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ARTICLE

Vegetation Ecology

Big trees burning: Divergent wildfire effects on large trees in open- vs. closed-canopy forests

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Abstract

Wildfire activity has accelerated with climate change, sparking concerns about uncharacteristic impacts on mature and old-growth forests containing large trees. Recent assessments have documented fire-induced losses of large-tree habitats in the US Pacific Northwest, but key uncertainties remain regarding contemporary versus historical fire effects in different forest composition types, specific impacts on large trees within closed versus open canopies, and the role of fuel reduction treatments. Focusing on the 2021 Schneider Springs Fire, which encompassed 43,000 ha in the eastern Cascade Range of Washington and burned during a period of severe drought, this study addresses three interrelated questions: (1) Are burn severity distributions consistent with historical fire regimes in dry, moist, and cold forest types? (2) How does burn severity vary among forest structure classes, particularly large trees with open versus closed canopies? (3) How do fuel reduction treatments influence forest structure and burn severity inside and outside of treated areas? Within each forest type, burn severity proportions were similar to historical estimates, with lower overall severity in dry forests than in moist and cold forests. However, across all forest types combined, high-severity fire affected 30% (4500 ha) of large-tree locations with tree diameters >50 cm. In each forest type, burn severity was lower in locations with large-open structure (<50% canopy cover) than in locations with large-closed structure (>50% canopy cover). Burn severity also was lower inside than outside treated sites in all structure classes, and untreated large-closed forests tended to burn at lower severity closer to treatments. These results highlight the susceptibility of dense, late-successional forests to contemporary fires, even in events with widespread potentially beneficial effects consistent with historical fire regimes. These results also illustrate the effectiveness of treatments that shift large-closed to large-open structures and suggest that treatments may help mitigate fire effects in adjacent large-closed forests. Long-term monitoring and adaptive management will be essential for conserving critical wildlife habitats and fostering ecosystem resilience to climate change, wildfires, and other disturbances.

KEYWORDS
adaptive management, burn severity, forest composition and structure, fuel reduction treatments, large trees, late-successional habitat, mature and old-growth forests, wildfire

INTRODUCTION

As wildfire activity accelerates with climate change, land managers, policy makers, and numerous interest-holders are concerned about sustaining biodiversity and ecosystem services. Mature and old-growth forests—composed of large, old trees and structural conditions that develop over centuries—are vital habitats for threatened and endangered species, representing a critical "backbone" of ecosystem resilience in frequent-fire forests globally (Bendall et al., 2023; Gaines et al., 2022; Halofsky et al., 2024; Hessburg et al., 2015; Lindenmayer et al., 2012). Such large-tree forests are highly variable in composition and structure across forest productivity gradients, with open canopy conditions associated with relatively dry, slower-growing forests and closed conditions associated with relatively moist, productive systems. In western North America, these vital ecosystems have continued to decline over the past 30 years despite policies intended to protect and restore large-tree forest habitats (Ayars et al., 2023; Davis et al., 2011, 2022; Spies et al., 2018; van Mantgem et al., 2009). A recent analysis of threats to late-successional forests on US Federal lands found that "wildfire, exacerbated by climate change and fire exclusion, is the leading threat to mature and old-growth forests, followed by insects and disease. Tree cutting (any removal of trees) is currently a relatively minor threat despite having been a major disturbance historically" (USDA & USDOI, 2024). These concerns have catalyzed widespread investment in fuel treatments and other proactive management strategies to reduce fire risk to ecosystems and communities, particularly in drier forests where wildfire was frequent historically (Hood et al., 2022; Kreider et al., 2024; USDA Forest Service, 2022; WA DNR, 2020). However, key uncertainties remain regarding contemporary versus historical fire effects in different forest composition types, how impacts on large trees differ between closed versus open canopies, and the role of fuel reduction treatments in mitigating fire severity, particularly in large-tree forests. In this era of rapid environmental change, it is especially important to quantify and understand the effects of fires and treatments across different types of large-tree forests that evolved with differing historical fire frequencies and severities.

Historical fire regimes and associated proportions of low, moderate, and high severity (i.e., overstory tree mortality) represent critical reference conditions to evaluate and understand contemporary fire effects in different forest types (Chamberlain, Bartl-Geller, et al., 2024; Donato et al., 2023; Greenler et al., 2023; Haugo et al., 2019). In this context, recent assessments have defined the positive and negative "work of wildfire" as the degree to which fire effects are consistent with historical fire regimes and/or landscape resilience and fire risk reduction objectives (Churchill et al., 2022; WA DNR, 2022). For example, in the eastern Cascade Range of Washington State, relatively frequent, low-severity fires ignited by lightning and Indigenous stewardship were characteristic in relatively dry, low-elevation forests historically, while less frequent, mixed-severity fire effects typified moist, mixed-conifer forests at intermediate elevations, and relatively less frequent, moderate- to high-severity fire shaped colder forests at high elevations (Agee, 1993; Churchill et al., 2022; Hagmann et al., 2021; Hessburg et al., 2007; Reilly et al., 2018). Fire effects that match historical severity proportions and patch sizes are assumed to accomplish restorative, beneficial work (Donato et al., 2023; Haugo et al., 2019; WA DNR, 2022). In contrast, fire effects exceeding the historical proportions or patch sizes of high-severity fire are especially concerning for the conservation of large trees in dry forests (Fornwalt et al., 2016; Hessburg et al., 2007; Mallek et al., 2013), as even high-severity proportions that fall within historical ranges may include extensive large-tree mortality. Large wildfire events that span broad gradients of forest types and associated historical fire regimes present critical opportunities to distinguish the positive and negative work of wildfire and to quantify fire impacts in a variety of large-tree forests.

In addition to evaluating contemporary wildfire outcomes across forest composition types, it is important to quantify fire effects in different forest structure classes, particularly large, old trees in open- versus closed-canopy conditions. Increasingly, land managers are grappling with the complex goal of sustaining dense, closed-canopy forests with large and old trees while simultaneously reducing forest density and fuel connectivity to restore landscape resilience to fire, drought, and other disturbances (Halofsky et al., 2024). Although large-tree abundance has declined in recent decades (USDA & USDOI, 2024), many large trees (e.g., >50 cm diameter; Hessburg et al., 2020) have persisted in both closed-canopy (e.g., >50% cover) and open-canopy (e.g., <50% cover) settings. Over the past century, some large-tree, open-canopy forests (referred to as large-open hereafter) have densified to large-tree, closed-canopy (large-closed hereafter) structure conditions, increasing susceptibility to high-severity, stand-replacing fire, drought stress, and insect outbreaks (Bennett et al., 2023; Hagmann et al., 2021; Hessburg et al., 2000; Prichard et al., 2021). At the same time, large-closed structures provide essential late-successional habitat with varying "hang time" for threatened species, such as the northern spotted owl (NSO; *Strix occidentalis caurina*; Halofsky et al., 2024; Lehmkuhl et al., 2015), whereas large-open structures support indicator species like the white-headed woodpecker (WHWP; *Dryobates albolarvatus*). Parsing out these structure classes in different forest composition types is challenging, but recently established remote sensing approaches enable high-resolution mapping of forest structure—especially sites with large trees in open versus closed canopies—at broad spatial extents (Kane et al., 2023; WA DNR, 2024). Spatially explicit structure datasets and statistical models support comparisons of burn severity among forest structure classes while accounting for primary fire drivers, including forest type, topography, fire weather, and treatment history (Cansler et al., 2022; Chamberlain, Meigs, et al., 2024; Povak et al., 2020; Prichard et al., 2020). Because age is very difficult to ascertain through remotely sensed datasets and most old trees are also large, this study focuses on large-tree structure, recognizing that old trees have unique characteristics, functions, and values (Lindenmayer et al., 2012).

Treatments that reduce the risk of high-severity fire can change wildfire from a threat to large-tree populations to a positive feedback mechanism that helps sustain and cultivate them by consuming surface and ladder fuels and maintaining lower tree densities. Previous studies have shown that restoring large-open structure found in historical or contemporary frequent-fire forests with active fire regimes can increase the likelihood of non-stand-replacing wildfire, particularly when prescribed fire is included and under moderate fire weather conditions (Cansler et al., 2022; Chamberlain, Meigs, et al., 2024; Collins et al., 2023; Davis et al., 2024; Hood et al., 2024; Kalies & Kent, 2016; Shive et al., 2024; Waltz et al., 2014). Less well understood, however, is the role of treatments in reducing severity in the portions of landscapes where maintaining large-closed forest structure supports habitats, riparian functions, or other objectives (Lehmkuhl et al., 2015). Treatments in large-closed sites typically focus on noncommercial thinning or mastication of smaller understory trees and shrubs, followed by reduction of surface fuels through piling and burning and/or broadcast burning. Light to moderate commercial thinning is another strategy where some overstory trees are removed while maintaining higher canopy cover (e.g., >50%; Lehmkuhl et al., 2015). Low- and moderate-severity fire effects can achieve similar outcomes and result in both large-open and large-closed structures. In addition, treatments can have "shadow effects" where they reduce severity in adjacent, untreated forests (Hood et al., 2022, 2024; McKinney et al., 2022; Ott et al., 2023; Prichard et al., 2020). Thus, management guidelines recommend placing treatments around high-value, large-closed forests, such as nest sites (Lehmkuhl et al., 2015). Although fire simulation studies have investigated the effects of treatments on fire behavior in adjacent areas and thresholds beyond which a proportion of treated area affects landscape-level fire risk (Ager et al., 2021; Barros et al., 2019; Finney et al., 2007; Furniss et al., 2023; McKinney et al., 2022; Ott et al., 2023), relatively little empirical evidence for this shadow effect exists, particularly for adjacent, untreated large-closed forest. To sustain large trees in both open- and closed-canopy forests, managers need robust quantification of the effectiveness of different treatment types at reducing burn severity both within and adjacent to treated areas.

This study investigated wildfire impacts on large trees across the 43,000-ha Schneider Springs Fire, which encompassed a gradient of forest types, historical fire regimes, forest structure classes, and pre-fire treatments in the eastern Cascades of Washington State, USA. Schneider Springs was the largest forest fire in Washington in 2021, a major fire year with widespread, severe drought. Much of the pre-fire landscape was large-tree forest in both closed and open structure classes, and extensive treatments had been implemented recently within the fire footprint. Our study builds on a complementary analysis of treatment effectiveness and drivers of burn severity across the Schneider Springs Fire (Chamberlain, Meigs, et al., 2024) by accounting for forest type, forest structure (especially large-closed and large-open classes), and distance to treatment edge.

Our specific research questions were:

1. Are burn severity distributions consistent with historical fire regimes in dry, moist, and cold forest types?
2. How does burn severity vary among forest structure classes, particularly large trees with open versus closed canopies?

3. How do fuel reduction treatments influence forest structure and burn severity inside and outside of treated areas?

Collectively, these questions address the susceptibility of forest structures (and large trees in particular) to contemporary wildfires and the influence of treatments on burn severity both within and beyond treated areas. This novel analysis also provides a holistic way to integrate historical reference conditions, forest composition and structure, and burn severity to assess the positive and negative work of wildfire and associated landscape resilience. The scope of this study is mixed-conifer forests of the interior Pacific Northwest, especially drier forests where treatments have been implemented more extensively. Our empirical approach is scalable to other fire events and ecoregions and could support ongoing and future efforts to conserve mature and old-growth forests across the western United States.

METHODS

Overview

We leveraged multiple spatial datasets and statistical tools to address our three interrelated research questions. We mapped forest types based on potential vegetation type (PVT), burn severity with satellite-based change detection, pre-fire forest structure with high-resolution digital aerial photogrammetry (DAP), and pre-fire treatments from landowner spatial data and records. We estimated forest type-specific historical fire regimes from the literature (Question 1). We compared severity among structure classes using generalized additive models (GAMs) to control for covarying predictors (Question 2). Finally, we assessed the association among treatments, forest structure, and burn severity using geospatial overlays and GAMs (Question 3).

Study area

The 2021 Schneider Springs Fire burned ~43,000 ha, encompassing a broad range of forest types, historical fire regimes, forest structure classes, and pre-fire treatments that are emblematic of the eastern Cascades of Washington (Figure 1). Elevations within the fire ranged from 600 to 2300 m, spanning warm, dry mixed-conifer forests and woodlands at lower elevations and southerly aspects, moist mixed-conifer forests at intermediate elevations and aspects, and cold forests at higher elevations and northerly aspects (species described below). Local forest composition, structure, and associated historical fire regimes vary with climate, topography, edaphic conditions, and disturbance and land use history (Agee, 2003; Franklin & Dyrness, 1973; Hessburg et al., 2000; Meigs et al., 2023).

Precipitation and temperature vary across the study area, but the overall climate is defined by relatively warm, dry summers and cold winters, with the majority of precipitation occurring as snow, resulting in seasonally dry conditions that are conducive to periodic natural disturbances (Franklin & Dyrness, 1973; Meigs et al., 2023). The Schneider Springs Fire was ignited by lightning and burned from August 4 to October 8, 2021, following a period of extraordinary drought in the eastern Cascades (Bumbaco et al., 2022; Office of the Washington State Climatologist, 2021). Like many large wildfires, Schneider Springs exhibited several large progression days driven by extreme weather, although large portions of Schneider Springs also burned under cooler and wetter conditions (e.g., much of the fire occurred under moderate relative humidity [RH] of 20%–40%; Chamberlain, Meigs, et al., 2024).

Land management objectives and historical fire regimes vary with ownership and forest type across the study area. Federal land that is managed for multiple uses by the USDA Forest Service is the most widespread ownership type (89% of fire footprint—mostly in late-successional reserve or wilderness areas at intermediate and high elevations; Figure 1). Washington State DNR lands managed for multiple uses but prioritizing timber revenue for state trust beneficiaries occur at lower elevations, primarily in dry forests (11% of fire footprint). Prior to Euro-American colonization, lightning-ignited fires and fires ignited by Indigenous burning practices were common in east Cascades landscapes (Haugo et al., 2019; Hessburg et al., 2000; Lake et al., 2017; Reilly et al., 2018; Turner et al., 2011). In dry forests at lower elevations, relatively frequent (5–25 years), low-severity fires predominated (Agee, 2003; Everett et al., 2000), while in moist mixed-conifer forests at intermediate elevations, less frequent (25–80 years), mixed-severity fires were more typical (Churchill et al., 2022; Hessburg et al., 2007). In cold forests at higher elevations, relatively infrequent (60–250 years), large patches of high-severity fire were more common (Agee, 1993; Donato et al., 2023; Hessburg et al., 2007), but low- and moderate-severity effects also played a role due in part to Indigenous burning, which supported the development of local- to landscape-scale patches of forest and non-forest (Charnley, 2008; Turner et al., 2011). Catalyzed by climatic warming and increasing forest density following decades of large-tree harvest, dense reforestation, fire exclusion, and increasing fire deficits (Donato et al., 2023), tree mortality

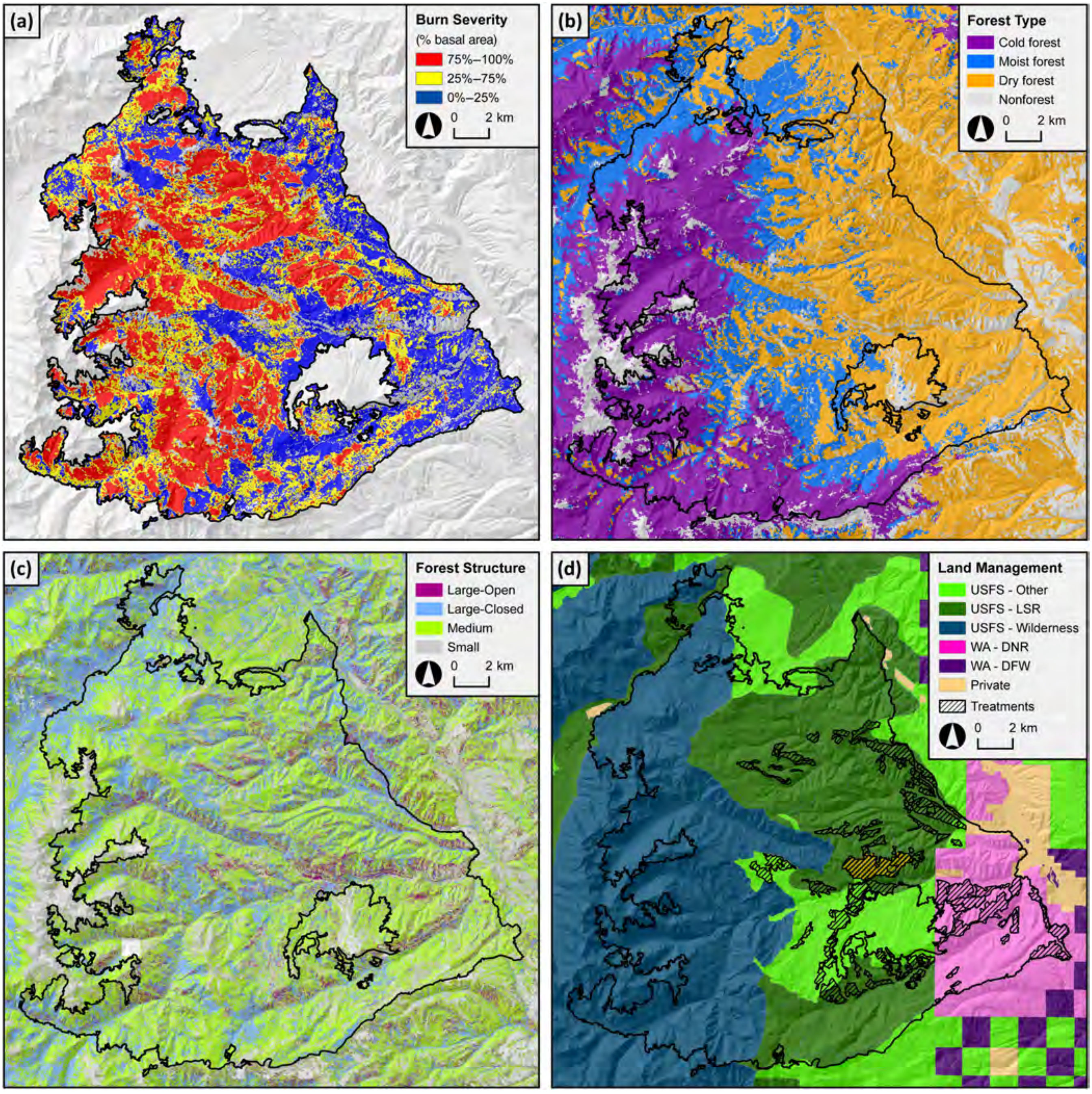

FIGURE 1 Schneider Springs (a) burn severity, (b) forest type, (c) pre-fire forest structure, and (d) land management. RdNBR burn severity classes are based on Reilly et al. (2017). Forest types are based on ILAP potential vegetation types (Halofsky et al., 2014), which are shown in Appendix S1: Table S1 and Figure S1. Study area location in central Washington State is shown in Appendix S1: Figure S1. Forest structure is based on digital aerial photogrammetry estimates of canopy cover and quadratic mean diameter (QMD) (WA DNR, 2024) in four classes: small (<25 cm QMD), medium (25–50 cm QMD), large-open (>50 cm QMD and <50% canopy cover), and large-closed (>50 cm QMD and >50% canopy cover) (see *Methods* for details on classes). Treatment locations are based on federal and state geospatial records (see *Methods*), and the only wildfire treatment (Meeks Table Fire) is shown with orange cross-hatching. USFS–LSR, USDA Forest Service Late-Successional Reserve; WA–DFW, Washington State Department of Fish and Wildlife; WA–DNR, Washington State Department of Natural Resources.

has increased in recent years due to wildfire, drought, and insect outbreaks (Bennett et al., 2023; Meigs et al., 2015, 2023; Reilly et al., 2017). Given the widespread extent of similar geographic conditions, these forests and their vulnerability to wildfire and other disturbances are broadly representative of seasonally dry-forest landscapes throughout western North America (Hessburg et al., 2019).

Burn severity

We mapped burn severity with Landsat satellite-based change detection, classifying low-, moderate-, and high-severity fire based on field inventories of tree mortality. As detailed by Chamberlain, Meigs, et al. (2024), we computed the Relative Differenced Normalized Burn Ratio (RdNBR) (Miller & Thode, 2007) using Google Earth Engine code described in Parks et al. (2018, 2021) to produce $RdNBR_{offset}$ (hereafter RdNBR) maps at 30-m resolution. RdNBR captures the relative fire-induced change in dominant vegetation and is appropriate for assessing fire effects across burned areas spanning heterogeneous pre-fire conditions (Cansler & McKenzie, 2014; Meigs & Krawchuk, 2018; Miller & Thode, 2007). Because treatments had been implemented within the fire footprint in 2020, we produced pre-fire mean imagery composites from June 1 to August 1, 2021, and post-fire mean composites from June 1 to August 1, 2022. We removed RdNBR values in the top and bottom 0.001 percentile of values to account for extreme outliers. We used continuous RdNBR as our primary response variable in statistical modeling (described below). To aid interpretation and for comparisons with historical fire regime estimates, we also classified RdNBR into low-, moderate-, and high-severity categories corresponding to 0%–25%, 26%–75%, and 76%–100% basal area mortality, respectively, using a regionally derived regression model (Meigs et al., 2020; Reilly et al., 2017).

Forest type

We evaluated burn severity in dry, moist, and cold forest types, all of which fall within the relatively dry East Cascades physiographic province (Table 1; Spies et al., 2018). We mapped these forest types based on PVT data from the Integrated Landscape Assessment Project (Halofsky et al., 2014), using the updated version from Henderson (WA DNR, 2020) and masking non-forest vegetation from the burn severity analyses. Each PVT represents the dominant forested vegetation present following succession and absent disturbance (Donato et al., 2023), and we combined PVTs into three general forest types to facilitate comparisons (Table 1, Figure 1; Appendix S1: Figure S1). First, dry forests include the ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & C. Lawson) and dry mixed-conifer PVTs, dominated by ponderosa pine, Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), as well as lesser amounts of western larch (*Larix occidentalis* Nutt.), grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), and lodgepole pine (*Pinus contorta* Douglas ex Loudon). Second, moist forests include the moist mixed-conifer PVT, with dominant species being Douglas-fir, grand fir, and western larch. Finally, cold forests include the Pacific silver fir (*Abies amabilis* Douglas ex J. Forbes), mountain hemlock (*Tsuga mertensiana* (Bong.) Carrière), and subalpine parkland PVTs, with additional dominant species being subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) and lodgepole pine. Broadleaf species, including quaking aspen (*Populus tremuloides* Michx.), black cottonwood (*Populus balsamifera* L. ssp. *trichocarpa* (Torr. & A. Gray ex Hook.) Brayshaw), Scouler's willow (*Salix scouleriana* Barratt ex Hook.), and red alder (*Alnus rubra* Bong.), are present in dry, moist, and some cold forest locations and provide important habitats and other functions. Across the Schneider Springs Fire, dry, moist, and cold forests represent 39%, 21%, and 31%, respectively (Table 1).

Forest structure

We mapped pre-fire forest structure using DAP data collected during summer and fall 2019 as part of the National Agricultural Imagery Program (NAIP) (Strunk et al., 2019). Images were processed as point clouds with the BAE Socet 175 GXP Auto Spatial Modeler (ASM) module (Walker & Pietrzak, 2015) using a customized configuration file (Strunk et al., 2019), the FUSION program (McGaughey, 2009), and vendor-supplied ground models derived from prior lidar acquisitions. All available FUSION grid metrics were calculated at 66-foot resolution (~20 m), using only points at least 6 feet (~1.8 m) above the ground.

To create DAP-based structure classes for this study, we created four forest structural classes based on canopy cover and quadratic mean diameter (QMD) of the top 25th percentile of trees by height (WA DNR, 2024): small (<25 cm QMD), medium (25–50 cm QMD), large-open (>50 cm QMD and <50% canopy cover), and large-closed (>50 cm QMD and >50% canopy cover). These structure classes are directly applicable to habitat assessments of closed versus open canopy (Halofsky et al., 2024) and large trees corresponding to the "21-inch rule" (53.3 cm) on National Forests (Hessburg et al., 2020). To create these classes, we used two separate modeled products representing tree size class and canopy cover. Both size class and canopy cover were modeled with random forest models considering a variety of environmental variables and all available DAP metrics (see appendix B in WA DNR (2024)). We applied several post-processing steps to

TABLE 1 Burn severity extent (area and percentage) of structure classes and forest types across the Schneider Springs Fire.

Attribute	Burn severity extent						Total area (ha)
	Low		Moderate		High		
	Area (ha)	Percentage	Area (ha)	Percentage	Area (ha)	Percentage	
Forest type							
Dry	7895	47	5315	32	3649	22	16,859
Moist	2802	31	3125	35	3033	34	8961
Cold	3465	26	4398	33	5395	41	13,258
Non-forest	2570	70	905	25	176	5	3652
Structure class							
Small	3784	49	2334	30	1574	20	7692
Medium	7442	37	6486	32	6193	31	20,121
Large-Closed	4026	33	3958	33	4116	34	12,100
Large-Open	1480	53	966	34	371	13	2817
Forest type × structure class							
Dry							
Small	1422	54	789	30	438	17	2648
Medium	3528	45	2485	32	1757	23	7770
Large-Closed	2083	42	1575	32	1315	26	4973
Large-Open	862	59	466	32	139	9	1467
Moist							
Small	600	40	477	32	415	28	1491
Medium	1204	30	1407	34	1468	36	4079
Large-Closed	845	28	1075	36	1066	36	2986
Large-Open	154	38	166	41	84	21	404
Cold							
Small	664	32	723	35	665	32	2052
Medium	1770	26	2210	32	2880	42	6860
Large-Closed	943	24	1245	32	1715	44	3903
Large-Open	89	20	220	50	135	30	443
Grand total	16,733	39	13,743	32	12,253	29	42,729

Note: Appendix S1: Table S1 presents severity extent across potential vegetation types, pre-fire old-growth structure index classes, and pre-fire northern spotted owl habitat. Low-, moderate-, and high-severity classes correspond to basal area mortality values of <25%, 25%–75%, and >75%, and RdNBR values of <235, 235–649, and >649, respectively, based on a regionally derived regression model (Reilly et al., 2017).

fix errors in the DAP data, which are described further in Appendix S2 and WA DNR (2024).

To complement the DAP-based structure comparison, we also assessed burn severity distributions across two other pre-fire structure maps with direct applications to late-successional forest conservation (Appendix S1). First, we used a 2019 map of old-growth structural index (OGSI) based on gradient nearest neighbor (GNN) imputation (Ohmann et al., 2012) designed to monitor mature (OGSI-80) and old-growth (OGSI-200) forests across the Northwest Forest Plan region (Davis et al., 2022). Second, we used a 2017 map of NSO habitat suitability that is based on lidar imagery (Halofsky et al., 2024). These maps provide fine- to intermediate-resolution insights into pre-fire late-successional forests with closed canopies, complementing the DAP-based maps of closed- and open-canopy forest.

Treatments and past wildfires

Obtaining, organizing, and quality-checking treatment information is a time-intensive process, especially when assessing treatment effects across multiple land ownerships. We utilized the same treatment history database described in detail by Chamberlain, Meigs, et al. (2024). Briefly, we acquired USDA Forest Service treatment data from the Forest Activity Tracker System and Washington State land data from the Washington DNR. We included past wildfire perimeters from the Monitoring Trends in Burn Severity dataset (one fire in this case; Figure 1d) and did not include treatments on private lands (<1% of the fire footprint). We grouped treatments into five major categories: thinning (Thinning; 1241 ha); thinning followed by surface fuel treatments such as piling of fuels, pile burning, and rearrangement of fuels that do not include broadcast burning (Thinning + FuelTx; 1691 ha); prescribed broadcast burning only (RxBurn; 141 ha); thinning followed by prescribed burning (Thinning + RxBurn; 293 ha); and past wildfire (WxBurn; 333 ha). We note that the thinning categories include both more intensive commercial treatments and noncommercial thinning treatments that focus on smaller trees that contribute to ladder fuels. We excluded regeneration harvest (76 ha) due to low total extent, as well as FuelTx-only (164 ha) treatments based on local manager observations that these were likely erroneous if they did not coincide with thinning treatments (Chamberlain, Meigs, et al., 2024). The only past wildfire we included as a treatment was the 2015 Meeks Table Fire, which burned primarily at low and moderate severity, since other past wildfires burned almost entirely at higher elevations in different biophysical settings than sites with mechanical thinning and prescribed burning treatments (Figure 1). We also excluded narrow road treatments (i.e., long slivers along roads with approximate widths <30 m).

After classifying all treatments into the five broad categories, we assigned dates to individual treatments using date-related field names in each database, working with local managers to verify dates and treatment types where data were missing or appeared inaccurate (J. Campbell, personal communications). We included all treatments with implementation dates prior to 2021 (with all treatments occurring after the year 1998 except for one 14-ha treatment in 1985) and used pre-fire NAIP imagery to verify and adjust treatment boundaries. We also applied a 30-m inward buffer to account for minor inconsistencies in mapped treatment boundaries. After verifying dates for each treatment, we merged the treatment layers to produce a single layer representing unique treatment combinations (see Table 1 in Chamberlain, Meigs, et al., 2024). Finally, we deleted any polygons with areas <5 ha to account for "slivers" created during the layer merging process and consolidated the total number of unique treatment categories by applying a standardized order: Thinning, FuelTx, RxBurn, and WxBurn.

Analysis

To evaluate burn severity distributions among dry, moist, and cold forest composition types in the context of forest type-specific historical fire regimes (Question 1), we summarized Schneider Springs severity and used LANDFIRE data as applied by WA DNR (2022). Specifically, we calculated ranges for historical fire severity (5th percentile, 50th percentile, and 95th percentile) for dry, moist, and cold forests using values from Haugo et al. (2019), which are based on LANDFIRE 2016 Biophysical Settings (www.landfirereview.org) and a refined simulation methodology from Blankenship et al. (2015). We crosswalked these values to PVTs and calculated weighted averages for the historical ranges using the area of each PVT for each forest type. We then compared the observed severity proportions for each fire by forested vegetation type with the historical ranges, excluding non-forest vegetation types (e.g., shrubland, grassland) from the analysis. Lastly, we computed the extent and relative abundance of each burn severity class in very large contiguous patches (>400 ha) to provide context for burn severity patterns among forest types.

To compare severity among forest structure classes with a specific focus on large-closed and large-open conditions (Question 2), we used geospatial overlays and GAMs, with GAMs specifically allowing us to control for covarying predictors. First, we evaluated burn severity distributions among forest types and structure classes, representing a full census of burn severity across the fire perimeter. We focused primarily on the DAP-based structure and also evaluated distributions among GNN-based OGSI and lidar-based spotted owl habitat classes. To aid interpretation of fire effects, we also graphed thresholds for low, moderate, and high severity based on Reilly et al. (2017).

We employed GAMs using the mgcv package (Wood, 2023) in R (R Core Team, 2023) to examine the effect of pre-fire forest structure classes on burn severity while controlling for key climate, fire weather, and topographic drivers. GAMs allow for fitting nonlinear relationships and assessing the magnitude and error of coefficients (Tortorelli et al., 2024; Wood, 2023). We produced modeling datasets by randomly sampling the Schneider Springs Fire using a 270-m sampling grid (centered on the RdNBR pixels) to reduce the effects of spatial autocorrelation (Kane et al., 2015). We then fit four GAMs: one for all forest types combined and one for each forest type. Specifically, we modeled RdNBR as a function of the categorical forest structure classes and three continuous predictors. We included the top climate, fire weather, and topographic drivers of severity identified by Chamberlain, Meigs, et al. (2024) for the Schneider Springs

Fire: long-term (1981–2010) climatic water deficit (CWD), day-of-burn RH, and topographic position index (TPI; measured using an 8010-m neighborhood). We fit forest structure classes as a linear term in each model. The three continuous predictors were smoothed using thin plate regression splines and a basis dimension of six. We included moderate-strength (gamma = 1.4) penalization terms in our models by setting select = "true," which allowed coefficients to shrink to zero or to be fit as linear terms (Marra & Wood, 2011; Tortorelli et al., 2024). Lastly, we used fitted models to predict RdNBR (and 95% CIs) for each forest structure class, while holding other covariates at their means.

To assess the influence of fuel reduction treatments on burn severity within and adjacent to treated areas (Question 3), we used geospatial overlays and GAMs. We evaluated treatment effects across all forest types (dry, cold, and moist), but we note that the majority of treatments occurred in the dry forest zone (Figure 1). First, we identified untreated reference areas as buffered regions 10–1000 m around each treated area (including all treatment categories). We then quantified the absolute and relative abundance of each forest structure class (small, medium, large-closed, and large-open) in each treatment category (Thinning, Thinning + FuelTx, Thinning + RxBurn, RxBurn, and WxBurn) and compared these estimates to untreated reference areas (randomly sampling untreated reference areas for 30,000 pixels so that the extent was similar to the total extent of the largest treatment category [Thinning + FuelTx]). Prior to modeling, we sampled treated and reference areas using a 270-m grid to reduce the effects of spatial autocorrelation on model errors. Then, we fit GAMs to (1) compare burn severity predictions inside and outside of treated areas for each structure class and (2) evaluate the influence of distance from treatment edge on burn severity for each forest structure class. In all models, we controlled for the same environmental covariates as used in Question 2. Specifically, for the first set of models, we modeled RdNBR as a function of treated versus untreated (a binary metric), CWD, RH, and TPI for each forest structure class. For the second set of models, we modeled RdNBR as a function of distance to treatment edge, CWD, RH, and TPI for each forest structure class. We smoothed all continuous predictors using thin plate regression splines and a basis dimension of six and the same model parameters as the GAMs for Question 2 (Tortorelli et al., 2024; Wood, 2023). Finally, we made predictions using the fitted models to show the relationship (and 95% CIs) between RdNBR and (1) treated versus untreated and (2) distance to treatment edge, while holding other covariates at their means.

RESULTS

Are burn severity distributions consistent with historical fire regimes in dry, moist, and cold forest types?

Within forest types, burn severity proportions were similar to historical estimates (Table 1, Figure 2). In dry forests, the extent of low-severity fire (47%) exceeded moderate-severity fire (32%), which in turn

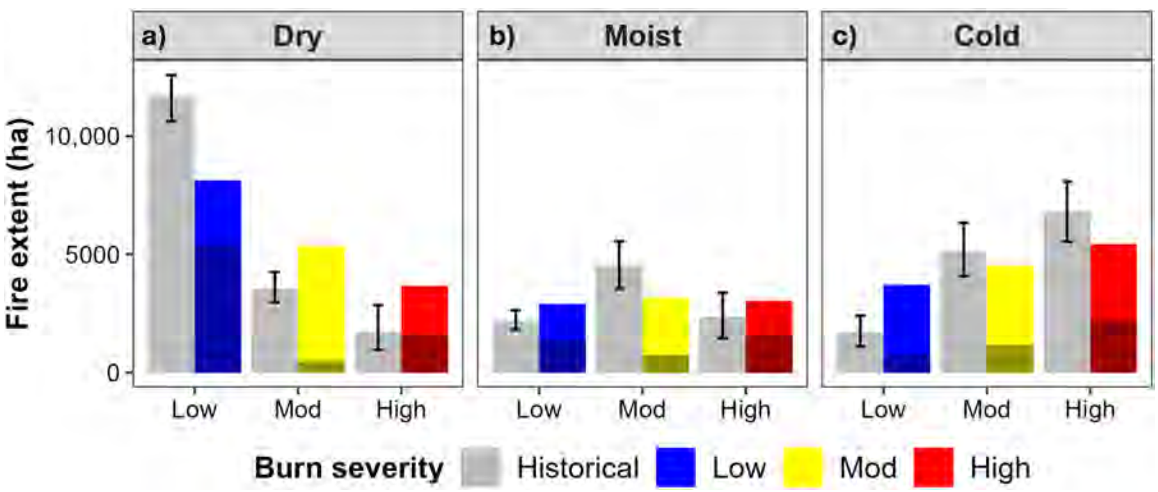

FIGURE 2 Burn severity across (a) dry, (b) moist, and (c) cold forest types of the Schneider Springs Fire and estimated historical distributions based on the 5th percentile, 50th percentile, and 95th percentile of severity using values from Haugo et al. (2019). Shaded portions of color bars indicate severity extent in very large patches (>400 ha) across the Schneider Springs Fire. Low-, moderate-, and high-severity classes correspond to basal area mortality values of <25%, 25%–75% and >75% and RdNBR values of <235, 235–649, and >649, respectively, based on a regionally derived regression model (Reilly et al., 2017).

exceeded high-severity fire (22%). Cold forests showed the opposite pattern, with high severity (41%) exceeding moderate (33%) and low (26%) severity (Table 1, Figure 2). In moist forests, severity classes were relatively evenly distributed (31%–35%), corresponding to less moderate severity than the historical estimate (Table 1, Figure 2). Despite the general similarity between the Schneider Springs Fire and historical estimates, the extent of moderate and high severity exceeded the historical range in dry forests (Figure 2). Moreover, the extent of moderate and high severity in all forest types has likely increased since the 2022 burn severity maps due to delayed tree mortality (Hood & Lutes, 2017).

Across the full extent of the Schneider Springs Fire, burn severity proportions were generally even, with low-, moderate-, and high-severity fire accounting for 39%, 32%, and 29% of forested areas, respectively (Table 1). High-severity effects were more abundant in western, higher elevation portions of the fire, where moist and cold forest types occur in more remote settings (i.e., late-successional reserve and wilderness areas; Figure 1). In comparison, low-severity fire was more abundant in eastern, lower elevation portions with more representation of dry forests and more active management (Figure 1). Accordingly, severity distributions increased from dry to moist to cold forest types (Appendix S1: Figure S2). Very large patches (>400 ha) represented a substantial proportion of high-severity effects in dry, moist, and cold forest types (45%, 53%, and 42%, respectively; Figure 2). Very large patches also were prevalent for low severity in dry and moist forests, and moderate severity exhibited the lowest abundance of very large patches (Figure 2).

How does burn severity vary among forest structure classes, particularly large trees with open versus closed canopies?

Although burn severity distributions overlapped among forest structure classes, there were important differences in both the overall distributions and median severity values among DAP-based structure classes in all three forest types (Figure 3). Across all forest types combined, large-closed sites had the highest median severity (RdNBR = 399), equivalent to the midrange of moderate severity (Appendix S1: Figure S2). In contrast, sites with large-open structure had the lowest median severity (RdNBR = 223), equivalent to the transition between low and moderate severity (Appendix S1: Figure S2). Within forest types, the same patterns were evident, with large-closed and large-open sites exhibiting a relatively consistent difference in burn severity (median RdNBR difference = 110, 124, and 102 in dry, moist, and cold forests, respectively; Figure 3, Table 1). Based on the imputed OGSI, median burn severity was generally higher in locations with old forests than in young and mature forests in dry forests (Appendix S1: Figure S3). Similarly, based on lidar, median burn severity was higher for moderate- and high-suitability NSO habitat than for low-suitability NSO habitat in dry forests (Appendix S1: Figure S3).

After accounting for key biophysical drivers with the GAM analysis, predicted burn severity varied significantly by structure class across all forest types combined and within forest types (Figure 4). Across all forest types combined, predicted severity was significantly lower in locations with large-open or small tree structure than in locations with large-closed or medium structure. These patterns were most pronounced in dry forests, while evident to a lesser degree in moist and cold forest types (Figure 4). Although differences in predicted severity between structure classes were significant for all forest types combined and individually, the magnitude of differences by severity class was relatively low, with all predicted values falling within the moderate severity class.

Collectively, high-severity fire affected 4500 ha (30%) of large-closed and large-open forests across all forest types (Table 1). In large-closed forests, burn severity classes were evenly distributed, with low-, moderate-, and high-severity fire each accounting for 33% (Table 1). In contrast, only 13% of large-open forests burned at high severity, with 34% burning at moderate severity and 53% at low severity (Table 1). High-severity fire was most prevalent in large-closed forests in each forest type, affecting 26%, 36%, and 44% of large-closed structure in dry, moist, and cold forests, respectively. In contrast, high severity was least prevalent in large-open forests, affecting 9%, 21%, and 30% of large-open structure in dry, moist, and cold forests (Table 1). Similarly, high-severity fire was widespread in old-growth forests (based on imputed OGSI) and high-quality NSO habitat (based on lidar), accounting for 36%–37% of these late-successional forest classes, while low severity was less abundant (28%–29%; Appendix S1: Table S1).

How do fuel reduction treatments influence forest structure and burn severity inside and outside of treated areas?

Across all forest types combined, treated areas included a variety of forest structure classes prior to fire (Figure 5). Untreated reference areas within 10–1000 m

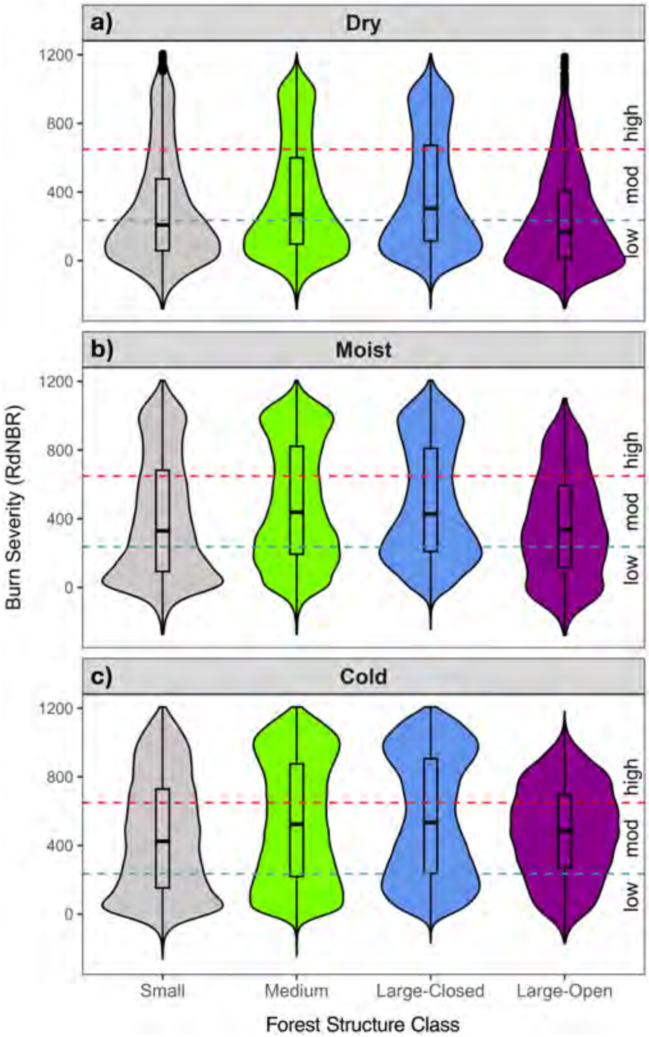

FIGURE 3 Observed burn severity across forest structure classes and forest types in the Schneider Springs Fire. The horizontal dashed lines correspond to thresholds between severity classes. See Figure 4 for predicted severity while accounting for primary fire drivers. See Appendix S1: Figure S2 for old-growth structural index and lidar-based northern spotted owl habitat suitability. Low-, moderate-, and high-severity classes correspond to basal area mortality values of <25%, 25%–75%, and >75%, and Relative differenced Normalized Burn Ratio (RdNBR) values of <235, 235–649, and >649, respectively, based on a regionally derived regression model (Reilly et al., 2017).

of treatments had the lowest relative abundance of large-open structure, whereas past wildfire as well as thinning plus prescribed burning had the highest relative abundance of large-open structure (Figure 5). Sites with thinning plus surface fuel treatments also exhibited relatively high abundance of large-open structure. All treatment categories exhibited lower severity distributions than untreated reference areas across all structure classes, with the lowest severity occurring in sites with prescribed fire (with and without thinning) or prior wildfire (Appendix S1: Figure S4).

For each structure class—including large-closed and large-open forests—burn severity distributions were consistently lower inside than outside of treated areas (10–1000 m; Figure 6). After accounting for key biophysical drivers with the GAM analysis, predicted burn severity was significantly lower inside than outside of treatments for the medium, large-closed, and large-open structure classes (Figure 6). For the large-closed structure class, burn severity was marginally lower at closer distances to treatments, increasing with distance from any treatment edge (Figure 6g). In contrast, the other structure classes exhibited similar severity irrespective of distance from treatment edge up to 1000 m (Figure 6). In large-open sites, predicted RdNBR remained at the threshold between low and moderate severity for up to approximately 1000 m from any treatment edge, reflecting the overall lower severity outcomes in this forest structure class (Figure 6).

DISCUSSION

Our findings highlight several key takeaways regarding the resistance and resilience of large-tree forests in an era of accelerating wildfire activity. First, burn severity distributions in dry, moist, and cold forest types were broadly consistent with historical fire regimes, reflecting both positive and negative work of wildfire; beneficial ecological outcomes include extensive wildfire treatment via low- and moderate-severity fire, while less desirable outcomes include high-severity fire affecting 30% of existing large-tree habitat across all forest types, with 92% of that occurring in large-closed forests. Second, burn severity was lower in locations with large-open tree structure than in locations with large-closed structure, particularly when accounting for primary biophysical and fire weather drivers. Third, burn severity was generally lower inside than outside treatment units, and severity in large-closed forests tended to be lower closer to treatments. Taken together, these results demonstrate that (1) mature and old-growth forests with large trees, especially with closed canopies, are vulnerable to large fires, even when overall landscape-scale fire effects are consistent with historical fire regimes and that (2) dry-forest landscapes with recent treatments are more likely to experience beneficial fire effects that sustain large-tree structures, including both open- and closed-canopy forests. Here, we interpret these results in the context of contemporary land management challenges.

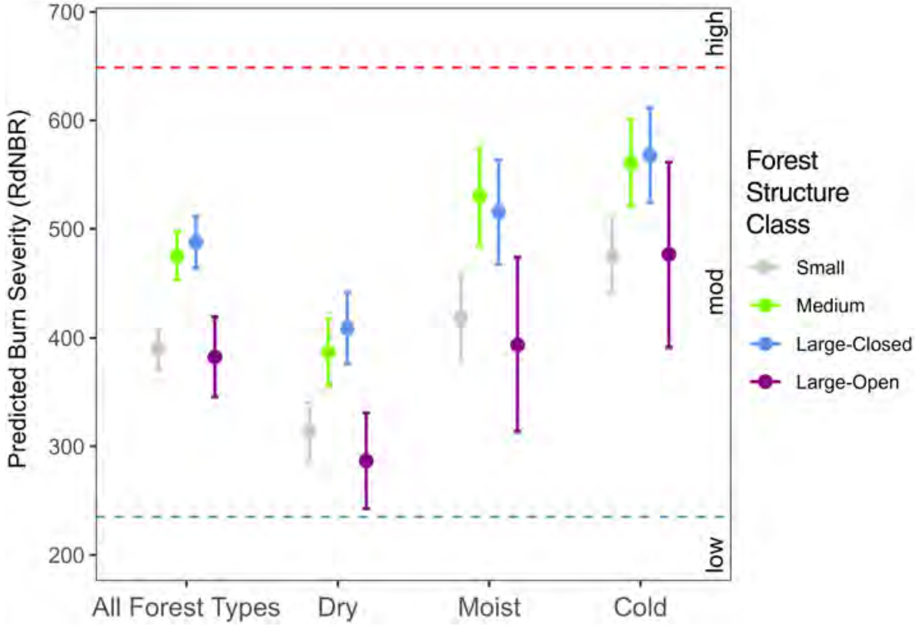

FIGURE 4 Predicted burn severity (mean and 95% CI) among pre-fire forest structure classes based on generalized additive models (GAMs) accounting for primary fire drivers. Separate GAMs were fit for all forest types combined and for each forest type. Severity (Relative differenced Normalized Burn Ratio [RdNBR]) was modeled as a function of forest structure class, climatic water deficit, relative humidity, and topographic position index. The latter three variables were held at their means here and were identified as the top drivers of burn severity in a fire-wide analysis by Chamberlain, Meigs, et al. (2024). See Figure 3 for observed severity distributions. Low-, moderate-, and high-severity classes correspond to basal area mortality values of <25%, 25%–75%, and >75% and RdNBR values of <235, 235–649, and >649, respectively, based on a regionally derived regression model (Reilly et al., 2017).

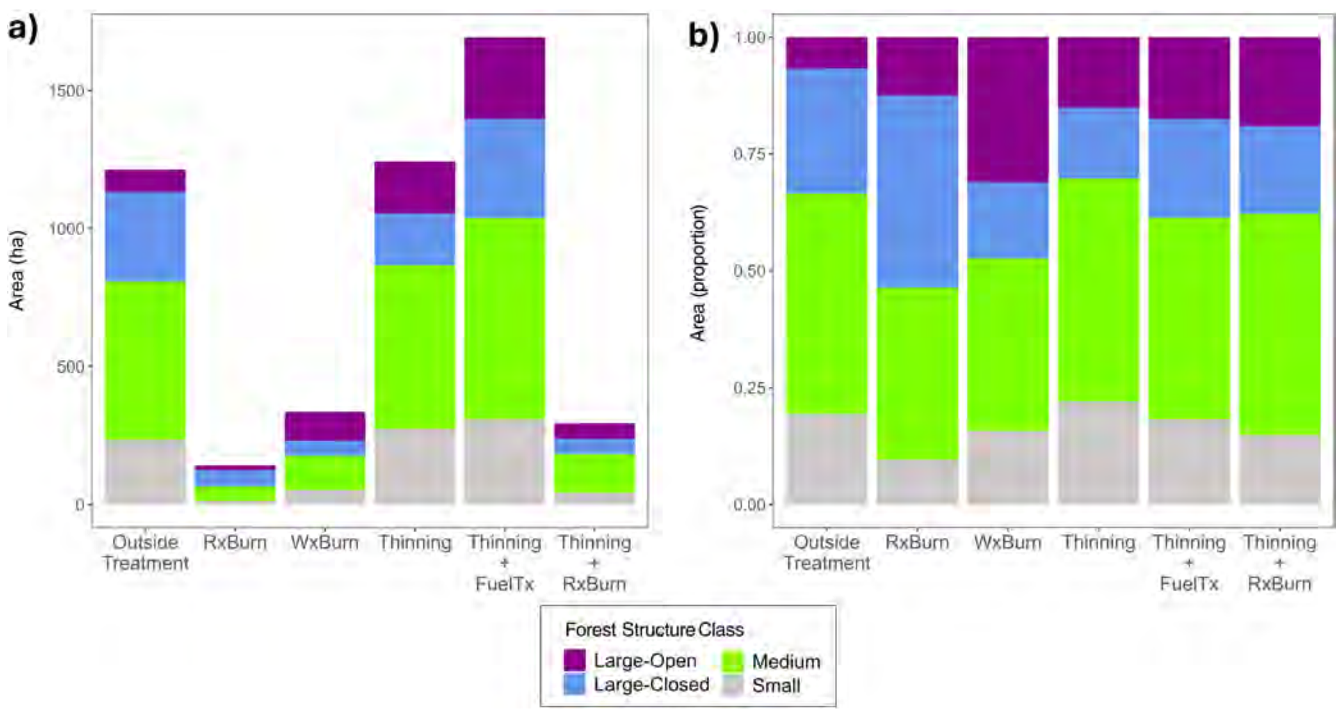

FIGURE 5 (a) Absolute and (b) relative abundance of pre-fire forest structure classes by pre-fire treatment categories. Outside treatment areas fall within a 10- to 1000-m buffer around treatment areas. Note that this figure does not stratify by forest type, but that the majority of treatments occurred in dry forests. The "WxBurn" category represents the 2015 Meeks Table Fire, which burned in dry forests in the eastern portion of the fire (Figure 1). RxBurn: prescribed broadcast burning. FuelTx: surface fuel treatment.

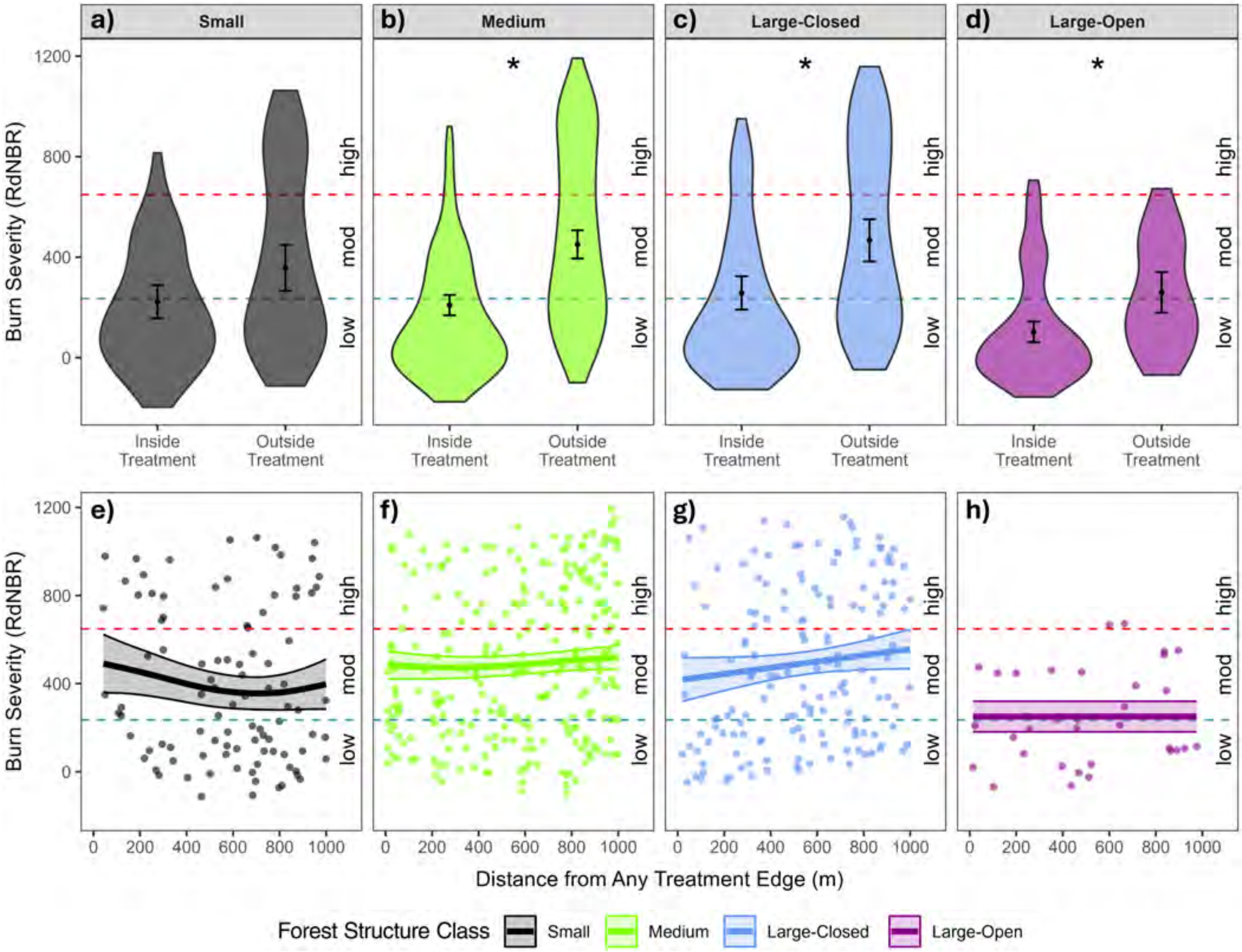

FIGURE 6 Burn severity inside treatments and outside in adjacent untreated areas (10- to 1000-m buffered region around treatments) across pre-fire forest structure classes. (a–d) Violin plots show the distribution of burn severity between treated (including all treatment types) and adjacent untreated areas for each structure class. Points within violin plots show model-predicted means (from generalized additive models) while holding climatic water deficit, relative humidity, and topographic position index at their respective means. Error bars show 95% CIs around predicted means. Asterisks (*) indicate nonoverlapping CIs. (e–h) Scatterplots show severity (Relative differenced Normalized Burn Ratio [RdNBR]) as a function of distance to any treatment edge (including all treatment types and all structure classes within treatments) for each structure class. Lines show predictions and 95% CIs for RdNBR as a function of distance to any treatment edge for each structure class based on generalized additive models. Predictions hold climatic water deficit, relative humidity, and topographic position index at their means. These were the top drivers of burn severity in a fire-wide analysis by Chamberlain, Meigs, et al. (2024). Low-, moderate-, and high-severity classes correspond to basal area mortality values of <25%, 25%–75%, and >75% and RdNBR values of <235, 235–649, and >649, respectively, based on a regionally derived regression model (Reilly et al., 2017). Note that this figure does not stratify by forest type but that the majority of treatments occurred in dry forests.

Contemporary work of wildfire in the context of historical fire regimes

Our findings that overall burn severity proportions were similar to historical estimates among forest types—yet also included extensive high-severity impacts on large trees—show how contemporary fires can have mixed outcomes for conservation and landscape resilience objectives. Despite being the largest fire in Washington State in 2021 (43,000 ha), a major fire year defined by severe drought (Chamberlain, Meigs, et al., 2024; WA DNR, 2022), the distribution of burn severity classes generally tracked historical fire regimes in dry, moist, and cold forest types (Figure 2). Moreover, low (26%–47%) and moderate (32%–35%) severity accounted for the majority of fire extent in dry, moist, and cold forest types (Table 1). Thus, across most of the Schneider Springs Fire, non-stand-replacing fire likely reduced surface fuels and bolstered the fire resistance of late-successional forests both inside and outside of fire refugia locations (Meigs et al., 2020). However, subsequent wildfires and follow-up treatments will likely be necessary to further reduce surface fuels, restore forest structure, and mitigate future fire risk (Chamberlain, Bartl-Geller, et al., 2024; Churchill et al., 2022; Greenler et al., 2023; North et al., 2021). Increasing fire extent in the East Cascades has begun to address the vast fire deficit that has accumulated following decades of fire exclusion (Donato et al., 2023), representing positive work of wildfire at the ecoregional scale.

At the same time, high-severity fire occurred in several very large patches (>400 ha) in all three forest types, especially in higher elevation, remote, and wilderness settings (Figures 1 and 2). Although some large high-severity patches are part of all fire regimes, high-severity patches are considered to have been smaller and more numerous historically, particularly in dry mixed-conifer forest types (Fornwalt et al., 2016; Hessburg et al., 2007; Mallek et al., 2013; Stephens et al., 2015). Additionally, high-severity fire affected 26%–44% (depending on forest type) of the large-closed forest structure that was present before the fire across forest types (Table 1), demonstrating that large fires can have negative effects on closed-canopy mature and old-growth forests. Thus, even when the proportion of high-severity fire is consistent with historical fire regimes, large patches of high severity that include large-closed forests can result in substantial loss of mature and old-growth structure that will take centuries to return (Bendall et al., 2023; Halofsky et al., 2024; Hessburg et al., 2015). In sum, although large fires like Schneider Springs can appear to be consistent with historical reference conditions, finer resolution analyses within structure classes reveal the degree to which they can substantially affect large trees and the critical habitats they provide.

Large tree sustainability in forests with open and closed canopies

We found that large-closed structures had the highest overall burn severity, which was significantly higher than severity in sites with large-open structures when accounting for covarying biophysical drivers (Figure 4). This divergence of fire effects on large trees in sites with open- versus closed-canopy structure is consistent with prior studies of fire effects in dry-forest landscapes where restoration treatments have been implemented (Hagmann et al., 2021; Prichard et al., 2020). The consistently lower severity in large-open structures in all forest types (Figure 4) demonstrates the fire resistance of this structure class, which is a primary objective of restoration strategies in western North America (e.g., USDA Forest Service, 2022; WA DNR, 2020). At the same time, the relatively higher severity in large-closed structures confirms that dense, late-successional forests are vulnerable to increasing wildfire activity in the Pacific Northwest region (Davis et al., 2022; Halofsky et al., 2024). Our findings of the highest overall burn severity in dry forests with old-growth structures (based on imputed OGSI) or high-suitability NSO habitat (based on lidar) further underscore this vulnerability.

This study also provides evidence that landscapes with extensive areas of large-closed forest are susceptible to large patches of high-severity fire. High-severity fire affected 24% and 34% of large-closed structure in dry and moist forests, respectively (~2400 total ha; Table 1), transforming large-tree to small-tree and non-forest structure for the next several decades and forgoing the opportunity to develop large-open structure that is more resistant to fire and other disturbances. Similarly, 37% of old-growth forest (OGSI-200) and 36% of high-suitability NSO habitat experienced high-severity fire, shifting them to young forest and low-suitability NSO habitat for the foreseeable future (Appendix S1: Table S1). Before the Schneider Springs Fire, 19.5% of the fire footprint was high-suitability NSO habitat, which is within the historical range (18%–24%) across dry and moist mixed-conifer forests of the eastern Cascades (Halofsky et al., 2024). Subtracting the areas affected by high-severity fire, the post-fire landscape was transformed such that only 12.5% supports high-suitability NSO habitat, and this estimate does not account for potential further effects of delayed tree mortality (Hood & Lutes, 2017; Reilly et al., 2023). As more fire events like Schneider Springs accumulate across the landscape, NSO habitat and other late-successional forests are vulnerable to further decline (Davis et al., 2022). As evidenced by this study, a single fire can have pervasive effects on large trees and late-successional habitats, and numerous large fires have occurred in the East Cascades in recent decades, resulting in widespread cumulative changes to large-open and large-closed forests (Reilly et al., 2018; WA DNR, 2022). Managers are thus left with the difficult decision of (1) retaining late-successional forests with a focus on suppression (a climate adaptation resistance strategy) while recognizing that future wildfires may be more severe, (2) reducing existing habitat through treatments to increase the likelihood of non-stand-replacing fire in the future (a climate resilience strategy), or (3) balancing these two approaches across a landscape (Halofsky et al., 2024).

Our findings underscore that restoring the balance of large-open and large-closed structure is key to the many values that large, old trees provide, including resistance to fire and drought, long-term sustainability of habitats for large-tree dependent species (e.g., NSO and WHWP), genetic diversity of old-growth trees, and other ecosystem services (Ayars et al., 2023; Bendall et al., 2023; DeWald & Kolanoski, 2017; Gaines et al., 2022; Lindenmayer et al., 2012). Before the Schneider Springs Fire, closed-canopy conditions represented 81% of large-tree areas (Table 1), setting the stage for the extensive losses of large trees and NSO habitat described above. However, although burn severity tended to be higher in large-closed than in large-open conditions, the range of predicted severity within each structure class and forest type was relatively narrow and fell within the moderate-severity category (Figure 4). This result likely stems from holding the bioclimatic and fire weather variables at their means in the GAM analysis, highlighting the strong influence of these variables on burn severity (Chamberlain, Meigs, et al., 2024). Thus, although large-open structures are less susceptible to high-severity fire, other factors (e.g., extreme fire weather) can override the influence of canopy structure on burn severity, increasing fire risk to all structure classes. In addition, the relatively low severity in locations with small structure further underscores the need for future research on the interactions among structure and other fire drivers, especially because small-tree locations were evenly mixed between open (53%) and closed (47%) conditions with differing fire sensitivity (Appendix S1: Table S2). Nevertheless, areas where surface fuels have been consumed by recent prescribed fires or wildfires have been shown to influence fire behavior and severity even during extreme fire weather and in locations with small trees (Cansler et al., 2022; Chamberlain, Bartl-Geller, et al., 2024; Chamberlain, Meigs, et al., 2024; Collins et al., 2023; Davis et al., 2024; Lyons-Tinsley & Peterson, 2012; Prichard et al., 2020; Shive et al., 2024; Taylor et al., 2022).

Treatment effects on burn severity in large-open and large-closed forests

In terms of treatment effects and effectiveness, our results support strategies to increase the pace of dry-forest treatments designed to restore historical structure and composition, including large, old trees. We found that locations with thinning plus prescribed burning, or past wildfire, had the highest relative abundance of pre-fire large-open structure (Figure 5), that burn severity was generally lower inside than outside treatment units (Figure 6), and that severity in large-closed forests tended to be lower closer to treatments (Figure 6). Our study shows that treatments that create and maintain large-open structures can be effective at mitigating burn severity, especially in dry forest types and even following periods of severe drought. These findings support the growing consensus that thinning combined with prescribed burning typically results in lower burn severity than untreated controls and less intensive treatments (Cansler et al., 2022; Chamberlain, Meigs, et al., 2024; Davis et al., 2024; Kalies & Kent, 2016; Shive et al., 2024; Taylor et al., 2022). Our study also highlights the beneficial effects of prior low- and moderate-severity fire, although the only wildfire treatment (Meeks Table Fire) was small relative to other treatments and the overall Schneider Springs Fire extent (Figures 1 and 5).

Interestingly, although locations with large-open structure had the lowest observed and predicted burn severity in all forest types (Figures 3 and 4), treatments were associated with lower severity in all structure classes (Figure 6). Large-open structures are considered to be more fire- and drought-resistant than large-closed structures (Chamberlain, Bartl-Geller, et al., 2024; Reilly et al., 2018), but our results indicate that treatments in large-closed forests that retain >50% canopy cover also can reduce burn severity. This result is consistent with studies showing that (1) burn-only treatments that treat surface and ladder fuels, (2) small-tree thinning combined with surface fuels treatments, and (3) light to moderate overstory thinning treatments combined with surface fuels treatments can all dampen fire effects (Chamberlain, Meigs, et al., 2024; Davis et al., 2024; Shive et al., 2024; Waltz et al., 2014). These results, along with our finding that treatments in any structure class can help protect adjacent areas with large-closed structure (Figure 6), illustrate that treatments can help to extend critical "hang time" for late-successional habitat (Furniss et al., 2023; Halofsky et al., 2024; Hessburg et al., 2015).

Collectively, these results highlight how treatments (a) in a variety of stand structure classes and (b) with a range of intensity and canopy cover retention in large-tree forests can provide multiple silvicultural options for land managers across high-priority landscapes. To be clear, however, large-closed forests are still more vulnerable than large-open forests to both wildfire and drought mortality (Stephens et al., 2015). Thus, sustaining large-tree populations and associated forests will require both silvicultural and wildfire treatments to restore extensive areas of large-open forest and surface fuel levels associated with historical reference conditions in dry, frequent-fire forests (Greenler et al., 2023; Hagmann et al., 2021; North et al., 2021; Reilly et al., 2018; Taylor et al., 2022). Additionally, our study indicates important variability both within and among treatments even though most of the treatments were in dry forests (ponderosa pine and dry mixed conifer). Specifically, treatment units within the Schneider Springs Fire included both open and closed structures, showing that treatments contain pockets of large-closed habitat and suggesting that landscape heterogeneity arises from the variety of treatment timing, type, and intensity (e.g., RxBurn, Thinning, and WxBurn; Figure 5).

Future research

Each of the datasets and analytical steps demonstrated here represent best estimates, but each also has uncertainties that set the stage for future research. First, remote sensing of severity can vary depending on imagery type, timing, and resolution (Howe et al., 2022), especially in open conditions with rapid regrowth of non-tree vegetation (e.g., lower severity estimates in small stands may be partially explained by less vegetation available to change, a factor that relativized burn indices [including RdNBR] are designed to address but may not fully resolve). Also, this study is a short-term assessment that should be coupled with long-term monitoring to track delayed mortality and regeneration trajectories (both multiyear remote sensing and field-based). Although low and moderate severity represented a strong majority of fire extent (71% total; Table 1), we note that these estimates do not account for delayed mortality (Hood & Lutes, 2017; Reilly et al., 2023). Future studies could build on this short-term spectral reflectance-based assessment to leverage post-fire field observations and high-resolution imagery (including DAP and lidar), enabling precise estimates of impacts on large trees and structural change relative to historical reference conditions. Future studies also could focus on multiple fire events spanning a broader range of fire weather to further elucidate treatment effectiveness at stand and landscape scales as well as threshold effects as fire weather becomes more extreme with climate change (Gaines et al., 2022).

Another key uncertainty is the historical estimates of fire frequency and severity by forest type. Our approach focuses on PVT and associated LANDFIRE biophysical settings, with historical fire regimes from the literature. However, the literature on fire regimes and forest types is not comprehensive, especially along steep environmental gradients like the Washington East Cascades. For example, the role of Indigenous fire stewardship in moist and cold forests has not been fully accounted for (Charnley, 2008; Lake et al., 2017; Turner et al., 2011), highlighting that fire may have been more frequent and with a broader variation of severities than the range presented in this analysis. Moreover, even where historical fire estimates are well supported by the fire history literature or reference sites, there is also important context for the historical prevalence of forest structure classes in different forest types (e.g., relative abundance of large-closed and large-open structures maintained by fire in relatively productive moist forests). Additionally, today's forests have more extensive, homogeneous, and lower quality late-successional, large-closed forest than historical conditions due to fire exclusion and high-grade logging (Davis et al., 2022; Halofsky et al., 2024). Future studies could determine how much and where high-severity fire in large-closed areas is restorative versus detrimental relative to historical landscapes and desired future conditions. Finally, because the majority of treatments were in relatively dry forest types of the East Cascades and previous studies have also focused on these forest types, future research should focus on treatment outcomes in moist mixed-conifer and other forest types not fully captured within the scope of this study.

Management implications and conclusions

In fire-prone landscapes of western North America, it is becoming increasingly challenging to restore large-open forests and sustain large-closed forests and associated habitats as large wildfires, droughts, and other disturbances accelerate with climate change (Ayars et al., 2023; Davis et al., 2022; Kreider et al., 2024; Spies et al., 2018; van Mantgem et al., 2009). Landscape-scale events like the Schneider Springs Fire have substantial positive and negative impacts on ecosystems and human communities, and they represent a powerful opportunity to evaluate impacts on large trees and implications for conservation strategies. The results of this study underscore the fire susceptibility of closed-canopy mature and old-growth forests with large trees and the role of forest health treatments within and adjacent to large-tree forests with both open and closed canopies. Several key conclusions and management implications emerged from our analyses:

1. Very large, contemporary wildfires can produce a range of burn severity impacts largely consistent with historical fire regimes across forest types, which can accomplish both positive and negative ecological work. This work of wildfire can include extensive treatment via low- and moderate-severity fire (especially in treated portions of dry forests) as well as substantial mortality of large, old trees via high-severity fire.
2. Large-open forest structure is more resistant to high-severity wildfire than large-closed structure.

Landscapes with a greater mix of open- versus closed-canopy large-tree structure are more likely to sustain large trees over time. Large, old trees have unique values (e.g., fire resistance, habitats, genetic diversity) that take centuries to develop.
3. Treatments can help restore the beneficial role of wildfire in sustaining large-tree populations in interior Pacific Northwest forests in several ways. Treatments that both restore large-open structure and reduce surface fuel loads are the most effective at reducing fire severity and sustaining large trees. Some treatments that maintain large-closed structure (canopy cover >50%) or include large-closed areas also can reduce severity. Finally, treatments in any structural condition can mitigate severity somewhat in adjacent, untreated large-closed forest.
4. Land managers have options to sustain and foster a resilient mosaic of open- and closed-canopy large-tree forests. In locations where large-closed habitats are a primary objective, light to moderate overstory thinning, mechanical surface and ladder fuel reduction, and prescribed fire are viable options. For high-priority, large-closed habitats (e.g., NSO nesting areas), managers can intersperse treatments that retain large trees and canopy cover and also apply variable treatments in adjacent areas. In other locations, more intensive thinning and surface fuel reduction or prescribed fire can restore large-open structure and habitat for associated wildlife species. To determine the appropriate mix and locations of different treatments, managers can utilize existing tools for landscape evaluation and decision support that combine target structural ranges from historical reference conditions, current forest conditions, projected future climate, fire risk, habitat, and other factors (Hessburg et al., 2015; WA DNR, 2022).

As forested landscapes become warmer and drier, strategies that recognize the geographic variability of mature and old forests with large trees will be increasingly important for habitat conservation and landscape resilience. In addition to beneficial effects, large fires can have substantial negative impacts on large, old trees, highlighting the urgency of proactive restoration to sustain these biological legacies and associated ecosystem services. Finally, long-term monitoring and adaptive management will continue to be essential for fostering ecosystem resilience to climate change, wildfires, and other disturbances.

AUTHOR CONTRIBUTIONS

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

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Data and code (Chamberlain et al., 2025) are available from Dryad: https://doi.org/10.5061/dryad.63xsj3vb9.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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