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Drought, wildfire and forest transformation: characterizing trailing edge forests in the eastern Cascade Range, Washington, USA

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Climate change and the compounding effects of drought and wildfire are catalyzing rapid ecosystem changes throughout the world. Relatively dry, trailing edge (TE) forests are especially vulnerable to ecological transformation when tree regeneration is moisture-limited following high-severity fire. Here, we illustrate the potential landscape-scale impacts of changing disturbance regimes by focusing on TE forests in the eastern Cascades of Washington, USA. Our specific objectives were to: (1) map TE forests based on climatic water deficit and forest cover; (2) characterize the composition, structure, and ownership of TE and non-TE forests; (3) quantify recent fire activity in TE and non-TE forests; (4) identify locations of potential forest loss where recent fires have burned severely in TE forests. Across the study area, TE forests encompassed 387 000 ha, representing a substantial portion (21 per cent) of the total forested landscape. TE forests generally were characterized by dry, mixed-conifer forest types with more open structure and less biomass than non-TE forests. The structural and compositional conditions within TE forests make them ideal locations for management strategies designed to enhance landscape resilience and sustain fire-resistant trees. TE forestland ownership is diverse (35 per cent federal, 19 per cent Tribal, 16 per cent Washington State, 14 per cent private non-industrial and 13 per cent private industrial), indicating that successful land management will require collaboration among numerous partners. Recent wildfires (1984–2020) cumulatively covered 84 300 ha (22 per cent) of TE forests and 363 500 ha (25 per cent) of non-TE forests. TE forests experienced less high-severity fire than non-TE forests (39 per cent vs. 46 per cent, respectively). Recent high-severity fire effects in TE forests occurred primarily in the northern portion of the study region, reflecting the distribution of individual large fires. By quantifying the variability of TE forests and their recent fire activity, this study supports adaptive management strategies for landscape restoration, post-disturbance reforestation and climate adaptation.

Introduction

Forests are dynamic, and their characteristics and benefits to society vary across space and time with climate, disturbance and land use. Anthropogenic climate warming and land use have altered natural disturbance regimes throughout the world, and recent increases of tree mortality are projected to accelerate, particularly in seasonally dry ecosystems (van Mantgem *et al.*, 2009; Millar and Stephenson, 2015; Ramsfield *et al.*, 2016; Stephens *et al.*, 2018; Hagmann *et al.*, 2021; McNellis *et al.*, 2021; Williams *et al.*, 2022). The compounding effects of high-severity, stand-replacing wildfire and drought stress may result in widespread tree regeneration limitation and associated conversion of forests to novel forest conditions or non-forest cover types (McKenney *et al.*, 2007; Donato *et al.*, 2016; Kemp *et al.*, 2019; Coop *et al.*, 2020). Such tipping points, where wildfires punctuate ongoing directional changes, are especially likely in trailing edge forests (hereafter 'TE forests') near current ecotones (Davis *et al.*, 2019; Parks *et al.*, 2019; Davis *et al.*, 2020; Urza *et al.*, 2020). Here, we define TE forests as locations between the current limits and projected future limits where trees are unlikely to persist due to moisture stress. Widespread changes within this critical interface could accelerate losses of ecosystem services and co-benefits, including carbon storage, water quantity and quality,

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wildlife habitat, recreation, and jobs associated with tourism and timber harvest (Buotte *et al.*, 2020; Case *et al.*, 2021). At the same time, reductions in tree cover may favour more drought-adapted vegetation, such as woodlands and shrub-steppe, or support the recovery of non-forest land cover types that declined following decades of fire exclusion (Hessburg *et al.*, 2019). Although TE forests have been mapped at regional scales using 1-km resolution climate data and forest inventory plots (Parks *et al.*, 2019), there is a need for higher resolution characterization of TE forest extent and variability at sub-regional levels. Specifically, fine-scale maps of TE forest location, composition and structure, and recent wildfire activity are necessary for updating land management objectives and supporting climate adaptation strategies.

Drought- and fire-induced forest transformation is a major policy concern with important implications for adaptive management strategies globally (Anderegg *et al.*, 2020). As land managers grapple with increasing rates of ecosystem change, increasing uncertainty, and novel post-disturbance environments, the resist-accept-direct (RAD) framework provides three complementary options to respond to changing ecological trajectories, support decisions and collaborative partnerships and manage for multiple objectives across scales (Lynch *et al.*, 2021; Schuurman *et al.*, 2022). The RAD framework is especially applicable in seasonally dry, TE forests, where ecological transformations are likely to manifest first (Parks *et al.*, 2019) and where elevated fire risk and smoke impacts have catalysed collaborative natural resource management (Ager *et al.*, 2019; WADNR, 2020). Although land managers and partners generally understand that the risk of forest transformation is increasing, they often lack spatially explicit information, especially in recently burned areas, to help them determine locations where reforestation efforts are likely to succeed (i.e. resist) *vs* locations where a shift to different forest or non-forest vegetation types might be unavoidable, socially acceptable, or even ecologically restorative (i.e. accept or direct). In addition, regulations and policies on both public and private land often mandate reforestation densities that are too high for projected future climates and do not allow managers to actively direct transitions to more drought-tolerant, open woodland conditions or non-forest cover types. Such constraints will be amplified by recent government initiatives like the US Federal REPLANT Act, which quadruples investments in reforestation projects on national forests (Infrastructure Investment and Jobs Act, 2021). Similarly, climate adaptation strategies that scale up the implementation of forest restoration treatments to reduce fire risk (i.e. resist) and increase community wildfire protection (e.g. WADNR, 2020) require information on which forests are most vulnerable to drought, wildfire and forest transformation. Thus, data-driven maps of vulnerable TE forests—especially locations within TE forests where high-severity fires have recently killed overstory trees—are needed to help decision makers prioritize resources for landscape restoration, post-disturbance reforestation and climate adaptation (Meyer *et al.*, 2021; Stevens *et al.*, 2021; Larson *et al.*, 2022), as well as to revise reforestation regulations to reflect changing climate and disturbance regimes.

There are multiple modelling approaches and datasets that can be integrated to identify and map vulnerable TE forests, their attributes, and recent wildfire activity. Empirical climatic niche models generally combine numerous climatic variables to project the presence or absence of species over multiple time periods (Guisan and Zimmermann, 2000; McKenney *et al.*, 2007). Although such models are widely recognized for their ability to predict species distributions at large spatial scales, they tend to incorporate climatic variables that are often highly correlated and may or may not be relevant to ecological processes of interest, such as tree regeneration and establishment (Pearson and Dawson, 2003). Given this limitation, an integrated, parsimonious approach that incorporates physiologically relevant, synthetic climate variables is warranted (Case and Lawler, 2017). One such variable, climatic water deficit (CWD; aka climatic moisture deficit; Parks *et al.*, 2019) is a key index of moisture balance that strongly corresponds with tree regeneration, vegetation composition and vulnerability to drought (Stephenson, 1998; Young *et al.*, 2017; Cansler *et al.*, 2022). Previous assessments have demonstrated clear associations between forest cover and CWD and have used future projections of CWD to project potential vegetation changes, including in TE forests (Parks *et al.*, 2019; WADNR, 2020). In addition to climatic and biophysical datasets, recently developed maps of forest composition, structure, biomass, land ownership, and fire effects enable the timely and robust analysis of TE forest characteristics and values at risk in the Pacific Northwest, US.

The eastern Cascade Range of Washington State is a vital location to investigate potential forest transformation because it contains drought-vulnerable forests with high fire risk and values at stake, including diverse ecosystems, socio-economic values and human communities in the wildland-urban interface. Fire activity has increased in recent decades (Reilly *et al.*, 2017; Haugo *et al.*, 2019; WADNR, 2022), as have insect and drought disturbances (Meigs *et al.*, 2015), prompting the Washington State legislature to initiate a 20-year Forest Health Strategic Plan to accelerate forest restoration, improve community wildfire protection and support climate change adaptation (WADNR, 2020). Steep environmental gradients found throughout this region facilitate rapid transitions among vegetation types, including TE forests. As such, drought-stressed forests could be highly vulnerable to transforming to novel forest or non-forest cover types following wildfire, and the potential for such transformation likely varies across spatiotemporal scales and gradients that have not been previously quantified. Although recent studies in the region have begun to disentangle the complex relationships among forest and fuel structure, drought and fire risk, fire behaviour and effects, and post-fire regeneration (e.g. Hessburg *et al.*, 2019; Stevens-Rumann and Morgan, 2019; Povak *et al.*, 2020; Prichard *et al.*, 2020), we fill an important knowledge gap by characterizing the variability and recent fire activity within TE forests. Our specific objectives are to:

1. Map the spatial distribution of TE forests based on climatic water deficit and forest cover;
2. Characterize the composition, structure, biomass, and land ownership of TE and non-TE forests;
3. Quantify the extent and severity (i.e. tree mortality) of recent wildfires in TE and non-TE forests;
4. Identify locations of potential forest loss and type conversion where recent high-severity fire has occurred in TE forests.

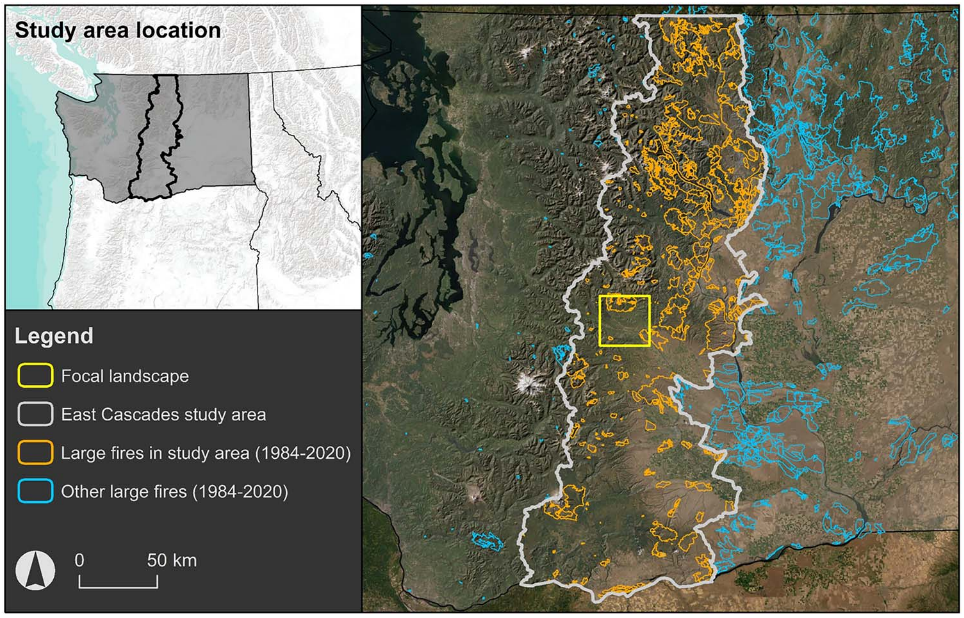

Figure 1 Study area map with recent large wildfires and aerial imagery of land cover. Green areas are generally forested. East Cascades study area is based on watersheds (HUC8). Fire perimeters are compiled by Washington State Department of Natural Resources from state and federal sources (e.g. MTBS; https://www.mtbs.gov/). Yellow box corresponds to zoom maps in Figures 3 and 7.

Methods

Study area

Our study area encompasses the eastern slopes of the Cascade Range in Washington State and is based on sub-watershed boundaries (Hydrologic Unit Code 8; Seaber *et al.*, 1987; Figure 1). This mountainous region is defined by steep topoclimatic gradients and a diverse array of vegetation conditions (Cansler and McKenzie, 2014). Forests are dominated by conifers, ranging from cold, subalpine forests at higher elevations and northerly aspects through moist and dry mixed-conifer forests at intermediate elevations and aspects to warm, dry forests and woodlands at lower elevations and southerly aspects (species described below). Local forest composition, structure and associated historical fire regimes vary with climate, topography, soil parent material and disturbance and land use history (Franklin and Dyrness, 1973; Hessburg *et al.*, 2000; Agee, 2003). At lower elevations (<500–1000 m, depending on latitude) and drier sites, dry forests transition to non-forest vegetation types including sparse woodlands, shrub-steppe and grasslands. Although precipitation and temperature are somewhat variable across the study area, the overall climate is defined by relatively warm, dry summers and cold winters, with the majority of precipitation occurring as snow. These climatic patterns generally result in seasonally dry conditions that are conducive to periodic natural disturbances (Franklin and Dyrness, 1973; Meigs *et al.*, 2015).

Land management objectives, natural disturbance regimes and forest co-benefits vary with forest composition, structure and ownership across the study area. Federal land that is managed for multiple uses by the USDA Forest Service is the most widespread ownership type, especially at moderate and high elevations. At lower elevations, there is a more diverse distribution of ownership, including Tribal lands, Washington State, municipal (local government), private industrial and private non-industrial (including non-governmental organizations). Lightning-ignited fires and cultural fires ignited by Native American Tribes were frequent in the study area prior to Euro-American colonization, especially in dry forests at lower elevations (Hessburg *et al.*, 2000; Lake *et al.*, 2017). Relatively frequent (5–25 years), low-severity fires at lower elevations (Everett *et al.*, 2000; Agee, 2003) and less frequent (25–80 years), mixed-severity fires at intermediate elevations (Hessburg *et al.*, 2007; Churchill *et al.*, 2022) created a heterogeneous mosaic of forest and non-forest vegetation types (Hessburg *et al.*, 2000). This multi-level patchwork of forest and non-forest vegetation exhibited dynamic, fire-facilitated feedbacks that enhanced resistance and resilience to large-scale disturbance and climatic fluctuations (Hessburg *et al.*, 2019; Churchill *et al.*, 2022). Spurred by climate warming and increasing forest density and homogeneity following decades of fire exclusion and increasing fire deficits (Haugo *et al.*, 2019), tree mortality has increased in recent years due to wildfire, drought and insect outbreaks (Meigs *et al.*, 2015; Reilly *et al.*, 2017). Given the widespread extent of similar geographic conditions, these forests and their vulnerability to projected drought and wildfire dynamics are broadly representative of seasonally dry forest landscapes throughout western North America.

Overall approach and geospatial datasets

In this study, we first combined current forest cover with current and future CWD to map the spatial distribution of TE forests (Objective 1). We then characterized TE and non-TE forests by overlaying existing maps of forest composition, structure, biomass and ownership (Objective 2). Finally, we mapped recent wildfire extent and severity (i.e. tree mortality) in TE and non-TE forests (Objective 3) to assess locations susceptible to type

Table 1 Key datasets for geospatial overlay analysis.

Variable (units)	Description	Reference
Climatic water deficit (CWD; mm year^{-1})	Process-based index of moisture stress in terrestrial ecosystems (also known as climatic moisture deficit)	Stephenson *et al*. (1998), Parks *et al.* (2019)
Current forest cover (binary)	Hybrid forest mask derived from three existing forest cover layers	This article
Biomass (Mg ha^{-1})	GNN-based aboveground tree biomass using the component ratio method for live trees ≥2.5 cm DBH (2017)	Ohmann *et al.* (2012)
Forest composition (categorical)	GNN-based forest type, which describes dominant tree species (based on basal area) of current vegetation; simplified to general types (2017)	Ohmann *et al.* (2012)
Forest structure (categorical)	GNN-based forest structural condition based on size class and cover class (2017)	Ohmann et al. (2012), O'Neil *et al.* (2001)
Land ownership (categorical)	Current land ownership of forested parcels compiled by WA State Department of Natural Resources	This article
Fire extent (ha)	Recent wildfire perimeters (1984–2020) compiled by WA State Department of Natural Resources	This article
Fire severity (categorical)	Low-, moderate-, high-severity fire based on RdNBR and field-based observations of tree basal area mortality (<25%, 25–75%, >75%)	Miller and Thode (2007), This article

conversion (Objective 4). Here, we describe the key geospatial datasets (Table 1) and how we combined them in a geospatial overlay analysis (Supplementary Figure 1).

Climate and forest cover

We obtained climate data, including precipitation, temperature and relative humidity, from Climate NA version 6.0 (downscaled to 90 m; http://www.climatewna.com; Wang *et al.,* 2016). We used CWD as our primary climatic variable because it is a widely used index of moisture stress in terrestrial ecosystems that integrates multiple climate and topography variables in a process-based way and is strongly related to potential vegetation composition (Stephenson *et al.,* 1998, Young *et al.,* 2017, Cansler *et al.,* 2022). We computed CWD as the difference between potential evapotranspiration (PET) and actual evapotranspiration (AET) after Dobrowski *et al.* (2013), as modified by Cansler *et al.* (2022) to incorporate the Priestley-Taylor equation (Priestley and Taylor, 1972). Along with elevation and latitude, the Priestley-Taylor equation uses albedo, solar radiation, relative humidity, and minimum and maximum temperatures to estimate daily PET. We computed annual CWD, PET and AET as the sum of monthly values and averaged the CWD values for current (1981–2010) and future (2041–2070) time periods based on projected future climate data from the average (ensemble) of 15 Global Circulation Models under the Representative Concentration Pathway (RCP) 8.5 (Figures 2 and 3). RCP 8.5 is a comparatively high warming scenario representing a future with no globally coordinated greenhouse gas mitigation (Riahi *et al.,* 2011; IPCC, 2013). We then selected the time period 2041–2070 an indicator of potential mid-21st century conditions. See Supplementary Data and Methods 1A for full CWD details.

We created a current forest cover layer encompassing the study area (Figure 2A) by leveraging the strengths of three widely used remotely sensed datasets: National Land Cover Dataset (NLCD; https://www.mrlc.gov/), LANDFIRE (https://landfire.gov/), and gradient nearest-neighbour forest type (GNN; Ohmann *et al.,* 2012; https://lemmadownload.forestry.oregonstate.edu/). We optimized our hybrid forest cover map by iteratively comparing different combinations of the existing datasets with 2017 NAIP 4-band imagery covering Washington (full details in Supplement 1B). This process allowed us to account for discrepancies, particularly locations that were misclassified as non-forest because of recent disturbance (including timber harvest and wildfires). We recognize that land use has altered forest cover; timber harvest has decreased forest cover in some locations, while fire exclusion has increased forest cover extensively (Hessburg *et al.,* 2007; Hessburg *et al.,* 2019). Although today's forests do not align fully with their biophysical potential, our main focus was to characterize current rather than historical forest conditions in the TE zone.

Forest composition, structure, biomass and ownership

We assessed forest characteristics of TE and non-TE forests using 2017 vegetation maps derived from GNN imputation (Ohmann *et al.,* 2012). GNN maps combine data from federal forest inventory plots, spatial predictors and Landsat time series to impute plot-level attributes for all forested pixels across Washington and other states; these maps are accurate across modelling regions but have limited accuracy at the pixel level, similar to other remote sensing approaches (Ohmann *et al.,* 2012). GNN maps include numerous plot variables (available online: https://lemmadownload.forestry.oregonstate.edu/), and we selected a subset of forest composition and structure variables for our analysis (Table 1). To simplify the overlay analysis (details below), we combined GNN forest types into six classes

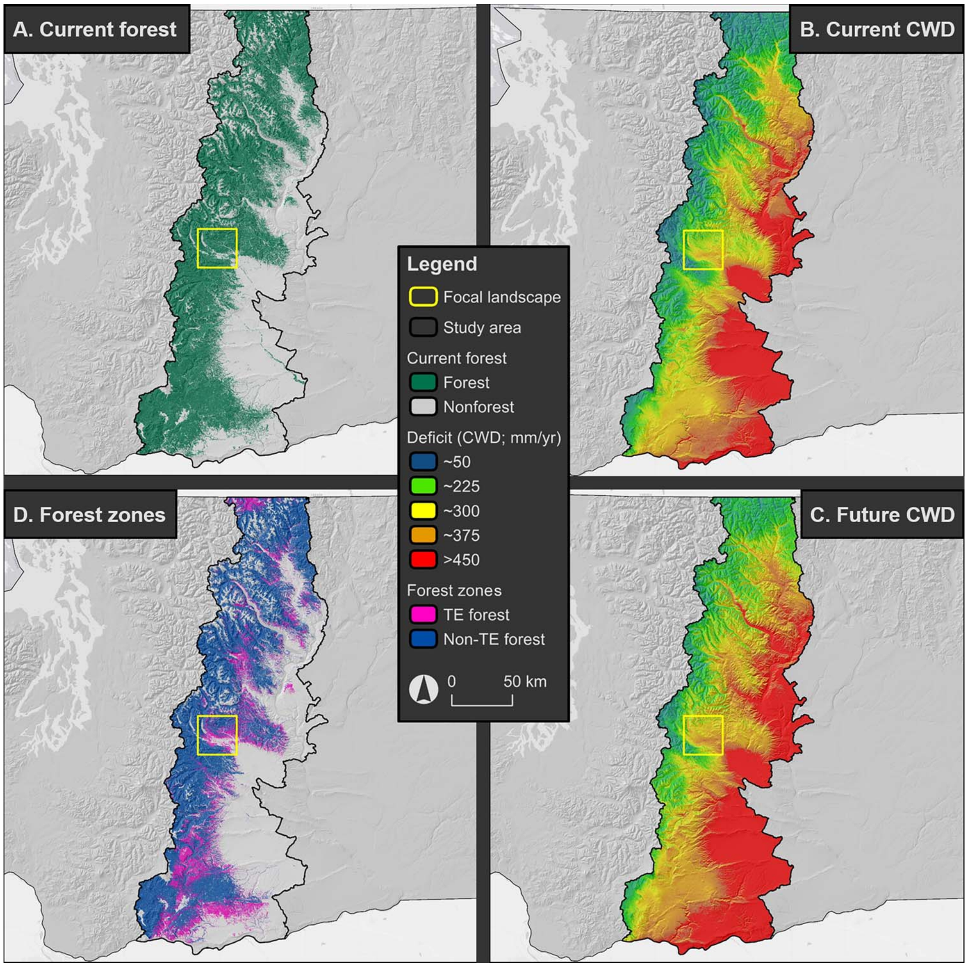

Figure 2 (A) Forest cover, (B) current climatic water deficit (CWD; 1981–2010), (C) future CWD (2041–2070) and (D) TE forests across the study area in the Washington East Cascades. (A) Tree cover declines with decreasing elevation in the Washington East Cascades. (B) Current (1981–2010) CWD declines rapidly from west to east and with decreasing elevation. (C) CWD is projected to increase from current to future periods (2041–2070), particularly in some higher elevation areas. (D) Magenta shows TE forests; blue shows non-TE forests. Yellow box corresponds to zoom map in Figures 3 and 7. Across the Washington East Cascades study region, TE forests encompass 386 893 ha (21 per cent) of the forested landscape.

applicable to forest composition types in the study area based on dominant tree species basal area (Meigs and Krawchuk, 2018; Figure 4, Supplementary Table 1). Specifically, we mapped all locations with the plurality of live tree basal area in the following species groups, ranging from warm/dry to cold/moist: ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & C. Lawson), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), silver fir (*Abies amabilis* Douglas ex J. Forbes), subalpine (including subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.), mountain hemlock (*Tsuga mertensiana* (Bong.) Carrière), and lodgepole pine (*Pinus contorta* Douglas ex Loudon)), and other forest (including western larch (*Larix occidentalis* Nutt.) and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), among others).

We combined GNN forest structural conditions into five structure classes based on live tree canopy cover and tree size after Meigs and Krawchuk (2018). Specifically, we defined sparse and open forest structure classes with canopy cover <10 and 10–40 per cent, respectively. We defined closed forest with canopy cover >40 per cent and further classified closed forest into three size classes based on dominant tree quadratic mean diameter: small: <25 cm QMD; medium: 25–50 cm QMD; large: >50 cm QMD. QMD is a standard metric of average tree size that gives greater weight to larger trees influencing basal area (Curtis and Marshall, 2000). We also calculated the total and average live tree (≥2.5 cm DBH) biomass in the TE and non-TE forest types based on the component ratio method (Ohmann *et al.*, 2012), classifying the continuous biomass values into discrete bins (50 Mg ha^{-1}) for the overlay analysis (described below).

We evaluated land ownership of TE and non-TE forests using 2019 Washington county tax parcel data and public land ownership layers compiled by the Washington State Department of Natural Resources (WADNR, 2020). We intersected these ownership layers hierarchically and removed overlapping slivers to create a seamless ownership map across eastern Washington. We illustrate fine-scale variability of the TE forest types, land ownership, forest type and forest structure for a centrally located focal landscape surrounding Cle Elum, WA (Supplementary Figure 3).

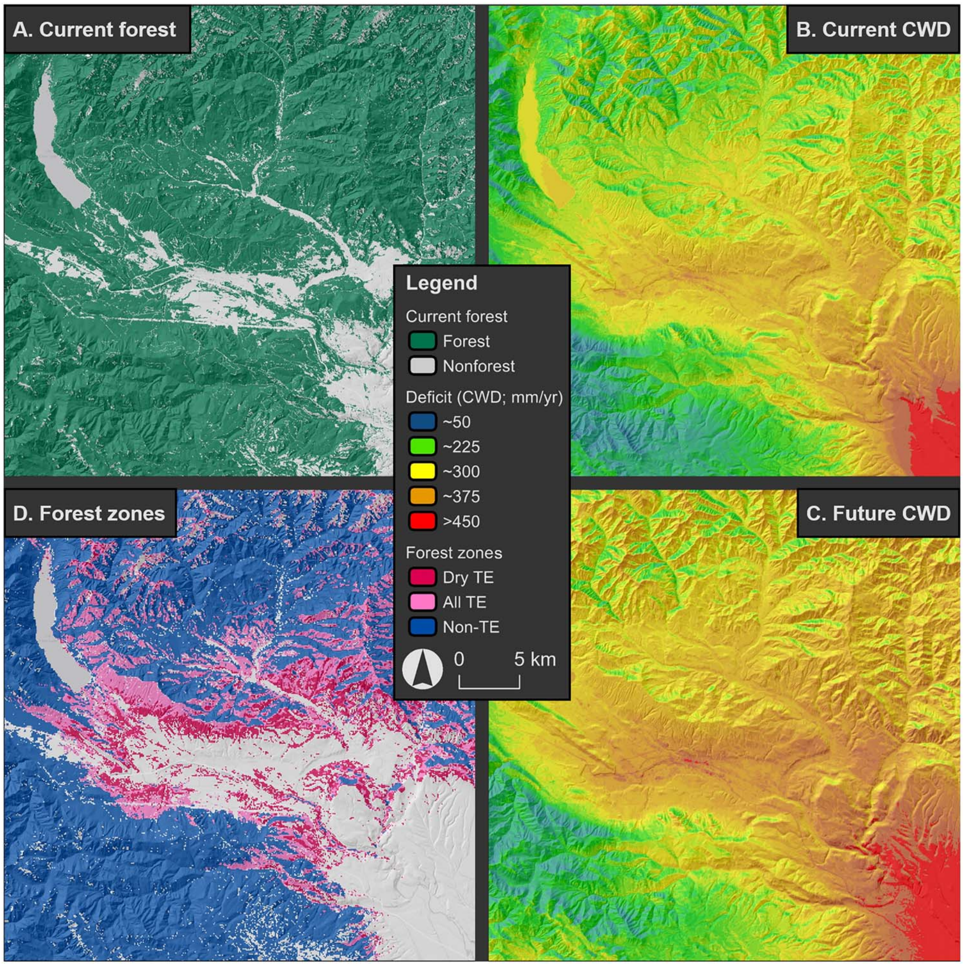

Figure 3 Focal landscape surrounding Cle Elum, WA, illustrating (A) forest cover, (B) current CWD (1981–2010), (C) future CWD (2041–2070) and (D) TE forest zones. TE forests in panel D are constrained to currently forested areas and include the dry TE forest subset (dark magenta) to identify the most drought-vulnerable locations (See Methods). Dry TE forests encompass 119 143 ha, equivalent to 30 per cent of the all TE forests.

Recent wildfire extent and severity

We used a database of recent wildfires (1984–2020) maintained by the Washington State Department of Natural Resources, which includes all large fires (>400 ha) represented by US Federal databases (e.g. MTBS; https://www.mtbs.gov/) and smaller events in Washington forests. We employed Google Earth Engine (GEE) to calculate the Relative differenced Normalized Burn Ratio (RdNBR; Miller and Thode, 2007) using annual mean composites of Landsat imagery (Parks *et al.*, 2018). RdNBR captures the relative fire-induced change in dominant vegetation and is appropriate for assessing fire effects across multiple events spanning heterogeneous pre-fire conditions (Miller and Thode, 2007; Cansler and McKenzie, 2014; Meigs and Krawchuk, 2018). We incorporated a dNBR offset based on a 180-m buffer around each fire and computed RdNBR using imagery from the years immediately before and after each fire (2-year interval; Parks *et al.*, 2018). We adapted the GEE code from Parks *et al.* (2018) to better capture the Washington growing season (15 June to 15 September rather than 1 June to 30 September). We classified continuous RdNBR maps into low-, moderate- and high-severity categories corresponding to <25, 25–75 and >75 per cent percent basal area mortality, respectively, based on 1-year post-fire forest inventory plots in Washington and Oregon ($n = 181$; RdNBR thresholds = 262, 600; Saberi and Harvey, 2022). We avoided double-counting areas burned more than once (i.e. reburn) by mosaicking all years together by maximum RdNBR value, thereby evaluating the highest-severity effects in any given location. Finally, we focused on the high-severity class to assess locations of potential type conversion to novel forest or non-forest conditions (full details in Supplement 1C).

Data analysis

Mapping trailing edge forests

We used a climate analogue modelling approach similar to Parks *et al.* (2019) to delineate TE forests as locations between the current limit and projected future limit where trees are unlikely to persist due to moisture stress. First, we computed the 95th percentile of current CWD of forested locations within a 50-km square moving window for each pixel using the current forest

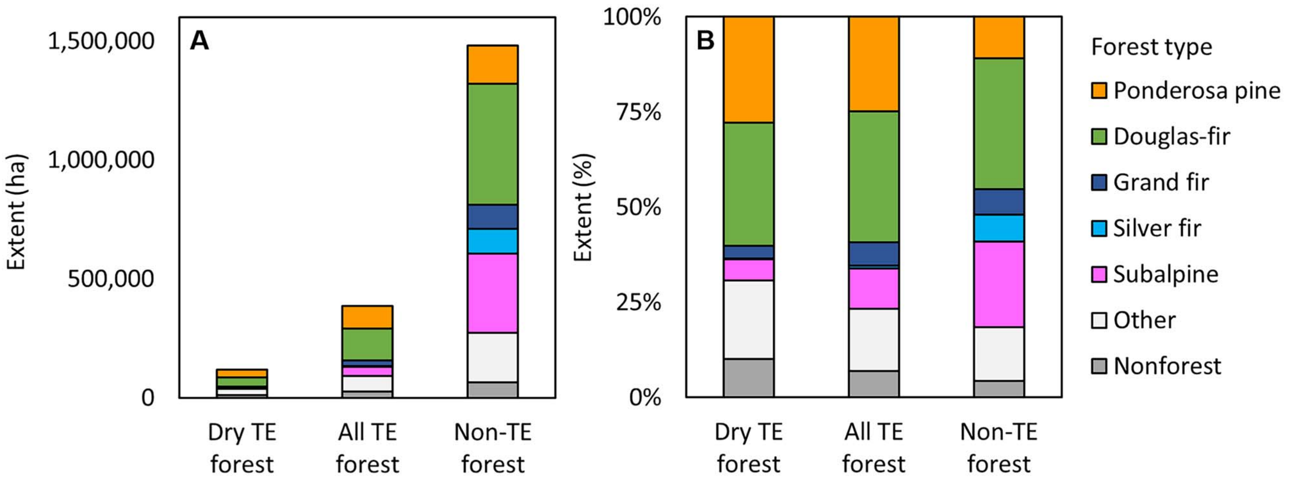

Figure 4 Composition of TE and non-TE forests on absolute (A) and relative (B) scales. Forest composition types are based on general groups of dominant tree species by basal area according to 2017 GNN maps (see Appendix Supplementary Table 1). TE forests are predominantly dry, mixed-conifer forests, with a higher proportion of ponderosa pine than non-TE forests (28, 25 and 11 per cent ponderosa pine for dry TE, all TE and non-TE forests, respectively). Forest types vary with topo-edaphic gradients and land use (Supplementary Figure 3).

cover map described above. We selected the 95th percentile based on high-resolution imagery of current tree line patterns across the study area and used a moving window approach because the relationship between CWD and forest composition varies with latitude and local topoedaphic conditions. We then mapped the current moisture-limited edge of TE forests (analogous to the current low elevation treeline) as locations beyond which current CWD exceeded the 95th percentile of current CWD of forested pixels. We classified the other edge of TE forests (analogous to the future low elevation treeline) as locations beyond which the projected future CWD exceeded the 95th percentile of current CWD (Figures 2 and 3). We then used the current forest cover map described above to constrain (i.e. mask) the TE forest zone and characterize contemporary forest conditions and fire effects. Finally, to identify the areas of TE forests with the highest moisture stress—that are more vulnerable to conversion to non-forest in the coming decades—we mapped 'dry TE' forests as a subset of 'all TE' forests based on the mean (midpoint) of current and projected future CWD (Supplementary Figure 1). We recognize that TE forests could be delineated using other approaches, climate variables, moving window sizes and CWD percentiles. Although our primary goal was to characterize the variability of TE vs non-TE forests rather than comparing different TE forest classifications, we assessed the sensitivity of the 95th percentile CWD threshold by comparing TE extent, fire extent and burn severity for TE maps based on 91st and 99th percentiles (Supplementary Figure 2, Supplementary Table 2). We created these TE and non-TE forest maps using the raster (Hijmans *et al.*, 2021) package in the R statistical environment (R Core Team, 2021).

Geospatial overlay analysis

Our final analytical step was to overlay the maps of dry TE forest all TE forest and non-TE forest with the maps of forest conditions and burn severity described above. We projected all geospatial layers to a common projection (Washington State Plane, North American 1983 HARN) and assessed unique combinations of categorical data using the Combine function in ArcMap 10.6.1 (Esri, 2018). For each TE forest category, we summarized forest composition, structure, biomass and ownership, as well as fire extent and severity after Meigs and Krawchuk (2018). Across all TE forests, we also assessed burn severity among ownership classes and mapped high-severity fire locations (>75 per cent tree mortality). Although dry TE forests are a subset of all TE forests, we report totals for each TE forest class separately to enable better comparison of forest composition, structure and fire activity among TE zones.

Results

Spatial distribution of TE forests

TE forests were a widespread and substantial component of the eastern Cascades study region. TE forests covered 386 893 ha, and non-TE forests covered 1 480 548 ha (Table 2). TE forests encompassed 21 per cent of the total forested landscape (1 867 441 ha; Table 2) and were especially widespread in southern portions of the study area (Figure 2). The dry TE forest subset covered 119 143 ha, representing a moderate portion (31 per cent) of all TE forests (Table 2). TE forests generally occupied drier locations at lower elevations and south-facing slopes, particularly the dry TE forests (Figures 2D and 3D, Supplementary Figure 3).

Composition, structure, biomass and ownership of TE forests

TE forests were predominantly dry, mixed-conifer forests, with a higher proportion of ponderosa pine than non-TE forests. Specifically, ponderosa pine represented 28, 25 and 11 per cent of the dry TE, all TE and non-TE forests, respectively (Figure 4). TE forests had similar relative abundance of Douglas-fir forest but a lower

Table 2 Summary of attributes of TE forests and non-TE forests

Variable (units)	Dry TE forest	All TE forest	Non-TE forest	Total forest
Extent (ha)	119 143	386 893	1 480 548	1 867 441
Total biomass (Mg)	8 663 361	32 068 400	200 134 911	232 203 311
Biomass per unit area (Mg ha^{-1})	72.71	82.89	135.18	124.34
Fire extent (ha)	25 277	84 348	363 546	447 894
Fire extent of area (%)	21	22	25	24
Low severity (%)	31	28	26	26
Moderate severity (%)	37	33	28	29
High severity (%)	32	39	46	45

Note that the dry TE forest category is a subset of all TE forests. See Table 1 for description of datasets and sources.

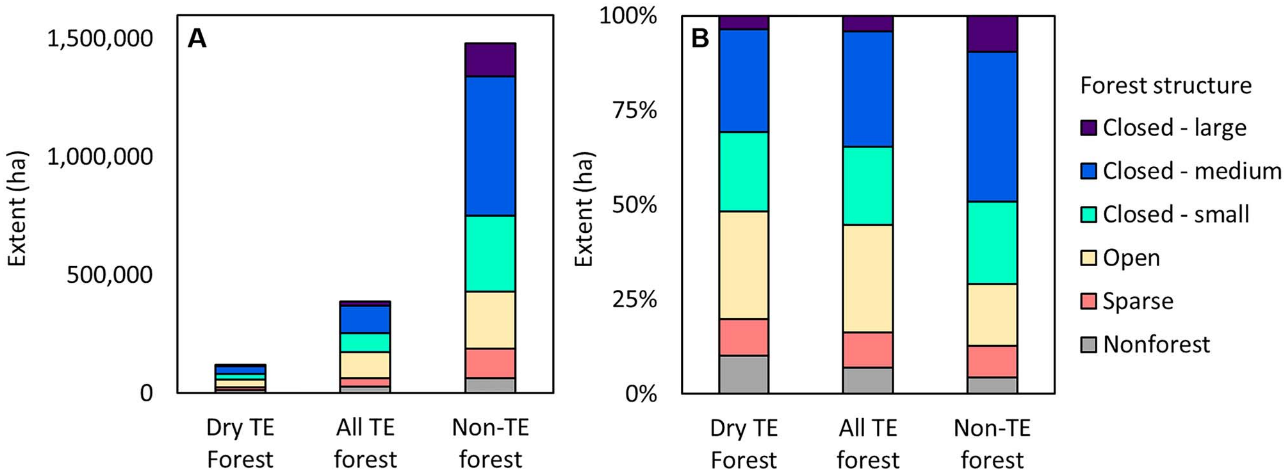

Figure 5 Structure of TE and non-TE forests on absolute (A) and relative (B) scales. Structural conditions are based on live tree canopy cover and size classes according to 2017 GNN maps. Structure classes are arranged in increasing order of tree cover and size and are defined in the Methods. TE forests generally have more open conditions than non-TE forests, which have more medium, close canopy forests. Forest structure varies with topo-edaphic gradients and land use (Supplementary Figure 3).

proportion of subalpine forest than non-TE forests, particularly in dry TE forests (Figure 4).

TE forests generally exhibited more open conditions than non-TE forests, which had more closed-canopy forests dominated by medium trees. The sparse and open conditions together accounted for 38 per cent in both zones of TE forests but only 24 per cent of non-TE forests (Figure 5). Dry TE forests contained 8.7 million Mg of biomass (72.71 Mg ha^{-1}), whereas all TE forests contained 32 million Mg of biomass (82.89 Mg ha^{-1}), and non-TE forests contained 200 million Mg of biomass (135.18 Mg ha^{-1}) (Table 2). Total biomass within dry TE forests was equivalent to 27 per cent of the biomass in all TE forests, which, in turn, was equivalent to 16 per cent of the total biomass in non-TE forests.

Land ownership in TE forests was 35 per cent federal, 19 per cent Tribal, 16 per cent Washington State, 14 per cent private non-industrial and 13 per cent private industrial, compared to 71, 10, 8, 5 and 4 per cent in non-TE forests, respectively (Table 3). As such, the relative abundance of ownership was higher in TE forests than in non-TE forests for all land owners with the exception of federal ownership, which was disproportionately lower in TE forests (Table 3). In addition, TE forests were distributed much more evenly among land ownerships than non-TE forests.

Recent wildfire extent and severity in TE and non-TE forests

Recent wildfires (1984–2020) cumulatively covered 84 300 ha (22 per cent) of TE forests and 363 500 ha (25 per cent) of non-TE forests (Table 2). Wildfire extent was relatively limited (25 000 ha) in the dry TE forest subset (Figure 6). TE forests experienced less high-severity fire than other forests, with the dry TE, all TE and non-TE forests exhibiting 32, 39 and 46 per cent high-severity fire, respectively (Figure 6, Table 2). Conversely, TE forests experienced more low-severity fire than other forests (31, 28 and 26 per cent low severity for dry TE, all TE and non-TE forests, respectively) (Table 2). Within TE forests, Tribal lands experienced the most high-severity fire (48 per cent), followed by federal (41 per cent), industrial (37 per cent) and state lands (35 per cent) (Figure 6).

High-severity locations susceptible to potential forest loss and type conversion

The northern portion of the study area has burned much more extensively in recent decades (1984–2020), resulting in more high-severity extent susceptible to potential type conversion in TE forests (Figure 7). Conversely, the southern portion of the

Table 3 Land ownership of TE forests and non-TE forests

Land ownership	All TE forest		Non-TE forest	
	Extent (ha)	Extent (%)	Extent (ha)	Extent (%)
Federal	137 265	35	1 054 693	71
Tribal lands	71 936	19	148 975	10
Washington State	61 808	16	122 375	8
Private non-industrial	53 855	14	68 373	5
Private industrial	48 456	13	60 236	4
Other	13 574	4	25 896	2
Total	386 893	100	1 480 548	100

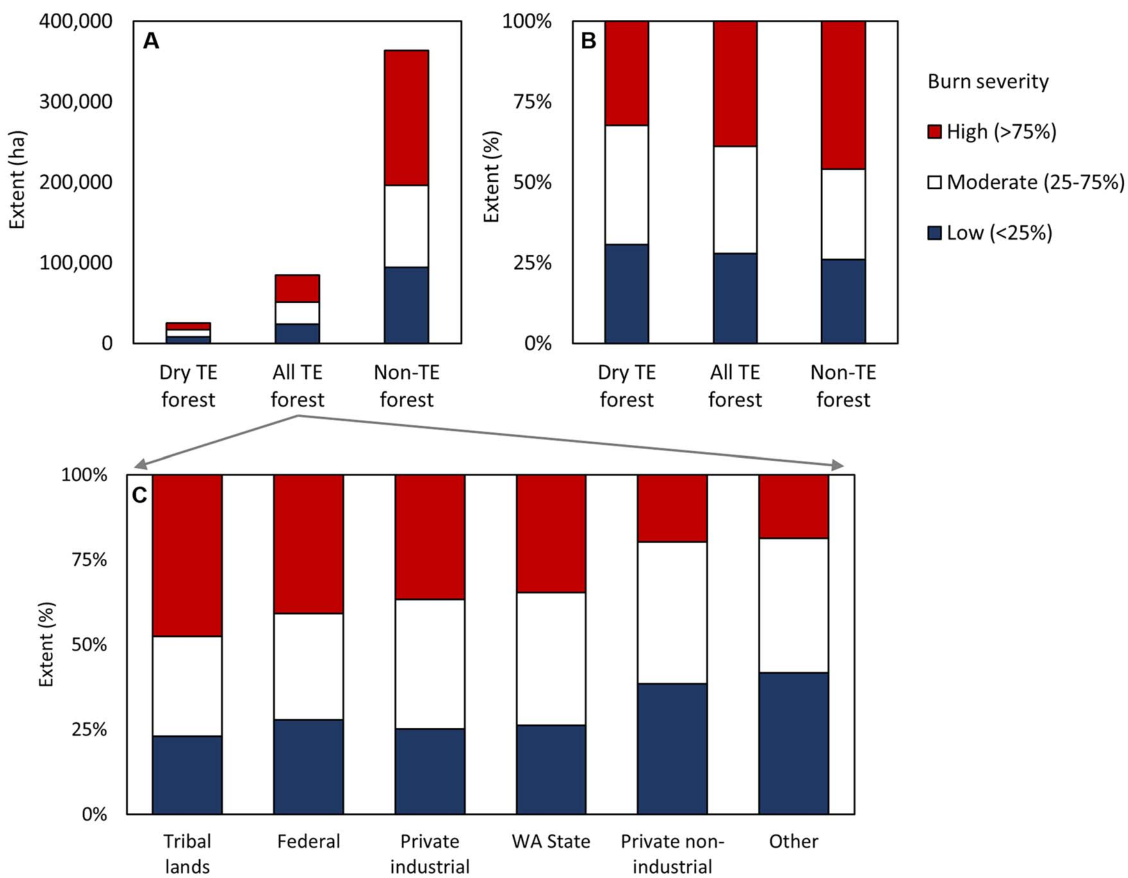

Figure 6 Recent wildfire extent and severity (1984–2020) in TE and non-TE forests on absolute (A) and relative (B) scales and across land ownership categories (C). Burn severity classes are derived from Landsat-based RdNBR and field-based observation of tree mortality. TE forests have experienced less high-severity fire than other forests (32, 39 and 46 per cent high severity for dry TE, all TE, and non-TE forests, respectively). Across All TE forests, Tribal lands have experienced the most high-severity fire (48 per cent), followed by federal (41 per cent), industrial (37 per cent) and Washington State lands (35 per cent).

study area has not experienced as much fire despite containing the most widespread TE forests (Figure 7A). A small number of fire events ($n = 5$) accounted for the vast majority of recent high-severity fire extent in TE forests across the study area, underscoring the episodic nature of fire and how large swaths of TE forest can burn in a short time span. Most of the TE forest zone has not experienced high-severity fire, but large fire events in the northern and southern ends of the study area (2006 Tatoosh Fire and 2015 Cougar Creek Fire, respectively) were two prominent examples of high-severity fire in TE forests (Figure 7C).

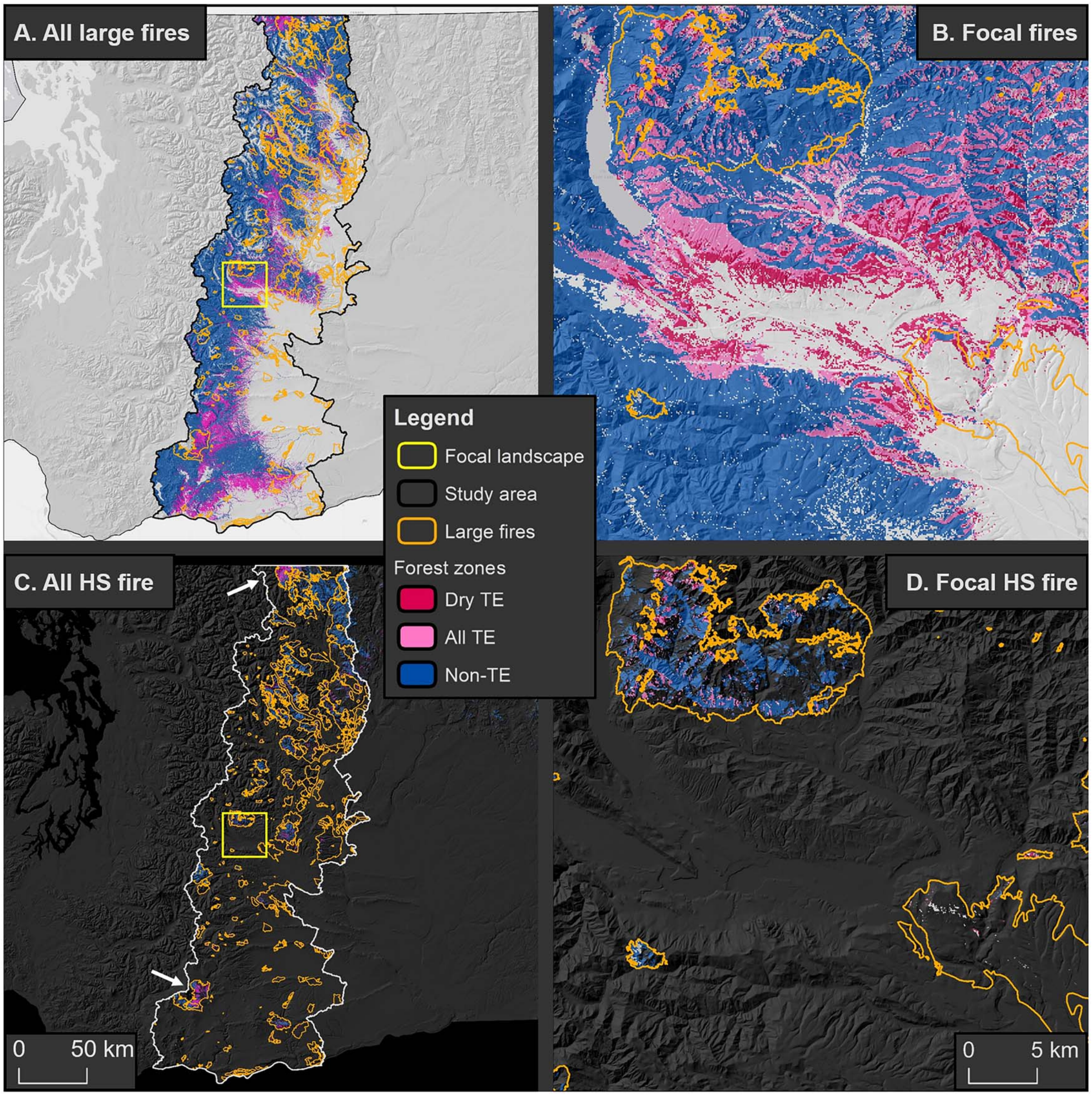

Figure 7 Fire extent (1984–2020) in TE and non-TE forests (A) and locations of high-severity fire in TE forests across the study area (C). Insets show fire extent (B) and high-severity locations (D) in the Cle Elum focal area. Note that wildfire extent in panels A and C is the same as Figure 1. TE zones in panels B and D include the dry TE forest subset to identify the most drought-vulnerable locations. The high-severity overlay excludes non-stand-replacing fire effects and corresponds to the high-severity category (>75 per cent tree mortality) in Figure 6. A small number of fire events ($n = 5$) accounted for the vast majority of recent high-severity fire in TE forests across the study region, underscoring the episodic nature of fire and how large swaths of TE forest can burn in a small window. White arrows in panel C indicate the two most prominent examples of large, high-severity fire in TE forests: 2006 Tatoosh Fire in the north; 2015 Cougar Creek Fire in the south.

Discussion

Characterizing trailing edge forests at risk of transformation

This study builds on broad-scale assessments of TE forests (e.g. Parks *et al.*, 2019) by leveraging downscaled climate data and characterizing the variability of forest composition, structure and ownership within multiple TE forest zones. We focused on the eastern Cascade Range of Washington, a mountainous landscape defined by steep topoclimatic and vegetation gradients where numerous social-ecological values are vulnerable to increasing drought and wildfire activity. Our results indicate that TE forests currently encompass 387 000 ha, representing a substantial portion (21 per cent) of the forested landscape across the study area and illustrating that projected drought and wildfire have the potential to transform forests to novel forest conditions or non-forest land cover types (Coop *et al.*, 2020). TE forests are generally dry, mixed-conifer forest types with more open structure, less biomass and more variable land ownership than non-TE forests, highlighting that land management will require multiple strategies and close collaboration among diverse partners. In addition, recent wildfires have occurred in relatively even proportions across TE and non-TE forests, while burn severity was lower in TE forests, consistent with expectations for drier, more open conditions with slower fuel accumulation (Agee, 2003). Finally, a few large wildfires have accounted for most of the high-severity fire locations in the TE zone, where tree regeneration likely will be limited by projected drought conditions (Stevens-Rumann and Morgan, 2019; Williams *et al.,* 2022). Our results offer a fine-scale projection of potential forest changes that are consistent with other vegetation change modelling studies, underscoring the high vulnerability of TE forests (e.g. Shafer *et al.,* 2015; Case *et al.,* 2020).

The widespread extent and variable composition and structure of TE forests set the stage for critical tipping points when future wildfires occur while increasing the uncertainty faced by land managers and residents. In the drier subset of TE forests, natural tree regeneration will be increasingly moisture limited, depending on the patch size distribution of tree mortality and post-fire climate (Donato *et al.,* 2016; Kemp *et al.,* 2019; Davis *et al.,* 2020; Povak *et al.,* 2020; McNellis *et al.,* 2021). Concurrently, the moist mixed-conifer forest types and closed-canopy structure in more mesic TE forests highlight that a wide range of forest conditions could be vulnerable to fire-catalysed transformation, although high-severity fire spanning a range of patch sizes occurred historically in the study area, especially in moist forests (Hessburg *et al.,* 2007; Haugo *et al.,* 2019). Our findings of relatively high variability of TE forests and recent fire effects underscore the value of locally adaptable management objectives and approaches (detailed implications below).

Because recent fires have been concentrated in the northern portion of the study area while TE forests are concentrated in the southern portion, the vast majority of TE forests have not experienced fire for decades. As such, recently unburned TE forests have not yet realized an opportunity for fire-catalysed forest transformation (Coop *et al.,* 2020; Davis *et al.,* 2020). Where fires have occurred in the study area, the overlap of high-severity patches with TE forest has been relatively uncommon and dispersed in small, disjunct patches, highlighting how spatiotemporal windows of disturbance co-occurrence can be narrow (Meigs *et al.,* 2015). As fires accumulate across the landscape, however, more TE forests will inevitably experience high-severity fire, sometimes in large patches as illustrated by recent large events, such as the 2006 Tatoosh Fire and 2015 Cougar Creek Fire. Large, high-severity patches are primed for ecological transformation—including transition to novel forest types or conversion to non-forest land cover types—when post-fire climate limits tree regeneration and alters successional trajectories (Kemp *et al.,* 2019; Stevens-Rumann and Morgan, 2019; Larson *et al.,* 2022).

The distribution of TE forest land ownership has important ramifications for ecosystem services and co-benefits, including carbon storage, water quantity and quality, biodiversity, recreation, jobs, and community wildfire protection (Buotte *et al.,* 2020; Case *et al.,* 2020). Specifically, the relatively even TE forest ownership (Table 3) makes multiple land owners vulnerable to economic and environmental losses when rapid changes occur, but it also increases the opportunity for innovative, cross-boundary partnerships to address changing disturbance regimes. In this context, a disproportionate burden is placed upon smaller land owners and those who rely more heavily on ecosystem services for livelihoods, since these owners have fewer options to mitigate risk and recover from rapid changes (UW Climate Impacts Group, 2018). Moreover, Indigenous communities that depend on place-based natural resources and ecosystem services for a variety of reasons, including food, water, medicine, spiritual needs and cultural identity may be disproportionately impacted by ecosystem transformation (Lynn *et al.,* 2013). For example, some Tribal lands contain substantial TE forest and recently have experienced large patches of high-severity fire, underscoring the need for investment in landscape restoration and climate adaptation strategies in these communities.

Uncertainties and future research needs

Throughout western North America, growing evidence of regeneration limitations in seasonally dry forests following high-severity fire indicates that ecological transformation is already occurring and will continue (Davis *et al.,* 2019; Coop *et al.,* 2020). Predicting which areas are most vulnerable to transformation is challenging, however, due to multiple sources of uncertainty in statistical models, geospatial data and climate projections (Parks *et al.,* 2019). Each dataset we utilized has inherent limitations, and there are potentially compounding uncertainties of our maps of TE forest zones, forest cover, forest composition and structure and burn severity. Because vegetation maps (e.g. Ohmann *et al.,* 2012) and fire maps (e.g. Miller and Thode, 2007) have been evaluated previously, we focused our sensitivity analysis on the uncertainty of the TE maps presented here. Decreasing the CWD threshold from the 95th to the 91st percentile increased TE forest extent from 387 000 (21 per cent of all forest) to 538 000 ha (29 per cent of all forest), including additional moist forest conditions (Supplementary Figure 2). Conversely, increasing the CWD threshold from the 95th to the 99th percentile decreased TE forest extent from 387 000 to 135 000 ha (7 per cent of all forest), constraining the map to the driest TE forest sites (Supplementary Figure 2, Supplementary Table 2). In addition, although fire extent varied substantially among CWD percentile maps (scaling with TE extent), the distribution of burn severity varied to a lesser degree (e.g. 2 per cent more and 9 per cent less high-severity fire for the 91st and 95th percentiles, respectively; Supplementary Table 2).

Future research will refine our ability to develop vulnerability maps to support land management and policy. In terms of our TE forest mapping and characterization analysis (Objectives 1 and 2), the univariate approach of delineating TE forests based on current forest and current and future CWD could be refined based on evaluation of the TE map sensitivity to different future climate scenarios (i.e. a broader range of RCPs and time periods). Comparison with other TE maps (e.g. Parks *et al.,* 2019) and simulation-based models of future vegetation change would provide additional lines of evidence to determine locations with the highest potential disturbance vulnerability (e.g. Shafer *et al.,* 2015; Case *et al.,* 2020). In addition, because the uncertainty of GNN-based maps of forest type and structure increases at fine scales (Ohmann *et al.,* 2012), alternative land cover datasets and LiDAR-based structure maps or field-based surveys of forest conditions would further refine our understanding of the variability of TE and non-TE forests. Finally, our analysis depends on the accuracy of the current forest cover map and the assumption that the current forest/non-forest ecotone is driven primarily by biophysical drivers like climate rather than land use history.

In terms of our fire mapping analysis (Objectives 3 and 4), alternative classifications of burn severity could reveal slightly different patterns of high-severity locations in TE forests, but the relatively low extent of fire in TE forests to date mitigates this uncertainty. Rather than assess actual fires retrospectively, future studies could incorporate quantitative fire risk maps that integrate fire probability and expected fire behaviour given current fuel conditions in different forest and non-forest cover types (Dunn *et al.,* 2020). Fire risk maps also could be combined with social vulnerability maps (e.g. Davies *et al.,* 2018) to highlight locations where ecological and social values at risk overlap. These overlapping locations may provide important opportunities for managers and conservation practitioners to engage with disproportionately impacted land owners and communities. Another key avenue for future research is targeted field sampling of post-fire tree regeneration in high-severity patches of varying size and time since fire (e.g. Donato *et al.,* 2016; Kemp *et al.,* 2019). Finally, although this study quantified recent fire patterns in TE forests at risk of fire-facilitated transformation, additional information is needed regarding fire's cumulative effects on the ecological and social values in TE and non-TE forests, particularly as fire occurrence and reburn increase over time.

Management implications

In seasonally dry forests of western North America, forest management objectives on most public and some other lands are currently focused on restoring the structure and resilience functions associated with historical, active fire regimes (Hagmann *et al.,* 2021). Climate adaptation and managing for a 'future range of variability' is also a key goal but is only beginning to be fully integrated into management planning (Hessburg *et al.,* 2019). The results of this study clearly demonstrate the need to anticipate and manage for future vs historical climate in a large portion of the eastern Cascades. As such, the innovative RAD framework (Lynch *et al.,* 2021; Schuurman *et al.,* 2022) offers a useful approach for adaptive management of TE forests. Before fires occur, it is essential to prioritize locations for maintaining current forest cover (i.e. resistance) *vs* passively accepting or proactively directing transitions to novel forest or non-forest habitats. Here, we provide example management strategies for TE forests to illustrate potential applications of the RAD framework, recognizing that specific options should be co-produced with land managers and partners and that multiple strategies, well-designed experiments, and flexibility will continue to be essential.

- **Resist ecological transformation to novel forest types or non-forest cover types**. Reduce canopy density and fuel abundance through ecologically appropriate thinning, prescribed burning or managed wildfire to increase resistance to drought and fire, especially in places with large, old trees. Favour existing drought- and fire-tolerant species (e.g. ponderosa pine, Douglas-fir, western larch). Thin to low densities to reduce tree competition and increase tree vigour (North *et al.,* 2022) while also retaining variable patterns of individual trees, clumps and openings (Churchill *et al.,* 2013) that can increase resistance and resilience to disturbance (Koontz *et al.,* 2020; Hagmann *et al.,* 2021). Such resistance strategies are most viable in relatively mesic TE forests, but interventions in dry TE forests could enable canopy tree persistence even when drought stress exceeds seedling physiological tolerances.
- **Accept transitions driven by high-severity wildfires compounded by drought**. After a disturbance, carefully evaluate whether reforestation is likely to be successful at that site or if conversion to novel forest or non-forest is very likely based on forest type and high-severity patch size. If planting is desired, plant at low densities using site-appropriate species and strategies to maximize survival (Meyer *et al.,* 2021; Stevens *et al.,* 2021). Reforestation is most likely to be successful in relatively mesic TE forests (Stevens-Rumann and Morgan, 2019), whereas forest conversions may be more acceptable in dry TE forests.
- **Direct transitions to new vegetation types.** In some sites, particularly, in the dry TE forest zone, intensively thin and retain large ponderosa pine trees to convert dense forest to open woodland structure (e.g. 10–20 per cent canopy cover), representing a new but more disturbance-tolerant structural condition. In other sites, consider planting drought-tolerant, non-conifer species (e.g. Oak (*Quercus spp.*)) at woodland densities or assisted migration of suitable conifer species where appropriate, especially following high-severity fire. In places where shrub and grassland communities are most appropriate, manage invasive grasses to direct transitions towards native shrub-steppe species that favour longer fire return intervals.

We emphasize that ecological transformations can have both positive and negative outcomes for different habitat types and associated values, and management objectives should integrate site-specific conditions with historical and projected future ranges of variability. Many of today's forested areas historically supported lower tree densities or non-forest vegetation types (e.g. herbland and shrub-steppe), which play a vital role in landscape patterns, fire flow and resilience (Hessburg *et al.,* 2019). TE landscapes are inherently dynamic, and the current mosaic of tree and non-tree conditions depends on climate, soils, disturbance and land use history. Moreover, species adapted to dry woodland habitats, such as Oregon white oak (*Quercus garryana* Douglas ex Hook.) may expand into the study area (McKenney *et al.,* 2007), which would provide critical habitat for species like the western gray squirrel *(Sciurus griseus)* while also restoring alignment between vegetation and the topoclimatic environment. Thus, adaptation strategies and forest regulations should be flexible enough to accommodate a range of potential pre- and post-fire management activities, including directing transitions to novel forest, low-density woodland and non-forest cover types. This flexibility will be essential to allow land managers to realize the potential benefits of wildfire, including fire managed for resource and cultural benefits (Lake *et al.,* 2017; Meyer *et al.,* 2021; Churchill *et al.,* 2022; Larson *et al.,* 2022; WADNR, 2022). As these landscapes continue to change, maps of TE forest variability and fire effects will help decision makers and partners prioritize resources for fire management and post-fire reforestation while supporting the long-term objectives of landscape restoration, community wildfire protection and climate adaptation.

Although fire-prone forests are intrinsically dynamic and resilient, recent and projected patterns of drought, wildfire and tree mortality (e.g. Millar and Stephenson, 2015; McNellis *et al.,* 2021) underscore the urgency of adaptive management actions and policies. This study demonstrates that TE forests are both widespread and variable in their composition, structure and ownership, indicating that management strategies will require cooperation among a wide range of partners and stakeholders. Collaborative natural resource management efforts have been increasing in the study area (e.g. WADNR, 2020), but establishing and sustaining zones of agreement regarding landscape-scale implementation of forest management projects (including post-fire tree planting) remains challenging (Hagmann *et al.,* 2021), especially across dynamic landscapes with multiple public and private land owners. These challenges are amplified by the reality that long-term trajectories may include permanent transitions from forest to non-forest conditions, which could be controversial for some community members. Nevertheless, it is critical to incorporate projected changes, equitable partnerships and culturally appropriate actions into planning efforts that will influence management activities and landscape dynamics for decades (e.g. landscape-scale NEPA analysis or US National Forest Plan revisions).

Conclusion

As temperatures, drought stress and disturbance activity continue to increase, western North American landscapes are increasingly vulnerable to forest loss and transformation (Davis *et al.,* 2019; Coop *et al.,* 2020), especially when drought increases wildfire severity and inhibits tree regeneration in TE forests, where trees live near their physiological limits (Kemp *et al.,* 2019; Parks *et al.,* 2019). At the same time, many currently forested landscapes have densified over the past century due to fire exclusion and land use, with conifer forests often replacing native woodland and non-forest land cover types (Hessburg *et al.,* 2019). Given the many social-ecological values that are susceptible to drought and wildfire in TE forests, it will be increasingly important to determine where and when disturbance-induced changes represent tipping points of ecosystem decline or the restoration of historical patterns and processes. This study improves our understanding of TE forests in the eastern Cascades of Washington by characterizing their composition, structure, ownership and recent fire activity. As TE forests continue to burn, it will be essential to monitor and adapt to changes in forest conditions and associated landscape resilience goals across multiple spatiotemporal scales.

Supplementary data

Supplementary data are available at *Forestry* online.

Data availability

The data underlying this article will be shared on reasonable request to the corresponding author.

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Conflict of interest statement

None declared.

References

Agee, J.K. 2003 Historical range of variability in eastern Cascades forests, Washington, USA. *Landsc. Ecol.* **18**, 725–740. https://doi.org/10.1023/B:LAND.0000014474.49803.f9.

Ager, A.A., Palaiologou, P., Evers, C.R., Day, M.A., Ringo, C. and Short, K. 2019 Wildfire exposure to the wildland urban interface in the western US. *Appl. Geogr.* **111**, 102059. https://doi.org/10.1016/j.apgeog.2019.102059.

Anderegg, W.R., Trugman, A.T., Badgley, G., Anderson, C.M., Bartuska, A., Ciais, P. *et al.* 2020 Climate-driven risks to the climate mitigation potential of forests. *Science* **368**, eaaz7005. https://doi.org/10.1126/science.aaz7005.

Buotte, P.C., Law, B.E., Ripple, W.J. and Berner, L.T. 2020 Carbon sequestration and biodiversity co-benefits of preserving forests in the western United States. *Ecol. Appl.* **30**, e02039. https://doi.org/10.1002/eap.2039.

Cansler, C.A., Kane, V.R., Hessburg, P.F., Kane, J.T., Jeronimo, S.M., Lutz, J.A. *et al.* 2022 Previous wildfires and management treatments moderate subsequent fire severity. *For. Ecol. Manag.* **504**, 119764. https://doi.org/10.1016/j.foreco.2021.119764.

Cansler, C.A. and McKenzie, D. 2014 Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern Cascade Range, USA. *Ecol. Appl.* **24**, 1037–1056. https://doi.org/10.1890/13-1077.1.

Case, M.J., Johnson, B.G., Bartowitz, K.J. and Hudiburg, T.W. 2021 Forests of the future: climate change impacts and implications for carbon storage in the Pacific Northwest, USA. *For. Ecol. Manag.* **482**, 118886. https://doi.org/10.1016/j.foreco.2020.118886.

Case, M.J., Kim, J.B. and Kerns, B.K. 2020 Using a vegetation model and stakeholder input to assess the climate change vulnerability of tribally important ecosystem services. *Forests* **11**, 618. https://doi.org/10.3390/f11060618.

Case, M.J. and Lawler, J.J. 2017 Integrating mechanistic and empirical model projections to assess climate impacts on tree species distributions in northwestern North America. *Glob. Chang. Biol.* **23**, 2005–2015. https://doi.org/10.1111/gcb.13570.

Churchill, D.J., Jeronimo, S.M., Hessburg, P.F., Cansler, C.A., Povak, N.A., Kane, V.R. *et al.* 2022 Post-fire landscape evaluations in Eastern Washington, USA: assessing the work of contemporary wildfires. *For. Ecol. Manag.* **504**, 119796. https://doi.org/10.1016/j.foreco.2021.119796.

Churchill, D.J., Larson, A.J., Dahlgreen, M.C., Franklin, J.F., Hessburg, P.F. and Lutz, J.A. 2013 Restoring forest resilience: from reference spatial patterns to silvicultural prescriptions and monitoring. *For. Ecol. Manag.* **291**, 442–457. https://doi.org/10.1016/j.foreco.2012.11.007.

Coop, J.D., Parks, S.A., Stevens-Rumann, C.S., Crausbay, S.D., Higuera, P.E., Hurteau, M.D. *et al.* 2020 Wildfire-driven forest conversion in western North American landscapes. *Bioscience* **70**, 659–673. https://doi.org/10.1093/biosci/biaa061.

Curtis, R.O. and Marshall, D.D. 2000 Why quadratic mean diameter? *West. J. Appl. For.* **15**, 137–139. https://doi.org/10.1093/wjaf/15.3.137.

Davies, I.P., Haugo, R.D., Robertson, J.C. and Levin, P.S. 2018 The unequal vulnerability of communities of color to wildfire. *PLoS One* **13**, e0205825. https://doi.org/10.1371/journal.pone.0205825.

Davis, K.T., Dobrowski, S.Z., Higuera, P.E., Holden, Z.A., Veblen, T.T., Rother, M.T. *et al.* 2019 Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. *Proc. Natl. Acad. Sci.* **116**, 6193–6198. https://doi.org/10.1073/pnas.1815107116.

Davis, K.T., Higuera, P.E., Dobrowski, S.Z., Parks, S.A., Abatzoglou, J.T., Rother, M.T. *et al.* 2020 Fire-catalyzed vegetation shifts in ponderosa pine and Douglas-fir forests of the western United States. *Environ. Res. Lett.* **15**, 1040b1048.

Dobrowski, S.Z., Abatzoglou, J., Swanson, A.K., Greenberg, J.A., Mynsberge, A.R., Holden, Z.A. *et al.* 2013 The climate velocity of the contiguous United States during the 20th century. *Glob. Chang. Biol.* **19**, 241–251. https://doi.org/10.1111/gcb.12026.

Donato, D.C., Harvey, B.J. and Turner, M.G. 2016 Regeneration of montane forests 24 years after the 1988 yellowstone fires: a fire-catalyzed shift in lower treelines? *Ecosphere* **7**, e01410. https://doi.org/10.1002/ecs2.1410.

Dunn, C.J., D, O'Connor, C., Abrams, J., Thompson, M.P., Calkin, D.E., Johnston, J.D. *et al.* 2020 Wildfire risk science facilitates adaptation of fire-prone social-ecological systems to the new fire reality. *Environ. Res. Lett.* **15**, 025001. https://doi.org/10.1088/1748-9326/ab6498.

ESRI 2018 *ArcGIS Desktop: Release 10*. Redlands, CA: Environmental Systems Research Institute.

Everett, R.L., Schellhaas, R., Keenum, D., Spurbeck, D. and Ohlson, P. 2000 Fire history in the ponderosa pine/Douglas-fir forests on the east slope of the Washington Cascades. *For. Ecol. Manag.* **129**, 207–225. https://doi.org/10.1016/S0378-1127(99)00168-1.

Franklin, J.F. and Dyrness, C.T. 1973 *Natural Vegetation of Oregon and Washington Gen. Tech. Rep. PNW-GTR-8*. USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. USDA Forest Service, Portland, OR, pp. 1–452.

Guisan, A. and Zimmermann, N.E. 2000 Predictive habitat distribution models in ecology. *Ecol. Model.* **135**, 147–186. https://doi.org/10.1016/S0304-3800(00)00354-9.

Hagmann, R.K., Hessburg, P.F., Prichard, S.J., Povak, N.A., Brown, P.M., Fule, P.Z. *et al.* 2021 Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests. *Ecol. Appl.* **31**, 1–34.

Hijmans, R.J., van Etten, J., Sumner, M., Cheng, J., Bevan, A., Bivand, R. *et al.* 2021 *Raster: Geographic Data Analysis and Modeling*. R package version 3.4-10. Available from: https://CRAN.R-project.org/package=raster.

Haugo, R.D., Kellogg, B.S., Cansler, C.A., Kolden, C.A., Kemp, K.B., Robertson, J.C. *et al.* 2019 The missing fire: quantifying human exclusion of wildfire in Pacific Northwest forests, USA. *Ecosphere* **10**, e02702. https://doi.org/10.1002/ecs2.2702.

Hessburg, P.F., Miller, C.L., Parks, S.A., Povak, N.A., Taylor, A.H., Higuera, P.E. *et al.* 2019 Climate, environment, and disturbance history govern resilience of western North American forests. *Front. Ecol. Evol.* **7**, 239. https://doi.org/10.3389/fevo.2019.00239.

Hessburg, P.F., Salter, R.B. and James, K.M. 2007 Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. *Landsc. Ecol.* **22**, 5–24. https://doi.org/10.1007/s10980-007-9098-2.

Hessburg, P.F., Smith, B.G., Salter, R.B., Ottmar, R.D. and Alvarado, E. 2000 Recent changes (1930s-1990s) in spatial patterns of interior northwest forests, USA. *For. Ecol. Manag.* **136**, 53–83. https://doi.org/10.1016/S0378-1127(99)00263-7.

Infrastructure Investment and Jobs Act 2021 *Public Law 117–58, Sect. 70302 (REPLANT Act: Repairing Existing Public Land by Adding Necessary Trees Act)*.

IPCC (Intergovernmental Panel on Climate Change) 2013 In *Climate Change 2013: The Physical Science Basis*. T.F., Stocker, D., Qin, G.-K., Plattner, M., Tignor, S.K., Allen, J., Boschung *et al.* (eds.). Cambridge University Press, Cambridge, UK.

Kemp, K.B., Higuera, P.E., Morgan, P. and Abatzoglou, J.T. 2019 Climate will increasingly determine post-fire tree regeneration success in low-elevation forests, Northern Rockies, USA. *Ecosphere* **10**, e02568.

Koontz, M.J., North, M.P., Werner, C.M., Fick, S.E. and Latimer, A.M. 2020 Local forest structure variability increases resilience to wildfire in dry western US coniferous forests. *Ecol Lett* **23**, 483–494. https://doi.org/10.1111/ele.13447.

Lake, F.K., Wright, V., Morgan, P., McFadzen, M., McWethy, D. and Stevens-Rumann, C. 2017 Returning fire to the land: celebrating traditional knowledge and fire. *J. For.* **115**, 343–353. https://doi.org/10.5849/jof.2016-043R2.

Larson, A.J., Jeronimo, S.A., Hessburg, P.F., Lutz, J.A., Povak, N.A., Cansler, C.A. *et al.* 2022 Ecological principles to guide post-fire forest landscape management in the Inland Pacific and Northern Rocky mountain regions. *For. Ecol. Manag.* **504**Special Issue, 119680. https://doi.org/10.1016/j.foreco.2021.119680.

Lynch, A.J., Thompson, L.M., Beever, E.A., Cole, D.N., Engman, A.C., Hawkins Hoffman, C. *et al.* 2021 Managing for RADical ecosystem change: applying the Resist-Accept-Direct (RAD) framework. *Front. Ecol. Environ.* **19**, 461–469. https://doi.org/10.1002/fee.2377.

Lynn, K., Daigle, J., Hoffman, J., Lake, F., Michelle, N., Ranco, D. *et al.* 2013 The impacts of climate change on tribal traditional foods. In *Climate Change and Indigenous Peoples in the United States*. New York: Springer, pp. 37–48.

McKenney, D.W., Pedlar, J.H., Lawrence, K., Campbell, K. and Hutchinson, M.F. 2007 Potential impacts of climate change on the distribution of North American trees. *Bioscience* **57**, 939–948. https://doi.org/10.1641/B571106.

McNellis, B.E., Smith, A.M., Hudak, A.T. and Strand, E.K. 2021 Tree mortality in western US forests forecasted using forest inventory and Random Forest classification. *Ecosphere* **12**, e03419.

Meigs, G.W., Campbell, J.L., Zald, H.S.J., Bailey, J.D., Shaw, D.C. and Kennedy, R.E. 2015 Does wildfire likelihood increase following insect outbreaks in conifer forests? *Ecosphere* **6**, art118. https://doi.org/10.1890/ES15-00037.1.

Meigs, G.W. and Krawchuk, M.A. 2018 Composition and structure of forest fire refugia: what are the ecological legacies across burned landscapes? *Forests* **9**, 243. https://doi.org/10.3390/f9050243.

Meyer, M.D., Long, J.W. and Safford, H.D. 2021 *Postfire Restoration Framework for National Forests in California*. Gen. Tech. Rep. PSW-GTR-270. Vol. **204**. US Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA, p. 270.

Millar, C.I. and Stephenson, N.L. 2015 Temperate forest health in an era of emerging megadisturbance. *Science* **349**, 823–826. https://doi.org/10.1126/science.aaa9933.

Miller, J.D. and Thode, A.E. 2007 Quantifying burn severity in a heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). *Remote Sens. Environ.* **109**, 66–80. https://doi.org/10.1016/j.rse.2006.12.006.

North, M.P., Tompkins, R.E., Bernal, A.A., Collins, B.M., Stephens, S.L. and York, R.A. 2022 Operational resilience in western US frequent-fire forests. *For. Ecol. Manag.* **507**, 120004. https://doi.org/10.1016/j.foreco.2021.120004.

Ohmann, J.L., Gregory, M.J., Roberts, H.M., Cohen, W.B., Kennedy, R.E. and Yang, Z. 2012 Mapping change of older forest with nearest-neighbor imputation and Landsat time-series. *For. Ecol. Manag.* **272**, 13–25. https://doi.org/10.1016/j.foreco.2011.09.021.

O'Neil, T.A., Bettinger, K.A., Vander Heyden, M., Marcot, B., Barrett, C., Mellen, T.K. *et al.* 2001 *Structural conditions and habitat elements of Oregon and Washington*. Wildlife habitats and relationships in Oregon and Washington. OSU Press, Corvallis, Oregon, p. 115–139.

Parks, S.A., Dobrowski, S.Z., Shaw, J.D. and Miller, C. 2019 Living on the edge: trailing edge forests at risk of fire-facilitated conversion to non-forest. *Ecosphere* **10**, e02651. https://doi.org/10.1002/ecs2.2651.

Parks, S.A., Holsinger, L.M., Voss, M.A., Loehman, R.A. and Robinson, N.P. 2018 Mean composite fire severity metrics computed with Google Earth Engine offer improved accuracy and expanded mapping potential. *Remote Sensing* **10**, 879. https://doi.org/10.3390/rs10060879.

Pearson, R.G. and Dawson, T.P. 2003 Predicting the impacts of climate change on the distribution of species: are bioclimate envelope models useful? *Glob. Ecol. Biogeogr.* **12**, 361–371. https://doi.org/10.1046/j.1466-822X.2003.00042.x.

Povak, N.A., Churchill, D.J., Cansler, C.A., Hessburg, P.F., Kane, V.R., Kane, J.T. *et al.* 2020 Wildfire severity and postfire salvage harvest effects on long-term forest regeneration. *Ecosphere* **11**, e03199.

Prichard, S.J., Povak, N.A., Kennedy, M.C. and Peterson, D.W. 2020 Fuel treatment effectiveness in the context of landform, vegetation, and large, wind-driven wildfires. *Ecol. Appl.* **30**, e02104. https://doi.org/10.1002/eap.2104.

Priestley, C.H.B. and Taylor, R.J. 1972 On the assessment of surface heat flux and evaporation using large-scale parameters. *Mon. Weather Rev.* **100**, 81–92. https://doi.org/10.1175/1520-0493(1972)100<0081:OTAOSH>2.3.CO;2.

R Core Team 2021 *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria, Available from: https://www.R-project.org/.

Ramsfield, T., Bentz, B., Faccoli, M., Jactel, H. and Brockerhoff, E. 2016 Forest health in a changing world: effects of globalization and climate change on forest insect and pathogen impacts. *Forestry* **89**, 245–252. https://doi.org/10.1093/forestry/cpw018.

Reilly, M.J., Dunn, C.J., Meigs, G.W., Spies, T.A., Kennedy, R.E., Bailey, J.D. *et al.* 2017 Contemporary patterns of fire extent and severity in forests of the Pacific Northwest, USA (1985–2010). *Ecosphere* **8**, 1–28.

Riahi, K., Rao, S., Krey, V., Cho, C., Chirkov, V., Fischer, G. *et al.* 2011 RCP 8.5—A scenario of comparatively high greenhouse gas emissions. *Clim. Chang.* **109**, 33–57. https://doi.org/10.1007/s10584-011-0149-y.

Saberi, S.J. and Harvey, B.J. 2022 in review after revision What is the color when black is burned? Quantifying forest (re)burn severity in short-interval fires using field and satellite indices. *Fire Ecology*.

Schuurman, G.W., Cole, D.N., Cravens, A.E., Covington, S., Crausbay, S.D., Hoffman, C.H. *et al.* 2022 Navigating ecological transformation: resist–accept–direct as a path to a new resource management paradigm. *Bioscience* **72**, 16–29. https://doi.org/10.1093/biosci/biab067.

Seaber, P.R., Kapinos, F.P. and Knapp, G.L. 1987 *Hydrologic Unit Maps: US Geological Survey Water Supply Paper 2294*. US Geological Survey.

Shafer, S.L., Bartlein, P.J., Gray, E.M. and Pelltier, R.T. 2015 Projected future vegetation changes for the northwest United States and southwest Canada at a fine spatial resolution using a dynamic global vegetation model. *PloS one* **10**, e0138759. https://doi.org/10.1371/journal.pone.0138759.

Stephens, S.L., Collins, B.M., Fettig, C.J., Finney, M.A., Hoffman, C.M., Knapp, E.E. *et al.* 2018 Drought, tree mortality, and wildfire in forests adapted to frequent fire. *Bioscience* **68**, 77–88. https://doi.org/10.1093/biosci/bix146.

Stephenson, N. 1998 Actual evapotranspiration and deficit: biologically meaningful correlates of vegetation distribution across spatial scales. *J. Biogeogr.* **25**, 855–870. https://doi.org/10.1046/j.1365-2699.1998.00233.x.

Stevens-Rumann, C.S. and Morgan, P. 2019 Tree regeneration following wildfires in the western US: a review. *Fire Ecol.* **15**, 1–17.

Stevens, J.T., Haffey, C.M., Coop, J.D., Fornwalt, P.J., Yocom, L., Allen, C.D. *et al.* 2021 Tamm review: postfire landscape management in frequent-fire conifer forests of the southwestern United States. *For. Ecol. Manag.* **502**, 119678. https://doi.org/10.1016/j.foreco.2021.119678.

Urza, A.K., Weisberg, P.J. and Dilts, T. 2020 Evidence of widespread topoclimatic limitation for lower treelines of the Intermountain West. *U. S. Ecol. Appl.* **30**, e02158. https://doi.org/10.1002/eap.2158.

UW Climate Impacts Group (UW Department of Environmental and Occupational Health Sciences, Front and Centered and Urban@UW) 2018. *An Unfair Share: Exploring the Disproportionate Risks from Climate Change Facing Washington State Communities. Ae Report Prepared for Seattle Foundation*. University of Washington, Seattle, WA. Available from: https://cig.uw.edu/wp-content/uploads/sites/2/2018/08/AnUnfairShare_WashingtonState_August2018.pdf.

van Mantgem, P.J., Stephenson, N.L., Byrne, J.C., Daniels, L.D., Franklin, J.F., Fulé, P.Z. *et al.* 2009 Widespread increase of tree mortality rates in the western United States. *Science* **323**, 521–524. https://doi.org/10.1126/science.1165000.

WADNR (Washington State Department of Natural Resources) 2020 *Forest Health Assessment and Treatment Framework. RCW 76.06.200*, **268**. Olympia, WA. Available from: https://www.dnr.wa.gov/publications/rp_2020_fh_report.pdf.

WADNR (Washington State Department of Natural Resources) 2022 *Wildfire Season 2021—Work of Wildfire Assessment*. Washington State Department of Natural Resources. Olympia, WA. Available from: https://www.dnr.wa.gov/publications/rp_workofwildfire2021_march2022.pdf.

Wang, T., Hamann, A., Spittlehouse, D. and Carroll, C. 2016 Locally downscaled and spatially customizable climate data for historical and future periods for North America. *PLoS One* **11**, e0156720. https://doi.org/10.1371/journal.pone.0156720.

Williams, A.P., Cook, B.I. and Smerdon, J.E. 2022 Rapid intensification of the emerging southwestern North American megadrought in 2020–2021. *Nat. Clim. Chang.* **12**, 1–3. https://doi.org/10.1038/s41558-022-01290-z.

Young, D.J., Stevens, J.T., Earles, J.M., Moore, J., Ellis, A., Jirka, A.L. *et al.* 2017 Long-term climate and competition explain forest mortality patterns under extreme drought. *Ecol. Lett.* **20**, 78–86. https://doi.org/10.1111/ele.12711.