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Large Trees: *Oregon's Bio-Cultural Legacy Essential to Wildlife, Clean Water, and Carbon Storage*

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Executive Summary

Large trees represent the bio-cultural heritage of dry forests of Eastern Oregon and Washington. Historically, they were the dominant structural features in both open and closed canopy forests with up to three-quarters of the region's surveyed forest types consisting of large trees and older forests. Circa 1920s logging wiped out most of the largest trees through clear-felling and high-grade logging practices, necessitating the need for "Eastside Screens" that have protected large trees on Forest Service lands for over two decades. The recovery of these trees is far from complete, yet the Forest Service proposes controversial measures that are not scientifically founded. The agency omits the vast majority of the scientific literature that supports large-tree protections in regions where large tree populations remain at greatly reduced numbers such as the Eastside forests.

This report is written in response to the Forest Services proposed amendment to large tree protections in Eastern Oregon and Washington. The key findings of this report support ongoing protections (no change in the standard) due to the relative importance and rare status of large trees (young and old) as uniquely providing ecosystem services, wildlife habitat, and biological legacies, summarized as follows:

Large Tree Values (all tree species are vital)

1. Forest raptors, woodpeckers, songbirds, bats, and other small mammals depend on large trees to nest, forage, overwinter, roost, and den.
2. Large trees provide shelter and microclimates for countless invertebrates, epiphytes, herpetofauna, and rare plants.
3. Large trees in riparian areas provide stream-side shading and, when they fall into streams, hiding cover for aquatic species.
4. Large trees store the accumulation of decades to centuries of atmospheric carbon helping to reduce adverse consequences of global overheating.
5. Large trees are essential to nutrient cycling, soil stabilization, and below-ground processes that develop as they mature.
6. Large trees remain in short supply due to a legacy of logging.
7. When logged, large trees release most (up to two-thirds) of their stored carbon to the atmosphere (contributing to global overheating) and their emitted carbon takes decades to centuries to recover, if ever.

Introduction and Purpose and Intent of this Report

The USDA Forest Service is considering removing the interim "Eastside Screens" that have protected large trees and older (late-seral) forests for over two decades across >9 million acres in Eastern Oregon and Washington (Umatilla, Wallowa-Whitman, Malheur, Ochoco, Deschutes, and Fremont-Winema National Forests). The agency proposes a forest plan amendment for all six National Forests claiming it needs to take large trees for "restoration" and other purposes and that this action is supported by recently published science. We reviewed the Environmental Assessment (EA) and the General Technical Report PNW-GTR-990 "The 1994 Eastside Screens large-tree harvest limit: review of science relevant to forest planning 25 years later[2] (GTR) in our determination of best available science use. In our review, we document the importance of large trees, ≥21 inches in diameter at breast height (dbh) as irreplaceable elements of the region's bio-cultural heritage that are critical to proper ecosystem functions including sequestering and storing (long term) carbon wildlife habitat, and myriad ecosystem services (e.g., clean water, nutrient cycling, below-ground processes). Drawing from three-decades of expertise in global and regional forest ecosystems, wildfire ecology, forest carbon dynamics, wildlife management, landscape ecology and conservation biology, we argue for keeping large tree protections in place. Dr. DellaSala serves on the editorial boards of several leading scientific journals and is thoroughly familiar with the relevant literature and the application of best available science (e.g., evidence-based literature reviews). Our Curricula Vitae are attached as an Appendix to this report.

"In the dry forests of central, eastern and southwestern Oregon, we can reduce fuels by removing debris, branches, brush and small trees using machines and controlled burns. This lowers fire risk for up to 15 years."

- Tom Spies, Emeritus, USDA Forest Service Pacific Northwest Research Station [1]

In sum, the agency has not made a cogent scientific case for removing large trees. Instead, the proposed action is based on an incomplete presentation and disclosure of relevant scientific information that contradicts the assertions in the EA and fails to acknowledge the associated scientific uncertainty. The EA omits numerous studies that do not support the proposed action to remove large trees, as herein cited.

[1] *https://www.oregonlive.com/opinion/2020/09/opinion-look-to-wildfires-history-to-better-prepare-for-next-one.html*

[2] *https://www.fs.usda.gov/treesearch/pubs/60635*

Introduction and Purpose and Intent of this Report

This report presents overwhelming evidence for the conservation of large trees of all species and the forests within which they are found. It covers 9 major topics and deficiencies in the EA and GTR as follows:

1. **Large trees as irreplaceable biocultural legacies of the forest.**
2. **Assessment of the Forest Vegetation Simulation Model.**
3. **Role of wildfire and problematic assumptions made.**
4. **Large trees, old forests, and carbon.**
5. **Carbon stocks and flows.**
6. **Limitations of thinning in a changing climate.**
7. **Top threats on public lands especially livestock and climate change.**
8. **Importance of Complex Early Seral Forests (CESF).**
9. **Best Available Scientific Information (BASI) vs. Best Available Independent Science (BAIS).**

Large Trees are Irreplaceable Bio-Cultural Legacies

Historical Background (Henjum 1994) and Origin/Purpose of Eastside Screens

Interest in protecting large trees and the late-seral forests and watersheds that harbor them dates back to at least 1992 when a bipartisan group of members of Congress requested a panel of scientists, organized by scientific societies, assess the importance and status of late-seral forests and old trees (Henjum et al. 1994). The EA refers to this earlier review process as the “Everett Report,” a related but separate GTR publication. The main findings of the scientific societies (independent scientists) were covered in the Henjum et al. (1994) independent report and require more attention in the EA. For instance, despite Henjum’s report being a seminal document for the reasons the Eastside Screens were created, their report was cited only twice in the EA (in consecutive sentences of the same paragraph). This barely reflects the broad suite of large tree values presented in the 245-page report Henjum provided to Congress. Given the EA omits the Henjum report’s main findings (with no explanation as to why), we are including some of the relevant information from the Henjum report that should have been duly noted and factored into the historical accounting of large tree and late-seral deficits in the EA (the Forest Service has access to this document given it was cited in the EA):

- The geographical extent of old-growth forest ecosystems in eastside National Forests shrank dramatically during the twentieth century. Continued logging of old growth outside current reserves will jeopardize unknown numbers of native species. Forest harvest and other human actions have changed the character of many components of eastside landscapes, including rivers and their populations of resident and migratory salmonids. Many ecologists (as mentioned in Henjum) believe that the combined effects of logging old growth and fire prevention have significantly increased the vulnerability of eastside landscapes to catastrophic disturbances, further threatening what are already severely reduced and degraded habitats.

- The 1936 survey (Cowlin et al. 1942) found that old growth of all forest types made up 89% of sawlog-sized stands and 73% of all commercial forestlands in eastern Oregon and Washington.

- Originally, a virgin forest of this type extended the length of Oregon along the east slopes of the Cascade Range from within a few miles of the summit to the desert’s edge. From about 10 miles in width on the north, it ranged to nearly 100 miles on the

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Klamath Plateau in the south, interrupted only by comparatively small openings of non forested land. Extensive cutting from Bend south has broken it up with large areas of pine second growth (Cowlin et al. 1942, cited in Henjum et al. 1994).

- A 1936 inventory in the Blue Mountains (today's Malheur, Ochoco, Umatilla, and Wallowa-Whitman National Forests) found that forests containing a significant proportion of ponderosa pine occupied about 80% of commercial forestlands…, the great majority of pine forests consisted of old growth (86%). By the mid-1960s, however, the proportion of commercial forests…had dropped to 40% (Bolsinger and Gerger 1975 in Henjum et al. 1994), a loss of half over a 30-year period.

- In the mid-1950s the acreage in sawtimber trees $\geq$ 21 inches dbh and larger was 2.5 million acres [Blue Mountains], the present inventory shows 1.4 million acres – a reduction of 44%.

Based on historical documentation and photo interpretation at the time, Henjum et al. (1994) concluded that far more large ($\geq$ 21 inches dbh) trees were prevalent historically compared to contemporary forests, and logging in unprotected areas would have reduced late-seral forests to <10% of the total forest area (not unlike the loss of old forests throughout the Pacific Northwest, see Strittholt et al. 2006). The scientists raised serious concerns about risks to associated wildlife species and ecological processes from ongoing logging of large trees. The Eastside Screens were created to allow the large ($\geq$ 21 inches dbh) tree component time to recover. Henjum et al. (1994) made no reference to preferential logging of any particular species. In fact, they specifically mentioned the same high-grading problems existed for large Douglas-fir as pine. In sum, in the wildlife literature cited in Henjum et al., size and structure matter most, composition less so.

In contrast, the GTR notes the importance of large trees of all species, but then proposes removal of mainly large firs, a significant inconsistency between the science and the management. Likewise, the EA notes the importance of large trees in stating, "maintaining or increasing the abundance of large trees, particularly where old trees may be lacking, can be an important element of providing ecological functions" (p. 20). Then it completely discounts that statement by saying: "while some trees may be both old and large, not all old trees are large, and not all large trees are old." Of course, this was known at the time of Henjum et al.; the 21 inch dbh standard was intended as a reasonably simple rule that would allow trees to get to

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older ages to make up the old-tree deficits documented. Lifting the standard now because there are exceptions to old but not large trees (and vice versa) is not a valid rationale.

The concern over protecting large trees is even more relevant today because of their global role in carbon cycles (Lindenmayer et al. 2012, 2013) and in lessening the catastrophic effects of climate change, as discussed below.

In reality, the cumulative effects of logging late-seral forests and large trees significantly compromised the resistance and resilience of Eastside ecosystems (see Henjum et al. 1994, Paine et al. 1999, Lindenmayer et al. 2011, 2016 for how logging affects ecosystem integrity), and that historical resistance and resilience is not yet fully recovered or restored. Large old ponderosa pines were and still are the primary source of Eastside forest resistance to intense fires, since they have especially thick bark, high crown base height, and the ability to recover from high levels of crown scorch. Large trees that better survive fires are also key sources of abundant post-fire seed ("seed rain") essential for forest resilience through tree regeneration after fires (Baker 2009). Large Douglas-firs are also highly fire resistant and fire resilient (Baker 2009).

Notably, large trees and older forests, globally and regionally, remain at substantial deficits (Lindenmayer et al. 2012, 2013), and in places where they are recovering, this is cause for celebration. In Eastern Oregon, large trees and old forests are still recovering from a century of high-grade logging that followed the westward railroad expansion and rapid ramp-up of the logging industry. Prior to European colonization, Eastside forests were dominated by large trees with 76-78% of forests in late-seral condition (dry, moist, and mixed forest types) maintained against the backdrop of mixed-severity wildfires and insect outbreaks (Cowlin et al. 1942, Baker 2015a). Those large trees remain today at a major deficit due to past and ongoing logging (mostly nonfederal).

The Historical Baseline of Large Trees, Natural Disturbances, and Tree Densities are Misrepresented in the EA and GTR

Eastside forest natural processes resulted in large and small patches of severely disturbed areas in forests that were not maintained by just low-severity fire as often claimed (e.g., see Hessburg et. al 2007, Perry et al. 2011, Baker 2012, Williams and Baker 2012, Odion et al. 2014).

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The historical occurrence of substantial numbers of large trees and extensive older forests (Cowlin et al. 1942, Henjum et al. 1994, Baker 2015a) in Eastside areas was certainly possible because fire effects operated at landscape scales on long fire rotations of many decades-centuries (Baker 2012, 2017, Williams and Baker 2012) that allowed trees to commonly mature and reach statuesque sizes (Cowlin et al. 1942, Henjum et al. 1994). Limited fire-scar sampling (fire return intervals extrapolated from small unrepresentative stands to landscapes often without reconstructing fire severity) result in fire return intervals that are too short, and misrepresent the occurrence and ecological importance of mixed- and high-severity fires (Hessburg et al. 2007, Baker 2017). This has led some researchers and the agency to falsely conclude that Eastside forests were predominately open park-like pine forests, when, in fact, fire regimes and forest structure and composition were much more complex. This complexity is most evident when assessed from landscape-scale probabilistic evidence that found fires were highly varied in frequency and severity and quite complex (Hessburg et al. 2007, Perry et al. 2011, Baker 2012, Williams and Baker 2012, Baker and Williams 2018). Denser, closed canopy forests were more prevalent, and shade-tolerant trees were more common (Hessburg et al. 2007, Baker 2012, Williams and Baker 2012) than acknowledged in the EA or GTR, which instead reflects omission of this evidence and an agency confirmation bias.

The EA and GTR omitted all of this published, peer-reviewed evidence about historical forest structure and fire, so that a significantly incomplete historical baseline is presented. This incorrect understanding of the role of historical fire led to fire hyperbola ("megafires") that contemporary fires are too large or too severe, even though many regional publications show no statistical increases in high-severity fires (Odion et al. 2014, Parks et al. 2015, Baker 2015b, Law and Waring 2015) or problems with post-fire conifer establishment even in the largest high severity patches (DellaSala and Hanson 2019). Fear of megafires is driving forest management policies throughout the region that call for reducing tree density and removing large shade-tolerant firs thought to increase fire hazard, even though firs were historically common in Eastside mixed-conifer forests (Hessburg et al. 2007, Baker 2012, 2015a, Williams and Baker 2012) and the efficacy of thinning to reduce fire intensity is questionable in a changing climate (Moritz et al. 2014, Schoennagel et al. 2017, see below). These large sources of scientific evidence and their significant implications were omitted in the EA and GTR, as were calls from these and other scientists to place more emphasis on coexisting with wildfires by strategically focusing on home protection and community adaptation, instead of back-country logging (Moritz et al. 2014, Schoennagel et al. 2017).

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Further, in many places, large firs (in dry and moist forests) are the only complex structures remaining or recovering after past high-grade logging, and these large firs now represent the accumulation of carbon stored in trees and soils for many decades. Regardless of species composition, large firs are now invaluable biological legacies (DellaSala 2019) and carbon sinks (see discussion of carbon in large trees below). Large dead trees (snags, logs) of all species are even more valuable as wildlife habitat (e.g., the GTR references large hollow trees and logs 20-80 in dbh). They also contain substantial amounts of above ground carbon, as most of the carbon when a large tree, tree cohort, or forest stand dies is transferred from live to dead pools where it slowly (decades) decays while cycling nutrients, retaining moist microclimates, and providing wildlife habitat (see below). Contrary to common beliefs, these dead trees are not a significant source of emissions (Harris et al. 2016; especially compared to logging), nor are they associated with increases in fire intensity (see Bond et al. 2009, Black et al. 2013, Donato et al. 2013, Six et al. 2014, Hart et al. 2015, Meigs et al 2016).

Notably, large shade-tolerant trees provide significant natural bet-hedging against potentially unexpected future disturbances that could easily remove or significantly reduce pines over large areas. Large beetle outbreaks that attack ponderosa pines are increasing in the West with rising temperatures (e.g., Graham et al. 2016 describe a 388,000-acre 2004-2014 mountain pine beetle outbreak that killed a large percentage of ponderosa pine). It is clear that post-beetle pine forests are typically dominated by surviving smaller pines and non-target trees, including shade-tolerant firs and other species (Baker 2018). Historical evidence that Eastside mixed-conifer forests typically contained abundant smaller trees as well as diverse tree species, including shade-tolerant trees, suggests that historical disturbances and climatic fluctuations provided this natural bet-hedging (Baker and Williams 2015), that is increasingly important as temperatures rise, and all kinds of disturbances appear to be increasing (Baker 2018). The EA (section 3.1.4.2) mentions increasing insects and disease and explains (section 3.1.5.4) that 63% of large tree mortality was caused by insects and disease, only 24% by fire, and 13% by other disturbances. Yet, much of the EA focuses on this lesser fire risk, not recognizing the important bet hedging provided by a wide diversity of tree species and tree sizes in the context of insect and disease. Of course, it is sensible to also pay attention to fire risk and other risks too, but a wide diversity of forest structures and composition is needed. Both small and large shade-tolerant trees provide key natural forest resistance and resilience to future pine-beetle outbreaks, a very real threat to Eastside forests especially in relation to climate change.

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Historical accounts of Eastside forests show that dry and moist forest types were dominated by large, old trees typically >60 inches dbh with most of the stand volume in the 20 to 44 in dbh class (Henjum et al. 1994). Before the ramp-up of commercial logging, old-growth forests were inventoried to have covered 78% of Eastern Oregon's ponderosa pine commercial forest area (Cowlin et al. 1942). This inventory was validated as an estimate of historical old growth by land-survey reconstructions, which found that 76% of ~691,600 acres of dry forests along the eastern slope of the Oregon Cascades would have qualified as old growth in the late-1800s (Baker 2015a). Large trees had been extensively logged by the 1980s, but due to interim protections of the Eastside screens and other factors (e.g., fire suppression) some of the volume is now filling back in, mainly midsize shade-tolerant trees in the 20+ in dbh range. Rather than acknowledge the recovery success of a part of the abundant large trees that once characterized these forests before they were unsustainably logged, the Forest Service now proposes in the EA to remove mainly remnant and recovering shade-tolerant firs up to 30 inches dbh.

This sudden rule change effectively sets aside part of the old-tree restoration goals and ignores public support for restoring historical old trees and old-forest abundance, regardless of species composition. The plan amendment is demonstrated here to be based on incorrect scientific understanding of the historical abundance and current ecological importance of large trees, including large firs. **As a consequence, the EA does not have the support of the scientific community, and may even lead to social conflict.**

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Many scientists have been calling instead for protection of all remaining now rare large trees and generally intact forests, including remnant and recovering old forests (Lindenmayer et al. 2011, 2013, Moomaw et al. 2019, scientist letter attached) to help remedy the global biodiversity (IPBES 2019) and climate crises (Ripple et al. 2017, 2020). As mentioned throughout our review, when it comes to large trees, the EA and GTR are incorrect about historical species composition and tree densities, but species composition is also of secondary importance, particularly since the EA reports that old trees in the project area have already declined by 8% over the last 16 years (Section 3.1.5.3). The value of now rare and reportedly declining old trees of any species for habitat and as carbon stores needs to be thoroughly emphasized in management proposals and decisions, as explained in detail in this report.

Large Trees and the Shifting Baseline Used in the EA

Removing large trees as proposed by the EA and its narrowly scoped science support (self-citations, omissions of evidence, GTR) is a complete denial of extensive ecological literature, niche theory, and how wildlife focus on structure more than composition. In doing so, the EA and GTR view the forest just for the fuels (confirmation bias) rather than as a living interconnected ecosystem recovering from extensive past high-grade logging damage. This narrow view reflects a shifting baseline perspective where the baseline is advanced to meet a particular objective – in this case – to show a more recently perceived large tree surplus even though the pre-logging time period is the necessary ecologically relevant timeline that shows a substantial deficit in large trees remaining for all species (Box 1).

Box 1. The Shifting Baseline Syndrome (SBS) involves moving the baseline to a more contemporary period to fit a desired outcome or perspective (Soga and Gaston 2018). It is inherent in decisions involving confirmation bias and self-reinforcing feedback systems – in this case the Forest Service just cannot accept a no-action alternative because it is built into the agency culture that they must always find a way to log trees and that, in turn, feeds back on the science that is cited or requested by the agency. There is a complete lack of objectivity and independence.

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Abstract (from Soga and Gaston 2018): With ongoing environmental degradation at local, regional, and global scales, people's accepted thresholds for environmental conditions are continually being lowered. In the absence of past information or experience with historical conditions, members of each new generation accept the situation in which they were raised as being normal. This psychological and sociological phenomenon is termed shifting baseline syndrome (SBS), which is increasingly recognized as one of the fundamental obstacles to addressing a wide range of today's global environmental issues. Yet our understanding of this phenomenon remains incomplete. We provide an overview of the nature and extent of SBS and propose a conceptual framework for understanding its causes, consequences, and implications. We suggest that there are several self-reinforcing feedback loops that allow the consequences of SBS to further accelerate SBS through progressive environmental degradation. Such negative implications highlight the urgent need to dedicate considerable effort to preventing and ultimately reversing SBS.

Notably, the SBS is rampant throughout the EA, particularly in the use of large tree numbers (Section 3.1.5.4). The FIA data show recent increases, but this says nothing about the remaining large overall deficit compared to the period prior to Euro-colonization and the expansion of industrial logging (Cowlin et al. 1942, Baker 2015a). The EA claims that the change in large tree numbers is different “than in forests that only experienced natural disturbances,” and while that might be true in places, the agencies’ adherence to a more contemporary baseline where large trees are recently increasing – rather than the documented historical deficit – inappropriately justifies logging large trees.

Instead of allowing large trees of all species to recover, the agency now wants to log them during their recovery phase. The large trees now present are certainly an improvement since the time of the Henjum et al. report, but large trees are still declining in the project area as mentioned in the EA, and current forests still lack the substantial structure and carbon stores once present in the original forest regardless of whether the large trees are pines or firs (e.g., Figure 1a,b). Just because large trees are coming back, does not mean it is all a wash (zero sum) to remove many of them now, since the evidence is clear that these forests have certainly not yet fully recovered. The EA itself shows them to be still declining.

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Consider these two historical photos vs. current conditions.

(a)

(b)

Figure 1. (a) Nearly all large trees like this one (which may have been a single tree cut to fit on the logging platform) were taken by loggers circa 1920s and have yet to return in abundance. (b) Large trees like this were common yet are almost all gone from the forests of the Pacific Northwest. The Forest Service's proposal to log trees $\geq$ 21 inches dbh due to a perceived "surplus" of large trees represents a shifting baseline perspective (as in Box 1). That is, the baseline is shifted to a more contemporary time period rather than acknowledging the large tree deficit in this case remains substantial from the period prior to the onset of logging. Photos: (a) Logger in Klamath County, ca. 1920, pinterest.com; and (b) Logging in the Pacific Northwest; pinterest.com.

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False Claims Regarding Large Trees in the EA and GTR

The agency also implies that in forests with no mechanical treatment and natural disturbances more large trees were lost (p. 33), but this is not the correct conclusion from the data. First, “loss” is impossible in this case – large trees when killed naturally do not simply vanish from the landscape – they become large snags and large logs that persist for decades, allowing for natural processes in disturbance-adapted forests. The EA reflects a narrow perspective that dismisses the importance of severe natural disturbances in creating snags and complex early seral forests (see below). The dead trees’ legacies are even more valuable to wildlife, as snags and logs, than when they were living (DellaSala 2019).

Second, much of the data presented in this paragraph (p. 33) do not demonstrate that mechanically treated forests or undisturbed forests will have more surviving large trees, than do burned forests, when their resistance and resilience are actually tested, which will not be known until they are also burned. The only relevant data about resistance and resilience in this paragraph are those comparing forests that actually burned, which show that only a small (17% vs. 20%) difference in loss of large trees occurred if the stand was mechanically treated prior to fire. However, this difference is trivial in ecosystems that have mixed-severity fire regimes, because large trees of course will be reduced by these fires. Either a 17% or a 20% loss of large live trees is very small and certainly not unnatural even relative to infrequent historical fires that were mainly moderate in severity (Baker 2012, Williams and Baker 2012), which by definition killed many more trees, between 20-70% of the basal area. Similarly, the paragraph on p. 34 shows only that managed forests have more increase in large trees at this point, but is this also because the managed forests just have not yet experienced fires, whereas unmanaged forests have? There is nothing scientifically sound in either paragraph that shows managed forests will have an ecologically significant difference in the mortality of large trees when they do experience fires, relative to expected mortality from fires like historical fires in Eastside dry forests, which included extensive moderate- and high-severity fire as repeatedly noted herein but not covered in the EA (Hessburg et al. 2007, Baker 2012, Williams and Baker 2012).

Again, instead of celebrating the return of some of the large trees from historic deficits, the EA argues that there are too many in some ecosystem types precisely because it has shifted the baseline to a more recent contemporary period as noted above. And, it also shifts the baseline by assuming that there should be almost no mortality from fires, so that a reduction from 20%

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to 17% from logging is important and significant, when it is trivial and ecologically insignificant relative to historical moderate- and high-severity fires.

In sum, the projections of large trees going forward are fundamentally flawed because they assume thinning will increase large trees relative to natural areas and will lower mortality from insects and wildfires (see the section on thinning limitations below).

Large Trees and Wildlife Use

The large tree value for wildlife cannot be overstated. Unfortunately, wildlife values were not prioritized in the EA. For instance, the EA failed to note the following regarding the significance of large trees for wildlife. (These are all quotes from General Technical Report PNW-GTR-391 “Trees and logs important to wildlife in the interior Columbia River basin” (Bull et al. 1997)).

"A large-diameter tree is vital because the chamber typically consists only of the former heartwood. To be most useful, the chamber must be large enough for a swift to fly up and down, for a pileated woodpecker to enter; or for a bear to occupy. All of these wildlife species typically use entrances from 30 to 80 feet off the ground, where the heartwood cylinder needs to be large enough to provide a chamber of suitable size. The average size of 21 trees used by swifts for nesting was 27 inches d.b.h. and 85 feet in height in northeastern Oregon (Bull and Collins 1993). Bear dens in hollow trees with top entries averaged over 43 inches d.b.h. and 57 feet tall in northeastern Oregon (Akenson and Henjum 1994). Pileated woodpeckers roosted in hollow trees that averaged 28 inches d.b.h. and 74 feet tall in northeastern Oregon (Bull and others 1992)."

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"Large, hollow trees are uncommon in managed landscapes and typically are found only in late- and old-seral stands of grand fir and western redcedar. Although isolated hollow trees in young stands have significant value to wildlife, these young stands cannot reproduce this type of structure for at least 150 to 200 years. The late-seral, multilayer stands that produce hollow trees comprise less than 3 percent of the forested landscape in the interior Columbia River basin (Hann and others, in press). Retaining all hollow trees in managed landscapes can be justified in most areas because so little of the landscape has them, they have little commercial value, and they are of great value to wildlife. Rarely, where large blocks (in excess of 600 acres) of this habitat have many hollow trees (in excess of 10 per acre), does the number of hollow trees probably exceed the wildlife need for habitat."

"... Large-diameter trees that show some evidence of being hollow are of the greatest value. In the absence of apparently hollow trees, large trees of the species most commonly producing this structural feature, such as grand fir and western larch, should be conserved."

"In northeastern Oregon, grand fir and western larch make up most of the hollow trees used by wildlife. All 56 nest and roost trees used by swifts were grand fir (Bull 1996a). Sixty- one percent of 123 pileated woodpecker roost trees (Bull and others 1992), and 8 of 10 arboreal bear dens were in grand fir as well (Akenson and Henjum 1994). The remainder of hollow trees used by pileated woodpeckers or black bears were primarily western larch and a few ponderosa pine."

An Assessment of the Forest Vegetation Simulator (FVS) Analysis Used in the EA

The essence of the problem in the use of FVS in the EA is that this model has not been shown to be validated for this kind of application, and generally failed in its one accuracy validation in the region. Proposed treatment alternatives to current management have no sound scientific basis in the FVS modeling, as their effects cannot be shown to be accurately predicted.

Little to No Validation of FVS in the Region; Where Its Accuracy was Validated, the Model Failed.

Computer models can be used to understand ecological interactions, which might require less accuracy, but when models are used to predict the future, as in this application in the EA, it is essential that the prediction accuracy of the model is known and well documented. FVS is a complex computer model with many interacting parts. When used for prediction, it is essential to validate all the parts individually and also to validate that when the parts are combined, the model outputs have known and reported prediction accuracy (Rykiel 1996, Mladenoff and Baker 1999). Validating predictions typically means running the model and comparing the results to actual values in plots in forests to show that the predicted value closely matches the plot value.

We did not conduct an exhaustive review of the scientific literature on the validation of the FVS model for prediction across large landscapes, which is the application in the EA, but what we found strongly suggests that the model has not been validated for the kind of application in the EA. The one validation study we found suggests the model is very inaccurate at prediction, not even predicting within ±25% of the actual value found in plots where the model was used. Now, of course, there may be model validations that have been done that are not published, but it is clear that the EA does not present the public with sound scientific evidence that FVS can predict accurately enough for its application in the EA.

First, the FVS manual (Dixon 2020) does not contain the word “validation” in the document. This is not necessarily fatal if the model is used simply to gain understanding of interacting processes, but it is potentially a fatal flaw if the model is used for prediction, as in the EA.

An Assessment of the Forest Vegetation Simulator (FVS) Analysis Used in the EA

Second, a 2013 validation of the ability of the Eastside Variant of FVS to actually predict is very relevant to the EA's application of FVS, but this study, that includes two US Forest Service scientists (Hummel et al. 2013), is not cited or discussed in the EA. Here are some quotes from this key peer-reviewed paper that are germane:

"There are protocols for evaluating FVS that recommend sensitivity analysis prior to validating model performance (Casrse et al. 2010). Evidence is scarce, however, that these protocols are routinely followed to evaluate the full scope of the model" (p. 41)

After further discussing FVS, Hummel et al. say*: "Despite its widespread use, model performance is not well understood and is infrequently documented" (p. 41).* Indeed, nowhere in this paper is a published evaluation of model performance or a model verification or a model validation for prediction cited, suggesting there likely may have been none in the Pacific Northwest. Hummel et al. may be the only validation in the region of the EA. If it is not, the EA failed to reveal this evidence to the public.

Hummel et al. validated the accuracy of model predictions of postfire conditions measured by tree mortality and fuel loads versus measurements of these same variables in six plots. As a standard of accuracy, they evaluated whether the model output fell within the interquartile range, which is between the 1st quartile and 3rd quartile, of the distribution of plot values. That is a very large range, representing the center 50% of the distribution of plot values, 25% on each side of the median value in the plots.

It is noteworthy that for tree mortality, the summary was:

"Indeed, for most of the stands the median simulated value for mortality fell outside of the interquartile range of plot data" (p. 50).

This study shows that FVS could not predict median tree mortality within even ±25% of the actual median value. Tree mortality is the parameter that is most close to the usage of FVS in the EA, this is a pretty lax accuracy standard, but the model failed even this test.

An Assessment of the Forest Vegetation Simulator (FVS) Analysis Used in the EA

We conclude that the FVS model has not been shown by validation to produce accurate predictions, and therefore none of the FVS analysis and findings presented as part of the "Environmental Effects" on p. 35-41 of the EA were shown to be valid in the EA. The only published validation of the Eastern Cascades variant of the FVS model that we could find suggests FVS mostly fails to meet even weak accuracy standards of predicting within ± 25% of the actual value. But, there is no validation that we could find at all for the variables predicted in the EA. The predictions likely are not accurate.

Modeled Effect Sizes are Far Too Small to be Predicted Accurately with FVS

Given the FVS failure to come within 25% of the plot median in the one accuracy validation in the region, as reviewed above, FVS cannot be expected to accurately predict where effect sizes are small. Here we show that effect sizes are quite small across the alternatives and attributes:

The summaries of modeled effect sizes are quoted here for all the model results, and the set of effect sizes then is compiled and analyzed:

3.1.6.1.1 p 35 on **Species Composition**: "FVS indicates an overall slight increase from present conditions in the dominance of fire tolerant species like ponderosa pine, western larch, and Douglas-fir." The percentage changes among the alternatives that are reported are 1.5%, 1.5%, about 5%, and flexibility.

3.1.6.2.1.1 p. 36 on **Late and Old Structure Forest**: "Modeling indicates less than 2.8% difference between alternatives in the amount of open LOS forest that will be created over the analysis period."

3.1.6.2.1.2.1 p. 37 on **Open Conditions in Dry and Moist Forest Outside of LOS**: "All alternatives decrease canopy cover outside of LOS by about 15% with differences between alternatives within 0.2%."

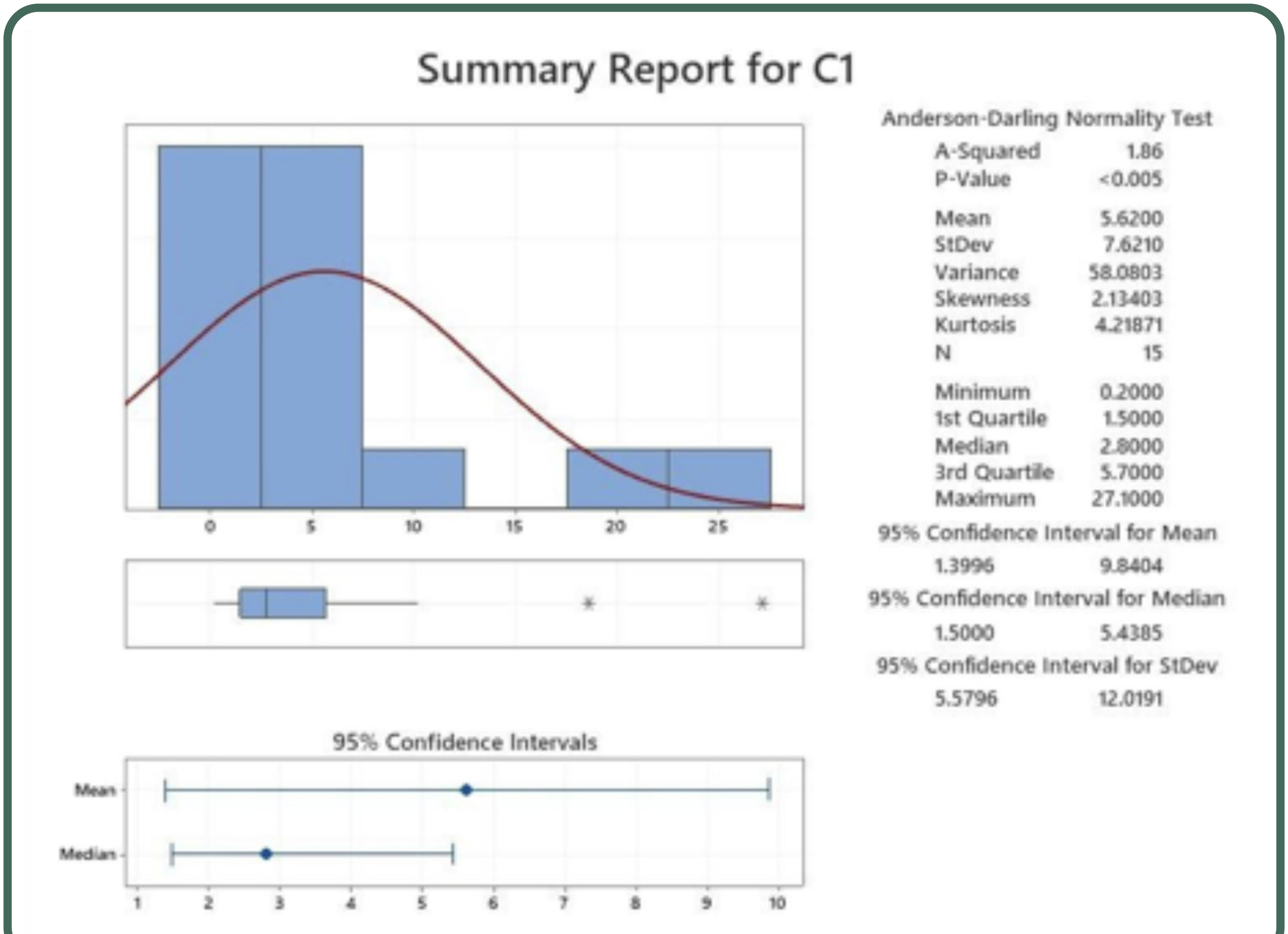
3.1.6.3.1.1 p. 39 on **Large and Old Trees:** "...the Current Management Alternative will result in an average of 8.8 large trees per acre remaining following thinning over the 25-year analysis period." The other alternatives are modeled to lead to decreases from this of 3.4%, 2.3%, and 10.2%.

An Assessment of the Forest Vegetation Simulator (FVS) Analysis Used in the EA

3.1.6.3.2.1 p. 40 on **Old Trees in Dry and Moist Forests**: "...the Current Management Alternative will result in an average of 7.0 old trees per acre following thinning." The other alternatives are modeled to lead to a 5.7% increase, a 27.1% increase, or an 18.6% decrease from the 7.0 old trees per acre under Current Management.

The upshot of all these modeling results are the following 15 effect sizes relative to Current Management, listed as absolute values of percentages in increasing order: 0.2, 0.2, 0.2, 1.5, 1.5, 2.3, 2.8, 2.8, 2.8, 3.4, 5.0, 5.7, 10.2, 18.6, 27.1.

Here is a Minitab graph and report of the distribution of these effect sizes:



An Assessment of the Forest Vegetation Simulator (FVS) Analysis Used in the EA

Of course, this assumes that this sample of effect sizes come from the same population, but there is similarity among the attributes that are modeled and among the management alternatives.

This analysis above shows that effect sizes have a median of only 2.8% and a mean of only 5.6%. And, 3/4 of effect sizes are less than 5.7%. Thus, these are mostly small effects from the alternatives, with only three exceptions (10.2%, 18.6%, and 27.1%).

To predict these small effect sizes accurately with FVS would have required a model with much higher accuracy than is shown by the one accuracy validation reviewed above, which could not come within 25% of the true median. Small effect sizes require a model with validated high predictive accuracy that can in most cases accurately predict a 2.8% to 5.6% difference in treatments. The evidence suggests FVS cannot come close to this level of predictive accuracy.

The failure of FVS in the one accuracy validation, combined with modeled small effect sizes that require highly accurate predictions, show that the FVS modeling part of the EA likely has no scientific validity.

The Role of Fire and Problematic Assumptions in the EA

This assumption in the EA is the key to understanding whether the EA is scientifically valid. Our review shows that the EA does not meet the Best Available Scientific Information (BASI) standard due to the proliferation of self-citations from scientists that only share the agency's limited perspectives. Evidence left out of the EA and GTR, and even evidence in studies cited to support the EA, show that large shade-tolerant trees have increased little, if at all. The EA's basic premise is thus deeply flawed.

Douglas-fir and other shade-tolerant trees (white fir/grand fir, incense cedar), may have increased since Euro-American colonization (e.g., Hagmann et al. 2014, Merschel et al. 2014), but *large* shade-tolerant trees and multilayered forests were clearly not minor components of historical Eastside forests, as falsely reported by Merschel et al. (2019) and cited in the EA and GTR. First, Merschel et al. (2014, 2019) was mostly based only on live extant trees—large stumps and down wood that could have been shade-tolerant trees, were not dated except at a single site (extremely limited site-specific example that cannot be extrapolated). Even if they had been dated, it is well known that firs decay more rapidly than pines (e.g., Scholl and Taylor 2010). Large firs extant in the middle-1800s could certainly have died and decayed or been burned in subsequent fires, a possible explanation for incongruity between the Merschel et al. age-structure and other direct records (see below) that do not rely on age-structures, but do show that firs were historically common.

These other sources, that show large firs were common, even in dry Eastside forests, were omitted in the EA and GTR, and in its primary sources (Johnston 2017, Johnston et al. 2018, Merschel et al. 2014, 2019). The original land surveys completed in the late-1800s, and early aerial photography both show that shade-tolerant trees were historically common. Over about 988,000 acres of dry forests on the eastern slope of the Cascades, shade-tolerant trees were 17% of trees and shade-tolerant trees dominated 12% of forest area, based on direct records by trained government surveyors in the late-1800s (Baker 2012). Independent evidence from early aerial photography confirms that mixed-conifer forests along the eastern slope of the Oregon Cascades had abundant area dominated by Douglas-fir and white fir/grand fir (Hessburg et al. 2007 Figure 4). Also, close to the Merschel et al. (2014) study area in the eastern Cascades, Douglas-fir and grand fir were >33% of large trees (≥21 inches dbh) in 1922-1925, before extensive high-grade logging (Hagmann et al. 2014). About 45% of firs were large (≥20 inches dbh) in the eastern Cascades, based on directly recorded historical diameters of bearing trees presented in Baker (2012 Fig. 3).

The Role of Fire and Problematic Assumptions in the EA

Thus, in total, large firs were likely historically dominant over other large trees in the 12% of area where firs dominated in general. And, outside this area of fir dominance, 17% of trees were shade-tolerant trees and about 45% of those were likely large trees. About 8% of trees, outside the area where shade-tolerant trees dominated, were also large firs or other large shade-tolerant trees. Thus, over the eastern Cascades as a whole, >10% of trees were likely historically large firs and other large shade-tolerant trees, and, in some large areas, firs were likely >33% of large trees.

Similarly, over about 753,000 acres of dry forests in Oregon's Blue Mountains, firs were common as 43% of forest area had >18% firs and 19% of forest area had >30% firs (Williams and Baker 2012). If a similar 45% of firs were also large here, then 43% of forest area had >8% large firs and 19% of forest area had >14% large firs, suggesting large firs and other shade-tolerant trees were also historically common in the Blue Mountains too. Forest understories also had small trees over 33% of Blue Mountains and 57% of Eastern Cascades dry-forest area, showing that historical forests had small shade-tolerant trees, which are thus not just an artifact of fire suppression. In the Eastern Cascades, concentrations of firs were widely distributed across elevation, slope, aspect, and topographic positions, but were somewhat favored on steeper slopes and at higher elevations (Baker 2012 Figure 2), thus firs were not restricted to just a few rare locations at all, but were widespread. Historical Eastside forests often also had a multi-layered structure, as reported in both Hessburg et al. (2007) and Baker (2015a), which Merschel et al. (2019) did not report when they said that multilayered forests were the result of fire suppression. None of this highly relevant evidence documenting that >10% of trees were likely large firs and other shade-tolerant trees in historical forests, based on evidence across >1.7 million acres of Eastside Oregon forests was cited or used in the EA, the GTR, or in Merschel et al. (2019). Again, this is indicative of confirmation bias and omission of relevant studies.

Most important, Merschel et al. (2014) did not find a substantial increase in large shade-tolerant trees over most of their study area, only in about 12.5% of their study area, which is likely not statistically different from the estimated >10% historical abundance documented above. The relative abundance of large shade-tolerant (Grand fir, Douglas-fir) and large shade-intolerant trees was no different between historical and current forests in most of the area (Merschel et al. 2014 Figure 2B), including "Persistent Shade Tolerant," "Persistent Ponderosa Pine," and "Recent Douglas-fir" in both the Eastern Cascades and the Ochoco

The Role of Fire and Problematic Assumptions in the EA

Mountains. Very little difference, just a slight increase in shade-tolerant trees, occurred in the last category, “Recent Grand Fir” in the Ochoco Mountains. The only combination of type and location that showed a significant relative increase in large shade-tolerant trees was in “Recent Grand Fir” in the Eastern Cascades, where Grand fir substantially increased. Thus, out of the 8 combinations of 4 types in 2 locations, only 1 (12.5%) showed a significant relative increase in large, shade-tolerant trees, likely no different from the historical estimate of >10% explained above. *Most importantly, this evidence does not support the need for the proposed EA action.*

Scientific evidence definitely does not support the premise underlying the EA that large shade-tolerant trees ascending into the canopy is unnatural and, therefore, these trees must be removed in “restoration.” Merschel et al. (2014), for example, did not conclude that the general increase in shade-tolerant trees they found, which was nearly all in just small trees (Merschel et al. 2014 Figure 2C), was entirely unnatural: “The wave of tree establishment that began in ~ 1900…was likely caused by a variety of factors…The relative importance of a favorable climate, grazing, and logging in determining contemporary age and stand structure inmixed-conifer forest is unknown” (p. 1684). However, other well-supported explanations were not considered by Merschel et al. (2014).

First, shade-tolerant white fir was shown to be able to ascend into the canopy in Eastside forests during a short 30-year fire-free interval (Agee 2003), showing that natural fluctuation in fire regimes is all that is needed for shade-tolerant trees to become large canopy trees. This is consistent with the estimate that >10% of historical trees were likely large firs, explained above. Second, there is abundant evidence from multiple sources (e.g., early land surveys, early aerial photographs, early scientific reports) that infrequent moderate- to high-severity fires strongly shaped Eastside ponderosa pine and mixed-conifer forests (Hessburg et al. 2007, Baker 2012, Williams and Baker 2012, Odion et al. 2014). It is ecologically within the natural range of variability for these fires to at times favor fire-resistant trees early in recovery after these fires and shade-tolerant trees later in recovery in mixed conifer forests across the West (Baker 2009), but shade-tolerant trees also can be favored early after severe fires simply because these trees can regenerate well in the shade of surviving pines (See Baker 2012 Quote 65 for a direct report of this by a government scientist in 1903 Eastside forests).

The Role of Fire and Problematic Assumptions in the EA

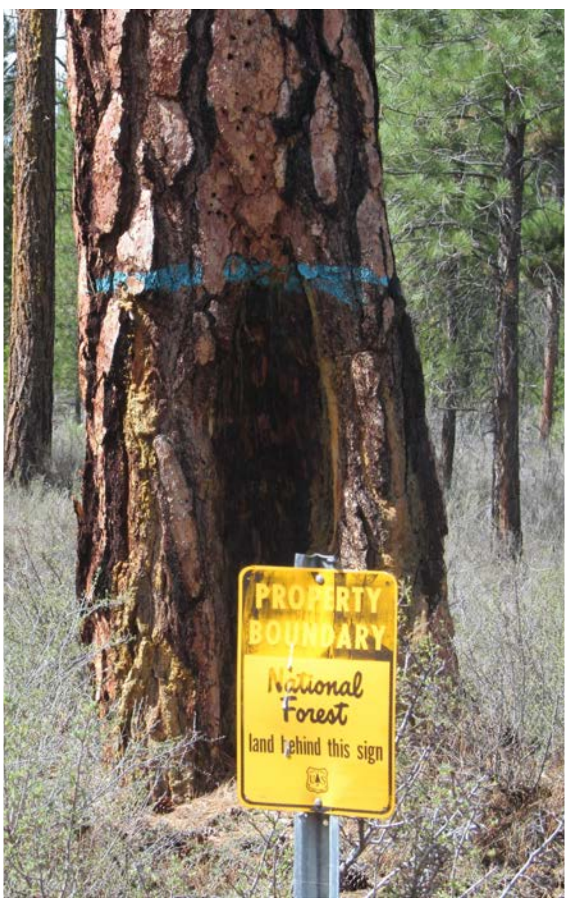
Also, there is abundant evidence that when beetle-outbreaks attack pines in mixed-conifer forests, then associated shade-tolerant trees naturally increase throughout the West (Baker and Williams 2015, Baker 2018). Evidence that large shade-tolerant trees have increased can be explained by multiple natural processes that have long historically shaped these forests, including: (1) climatic episodes that favor shade-tolerant trees; (2) fluctuations in fire-free intervals that enable shade-tolerant trees to grow into the canopy; (3) natural recovery later in succession after moderate- to high-severity fires, beetle outbreaks, droughts etc.; (4) increases favored after beetle outbreaks; and (5) favorable conditions for shade-tolerant trees to regenerate after moderate-severity fires in the shade of surviving pines. Of course, logging of ponderosa pines and fire suppression are widely known to contribute to increases in shade-tolerant trees as well. However, the EA, GTR, and none of the publications that are cited in support of the proposed action (Merschel et al. 2014, 2019, Johnston 2017, Johnston et al. 2018) systematically reviewed the evidence for these natural causes of increased shade-tolerant trees in Eastside forests. Merschel et al. (2014) mentions climate, but the others mention none, assuming that fire suppression was the cause. Thus, not only is there no valid evidence that large shade-tolerant trees have increased, as reviewed above, but even if they had, an increase can be fully explained by natural processes, so that logging is certainly unwarranted.

Similar to Merschel et al. (2014), evidence for little relative increase in large shade-tolerant trees was also found in a small area in the Blue Mountains. Johnston et al. (2018) reconstructed basal area in 5 stands covering just under 4 acres in a small part of the Blue Mountains. In stand 1, a ponderosa pine stand (Johnston et al. 2018 Figure 2), large trees (≥21 inches) were lacking in both 1880 and 2016, so this stand showed no change in composition of large trees. In stand 2, also a ponderosa stand, large trees were entirely ponderosa in 1880 and nearly all ponderosa in 2016, clearly indicating little to no change in composition of large trees. In stand 3, a grand fir site, large trees were nearly all ponderosa in 1880, but about 2/3 firs in 2016, which clearly fits the concept of increasing large shade-tolerant trees. In stand 4, also a grand fir site, large trees were entirely grand fir in 1880, but also included Douglas-fir and larch in 2016, thus clearly indicating no change in dominance of large trees by shade-tolerant species. In stand 5, also a grand fir site, large trees were entirely shade-tolerant western white pine, grand fir, and Douglas-fir in both 1880 and 2016, clearly indicating no change in dominance of large trees by shade-tolerant species. Overall, this study showed little to no change in four stands, and substantial compositional change from shade-intolerant to shade-tolerant large trees in only one stand (20%) in this small study area that cannot be extrapolated or inferred

The Role of Fire and Problematic Assumptions in the EA

outside the five areas of analysis, since they were not random samples. No natural explanations for these changes were considered.

A third study with *just 20 reconstruction plots* in the southern Blue Mountains (Johnston 2017) showed no increase in large (>25 inches dbh) shade-tolerant trees in 13 of 16 plots in ponderosa pine and dry mixed-conifer forests. This means that, again, the forests showing an increased percentage of large shade-tolerant trees is small, only about 19% in this case, similar to the last study. Also, it is plausible that in these other three plots in dry mixed conifer in 1860, which graphs show had relatively low basal areas, basal area had been reduced by preceding moderate- to high-severity disturbances (e.g., fire, drought, beetles). However, this and other natural explanations for changes were not considered either. In scientific studies, alternative explanations need to be at least discussed and therefore citing this study with limited sampling as evidence of an increase in large firs does not reflect the best available science as alternative explanations were not ruled out and uncertainties were not even included.



None of the three studies evaluated the possible natural-recovery explanation, even though it had been documented in previous studies of Eastside forests (e.g., Hessburg et al. 2007). Only Merschel et al. (2014) mentioned climatic fluctuations as a natural explanation of an increase in shade-tolerant trees. As a consequence, restoration proposals in these forests may assume it is sensible to remove shade-tolerant trees and to even restore the low basal area and more pine-dominated early-successional state of mixed conifer forests, when, in fact, they likely are now just recovering to their pre-disturbance old-growth state. These studies and the EA and GTR did not adequately evaluate natural causes of the small increases that were documented in large shade-tolerant trees. Also, the evaluation of the extent of change in large shade-tolerant trees in Eastside forests in the EA/GTR and the tree-ring studies revealed that they did not compare current

The Role of Fire and Problematic Assumptions in the EA

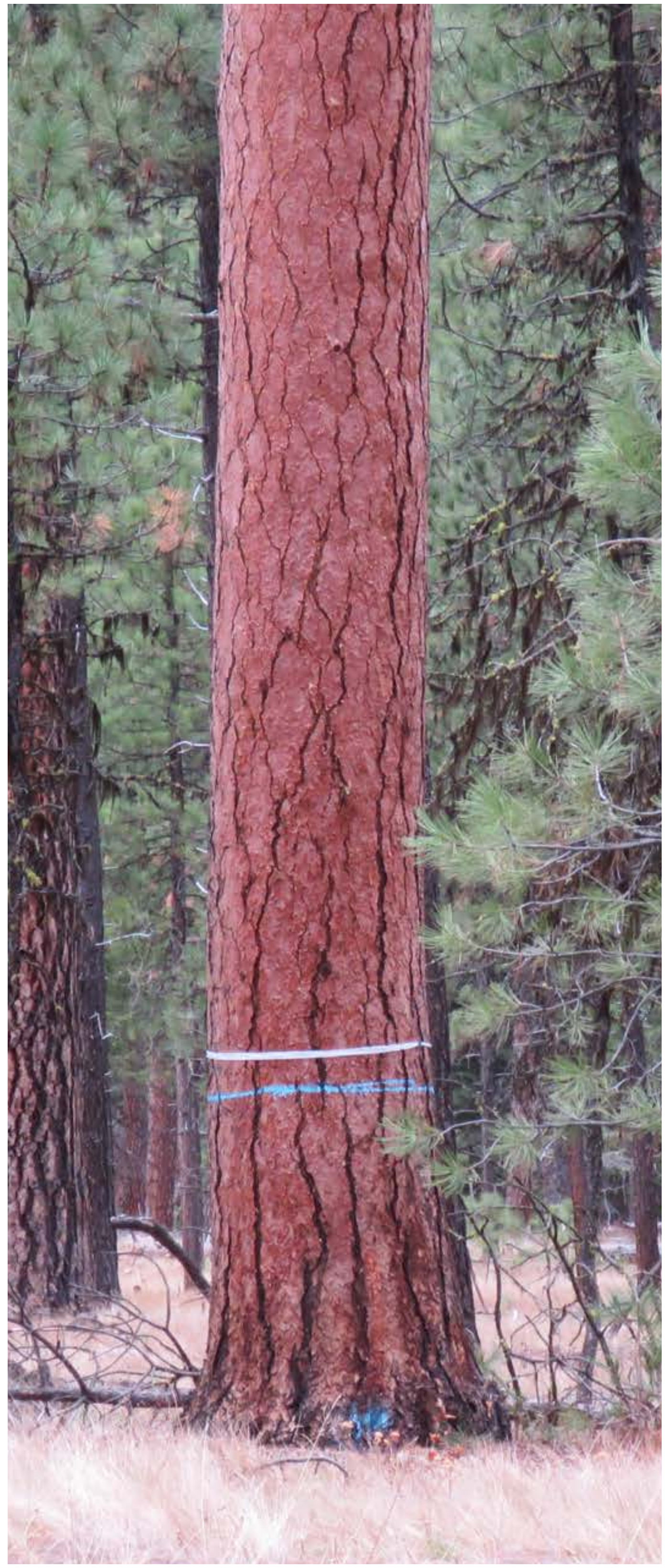
large trees to historical large trees using the much more extensive datasets from the land surveys and early aerial photography. As reviewed above, these datasets showed extensive historical shade-tolerant large trees, often >10% of all trees, much more prevalent than reported in the tree-ring reconstructions.

Together, these three studies (Merschel (2014); Johnston (2017); and Johnston et. al. (2018)), which are the primary scientific sources for the EA/GTR argument that large trees in Eastside forests are increasingly dominated by shade-tolerant trees, found a substantial increase in only roughly 12.5% of the Eastern Cascades (Merschel et al. 2014) and 19-20% of the Blue Mountains (Johnston 2017, Johnston et al. 2018). In the eastern Cascades, the 12.5% is likely not statistically different from the >10% historical large trees; in the Blue Mountains, the small sample sizes likely also would not show the 19-20% sample estimate to be statistically different, but in any case, the true difference is likely very small. Moreover, these small increases in the Blue Mountains could represent natural recovery after moderate-to high-severity disturbances in the 1800s, could also have been driven by wet periods in other cases, could represent periods without fire, and of course could represent the effects of fire suppression or logging. None of these studies even addressed these natural causes, much less excluded them as the actual cause of increased large shade-tolerant trees in the small percentage of stands where this change was found. This means it is not possible to be sure that removing large shade-tolerant trees would constitute restoration. It could instead be reversing an entirely natural change process, which also represents ecologically beneficial restoration and increased carbon storage. Tree-ring evidence does not support the proposed action; the effect is nonexistent or small and its causes are unknown and possibly natural. Finally, removing large shade-tolerant trees reduces resistance and resilience to pine-beetle outbreaks--a more significant risk to Eastside forests than the risk of fires.

The 21-inch rule does not impede restoration, as suggested in the EA/GTR and Merschel et al. (2019). It is clear that the rule does not prevent reduction in the small shade-tolerant trees that are the primary size class of shade-tolerants that have increased, as reviewed above. The evidence above shows there is no need to remove large shade-tolerant trees, particularly since, in many areas, large shade-tolerant trees were removed during earlier high-grade logging. Also, the EA itself indicates that a no-action alternative could lead to a reduction in

The Role of Fire and Problematic Assumptions in the EA

shade-tolerant trees, suggesting it is more sensible to keep the ones present now. Old trees are shown in the EA (section 3.1.5.3) to have declined overall by about 8% between 2001 and 2017. Indeed, old Douglas-fir declined by 14% over these 16 years, *while old grand/white fir have only increased by 5%*. That is a net decline in old shade-tolerant trees of 9% (14%-5%) in 16 years (2001-2017). Since the tree-ring evidence shows that none of the area in the Eastern Cascades and only up to about 10% of the area in the Blue Mountains has experienced an increase in large shade-tolerant trees, the evidence suggests that **a no-action alternative, where this current trend in loss of shade-tolerant trees continues as it has over the last 16 years, could completely reverse the documented increase within about 16 years at no cost**. Moreover, it is clear the EA is incorrect in forecasting further increases in shade-tolerant trees as the data clearly shows further loss is much more likely.



Large Trees, Older Forests, and Carbon

In general, large trees are big sticks of carbon, accumulated over decades and, in some cases, centuries. The forests that harbor them contain substantial amounts of above and below ground carbon in living and dead pools. When a forest dies from natural causes, most of its carbon transfers from live to dead pools. In contrast, logging overall decreases the forest carbon reservoir by up to two-thirds with 40-60% emitted quickly through rapid decomposition (Law et al. 2018, Hudiburg et al. 2019). Logging related emissions overall are 8 times that of natural disturbances nationally (Harris et al. 2016) and, in Oregon, emissions from logging are 10 times greater than that of wildfires (Law et al. 2018). In sum, in a rapidly changing climate it is of utmost importance to keep as much atmospheric carbon tied up in the forest, which also benefits biodiversity and water quality (Brandt et al. 2014). Oregon will not meet a net-carbon neutral goal without keeping emissions from logging and energy use out of the atmosphere – there is simply no time left to permit additional emissions from the logging of large trees.

The following reflects the disproportionate role large trees play in forest-climate feedbacks.

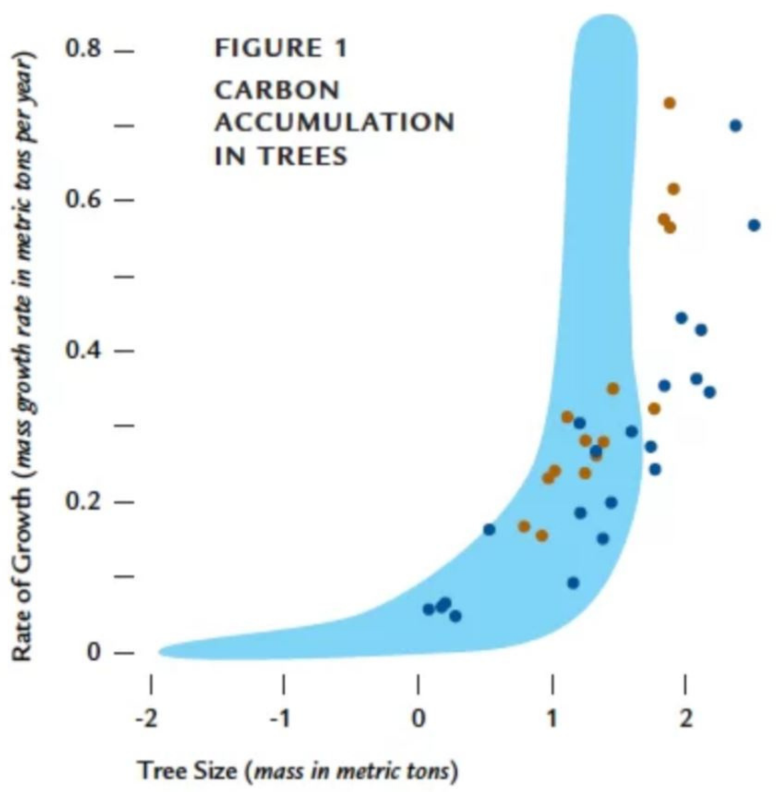
Trees accumulate carbon over their entire lifespan. While growth efficiency declines as trees mature, corresponding increases in a tree's total leaf area overcome this slow growth period as the whole-tree carbon accumulation rate increases with age and tree size. For instance, a study of 673,046 trees across six countries and 403 species found that a large old tree may sequester as much carbon in one year as growing an entire medium size tree (Stephenson et al. 2014). At one site, large trees comprised 6% of the trees but 33% of the annual forest growth. More recent studies show the largest 1% of trees in old-growth forests worldwide store ~50% of the total live-tree stand level carbon (Lutz et al. 2018). In sum, young trees grow fast, but old trees store a disproportionate amount of carbon over time given the larger leaf surface area for absorption and massive tree trunks and root wads that represent centuries of accumulated carbon.

Quoting directly from the abstract in Lutz et al. (2018):

"*Main conclusions: Because large-diameter trees constitute roughly half of the mature forest biomass worldwide, their dynamics and sensitivities to environmental change represent potentially large controls on global forest carbon cycling. We recommend managing forests for conservation of existing large-diameter trees or those that can soon reach large diameters as a simple way to conserve and potentially enhance ecosystem services.*"

Large Trees, Older Forests, and Carbon

Old forests accumulate carbon and contain vast quantities of it. Although individual trees experience an increasing rate of carbon sequestration, forest stands experience an "S-curve" of net sequestration rates (e.g. slow, rapid, slow) (Lutz et al. 2018). The expected decline in older stands is due to tree growth balanced by mortality and decomposition. For instance, an international team of scientists reviewed published forest carbon-flux estimates from stands 15 to 800 years old and found that, in fact, net carbon storage was positive for 75 percent of the stands over 180 years old and the chance of finding an old-growth forest that was carbon neutral was < 1 in 10 (Luyssaert et al. 2014). They concluded that older forests are substantial carbon sinks, steadily accumulating carbon over centuries and containing vast quantities of it in relatively stable form (also see DellaSala and Moomaw 2020).



Aboveground mass growth rates for 58 species (shaded area) juxtaposed with two of the most massive tree species on earth: Swamp Gum (*Eucalyptus regnans*—brown dots) and Coast Redwood (*Sequoia sempervirens*—blue dots). Mass growth rate equals the total mass accumulated each year after accounting for respiration. The mass of a tree is primarily carbon, so the figure shows that annual carbon accumulation increases with the size of the tree. (Adapted from Stephenson et al. 2014.)

Old forests accumulate carbon in soils. Soil organic carbon levels in old forests are generally thought to be in a steady state. However, protecting remaining unlogged forests and large trees provides for more stable microclimates (with less desiccation and lower temperatures) that will help mitigate local adverse climate effects. In fact, recent research shows that older forests may act as an important climate buffer (Frey et al. 2016). Thus, scientists have repeatedly acknowledged the superior climate benefits inherent to older forests that are irreplaceable in human lifetimes.

Large Trees, Older Forests, and Carbon

Old forests and trees share carbon among species. Trees compete for sunlight and soil resources, and competition for resources is commonly considered the predominant tree species interactions in forests. However, recent research on carbon isotope labeling has shown that trees interact in more complex ways, including substantial exchange and sharing of carbon below ground. Aided by mycorrhiza networks, interspecific transfer among trees accounts for 40% of the fine root carbon: totally ~280 kg ha-1 per year tree-to-tree transfer (Klein et al. 2016). Morrien et al. (2017) found that mycorrhiza soil networks become more connected and take up more carbon as forest succession progresses even without major changes in dominant species composition. Notably, older forests, compared to young growth, contain more complex below-ground processes that connect trees at the subsurface level (Morrien et al. 2017). Thus, the Forest Service needs to provide information on the impacts of logging on soil microbial and mycorrhizae carbon exchange. This may be especially important given many Eastside trees grow together in cluster cohorts and, thus, removing some of the cohort can impact this below-ground carbon and chemical/nutrient exchange network at the detriment of the remaining trees within the cohort (see Beck et al. 2020).

Conservation of forest carbon stocks can help slow climate change. Griscom et al. (2017) systematically evaluated 20 conservation, restoration, and improved land management actions that increase carbon storage and avoid greenhouse gas emissions. They found that the maximum potential of natural climate solutions was ~2.4 Petagrams (billion tons) of carbon per-year while safeguarding food security and biodiversity (Griscom et al. 2017). Thus, when carbon is properly maintained, the entire national forest benefits through maintenance of linked ecosystem services and biodiversity (i.e., multifunctionality of forests maintained via carbon management; see Brandt et al. 2014). In addition to carbon, older forests and large trees accrue soil depth, cycle nutrients, mitigate pollution, purify water, release oxygen, and provide habitat for wildlife at levels far superior than logged forests (Mackey et al. 2013, Mackey 2014, Mackey et al. 2014; Brandt et al. 2014).

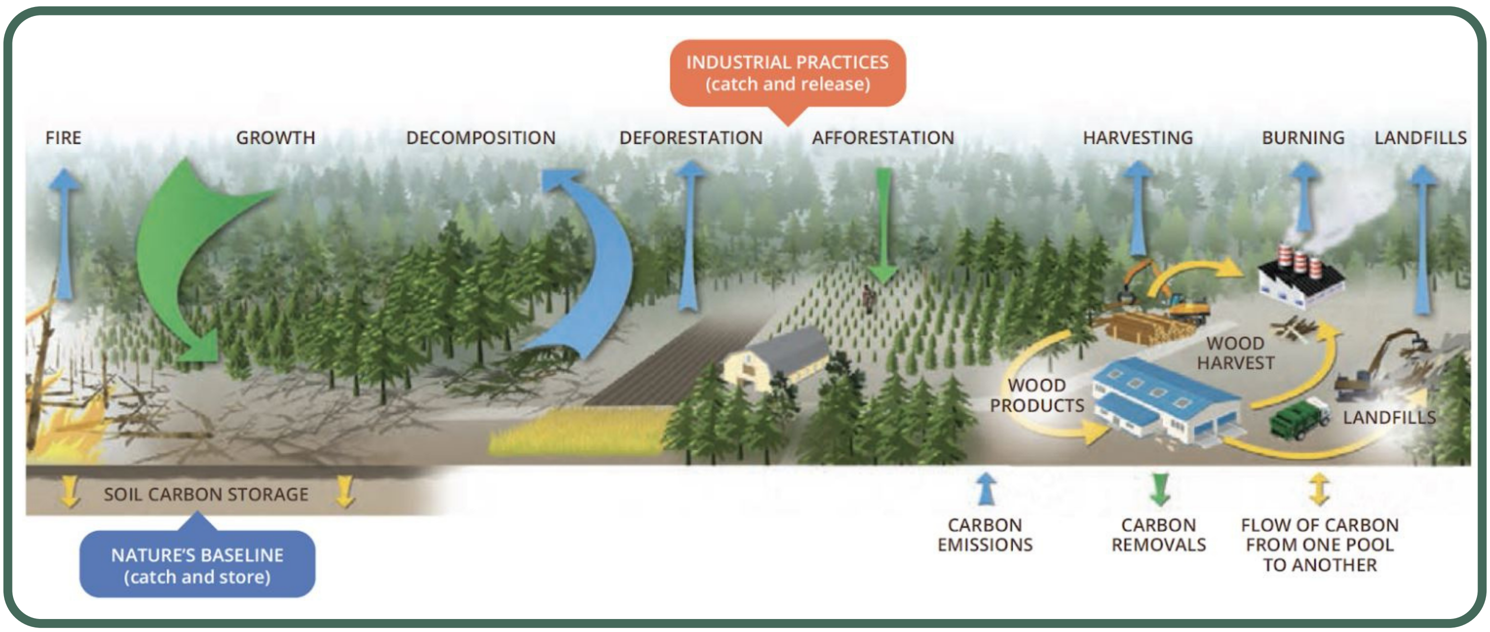
Older forests and large trees are far superior to logged forests in climate mitigation and biodiversity benefits. Globally, mature forests store 30-50% more carbon than logged forests and up to half of the live-tree carbon stored in a forest is represented by the largest/oldest 1% of trees at the stand level (Lutz et al. 2018). Logging late-seral forests and large trees results in a net carbon debt and other irreplaceable losses that are not made up for via reforestation or wood product stores as the carbon present in older forests and soils takes centuries to

Carbon Stocks and Flows

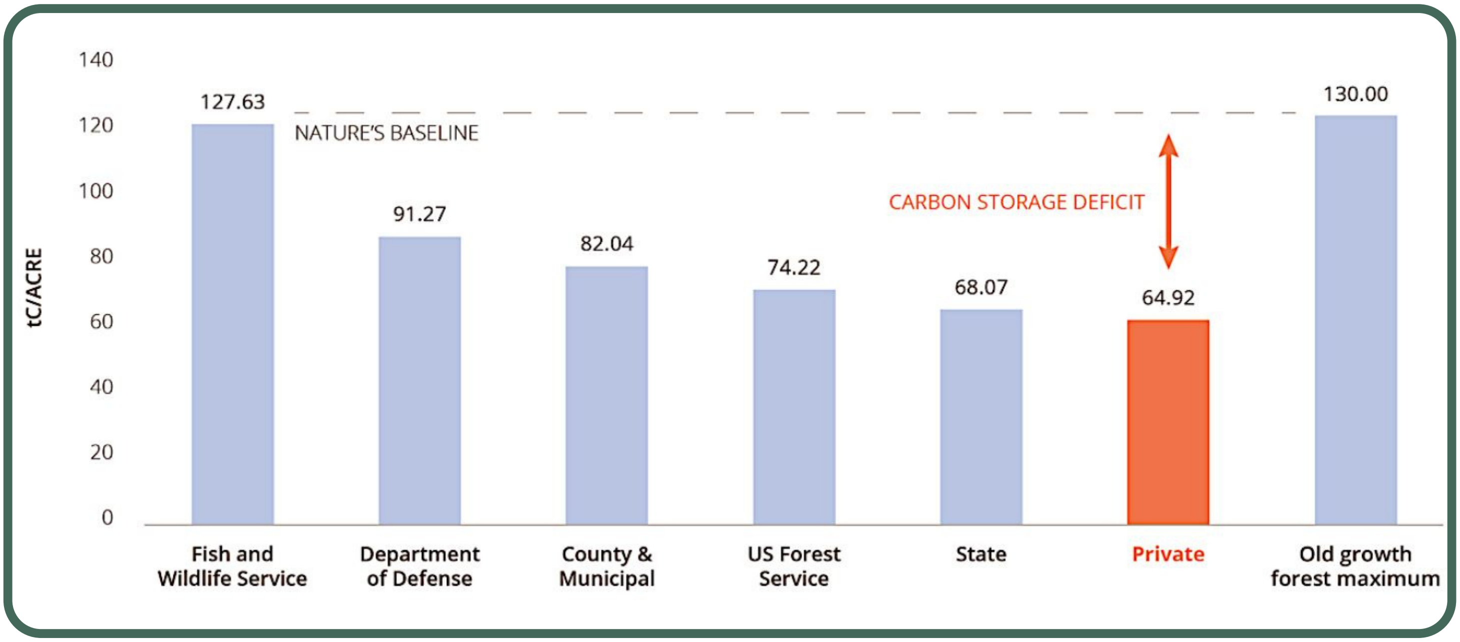
accumulate compared to much shorter-lived wood products that represent only a fraction of the original forest store (see Harmon 2019, Hudiburg et al. 2019).

Taking a hard look at carbon stocks and flows means using the proper baseline and models, and assessing the plan amendment in the context of the regional background (cumulative stocks and fluxes). The EA entirely fails to address carbon issues. The following explains how a carbon analysis could be undertaken with respect to the proposed action. The baseline for carbon stocks should be an approximation of the pre-industrial forest conditions as noted by the DACSE (2019) illustrations below.

Slow In, Fast Out (Life Cycle Analysis) - The illustration from the DACSE (2019) report below provides a schematic of what is generally involved with a carbon life cycle analysis, which the EA needs to present in order to take a hard look at emissions. In general, DACSE (2019) states that native forests can be thought of as a "catch and store" for carbon while industrial forestry represents a "catch and release" of carbon. Another way to look at this is Dr. Beverly Law's (Oregon State University) use of the slow-in (sequestration) and fast-out (emissions) analogy for how forestry practices affect carbon stocks and flows. That is – the process of photosynthesis results in a slow accumulation of carbon in trees over decades to centuries, while logging removes most of that carbon within years to decades.



Carbon Stocks and Flows



DACSE (2019) provides the Forest Service with a means for assessing the cumulative emissions effects of dead zones (Table below) in the surroundings (clearcuts and young plantations) with a recommendation to first and foremost allow forests on public lands to reach their maximum carbon potential (recovering stocks, minimizing flows) and to avoid additional emissions (Moomaw et al. 2019), especially in the context of private lands logging. The figures and tables provided are an example of what the Forest Service needs to include in the EA in order to satisfy the BASI requirements of the planning rule with respect to cumulative (direct, indirect, surroundings) emissions in a regional context.

Carbon Stocks and Flows

NET LANDSCAPE SCALE (10,000 ACRES) SEQUESTRATION BY ROTATION AGE

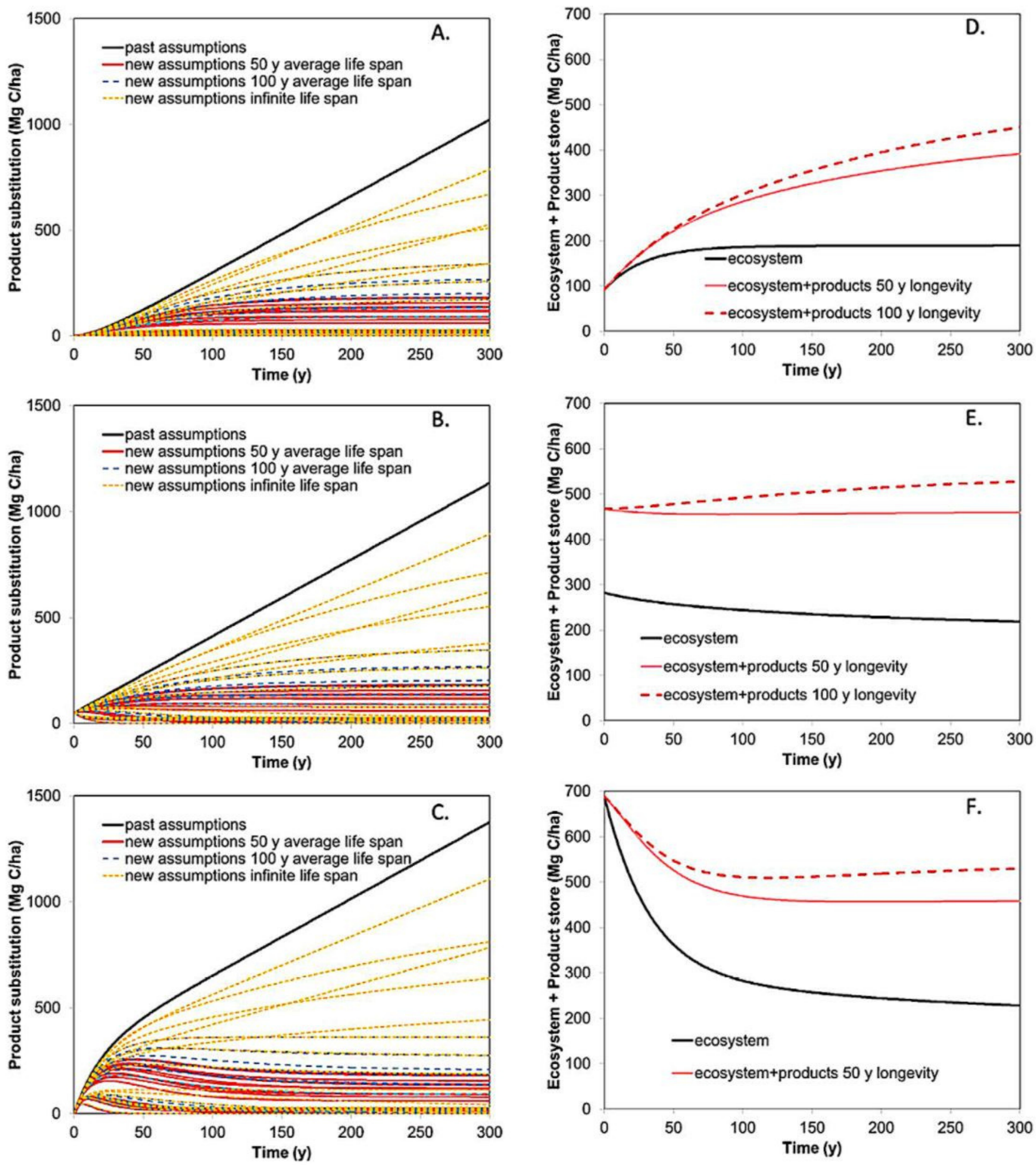
ROTATION AGE (YRS)	CLEARCUT ACRES/ YEAR	DEAD ZONE ACRES	DEAD ZONE %	DZ EMIT (TCO_2/YR)	RESID NEP (TCO_2/AC)	RESID SEQ (TCO_2/YR)	NET SEQ (TCO_2/YR)
20	500	6,500	65.00%	16,250	7	24,500	8,250
30	333	4,333	43.33%	10,833	6	34,000	23,167
40	250	3,250	32.50%	8,125	5	33,750	25,625
50	200	2,600	26.00%	6,500	5	37,000	30,500
60	167	2,167	21.67%	5,417	5	39,167	33,750
70	143	1,857	18.57%	4,643	4	32,571	27,929
80	125	1,625	16.25%	4,063	4	32,571	29,438
90	111	1,444	14.44%	3,611	4	34,222	30,611
100	100	1,300	13.00%	3,250	3	26,100	22,850
120	83	1,083	10.83%	2,708	3	26,750	24,042

Carbon stored in wood products or wood substitution for other materials does not offset carbon losses from logging – Harmon (2019) challenged the wood substitution argument commonly used in forestry circles as grossly inflated – emissions reductions were off by a factor of 2 to 100 as noted.

Abstract (Harmon (2019): Substitution of wood for more fossil carbon intensive building materials has been projected to result in major climate mitigation benefits often exceeding those of the forests themselves. A re-examination of the fundamental assumptions underlying these projections indicates long-term mitigation benefits related to product substitution may have been overestimated 2- to 100-fold. This suggests that while product substitution has limited climate mitigation benefits, to be effective the value and duration of the fossil carbon displacement, the longevity of buildings, and the nature of the forest supplying building materials must be considered.

Carbon Stocks and Flows

The figures from Harmon (2019) can provide the Forest Service with a proper means for assessing harvest wood product pools compared to leaving the carbon in the forest – which logging creates a net carbon debt emitted to the atmosphere.



Carbon Stocks and Flows

Harmon (2019) concludes:
Past analyses suggest product substitution benefits at the landscape level continue to increase at a constant rate into the future [6–16]. Moreover, they imply that while a carbon debt can be created in some situations (e.g. harvest of primary forests), that this debt is eventually paid back via product substitution [10 , 12 , 32]. While I examined only a few illustrative cases, in the case of product substitution, these debts would not be paid back if the displacement declines or there are losses via cross-sector leakage or related to product replacement. That is because negative feedbacks associated with losses can prevent product substitution from accumulating forever. These negative feedbacks could exist regardless of the forest ecosystem, the harvest system, and the efficiency of processing harvests into products as well as the proportion allocated to buildings. Thus, while I did not examine the effect on a wide range of ecosystems, or alternative harvest systems, or systems in which buildings are a minor faction of harvested carbon, these underlying relationships would not be altered for these new situations.

Full Life Cycle Analysis needs to be incorporated in plan revisions – Rigorous accountability methods, and not qualitative assessments, are necessary to evaluate options for minimizing emissions from forestry operations and harvested wood product pools (Hudiburg et al. 2019). Researchers warn against the use of *a priori* (or qualitative) assumptions about net carbon neutrality where biogenic emissions are underestimated because the carbon released from wood use is assumed displaced by regrowth. This assumption is fundamentally flawed (multiple citations in Hudiburg et al. 2019). Hudiburg et al. (2019) summarize the need for this analysis before claims can be made that wood product use or substitution is a valid way to store carbon compared to carbon stored in unlogged forests.

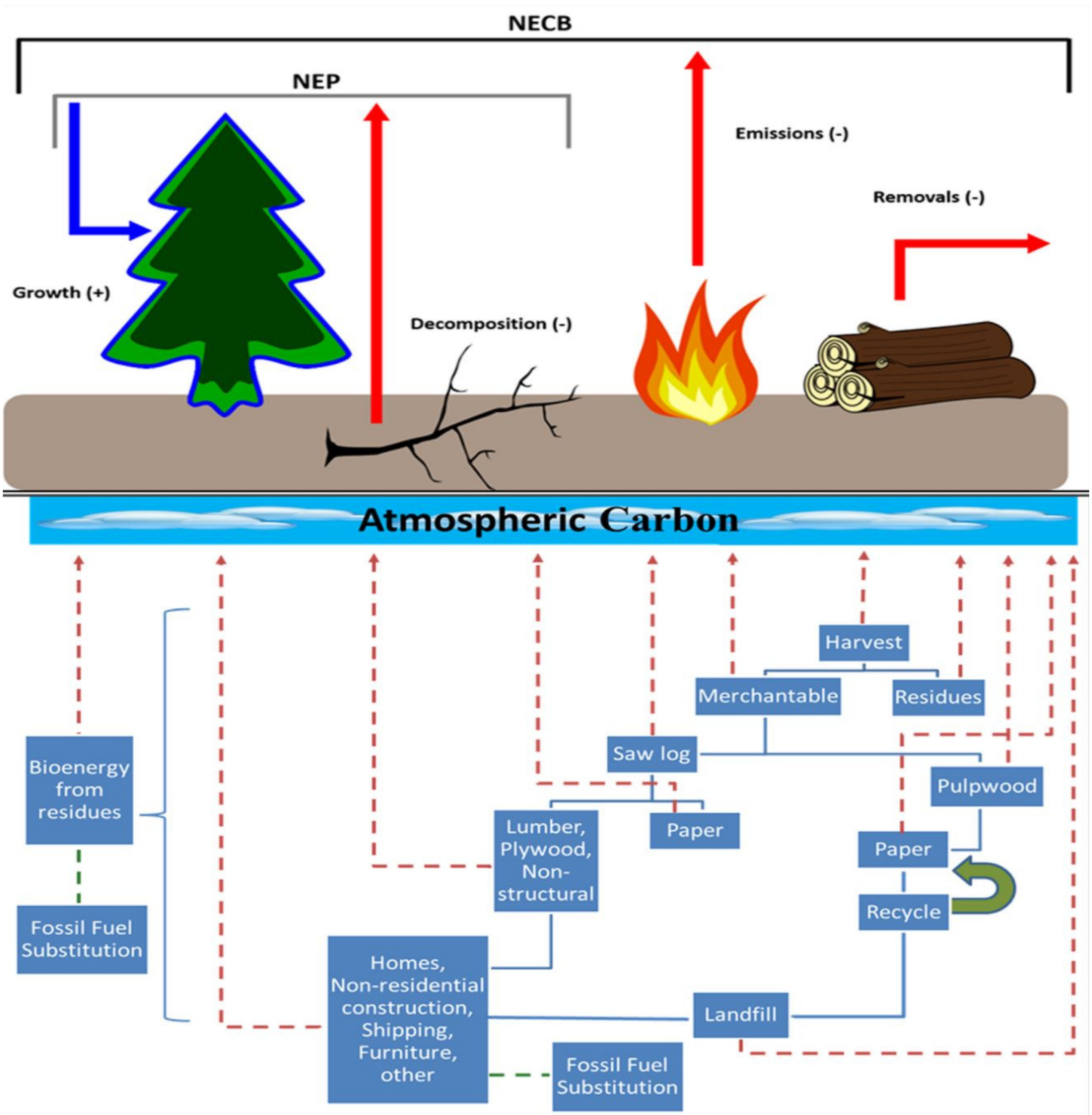
Carbon Stocks and Flows

Abstract (Hudiburg et al. 2019): Atmospheric greenhouse gases (GHGs) must be reduced to avoid an unsustainable climate. Because carbon dioxide is removed from the atmosphere and sequestered in forests and wood products, mitigation strategies to sustain and increase forest carbon sequestration are being developed. These strategies require full accounting of forest sector GHG budgets. Here, we describe a rigorous approach using over one million observations from forest inventory data and a regionally calibrated life-cycle assessment for calculating cradle-to-grave forest sector emissions and sequestration. We find that Western US forests are net sinks because there is a positive net balance of forest carbon uptake exceeding losses due to harvesting, wood product use, and combustion by wildfire. However, over 100 years of wood product usage is reducing the potential annual sink by an average of 21%, suggesting forest carbon storage can become more effective in climate mitigation through reduction in harvest, longer rotations, or more efficient wood product usage. Of the~10 700 million metric tonnes of carbon dioxide equivalents removed from west coast forests since 1900, 81% of it has been returned to the atmosphere or deposited in landfills. Moreover, state and federal reporting have erroneously excluded some product-related emissions, resulting in 25%–55% underestimation of state total CO2 emissions. For states seeking to reach GHG reduction mandates by 2030, it is important that state CO2 budgets are effectively determined or claimed reductions will be insufficient to mitigate climate change.

It should be noted that up to 40% of the harvested wood does not become a product and products decay over time, resulting in product accumulation much smaller than the total amount harvested (Hudiburg et al. 2019). Emissions need to be tracked from combustion of wood that does not become product, combustion for energy, decomposition and/or combustion at end-of-product chain.

Carbon Stocks and Flows

To illustrate the mechanics involved in life cycle analysis, Hudiburg et al. (2019) note the following modeling inputs that the Forest Service now needs to analyze in taking a hard look at emissions:



Carbon Stocks and Flows

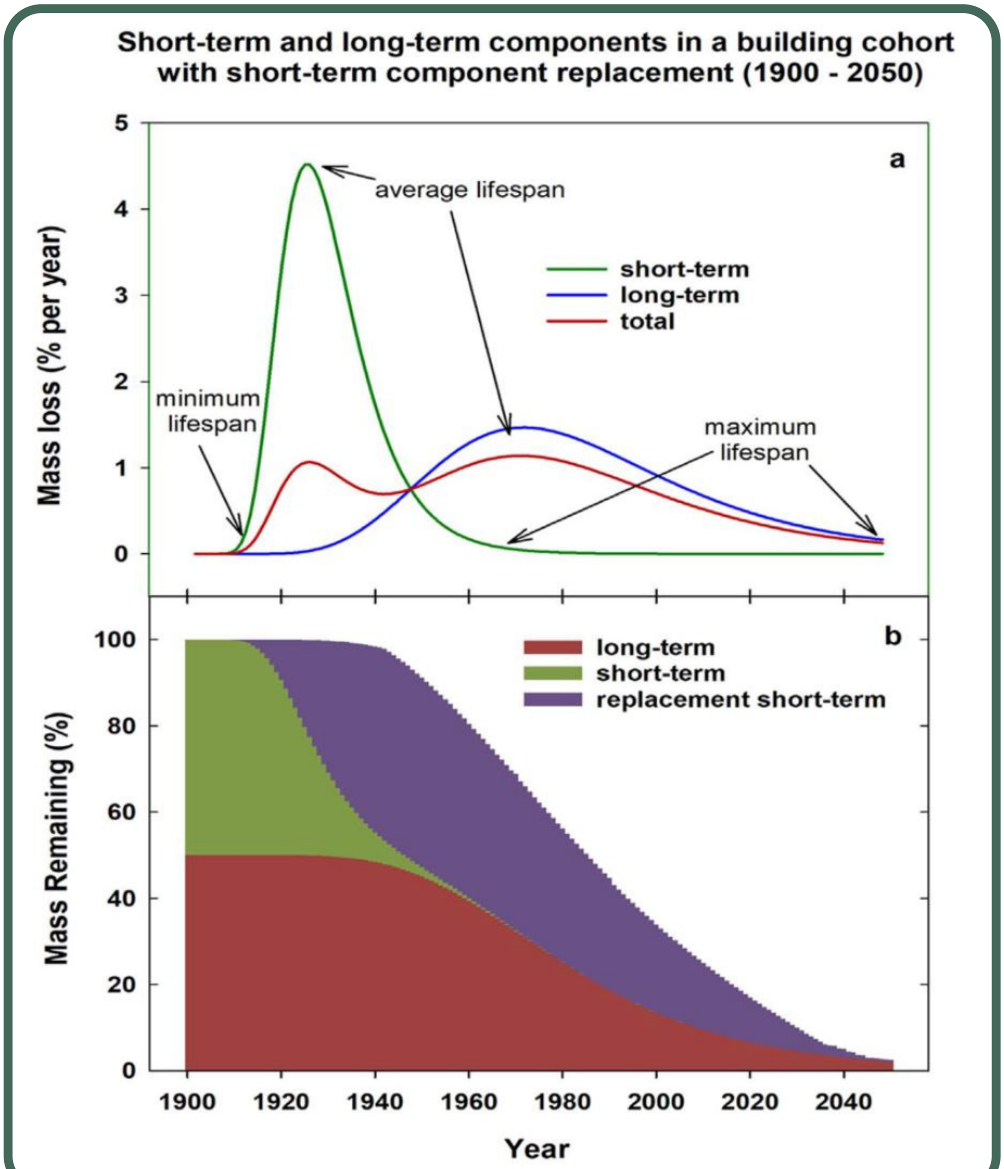
The model used by Hudiburg et al. (2019) relied on data inputs from the Forest Inventory Analysis (FIA) program of the Forest Service that is readily available and needs to be used in life cycle analysis in order to estimate emissions. Using this modeling approach, Hudiburg et al. (2019) provided average annual total flux estimates from logging and wood product use, which can readily be adopted for use in forest planning alternatives. Table 1 from Hudiburg et al. (2019) is an example of model outputs in a rigorous life cycle analysis that the Forest Service can use to assess planning alternatives given the agency has access to the same datasets.

Table 1. Average annual total fluxes by state and region from 2001 to 2016. All units are in million MT CO_2e. Negative numbers indicate a carbon sink (CO_2 is being removed from the atmosphere). The more negative the number, the stronger the sink. Grey shading is used to indicate net values that represent carbon sink strength both before and after removals are accounted for.

Ecosystem	Washington	Oregon	California	Total
1. Forested area (million hectares)	9.7	12.4	11.9	34.0
2. Net ecosystem production (NEP)	**−89.9**	**−102.0**	**−99.8**	**−291.6**
3. Fire emissions	5.1	5.3	10.3	20.7
4. Harvest removals	18.5	30.5	11.5	60.5
Net ecosystem carbon balance (NECB) (sum of rows 1 through 4)	**− 66.4**	**−66.2**	**−78.0**	**−210.5**
Forest industry	**Washington**	**Oregon**	**California**	**Total**
5. Harvest residue combustion (onsite)	3.9	6.5	2.5	12.9
6. Harvest, transportation, manufacturing (FFE emissions)	2.8	4.6	1.6	9.0
7. Wood product pool annual inputs	−18.5	−30.5	−11.5	−60.5
8. Landfill annual inputs (from products)	−6.8	−11.9	−4.2	−22.9
9. Wood manufacturing losses	3.9	6.5	3.9	14.3
10. Wood product and landfill decomposition	21.4	36.2	13.3	71.0
Net forest sector carbon balance (NECB + sum of rows 5 through 10)	**−59.5**	**−54.7**	**−72.4**	**−186.6**
11. Wood product substitution (wood)	−3.0	−4.9	−1.6	−9.4
12. Wood product substitution (energy)	−1.8	−3.0	−1.8	−6.6
Net forest sector carbon balance (with credits; NECB + sum of rows 5 through 12)	**−64.3**	**−62.6**	**−75.8**	**−202.7**

Using life cycle analysis and realistic wood product substitution estimates (in Harmon 2019), Hudiburg's analysis shows a decay rate of wood product stores that should provide reliable estimates of flux from logging and wood use in forest plan alternatives.

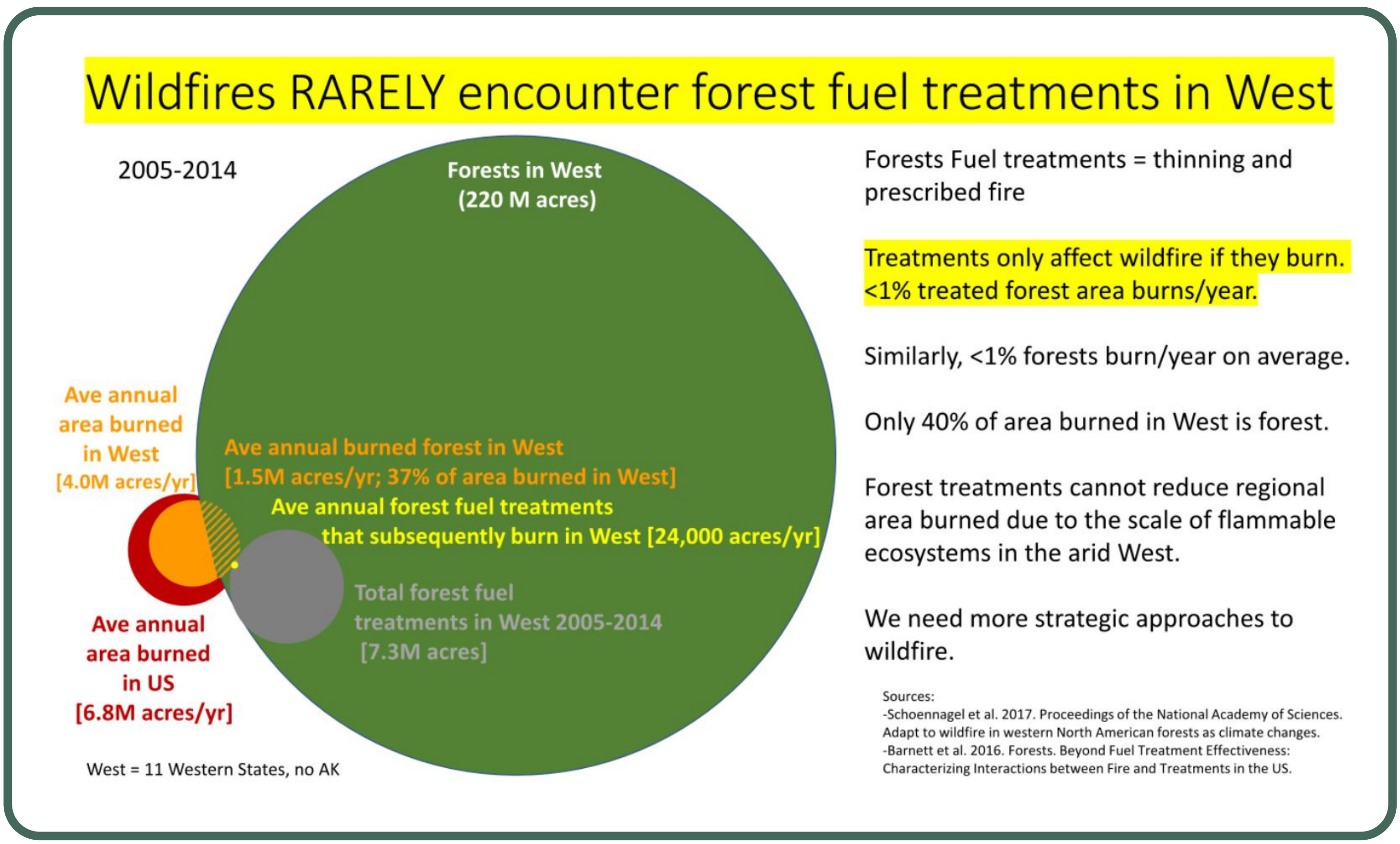
Carbon Stocks and Flows



It should be noted that Hudiburg et al. (2019) life cycle analysis estimated that harvest related emissions overall reduced the natural carbon sink by 27-46% in the three western states. Further, these researchers reported that not a single study has tracked the reliability of wood product substitution claims (i.e., degree of leakage occurring in "displaced" products then used elsewhere). Both the Harmon (2019) and Hudiburg et al. (2019) approaches illustrate fundamental flaws in wood product substitution and emissions claims commonly made in forestry circles.

Limitations of Thinning in a Changing Climate

The EA makes use of a Stand Density Index model (p. 22) to assess fire risk under different thinning scenarios, yet it does not include realistic probability assumptions regarding treatment efficacy. The figure below illustrates uncertainties of relying on thinning to reduce wildfire intensity given that the period when fuels are lowest is generally short lived in forests, and wildfires rarely encounter thinned sites when fuels are lowest (Schoennagel et al. 2017). The extremely low probability of fire and thinned site co-occurrence invalidates the EA assumptions about lowering fire intensity, and further invalidates presumed reduced tree mortality assumptions, through thinning and prescribed fire. Simply increasing the area thinned or burned under prescription does not change these odds appreciably given one cannot accurately predict when and where a fire will occur and many areas are inaccessible (Schoennagel et al. 2017). And, finally, thinning of pines and removal of non-pine tree species reduces stand resilience to pine-beetle outbreaks; retaining a diversity of tree sizes and tree species leads to natural bet-hedging against unpredictable future disturbances (Baker and Williams 2015), like beetle outbreaks, that could kill ponderosa pines over large areas.



Limitations of Thinning in a Changing Climate

Excessive opening of the tree canopy can lead to higher incidence of wind penetrance and rapid-fire spread, particularly if thinning is conducted on steep slopes and in remote areas with limited access making fine fuel consumption via prescribed fire impractical. In a warming climate where more extreme fire weather is likely, thinning is even less likely to alter fire behavior (Abatzoglou and Williams 2017, Schoennagel et al. 2017). In sum, thinning and/or prescribed fire cannot be assumed to lower fire intensity given significant limitations (Box 2) nor does it mimic the high quality of habitat conditions produced by natural disturbance events that generate biological legacies (as discussed above).

Box 2. Limitations of thinning.

(1) Thinning reduces habitat for canopy dependent species, including many raptors, requires an expansive road network damaging to aquatics, can spread invasive and flammable weeds, and, when implemented over large landscapes, releases more carbon emissions than fires, even severe ones (Campbell et al. 2012).
(2) As noted, there is a very low probability (<1%) that a thinned forest will encounter a fire during the narrow period (10-20 years depending on site factors) of reduced "fuels," resulting in large-scale thinning proposals that alter forest conditions over large areas.
(3) Excessive thinning (e.g., reducing bulk crown density below 60%) can increase wind speeds and solar radiation to the ground causing increased flammable vegetation growth and fire spread.
(4) Thinning needs to be followed by prescribed fire to reduce flammable slash but this can cause soil damage especially if burning is concentrated in piles (intensifies heat effects to soils).
(5) Thinning is seldom cost effective without public subsidies or removing large fire-resistant trees.
(6) In some regions (Sierra, Klamath-Siskiyou), time since fire is not associated with increasing fire risks (i.e., as forests mature, they become less flammable).
(7) Thinning efficacy is limited under extreme fire weather (principal factor governing large fires).

Limitations of Thinning in a Changing Climate

Box 2. Limitations of thinning cont.

(8) At regional scales, active management (unspecified forms of logging) combined with extreme fire weather have been associated with uncharacteristic levels of high severity fires (Bradley et al. 2016, Zald and Dunn 2018).

(9) Preemptive thinning for insect concerns may reduce susceptibility to small outbreaks but is unlikely to reduce susceptibility to large, landscape-scale epidemics. Once beetle populations, for instance, reach widespread epidemic levels, silvicultural strategies aimed at stopping them are not likely to reduce forest susceptibility to outbreaks. Furthermore, such silvicultural treatments could have substantial, unintended short- and long-term ecological costs associated with road access and an overall degradation of natural areas (Black et al. 2013).

Black et al (2013) looked at this issue in depth in a comprehensive literature review of thinning efficacy in terms of insect outbreaks. The authors raise substantial uncertainties in thinning assumptions not recognized in the EA, which instead assumes thinning will lower tree mortality. The following is in except from that review.

Prior to Outbreaks

The effectiveness of thinning to reduce forest susceptibility to bark beetles is believed to be related to tree vigor (Fettig et al. 2007); which may increase as moisture stress is decreased, and which in turn may make trees less susceptible to insect infestation. The premise is that if the trees are healthy and vigorous, they may be able to "pitch out" the attacking beetles, essentially flooding the entrance site with resin that can push out or drown the beetle [...].

Some studies have suggested that competition for light and water may reduce the vigor of surviving trees and increase susceptibility to bark beetle attacks (Fettig et al. 2007) and that thinning may, therefore, improve outbreak resistance. For instance, low-vigor ponderosa pine (Pinus ponderosa) in central Oregon was more often attacked by beetles than high-vigor trees during early stages of outbreaks (Larsson et al. 1983). Similarly, beetle activity has been associated with high tree densities in ponderosa pine and Douglas-fir (Pseudotsuga sp.) stands (Negrón et al. 2001; Negrón and Popp 2004). Ponderosa pine study plots in Colorado's Front Range infested by mountain pine *beetle had significantly higher tree basal area and density (Negrón and Popp 2004). Douglas-fir beetles (D. pseudotsugae) more often attacked stands containing a high percentage of basal area represented by high densities of Douglas fir and slow growth during the five years prior to attack in Colorado's Front Range (Negrón et al. 2001).*

Studies that have looked directly at thinning and its effects on tree vigor in Western forests have shown mixed results. While some studies have found that thinning reduces stand susceptibility in some circumstances (Fettig et al. 2007), other research has found bark beetles do not preferentially infest trees with declining growth. For example, Sánchez-Martínez and Wagner (2002) found that ponderosa pine forests of northern Arizona growing in dense stands were not more likely to be colonized by bark beetles.

Under some circumstances, thinning may alleviate tree stress at the stand level but is unlikely to be effective at mitigating susceptibility against extensive or severe outbreaks (Safranyik and Carroll 2006). Preisler and Mitchell (1993) found that thinned plots of lodgepole pine in Oregon were initially unattractive to mountain pine beetles; but when large numbers of attacks occurred, colonization rates were similar to those in unthinned plots. Similarly, Amman et al. (1988) studied the effects of spacing and diameter of trees and concluded that tree mortality was reduced as basal area was lowered. However, if the stand was in the path of an ongoing mountain pine beetle epidemic, spacing and density of trees had little effect (Amman et al. 1988).

While thinning has the potential to reduce tree stress, which can reduce susceptibility to insect attack, it also has the potential to bring about other conditions that can increase susceptibility. For example, thinning may injure surviving trees and their roots, which can provide entry points for pathogens and ultimately reduce tree resistance to other organisms (Hagle and Schmitz 1993; Paine and Baker 1993; Goyer et al. 1998). Although thinning can be effective in maintaining adequate growing space and resources, there is accumulating evidence to suggest that tree injury, soil compaction, and temporary stress due to changed environmental conditions caused by thinning may increase susceptibility of trees to bark beetles and pathogens (Hagle and Schmitz 1993).

During Outbreaks

There is general agreement that silvicultural treatments cannot effectively stop outbreaks once a large-scale insect infestation has started. Citing multiple sources, Hughes and Drever (2001) found that most control efforts have had little effect on the final size of outbreaks. In another

Limitations of Thinning in a Changing Climate

review, Romme et al. (2006) point out that once an extensive outbreak has started, timber management is unlikely to stop it. Control of such outbreaks is theoretically possible, but it would require treatment of almost all of the infected trees (Hughes and Drever 2001). Amman and Logan (1998) point to failed attempts to use direct control measures, such as pesticides and logging, after an infestation starts. They suggest that by the early 1970s, it was apparent that attempts to control the extensive mountain pine beetle outbreaks that were occurring in the northern Rockies, by directly killing the beetles, were not working.

If a bark beetle infestation is relatively restricted and concentrated in a limited area, it may be feasible to reduce the impact of that outbreak by removing infested trees from a forest stand, or by thinning a stand to reduce stress of trees competing for limited nutrients, sunlight, and moisture. However, specific climatic conditions are believed to be required for beetle populations to reach epidemic levels. As such, a small population of beetles is not sufficient for an outbreak to occur and would not necessarily lead to an outbreak. Conversely, under climatic conditions favorable for an outbreak, bark beetle outbreaks can erupt simultaneously in numerous, dispersed stands across the landscape. Thus, even if a growing population of beetles is successfully removed from one stand or the stand is thinned to increase vigor, under climatic conditions suitable for outbreaks, beetles from other stands are likely to spread over a landscape. Given that climate typically favors beetle populations and stresses trees over very large areas, successfully identifying and treating stands over a large enough region to have a significant impact on the overall infestation is impractical and costly.

Following Outbreaks

Post-disturbance harvest is common practice on forest lands and is designed to remove trees or other biomass in order to produce timber or other resources. This type of resource extraction has the potential to inadvertently lead to heightened insect activity (Nebeker 1989; Hughes and Drever 2001; Romme et al. 2006). In particular, snags and fallen logs contribute to the protection of soils and water quality and provide habitat for numerous cavity and snag-dependent species (Romme et al. 2006), many of which prey on bark beetles and other economically destructive insects. Therefore, outbreaks could be prolonged because of a reduction in the beetle's natural enemies (Nebeker 1989), including both insects and bird species that feed on mountain pine beetles (Koplin and Baldwin 1970; Shook and Baldwin 1970; Otvos 1979). Furthermore, post-disturbance harvest can damage soil and roots by compacting them (Lindenmayer et al. 2008) leading to greater water stress in trees, which may reduce conifer regeneration by increasing sapling mortality (Donato et al. 2006) and, in general, may cause more damage to forests than that caused by natural disturbance events (DellaSala et al. 2006).

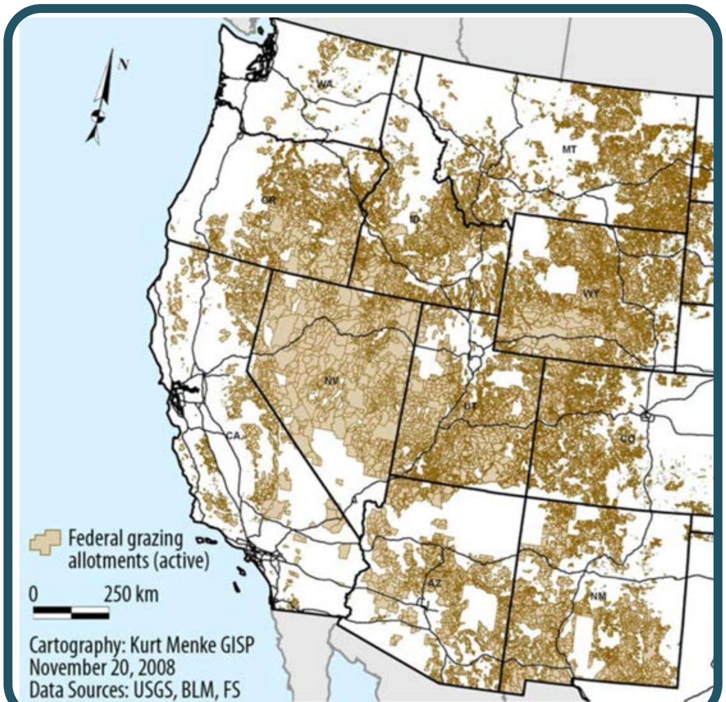
Inadequate Attention to the Top Threat on National Forests: Livestock and Climate Change

The EA fails to acknowledge (and reduce) the cumulative impacts of livestock grazing's interaction with climate change in greatly compromising ecological integrity. These stressors have synergistically transformed native grasslands, oak woodlands, stream banks, and riparian vegetation, representing the top threat on public lands across the West (Beschta et al. 2012). The omission of available information, like the Beschta et al. peer reviewed study, creates scientific uncertainties and public controversy, which compromise the agency's ability to maintain ecological integrity--particularly by not meeting the BASI standard or higher-level confidence in the information presented in the EA for a project of significant impacts to the environment (i.e., failure to take a hard look at the evidence).

Excerpted from Beschta et al. (2012)

Abstract: Climate change affects public land ecosystems and services throughout the American West and these effects are projected to intensify. Even if greenhouse gas emissions are reduced, adaptation strategies for public lands are needed to reduce anthropogenic stressors of terrestrial and aquatic ecosystems and to help native species and ecosystems survive in an altered environment. Historical and contemporary livestock production—the most widespread and long-running commercial use of public lands—can alter vegetation, soils, hydrology, and wildlifespecies composition and abundances in ways that exacerbate the effects of climate change on these resources. Excess abundance of native ungulates (e.g., deer or elk) and feral horses and burros add to these impacts. Although many of these consequences have been studied for decades, the ongoing and impending effects of ungulates in a changing climate require new management strategies for limiting their threats to the long-term supply of ecosystem services on public lands. Removing or reducing livestock across large areas of public land would alleviate a widely recognized and long-term stressor and make these lands less susceptible to the effects of climate change. Where livestock use continues, or where significant densities of wild or feral ungulates occur, management should carefully document the ecological, social, and economic consequences (both costs and benefits) to better ensure management that minimizes ungulate impacts to plant and animal communities, soils, and water resources. Reestablishing apex predators in large, contiguous areas of public land may help mitigate any adverse ecological effects of wild ungulates.

Inadequate Attention to the Top Threat on National Forests: Livestock and Climate Change



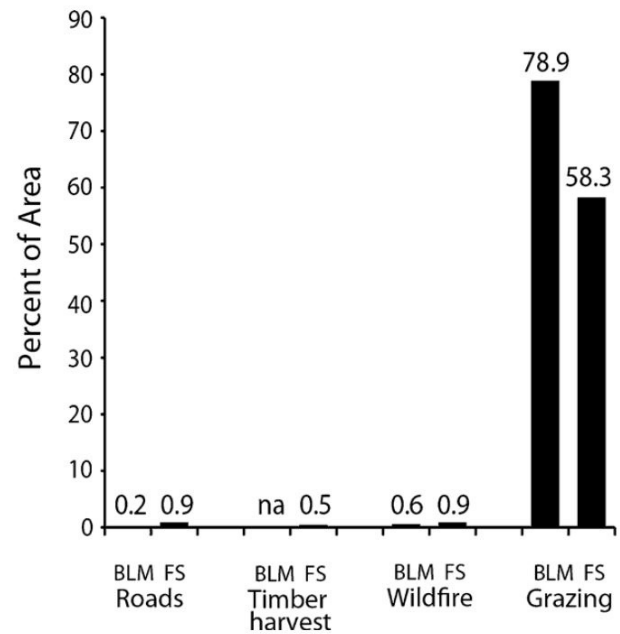


Fig. 3 Percent of Bureau of Land Management (BLM) and US Forest Service (FS) lands in eleven western states that are occupied by roads or are affected annually by timber harvest, wildfire, and grazing. *Data sources* Roads, BLM (2009) and FS, Washington Office; Timber harvest (2003–09), FS, Washington Office; Wildfire (2003–09), National Interagency Fire Center, Missoula, Montana; Grazing, BLM (2009) and GAO (2005). ''na'' = not available

From Beschta et al. (2012, Table 1)

Table 1 Generalized climate change effects, heavy ungulate use effects, and their combined effects as stressors to terrestrial and aquatic ecosystems in the western United States

Climate change effects	Ungulate use effects	Combined effects
Increased drought frequency and duration	Altered upland plant and animal communities	Reduced habitat and food-web support; loss of mesic and hydric plants, reduced biodiversity
Increased air temperatures, decreased snowpack accumulation, earlier snowmelt	Compacted soils, decreased infiltration, increased surface runoff	Reduced soil moisture for plants, reduced productivity, reductions in summer low flows, degraded aquatic habitat
Increased variability in timing and magnitude of precipitation events	Decreased biotic crusts and litter cover, increased surface erosion	Accelerated soil and nutrient loss, increased sedimentation
Warmer and drier in the summer	Reduced riparian vegetation, loss of shade, increased stream width	Increased stream temperatures, increased stress on cold-water fish and aquatic organisms
Increased variability in runoff	Reduced root strength of riparian plants, trampled streambanks, streambank erosion	Accelerated streambank erosion and increased sedimentation, degraded water quality and aquatic habitats
Increased variability in runoff	Incised stream channels	Degraded aquatic habitats, hydrologically disconnected floodplains, reduced low flows

Failure to Consider the Importance of Complex Early Seral Forests (CESF) and Severe Disturbances That Generate Them

Both the EA and GTR recognize the importance of early seral species but are silent on complex early seral forests even though this forest type is as biodiverse as old-growth forests, the seral stages are connected through time (what happens to one affects the other), and they are only generated by large and small severe disturbances in forests already having structure before the disturbance (Swanson et al. 2011, Donato et al. 2012, DellaSala and Hanson 2015, DellaSala et al. 2017). This lack of recognition in the EA for an important forest type is reflected in the problems noted in the outdated (1996) structure class diagram (EA:Figure 2) used to describe seral stages and fuel loads in the EA. For instance, the stand initiation stage illustration is an oversimplification that lacks recognition of the critically important pulse of biological legacies created by severe disturbance in these forests. While the EA mainly considers them "fuels," these structures are the rarest and most important forest elements on the landscape and part of the uniqueness of this seral stage (Swanson et al. 2011). The diagram needs to be updated and the importance of complex early seral forests generated largely by severe burns (large and small high-severity patches) fully acknowledged as a biologically distinct community in the EA.

Large and small high-severity patches provide important foraging habitat for Northern Goshawks (at-risk species), ungulate foraging habitat (Bond 2015), snowshoe hare/lynx dynamics, woodpeckers (including at-risk species: Lewis's woodpecker), songbirds (Hutto et al. 2015), bats (Chambers and Saunders 2013), and boreal owls (at-risk species) in upper elevation spruce-fir forests. The EA inappropriately and arbitrarily assumes high-severity patches constitute habitat loss.

Ecological Importance of High Severity Patches – the hyperbola surrounding "megafire" claims centers largely around presumed patch size increases. However, in the largest dataset analyzed, high severity patches increased briefly (1984-91) and then leveled off throughout western forests (DellaSala and Hanson 2019, peer reviewed). Additionally, these researchers rejected the notion that conifer establishment was insufficient even in the largest patches. Such assumptions underline forest management approaches in the EA and are refuted.

From DellaSala and Hanson (2019):

> "*Over the entire time series, 1984-2015, there was a significant increasing trend in the combined total area of CESF [complex early seral forests] patches >400 ha in each year (τ = 0.407, p =*

Failure to Consider the Importance of Complex Early Seral Forests (CESF) and Severe Disturbances That Generate Them

0.001), but no trend in patch size (τ = 0.009, p = 0.802). However, when the data were analyzed by time periods, there was only a significant difference in the annual area of CESF habitat created by high-severity fire relative to the earliest time period (1984-1991), but no significant differences were detected among time periods since the early 1990s (Table 1, Figure 2). With regard to the size of individual large CESF patches, there were no significant differences detected among time periods [...]."

Table 1. Critical values (q0.05, 4), absolute difference between mean of ranks (|RA-RB|), standard errors (SE), and test statistics (q) to assess statistical significance, at α = 0.05, of any differences between the four time groups ("1" = 1984-1991, "2" = 1992-1999, "3" = 2000-2007, and "4" = 2008-2015) for total annual area of CESF patches >400 ha using the Nemenyi non-parametric test for multiple comparisons between groups with an equal sample size (n = 8 years for each time group). Statistical significance of levels of q are shown as "Y" (significant) or "N" (not significant).

Time group comparison	$q_{0.05,4}$	$\lvert R_A - R_B \rvert$	SE	q	Significant? ($q > _{0.05,4}$?)
1-2	3.63	45.0	26.53	1.70	N
1-3	3.63	108.0	26.53	4.07	Y
1-4	3.63	107.0	26.53	4.03	Y
2-3	3.63	63.0	26.53	2.37	N
2-4	3.63	62.0	26.53	2.34	N
3-4	3.63	1.00	26.53	0.04	N

Failure to Consider the Importance of Complex Early Seral Forests (CESF) and Severe Disturbances That Generate Them

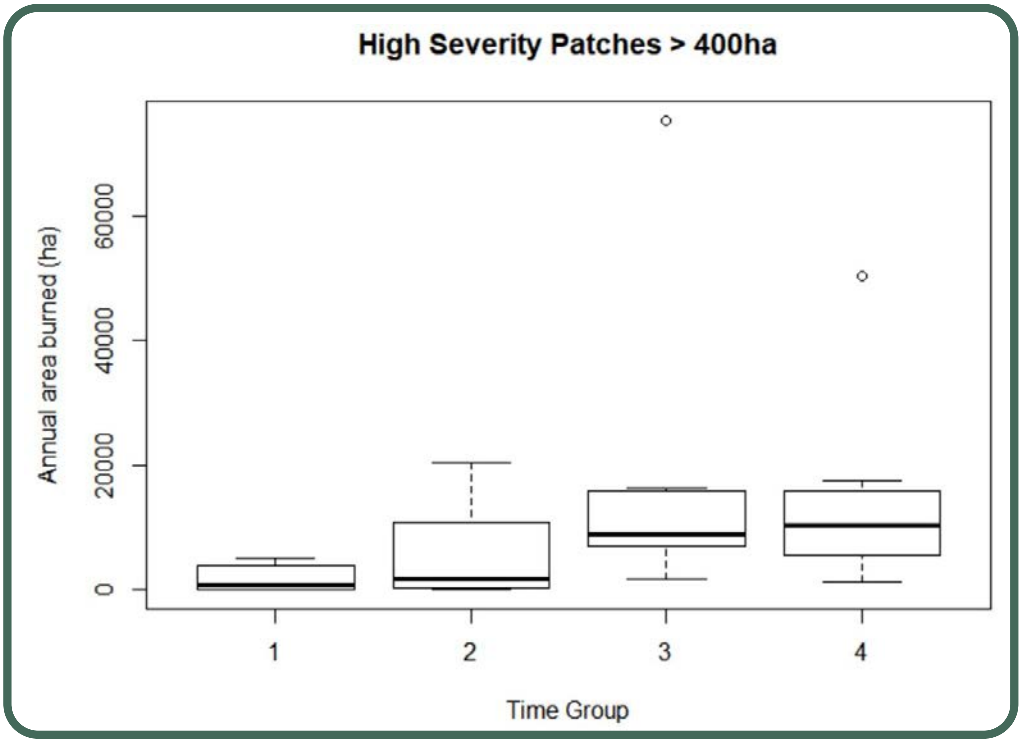


Figure 2 from DellaSala et al. Annual area of large patches (>400 ha) of high-severity fire in the four time periods ((“1” = 1984-1991, “2” = 1992-1999, “3” = 2000-2007, and “4” = 2008-2015).

Importantly, Hutto et al. (2016) recommended that managers maintain ecological integrity of western dry pine and mixed-conifer forests through a more informed approach to the importance of mixed and high-severity fires. Here is their abstract:

Abstract (Hutto 2016): We use the historical presence of high-severity fire patches in mixed-conifer forests of the western United States to make several points that we hope will encourage development of a more ecologically informed view of severe wildland fire effects. First, many plant and animal species use, and have sometimes evolved to depend on, severely burned forest conditions for their persistence. Second, evidence from fire history studies also suggests that a complex mosaic of severely burned conifer patches was common historically in the West. Third, to maintain ecological integrity in forests born of mixed-severity fire, land managers will have to accept some severe fire and maintain the integrity of its aftermath. Lastly, public education messages surrounding fire could be modified so that people better understand and support management designed to maintain ecologically appropriate sizes and distributions of severe fire and the complex early-seral forest conditions it creates.

Failure to Disclose and Consider Best Available Scientific Information (BASI) and Independent Scientific Information

DellaSala et al. (2017) recommend that managers include mixed-severity effects in dry pine and mixed conifer forests to achieve ecological integrity and plant diversity. And while much of the EA project area can be assumed to be in a xeric condition conducive to pines, mixed-severity fire effects, including large and small high-severity patches are indeed characteristic (Hessburg et al 2007, Odion et al. 2014), need to be maintained, and are being grossly underestimated in ecological importance throughout the EA and GTR.

The 2012 planning rule requires forest plans to meet the BASI standard during plan updates (Box 3). Importantly, the agency also should give greater weight to independent science that has no connection to agency funding, GTR reports, or agency publications. We refer to this as the Best Available Independent Science (BAIS). The EA misses the mark on both criteria by failing to disclose the vast majority of *accurate, reliable, and relevant* science that conflicted with the proposed action.

Box 3. 36 C.F.R. § 219.3 Role of science in planning.

The responsible official shall use the best available scientific information to inform the planning process required by this subpart. In doing so, the responsible official shall determine what information is the *most accurate, reliable, and relevant to the issues being considered.* The responsible official shall document how the best available scientific information was used to inform the assessment, the plan decision, and the monitoring program as required in §§ 219.6(a)(3) and 219.14(a)(4). Such documentation must: identify what information was determined to be the best available scientific information, explain the basis for that determination, and explain how the information was applied to the issues considered.

Ryan et al. (2018) provide specifics on how best to at least meet the BASI standard as illustrated in their Figure below. They specify “the definition of BASI is contained in the “zero code” chapter of the handbook and specifies three primary criteria: *accuracy,*

Failure to Disclose and Consider Best Available Scientific Information (BASI) and Independent Scientific Information

reliability, and relevance (FSH 1909.12.07.12), in addition to the Data Quality Act (PL 106–554) for guidance on evaluating available information. *Available* is defined as information that currently exists in a form useful for the planning process without further data collection, modification, or validation (FSH 1909.07.01). All the citations provided in this report were *available* to the agency, yet most were not considered in the EA presumably because they do not support the prevailing views of the authors of the GTR (confirmation bias). Thus, the agency cannot claim it is meeting either BASI or the more rigorous BAIS standard as there is a critical gap in the literature coverage and a lack of independence that is evident in a strong underlying bias about wildfires, large tree logging, and thinning.

For example, the no action alternative in the EA was developed, in part presumably, from input given by independent scientists at the May 11 Science Forum, yet, the input was readily dismissed by the agency in the EA. The science forum reflected major differences between the independent scientist panel vs. those with prior involvement in Forest Service studies and agency funding (e.g., GTR scientists; especially Hessburg, Johnston, and Spies cited extensively). That controversy reflects a lack of agreement in the science community which should be reflected in the NEPA analysis and GTR as *uncertainties* with risky outcomes; it is not.

In our review of the supporting materials of the EA the Forest Service, for instance, we noted serious flaws in the GTR. The report is tied mainly to a similar set of viewpoints on fire and large tree removal (Hessburg, Spies, Johnston cited extensively), while science not supportive of this view was omitted. While the supportive publications provide an important perspective within the larger constellation of literature, the EA and GTR are out of balance with the wider body of evidence that does not agree with these narrow agency-supportive positions.

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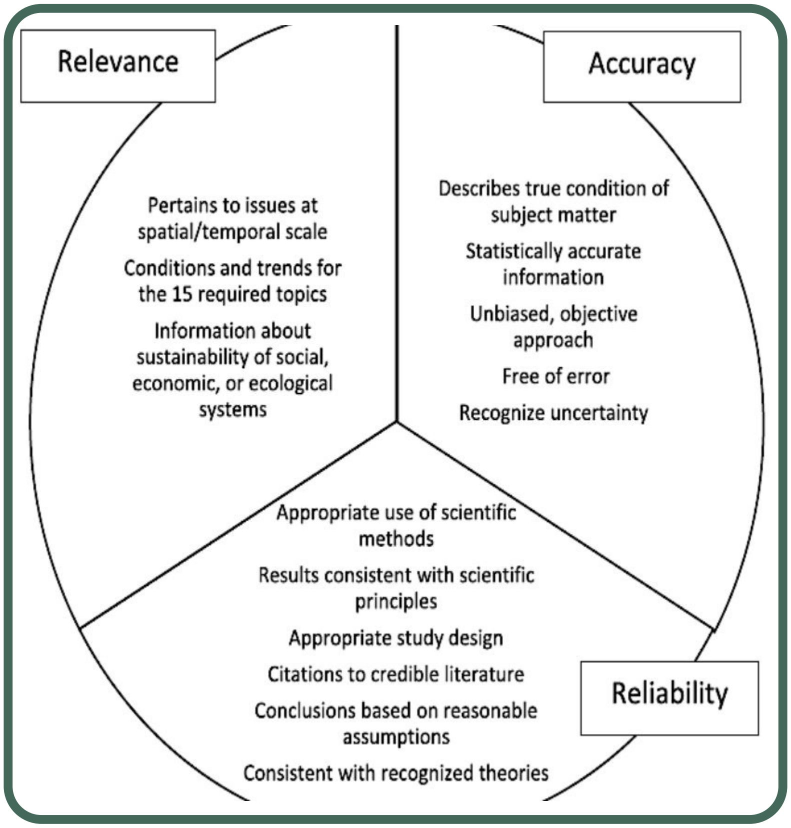

For example, on p. 8, the EA states that a variety of empirical studies and science syntheses demonstrate that protection of all large, greater than 21 inch, trees "has prevented restoration of historical conditions." It should be noted that of the 6 cited studies in support of this supposition, one study was in review (not published or available to the public), 4 were the same author (Johnston), and one was an agency GTR (Stine) non-peer reviewed publication loaded up with some of the same authors as the GTR. This is hardly a "variety of studies" particularly since the cited Johnston studies (4 of them) are based on limited fire scar sampling (as noted) and one (Merschel) is from a single locality (central OR) with sampling problems (as noted) and involves a Forest Service scientist (Spies) with the same perspective as the GTR authors. Thus, the Forest Service has skipped over much of the *relevant* and readily *available* independent literature and the EA suffers from confirmation bias and self-citations that have resulted in inconsistencies regarding the ecological importance of large trees vs. the agencies' preference to log them now.

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For example, on the one hand the EA and GTR aptly note the importance of large trees to wildlife but then the agency turns around and concludes that there are too many large firs, even though large trees overall are at a substantial deficit compared to the period before European colonization and increase in logging pressures circa 1920s. If the purpose and intent of forest planning is to grow more large trees, then taking the accumulated growth of decades of large firs off the board (so to speak) reflects an inherent bias that does not comport with the structural, wildlife, and carbon values of these critical legacy elements (see below).

The GTR and EA repeatedly underestimate the importance of large trees of *all species* in sequestering and storing carbon. In particular, the EA is silent on emissions from logging large trees. This serious omission of carbon-climate related impacts from logging is directly contrary to what many scientists are calling for regarding increased protections for large trees and old forests as nature-based climate solutions (Mackey et al. 2012, 2014, Lindenmayer et al. 2012, 2013, Griscom et al. 2017, Law et al. 2018, Moomaw et al. 2019). Notably, the Intergovernmental Panel on Climate Change (IPCC 2018) called for countries to reduce emissions, including from the land sector, and discussed the important role of forests in uptake and storage of atmospheric carbon. The EA skips over this literature with no discussion of emissions or importance of retaining accumulated carbon stores.

Although the Forest Service invited independent scientists to participate in their May 11, 2020 Science Forum Webinar prior to development of a proposed action, the agency ultimately failed to disclose and consider the information provided. Dr. DellaSala participated in the webinar along Drs. Bev Law, John Alexander, and Chad Hanson. The information presented by these scientists was not analyzed in the EA. For example, Dr. DellaSala made the case for the drop and leave tree option, pruning lower branches, and girdling trees in cohorts as ways to meet the agency's goals. None of those options were analyzed as the agency claims it is not practical and that it would raise fuel hazards. The agency reached this conclusion without providing any evidence. Neither did the Forest Service consider the hazardous fuels created by removing the tree bole (least flammable). After removal of tree boles and other large tree structure components, forests are left much more open, subject to winds and solar radiation, and are hotter and drier. Smaller diameter material such as twigs and branches, slash, and slash piles are typically left on the ground, further adding to increased fire hazards after logging. Thus, in dismissing a non-yarding

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alternative, the EA reflects a lack of consideration for any science that conflicts with the agency's desires to log big trees. The EA claims there is "new science" to justify lifting the screens instead of leaving large trees on site, yet, again citing only the limited science criticized above for confirmation bias (Johnston, Merschel, Spies, Hessburg).

The EA fails to disclose relevant state scientific disputes (disagreements, controversy) and instead includes biased perspectives on fire, as generally noted by Iftekhar and Pannell (2015) and Moritz et al. 2018 (below). The following biased perspectives summarized from Iftekhar and Pannell (2015) are widespread in the EA and GTR:

- Action bias – tendency to take actions even when it is better to delay action (in this case the impacts of aggressive thinning may be more significant than effects of fire on ecosystems given uncertainties of treatment effectiveness as noted below).

- Framing effect – tendency to respond differently to alternatively worded but objectively equivalent descriptions of the same item (failure to account for ecosystem benefits of mixed-severity fires, including periodic flare-ups of high severity patches).

- Reference-point bias – tendency to overemphasize a predetermined benchmark for a variable when estimating the level of that variably (i.e., over-reliance on fire scar sampling rather than presenting more robust and multiple lines of evidence, noted below).

- Satisficing rule – tendency to stop searching for a better decision (i.e., a NEPA based range of alternatives) once a decision that seems sufficiently good is identified.

- Loss aversion – tendency to value losses more highly than similar gains (i.e., managing wildfire of moderate-high intensity effects for ecosystem benefits instead of avoiding it by mechanical thinning and fire suppression).

- Limited reliance on systematic learning – tendency to use information from past

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successful efforts rather than using information from both successful and failed efforts via extensive and well-funded ecosystem monitoring (adaptive management and learning is not possible without well-funded monitoring).

Each of these biases need to be resolved by referring to multiple lines of evidence in reconstructing fire regimes, not relying mainly on fire scars, and conducting well-funded monitoring studies that fully assess project effects on species of conservation concern and ecological and cultural values. Multiple lines of evidence and monitoring are discussed in Odion et al. (2016) and Moritz et al. (2018) in the Common Ground Report (see below and section on problems with fire scar sampling).

Areas of Agreement/Disagreement (Common Ground): We participated as respondents in the so-called "Common Ground" report (Moritz et al. 2018) and are thoroughly familiar with the report's findings. The EA should pay particular attention to the following key findings in relation to areas of agreement, uncertainty, and disagreement and evaluate project actions based on certainty levels (full public disclosure) as follows.

Areas of Agreement (high certainty):

- The role of changing climatic conditions is increasingly important in influencing fires.
- Multiple fire ecology and fire history research can be useful.
- Heterogeneity of fire effects, including patterns of patches created by fires of *all severities*, is important to forest resilience to future fires.
- Generalized models of historical fire regimes vary by ecoregion and forest type.
- Even within the same ecoregion and forest type, there is variation in historical fire regimes among differing environmental gradients.
- Historically, *some degree of low-, moderate-, and high-severity fire* has occurred in all forest types, but in substantially different proportions and patch size distributions at different locations.

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- Classification of historical fire regimes according to forest types can be coarse; thus, failure to recognize variation of *historical fire regimes within forest types* can lead to overgeneralization and oversimplification of landscape conditions.

- *Presence of roads, road density and railways, livestock grazing, invasive species, and mining can alter fire regimes.* Even a single one of these influences can have profound effects on vegetation and fire behavior conditions. When present in combinations, cumulative effects will arise that may push ecosystems past tipping points (Paine et al. 1998, Lindenmayer et al. 2011).

- Knowledge of historical range of variability (HRV) is useful but *does not dictate land management goals.* HRV findings from one area may or may not have relevance elsewhere.

- Recent trends in many western forest regions of more large fires and more area burned are linked to recent climatic trends of hotter droughts and longer, more severe fire seasons.

- Respondents who emphasized the longer time scales of charcoal records noted that *most areas of predominantly low-severity fires showed some incidence of moderate- or high-severity fire* over longer time frames.

- It is desirable to use *multiple methods to reconstruct historical fire regimes.* More can be learned using multiple approaches and considering data from diverse temporal and spatial scales.

- Importance of local context in management of fire-prone landscapes underscores the need to move away from oversimplified narratives that encourage application of fire research beyond its original scope of inference. *Note: the scope of inference is of particular concern here as over reliance on fire scar sampling for landscape scale interpolation has inherent biases and uncertainties.*

Areas of Disagreement and High Uncertainty:

- Fire regime inferences from historical and modern tree inventory data, simulation models, and other approaches have *levels of uncertainty.*

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- Whether large, high-severity fires have increased to a significant and measurable degree in all forest types *in comparison to historical fire regimes* (i.e., prior to modern fire suppression) remains debatable.

- If f*uel treatments* are urgently needed across nearly all forests *remains debatable.*

- Fuel treatments should be focused around communities and plantations; but *hazard fuel reduction elsewhere remains debatable.*

- There is *high uncertainty* about where and when fuel treatments are beneficial.

- Commonly used vegetation classification schemes as a suitable basis for generalizing about fire regimes remains debatable. *Known geographic variation in fire regimes within forest types argues for improved forest and fire regime classifications.*

- *Tree-ring evidence sometimes supports conclusions that contrast with those derived from landscape-scale inventory and monitoring data using different sampling frames creates uncertainty.*

- General applicability of *"thinning and prescribed burning remedies"* to offset human influences is *debatable.*

- Human impacts on forest successional conditions in moist and cold forests remains debatable.

- Extent to which landscape tipping points have been reached as a result of high-severity fires is debatable.

- Effectiveness of *fuel treatments under projected climate futures and associated more extreme fire weather is uncertain.*

- Interpretation of any research evidence and the *scope of related inferences* is limited by scaling (uncertainty) and sampling concerns associated with the methods, and these limitations apply to all research methods.

- All methods for reconstructing historical fire regimes are necessarily indirect and

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have degrees of uncertainty. They may include, but are not limited to, interpreting evidence of past fires or the extent of fire-dependent ecosystems from historical documents, land surveys, aerial photographic reconstructions, *fire-scar and growth-release data* from tree rings, tree age and death dates from tree-ring data, climatic data linked with past fires, charcoal and pollen deposits, current characteristics of stands (i.e., structure, species, and stand age distribution), fire perimeter mapping, historical timber survey data, and use of statistical distributions for modeling stand-replacing fire.

In closing, the Forest Service cannot present to the public that they are using BASI or BAIS until the above issues are fully resolved, levels of uncertainty and risks to large trees and associated values are fully disclosed and properly avoided, and the publications of the wider scientific community are disclosed and addressed.

It is especially troubling that one of the principal supporters, Dr. Tom Spies, for removing large tree protections in Eastside Forests stated in his recent Oregonian Op-ed that fuel reduction is about brush and s*mall trees*, not large trees. His apparent contradiction of being for large tree removal in the GTR vs. then stating publicly the fuel reduction need is s*mall trees*, branches, and brush reflects uncertainty in the science underlining the EA. In fact, it argues for an alternative that prunes lower branches, girdles and leaves firs on site, and focuses on understory reductions – not the large tree overstory.

"In the dry forests of central, eastern and southwestern Oregon, we can reduce fuels by removing debris, branches, brush and small trees using machines and controlled burns. This lowers fire risk for up to 15 years."

-Tom Spies, Emeritus, USDA Forest Service Pacific Northwest Research Station [3]

[3] *https://www.oregonlive.com/opinion/2020/09/opinion-look-to-wildfires-history-to-better-prepare-for-next-one.html*

Conclusion

The EA and GTR are not based on BASI or BAIS principally because they have skipped over the vast majority of independent science that does not support the proposed action creating management uncertainties, high risks to forests from misguided actions, and conflict with the public and scientists. One-hundred and fifteen scientists recently demonstrated support for continued large tree protections in an open letter to the Forest Service and have requested that protections remain in place. Notably, public trust is earned and the agency's track record is poor on this account as summarized by Dr. DellaSala's presentation at the May 11, 2020 Science Forum:

- 1960 belief that LS/OG was "decadent" and needed to be "regenerated."
- 1990s "forest health" that insects and fire risks can be reduced by logging.
- 1990s belief that certain forms of logging "mimic" natural disturbance processes.
- 2000s postfire logging of late-successional reserves and roadless areas to "restore" old growth conditions.
- Widespread expansion of Categorical Exclusion authorities in NEPA for "active management."
- Trump's executive order for more logging and "active management."
- This is why standards are needed - for agency accountability and transparency.

Because nearly all of the independent science was ignored at the Science Forum and is not reflected in the EA or GTR, Dr. DellaSala's recommendations are included herein.

- Start by reaffirming the 21-inch rule as critical to ecological integrity and restoration of depleted large tree structures.
- Protect large tree cohorts for below-ground connectivity and ecosystem functionality.

Conclusion

- Develop a reserve network based on redundancy, connectivity, coarse/fine filter conservation approaches as in the Northwest Forest Plan (DellaSala et al. 2015, 2017).

- Protect ecologically important areas where fir is more suited and historically occurred in abundance.

- Identify and protect climate refugia (north and east facing slopes, ashy soils, gulches, elevational connectivity, riparian areas, unroaded/roadless areas, remaining LS/OG)

- Wildlife don't care if it's a fir or a pine - martens, bats, goshawks, woodpeckers use large trees and large firs that are all that is left in places.

- Narrow exception (based on historical evidence) for encroached firs in canopy drip line of old pines but only girdle, fell (not yard), and tip into streams for bull trout and steelhead.

- Reduce anthropogenic stressors especially cows and climate change as the biggest threats to ecological integrity on public lands (Beschta et al. 2012).

- Fire suppression and logging exacerbate fire intensity (Bradley et al. 2016).

- Roads contribute to ignitions, aquatic impacts, habitat fragmentation, invasives (Ibisch et al. 2017).

- Thinning alters stand dynamics (blow down, invasives, fire spread) and increases emissions (Dr. Law's presentation).

- Postfire logging disrupts natural successional processes (Lindenmayer et al. 2008).

- Use a historical baseline of trees >60 inches dbh with most stands 20-44 inches dbh (Henjum et al. 1994).

Conclusion

To underline the main points so it is clear in analysis of alternatives, as the Forest Service revises its Eastside forest plans, it needs to incorporate the following:

- A reserve network that protects all remaining late-seral forests and native communities following the conservation biology principles of the Northwest Forest Plan as a model (see DellaSala et al. 2015).
- Protection of large trees ≥21 inches dbh – all species – for their wildlife, carbon, and legacy values.
- Maintenance of diversity and ecological integrity via managed wildlife fire of ALL severities – not just low severity.
- Removal of livestock from native grasslands, oak woodlands, dry forests, and especially seeps and riparian areas with reintroduction of beavers to facilitate stream recovery and natural ecosystem processes.
- Choose an alternative that minimizes emissions and carbon losses from logging.

In closing, the Forest Service is overly focused on large tree removals and cannot see the forest for the ecosystem. There is new evidence that tree cohorts are connected by below-ground processes that regulate resistance to drought and climate susceptibility of conifers (Beck et al. 2020 – not cited or discussed in the EA). Ectomycorrhizal fungi connect tree cohorts that exchange nutrients and chemicals at the subsurface level and these connections are disrupted when trees within a cohort are logged (soil disturbance, Beck et al. 2020). The agency needs to take into consideration this new information and manage the forest for more than mainly fuels. The agencies' understanding of resilience is narrowly focused and out of step with the broader scientific community and importance of recovering large trees.

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