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Implications of recent wildfires for forest management on federal lands in the Pacific Northwest, USA

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ABSTRACT

Adoption of the Northwest Forest Plan (NWFP) in 1994 marked a pivotal moment in federal forest management in the Pacific Northwest, shifting focus away from intensive timber harvest toward an ecosystem management approach that emphasized late successional and old forest habitat with the creation of a reserve network across moist and dry forest zones. Thirty years after implementation, concerns over accelerating wildfire threats have prompted efforts to adapt the Plan to a warming climate, yet the actual effects of recent fires on NWFP forests are not well understood. In this study, we evaluated over 2200 fires that have burned in the NWFP area over the last four decades to inform conservation efforts and Plan amendments. We quantified patterns and drivers of fire severity across different land use allocations and major forest zones within the NWFP. We found that annual area burned and mean high severity patch size increased across the study area, and historically frequent-fire forest types experienced the most severe wildfire effects. Although moist forest types were less affected by wildfire than dry forests, we observed large-scale forest cover loss in late successional reserves. Weather was a prominent driver of fire severity across much of the region, but bottom-up influences including vegetation type, topography, and pre-fire forest structure exerted strong controls outside of large high severity patches. Our results present a comprehensive analysis of wildfire effects across the NWFP, providing context for future Plan amendments and climate adaptation strategies.

1. Introduction

Globally, forests are changing rapidly as wildfire seasons grow longer and more severe (Flannigan et al., 2013; Parks and Abatzoglou, 2020) and forests are further challenged by severe drought (Dai, 2013; Swain, 2015), widespread insect outbreaks (Raffa et al., 2008), expansion of the wildland urban interface (Radeloff et al., 2018), and resource extraction (Laurance et al., 2000). In response to accelerating threats to forests, dominant conservation strategies throughout much of the last century aimed to protect late successional and old-growth (LSOG) forest habitat through the designation of wilderness areas and reserves (Massip, 2020). While forest reserves play a strong role in curbing forest losses to development and commercial exploitation (Talty et al., 2020), a warming climate and shifting disturbance regimes have continued to reshape forest landscapes and erode LSOG forests (Jones et al., 2025; Seidl et al., 2017). As climate change is expected to amplify existing forest stressors in the coming decades (Abatzoglou et al., 2021; Cook et al., 2018; Keenan, 2015), there is significant need to assess the role of static reserve systems in achieving forest conservation goals (Bengtsson et al., 2003; Hessburg et al., 2021; North et al., 2015). To sustain forest habitat in an era of rapid environmental change, informing adaptive management strategies is critically important (Prichard et al., 2021; Wildland Fire Mitigation and Management Commission, 2023).

In the Pacific Northwest, the Northwest Forest Plan (hereafter, the NWFP or the Plan) directs the management of federal forests across nearly 10 million ha in Washington, Oregon, and northern California. Implementation of the Plan in 1994 marked a pivotal moment following the ‘timber wars’ – intense conflict between conservation groups and timber companies over the fate of remaining old growth forests – and decades of debate about the primary role of public land management in the region (Johnson et al., 2023; Winkel, 2014). Unprecedented in its geographic scale and complexity, the Plan represented the most

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ambitious forest management, conservation, and monitoring effort ever implemented for the US national forest system and shifted focus away from intensive timber harvest toward ecosystem management that emphasized late-successional forest habitat for threatened and endangered species (Johnson et al., 2023; Thomas et al., 2006). A network of forest reserves strategically located across both moist and fire-frequent dry forest zones was the primary strategy to maintain and restore forest habitat of species such as the northern spotted owl (*Strix occidentalis caurina*) and marbled murrelet (*Brachyramphus marmoratum*).

Across the NWFP, management guidance for federal forests is determined by a set of land use allocations developed as part of the Plan. A range of management strategies are represented across allocations, ranging from Congressional Reserves (lands designated by US Congress, including wilderness areas and national parks where active forest management is restricted) to Matrix lands (non-reserved forests where most silvicultural and timber harvest activities were expected to occur). Of particular importance to the Plan's conservation strategy are forests allocated as Late Successional Reserves (LSRs), where management emphasizes the protection and enhancement of LSOG conditions (Johnson et al., 2023). While potential impacts from natural disturbances such as wildfire were considered as part of the NWFP's reserve design – particularly in the dry forest zone – the effects of climate change on the frequency, size, and severity of recent wildfires were not anticipated in the Plan (Gaines et al., 2022; Spies et al., 2019). Three decades after its adoption, ongoing efforts to amend the Plan have aimed to address the effects of climate change and implement adaptation strategies to better sustain old forests and the cultural significance, economic and resource values, habitat, and carbon sequestration they provide (US Forest Service, 2023).

Wildfire is currently the driving agent of changing forest conditions across the diverse landscapes of the NWFP (Davis et al., 2015, 2022). Increasingly large and severe recent fires have rapidly reshaped fire-prone dry forests of the region, consistent with broader trends observed across western North America (Cansler and McKenzie, 2014; Cova et al., 2023; Harvey et al., 2016; Parks and Abatzoglou, 2020; Reilly et al., 2017; Steel et al., 2018). The unprecedented scale of large, stand-replacing patches observed within these fires presents critical challenges to the regeneration and persistence of future forests, particularly in a warming climate (Coop et al., 2020; Davis et al., 2019). Moist forests within the region have likewise recently burned in large and severe fires, though these events are generally more consistent with historical fire regimes of the moist forest zone (Reilly et al., 2022).

Given current wildfire impacts to forests across western North America, forest conservation strategies need to account for the effects of recent wildfires and anticipated trends under warmer and drier conditions. This is pertinent across a range of post-fire effects, including areas where stand-replacing fire may challenge reforestation and catalyze conversion to non-forest cover (Coop et al., 2020; North et al., 2019), in areas where forests are maintained through more frequent wildfires with low and mixed severity ecological effects (Spies et al., 2006), and in patches of unburned fire refugia (Meddens et al., 2018). Analysis of trends and drivers (i.e., fuels, topography, and weather) of wildfire effects on forests can be used to inform future conservation and management strategies that consider recent forest loss, prioritize stewardship of remaining forests, and anticipate post-fire forest recovery in a future with more frequent fire (Abatzoglou et al., 2021; Dye et al., 2024).

In this study, we evaluated recent wildfires within the NWFP area to inform conservation efforts and recommendations for Plan adaptations. We analyzed burn severity data for over 2200 fires that have burned across the region over the last four decades to quantify patterns and drivers of fire severity across different land use allocations and major forest zones. Our study was guided by three central research questions: 1) What have been the ecological effects of recent wildfires (in terms of area burned at different burn severities) on NWFP forests across land use allocations and forest zones?; 2) Within the NWFP area, what are the primary drivers (in terms of fire weather, fuels, and topography) of fire severity across land use allocations and forest zones?; and 3) What are the management and policy implications of these trends?

2. Methods

2.1. Study area

We evaluated fires that burned at least partially within the administrative boundaries of the NWFP across forests in Washington, Oregon, and northern California (Fig. 1). Forests in this region span diverse gradients of climate and topography and include a broad range of forest types from coastal rainforests to historically frequent-fire dry mixed-conifer forests and pine-oak woodlands (Hessburg et al., 2019). We evaluated fires across two broad physiographic regions as defined in the NWFP: a moist forest zone encompassing forests west of the Cascade Mountain crest and along the coast of northern California, and a dry forest zone comprised of the forests east of the Cascade Mountain crest and within the Klamath Mountain ecoregion of southwestern Oregon and interior northern California (Franklin and Johnson, 2012).

The moist forest zone of the NWFP is largely characterized by high-biomass, highly productive conifer forests and rainforests dominated by western hemlock (*Tsuga heterophylla*), Douglas-fir (*Pseudotsuga menziesii*), western red cedar (*Thuja plicata*), and Sitka spruce (*Picea sitchensis*), with abundance of bigleaf maple (*Acer macrophyllum*), black cottonwood (*Populus trichocarpa*), and red alder (*Alnus rubra*) (Franklin and Dyrness, 1973). Along the coast in northern California, long-lived coastal redwoods (*Sequioia sepervirens*) are present. At higher elevations, cold forests are dominated by Pacific silver fir (*Abies amabilis*), mountain hemlock (*Tsuga mertensiana*), and subalpine fir (*Abies lasiocarpa*). Although moist conifer, moist mixed conifer, and temperate rainforests comprise much of the moist forest zone area, drier mixed conifer forests and mixed evergreen forests are common in warmer, drier locations of the moist forest zone. Riparian forests are locally abundant in wetlands and along streams and rivers. The moist forest zone is broadly characterized by mild, wet winters and a pronounced summer dry season. Large fires in this region were historically infrequent (Agee, 1996; Weisberg and Swanson, 2003), but when large fires did occur – often under extreme dry east wind events – high severity fire effects were common (Reilly et al., 2022). Though the region is often described as an infrequent fire regime, low- to moderate-severity fires associated with Indigenous stewardship and microclimatic variation were widespread and historically frequent (Boyd, 1999; Lorimer et al., 2009; Merschel et al., 2024). Along with lightning ignitions, Indigenous stewardship practices including fuel harvesting and cultural burning played an important role in the development of local- to landscape-scale patches of forest and non-forest vegetation such as meadows and grasslands (Charnley et al., 2017; Kimmerer and Lake, 2001; Lorimer et al., 2009).

The dry forest zone of the NWFP is dominated by semi-arid mixed conifer forests – particularly in the Klamath ecoregion – with areas of cold forests, mixed evergreen forests, and moist mixed conifer forests present across the region. Riparian forests, pine-oak woodlands, pine savannas, and oak woodlands are also present in the region but are less abundant. In the southern Cascades, mixed conifer forests contain Jeffrey pine (*Pinus jeffreyi*), incense cedar (*Calocedrus decurrens*), sugar pine (*Pinus lambertiana*), and white fir (*Abies concolor*) (Skinner and Taylor, 2018). From the central Cascades of Oregon to northern Washington state, dry zone mixed conifer forests are dominated by variable assemblages of fire-tolerant Douglas-fir, ponderosa pine (*Pinus ponderosa*), and western larch (*Larix occidentalis*), intermixed with aspen (*Populus tremuloides*) and grand fir (*Abies grandis*) on relatively moist sites (Franklin and Dyrness, 1973; Sorenson, 2012). At higher elevations, forests dominated by mountain hemlock are prevalent in the north, with red fir (*Abies magnifica*) common in the south. The dry forest zone is the more frequent-fire region of the NWFP; historical fire regimes were broadly characterized by low- to moderate-severity fire effects dominated by

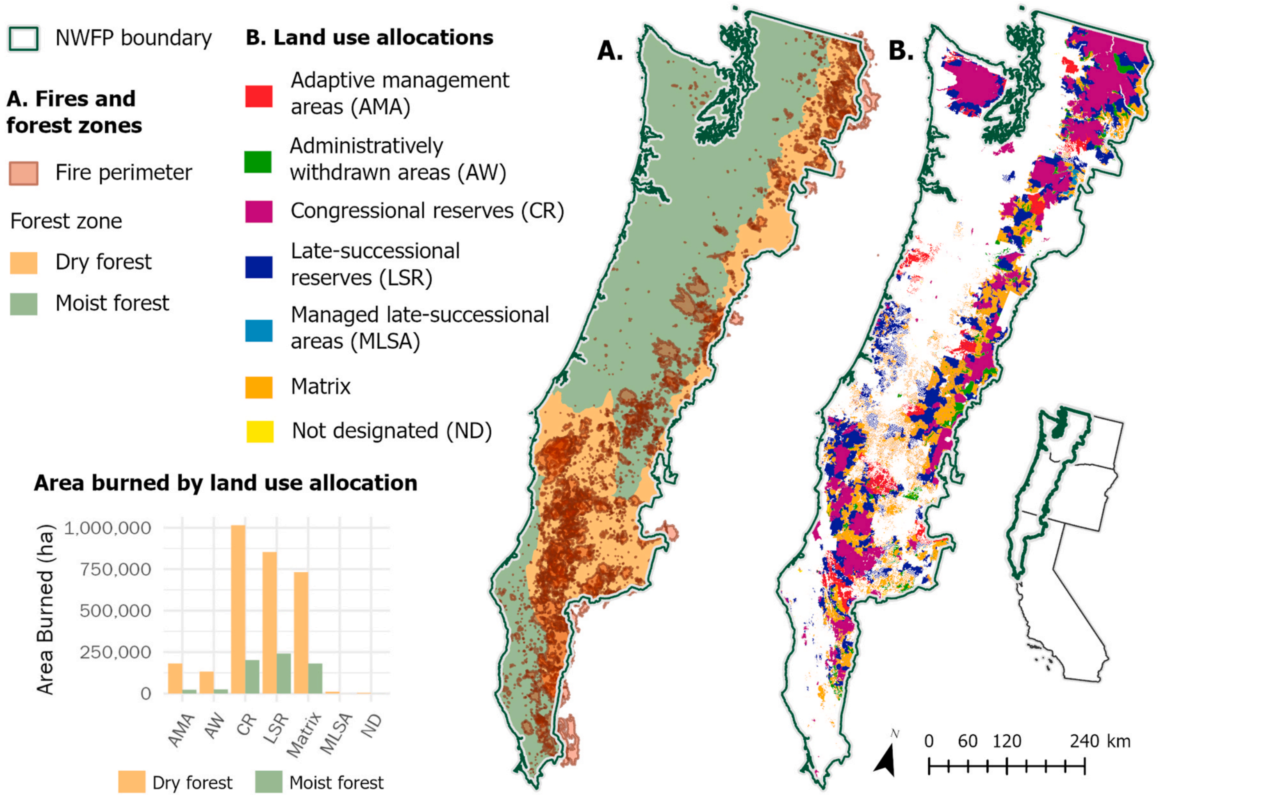

Fig. 1. Map of the study area spanning the bounds of the Northwest Forest Plan (NWFP) across federal lands in Washington, Oregon, and northern California, USA. Map A (left): major physiographic forest zones and over 2200 wildfires that have burned between 1985 and 2022 at least partially within the bounds of the NWFP. Map B (right): federal land use allocations designated within the NWFP. Total area burned by land use allocation and major forest zone (dry versus moist) is shown in the bottom left. The majority of area burned occurred in congressional reserves (CR), late successional reserves (LSR), and Matrix designations.

frequent (less than 35 year) return intervals and mixed-severity fires with fire return intervals less than 75 years (Agee, 1996; Hessburg et al., 2016; Perry et al., 2011).

The NWFP recognized a potential need for different management strategies and reserve design between forest zones (Franklin and Johnson, 2012). In moist forest zone LSRs, wildfire impacts were generally not considered, but management activities such as forest thinning to accelerate old forest structural characteristics and promote ecological diversity in young (< 80 years) second-growth forests were allowed (U. S. Forest Service USFS, Bureau of Land Management BLM, 1994). In the dry forest zone, LSRs were designed as larger, more contiguous areas than in the moist forest zone to maintain a baseline level of wildlife habitat connectivity while providing some redundancy to the potential wildfire effects that could erode forest cover (Johnson et al., 2023). The Plan recognized that active management (referred to as risk-reduction treatments), including forest thinning and prescribed burning, would be required to maintain ecological function in frequent-fire dry forest landscapes. While risk-reduction treatments within dry forest zone LSRs were intended in the Plan, rates of restoration treatments such as thinning and burning are generally below the levels needed to maintain resilience of dry forest landscapes (Franklin and Johnson, 2012; Gaines et al., 2022). Implementing adaptive management strategies throughout the Plan area has proven challenging (Gaines et al., 2022; Spies et al., 2018c), though recent efforts to amend the Plan have proposed actions in both moist and dry forests aimed at improving wildfire resilience and climate adaptation (US Forest Service, 2023).

2.2. Fire perimeters

We compiled a dataset of fire perimeters from datasets maintained by the California Department of Forest and Fire Protection (CAL FIRE) Fire and Resource Assessment Program, the Washington Department of Natural Resources, and the National Interagency Fire Center. We identified all recorded fires that burned between 1985 and 2022 within the administrative boundaries of the Plan, retaining all fires greater than 4 ha to minimize potential data entry errors and ensure each burn severity image contained a sufficient number of pixels to analyze spatial patterns of burn severity. Although the NWFP was adopted in 1994, we chose to evaluate severity for all possible fires in the modern Landsat satellite record – back to 1985 – to provide a broader assessment of how forests within the NWFP have fared following wildfire. A total of 2254 fires met our criteria; 352 fires burned prior to 1994 representing 7 % of the total area burned across the study period.

2.3. Patterns of fire severity

We generated a Landsat-derived burn severity image for each of the 2254 fires in our dataset using a methodology developed by Parks et. al (2019) in Google Earth Engine (Gorelick et al., 2017). The workflow produces a 30-m resolution predicted Composite Burn Index (CBI) image for a given fire by combining the Relativized Burn Ratio (Parks et al., 2014) – a spectral index used to measure burn severity developed from pre- and post-fire Landsat imagery – with climatic variables, latitude, and other spectral indices such as the Normalized Difference Vegetation Index and Mid-Infrared Bi-Spectral Index in a Random Forest model (Breiman, 2001) calibrated by over 8000 field sampling plots. We chose to use predicted CBI to examine patterns of burn severity as it is a more ecologically interpretable measure of post-fire vegetation change compared to unitless spectral indices such as RBR or the Relativized delta Normalized Burn Ratio (RdNBR – Miller and Thode, 2007). For further ecological interpretability, continuous predicted CBI values were classified into categories using established thresholds: unburned/very low severity - CBI values below 0.1; low severity - values 0.1–1.25; moderate severity - values 1.25–2.25; and high severity - values greater than 2.25 (Miller and Thode, 2007). Because CBI is a field-based sampling protocol developed to evaluate fire severity in forests (Key and Benson, 2006), all non-forested pixels were removed from our burn severity images using a forest capability mask originally developed as part of the NWFP Monitoring Program (Ohmann et al., 2012). Additionally, because we were interested specifically in federal forests managed as part of the NWFP, we excluded non-federal lands from our analysis.

Patterns of fire severity have important implications for a variety of ecosystem processes, including potential post-fire successional dynamics, loss of forest cover and wildlife habitat, and tree regeneration following stand-replacing fire (Collins et al., 2017b; Stevens et al., 2017; Stevens-Rumann and Morgan, 2019). To evaluate the ecological effects of recent wildfires on NWFP forests, we evaluated patterns of fire severity using four landscape metrics: total area burned by severity class, core area of high severity fire patches, mean patch size of contiguous areas of high severity and combined low and unburned (unburned-low) patches, and connectivity of unburned-low patches (Table 1). Across the NWFP, we observed the majority of area burned within Congressional Reserves (CRs), LSRs, and Matrix designations (Fig. 1); because of this, we focused our analysis on these three land use allocations. All landscape metrics were calculated using the *landscapemetrics* package in R (Hesselbarth et al., 2019).

Class Area - We calculated total area burned by severity class (unburned/very low, low, moderate, and high severity) to assess the ecological effects of recent wildfires in forests of the NWFP. Fire severity was evaluated across the full study period by land use allocation (CR, LSR, and Matrix), forest zone (dry versus moist), and major forest type based on LANDFIRE Biophysical Settings potential vegetation types (Supplemental Table S1, Rollins and Frame, 2006). We additionally evaluated trends in annual area burned by land use allocation and forest zone. Trends were tested for statistical significance using Theil-Sen (T-S) slope estimators – a nonparametric technique to evaluate the median slope across a time series – via the ‘trend’ package in R (Pohlert, 2019). Following previous studies, we evaluated the statistical significance of trends using a p-value of 0.10 (Cova et al., 2023; Dennison et al., 2014; Holden et al., 2018; Parks and Abatzoglou, 2020).

Patch Size - We evaluated trends in area-weighted mean annual patch size by forest zone using an 8-cell neighborhood to define a patch. Area-weighted means weight each patch by their proportional contribution to the total area of all patches, and are generally preferred for ecological interpretations over arithmetic mean patch size as they better reflect the largest patches present on the landscape (Li and Archer, 1997). Patch size trends were assessed for the high severity and unburned-low classes, and tested for statistical significance using T-S slope estimators. Because large patches often span designations, we did not evaluate patch size by land use allocation. We focused on high severity burn patches because they are associated with high (> 75 %) tree mortality, can have strong effects on forest recovery and successional dynamics, and influence wildlife habitats (Coop et al., 2020; Jones et al., 2020). We grouped unburned-low severity pixels into patches (corresponding to areas that have likely experienced < 25 % overstory tree mortality, Miller and Thode, 2007) as they serve important functions as biological legacies (Johnstone et al., 2016; Meddens et al., 2018), habitat refugia (Robinson et al., 2013), and are an important component of restoring fire-resilient structure and composition in dry forests (Becker and Lutz, 2016; Hood et al., 2015). Moderate severity pixels were not considered within patch analyses because they tend to have mixed effects and represent relatively high uncertainty in burn severity classifications (Furniss et al., 2020).

Core Area - The interior core area of high severity patches is often used as a proxy for understanding where forest regeneration may be threatened following wildfire due to distance from live seed sources at the patch edge (Collins et al., 2017a, 2017b; Stevens et al., 2017). We evaluated the core area of high severity patches by forest zone in CRs, LSRs, and Matrix designations to understand how recent wildfires may influence post-fire successional dynamics. Because a single high severity patch may span multiple designations, this metric represents the amount of core area within a given land use allocation, and not necessarily the size of the entire patch core. We define core area as the interior of a high severity patch at least 120 m from the patch edge, where wind-driven seed dispersal for non-serotinous and relatively heavy-seed species such as ponderosa pine becomes unlikely (Clark et al., 1999). Because seed dispersal distances vary widely and can exceed 120 m for tree species with wind-borne seeds (Laughlin et al., 2023), we recognize our core area threshold as a conservative proxy for identifying areas where tree regeneration may be challenged, particularly in moist forests.

Connectivity - Late successional reserves of the NWFP were arranged to provide a network of habitats for wildlife and plant species associated with LSOG forests, including the Northern Spotted Owl (NSO) (Johnson et al., 2023). In the dry forest zone, LSRs were designated in larger areas than in the moist forest zone to provide redundancy in anticipation of fires that may erode forest cover within reserves. Connectivity of unburned-low patches can inform where habitat may persist following fire, and how it may have shifted over time. To evaluate post-fire patterns of habitat connectivity, we calculated an aggregation index for unburned-low patches of both dry and moist forest zone LSRs and tested for statistical significance of annual trends in aggregation using T-S slope estimators. The aggregation index is a unitless metric that measures the number of within-class patch adjacencies divided by the theoretical maximum possible number of adjacencies for that class (He et al., 2000; McGarigal and Marks, 1995).

2.4. Drivers of fire severity - datasets

We evaluated drivers of fire severity as a function of predictor variables representing fuels, topography, and weather by forest zone separately in CRs, LSRs, and Matrix designations (Table 2). Because

Table 1
Landscape metrics calculated for 2254 fires to evaluate the ecological effects of recent wildfires on NWFP forests. Metrics were calculated by land use allocation and major forest zone (dry versus moist). Table adapted from Singleton et al. (2021) and Cova et al. (2023).

Metric	Description	Interpretation of low values	Interpretation of high values	Units	Range
Class Area	*Area burned:* Total area belonging to severity class *i*.	Less area burned	More area burned	Ha	Class Area ≥ 0
Patch Size	*Area-weighted mean patch size:* Measure of mean patch size for class *i*. Only calculated for high-severity and combined low and unburned (unburned-low) patches.	Generally smaller patch sizes with few or no large patches	Generally larger patch sizes or few large patches among many smaller patches	Ha	Patch Size ≥ 0
Core Area	*Total core area:* Total core area of class *i* > 120 m from patch edge. Only calculated for high-severity class.	Less interior area burned	More interior area burned	Ha	Core Area ≥ 0
Connectivity	*Aggregation Index:* The number of like adjacencies of patches for class *i* divided by the theoretical maximum possible number of like adjacencies for that class. Only calculated for unburned-low patches.	Disaggregated patches with lower landscape-level connectivity.	Aggregated patches with higher landscape-level connectivity.	None / Index	100 ≥ Connectivity ≥ 0

Table 2
Predictor variables used to assess drivers of fire severity by forest zone (dry versus moist) and land use allocation (Congressional Reserves, Late Successional Reserves, and Matrix designations).

Category	Variable	Source	Resolution
Topography	Topographic position index (TPI) fine - 270 m	(Evans and Murphy, 2021)	30 m
	TPI coarse - 2070 m	(Evans and Murphy, 2021)	30 m
	Slope	(Farr et al., 2007; Gorelick et al., 2017)	30 m
	Aspect	(Farr et al., 2007; Gorelick et al., 2017)	30 m
	Elevation	(Farr et al., 2007; Gorelick et al., 2017)	30 m
	Heat load index	(Evans and Murphy, 2021)	30 m
Forest structure	Cover type	(Rollins and Frame, 2006); see Table S1 within Supplement for crosswalk	30 m
	Stand age of dominant trees (Stand age)	(Ohmann and Gregory, 2002)	30 m
	Component Ratio Method biomass of live trees (Biomass)	(Ohmann and Gregory, 2002)	30 m
	Canopy cover	(Ohmann and Gregory, 2002)	30 m
	Diameter diversity index (DDI)	(Ohmann and Gregory, 2002)	30 m
	Old growth structural index (OGSI)	(Ohmann and Gregory, 2002)	30 m
	Snag volume (Snag)	(Ohmann and Gregory, 2002)	30 m
	Time since most recent disturbance (Time since)	(Healey et al., 2018)	30 m
	Most recent disturbance type (Disturbance)	(Healey et al., 2018)	30 m
Weather	Wind velocity (Wind)	(Abatzoglou, 2013)	30 m
	Vapor pressure deficit (VPD)	(Abatzoglou, 2013)	30 m
	Energy resource component (ERC)	(Abatzoglou, 2013)	30 m

many of our predictor variables included datasets derived from satellites with shorter temporal time spans or coarser spatial resolutions than Landsat (i.e., daily fire progression maps used to acquire weather variables are derived from MODIS satellite data and are not available before 2000), we constrained our models to the 407 large fires (> 500 ha) that burned between 2001 and 2021. We used continuous fire severity values from the Relativized Burn Ratio (Parks et al., 2014) as the response variable in our models rather than predicted CBI, as predicted CBI values are derived from models that already include climate and site moisture predictors (Parks et al., 2019), which could violate the assumption of independence in our analysis.

Topographic datasets on elevation, slope, aspect, heat load index, and topographic position index (TPI) were generated for each fire derived from a 30-m digital elevation model (Farr et al., 2007). Slope and aspect were calculated using the *terra* package in R (Hijmans et al., 2024). Heat load index was calculated using the *spatialEco* package in R (Evans and Murphy, 2021) and derived from slope, aspect, and latitude following McCune and Keon (2002), where values near 0 represent cooler and wetter pixels and values near 1 represent warmer and drier pixels. TPI was calculated as the difference between the elevation of a given pixel and the mean surrounding elevation within a moving window surrounding the pixel; we calculated two separate TPI variables using a fine- (270 m) and coarse- (2070 m) scale moving window using the *spatialEco* package (Evans and Murphy, 2021).

We obtained variables on pre-fire canopy cover, biomass per hectare, stand age of dominant trees, snag volume, diameter diversity index, and old growth structural index (OGSI) for the year before each fire developed as part of the NWFP Monitoring Program using the gradient nearest neighbor method integrating Forest Inventory Analysis plots, spectral data, topographic data, and climate data (Ohmann et al., 2012; Ohmann and Gregory, 2002). Using a change attribution dataset developed by the Landscape Change Monitoring System (Healey et al., 2018), we mapped the most recent disturbance type prior to each fire (one of fire, harvest, insect/drought stress, other, or no detected disturbance) and years since the most recent disturbance. LCMS datasets are developed from spectral changes and are only available for the modern Landsat record; where no disturbance was detected since 1985, we used stand age from GNN to attribute years since the most recent disturbance. Finally, we incorporated a predictor variable for major forest type by grouping LANDFIRE Biophysical Settings potential vegetation types into broad categories of vegetation (Supplemental Table S1, Rollins and Frame, 2006).

Information on daily fire weather was obtained by first producing MODIS-derived interpolated day-of-burn maps for each fire using a methodology developed by Parks (Parks, 2014), then acquiring gridded surface meteorological data from GRIDMET (Abatzoglou, 2013) for the corresponding day and area burned. We used this workflow to produce 30-m maps of daily wind velocity, vapor pressure deficit (VPD), and energy resource component (ERC) for each fire. VPD is calculated as the difference between the amount of moisture in the atmosphere and the amount of maximum moisture it can hold and has strong effects on wildfire behavior (Abatzoglou and Williams, 2016), and ERC is a composite fuel moisture index and can be used to gauge fuel dryness.

2.5. Drivers of fire severity – statistical modeling

To minimize effects of short-distance spatial autocorrelation in our statistical models, we extracted mean response and predictor variables within a 3 × 3 pixel window on a grid of points spaced 270 m apart across the whole study area (Kane et al., 2015). We used tree-based Random Forest (RF) machine learning algorithms within the *ranger* package in R (Wright et al., 2023) to model relationships between our subsampled predictor datasets and RBR response variable. Separate RF models were constructed for each forest zone (dry versus moist) and land use allocation (CR, LSR, and Matrix designations) for a total of six models. For each model, we first evaluated predictor variable importance by running RF with all 18 predictor variables (Table 2) and calculating the percent increase in mean squared error (MSE) for each predictor variable present in the model. We then applied a variable selection for interpretation workflow implemented in the Variable Selection Using Random Forests (*VSURF*) package in R to refine model predictors (Genuer et al., 2015). A final set of RF models were run using only the selected predictors from the variable selection step and evaluated using out-of-bag error. We ran all RF models at each step using 1000 bootstrapped samples in which one-third of the predictor variables were randomly selected and evaluated at each node split of the decision tree. We visualized relationships between individual predictor and response variables for each model using partial dependence plots. Lastly, we used Shapley additive explanation (SHAP) values to explore the local importance of our predictors at the individual pixel-level. SHAP values are based on cooperative game theory and, for each sample, quantify the impact of each predictor variable (positive or negative) on the response. To contextualize our results, we created categorical maps of the most influential driver (greatest magnitude SHAP value, positive or negative) of severity at the pixel-level for a representative set of case-study fires. SHAP values were calculated using the *treeShap* package in R (Komisarczyk et al., 2024).

3. Results

3.1. *What have been the ecological effects of recent wildfires on NWFP forests across land use allocations and forest zones?*

Between 1985 and 2022, over 3.6 million hectares of forest burned within the NWFP area, with most of that (90 %) occurring in CRs, LSRs, and Matrix land allocations (Table 3). Within each of these allocations, the dry forest zone accounted for 4–5 times the area burned than the moist forest zone, with the greatest difference observed in CRs. The greatest fire activity was observed in dry forest CRs and dry forest LSRs – 67.5 % (1,015,009 ha) of all dry forest CR area and 59.4 % (853,469 ha) of dry forest LSR area burned over the study period. Over the 37-year study period, over one-fifth (22 %, 330,850 ha) of the total dry forest CR area – including burned and unburned area – experienced high severity fire effects, and 19 % of the total dry forest LSR area (273,816 ha) experienced high severity fire. Of the moist forest zone land use allocations, the most area burned was in LSRs at 16.9 % (241,083 ha). However, Matrix lands experienced the most severe proportional wildfire impacts in the moist first zone, with 5.4 % of the total Matrix land area burning as high severity fire (60,197 ha).

From 1985–2022, annual area burned increased in each forest zone and land use allocation, with the greatest area having burned in the last decade (Fig. 2). In dry forests, fire activity was observed across the full study period. By contrast, much less fire activity was observed in moist zone forests from 1985 to 2015, followed by a pronounced increase in annual area burned between 2015 and 2022. Temporal trends in annual area burned were statistically significant in each forest zone and land use allocation per T-S slope estimators, though trends in moist forest zone LSRs were marginally significant.

Area burned in the moist forest zone was concentrated in moist mixed conifer, moist conifer/rainforest, and cold forest cover types (Fig. 3, Supplemental Table S1), but represented an overall small proportion of the total forested area relative to the dry forest zone. In moist forest zone LSRs, for example, wildfires burned 132,564 acres in moist mixed conifer forests dominated by Douglas-fir, western hemlock, and Pacific silver fir but accounted for only 26 % of the total moist mixed conifer area within the allocation. In moist forest zone CRs and Matrix lands, less than 20 % of the total area of moist conifer/rainforest and moist mixed conifer forest types burned over the study period, and less than 7 % of this area burned with high severity effects.

Area burned was distributed over a wide variety of forest types across the land use allocations in the dry forest zone (Fig. 3). In dry forest zone CRs, we observed large (> 40,000 ha) extents of area burned in the Cascades dry mixed conifer, nwCA mixed conifer, cold forest, nwCA mixed evergreen, and 'other' cover types (where 'other' was dominated by high elevation barren rock, shrubland, and grassland, Supplemental Table S1). Dry forest LSRs contained large areas burned in cold forests, moist mixed conifer, Cascades dry mixed conifer, nwCA mixed evergreen, and nwCA mixed conifer cover types. In dry forest Matrix land, large area burned was observed in the Cascades mixed conifer, nwCA mixed evergreen, and nwCA mixed conifer forest types. The greatest

Table 3
Area burned by severity class (in hectares) by land use allocation and forest zone. Values in the "% total area burned" columns describe the proportion of area burned in each severity class as a function of total burned area - e.g., in dry forest zone Congressional Reserves, 27.5 % of the total area burned yielded low severity effects. The next column ("% total CR area") describes the proportion of area burned in each severity class as a function of the total available area (both burned and unburned) in that forest zone and allocation - e.g., in dry forest zone Congressional Reserves, 18.5 % of the entire area within dry forest zone Congressional Reserves burned with low severity effects. Table continues on the next page.

Congressional Reserves (CRs)									
	Dry forest zone			Moist forest zone			All CRs		
	Area burned (ha)	% total area burned	% total CR area	Area burned (ha)	% total area burned	% total CR area	Area burned (ha)	% total area burned	% total CR area
Unburned/ Very Low	89,369	8.8	5.9	23,696	11.7	1.4	113,065	9.3	3.5
Low	278,636	27.5	18.5	46,771	23.2	2.8	325,407	26.7	10.2
Moderate	316,154	31.1	21.0	52,467	26.0	3.1	368,621	30.3	11.5
High	330,850	32.6	22.0	78,906	39.1	4.7	409,756	33.7	12.8
Total Area Burned	**1,015,009**	**-**	**67.5**	**201,839**	**-**	**11.9**	**1,216,848**	**-**	**38.1**
Total CR Area	**1,503,937**			**1,689,768**			**3,193,705**		
Late Successional Reserves (LSRs)									
	Dry forest zone			Moist forest zone			All LSRs		
	Area burned (ha)	% total area burned	% total LSR area	Area burned (ha)	% total area burned	% total LSR area	Area burned (ha)	% total area burned	% total LSR area
Unburned/ Very Low	64,179	7.5	4.5	40,317	16.7	2.8	104,496	9.5	3.6
Low	240,742	28.2	16.7	80,670	33.5	5.6	321,412	29.4	11.2
Moderate	274,732	32.2	19.1	64,508	26.8	4.5	339,240	31.0	11.8
High	273,816	32.1	19.0	55,589	23.0	3.9	329,405	30.1	11.5
Total Area Burned	**853,469**	**-**	**59.4**	**241,083**	**-**	**16.9**	**1,094,552**	**-**	**38.2**
Total LSR Area	**1,437,790**			**1,430,048**			**2,867,838**		
Matrix									
	Dry forest zone			Moist forest zone			All Matrix		
	Area burned (ha)	% total area burned	% total Matrix area	Area burned (ha)	% total area burned	% total Matrix area	Area burned (ha)	% total area burned	% total Matrix area
Unburned/ Very Low	47,140	6.4	3.0	31,836	17.6	2.9	78,977	8.6	3.0
Low	169,894	23.2	11.0	45,520	25.1	4.1	215,415	23.6	8.1
Moderate	248,895	34.0	16.1	43,771	24.1	3.9	292,666	32.0	11.0
High	266,472	36.4	17.2	60,197	33.2	5.4	326,668	35.8	12.3
Total Area Burned	**732,401**	**-**	**47.4**	**181,325**	**-**	**16.3**	**913,726**	**-**	**34.4**
Total Matrix Area	**1,545,594**			**1,110,071**			**2,655,665**		

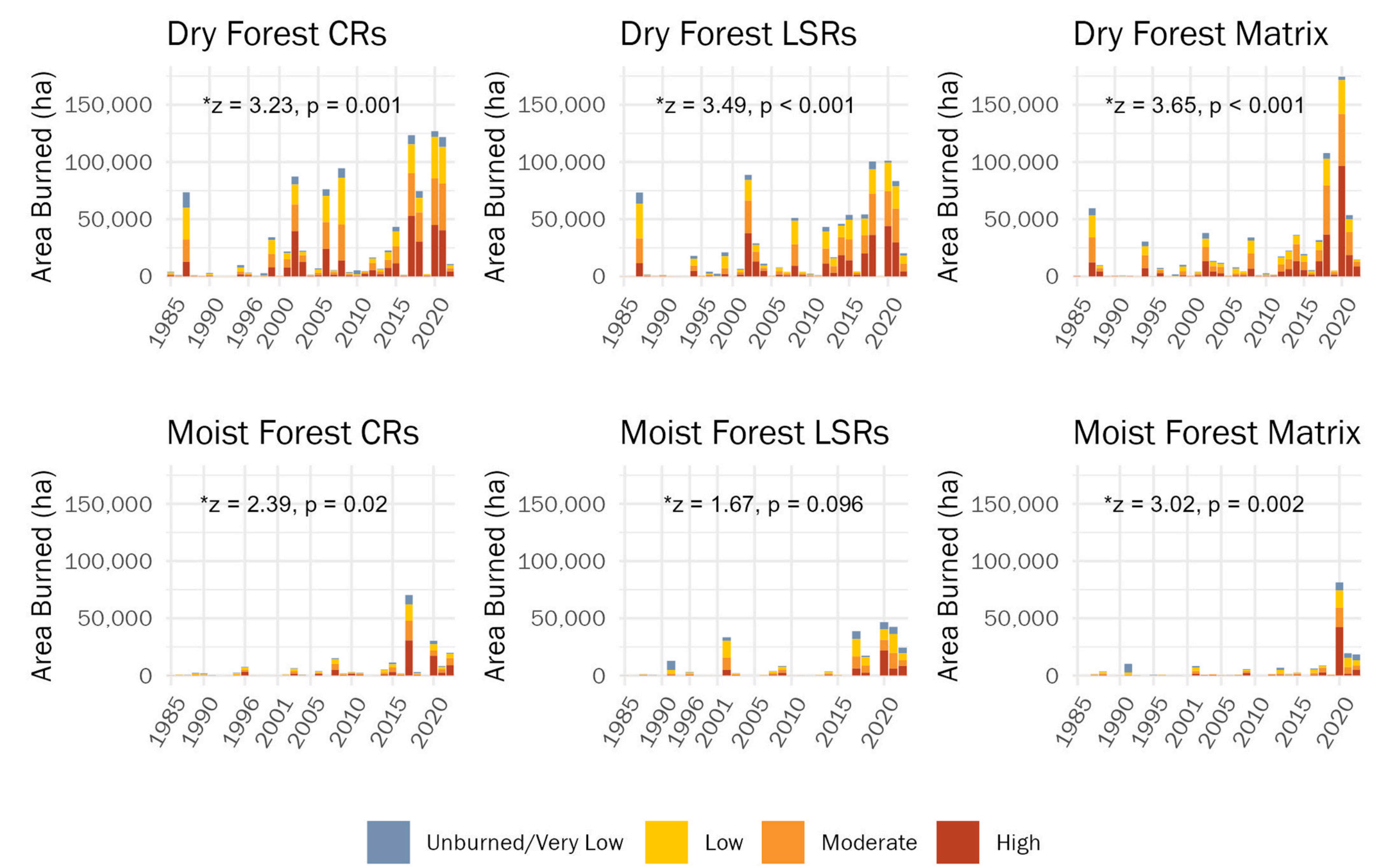

Fig. 2. Trends in annual area burned by severity class in dry forest zone (top row) and moist forest zone (bottom row) across land use allocations. CRs - Congressional Reserves, leftmost column; LSRs - Late Successional Reserves, middle column; Matrix - Matrix land designation, rightmost column. Z statistic and p-value printed on plots represent model outputs from Theil-Sen slope estimators to assess statistically significant trends in annual area burned. Asterisks (*) represent plots with statistically significant trends. Statistical significance was assessed at $p < 0.1$ following previous studies (Dennison et al., 2014; Holden et al., 2018; Parks and Abatzoglou, 2020).

extent of high severity area burned in each dry forest zone allocation was the nwCA mixed conifer cover type, which burned 146,701 ha in Matrix, 135,659 ha in CRs, and 134,068 ha in LSRs.

We observed the greatest high severity impacts in pine-oak woodlands, oak woodlands, pine forests, nwCA mixed conifer, nwCA mixed evergreen, and Cascades dry mixed conifer forest types (Fig. 3). Pine-oak woodlands in dry forest zone CRs experienced the greatest proportional impacts, where 61 % of their total area within the allocation (both burned and unburned) burned at high severity. Over half (54 %) of all oak woodlands within dry forest zone CRs burned with high severity effects, and 41 % of nwCA mixed conifer extent in the allocation burned at high severity. In dry forest zone LSRs, oak woodlands had the greatest proportional impacts (32 % of the total extent burned at high severity), followed by nwCA mixed conifer (26 % burned at high severity) and Cascades dry mixed conifer (25 % burned at high severity). In Matrix lands, 25 % of all oak woodlands burned with high severity effects, and 24 % of nwCA mixed conifer and 22 % of nwCA mixed evergreen burned at high severity.

Mean high severity patch size increased over the study period in both dry and moist forest zones (Fig. 4A). T-S model fits indicated a nearly 4-fold increase in dry forest zone mean high severity patch size (from 40.1 ha in 1985–154.1 ha in 2022), and a 6-fold increase in the moist forest zone (6.12 ha in 1985–39.79 ha in 2022). Both trends were statistically significant. In the moist forest zone, annual mean size of unburned-low severity patches (Fig. 4B) significantly increased over the study period (from a predicted mean patch size of 5.3 ha in 1985–51.8 ha in 2022 per T-S models). There was no discernible trend in annual mean size of unburned-low severity patches in the dry forest zone over the study period. In dry forest zone LSRs, unburned-low severity patches grew increasingly disaggregated over the study period (Fig. 4C). There were no significant trends in aggregation of unburned-low severity patches in moist forest zone LSRs.

Across the study area, high severity interior core area was distributed in many small patches with relatively few large patches (Fig. 5). Dry forest zone allocations overall contained both a greater number of patches and larger cumulative extent of high severity core area than moist forest zone allocations. Moist forest zones had generally wider core area patch size distributions (i.e., a relatively greater proportion of large patches) than dry forest zones. In all forest zones and allocations, a relatively small number of the largest patches of core area accounted for the greatest cumulative area burned.

3.2. Within the NWFP area, what are the primary drivers of fire severity across land use allocations and forest zones?

Our variable selection workflow retained between 6 and 9 predictors for each final RF model (Fig. 6). Weather variables (wind, VPD, and ERC) were the most important predictors of fire severity (RBR) in all models except for the dry forest zone CR model, in which cover type, elevation, biomass, and canopy cover were more important than weather. Wind was associated with the greatest increase in model MSE across the study area (116 % in the moist forest Matrix model). Elevation was the second most important variable driving severity in the dry forest CR model, and the most important predictor after weather variables in all other models. Cover type was an important predictor in the dry forest CR (increasing MSE by 29.9 %), dry forest LSR (16.3 %), moist

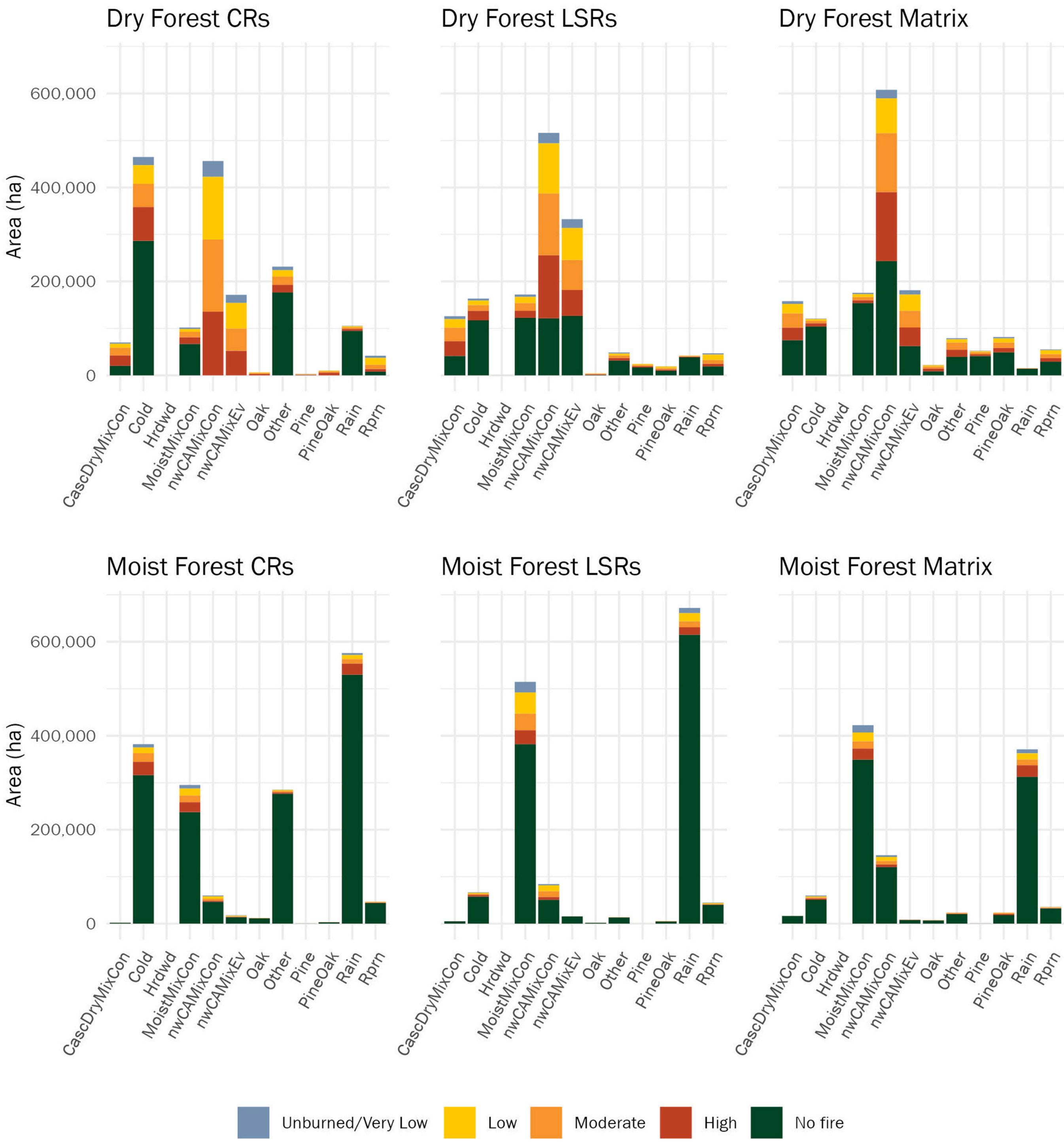

Fig. 3. Area burned by severity class within each forest type by dry (top row) and moist (bottom row) forest zones and land use allocations (CRs: Congressional Reserves, left column, LSRs: Late Successional Reserves, middle column; Matrix: Matrix land allocation, right column). Dark green represents the area within that forest type that has not burned at all; the remaining colors are symbolized by severity class. CascDryMixCon - dry mixed conifer forests within the Cascade mountain range; Cold - cold forests; Hrdwd - hardwood forests; MoistMixCon - moist mixed conifer forests; nwCAMixCon - mixed conifer forests in northwestern California; nwCAMixEv - mixed evergreen forests in northwestern California; Oak - oak woodlands, Other - non-forest or other forest; Pine - pine forests and savannas; PineOak - pine-oak woodlands, Rain - temperate rainforests; Rprn - riparian forests. For a full breakdown of these forest type groups, see Supplemental Table S1.

forest CR (35 %), and moist forest Matrix models (13.3 %). Time since the last pre-fire disturbance, biomass, and canopy cover were also important predictors, but order of importance varied by model. Coarse-scale TPI was selected as a final variable only in moist forest zone models, increasing MSE by 13.4 % in moist CR, 12.1 % in moist LSR, and 12.9 % in moist Matrix models. DDI was only selected in the final model for moist forest CRs, increasing MSE by 17.6 %. Aspect, fine scale TPI, most recent disturbance type, OGSI, heat load, and stand age were generally less important and were not selected in final models. RF models explained 55 % of the variability in RBR across moist forest zone CRs, 44 % in moist forest zone LSRs, 52 % in moist forest zone Matrix, 39 % in dry forest zone CRs, 43 % in dry forest zone LSRs, and 50 % in

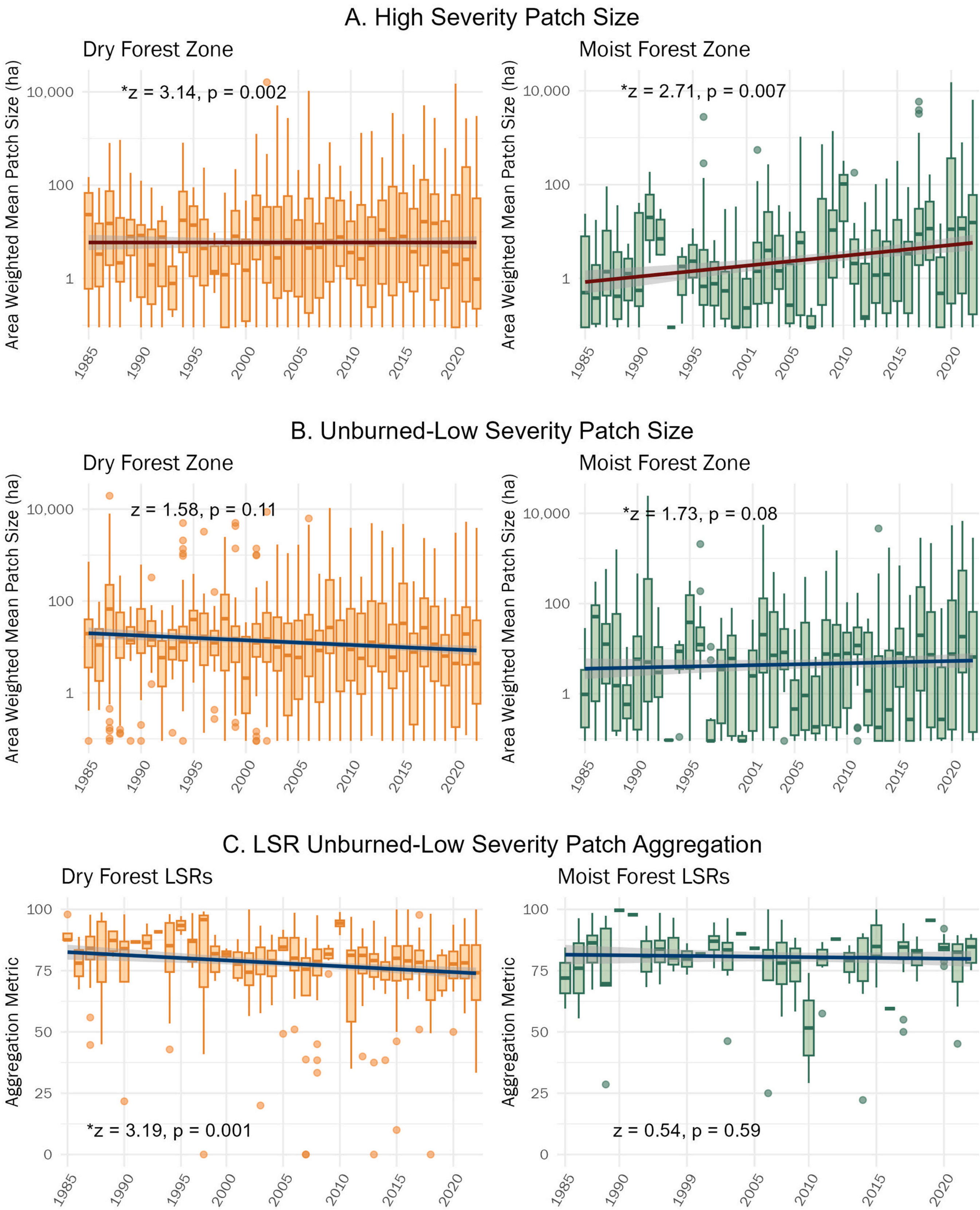

Fig. 4. Patch configurations in dry forest zone (left column) and moist forest zone (right column) fires. Panel A (top row): Annual mean high severity patch size (area-weighted), regardless of land use allocation; Panel B (middle row): Annual mean size (area-weighted) of unburned and low severity patches combined, regardless of land use allocation; Panel C (bottom row): Annual mean aggregation index of unburned and low severity patches combined in LSRs only. High aggregation index values indicate patches that are more aggregated; low index values indicate patches that are more isolated.

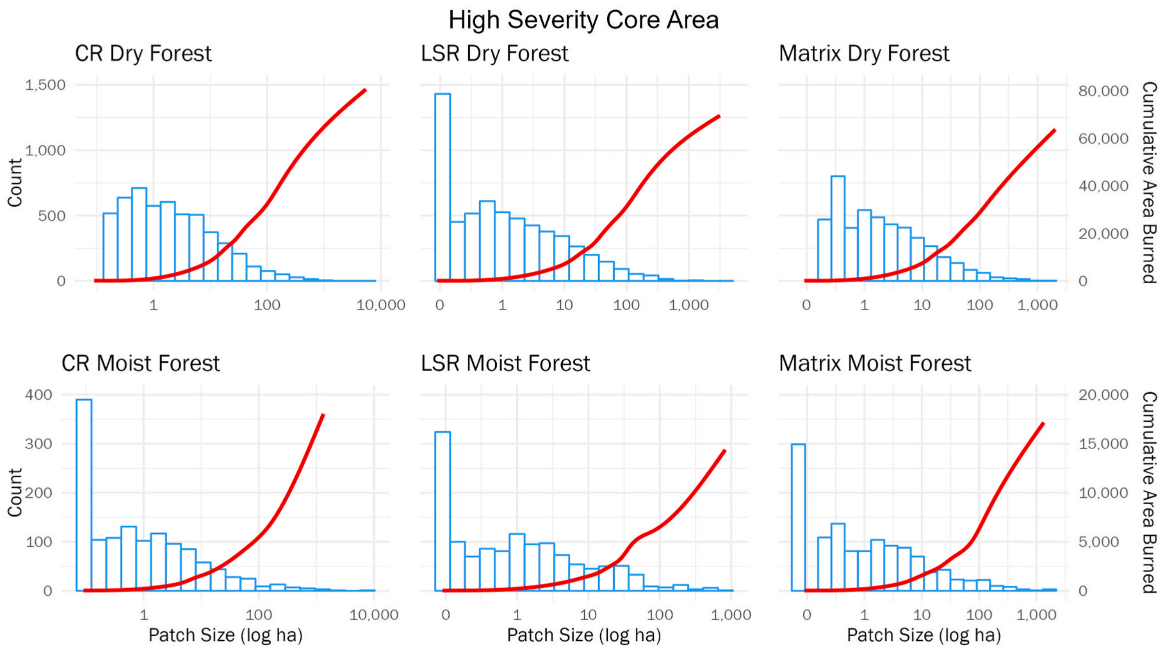

Fig. 5. Histogram of high severity core area patches (left axis) and cumulative area burned within high severity core area (red line, right axis) by dry forest zone (top row) and moist forest zone (bottom row) by allocation. CR - Congressional Reserve; LSR - Late Successional Reserve; Matrix - Matrix designation. Note that a single contiguous high severity patch can span multiple designations - therefore "patch size" here does not necessarily refer to whole patches (i.e., a 100 ha patch present in LSR moist forests may actually be part of a larger 500 ha patch).

dry forest zone Matrix.

Partial dependence plots revealed relationships between continuous predictor variables and fire severity (RBR) (Supplemental Figures S1-S6). In all models, RBR increased as wind velocity and VPD increased. Relationships between RBR and ERC across the study area were more variable – particularly at the tail ends of the distribution of ERC values – but suggested a general increase in severity as ERC increased across much of the study area. In all models, RBR generally increased with elevation, but tended to plateau or decrease above elevations around 2000 m (Figure S1, Figure S4). RBR steadily increased with pre-fire canopy cover in moist and dry forest zone CRs; in moist forest LSRs, RBR increased as canopy cover increased between 0 % and 20 %, plateaued between 20 % and 80 %, then increased again between 80 % and 100 % cover (Figure S2). In all models containing pre-fire biomass as a predictor, RBR increased with biomass up to 250 Mg/ha, where severity then plateaued or decreased as biomass increased. RBR increased in all models as time since disturbance increased for the first 25 years. In moist forest models, RBR then decreased between 25 and 75 years following disturbance, then continually increased; in dry forest models, RBR continued to decrease with increasing time since disturbance after the first 25 years.

In moist forest zone models where cover type was selected as a final predictor (Matrix and CR), cold forests, moist mixed conifer, and rainforest cover types were associated with the highest severity. The 'other' cover type (dominated by high elevation shrublands and grasslands) was associated with high severity in the moist Matrix model, and Cascades dry mixed conifer forests were associated with high severity in the moist CR model, though both cover types represented a small proportion of total land area in each designation. In dry forest zone models, Cascades dry mixed conifer, cold forests, moist mixed conifer, and rainforests were associated with the highest burn severity. Hardwood forests were additionally associated with high severity but comprised less than 0.1 % of the total area burned. Northwest California (nwCA) mixed conifer, nwCA mixed evergreen, oak woodlands, pine forests, pine-oak woodlands, and the 'other' category were also associated with increased RBR, though not to the extent of the aforementioned cover types.

Maps of SHAP values represent the pixel-level variability of local drivers of fire severity (Fig. 7). While we observed that many large, high severity patches across the study area were associated with top-down weather drivers, local importance maps of case-study fires also revealed a diversity of bottom-up drivers. In the 2020 Big Hollow fire, for example, large patches of high severity in moist forest zone LSRs were primarily associated with wind in a topographically complex landscape in the western Cascades of Washington (Fig. 7A). By contrast, in the 2003 Booth fire in the eastern Cascades of Oregon, high severity patches in dry forest zone CRs were driven largely by pre-fire canopy cover and cover type on the leeward side of a large ridge (Fig. 7B).

4. Discussion

We examined patterns and drivers of severity in wildfires that have burned between 1985 and 2022 within the NWFP area. Over the 37 year period, we observed significantly greater fire activity and burn severity patterns indicative of more severe ecological effects in the dry forest zone than moist forest zone (Fig. 2, Table 3), with trends in annual area burned and high severity in the dry forest zone consistent with those of frequent-fire forests in western North America more broadly (Cova et al., 2023; Harvey et al., 2016; Parks and Abatzoglou, 2020; Reilly et al., 2022). While the moist forest zone experienced relatively little fire for the first 30 years of the study period (Fig. 2), we observed significant shifts in recent wildfire activity and high severity patch sizes that are likely to continue under future climates (Abatzoglou et al., 2021; Cullen et al., 2023; Rupp et al., 2017). We found that across the NWFP area, forest types with historically frequent fire regimes such as pine-oak woodlands, oak woodlands, and dry mixed conifer forests experienced the greatest impacts from high severity fire (Fig. 3), which have important implications for the efficacy and adaptation of the NWFP in a warmer climate. Finally, while weather variables were the most influential predictor of severity across much of the study area and exerted strong top-down controls, we observed evidence of strong bottom-up controls such as pre-fire canopy cover, biomass, and cover type on severity at local scales that are relevant to place-based management strategies.

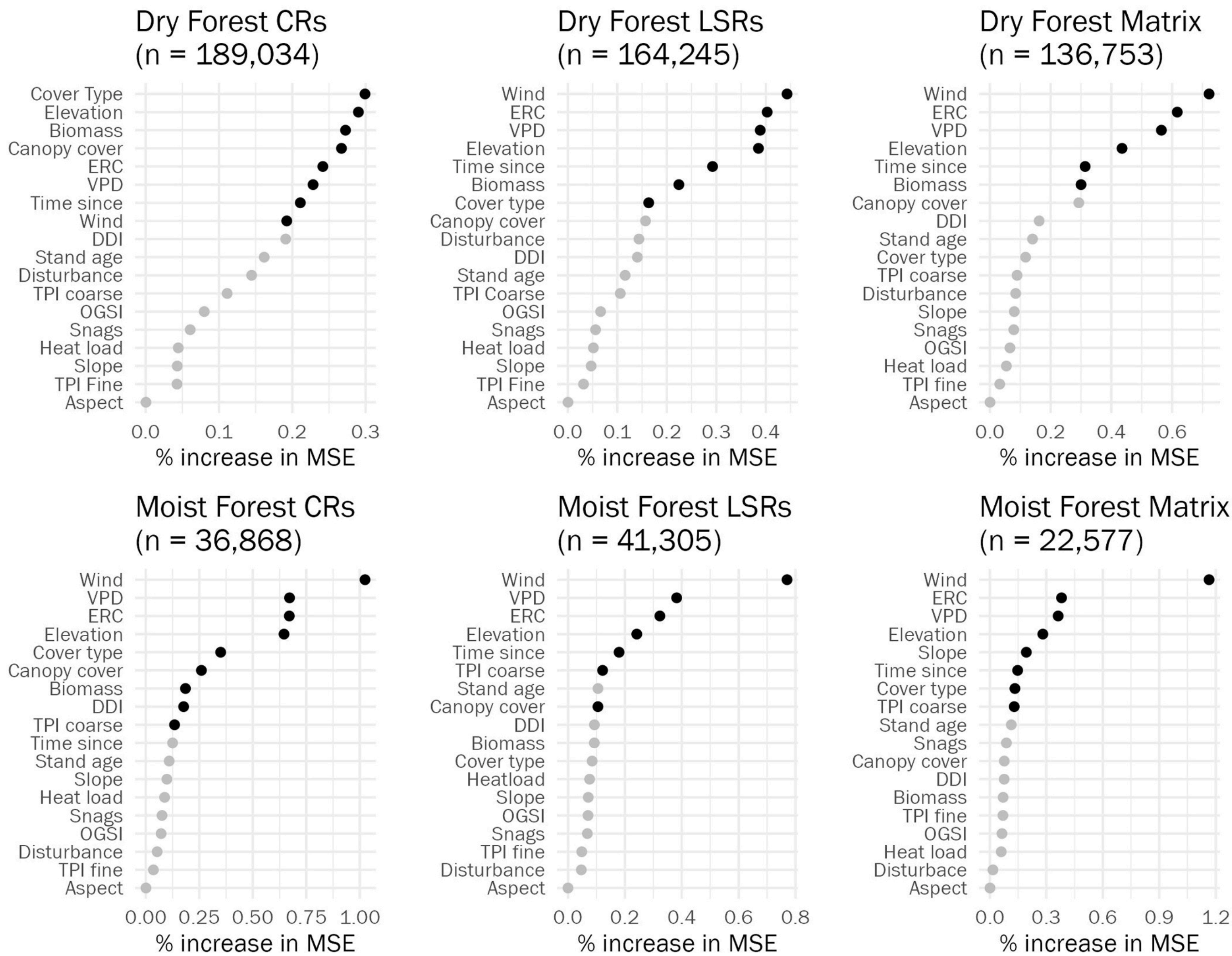

Fig. 6. Variable importance plots for predictor variables from Random Forest (RF) models of RBR for dry forest (top row) and moist forest (bottom row) zones and allocations. Black circles denote variables retained in the variable selection process; gray circles denote variables removed from the final RF models during variable selection. ERC - energy resource component; VPD - vapor pressure deficit; Time since - time (years) since last pre-fire disturbance; DDI - diameter diversity index; TPI - topographic position index; OGSI - old growth structural index; MSE, Mean Squared Error.

4.1. What have been the ecological effects of recent wildfires on NWFP forests across land use allocations and forest zones?

The NWFP recognized differences in historical disturbance regimes of dry and moist forests, and anticipated potential wildfire effects – especially in the dry forest zone – that could reduce forest cover and habitat associated with old and mature forest conditions (Franklin and Johnson, 2012). However, the expected wildfire impacts were based on the area burned in preceding decades leading up to the Plan's adoption in 1994 (Davis et al., 2016, 2011) and did not explicitly account for how fire regimes may shift under a changing climate (Gaines et al., 2022). Past studies have connected trends in annual area burned with severe droughts and higher temperatures, both anticipated to intensify under continued climate change (Dennison et al., 2014; Westerling, 2016). Given that across the NWFP area, fire activity has significantly increased over the last four decades (Fig. 2) and will likely continue to increase (Abatzoglou et al., 2021), our results suggest that effects of recent wildfires have outpaced Plan expectations, especially in the dry forest zone. Within the NWFP area, the dry forest zone experienced substantially greater fire activity and extent of high severity effects from wildfire than the moist forest zone (Table 3), with over four times both the total area burned and total high severity area across CRs, LSRs, and Matrix lands.

Observed increases in mean high severity patch size (Fig. 4A) may additionally challenge Plan expectations and existing reserves. This is particularly the case in the dry forest zone, where LSR boundaries were delineated as contiguous areas designed to withstand large wildfire events over at least the first half century of the Plan, such that unburned portions could maintain a well-connected network of LSOG forests and provide habitat for species such as the Northern Spotted Owl (NSO) (Johnson et al., 2023). Increasingly large high severity patches – with single patches as large as 10,000 ha in some areas – may challenge this reserve design as multi-storied, closed-canopy forests that are highly valued for NSO habitat (Sovern et al., 2019) are also highly susceptible to stand-replacing fire and extensive tree mortality. Concomitant with increases in high severity patch size, forest patches that burned with low severity effects or were unburned following wildfire (unburned-low severity patches) grew increasingly fragmented over the study period in the dry forest zone (Fig. 4C). It is important to note that we did not evaluate the connectivity of unburned forest patches *outside* of known fire perimeters (i.e, all potential habitat). Likewise, unburned-low severity patches within fire perimeters may not always contain suitable old forest habitat because pre-fire forest conditions within LSRs are variable. Our results suggest that, as more forested area burns, LSOG

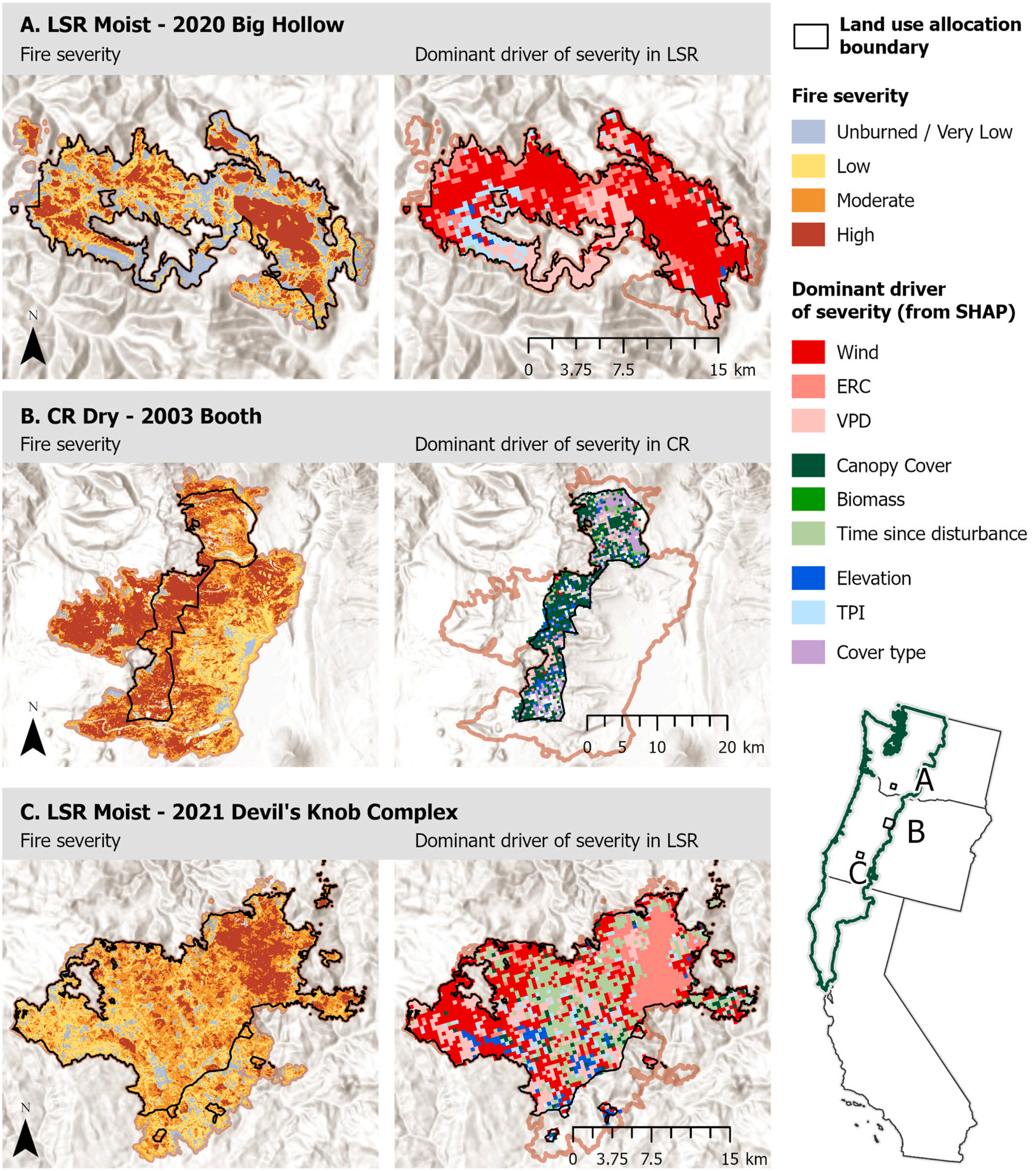

Fig. 7. Fire severity (left panel) and locally dominant driver of severity at the pixel level (right panel) for three select fires according to SHAP values.

habitats will likely erode due to the combined influences of large stand replacing patches and increased fragmentation of unburned areas.

In the moist forest zone, large fire years were relatively rare in the first 30 years of the study period, but recent large wildfires have driven observed increases in annual burned area (Fig. 2), mean high severity patch size (Fig. 4A), and extent of high severity interior core area in CRs, LSRs, and Matrix designations (Fig. 5). Although recent fires are largely consistent with historical fire regimes in the moist forest zone (Reilly et al., 2022), they have had important impacts to NWFP forests, particularly within LSRs. As a whole, only a small fraction (3.9 %) of the total LSR network in the moist forest zone experienced high severity fire effects (Table 3). However, in some places, high severity fire affected large portions of individual LSRs. For example, in the Willamette and Mt. Hood National Forests of western Oregon, we observed several large (>1000 ha) LSR units that burned almost entirely at high severity (Fig. 8A). Further south in the 2020 Archie Creek fire in the Oregon

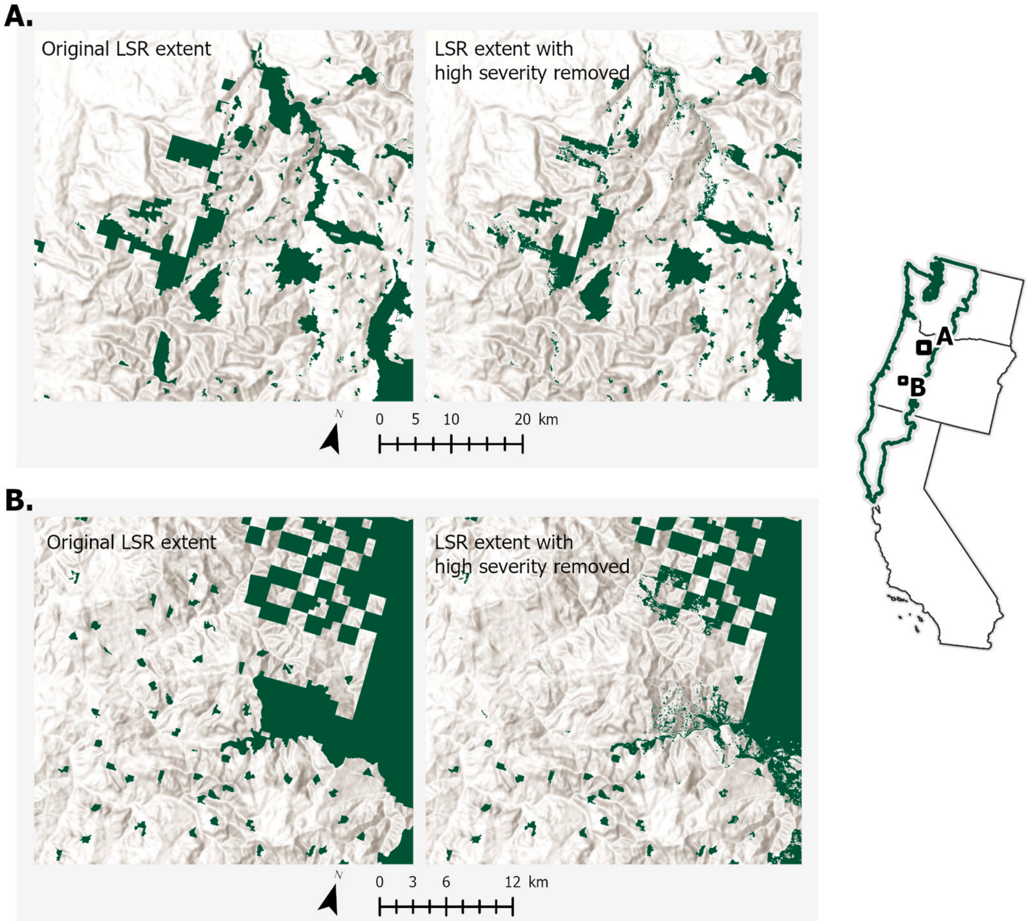

Fig. 8. Extent of high severity fire effects on Late Successional Reserve (LSR) boundaries. Areas shown in dark green represent LSR designations. The left image of each panel shows the true extent of LSR designations, and the right image of each panel shows what that extent would look like if areas that have burned with high severity effects were removed. Panel A (top) shows an area near the Willamette and Mt. Hood National Forests in the Oregon western Cascades; Panel B (bottom) shows an area near the Umpqua National Forest in the Oregon western Cascades.

western Cascades, 24 individual small (approx. 40 ha) LSRs burned completely as high severity fire (Fig. 8B). These severe wildfire effects to entire reserves have important implications for future Plan considerations. Specifically, designations of LSRs in the Plan were largely driven by where remaining LSOG forests still existed following an extended period of widespread timber harvests. The location and size of the reserve network was informed by the NSO conservation strategy (Thomas et al., 1990), then revised to provide for the conservation of other fish, wildlife, and plant species associated with LSOG habitats (Thomas et al., 1993). Our findings can inform considerations for adapting the design and management of existing reserves that may be necessary to meet Plan objectives amid substantial LSOG habitat loss.

Pine forests, oak woodlands, pine-oak woodlands, and dry mixed conifer forests consistently had the greatest high severity impacts across the NWFP area (Fig. 3). In some areas – such as pine-oak woodlands in dry forest zone CRs – over half the total forest extent has been impacted by high severity fire. In dry forest types historically characterized by frequent, low-intensity fire regimes, large extents of stand-replacing effects represent significant departures from historical disturbance regimes (Agee, 1996; Hessburg et al., 2019; Skinner et al., 2018). In these ecosystems, frequent fire maintained by active Indigenous stewardship and lightning ignitions historically reduced understory fuels and maintained relatively open canopies and dynamic, heterogenous forest structures that conferred resilience to wildfire and drought (Agee, 2003; Chamberlain et al., 2023; Hagmann et al., 2013; Hessburg et al., 2019; Taylor and Skinner, 2003). Dry forests with restored structural characteristics and large, fire-resistant trees are likely to be more resilient to climate change (Liang et al., 2017; Murphy et al., 2021), but continued fire suppression and loss of cultural burning leave them increasingly vulnerable to conversion to non-forest vegetation (Collins et al., 2011; Coop et al., 2020; Kreider et al., 2024). Based on these findings and related literature (Kalies and Yocom Kent, 2016; Prichard et al., 2021), conserving large old trees in dry forests and reducing the risk of severe wildfires will require a combination of active stewardship and ecological restoration treatments that include managed wildfire, mechanical thinning, and prescribed fire at a greatly accelerated pace and scale.

4.2. Within the NWFP area, what are the primary drivers of fire severity across land use allocations and forest zones?

Synoptic weather patterns exert strong controls on fire behavior, and previous studies have linked increases in vapor pressure deficit (Mueller et al., 2020), wind speed (Prichard et al., 2020), and ERC (Parks et al., 2018) to large fire growth and severe fire effects. While fire weather was a dominant driver across much of the study area, consistent with other studies in the region (Cansler et al., 2022; Evers et al., 2022), only fires greater than 500 ha were considered in our models. Fires that burn under more moderate weather conditions are typically suppressed and remain small (Calkin et al., 2015; Katuwal et al., 2016), reinforcing a 'suppression bias' (*sensu* Kreider et al., 2024) in which fires typically burning under the most extreme conditions are able to grow large. Because of this, our fires – and therefore our model results – likely reflect more extreme weather conditions.

Although our study considered only large fires in which weather variables would be expected to be dominant drivers of fire severity, bottom-up controls including vegetation cover type, elevation, pre-fire biomass, and pre-fire canopy cover were also important and were the dominant drivers in areas such as dry forest zone CRs (Fig. 6). Dry mixed conifer forests in particular were associated with high severity effects (Fig. 3, Figure S4), where the effects of climate change combined with fire exclusion and forest densification (Hessburg et al., 2019) have led to recent large wildfires that have rapidly homogenized forest structure with large extents of stand replacing effects and uncharacteristically large high severity patches (Cansler and McKenzie, 2014; Churchill et al., 2022). Drivers of fire severity derived from regional-scale models (such as those present in this study) can often overemphasize top-down controls with broad gradients in climate and elevation, masking signals from bottom-up controls operating at more local scales (Povak et al., 2025). Here, the dominance of bottom-up controls at the global model level within dry forest zone CRs – where the use of prescribed and managed wildfires can be difficult to implement (Miller et al., 2020) – is particularly notable, and may reflect profound departures in the structure, function, and composition of dry forest types (Hagmann et al., 2021) exacerbated by the effects of climate change.

Across the broader study area, wind was often the most important driver within large, high severity patches (Fig. 7A), but we observed strong bottom-up controls on severity at local scales within our case-study fires (Fig. 7B). Maps produced from SHAP values – representing the unique influence of each predictor variable on the response at the pixel-scale – demonstrated a range of dominant local drivers. Even within regions where weather variables were the most important drivers at the full model level, we observed variation in local dominance of predictors associated with pre-fire fuels, topography, and forest structure (Fig. 7A, B). For example, in the moist forest zone LSR model, outside of the largest, wind-driven high severity patch in the 2021 Devil's Knob Complex fire (Fig. 7C), time since disturbance, topography, and canopy cover were dominant drivers within smaller high severity patches along upper slopes and valley bottoms. As evidenced by studies on fuel treatment effectiveness, bottom-up controls can moderate fire severity even under extreme weather conditions (Lydersen et al., 2017; Prichard et al., 2020). Reliance on global variable importance alone may mask signals from important – but underrepresented – fine-scale controls at local scales (Povak et al., 2025), and local variables may be important in small domains of a fire but have limited influence on global model results (Dormann et al., 2007; Prichard et al., 2020). While computationally intensive, including assessments of local drivers of fire severity may be important tools to evaluate the strength of bottom-up controls to inform management strategies.

4.3. Management and policy implications

Adaptive management is the systematic and iterative process of planning and decision-making based on learned outcomes and monitoring that measure the effectiveness of existing management approaches (Holling, 1978). Investments in adaptive management within the NWFP supported a robust monitoring program (Davis et al., 2022, 2016, 2011), but enacting changes to management strategies with monitoring data has been difficult to implement in practice (Gaines et al., 2022; Spies et al., 2018), in part due to funding limitations, a legacy of distrust in active management, and staffing (Bormann et al., 2007; Spies et al., 2019, 2018b; Stankey et al., 2003).

Following decades of widespread timber harvest across much of the area, the NWFP was successful in conserving and enhancing LSOG forests by limiting logging on federal lands, particularly within LSRs (Spies et al., 2018c). However, climate change and increasingly large and severe wildfires since the Plan's adoption have profoundly reshaped landscapes across the NWFP area and now threaten the future of LSOG habitats (Table 3). While disturbances such as episodic drought, insect and pathogen outbreaks, and land development additionally threaten NWFP forests and will likely increase under climate change (Halofsky et al., 2020), wildfire is currently the driving agent of forest change across the region (Davis et al., 2022, 2015). As fire frequency, extent, and severity is predicted to increase through at least the next half century (Abatzoglou et al., 2021; Dye et al., 2024; Parks et al., 2016), it is critical that forest management plans account for and anticipate the effects of wildfire.

Rapid erosion of LSOG forests over substantial portions – or in some cases, the entirety – of LSRs (Fig. 8) poses a grave challenge for critical wildlife habitat connectivity, climate refugia, and remaining old and mature forests (Spies et al., 2019). Current Plan objectives to promote multi-layered, dense forest structures within LSRs are largely in line with the disturbance ecology and historical old-growth forest structures within much of the broader moist forest zone (Agee, 1996; Spies et al., 2018a), but adjustments to the existing design and management of reserves may still be required to meet Plan goals. For example, within moist forest zone LSRs where old forest habitat has been affected by substantial high severity fire effects, managers could implement variable density thinning in remaining second-growth stands to accelerate the development of old growth structural conditions (Halofsky et al., 2018; Spies et al., 2019). Site-specific fire suppression may be desired to protect LSOG forests from large, high severity fire effects (Halofsky et al., 2018), but as evidenced by the 2020 wildfire season in western Oregon, extreme fire weather may overwhelm the capacity for operations to contain and suppress active wildfires. Where entire LSR units have been impacted by high severity fire, management strategies could consider adjusting reserve boundaries to emphasize existing LSOG patches on surrounding Matrix lands (Halsey, 2024) or alternatively promote old forest habitat by ensuring greater protection of LSOG forest patches within Matrix designations, independent of reserve boundaries. Alternatively, following the recommendations of the National Cohesive Wildland Fire Management Strategy (US Department of Interior DOI, US Department of Agriculture USDA, 2014), an 'all hands, all lands' approach to LSOG conservation across land ownerships could enhance the resilience of old and mature trees and forests across the region.

In the dry forest zone, management strategies aimed at maximizing dense, multi-layered forest structures are generally inconsistent with maintaining ecological integrity in historically frequent-fire forests (Spies et al., 2019). In these systems, old forest conditions maintained by frequent-fire generally supported more open structures and could be more clearly emphasized in management direction for LSRs. Protection of large, old trees can serve as anchors in a landscape of dynamically shifting burned and unburned areas (Hessburg et al., 2015, 2019, 2016). Ecological restoration in dry forest types – including the use of thinning, prescribed and cultural burning, and managed wildfire – can reduce fuel continuity, promote the retention of climate- and fire-resistant large trees, and restore historical fire regimes (Hessburg et al., 2015, Prichard et al., 2021). Restoration of fire- and climate-resilient forest structure and composition is especially relevant for forests that have already been profoundly impacted by high severity fire such as pine-oak woodlands and dry mixed conifer forests (Fig. 3). Within reserves in the dry forest zone, adaptive management options could maintain existing LSR boundaries and implement proactive, continuous management to maintain ecological integrity and restore the role of frequent understory burning. As an alternative, retention and recruitment of large and old trees could be prioritized independent of land use allocations to promote fire- and climate-resilient forest structure and composition through proactive treatments across a broader landscape (Hessburg et al., 2015, Gaines et al., 2022).

Providing managers with the flexibility to manage wildfires under moderate weather conditions could also serve landscape-scale restoration goals both within and outside the existing reserve network. Particularly in frequent-fire dry forests, the pace and scale of prescribed burning and mechanical treatments is below what is needed to restore forest landscapes across the broader west (North et al., 2021; Prichard et al., 2021). Despite profound high severity effects in NWFP dry zone forests (Fig. 2, Table 3), recent fires have also done a substantial amount of ‘work’ to reshape forests at low and moderate severities. Low-to-moderate severity fire can have beneficial effects, shifting closed-canopy forests to more fire-resilient open structural conditions via understory fuel consumption and fire-induced thinning of mainly fire-intolerant small and medium-sized trees that may also be vulnerable to drought stress, forest insects, and pathogens (Churchill et al., 2022; Lydersen et al., 2016). Managing fires under moderate weather conditions can maximize this work and accelerate achievement of restoration objectives (North et al., 2021). Similar strategies have been adopted in other dry forested regions such as the Sierra Nevada (Keeley et al., 2021). In fire-frequent areas that have already burned, allowing managers the flexibility to conduct post-fire fuel treatments such as removing remaining ladder fuels (Collins et al., 2018) and creating or accentuating tree spatial patterns associated with fire and climate resilience could maximize beneficial outcomes in subsequent fire events (Chamberlain et al., 2023; Churchill et al., 2013; Koontz et al., 2020; Stevens et al., 2021).

Lastly, a key challenge for managers will be identifying adaptive management ‘triggers’, or predetermined commitments to initiate a shift in management strategy if monitoring data reveals undesirable ecological outcomes (Nie and Schultz, 2012). Necessary management interventions can be difficult to recognize, especially as ecosystems experience “shifting baselines,” in which accepted norms for environmental conditions gradually change (Pauly, 1995; Soga and Gaston, 2018). Evaluating trends in the spatial configurations of fire severity to understand impacts to forest habitat can be valuable tools that can be used to inform management triggers. For example, potential management triggers could incorporate monitoring of high severity patch size, proportion of high severity effects, or erosion of forest cover above a particular threshold (i.e., high severity impacts to greater than a specified proportion of a given forest type over a monitoring period). Indicators used as management triggers could be informed by pre-fire forest conditions, historical ranges of variability, and specific management goals, such as conserving LSOG forest habitat. Assessments of these post-fire patterns at a watershed level can be particularly valuable, as mid-scales are small enough to understand changing local conditions and identify restoration needs and priorities, yet are large enough to evaluate cumulative effects and scale down broad-scale management directives (Hessburg et al., 2013).

5. Conclusion

Wildfire has had profound effects on forests across the NWFP area. Across dry and moist forest zones and LSR, CR, and Matrix land allocations, annual area burned and mean high severity patch size has significantly increased. The dry forest zone – dominated by dry mixed conifer forests in the Klamath Mountains and eastern Cascades, mixed evergreen forests, and cold forests at high elevations – experienced the greatest fire activity in terms of total area burned. Dry mixed conifer forests and relatively rare, culturally important dry forest types such as pine-oak forests and oak woodlands were most severely impacted in terms of proportion of total forest extent burned at high severity.

While the moist forest zone had relatively small area burned compared to dry forests, recent fires including the 2020 wildfires in western Oregon resulted in a large-scale erosion of forest cover, particularly in LSRs, with a substantial or complete loss of forest cover in smaller networks of reserves. Our results have important implications for NWFP revisions aimed at integrating adaptive management. Adapting the design and management of reserves, increasing pre- and post-fire forest restoration activities, expanding opportunities for wildland fire use based on existing fire scars, and identifying ‘triggers’ to inform adaptive management may be necessary to achieve Plan goals.

CRediT authorship contribution statement

Cova Gina Rosa: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Susan Prichard:** Writing – review & editing, Writing – original draft, Resources, Methodology, Investigation, Conceptualization. **Harold Zald:** Writing – review & editing, Writing – original draft, Resources, Methodology, Investigation, Funding acquisition, Conceptualization. **William L. Gaines:** Writing – review & editing, Conceptualization. **Van R. Kane:** Writing – review & editing, Writing – original draft, Resources.

Declaration of Competing Interest

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

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.foreco.2025.123262.

Data availability

Data will be made available on request.

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