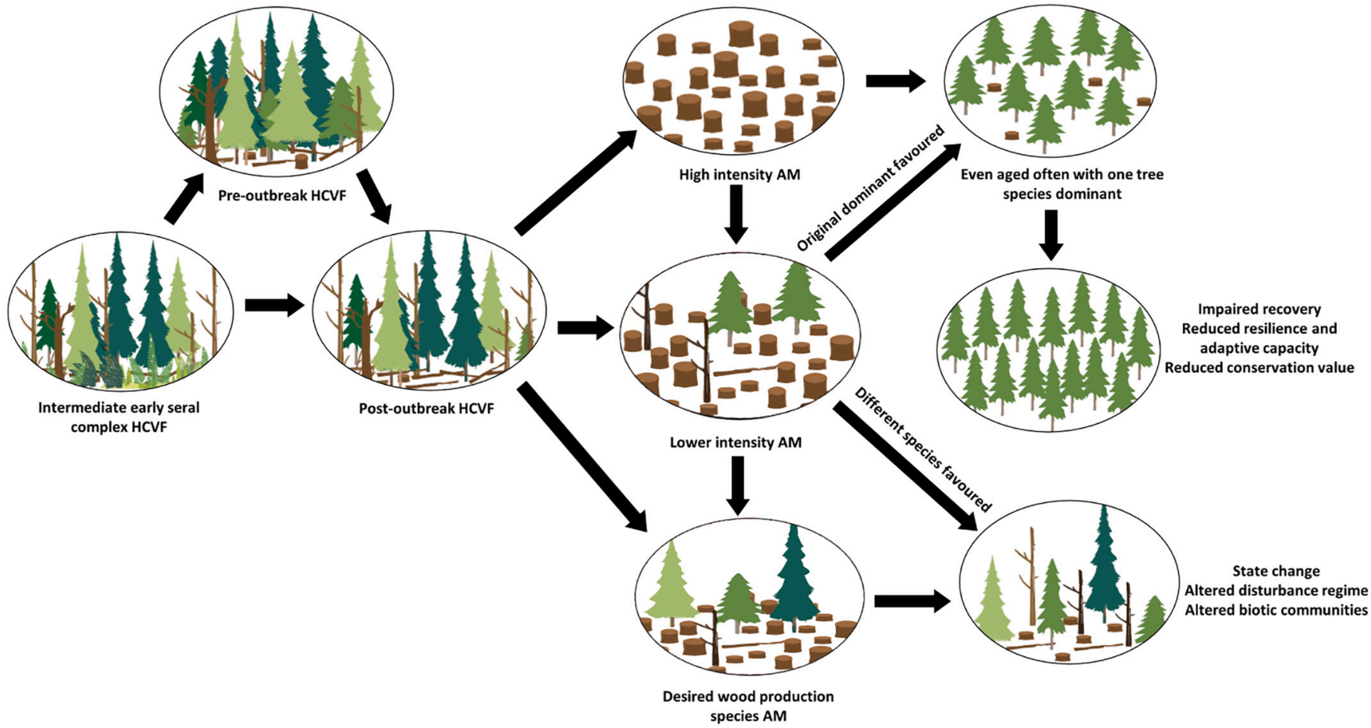

Fig. 3. Circular succession in beetle-affected forests in the absence of AM showing post outbreak recovery versus shifts to different trajectories and potentially different states as well as losses of key ecological values of HCVF associated with AM.

biological communities and successional pathways are disrupted (Choma et al., 2023; Sullivan et al., 2010). Adaptive capacity of HCVF to beetle kill may be reduced by the removal of surviving trees with greater genetic resistance to beetles and tolerance to new climatic conditions (Six et al., 2021). Forests treated with intensive fuel reduction or salvage logging tend to regrow as even-aged, relatively homogenized stands (Fig. 3) with lower biodiversity and resilience to future disturbances, including from bark beetles and fire. Further, in logged stands, slash and drier conditions can increase flammability and fire spread. As regeneration progresses, a continuous fuel bed of young trees often results, quite different than the heterogenous dead wood/multi-aged living tree structure maintained in most unmanaged stands (Hood et al., 2017; McIver and Ottmar, 2018). AM treatments that preferentially retain or remove particular tree species to promote 'desired species' and presumably reduce risk of future outbreaks can change biological communities, circumvent natural succession, and hinder adaptive shifts in composition responsive to a changing climate (Morin et al., 2018; Popkin, 2021). Such changes can alter disturbance dynamics far into the future (Leverkus et al., 2021).

5.3. Degradation effects on carbon

While post-outbreak forests were once viewed as major sources of carbon (Kurz et al., 2008), subsequent studies have found release is often slow and quickly offset by regeneration that typically occurs more rapidly in unlogged stands (Hansen, 2014). This is mainly because carbon shifts from live to dead pools where it slowly decomposes. In contrast, salvage logging and cut and burn treatments release large amounts of carbon, even when a portion of the cut wood is converted to timber products (Campbell et al., 2012; Donato et al., 2013).

6. General discussion

Our case studies from south-east Australia and western North America illustrate that there are circumstances where AM, particularly in HCVF have cumulative negative impacts as a form of forest degradation that undermines ecological integrity, and impairs the capacity of ecosystems and species to adapt to compounded disturbances (DellaSala et al., 2025). Indeed, there can be marked differences in structural complexity, habitat suitability, disturbance regimes, and post-disturbance recovery trajectories between: (A) HVCF subject to natural disturbances where biodiverse, complex early successional environments are created and stands are maintained as high conservation value areas, just at a different seral stage (Fig. 4), and (B) HCVF where pre-disturbance AM (e.g. commercial thinning) and post-disturbance AM (e.g. salvage logging) degrades habitat suitability for threatened species, elevates forest flammability, generates large carbon emissions, and impairs post-disturbance recovery (Fig. 4). These impacts include the potential to disrupt natural "circular succession" processes (see Fig. 3, DellaSala et al., 2025) and thereby undermine the biodiversity values of, and key ecological processes in, HCVF. Our findings highlight the need for caution in the application of AM in HCVF such as biodiversity hotspots (e.g. Mikusiński et al., 2018), complex early seral forests (Swanson et al., 2011), and old growth stands (Lindenmayer et al., 2019).

6.1. Consideration of broader set of issues in guiding appropriate management of HCVF

Whilst the primary focus of most AM programs has been to limit the risk of wildfires and beetle outbreaks, we argue that the ways AM may contribute to forest degradation needs greater recognition. First, there is a need to consider the impacts of AM on forest biodiversity via collateral ecosystem damages and carbon emissions (DellaSala et al., 2025). This includes not only increased losses of important habitat structures (e.g. large trees and other key biological legacies) as a result of activities such as post-disturbance salvage logging (e.g. see Mikusiński et al., 2018; Bettega et al., 2024), establishing firebreaks, and recurrent prescribed burning, but also increased levels of forest fragmentation as a result of extensive networks of firebreaks and roads through otherwise intact forests. In addition, modification of forests due to AM activities like commercial thinning may: (1) result in soil compaction (Marchi et al., 2014; Picchio et al., 2012; Hwang et al., 2020); (2) alter microclimate

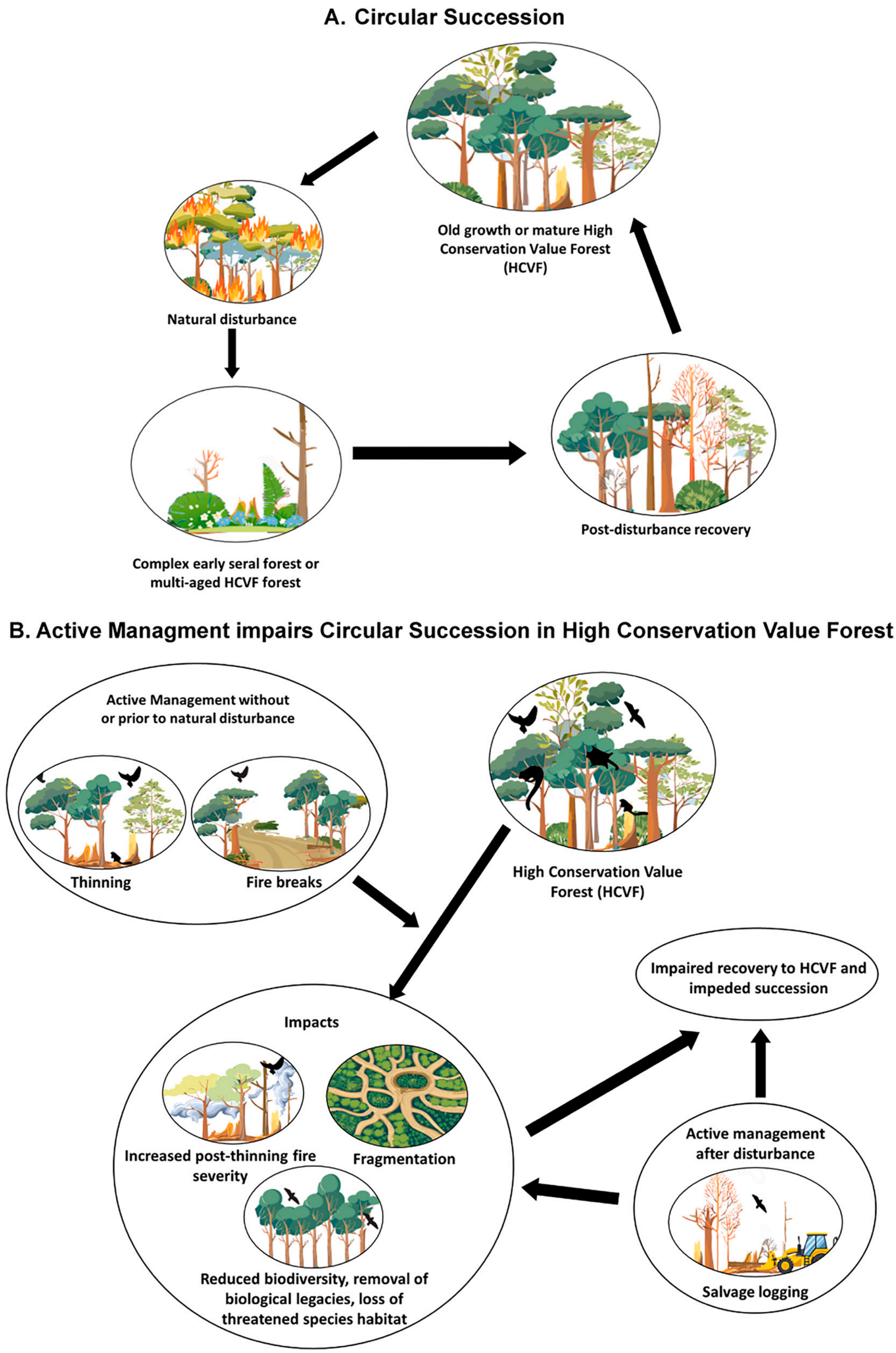

Fig. 4. Conceptual representation of the differences in patterns of "circular succession" in HCVF subject to natural disturbance that then recovery via complex early successional stages (Part A) vs. HCVF where AM is applied either prior to disturbances, such as insect outbreak or wildfire (e.g. thinning operations), and/or following natural disturbances (e.g. salvage logging) (Part B). For further discussion of "circular succession" (see DellaSala et al., 2025).

conditions (Hao et al., 2024; Trentini et al., 2017); and (3) make forests more flammable (e.g. coastal rainforests of western Canada, Millikin et al., 2024). Changes in microclimate also may have negative impacts on, for example, old growth stands that can act as refugia for temperature-sensitive bird species against the backdrop of climate change and increasing wildfires (Betts et al., 2017; Lesmeister et al., 2019).

A second issue is that some forms of AM such as intensive thinning can be especially degrading to HCVF (see Baker et al., 2023) compared with self-thinning that will occur without human intervention (Westoby, 1984), and potentially enable forests to transition to a less flammable state as they mature (Zylstra et al., 2023). Thinning as part of AM also ignores the need for, and circumvents, natural selection to climate change and increased disturbance severities (Kuparinen et al., 2010) because humans select which trees die or persist rather than natural stressors.

Third, the Green House Gas (GHG) emissions from thinning, all forms of logging, roads, and the establishment of firebreaks need to be reconsidered, especially as increasing carbon emissions are a contributor to climate change and, in turn, to greater prevalence of extreme fire weather (Canadell et al., 2021). In the global climate emergency, it is important to keep carbon in forests even after severe wildfire (Harmon et al., 2022; Keith et al., 2014a), and avoid logging that generates large amounts of emissions (Keith et al., 2014b).

6.2. *Consideration of overall disturbance burden*

AM may add cumulative disturbances to HCVF, some of which already have been disturbed by logging and wildfire, are at risk of elevated fire severity following past logging (Thompson et al., 2007; Wilson et al., 2022; Mackey et al., 2023; see Lindenmayer and Zylstra, 2024), and post-disturbance salvage logging. We argue that land managers need to carefully consider the total and cumulative disturbance burden in HCVF (Driscoll, 2024). That is, perturbations arising from all kinds of disturbances (natural disturbance, human disturbance, climate change, and their interactions) and especially where that burden is very high and may leave HCVF susceptible to yet more disturbances that lead to their degradation.

The total disturbance burden can include cumulative changes to forest structures (i.e. stand level removals of too much biomass, large trees, legacies) and spatial patterns of forest cover (e.g., landscape fragmentation from logging, roads, fire breaks). The overall disturbance burden can be important in increasing the risks of invasion by exotic plants that undermine forest integrity (e.g. Jo et al., 2024) and can shift HCVF into novel states (Lindenmayer et al., 2011). We note that some advocates of AM suggest that disturbance exclusion can be important (Bennett et al., 2024), and we argue that this may be especially appropriate in some HCVF, including where old growth stands are especially rare and post-disturbance regrowth support many kinds of biological legacies that provide habitat for threatened species. However, we suggest that the exclusion of disturbances focus first and foremost on AM factors that trigger cumulative disturbance loads, as they are the main disturbances that are within our control.

6.3. *Re-examination of pre-European colonization of forest conditions in Australia and western North America*

Often the rationale for AM (see Keenan, 2024) is that forests prior to European colonization were more open and park-like relative to current conditions (Gammage, 2011; Hessburg et al., 2021) and interventions such as commercial thinning are merely returning forests to their "natural state" (Pascoe and Gammage, 2021). This is despite evidence that some areas of forest were much denser than previously believed (Baker et al., 2023). The extent of historic versus contemporary conditions, and the collateral damages of moving ecosystems too quickly in a given direction, can shift baseline conditions inappropriately to more recent timelines that may not reflect historic periods (i.e., shifting baseline perspective in DellaSala et al., 2022). Indeed, for tall, wet forests of south-eastern Australia, the concept of open and park-like forest is a flawed baseline as the "natural" state comprised dense stands of tall trees with a mesic understorey and shrub layer (Lindenmayer et al., 2024b). AM could produce a forest structure that is the anti-thesis of "natural" forest conditions, amounting to a clear form of degradation. Moreover, the scale and intensity of some kinds of AM (e.g. such as over-burning and excessive thinning in western North America (see DellaSala et al., 2022)) and south-eastern Australia appears likely to be well beyond the kinds of activities practiced by Indigenous peoples before colonization (e.g. Gott, 2005). A related issue is that the activities aimed at creating forest conditions now that are similar to those believed to have been "natural" approximately 220–400 years ago, may be mismatched to the quite different climatic regimes that characterize many contemporary forest environments.

6.4. *The need for long-term studies and the protection of large reference HCFV areas*

There is increasing pressure to implement AM in many HCVF globally, in part as a response to climate change and altered fire regimes and beetle outbreaks (Hessburg et al., 2021; Prichard et al., 2021; Popkin, 2021). However, some activities under AM may have limited scientific evidence to support their application (Mikusiński et al., 2018; Popkin, 2021). We therefore call for more well-designed empirical studies to better evaluate the efficacy of degrading practices such as the establishment of extensive networks of firebreaks, roads, and pre- and post-disturbance logging. These studies need to be long term investigations as some forms of AM such as commercial thinning, post-disturbance logging, and/or repeated prescribed burning (i.e., over burning) may contribute both to forest degradation (DellaSala et al., 2025) (Table 1) and have limited effectiveness in the medium term (e.g. 10 years and longer) (Zylstra et al., 2022).

Given uncertainty in the efficacy of AM, coupled with its potential to contribute to degradation in HCVF, we argue there is a need to promote heterogeneity in management actions, including ensuring there are large areas (e.g., landscape scale, intact forest blocks and patches of old growth and complex early seral forest) with no intervention. These can act as reference areas in studies that contrast the impacts of intervention versus no intervention.

6.5. *Examining alternatives to some kinds of AM practices*

There is an emerging body of research suggesting that there are ecological controls on wildfires that can strengthen as some types of forest mature (Lesmeister et al., 2019; Zylstra et al., 2023). Similarly, we suggest there will be an increasing role for new technologies such as drones and autonomous aerial vehicles to help more quickly detect human-caused ignitions and assist in rapid management responses (Roldán-Gómez et al., 2021). Such technologies could reduce the need for (and hence the recurrent costs of) AM practices like widespread thinning and road building. Indeed, cost-effectiveness needs to be a key part of exploring approaches and technologies that are an alternative to AM. For example, the costs of repeated thinning operations and/or the ongoing maintenance of firebreaks may exceed those of natural ecological controls on fire behaviour (Zylstra et al., 2023) (including following beetle outbreaks, Six et al., 2021). Similarly, AM practices might be more cost-effective and effective for human safety if they are implemented close to structures and not in remote areas where there are no human communities (Gibbons et al., 2012).

6.6. *Cases where AM is appropriate*

There are certainly cases where AM is appropriate and is a necessary part of ecological restoration (Bernes et al., 2015; Prichard et al., 2021).

Examples include the control of exotic animals and invasive plants that have colonised natural forests, including the vectors of their spread (e.g., livestock, roads); reduction of native herbivores that have become overabundant (e.g. see Jandl et al., 2024); restoring structure to replanted forests; removing roads to increase connectivity and hydrological functions; and returning extirpated species. Where there is a deficit of wildfire as part of natural fire regimes (Prichard et al., 2021), this could also include working with wildfire for ecosystem benefits (DellaSala et al., 2022). However, we recognize that replacing high-intensity wildfire (which is part of the natural fire regime for some ecosystems) with frequent low intensity planned fire could have significant negative impacts on a wide range of key elements of biodiversity, including those associated with complex early seral forests (Swanson et al., 2011; Lindenmayer et al., 2019).

7. Conclusions

Forests are receiving increased attention from AM proponents (Hessburg et al., 2021; Popkin, 2021; Prichard et al., 2021; Keenan, 2024), in part, as a response to climate change, but also due to altered fire regimes and the increasing prevalence of bark beetle outbreaks. However, AM has can degrade the ecological integrity of HCVF and could convert HCVF to permanently altered states via the cumulative disturbance burden. This is despite commitments on the part of many governments to end forest degradation globally by 2030 (see European Commission, 2022; DellaSala et al., 2025)

There is an inherent need for forest managers to grapple with the underlying drivers of altered fire regimes, patterns of flammability, and forest susceptibilities to insect outbreaks in HCVF rather than just the effects of these changes. For example, climate change is an obvious factor driving extreme fire weather and bark beetle outbreaks, yet thinning and post-disturbance logging treats the effects. Therefore, efforts to reduce GHG emissions along with protecting natural climate solutions like HCVF are fundamental to climate mitigation and adaptation. Similarly, the past history of logging has contributed to the increased flammability of some HCVF following their regeneration, both in south-eastern Australia (Taylor et al., 2014; Wilson et al., 2022) and parts of North America (Thompson et al., 2007; Bradley et al., 2016; Levine et al., 2022; Mackey et al., 2023). This suggests that some of the ecological conditions in forests that have been created by industrial forestry practices are not solvable by more of the same practices that encompass intensive AM.

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CRediT authorship contribution statement

David Lindenmayer: Writing – review & editing, Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Conceptualization. **Philip Zylstra:** Writing – review & editing, Methodology, Investigation, Formal analysis, Conceptualization. **Chad T. Hanson:** Writing – review & editing, Methodology, Investigation. **Diana Six:** Writing – review & editing, Methodology, Investigation. **Dominick A. DellaSala:** Writing – review & editing, Validation, Supervision, Methodology, Investigation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

No data was used for the research described in the article.

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