Received: 21 June 2022 | Revised: 21 December 2022 | Accepted: 28 December 2022

DOI: 10.1111/geb.13634

RESEARCH ARTICLE

Global Ecology and Biogeography A Journal of Macroecology WILEY

Spatial interactions among short-interval fires reshape forest landscapes

Brian J. Harvey[1,2] | Michele S. Buonanduci[1,2] | Monica G. Turner[3]

[1]School of Environmental and Forest Sciences, University of Washington, Seattle, Washington, USA

[2]Quantitative Ecology and Resource Management, University of Washington, Seattle, Washington, USA

[3]Department of Integrative Biology, University of Wisconsin, Madison, Wisconsin, USA

Correspondence
Brian J. Harvey, School of Environmental and Forest Sciences, University of Washington, Box 352100, UW-SEFS, Seattle, WA 98195, USA.
Email: bjharvey@uw.edu

Funding information
National Science Foundation, Grant/Award Number: 1719905 and 1720620; Northwest Climate Adaptation Science Center; U.S. Forest Service, Grant/Award Number: 17-CS-11130400-010

Handling Editor: Thomas Gillespie

Abstract

Aim: Ecological disturbances are increasing as climate warms, and how multiple disturbances interact spatially to drive landscape change is poorly understood. We quantified burn severity across fire regimes in reburned forest landscapes to ask how spatial patterns of high-severity fire differ between sequential overlapping fires and how landscape heterogeneity is shaped by cumulative disturbance patterns. We also characterized the amount and configuration of an emerging phenomenon: areas burned as high-severity fire twice in successive fires.

Location: Northwest USA.

Time period: 1984–2020.

Major taxa studied: Forests of western continental USA.

Methods: We used a field-calibrated atlas of satellite-measured burn severity across diverse fire regimes (more than three decades, >200 short-interval fires) to quantify landscape metrics of high-severity (>75% tree mortality) fire in sequential overlapping short-interval fires. We used generalized linear models to test differences in individual and cumulative landscape patterns of burn severity following the first and second fires.

Results: The amount of severe wildfire and patch size/configuration were generally similar between successive overlapping fires and across fire regimes. However, overlapping individual fires produced cumulative landscape patterns of recent high-severity fire that were consistently more homogeneous after two fires, with greater distances to remaining mature forest. Additionally, 19–25% of landscapes affected by short-interval fires burned at high severity in both fires, highlighting the spatial extent of repeatedly and severely disturbed forests.

Main conclusions: When two individually heterogeneous fires overlap, burn mosaics can fit together like puzzle pieces, whereby twice-burned landscapes are composed of large and simple-shaped patches of cumulative recent high-severity fire interspersed with small patches of mature/old forest. These cumulative spatial outcomes of interacting disturbances can be mechanisms of shifting ecosystem dynamics as global change unfolds and reburns continue.

KEYWORDS
climate warming, disturbance, disturbance interactions, heterogeneity, northern Rocky Mountains, Pacific Northwest, resilience, scale, wildfire

1 | INTRODUCTION

The 21st century has already seen increasing disturbance activity across forested regions world-wide (Seidl et al., 2017), with multiple disturbances interacting (Burton et al., 2020). Spatial heterogeneity is a key feature of disturbances (Turner, 2010), and a common refrain in ecology is that heterogeneity begets heterogeneity. For example, spatial patterns from one disturbance constrain or promote subsequent disturbance occurrence, size, magnitude and heterogeneity via ecological memory in landscapes (Peterson, 2002). Interacting disturbances of the same type are often characterized by negative or inhibitory feedbacks at fine scales, because one disturbance reduces necessary ingredients for subsequent disturbances [e.g., one fire removing fuel for the next fire (Burton et al., 2020)]. Such negative feedbacks are expected to attenuate otherwise accelerating climate-driven disturbance activity, yet disturbance interactions that unfold at fine scales might produce qualitatively different outcomes when scaled across broader landscapes.

Across both hemispheres, wildfire occurrence, size and severity have increased sharply with climate warming (Collins et al., 2022; Parks & Abatzoglou, 2020). Trends of increasing fire activity are expected globally (Ellis et al., 2022), although negative feedbacks from fuel limitations might diminish the increases expected from climate alone (Abatzoglou et al., 2021). As these changes unfold, understanding how multiple fires interact to reshape forested landscapes has emerged as a high priority for research, management and policy. Rapidly changing fire regimes might soon be misaligned with fire-adapted traits of dominant tree species (Nolan et al., 2020; Pausas & Keeley, 2021), eroding forest resilience (Johnstone et al., 2016) and catalysing transitions to non-forest ecosystems (Coop et al., 2020; Seidl & Turner, 2022).

A key spatial dimension of fire regimes is the landscape heterogeneity of burn severity [i.e., the spatial arrangement of ecological effects of fire, typically measured as vegetation killed by fire (Keeley, 2009; Morgan et al., 2014)]. Increases in burn severity (Parks & Abatzoglou, 2020) have been accompanied by changes in the spatial configuration of high-severity or stand-replacing fire (e.g., where fire kills all or most of the pre-fire live vegetation). In western North America, trends of increasing homogeneity towards larger and simpler-shaped patches of stand-replacing fire have emerged in the Rocky Mountains (Harvey et al., 2016b), south-west (Miller et al., 2009; Stevens et al., 2017) and north-west (Cansler & McKenzie, 2014; Reilly et al., 2017). Similar trends of increasing fire extent and severity have been documented in other regions world-wide (Collins, Bradstock, et al., 2021; Nolan et al., 2020; Pausas & Fernández-Muñoz, 2012; Whitman et al., 2022). The proportion, mean patch size and patch shape complexity of high-severity fire characterize important aspects of post-fire landscapes that underpin mechanisms of resilience, such as the extent and diversity of remnant forest patches (Meddens et al., 2018) and the distance to seed sources required for post-fire forest recovery (Gill et al., 2022). Spatial metrics have been quantified in many individual fires (Cansler & McKenzie, 2014; Collins et al., 2017; Harvey et al., 2016b), but how overlapping fires interact and produce cumulative patterns in landscapes burned by sequential fires is poorly understood.

As climate warms and fire regimes change, increased fire potential in many ecosystems leads to multiple fires overlapping as short-interval reburns, or locations that have burned more than once within three to four decades (Prichard et al., 2017). Overlapping fires are often linked disturbances (Simard et al., 2011), whereby one fire affects the likelihood or magnitude of a subsequent fire through feedbacks. For example, one fire can increase (via elevated flammability and susceptibility to fire) or decrease (via removal of fuels or decreased susceptibility to fire) the severity of a subsequent fire (Prichard et al., 2017). Notably, such feedbacks are dynamic through time; they are largely negative when intervals between fires are very short and shift towards neutral or positive as fire intervals approach several decades (Harvey et al., 2016a; Parks et al., 2014). Feedbacks between fires also vary in strength and effect among fire regimes. For example, in forests adapted to frequent low-severity fire, reburns can foster resilience if the occurrence of one fire reduces severity in a subsequent fire (Harvey et al., 2016a; Parks et al., 2014) and/or moves forest conditions towards those present during pre-colonization fire regimes (Cansler et al., 2022; Larson et al., 2013; Laughlin et al., 2023). Conversely, in high-severity, infrequent-fire regimes, where fuels are less limiting and weather/climate are the main constraints on fire, negative feedbacks are weaker and shorter in duration (Collins et al., 2019; Prichard et al., 2017). Across fire regimes, negative feedbacks imposed by fuel limitations from prior fires might be weakened as climate and weather drivers of fire intensify under climate warming (Bessie & Johnson, 1995; Turner & Romme, 1994), although this has not been tested widely.

How spatial and temporal dimensions of burn severity interrelate in reburns is a crucial knowledge gap in understanding global change, because spatial patterns of burn severity underpin key mechanisms of forest resilience (Downing et al., 2021; Gill et al., 2022). Linked disturbance interactions among fires have been well studied at the scale of forest stands (Collins, Hunter, et al., 2021; Harvey et al., 2016a; Parks et al., 2014), but how spatial heterogeneity among fires might be linked is poorly understood. Furthermore, short-interval disturbances can produce synergistic or compound disturbance effects that erode ecosystem resilience (Paine et al., 1998). For example, in forests adapted to infrequent fires, high-severity fires at very short intervals can produce uncharacteristic levels of biomass consumption and delay post-fire recovery (Brown & Johnstone, 2012; Turner et al., 2019; Whitman et al., 2019). Although locations that persist through multiple fire events as unburned fire refugia are well documented (Collins et al., 2019; Downing et al., 2021; Meddens et al., 2018), the amount and spatial configuration of areas that burn repeatedly at high severity have received relatively little study.

Here, we asked multiple questions about how landscape patterns of severe fire interact across forest ecosystems and fire regimes in the North-west USA (Supporting Information Figure S1).

To characterize the context of changing climate and fire weather in reburned landscapes, we quantified differences in average drought and fire-weather indices between the first and second fire. We then asked:

1. How do feedbacks among fires unfold spatially, such that the amount and spatial pattern of stand-replacing fire differ between the first and second fire, and how do these differences vary with fire interval and among fire regimes?
2. Are the cumulative spatial patterns of stand-replacing fire and distance to mature forest after the second fire different from those after the first fire?
3. How much area within reburned landscapes is potentially experiencing compound disturbance effects by burning at high severity in both fires, and how are these areas configured spatially?

2 | METHODS

2.1 | Study area

Forests and woodlands of the Northwest USA cover a gradient of fire regimes and associated fire-adapted traits of dominant tree species (Supporting Information Figure S1). We organized forest zones within the study area by the prevailing historical fire regime, using fire regime groups (FRGs) from the LANDFIRE, 2016 biophysical settings review (LANDFIRE, 2016; Rollins, 2009), as used in similar applications (Haugo et al., 2019). We grouped FRGs into three general fire regimes: low-severity fire (LSF) regime, which is FRG I [frequent (0–35 years) and primarily low severity (<50% overstorey tree mortality)]; mixed-severity fire (MSF) regime, which is FRG III [intermediate (0–200 years) and mixed severity]; and high-severity fire (HSF) regime, which is FRG IV and V [infrequent (35–200+ years) and high severity (>75% overstorey tree mortality)]. In general, fire regimes vary geographically within the region by climate conditions conducive to different frequencies and severities of fire and are dominated by conifer trees in the *Pinaceae* family with fire adaptations corresponding to the historical fire regime (Supporting Information Appendix S1; Agee, 1993; Baker, 2009; Stevens et al., 2020). LSF regimes occur in warm and dry (often lower-elevation) locations and are characterized by forests and woodlands dominated by thick-barked fire-resistant conifers (e.g., *Pinus ponderosa* and *Pseudotsuga menziesii* var. *glauca*) and resprouting angiosperms (e.g., *Quercus garryana*). HSF regimes occur in cool (often high-elevation) or wet (often coastal) locations and are characterized by forests dominated by thin-barked and fire-sensitive conifers and/or those adapted to reproduce prolifically after fire (e.g., *Pinus contorta*, *Pseudotsuga menziesii* var. *menziesii*, *Tsuga heterophylla*, *Thuja plicata*, *Picea engelmannii* and *Abies* spp.). MSF regimes occur in mid- to high-elevation forests dominated by a wide range of species with adaptations to variable fire effects, often corresponding to complex variations in topography. Within the study area, we focused on all forest and woodland areas that burned at least twice between the years 1984 and 2020.

2.2 | Data acquisition and processing

We generated a field-calibrated atlas of burn severity for all reburn areas during 1984–2020 in the study area with the following steps. Initially, all fire perimeters that intersected the study area were obtained from Monitoring Trends in Burn Severity (MTBS; https://mtbs.gov/) (Eidenshink et al., 2007) for the years 1984–2019 and from Rapid Assessment of Vegetation Condition after Wildfire (RAVG; https://burnseverity.cr.usgs.gov/ravg/) for the year 2020. Reburned landscapes were then identified by intersecting all fires. In areas where three or more fire perimeters overlapped, we retained the reburn area associated with the first two fires occurring within our study period. We retained only conterminous reburn polygons ≥1000 ha in size that were designated wildfires (i.e., we excluded prescribed fires) and that were ≥50% forested (Supporting Information Appendix S1). Each reburned landscape was assigned a dominant fire regime (LSF, MSF or HSF) based on the LANDFIRE FRG that represented the plurality of pixels within that landscape.

For each reburned landscape, we produced burn severity maps for the first and second fire using established methods (Parks et al., 2018). Using Google Earth Engine and Google Colaboratory, we generated burn severity maps for each fire using the Relative difference Normalized Burn Ratio (RdNBR) (Miller & Thode, 2007). This method uses a composite of Landsat imagery from the pre- and post-fire growing seasons to calculate changes in spectral reflectance in each 30 m pixel. We included an offset term in our RdNBR calculations to account for phenological differences between growing seasons (Parks et al., 2018). Maps of RdNBR were converted to categorical maps of two burn severity classes within each burn perimeter: stand-replacing (≥75% tree basal area killed by fire) and less than stand-replacing (<75% tree basal area killed by fire). To determine the value of RdNBR corresponding to ≥75% basal area killed by fire, we used statistical models developed from a regional dataset of 315 field plots paired with RdNBR values (Saberi, 2019; Saberi et al., 2022), following methods outlined by Harvey et al. (2019). Prior analyses have demonstrated that the value of RdNBR corresponding to 75% basal area mortality differs by forest structure at the time of fire (Saberi, 2019), which we accounted for in our burn severity maps. Specifically, we used RdNBR ≥ 542 as the cut-off for 75% basal area killed by fire (stand-replacing) for all locations burned in the first fire and for any locations burned in the second fire that were less than stand-replacing (RdNBR < 542) in the first fire. If a location burned as stand-replacing (RdNBR ≥ 542) in the first fire, the cut-off for 75% basal area killed by fire (stand-replacing) in the second fire was lowered to RdNBR ≥ 304 (Saberi, 2019; S.J. Saberi & B.J. Harvey, unpublished data). To evaluate the sensitivity of our findings to these thresholds for stand-replacing fire, we also calculated landscape metrics and performed all analyses using set values for all pixels in all fires of either 75% (RdNBR ≥ 542) or 90% (RdNBR ≥ 673) basal area killed by fire.

We evaluated a total of 221 reburned landscapes (n = 221 landscapes from n = 210 unique fire event pairs), covering a total area of 931,348 ha (Supporting Information Table S1). Most reburns (by number and total area reburned) were in the historical LSF regime, as were the largest median, mean and maximum reburn event sizes (Supporting Information Table S1).

2.3 | Climate and weather at the time of each fire

To characterize how the average climate and weather conditions might have differed between the first and second fires in each reburned landscape, we used two common indices related to wildfire potential. To characterize the drought severity at the time of fire, we used the Evaporative Demand Drought Index (EDDI; Hobbins et al., 2016; McEvoy et al., 2019), which incorporates atmospheric measures of temperature, humidity, wind speed and solar radiation near the ground surface. The EDDI characterizes locally scaled anomalies in evaporative demand or the amount of evapotranspiration that would occur given unlimited soil moisture, with values ranging from −2.5 (wet) to +2.5 (dry). To characterize potential fire intensity at the time of fire, we used the Energy Release Component (ERC; Bradshaw et al., 1984), which is a composite fuel-moisture index of the contribution of live and dead fuels to the available energy released per unit area during a fire. Higher ERC values are related to greater potential fire intensity. Both EDDI and ERC have strong demonstrated relationships with many wildfire characteristics, including spread rate, intensity and severity (McEvoy et al., 2019; Reilly et al., 2022).

The EDDI and ERC values were obtained for each fire from gridMET, a daily surface meteorological dataset available at 4 km resolution for the contiguous USA (Abatzoglou, 2013). For each 30 m pixel within each reburned landscape (first and second fire in each of the 221 reburned locations), values were extracted for a 30-day period centred on the ignition date of each fire (15 days before ignition and 15 days after ignition). We selected aggregate values over the 30-day period for EDDI and mean values over the 30-day period for ERC. We then averaged the aggregate EDDI and mean ERC values within the spatial footprint of each reburned landscape. Differences between first and second fires for EDDI and ERC within each burned landscape were tested using Student's two-sided *t*-test. These climate and weather data represent general conditions when each of the first and second fires occurred, but do not capture important spatially and temporally dynamic variables (e.g., local wind speeds) that are key drivers of fire spread, intensity and severity. Linking such data with daily fire progression maps is possible for fires occurring after but not before 2001 (Parks, 2014), limiting the capacity to assign daily weather variables for all fires in our dataset. Therefore, we present these data to provide context for average changes in weather and climate between first and second fires occurring in reburned landscapes but do not include these data as covariates in our models.

2.4 | Landscape metric calculation

For question 1 (comparing heterogeneity of burn severity between the first and second fires in each reburned landscape), we calculated four metrics for each landscape, following established methods (Harvey et al., 2016b). First, we calculated the proportion of each fire that was stand-replacing. Second, to characterize the spatial configuration of stand-replacing fire, we calculated the area-weighted mean patch size within each fire. Third, to characterize the complexity of stand-replacing patches, we calculated the area-weighted mean edge-to-area ratio of stand-replacing fire patches with each fire. Fourth, we calculated the total core area and the proportion of the fire that was core area within stand-replacing fire patches. Core area is defined as the area within the interior of stand-replacing patches that is ≥150 m from the patch edge, which exceeds the probable seed dispersal distance for most wind-dispersed conifers in western North America (Donato, Fontaine, Campbell, et al., 2009; Donato et al., 2016; Greene & Johnson, 1989; Harvey et al., 2016c; Kemp et al., 2016).

For question 2 (comparing the cumulative landscape patterns after both fires with those produced by the first fire), we used the same landscape metrics outlined for question 1 but replaced the second individual fire with landscape metrics of the cumulative patterns from both fires. That is, any area that burned as a stand-replacing fire in either the first fire or the second fire was considered part of the cumulative stand-replacing fire area, and the same four landscape metrics were calculated as for question 1. In addition, we used the landscape patterns of stand-replacing fire in the first fire and the cumulative (post-second fire) landscapes to calculate distances to post-fire live mature forest (a proxy for post-fire live tree seed source) (Supporting Information Appendix S1). We summarized the distributions of distance to live forest following each fire as pixel-based frequency distributions using 50 m bins of distance to live forest.

For question 3 (characterizing areas burned twice at high severity), maps were generated by identifying the area of overlap of stand-replacing fire in the first fire and the second fire. Using those maps, we calculated the same landscape metrics as for question 1 for the areas within reburn landscapes burning twice as stand-replacing. All landscape metrics for questions 1–3 were calculated using the *sf* (Pebesma, 2018) and *raster* (Hijmans et al., 2022) packages in the R statistical computing environment (R Core Team, 2021); all patches were characterized using the eight-neighbour rule after running a 3 × 3 majority smoothing filter on classified images to minimize the occurrence and impact of single-pixel patches.

2.5 | Landscape analyses

To address question 1 (Figure 1a), we treated each reburned landscape as a paired sample of burn severity patterns (landscape metrics characterizing the first and second fires) and quantified differences between the fires using the difference for each metric. Thus, zero represents no difference in a given landscape metric between the first and second fire, positive values indicate a metric being a higher in the second fire, and negative values indicate a metric being lower in the second fire.

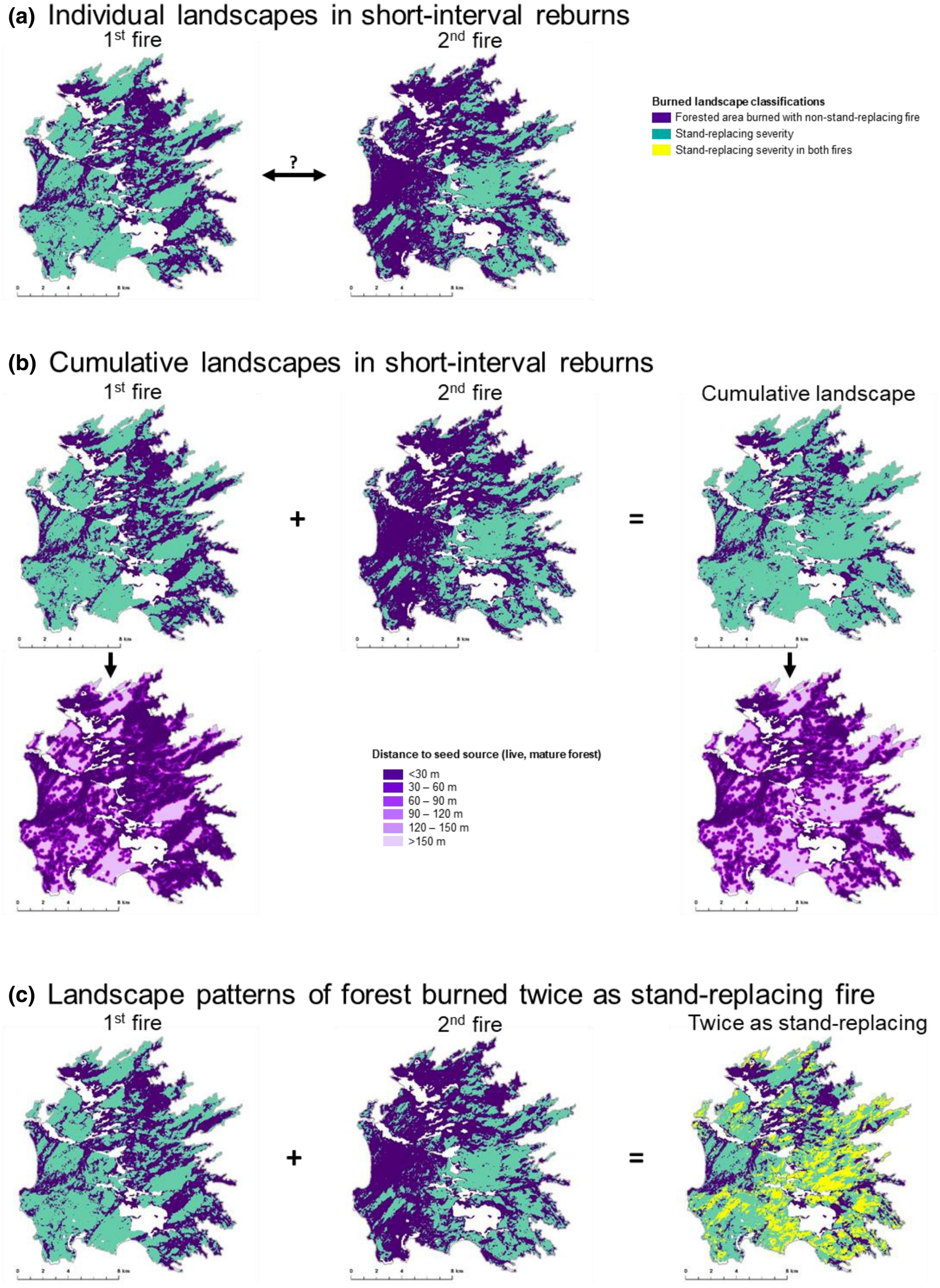

FIGURE 1 Graphical example of the approach to addressing each research question in the overlapping area where the 2016 Maple fire reburned the 1988 North Fork fire in Yellowstone National Park. (a) Question 1 compares landscape metrics of stand-replacing fire between the first fire and second fire in reburned landscapes. (b) Question 2 compares the cumulative landscape metrics of stand-replacing fire following both fires with those of the first fire, in addition to two measures of distance to seed source. (c) Question 3 characterizes trends in areas within reburned landscapes that burned as stand-replacing fire twice (in both fires)

To address question 2 (Figure 1b), we quantified differences in landscape metrics of stand-replacing fire and distance to live mature forest in each landscape following the first fire and the cumulative (first and second) fires. The same metrics as for question 1 were used to assess differences in landscape patterns of stand-replacing fire, and the proportion core area and continuous distributions of distance to live mature forest were compared between fires to assess differences in distance to a potential seed source. Proportion core area is a complement to total core area but quantifies the proportion of a landscape that has stand-replacing burned area that exceeds the reliable dispersal distance of most conifer trees (>150 m from a remaining live mature forest). We also compared the continuous distributions of distance to potential seed sources post-first fire and post-both fires using two metrics. Initially, we used a test statistic [Menning Departure Index (*M*); Menning et al., 2007] that characterizes the degree of difference in each pair of distributions. The value of *M* is useful in comparing differences in shape and skewness direction between two distributions that can have different absolute frequencies or ranges, such as tree size distributions among forests (Morris et al., 2022). Differences between two distributions are indicated by the sign of *M* (a right shift from a reference to a test distribution is positive, whereas a left shift from a reference to a test distribution is negative), with the magnitude of *M* indicating the distance of the shift between distributions (Supporting Information Figure S2). In addition to *M*, we also used the Stand-replacing Decay Coefficient (SDC), a parameter that characterizes the rate of loss of patch interior area with increasing distance to patch edge (Collins et al., 2017; Stevens et al., 2017). With SDC, smaller values reflect larger and simpler-shaped stand-replacing patches with greater interior patch area far from patch edges. The analytical approach in question 2 mirrored question 1, because each landscape was a paired difference between the post-first fire landscape and the cumulative landscape after both fires for each of these seven metrics.

For questions 1 and 2, analyses were designed to provide the most interpretable evaluation of differences in landscape patterns across reburned landscapes by accounting for differences in landscape sizes among reburns. We did this by using normalized or proportional differences wherever possible. For the proportion stand-replacing and proportion core area, given that the raw values are already proportions, we compared the proportion in the first fire with the proportion in the second fire. In other words, if the first fire was 50% stand-replacing and the second fire was 25% stand replacing, the difference would be −25% stand-replacing (i.e., difference = proportion in second fire minus proportion in first fire). For area-weighted mean patch size and total core area, we compared paired fires in reburned landscapes using proportional differences by calculating differences in $\log_{10}$-transformed values between fires. In other words, if the area-weighted mean patch size of stand-replacing fire was 1000 ha in the first fire and 10 ha in the second fire, the proportional difference would be −2, or two orders of magnitude lower [i.e., difference = $\log_{10}$(value in second fire) minus $\log_{10}$(value in first fire)]. For the edge-to-area ratio of stand-replacing patches, and for the *M* index and SDC (for differences in distance-to-seed-source distributions), we compared raw differences in these metrics between paired fires in a landscape, because the metrics are already ratios or indexes, respectively.

For question 3 (Figure 1c), we evaluated how two key metrics (proportion twice stand-replacing and area-weighted mean patch size of twice stand-replacing locations) varied with the interval between fires.

Analyses for all three research questions followed a similar structure and were conducted with each reburned landscape as the focal unit of analysis (Figure 1). Differences between the first and second fires (question 1) and between the first fire and the cumulative burned landscape (question 2) were assessed using general linear models fitted to the differences between paired metrics in each of the reburned landscapes. Models included a categorical term for fire regime (low, mixed or high) and a continuous term for the interval between fires (1–35 years) to test how effects varied among these two important dimensions. Models also included an interaction term (FRG × reburn interval) to allow for varying slopes across fire regimes. For question 3, the analysis focused on areas burned twice as stand-replacing (only possible after the second fire). Therefore, we did not use a paired analysis of differences for question 3, but our models retained the same explanatory terms for fire regime and reburn interval used for questions 1 and 2. Model fits were plotted with the *ggplot2* package (Wickham, 2016) in R, with mean linear model fits and 95% confidence intervals (CIs; shaded envelopes). Non-overlap of 95% CIs with the zero line were interpreted as average increases or decreases in landscape metrics between paired fires in reburned landscapes. Tabular model outputs for each analysis are presented in Appendix S2.

3 | RESULTS

3.1 | Drought and fire weather in reburned landscapes

Reburned landscapes were characterized on average by modestly greater drought and fire-weather severity during the second fire compared with the first fire (Figure 2). Paired differences in drought and fire weather between first and second fires were variable among reburned landscapes, and average differences were not substantially greater than zero across all three fire regimes. Drought at the time of fire was on average more severe during the second fire, with EDDI on average +0.30 greater in the second fire than the first fire (95% CI from +0.16 to +0.44, $p < .001$; Figure 2, top panel). Likewise, fire weather at the time of fire was on average more severe during the second fire, with ERC on average +3.9 greater in the second fire than the first fire (95% CI from +2.6 to +5.2, $p < .001$; Figure 2, bottom panel). An average difference of +0.30 for EDDI between the first and second fire represents a 9% increase across the range of EDDI values observed among all fires (−1.28 to 2.09), and an average difference of +3.9 for ERC represents a 6% increase across the range of ERRC values observed among all fires (26.9–89.8).

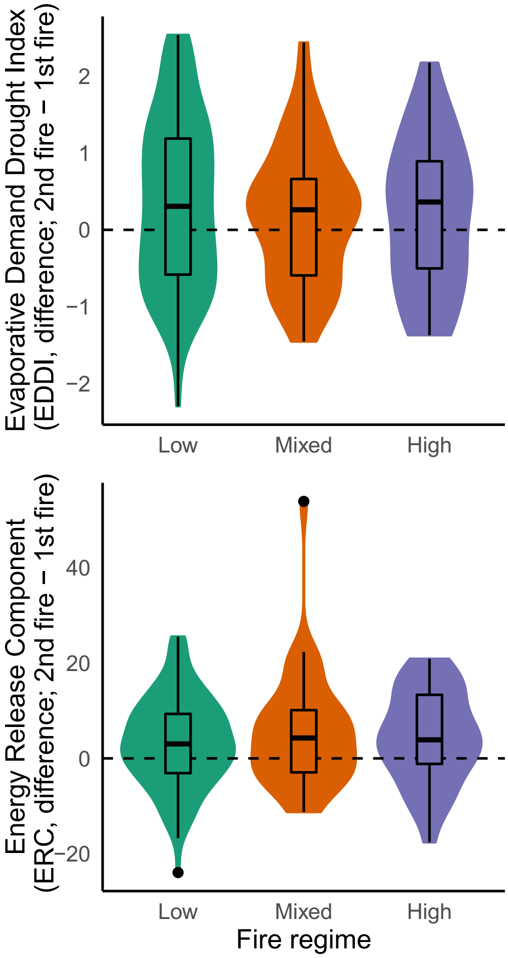

FIGURE 2 Difference between the first and second fires in reburn landscapes for two common indices characterizing climate and weather conditions related to wildfire potential. The top panel shows the difference in evaporative demand drought index (EDDI) between fires (positive values are greater drought severity in the second fire). The bottom panel shows the difference in the energy release component (ERC) between fires (positive values reflect warmer and drier weather associated with greater potential fire intensity in the second fire). Each column is a fire regime (LSF, low-severity fire, left column, green; MSF, medium-severity fire, centre column, orange; HSF, -high-severity fire, right column, purple). Inset boxplots denote the median (centre line), 25th and 75th percentiles (lower and upper bounds of the box), largest value within 1.5 times below and above the 25th and 75th percentiles (lower and upper ends of lines, respectively), and black dots represent any individual value outside these bounds. Violin plots (coloured/shaded areas around boxplots) are the continuous distribution of values along the *y*-axis for each category. The EDDI and ERC values were assigned to each fire by spatially averaging values over the reburn landscape from a 30-day period centred on the fire ignition date for each fire individually

3.2 | Question 1: Individual landscape patterns of stand-replacing fire, second fire versus first fire

On average, differences in landscape patterns of stand-replacing fire between the first and second fires were subtle, although there was high variability in paired differences in landscape metrics across fire regimes and reburn intervals (Figure 3; Table 1; Supporting Information Table S2). In the LSF regime, second fires had lower proportions stand-replacing, smaller and more complex-shaped patches and less core area than the first fire on average if they occurred within *c*. 5–10 years of the first fire (Figure 3, left column). However, when reburns occurred in the LSF regime at intervals exceeding *c*. 10 years, second fires had, on average, greater proportions stand-replacing, larger and less-complex-shaped patches and greater core area than the first fire. For the MSF and HSF regimes, across nearly all reburn intervals, landscape patterns of the second fire were on average indistinguishable from the first fire (Figure 3, centre and right columns); one exception was greater proportion stand-replacing for the second fire in the HSF regime when reburns occurred at intervals exceeding *c*. 15 years (Figure 3, right column, upper row).

3.3 | Question 2: Cumulative landscape patterns of stand-replacing fire and distance to seed source

Cumulative patterns of stand-replacing fire (i.e., landscape patterns of areas burned as stand-replacing in either fire) had greater proportions stand-replacing, larger and less complex-shaped patches and greater core area than the first fire in each landscape (Figure 4; Table 1; Supporting Information Table S3). On average across reburn intervals, the mean cumulative proportion of stand-replacing fire after the second fire was 0.2 (i.e., 20%) greater than after the first fire (Figure 4). Across fire regimes, the average proportion stand-replacing fire in cumulative burned landscapes was *c*. 1.5 times the average proportion in first fire landscapes, resulting in 42–61% of reburn landscapes burned as stand-replacing in at least one of the two fires (Table 1). Patch sizes responded in a similar way, with increases in the average area-weighted mean patch size from first fires to cumulative burned landscapes ranging from more than two times in the LSF and MSF regimes to approximately three times in the HSF regime. Patch edge-to-area ratio decreased from the first fire to the cumulative burned landscape across fire regimes. Finally, the average total core area in cumulative burned landscapes was approximately two times the total in the first fire landscapes across fire regimes (Table 1). Trends in these cumulative patterns of stand-replacing fire were stronger with longer reburn intervals for LSF and HSF regimes but were unrelated to reburn interval in the MSF regime (Figure 4).

In addition to the total core area, reburns increased the proportion of burned landscapes that was core area within stand-replacing patches, in addition to the average distances to potential seed source in burned landscapes (Figure 5; Supporting Information Tables S4 and S5). On average, reburns more than doubled the cumulative proportion of burned landscape that was core area and consistently shifted distributions of distance to live mature forest within burned landscapes to the right and with longer tails (i.e., greater area at longer distances to live trees). The largest increase in proportion core area and distance to live forest was in the HSF regime, but differences among fire regimes were minimal. The magnitude of differences for core area and distance to live forest increased with reburn

Global Ecology and Biogeography A Journal of Macroecology -WILEY

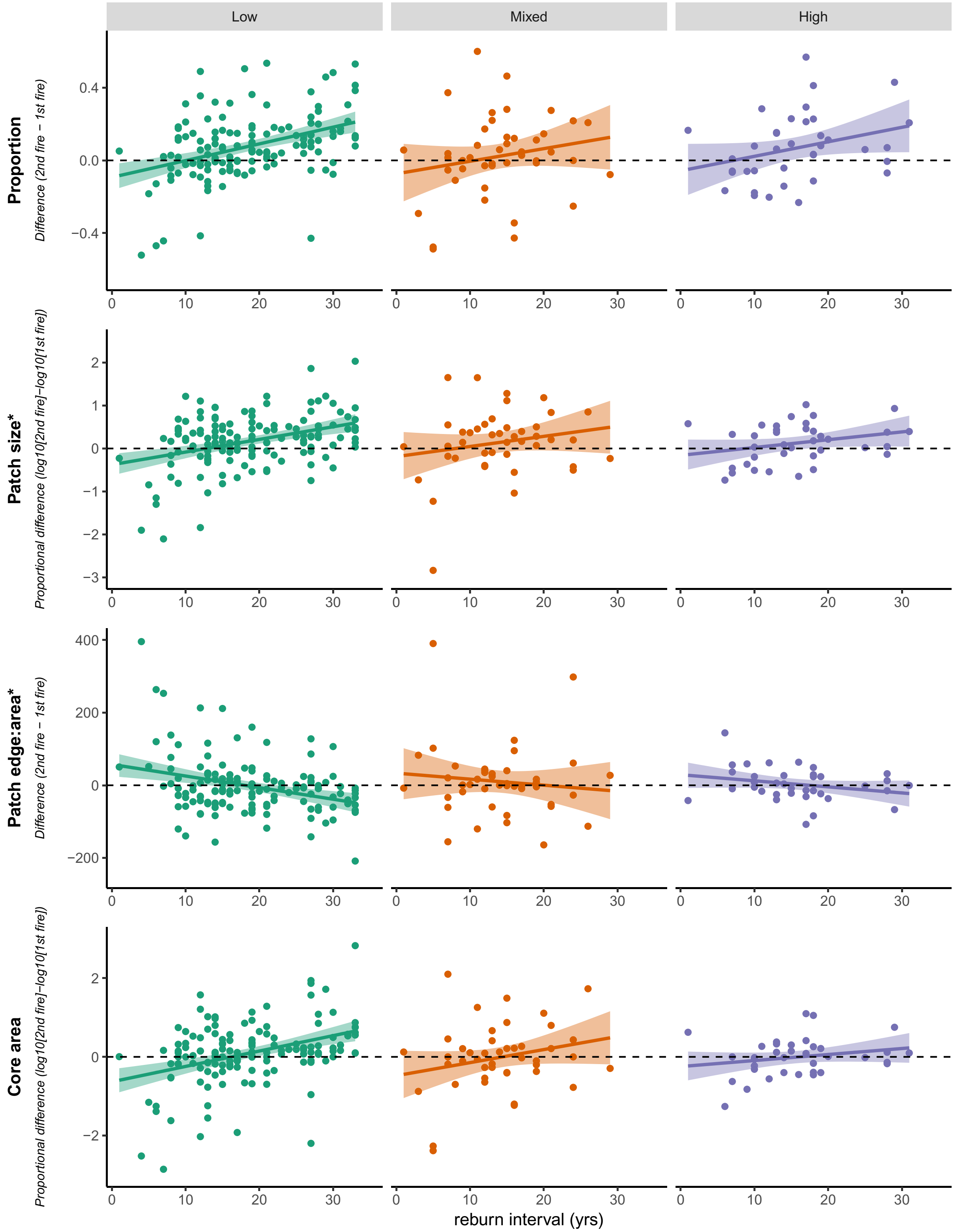

FIGURE 3 Model results for question 1, comparing the difference in landscape metrics between the first and second fires in reburn landscapes. Each row is a separate landscape metric, and each column is a fire regime (LSF, low-severity fire, left column, green; MSF, medium-severity fire, centre column, orange; HSF, high-severity fire, right column, purple). Each point is a single reburned landscape, representing the paired difference (second fire minus first fire) in each landscape metric, with the zero line (dashed) representing no difference. Model fits are represented by a continuous line for the mean model fit and shaded envelope for the ±95% confidence interval. See the Supporting Information (Table S2) for full model results. *Area-weighted metrics

TABLE 1 Landscape metrics of stand-replacing fire in each individual fire (first and second), the cumulative pattern of stand-replacing fire (areas that burned as stand-replacing fire in either fire) and stand-replacing fire in both fires (areas that burned as stand-replacing fire in first and second fires)

Fire regime	First fire			Second fire			Cumulative (either fire)			Stand-replacing in both fires		
	Range	Mean	Median	Range	Mean	Median	Range	Mean	Median	Range	Mean	Median
Proportion stand-replacing												
LSF	<.01–.79	.26	.24	<.01–.81	.34	.32	.03–.88	.42	.40	.00–.65	.19	.17
MSF	.00–.85	.39	.41	<.00–.86	.42	.42	<.01–.91	.59	.63	.00–.72	.22	.22
HSF	.03–.78	.40	.40	.01–.84	.47	.48	.06–.89	.61	.67	.00–.53	.25	.27
Area-weighted mean patch size (ha)												
LSF	1–7850	435	147	3–21,656	785	213	4–19,922	985	328	0–6399[a]	275[a]	67[a]
MSF	0–1613[b]	423[b]	256[b]	1–2511	554	383	2–2824	996	852	0–1547[a]	142[a]	67[a]
HSF	4–3022	525	289	2–5701	797	437	6–10,975	1499	823	0–1689[a]	200[a]	105[a]
Area-weighted mean patch shape complexity (edge-to-area ratio; m/ha)												
LSF	58–558	166	154	35–473	164	144	34–450	138	129	90–1167[a]	214[a]	186[a]
MSF	57–431[b]	144[b]	118[b]	49–729	157	133	45–571	115	88	62–833[a]	233[a]	196[a]
HSF	71–378	138	126	52–434	141	115	44–384	110	88	98–594[a]	207[a]	186[a]
Total core area within stand-replacing patches >150 m from edge (ha)												
LSF	0–7515	369	85	0–9320	519	112	0–11,150	723	209	0–4249	191	24
MSF	0–717	230	153	0–934	195	127	0–1346	439	372	0–639	42	13
HSF	0–3780	340	120	0–2538	341	160	0–6746	734	377	0–512	64	28

Abbreviations: HSF, high-severity fire; LSF, low-severity fire; MSF, mixed-severity fire.

[a]One fire in the LSF regime, three fires in the MSF regime and one fire in the HSF regime did not have any area burned twice as stand-replacing fire and were therefore removed from calculations of stand-replacing patch size and shape.

[b]One fire in the MSF regime did not have any stand-replacing fire and was therefore removed from calculations of stand-replacing patch size and shape.

interval for LSF and HSF regimes but was unrelated to reburn interval in the MSF regime (Figure 5).

3.4 | Question 3: Landscape patterns of areas burned twice as stand-replacing fire

Forest that burned twice as stand-replacing accounted for *c*. 19–25% of reburned landscapes across fire regimes (Figure 6; Table 1; Supporting Information Table S5). Patches of forest burned twice as stand-replacing fire were smaller and more complex in shape than patches of stand-replacing fire in either of the fires individually (Table 1), but accounted cumulatively for 160,225, 22,586 and 35,152 ha of burned landscapes in LSF, MSF and HSF regimes, respectively; collectively, 217,963 ha across all reburned landscapes. The proportion of reburned landscapes and the mean patch size of twice stand-replacing patches increased with reburn interval across fire regimes (Figure 6; Supporting Information Table S5).

4 | DISCUSSION

Our findings highlight key dimensions of disturbance regime dynamics and have important implications for how changing fire regimes and their landscape patterns can influence forest resilience. First, feedbacks among landscape patterns of burn severity were variable and weaker than expected, such that spatial patterns of stand-replacing fire in second fires were generally similar to previous fires; negative landscape feedbacks were short lived and evident only in limited contexts. Second, the cumulative landscape patterns of recent high-severity fire produced by sequential overlapping fires were on average more homogeneous than those produced by each fire individually, with important implications for post-fire resilience mechanisms in forests. Third, areas that burned twice as stand-replacing in successive fires are widespread and suggest an important, yet poorly understood, signal of changing forest fire regimes. Collectively, these insights help to refine our understanding of how spatial disturbance interactions contribute to disturbance regime change as climate warms.

4.1 | Feedbacks among landscape patterns of fire in reburns were variable, weak and brief

Across fire regimes, our findings suggest that landscape memory (Peterson, 2002) of burn severity from one fire to the next is ephemeral and context dependent during a period of warming climate. High variability and minimal average differences between landscape patterns of stand-replacing fires in second fires versus preceding fires suggest weaker than expected spatial feedbacks between fire

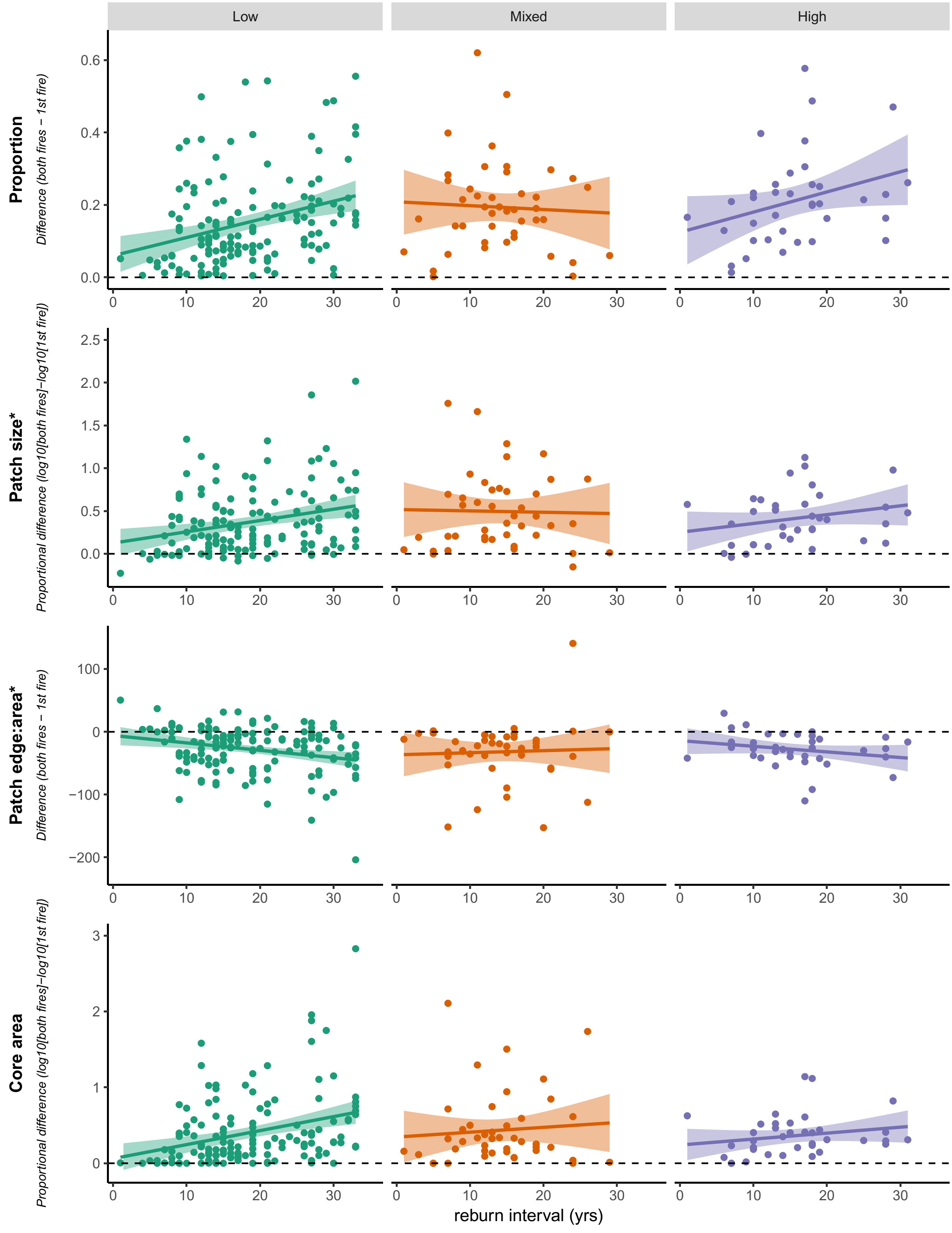

FIGURE 4 Model results for question 2, comparing the difference in landscape metrics between the first fire and cumulative burn severity patterns (both fires) in reburn landscapes. Each row is a separate landscape metric, and each column is a fire regime (LSF, low-severity fire, left column, green; MSF, medium-severity fire, centre column, orange; HSF, high-severity fire, right column, purple). Each point is a single reburned landscape, representing the paired difference (cumulative patterns from both fires minus first fire) in each landscape metric, with the zero line (dashed) representing no difference. Model fits are represented by a continuous line for the mean model fit and shaded envelope for the ±95% confidence interval. See the Supporting Information (Table S3) for full model results. *Area-weighted metrics

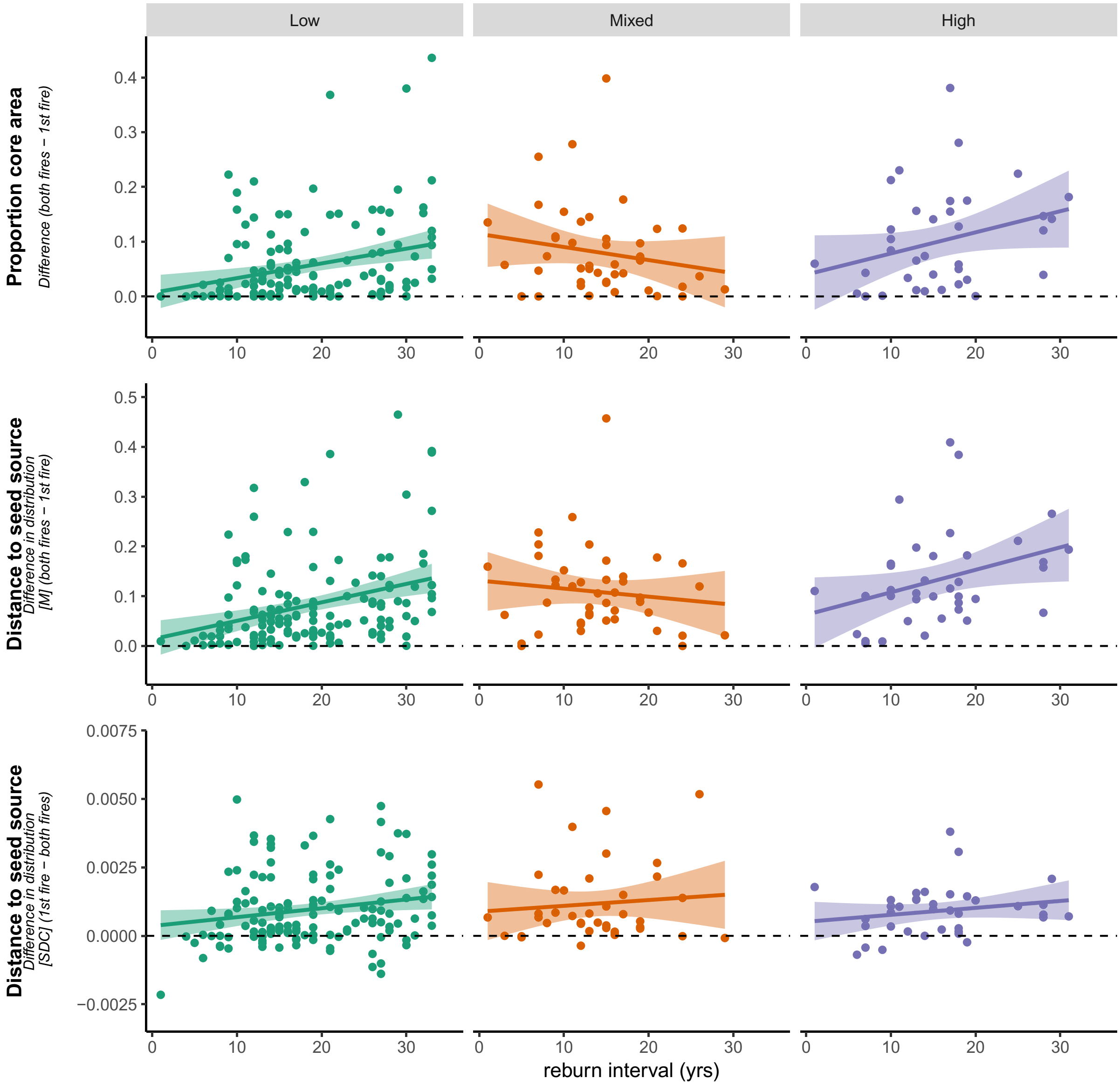

FIGURE 5 Model results for question 2, comparing the difference in three measures of distance to potential seed source between the first fire and cumulative burn severity patterns (both fires) in reburn landscapes. The top row is proportion core area (>150 m from edge of a stand-replacing patch), the middle row is the Menning index (*M*), which characterizes the shift (right = positive; left = negative) in the distribution of distance to seed source in stand-replacing burned areas, and the bottom row is the stand-replacing decay coefficient (SDC), which characterizes the rate of loss of patch interior area with increasing distance to patch edge. Each column is a fire regime (LSF, low-severity fire, left column, green; MSF, medium-severity fire, centre column, orange; HSF, high-severity fire, right column, purple). Each point is a single reburned landscape, representing the paired difference (cumulative patterns from both fires minus first fire) in each landscape metric, with the zero line (dashed) representing no difference. Model fits are represented by a continuous line for the mean model fit and shaded envelope for the ±95% confidence interval. See the Supporting Information (Table S3) for full model results

events, and landscape constraints from fuel limitations occurred in a subset of conditions that correspond to differences among fire regimes. Our findings were robust to the threshold used for stand-replacing fire. Results were qualitatively similar whether 90 or 75% basal area killed by fire was used to characterize stand-replacing (high-severity) fire and whether stand-replacing thresholds for locations in a second fire accounted for areas that had burned in a prior stand-replacing fire (Supporting Information Appendix S3).

Spatial patterns that were lower severity and more heterogeneous in the second fire than the previous fire were restricted to the LSF regime when reburn intervals were < 10 years. This is the post-fire period when fuel levels are greatly reduced from the preceding fire (Eskelson & Monleon, 2018; Stevens-Rumann & Morgan, 2016; Stevens-Rumann et al., 2020) and when negative feedbacks among fires are expected to be strongest. Given mean fire-return intervals (MFRIs) of *c*. 0–35 years in LSF regimes, short-interval fires in our study period (a window of 35 years) are expected as a normal part of the fire regime and often serve as stabilizing forces in maintaining resilience to fire (Hessburg et al., 2015, 2019). However, this feedback flipped from negative

Global Ecology and Biogeography A Journal of Macroecology

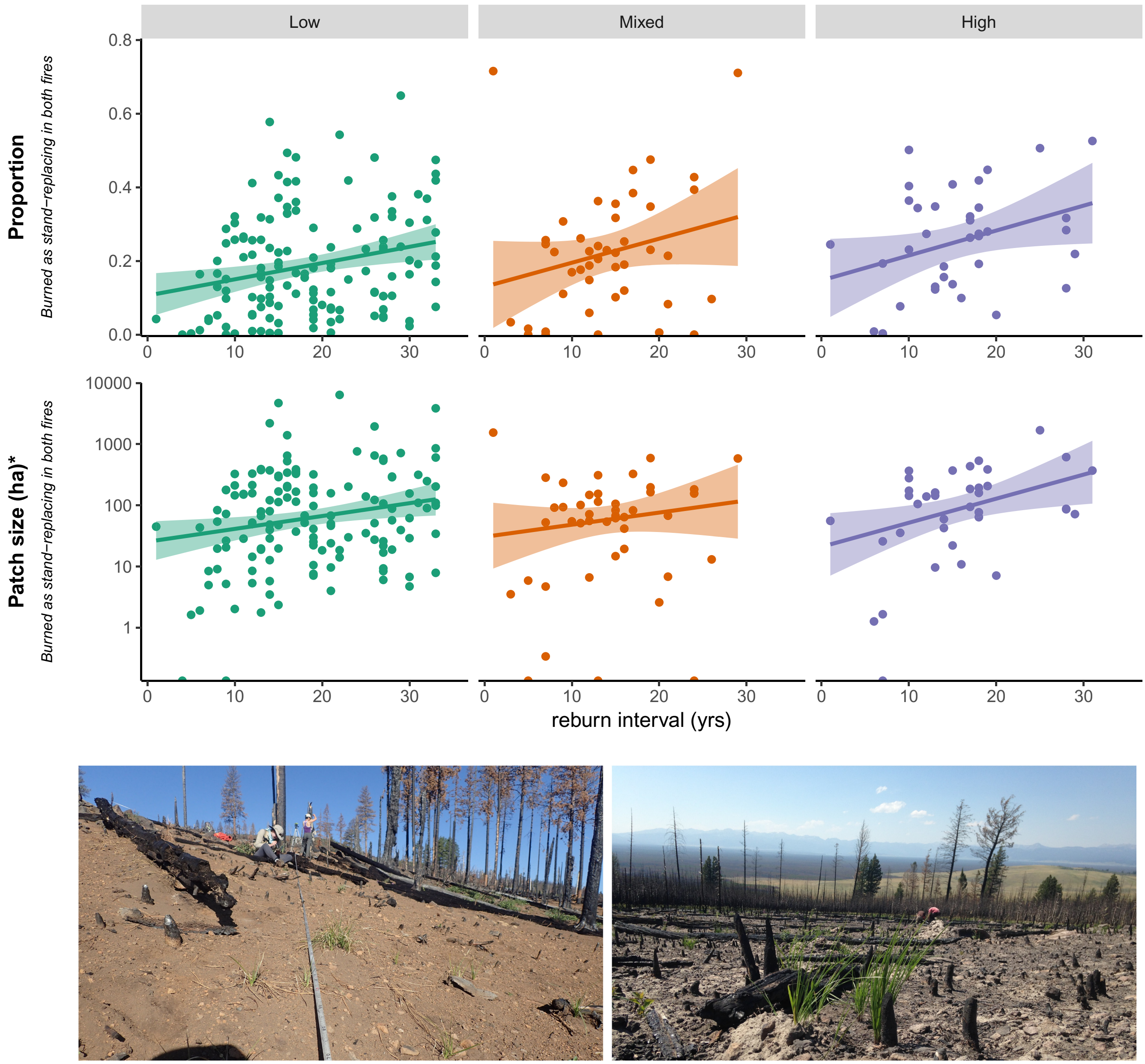

FIGURE 6 Model results for question 3, characterizing two landscape metrics of areas that burned as stand-replacing fire twice (in both fires). The top row is the proportion burned twice as stand-replacing within reburned landscapes, and the bottom row is the area-weighted mean patch size of such areas. Each column is a fire regime (LSF, low-severity fire, left column, green; MSF, medium-severity fire, centre column, orange; HSF, high-severity fire, right column, purple). Each point is a single reburned landscape, representing the value in each of the 221 reburned landscapes. Model fits are represented by a continuous line for the mean model fit and shaded envelope for the ±95% confidence interval. Photographs at the bottom illustrate locations burned twice as stand-replacing in the LSF regime (left) and HSF regime (right). Photograph credits: S. J. Saberi (left); B. J. Harvey (right). See the Supporting Information (Table S5) for full model results. *Area-weighted metrics

to positive when fire intervals exceeded 10–15 years, such that second fires were more severe and landscape patterns were more homogeneous than preceding fires. This suggests that fuels are sufficient to support equal or greater burn severity patterns by this fire interval, which could be exacerbated by increases in average drought and fire weather conditions when reburns occur (Figure 2). This fading constraint of past fire limiting subsequent fires has been documented at stand (point) scales (Cansler et al., 2022; Harvey et al., 2016a; Parks et al., 2014). Through the lens of spatially interacting fires, our analysis demonstrates how local ecological memory (Peterson, 2002) emerges across broader scales and landscapes.

For MSF and HSF regimes, there was little detectable difference between average landscape patterns of burn severity in the second fire versus the previous fire across intervals, which is probably attributable to different controls of burn severity compared with LSF regimes. Fires are less frequent in MSF and HSF regimes, because the climate and weather conditions are less routinely conducive to fire (Agee, 1993; Baker, 2009). As such, there were fewer reburn occurrences with very short intervals (<15 years, when fuel limitations from past fires are strongest) compared with LSF regimes, as expected. By the time most reburns occurred in MSF and HSF regimes (>10–15 years), subsequent stand-replacing fire can be supported by abundant post-fire live fuels and coarse surface fuel accumulation from snagfall (Donato et al., 2013; Nelson et al., 2016, 2017; Stevens-Rumann et al., 2020). That is, local inhibitory feedbacks have been lifted (Burton et al., 2020), and ecological memory fades to have less effect on emergent landscape patterns (Peterson, 2002). The modestly greater drought severity and fire weather conditions during second fires compared with previous fires, even in HSF regimes (Figure 2), suggests that climate-imposed limits, once historically important in infrequent-fire regimes, are also relaxing.

Tracking trends of burn severity in future overlapping fires is increasingly important as fire activity continues to rise, because different dynamics might potentially unfold if the number of overlapping fires continues to increase. For example, short-interval fires can reduce fine- and coarse-fuel loads by more than half compared with single fires (Stevens-Rumann & Morgan, 2016; Stevens-Rumann et al., 2020; Turner et al., 2019), imposing stronger fuel limitations on additional (third or more) fires. In our dataset, there were only 13 thrice-burned landscapes with ≥1000 ha overlap for all three fires, and landscape patterns in the third fire did not exhibit a consistent qualitative difference from the first or second fire (Supporting Information Appendix S4). This suggests that so far, at least, dynamics in thrice-burned landscapes are similar to our findings in twice-burned landscapes. However, the shifting fire environment between successive fires in a given location (Figure 2) suggests that our analyses are capturing emerging dynamics of fire regimes that are on a trajectory of change (Hessburg et al., 2019), and future outcomes with more reburns under continued warming climate might differ.

4.2 | Reburn mosaics fit like puzzle pieces that alter heterogeneity and resilience

Wildfires are well known for being spatially heterogeneous, but our results demonstrate that short-interval fires produce cumulative patterns of recent stand-replacing fire that can homogenize important components of landscapes. Negative feedbacks from strong ecological memory at the local (pixel) scale (Cansler et al., 2022; Harvey et al., 2016a; Parks et al., 2014) emerge as different phenomena across burned landscapes, such that on average, more than half of the area within reburned landscapes is recently burned as stand-replacing in one fire or another (Table 1). Heterogeneous mosaics from each fire can fit together like puzzle pieces to produce a forested landscape that is considerably more homogeneous in recent burn severity patterns than either fire individually. The characterization of these emergent patterns demonstrates the importance of spatially explicit approaches for understanding the consequences of disturbance interactions (Gill et al., 2022; Peterson, 2002).

The increased homogenization of landscapes after short-interval reburns has important implications for forest resilience. Reburns reduce overall forest cover and stand age within burned landscapes, because reburn intervals (1–35 years) are much shorter than the time required for recovery of functionally 'mature' forests [often >100 years in most western North American conifer forests (McDowell et al., 2020)]. Continued erosion of remnant mature or old-growth forest patches with each subsequent fire results in post-reburn landscapes with greater distances to live tree seed sources in severely burned locations, which can slow post-fire forest regeneration (Donato et al., 2016; Harvey et al., 2016c; Kemp et al., 2016) and promote expansion of early-seral plant communities. Furthermore, if the nearest surviving forest is itself young and immature, seed dispersal into stand-replacing patches is drastically reduced (Gill et al., 2021).

The effects of short-interval reburns in reshaping forest landscapes vary among forest ecosystems and fire regimes. In LSF regimes, these emergent patterns could promote resilience to fire if they restore forest structure and composition that is better adapted to frequent fire (Larson et al., 2013). This can be especially true in locations where contemporary forest cover (before recent short-interval reburns) is greater and more homogeneous than under pre-colonial fire regimes (Hessburg et al., 2019). However, if cumulative patches of severe fire from overlapping fires exceed the capacity for recovery of dominant tree species, such patterns can instead catalyse resilience loss and transition to grassland or shrubland ecosystems (Steel et al., 2021). In HSF regimes where tree survival through fire is generally lower and post-fire seedling establishment is a dominant mechanism of resilience, emergent burn severity patterns with long distances to seed sources can substantially slow post-fire tree regeneration and forest re-establishment (Donato, Fontaine, Robinson, et al., 2009; Gill et al., 2021; Turner et al., 2019).

4.3 | Forests burned twice at high severity are widespread and have important ecological effects

Our study documents the spatial extent and landscape configuration of an ecologically important product of reburns, namely areas burned twice at high severity over a short interval. Totalling >23% of all reburned landscapes across a large study region, these locations are the high-severity counterpart to fire refugia (Collins et al., 2019; Meddens et al., 2018) and are areas where compound disturbance effects (Paine et al., 1998) can alter forest resilience. These locations can have substantially reduced post-fire tree regeneration and woody carbon stocks (Brown & Johnstone, 2012; Stevens-Rumann & Morgan, 2016; Turner et al., 2019), compositional shifts towards resprouting or precocious obligate-seeding woody plants (Donato, Fontaine, Robinson, et al., 2009; Hoecker & Turner, 2022; Whitman et al., 2019), less physical material remaining (e.g., snags, branches and litter) and extremely hot and dry microclimates that can exceed the tolerance of young post-fire tree seedlings (Hoecker et al., 2020; Whitman et al., 2019; Wolf et al., 2021). As such, areas burned severely twice in successive fires are qualitatively different from forests burned severely in either the first or the second fire only, and they represent unique conditions in post-fire landscapes. Tracking the fate of these areas through time will be important, because their different post-fire structural legacies and successional pathways might contribute important future heterogeneity to reburned landscapes. Our results can be used to identify trends in the extent and configuration of these locations as additional short-interval reburns occur in the future.

5 | CONCLUSIONS

Interacting disturbances and their emergent spatial patterns are crucial outcomes of changing disturbance regimes. Our spatially explicit analysis of overlapping fires across a broad region provides an important lens for understanding disturbance interactions. Specifically, spatial patterns of burn severity from sequential overlapping fires that are heterogeneous individually can interact in surprising ways to reshape and homogenize aspects of forest landscapes. As climate warms and more fires occur in short succession, landscape reductions in burn severity might be less common than expected, as negative spatial feedbacks between overlapping short-interval fires were ephemeral and occurred in limited contexts. Overlapping short-interval fires affect key mechanisms of forest resilience by increasing the cumulative area of recent high-severity fire, reducing the extent of residual live mature forest and increasing the distance between severely burned areas and nearby seed sources. Spatial patterns emerging from cumulative effects of interacting disturbances contribute to shifts in ecosystem dynamics as global change unfolds.

AUTHOR CONTRIBUTIONS

B.J.H and M.G.T designed the research and acquired funding; B.J.H., M.S.B., and M.G.T. performed the research; M.S.B. processed burn severity data to prepare for analysis; B.J.H. analyzed data; and B.J.H. wrote the paper with input from all authors.

ACKNOWLEDGEMENTS

This project was supported by the National Science Foundation (NSF-DEB grant 1720620 to B.J.H. and NSF-DEB grant 1719905 to M.G.T.), the United States Geological Survey Northwest Climate Adaptation Science Center (Fellowship to M.S.B.), the Joint Fire Science Program Graduate Research Innovation Award (Fellowship to M.S.B.) and the United States Forest Service Geospatial Technology Application Center (Agreement 17-CS-11130400-010). B.J.H. acknowledges support from the Jack Corkery and George Corkery Jr. Endowed Professorship in Forest Sciences. M.G.T. acknowledges support from the University of Wisconsin-Madison Vilas Trust. We thank S. Saberi for technical help with calibrating models to convert RdNBR to basal area killed by fire. We thank W.C. Addle for additional inspiration. We thank D. Abendroth for valuable insights via conversation in early stages of this project, and we thank S. Prichard, T. Gillespie and two anonymous reviewers for constructive comments on an earlier version of this manuscript.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data are available on Zenodo at: https://zenodo.org/record/7469188.

ORCID

Brian J. Harvey https://orcid.org/0000-0002-5902-4862
Michele S. Buonanduci https://orcid.org/0000-0003-3646-9954
Monica G. Turner https://orcid.org/0000-0003-1903-2822

REFERENCES

Abatzoglou, J. T. (2013). Development of gridded surface meteorological data for ecological applications and modelling. *International Journal of Climatology*, *33*, 121–131.

Abatzoglou, J. T., Battisti, D. S., Williams, A. P., Hansen, W. D., Harvey, B. J., & Kolden, C. A. (2021). Projected increases in western US forest fire despite growing fuel constraints. *Communications Earth & Environment*, *2*, 1–8.

Agee, J. K. (1993). *Fire ecology of Pacific northwest forests* (2nd ed. ed.). Island Press.

Baker, W. L. (2009). *Fire ecology in Rocky Mountain landscapes*. Island Press.

Bessie, W. C., & Johnson, E. A. (1995). The relative importance of fuels and weather on fire behavior in subalpine forests. *Ecology*, *76*, 747–762.

Bradshaw, L. S., Deeming, J. E., Burgan, R. E., & Cohen, J. D. (1984). *The 1978 National Fire-Danger Rating System: Technical documentation (general technical report INT-169)*. USDA Forest Service, Intermountain Forest and Range Experiment Sation.

Brown, C. D., & Johnstone, J. F. (2012). Once burned, twice shy: Repeat fires reduce seed availability and alter substrate constraints on Picea mariana regeneration. *Forest Ecology and Management*, *266*, 34–41.

Burton, P. J., Jentsch, A., & Walker, L. R. (2020). The ecology of disturbance interactions. *Bioscience*, *70*, 854–870.

Cansler, C. A., Kane, V. R., Hessburg, P. F., Kane, J. T., Jeronimo, S. M. A., Lutz, J. A., Povak, N. A., Churchill, D. J., & Larson, A. J. (2022). Previous wildfires and management treatments moderate subsequent fire severity. *Forest Ecology and Management*, *504*, 119764.

Cansler, C. A., & McKenzie, D. (2014). Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern Cascade Range, USA. *Ecological Applications*, *24*, 1037–1056.

Collins, B. M., Stevens, J. T., Miller, J. D., Stephens, S. L., Brown, P. M., & North, M. P. (2017). Alternative characterization of forest fire regimes: Incorporating spatial patterns. *Landscape Ecology*, *32*, 1543–1552.

Collins, L., Bennett, A. F., Leonard, S. W. J., & Penman, T. D. (2019). Wildfire refugia in forests: Severe fire weather and drought mute the influence of topography and fuel age. *Global Change Biology*, *25*, 3829–3843.

Collins, L., Bradstock, R. A., Clarke, H., Clarke, M. F., Nolan, R. H., & Penman, T. D. (2021). The 2019/2020 mega-fires exposed Australian ecosystems to an unprecedented extent of high-severity fire. *Environmental Research Letters*, *16*, 044029.

Collins, L., Clarke, H., Clarke, M. F., McColl Gausden, S. C., Nolan, R. H., Penman, T., Bradstock, R., & Varner, M. (2022). Warmer and drier conditions have increased the potential for large and severe fire seasons across South-Eastern Australia. *Global Ecology and Biogeography*, *31*, 1933–1948.

Collins, L., Hunter, A., McColl-Gausden, S., Penman, T. D., & Zylstra, P. (2021). The effect of antecedent fire severity on reburn severity and fuel structure in a resprouting eucalypt forest in Victoria, Australia. *Forests*, *12*, 450.

Coop, J. D., Parks, S. A., Stevens-Rumann, C. S., Crausbay, S. D., Higuera, P. E., Hurteau, M. D., Tepley, A., Whitman, E., Assal, T., Collins, B. M., Davis, K. T., Dobrowski, S., Falk, D. A., Fornwalt, P. J., Fulé, P. Z., Harvey, B. J., Kane, V. R., Littlefield, C. E., Margolis, E. Q., … Rodman, K. C. (2020). Wildfire-driven Forest conversion in Western North American landscapes. *Bioscience*, *70*, 659–673.

Donato, D. C., Fontaine, J. B., Campbell, J. L., Robinson, W. D., Kauffman, J. B., & Law, B. E. (2009). Conifer regeneration in stand-replacement portions of a large mixed-severity wildfire in the Klamath–Siskiyou Mountains. *Canadian Journal of Forest Research*, *39*, 823–838.

Donato, D. C., Fontaine, J. B., Kauffman, J. B., Robinson, W. D., & Law, B. E. (2013). Fuel mass and forest structure following stand-replacement fire and post-fire logging in a mixed-evergreen forest. *International Journal of Wildland Fire*, *22*, 652.

Donato, D. C., Fontaine, J. B., Robinson, W. D., Kauffman, J. B., & Law, B. E. (2009). Vegetation response to a short interval between high-severity wildfires in a mixed-evergreen forest. *Journal of Ecology*, *97*, 142–154.

Donato, D. C., Harvey, B. J., & 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.

Downing, W. M., Meigs, G. W., Gregory, M. J., & Krawchuk, M. A. (2021). Where and why do conifer forests persist in refugia through multiple fire events? *Global Change Biology*, *27*, 3642–3656.

Eidenshink, J., Schwind, B., Brewer, K., Zhu, Z., Quayle, B., & Howard, S. (2007). A project for monitoring trends in burn severity. *Fire Ecology*, *3*, 3–21.

Ellis, T. M., Bowman, D. M. J. S., Jain, P., Flannigan, M. D., & Williamson, G. J. (2022). Global increase in wildfire risk due to climate-driven declines in fuel moisture. *Global Change Biology*, *28*, 1544–1559.

Eskelson, B. N. I., & Monleon, V. J. (2018). Post-fire surface fuel dynamics in California forests across three burn severity classes. *International Journal of Wildland Fire*, *27*, 114.

Gill, N. S., Hoecker, T. J., & Turner, M. G. (2021). The propagule doesn't fall far from the tree, especially after short-interval, high-severity fire. *Ecology*, *102*, e03194.

Gill, N. S., Turner, M. G., Brown, C. D., Glassman, S. I., Haire, S. L., Hansen, W. D., Pansing, E. R., St Clair, S. B., & Tomback, D. F. (2022). Limitations to propagule dispersal will constrain postfire recovery of plants and fungi in Western coniferous forests. *Bioscience*, *72*, 347–364.

Greene, D. F., & Johnson, E. A. (1989). A model of wind dispersal of winged or plumed seeds. *Ecology*, *70*, 339–347.

Harvey, B. J., Andrus, R. A., & Anderson, S. C. (2019). Incorporating biophysical gradients and uncertainty into burn severity maps in a temperate fire-prone forested region. *Ecosphere*, *10*, e02600.

Harvey, B. J., Donato, D. C., & Turner, M. G. (2016a). Burn me twice, shame on who? Interactions between successive forest fires across a temperate mountain region. *Ecology*, *97*, 2272–2282.

Harvey, B. J., Donato, D. C., & Turner, M. G. (2016b). Drivers and trends in landscape patterns of stand-replacing fire in forests of the US northern Rocky Mountains (1984–2010). *Landscape Ecology*, *31*, 2367–2383.

Harvey, B. J., Donato, D. C., & Turner, M. G. (2016c). High and dry: Post-fire tree seedling establishment in subalpine forests decreases with post-fire drought and large stand-replacing burn patches. *Global Ecology and Biogeography*, *25*, 655–669.

Haugo, R. D., Kellogg, B. S., Cansler, C. A., Kolden, C. A., Kemp, K. B., Robertson, J. C., Metlen, K. L., Vaillant, N. M., & Restaino, C. M. (2019). The missing fire: Quantifying human exclusion of wildfire in Pacific northwest forests, USA. *Ecosphere*, *10*(4), 1–16.

Hessburg, P. F., Churchill, D. J., Larson, A. J., Haugo, R. D., Miller, C., Spies, T. A., North, M. P., Povak, N. A., Belote, R. T., Singleton, P. H., Gaines, W. L., Keane, R. E., Aplet, G. H., Stephens, S. L., Morgan, P., Bisson, P. A., Rieman, B. E., Salter, R. B., & Reeves, G. H. (2015). Restoring fire-prone inland Pacific landscapes: Seven core principles. *Landscape Ecology*, *30*, 1805–1835.

Hessburg, P. F., Miller, C. L., Parks, S. A., Povak, N. A., Taylor, A. H., Higuera, P. E., Prichard, S. J., North, M. P., Collins, B. M., Hurteau, M. D., Larson, A. J., Allen, C. D., Stephens, S. L., Rivera-Huerta, H., Stevens-Rumann, C. S., Daniels, L. D., Gedalof, Z., Gray, R. W., Kane, V. R., … Salter, R. B. (2019). Climate, environment, and disturbance history govern resilience of Western North American forests. *Frontiers in Ecology and Evolution*, *7*, 1–27.

Hijmans, R.J., van Etten, J., Sumner, M., Cheng, J., Baston, D., Bevan, A., Bivand, R., Busetto, L., Canty, M., Fasoli, B., Forrest, D., Ghosh, A., Golicher, D., Gray, J., Greenberg, J. A., Hiemstra, P., Hingee, K., Ilich, A., Institute for Mathematics Applied Geosciences, … Wuees, R. (2022). *raster: Geographic data analysis and modeling*. https://cran.r-project.org/web/packages/raster/index.html

Hobbins, M. T., Wood, A., McEvoy, D. J., Huntington, J. L., Morton, C., Anderson, M., & Hain, C. (2016). The evaporative demand drought index. Part I: Linking drought evolution to variations in evaporative demand. *Journal of Hydrometeorology*, *17*, 1745–1761.

Hoecker, T. J., Hansen, W. D., & Turner, M. G. (2020). Topographic position amplifies consequences of short-interval stand-replacing fires on postfire tree establishment in subalpine conifer forests. *Forest Ecology and Management*, *478*, 118523.

Hoecker, T. J., & Turner, M. G. (2022). A short-interval reburn catalyzes departures from historical structure and composition in a Mesic mixed-conifer forest. *Forest Ecology and Management*, *504*, 119814.

Johnstone, J. F., Allen, C. D., Franklin, J. F., Frelich, L. E., Harvey, B. J., Higuera, P. E., Mack, M. C., Meentemeyer, R. K., Metz, M. R., Perry, G. L. W., Schoennagel, T., & Turner, M. G. (2016). Changing disturbance regimes, ecological memory, and forest resilience. *Frontiers in Ecology and the Environment*, *14*, 369–378.

Keeley, J. E. (2009). Fire intensity, fire severity and burn severity: A brief review and suggested usage. *International Journal of Wildland Fire*, *18*, 116.

Kemp, K. B., Higuera, P. E., & Morgan, P. (2016). Fire legacies impact conifer regeneration across environmental gradients in the U.S. northern Rockies. *Landscape Ecology*, *31*, 619–636.

LANDFIRE. (2016). Landfire: Landfire fire regime group layer.

Larson, A. J., Belote, R. T., Cansler, C. A., Parks, S. A., & Dietz, M. S. (2013). Latent resilience in ponderosa pine forest: Effects of resumed frequent fire. *Ecological Applications*, *23*, 1243–1249.

Laughlin, M. M., Bakker, J. D., Churchill, D. J., Gregory, M. J., DeMeo, T., Alvarado, E. C., & Harvey, B. J. (2023). Trends in forest structure restoration need over three decades with increasing wildfire activity in the interior Pacific northwest US. *Forest Ecology and Management*, *527*, 120607.

McDowell, N. G., Allen, C. D., Anderson-Teixeira, K., Aukema, B. H., Bond-Lamberty, B., Chini, L., Clark, J. S., Dietze, M., Grossiord, C., Hanbury-Brown, A., Hurtt, G. C., Jackson, R. B., Johnson, D. J., Kueppers, L., Lichstein, J. W., Ogle, K., Poulter, B., Pugh, T. A. M., Seidl, R., … Xu, C. (2020). Pervasive shifts in forest dynamics in a changing world. *Science*, *368*, eaaz9463.

McEvoy, D. J., Hobbins, M., Brown, T. J., VanderMolen, K., Wall, T., Huntington, J. L., & Svoboda, M. (2019). Establishing relationships between drought indices and wildfire danger outputs: A test case for the California-Nevada drought early warning system. *Climate*, *7*, 52.

Meddens, A. J. H., Kolden, C. A., Lutz, J. A., Smith, A. M. S., Cansler, C. A., Abatzoglou, J. T., Meigs, G. W., Downing, W. M., & Krawchuk, M. A. (2018). Fire refugia: What are they, and why do they matter for global change? *Bioscience*, *68*(12), 944–954.

Menning, K. M., Battles, J. J., & Benning, T. L. (2007). Quantifying change in distributions: A new departure index that detects, measures and describes change in distributions from population structures, size-classes and other ordered data. *Oecologia*, *154*, 75–84.
Miller, J. D., Safford, H. D., Crimmins, M., & Thode, A. E. (2009). Quantitative evidence for increasing Forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. *Ecosystems*, *12*, 16–32.
Miller, J. D., & Thode, A. E. (2007). Quantifying burn severity in a heterogeneous landscape with a relative version of the delta normalized burn ratio (dNBR). *Remote Sensing of Environment*, *109*, 66–80.
Morgan, P., Keane, R. E., Dillon, G. K., Jain, T. B., Hudak, A. T., Karau, E. C., Sikkink, P. G., Holden, Z. A., & Strand, E. K. (2014). Challenges of assessing fire and burn severity using field measures, remote sensing and modelling. *International Journal of Wildland Fire*, *23*, 1045.
Morris, J. E., Buonanduci, M. S., Agne, M. C., Battaglia, M. A., & Harvey, B. J. (2022). Does the legacy of historical thinning treatments foster resilience to bark beetle outbreaks in subalpine forests? *Ecological Applications*, *32*, e02474.
Nelson, K. N., Turner, M. G., Romme, W. H., & Tinker, D. B. (2016). Landscape variation in tree regeneration and snag fall drive fuel loads in 24-year old post-fire lodgepole pine forests. *Ecological Applications*, *26*, 2424–2438.
Nelson, K. N., Turner, M. G., Romme, W. H., & Tinker, D. B. (2017). Simulated fire behaviour in young, postfire lodgepole pine forests. *International Journal of Wildland Fire*, *26*, 852.
Nolan, R. H., Boer, M. M., Collins, L., Resco de Dios, V., Clarke, H., Jenkins, M., Kenny, B., & Bradstock, R. A. (2020). Causes and consequences of eastern Australia's 2019–20 season of mega-fires. *Global Change Biology*, *26*, 1039–1041.
Paine, R. T., Tegner, M. J., & Johnson, E. A. (1998). Compounded perturbations yield ecological surprises. *Ecosystems*, *1*, 535–545.
Parks, S. A. (2014). Mapping day-of-burning with coarse-resolution satellite fire-detection data. *International Journal of Wildland Fire*, *23*, 215.
Parks, S. A., & Abatzoglou, J. T. (2020). Warmer and drier fire seasons contribute to increases in area burned at high severity in Western US forests from 1985 to 2017. *Geophysical Research Letters*, *47*, e2020GL089858.
Parks, S. A., Holsinger, L. M., Voss, M., Loehman, R., & Robinson, N. (2018). Mean composite fire severity metrics computed with Google earth engine offer improved accuracy and expanded mapping potential. *Remote Sensing*, *10*, 879.
Parks, S. A., Miller, C., Nelson, C. R., & Holden, Z. A. (2014). Previous fires moderate burn severity of subsequent wildland fires in two large Western US wilderness areas. *Ecosystems*, *17*, 29–42.
Pausas, J. G., & Fernández-Muñoz, S. (2012). Fire regime changes in the Western Mediterranean Basin: From fuel-limited to drought-driven fire regime. *Climatic Change*, *110*, 215–226.
Pausas, J. G., & Keeley, J. E. (2021). Wildfires and global change. *Frontiers in Ecology and the Environment*, *19*, 387–395.
Pebesma, E. (2018). Simple features for R: Standardized support for spatial vector data. *The R Journal*, *10*, 439–446.
Peterson, G. D. (2002). Contagious disturbance, ecological memory, and the emergence of landscape pattern. *Ecosystems*, *5*, 329–338.
Prichard, S. J., Stevens-Rumann, C. S., & Hessburg, P. F. (2017). Tamm review: Shifting global fire regimes: Lessons from reburns and research needs. *Forest Ecology and Management*, *396*, 217–233.
R Core Team. (2021). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing.
Reilly, M. J., Dunn, C. J., Meigs, G. W., Spies, T. A., Kennedy, R. E., Bailey, J. D., & Briggs, K. (2017). Contemporary patterns of fire extent and severity in forests of the Pacific northwest, USA (1985-2010). *Ecosphere*, *8*, e01695.
Reilly, M. J., Zuspan, A., Halofsky, J. S., Raymond, C., McEvoy, A., Dye, A. W., Donato, D. C., Kim, J. B., Potter, B. E., Walker, N., Davis, R. J., Dunn, C. J., Bell, D. M., Gregory, M. J., Johnston, J. D., Harvey, B. J., Halofsky, J. E., & Kerns, B. K. (2022). Cascadia burning: The historic, but not historically unprecedented, 2020 wildfires in the Pacific northwest, USA. *Ecosphere*, *13*, e4070.
Rollins, M. G. (2009). LANDFIRE: A nationally consistent vegetation, wildland fire, and fuel assessment. *International Journal of Wildland Fire*, *18*, 235.
Saberi, S. J. (2019). *Quantifying burn severity in forests of the interior Pacific northwest: From field measurements to satelite spectral indices*. [M.S. Thesis]. University of Washington.
Saberi, S. J., Agne, M. C., Harvey, B. J., Saberi, S. J., Agne, M. C., & Harvey, B. J. (2022). Do you CBI what I see? The relationship between the composite burn index and quantitative field measures of burn severity varies across gradients of forest structure. *International Journal of Wildland Fire*, *31*, 112–123.
Seidl, R., Thom, D., Kautz, M., Martin-Benito, D., Peltoniemi, M., Vacchiano, G., Wild, J., Ascoli, D., Petr, M., Honkaniemi, J., Lexer, M. J., Trotsiuk, V., Mairota, P., Svoboda, M., Fabrika, M., Nagel, T. A., & Reyer, C. P. O. (2017). Forest disturbances under climate change. *Nature Climate Change*, *7*, 395–402.
Seidl, R., & Turner, M. G. (2022). Post-disturbance reorganization of forest ecosystems in a changing world. *Proceedings of the National Academy of Sciences of the United States of America*, *119*, e2202190119.
Simard, M., Romme, W. H., Griffin, J. M., & Turner, M. G. (2011). Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? *Ecological Monographs*, *81*, 3–24.
Steel, Z. L., Foster, D., Coppoletta, M., Lydersen, J. M., Stephens, S. L., Paudel, A., Markwith, S. H., Merriam, K., & Collins, B. M. (2021). Ecological resilience and vegetation transition in the face of two successive large wildfires. *Journal of Ecology*, *109*, 3340–3355.
Stevens, J. T., Collins, B. M., Miller, J. D., North, M. P., & Stephens, S. L. (2017). Changing spatial patterns of stand-replacing fire in California conifer forests. *Forest Ecology and Management*, *406*, 28–36.
Stevens, J. T., Kling, M. M., Schwilk, D. W., Varner, J. M., & Kane, J. M. (2020). Biogeography of fire regimes in western U.S. conifer forests: A trait-based approach. *Global Ecology and Biogeography*, *29*, 944–955.
Stevens-Rumann, C., & Morgan, P. (2016). Repeated wildfires alter forest recovery of mixed-conifer ecosystems. *Ecological Applications*, *26*, 1842–1853.
Stevens-Rumann, C. S., Hudak, A. T., Morgan, P., Arnold, A., & Strand, E. K. (2020). Fuel dynamics following wildfire in US northern Rockies forests. *Frontiers in Forests and Global Change*, *3*, 51.
Turner, M. G. (2010). Disturbance and landscape dynamics in a changing world. *Ecology*, *91*, 2833–2849.
Turner, M. G., Braziunas, K. H., Hansen, W. D., & Harvey, B. J. (2019). Short-interval severe fire erodes the resilience of subalpine lodgepole pine forests. *Proceedings of the National Academy of Sciences of the United States of America*, *116*, 11319–11328.
Turner, M. G., & Romme, W. H. (1994). Landscape dynamics in crown fire ecosystems. *Landscape Ecology*, *9*, 59–77.
Whitman, E., Parisien, M.-A., Thompson, D. K., & Flannigan, M. D. (2019). Short-interval wildfire and drought overwhelm boreal forest resilience. *Scientific Reports*, *9*, 18796.
Whitman, E., Parks, S. A., Holsinger, L. M., & Parisien, M.-A. (2022). Climate-induced fire regime amplification in Alberta, Canada. *Environmental Research Letters*, *17*, 055003.
Wickham, H. (2016). *ggplot2: Elegant graphics for data analysis*. https://link.springer.com/book/10.1007/978-3-319-24277-4#bibliographic-information
Wolf, K. D., Higuera, P. E., Davis, K. T., & Dobrowski, S. Z. (2021). Wildfire impacts on forest microclimate vary with biophysical context. *Ecosphere*, *12*(5), 1–16.

BIOSKETCH

Brian J. Harvey is an Assistant Professor and the Jack Corkery and George Corkery Jr. Endowed Professor in Forest Sciences at the University of Washington, School of Environmental and Forest Sciences. His research focuses on fire and disturbance ecology in forested landscapes at spatial scales ranging from trees to regions, using a combination of field studies, remotely sensed geospatial datasets and spatially explicit landscape analyses. His work builds an understanding of the nature of forest disturbances, how disturbance regimes are changing and multiple disturbances interact, and how forest structure and function are shaped by disturbances and climate change.

SUPPORTING INFORMATION

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

How to cite this article: Harvey, B. J., Buonanduci, M. S., & Turner, M. G. (2023). Spatial interactions among short-interval fires reshape forest landscapes. *Global Ecology and Biogeography*, *32*, 586–602. https://doi.org/10.1111/geb.13634