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ORIGINAL ARTICLE

Repeated High-severity Fire in the Sierra Nevada and Southern Cascades of California, United States: Landscape Trends and Belowground Effects

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Abstract

Repeated high-severity fire is threatening forest resilience in dry and mesic forests of the Sierra Nevada and southern Cascades of California, USA. While impacts to plant communities have been described, the belowground effects of this fire regime change are less known. Here, we: (1) describe trends in reburn activity in the Sierra Nevada and southern Cascades, (2) conduct a case study on the effect of repeated high-severity fire on soil biogeochemistry within Plumas National Forest, California, and (3) propose a conceptual framework to describe hypothetical belowground effects of repeated high-severity fire on soil biogeochemistry and the soil microbiome by drawing on our case study and available literature. We found that yearly ≤ 20 y reburn area (yearly burned area that has also burned within 20 y prior) and yearly ≤ 20-y *high-severity* reburn area (where both initial and repeated burn events were of high severity) are increasing across the Sierra Nevada and Southern Cascades. Our framework suggests that soil carbon and nitrogen (in bulk soil and soil organic matter fractions) may be resistant to additional decline with recurrent fire. However, declining post-fire inorganic N pulses with recurrent fire, the inability of soil organic matter to recover between fires, and an altered microbial community may impact ecosystem recovery. These findings may be pertinent to dry and mesic forests beyond our study region.

Key words: severe wildfire; reburn; soil organic matter; montane chaparral; ecosystem resilience; biogeochemistry.

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Highlights

- Severe reburn activity is increasing in the Sierra Nevada and southern Cascades.
- Soil C and N resisted continued loss with reburns but may not recover between fires.
- Changes to SOM substrate could impact microbial communities and soil fluxes.

Introduction

The occurrence of large and extreme wildfires is increasing in forests of the Western US (Dennison and others 2014; Abatzoglou and others 2021; Hagmann and others 2021). While fire is an integral component of many Western landscapes, increasing wildfire activity beyond the historical range of variability for a given region can hinder forest resilience (Coop and others 2020). As such, there is growing concern over the fate of Western forests (Turner 2010; Johnstone and others 2016; Stevens-Rumann and others 2018).

Forest wildfire regimes are often categorized as fuel- or climate-limited, with tradeoffs between expected fire recurrence and severity (defined here as degree of fire-induced plant mortality or soil damage; Agee 1998; Keeley 2009; Steel and others 2015), though forests often display characteristics of both fuel and climate limitation over larger geographic and temporal scales (Agee 1998; Safford and Stevens 2017). The self-limiting nature of wildfire in forests (Hurteau and others 2019) should prevent widespread frequent *and* severe fire. However, repeated high-severity fire is impacting conifer forests in both historically climate-limited fire regimes (for example, subalpine lodgepole (*Pinus contorta*) forests; Turner and others 2019, 2025) and in those with mixed fire regimes across the West (for example, dry and mesic conifer forests of the Rocky, Sierra Nevada, and Klamath mountain ranges; Donato and others 2009, 2016; Miller and others 2009; Buma and others 2020; Hoecker and Turner 2022; Paudel and others 2022). With short-interval severe fire, forests risk conversion to a non-forested shrub- or grass-dominated state that is maintained by positive fire-vegetation feedbacks (Coppoletta and others 2015).

Repeated high-severity fire may prevent forest recovery through a combination of aboveground and belowground mechanisms. Aboveground mechanisms for long-term conversion to a non-forested state include limited time for conifer regeneration between fire events, post-fire microclimatic conditions that are not conducive to conifer reestablishment, and increased conifer propagule limitation (Johnstone and others 2016; Urza and Sibold 2017; Rodman and others 2020; Nolan and others 2021; Prichard and others 2021; Gill and others 2022). Belowground, recovery may be further inhibited by changes in soil carbon (C) and nitrogen (N) cycling (Agbeshie and others 2022) and the soil microbiome. For example, because C and N both have relatively low oxidation temperatures, the combustion of soil C and N in shallow surface soils can reduce soil organic matter (SOM) for decades after severe fire (Wan and others 2001; Knicker 2007; Nave and others 2011; Dove and others 2020). Soil organic matter provides several services that might impact ecosystem recovery, such as soil structure (which prevents erosion), water holding capacity (which confers drought resilience), and nutrient storage (which confers resources to recovering vegetation; Blanco-Canqui and others 2013; Bossio and others 2020).

The effects of frequent, low-severity fire or infrequent, high-severity fire on aboveground and belowground ecosystem properties and processes have been synthesized and reviewed extensively (for example, Covington and Sackett 1986; Smithwick and others 2005; Johnson and others 2012; Dove and others 2020, 2022; Pellegrini and Jackson 2020; Pellegrini and others 2020, 2021, 2022). We lack sufficient information, at present, to thoroughly review or synthesize the impacts of frequent severe fire on forest ecosystem properties or processes–especially belowground. Therefore, to encourage more research on this emergent topic, we identified three needs: (1) determine the spatial scope of the problem, (2) collect data showing possible belowground impacts of frequent severe fire, and (3) generate hypotheses that can be tested with subsequent research based on available data and what is known from other fire regimes. Here, we focus on the Sierra Nevada and southern Cascades of California, USA, a region which is vulnerable to repeated high-severity fire due to dense forests, steep terrain, and reliance on snowpack for soil and fuel moisture. Severe reburn activity has been described in the region (Coppoletta and others 2015; Lydersen and others 2019; Paudel and others 2022), but not yet quantified. We first analyze landscape trends in reburn activity to characterize the spatial and temporal extent of reburning in the Sierra Nevada and southern Cascades. We then present a case study on the impacts of recurrent high-severity fire on soil biogeochemical pools and fluxes on the Plumas National Forest, California, in which we characterized soil C and N storage in

different SOM fractions, soil CO_2 efflux (representative of heterotrophic microbial respiration), and inorganic N concentrations and cycling. We intend the case study to provide detailed insight into potential belowground responses to repeated high-severity fire in a subregion of the Sierra Nevada – a baseline of data to which future research can be compared. We then combine the case study results with extant literature to generate broader hypotheses regarding how repeated high-severity fires may influence soil properties and processes in conifer forests in and beyond the Sierra Nevada and southern Cascades.

Methods

Study Region

The Sierra Nevada and Southern Cascades of California have a historically mixed-severity fire regime (Hessburg and others 2016) and are comprised of varied forest types including mixed-conifer forest and upper montane forests, as well as montane chaparral (Airey Lauvaux and others 2016). Montane chaparral is dominated by sclerophyllous shrubs (for example, Greenleaf manzanita or *Arctostaphylos patula*) and found in areas with arid environmental conditions, such as dry, south-facing slopes, or in recent high-severity burn patches, where it is viewed as early successional habitat (Nagel and others 2005; Hessburg and others 2016).

Repeated high severity fire in this region has been attributed, in part, to past fire exclusion (lessened fuel limitation) which left large areas prone to severe fire during periods of prolonged drought or extreme fire weather conditions (lessened climate limitation; Parks and others 2015b; Prichard and others 2021). With typical wildfire limitations diminished, fires have burned past typical barriers, including into recently burned areas that might otherwise limit fuel connectivity (Parks and others 2015a; Taylor and others 2022). Severely burned areas often consist of standing and downed snags and, early in succession, chaparral shrubs. Both land cover types can carry subsequent fires (Coppoletta and others 2015; Lydersen and others 2017, 2019). Successive burning can hinder conifer recovery, promoting long-term conversion to a chaparral-dominated state (Collins and others 2007; Coppoletta and others 2015; Grabinski and others 2017; Davis and others 2020, 2024; Paudel and others 2022).

Spatial Analysis

To quantify forest area of the Sierra Nevada and Southern Cascades region experiencing recurrent wildfire, we analyzed publicly available fire perimeter and fire severity data from the Monitoring Trends in Burn Severity (MTBS) database (Eidenshink and others 2007). The MTBS database provides multiple metrics of fire severity, including a five-class thematic map of burn severity used here (classes include unburned to low severity, low severity, moderate severity, high severity, and increased greenness). The burn severity classes are generated through a threshold analysis of differenced normalized burn ratio (dNBR) which correlates with field-based assessments of burn severity both aboveground and belowground. While the severity classifications have limitations (Kolden and others 2015), they are useful for analyzing trends across broad spatial scales.

We used the MTBS data to calculate annual $\leq$ 20 y reburn area, or the area of land burned each year in wildland fire that had also burned in wildland fire within 20 y prior, as well as annual $\leq$ 20 y *high-severity* reburn area, or the area of land burned in wildland fire at high-severity each year that had also burned in wildland fire at high-severity within 20 y prior. For example, the $\leq$ 20 y reburn area for 2013 represents the area burned in 2013 that had also burned at some point between 1993 and 2013. It is noteworthy that our analyses only include areas that burned twice within a 20-year period, and additional areas that burned more than twice could lead to higher estimates of reburn area. The MTBS database currently includes the 1984–2024 fire years, which enabled us to calculate $\leq$ 20 y reburn area for 2004–2024. We conducted generalized linear regressions (Gamma family with log link) of each reburn area metric over time. All spatial and regression analyses were conducted in R version 4.5.0 (R Core Team 2025) and maps (Figure 1) were generated using ArcGIS Pro version 3.5 (Environmental Systems Research Institute (ESRI) 2025).

Case Study

Given the limited information on how repeated high-severity fire influences soil biogeochemistry, we conducted a case study in Plumas National Forest, California by comparing a chronosequence (that is, space-for-time substitution; Pickett 1989) of sites at varying stages of recovery from an initial high-severity fire event (1, 3, 15, and 100 + y post-fire) as well as two sites at early stages of recovery

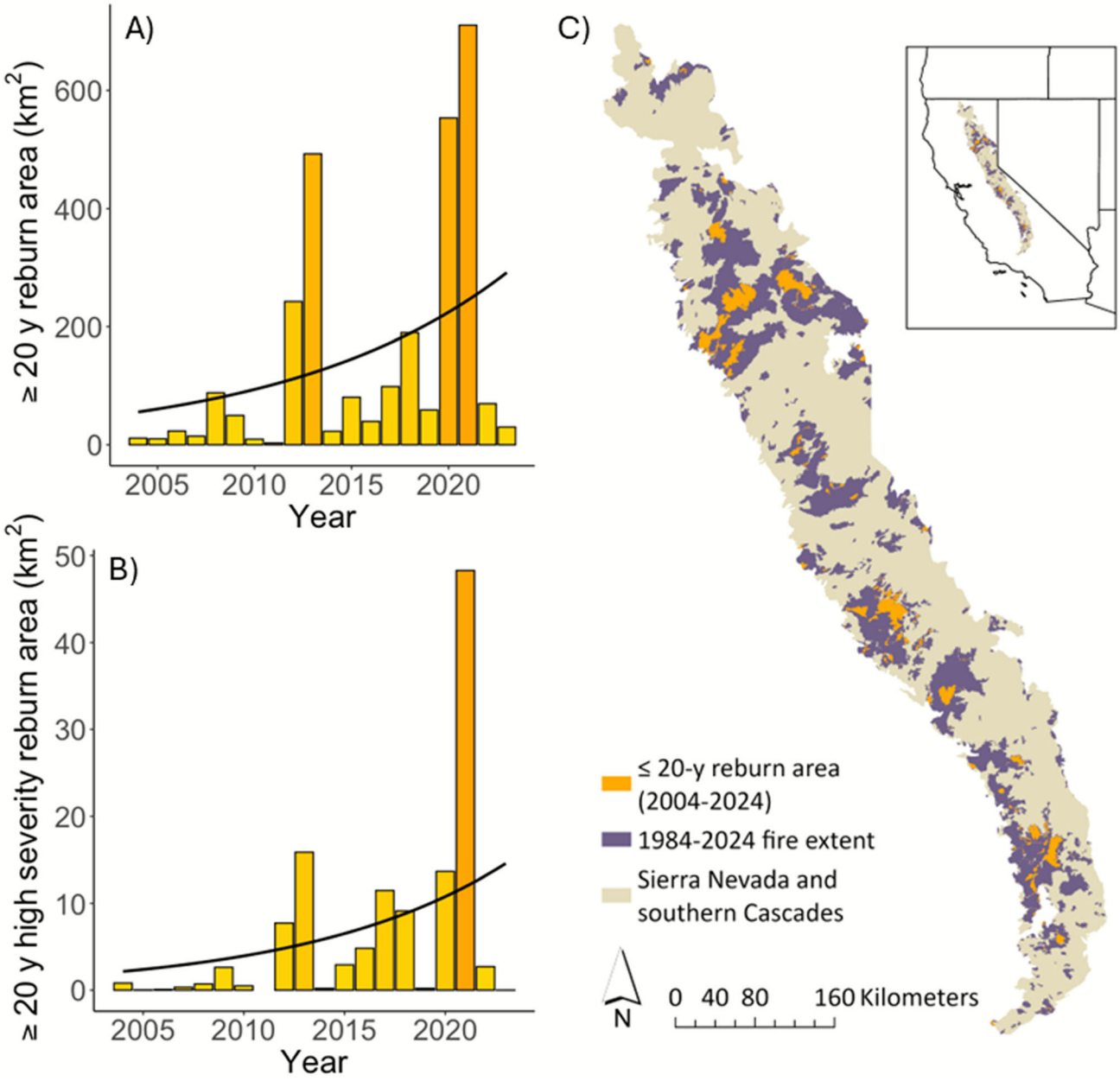


Figure 1. Trends in ≤ 20-y reburn area (area that has burned twice within 20 y; km^2; Panel **A**; colors scaled by reburn area) or ≤ 20-y high-severity reburn area (area that has burned at high-severity twice within 20 y; km^2; Panel **B**; colors scaled by reburn area) over time (2004–2024) and a depiction of ≤ 20-y reburn area extent (Panel **C**) for the Sierra Nevada and Southern Cascades bioregions. In Panel C, ≤ 20-y reburn area (orange) is shown over the extent of all wildfires that occurred between 1984 and 2024 (purple).

from a second high-severity fire event (1 and 3-y post-fire) that occurred within 15-y of the initial fire event (Figure 2; Appendix 1, Table S1). The single-burn chronosequence allowed us to examine soil recovery without repeated fire (as in, for example, Ross and others 2012; Kurth and others 2014; Dove and others 2020, 2022), providing region-specific context needed to determine how soil recovery after two severe fire events differs from soil recovery after one severe fire event.

Using a combination of remote sensing imagery, differenced normalized burn ratio (dNBR), and field validation, we identified six sites with different burn histories: sites that burned at high severity once in 2007 (hereafter 15-y single-burn), 2019 (hereafter 3-y single-burn), or 2021 (hereafter 1-y single-burn), sites that burned both in 2007 and

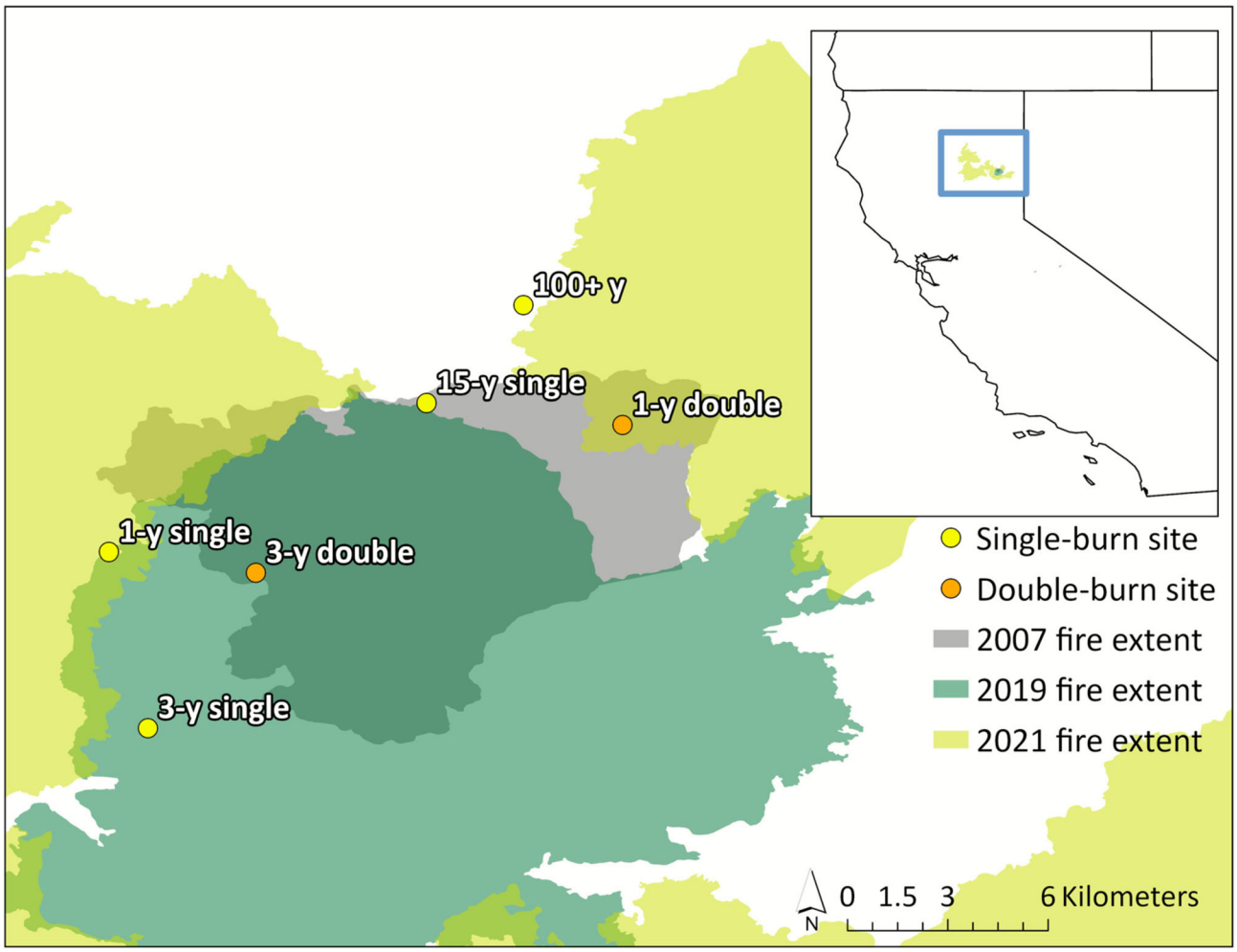


Figure 2. Case study plot locations laid over the approximate fire extent for the 2021 Dixie Fire (1-y prior to sampling), the 2019 Walker Fire (3-y prior to sampling), and the 2007 Wheeler Fire (15-y prior to sampling; part of the Antelope Complex). Each plot burned at high severity either once (for single-burn sites) or twice (for double-burn sites). Burn history was field validated at all sites. To see where plots are located on differenced normalized burn ratio (dNBR) maps for each fire, see Appendix 1, Fig. S1.

2019 (hereafter 3-y double-burn) or in 2007 and 2021 (hereafter 1-y double-burn), and a site that had not burned in at least 100 years (hereafter 100 + y single-burn; Figure 2). While there was substantial overlap among these fires (Figure 2), overlapping areas that burned at high severity in each fire while also retaining similar site characteristics were rare (see Appendix 1, Figure S1 for sites overlaid on dNBR maps of each fire), preventing us from identifying multiple sites for each burn history. Despite this limitation, we had the rare opportunity to study contrasting fire activity at six sites with similar state factors (climate, topography, soil characteristics, and vegetation [based on pre-fire remote imagery and surrounding unburned forests]) located within 13 km of each other.

The single-burn chronosequence included a mature mixed-conifer forest (100 + y since fire), a Greenleaf manzanita (*Arctostaphylos patula*)-dominated forest with some conifer regeneration (15-y since fire), and two sites at early stages of recovery, dominated by herbaceous vegetation (3-y since fire) or still devoid of live vegetation (1-y since fire). The 15-, 3-, and 1-y single-burn sites also each had abundant downed and/or standing snags. The 3-y double-burn site was dominated by herbaceous vegetation, similar to its single-burn counterpart. The 1-y double-burn site had patchy herbaceous vegetation. Both double-burn sites also included top-killed Greenleaf manzanita and evidence of snag consumption during the last fire (for example, log-shaped areas of complete combustion on the ground surrounded by ash). Site soil, cli-

matic, topographic, and additional plant community characteristics are described in Appendix 1, Table S1 and Appendix 2.

Detailed descriptions of all field sample collection and laboratory methods can be found in Appendix 2. Briefly, in June of 2022, we collected six 0–5 cm soil samples from each site (including bulk density samples). We then measured CO_2 production, nitrate (NO_3^-), and ammonium (NH_4^+) concentration, and net nitrification through a 7-day incubation of field moist soil (Hart and others 1994; Drury and others 2008) as well as soil C storage in particulate organic matter (POM; sand) and mineral-associated organic matter (MAOM; clay and silt) using soil size fractionations for each sample (Christensen 2001; Poeplau and others 2018).

We conducted statistical analyses in R version 4.5.0 (R Core Team, 2025) with statistical significance set a priori to $\alpha = 0.05$. We used analysis of variance (ANOVA) and Tukey post hoc tests to determine differences among single-burn sites for each measured variable. We used Welch's T-tests to determine differences between the 1-y single- and double-burn sites and between the 3-y single- and double-burn sites. We focused on C and N concentrations for these analyses, but bulk density and stock information can be found in Appendix 1, Table S2.

Results

Characterizing Regional Changes in Reburn Activity

We found an exponential increase in ≤ 20 y reburn area (irrespective of fire severity) across the Sierra Nevada and Southern Cascades (Figure 1A; $\beta = 0.16$, SE = 0.05, t = 3.18, $p = 0.005$) between 2004 and 2024. There was also an exponential increase in ≤ 20 y *high-severity* reburn area (considering both initial and repeat fire severity; Figure 1B; $\beta = 0.21$, SE = 0.05, t = 4.37, $p < 0.001$), providing evidence that repeated high-severity fire has increased across the Sierra Nevada and Southern Cascades. Figure 1C includes a map of ≤ 20 y reburn area throughout the region, illustrating that both burned areas and reburned areas have occurred throughout the study region.

Repeated High-Severity Fire Effects on Soil in Plumas National Forest

When we compare patterns across the chronosequence that experienced only single fires, soil C and N concentration (both in bulk soil and in different soil fractions) and cumulative R_h were each higher in the 100 + y site (longest time since fire) than all other sites, where soil properties did not differ (Figure 3; Appendix 1, Tables S3-S5). Particulate C and N were, on average, 53% and 62% lower, respectively, and mineral-associated C and N were 38% and 50% lower, respectively, in the three most recently burned sites compared to the 100 + y site. Conversely, soil NH_4^+, NO_3^-, and net nitrification were each elevated in the 1-y site (shortest time since fire) compared to all other sites (Figure 3; Appendix 1, Tables S3 and S5).

Soil C and N concentrations (both in bulk soil and in different soil fractions) were similar between single- and double-burn sites (1-y or 3-y post-fire; Figure 3, Appendix 1, Table S4). Cumulative R_h was lower in the 3-y double-burn site than the 3-y single-burn site (Welch's t = 2.84, $p = 0.03$). The 1-y double-burn site had lower soil NH_4^+ (Welch's t = 3.43, $p = 0.007$) but higher soil NO_3^- (Welch's t = − 2.4, $p = 0.05$) than the 1-y single-burn site. Plots for soil C and N concentrations in each soil fraction can be found in Appendix 1, Figure S2.

Discussion

We identified landscape trends in reburn activity across the Sierra Nevada and southern Cascades to determine the extent to which repeated high-severity fire may threaten forest recovery. Positive trends in both ≤ 20 y reburn area and ≤ 20 y high-severity reburn area provide, to our knowledge, the first evidence of these increases throughout the region. Both trends appeared strongly influenced by the 2020 and 2021 fire years, which were historically large fire years for California but also part of a decades-long trend in increasing fire activity (Keeley and Syphard 2021; Safford and others 2022). This suggests that large areas of forest may be at increasing risk of conversion to a shrub- or grass-dominated state that could change forest soils over time. In particular, our case study identified that recurrent severe fire may prevent the slow recovery of SOM and the critical ecosystem services it provides (Smith and others 2013; Scharlemann and others 2014), impact forest ecosystem C and N budgets, and potentially impact the soil microbial community.

Soil biogeochemical properties may not change uniformly with repeated high-severity fires, even within a given location. Our case study provided a glimpse into one set of outcomes, though a wide range of soil responses might be possible (Figure 4). We posit that possible responses of soil biogeochemical properties can be summarized in one of

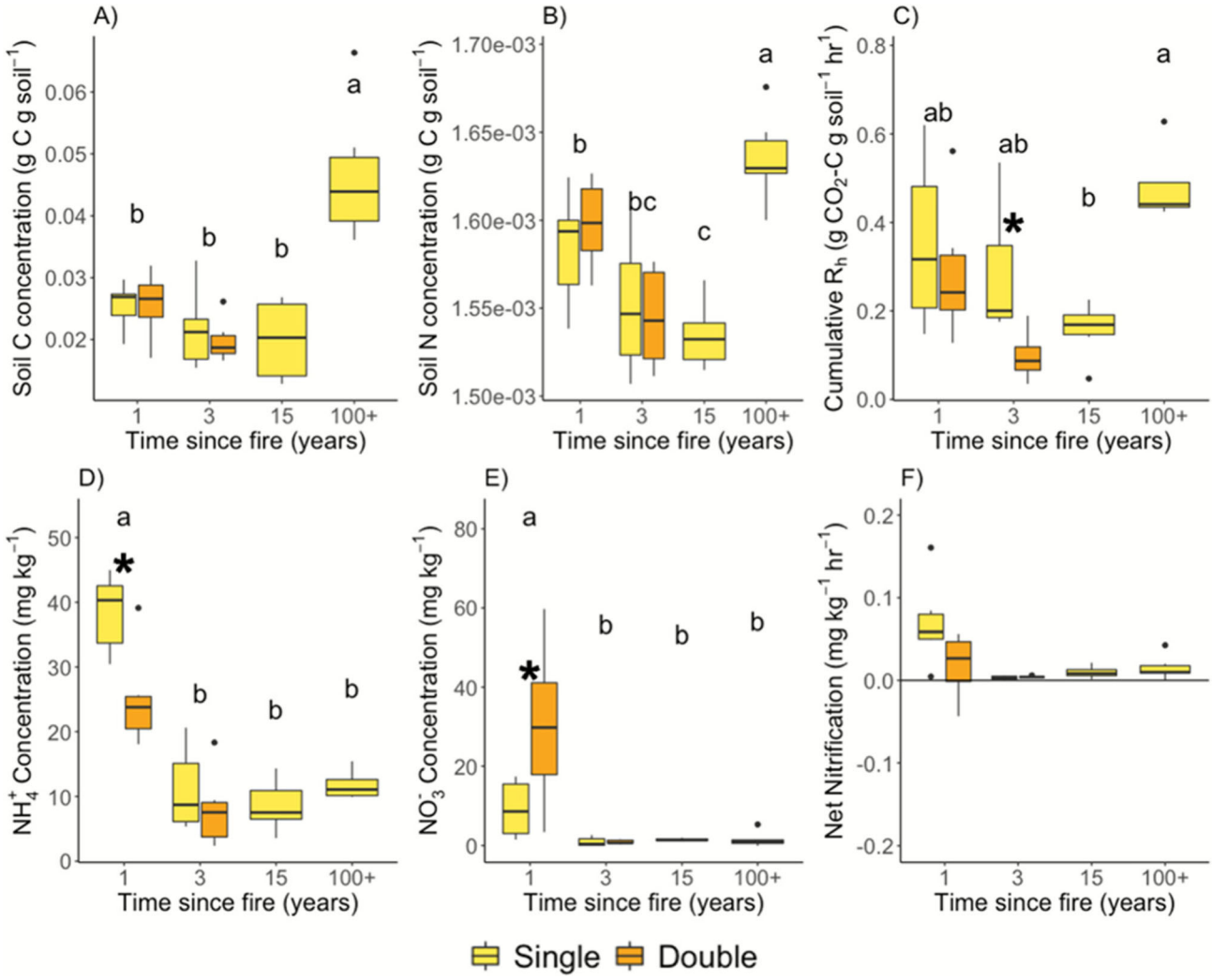
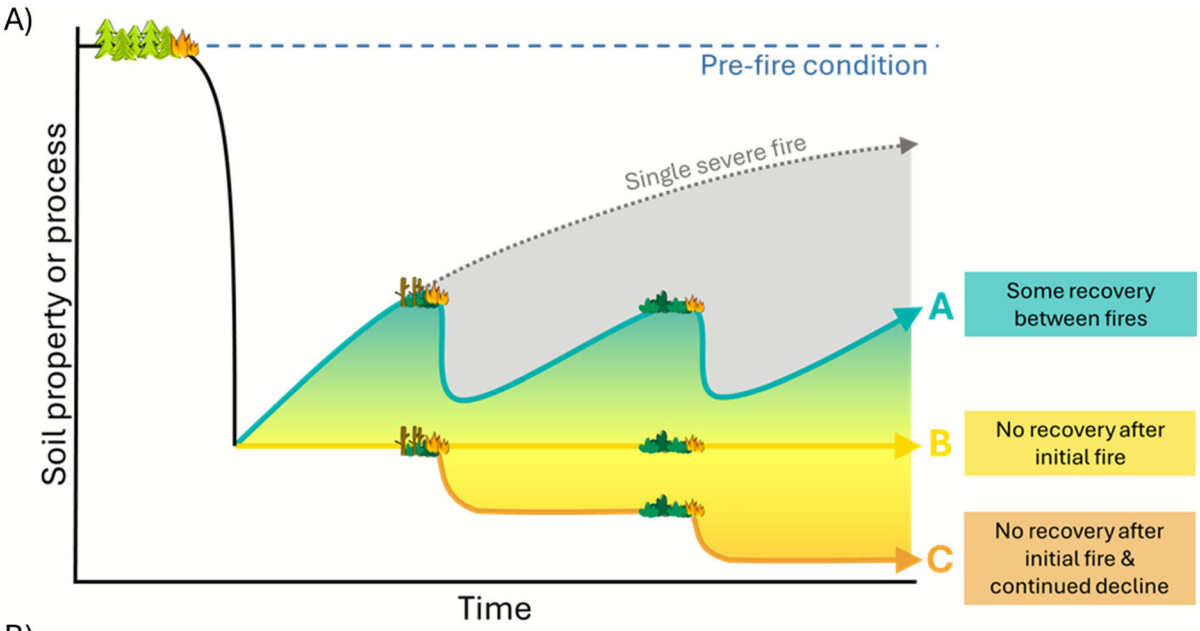


Figure 3. Soil total carbon (C; Panel **A**) and nitrogen (N; Panel **B**) concentrations, cumulative soil heterotrophic respiration (R_h; Panel **C**), ammonium (NH_4^+) concentration (Panel **D**), nitrate (NO_3^-) concentration (Panel **E**), and net nitrification (Panel **F**) of each site. The letters signify differences across single-burn sites (ANOVA) while asterisks (*) signify significant differences between single- and double-burn sites (Welch's T-Test) where $p \leq 0.05$.

three ways: soil properties that are resistant to loss beyond the effect of the initial high-severity fire and display some degree of resilience between fire events (Path A; Figure 4), soil properties that are resistant to decline beyond the effect of the initial high-severity fire but unable to recover between fire events (Path B; Figure 4), or soil properties that experience continued decline with repeated burning until a lower limit is reached (Path C; Figure 4). We will discuss examples of soil properties that may follow each of these pathways below, which are also listed in Figure 4B.

Given the slow recovery of SOM after severe fire (Dove and others 2020), there may be non-detectable soil organic C and N recovery between successive fire events under repeated high-severity fire, corresponding with path B. While SOM losses through fire are often limited to the top centimeters of soil, it can take several decades for the recovery of plant C inputs to allow shallow surface SOM to accumulate to pre-fire levels (Nave and others 2011; Dove and others 2020). We observed evidence of this for C and N in both bulk soil and soil fractions, where: (1) C and N concentrations showed no sign of recovery by 15-y post-fire and, (2) single- and double-burn sites (1-y or 3-y since fire) did not differ. The findings of our case study also align with a recent study by Turner and others (2025) in the greater Yellowstone ecosystem (Wyoming, USA). The authors hypothesized a ''ratcheting down'' effect of short-interval, stand-replacing fire on *ecosystem* N stocks. However, they

A)



B)

Soil property	Potential single severe fire effect		Hypothesized repeat severe fire effect
Soil carbon (C) and nitrogen (N)	Immediate and substantial losses (upper few cm), with several decades needed for recovery.	Johnson and others 2012; Miesel and others 2018; Dove and others 2020	With limited time for soil recovery between fires, additional losses of C and N (bulk soil, MAOM, or POM) and the ability of C and N to recover to a pre-fire condition are unlikely (Path B). Continued decline could occur if downed debris or vegetation after an initial fire increase soil heating effects in a repeat fire (Path C).
Mineral-associated and particulate organic matter (MAOM and POM)	Losses focused in POM, though MAOM losses possible with substantial soil heating.	Ulery and others 1996; Merino and others 2021; Pellegrini and others 2022	
Microbial community composition	Functional diversity declines while heat tolerant taxa become competitive.	Glassman and others 2016; Dove and others 2022; Nelson and others 2022	Repeated mortality of less fire-resistant organisms could lead to continued decline in microbial richness and functional diversity (Path C).
Decomposition and soil carbon flux	Fire-driven mortality and reduced/altered soil substrate reduces the abundance and activity of soil heterotrophs.	Dore and others 2008; Sullivan and others 2011; Ross and others 2012	Partial microbial recovery (especially taxa that are competitive post-fire) may increase microbial biomass and respiration between fires (Path A).

Figure 4. Panel A depicts pathways of repeated high-severity fire effects on soils (paths A–C) that prevent soil biogeochemical properties or processes from returning to a pre-fire condition (blue dashed line), as they eventually might after a single severe fire (gray-dotted line). Each fire event corresponds with a vegetation and fire depiction. Path A represents properties that are resistant to loss beyond the effect of the initial high-severity fire and display resilience between fire events. Path B represents properties that are resistant to decline beyond the effect of the initial high-severity fire but unable to recover between fire events. Path C represents properties that can experience continued decline with repeated burning until a lower limit is reached. The area between the pathways is shaded to represent the full range of possible outcomes. Panel **B** describes the potential effects of single severe fires on commonly measured soil biogeochemical effects (with example references) as well as hypotheses for potential repeated severe fire effects that correspond to Panel **A**. For a comprehensive review of single fire effects on soils, see Certini (2005), Knicker (2007), Nave and others (2011), Wang and others (2012), Pellegrini and Jackson (2020), or Agbeshie and others (2022).

instead found that soil N stocks were resistant to continued loss, which buffered effects on ecosystem N. Nevertheless, some degree of continued decline in soil C and N, corresponding with path C, could still occur in some forest types if the addition of downed snags or early successional vegetation (like chaparral) after an initial fire increases surface fuel loads and leads to additional soil heating effects in a repeat fire.

Fire-driven SOM losses are unlikely to be evenly distributed throughout SOM fractions. Particulate organic matter (POM) is often more abundant in forest soils and less resistant to fire than mineral-associated organic matter (MAOM; Pellegrini and others 2022), which is physically and chemically protected from loss by occlusion in fine aggregates and by sorption to mineral surfaces (Lavallee and others 2020; Sokol and others 2022). Losses of MAOM through fire can still occur, however, depending on the magnitude of soil heating and corresponding changes to mineral composition and soil aggregation (Ulery and others 1996; Merino and others 2021). The single-burn chronosequence aligned with this understanding, with greater relative losses of POM C and N than of MAOM C and N.

While MAOM and POM fractions experienced different degrees of C and N loss after the initial severe fire event relative to unburned conditions, both fractions were resistant to continued decline after the second fire event (path B). A shift in the relative abundance of SOM fractions could affect post-fire soil function, as MAOM is considered less easily accessible by microbes (given its protection mechanisms) than POM (Lavallee and others 2020; Daly and others 2021; Islam and others 2022) and has a lower C:N ratio, attributed in part to the microbial products that contribute to it (Sokol and others 2019; Whalen and others 2022). As such, recovery of the microbial community under re-burns could be especially important for MAOM recovery.

Severe fire can also influence soil N cycling (Turner and others 2007; Hanan and others 2016; Brady and others 2022). The deposition of NH_4^+-rich ash, loss of pine needle litter, and reduced plant N uptake in post-fire soils can result in elevated rates of nitrification that may last from a few years in mixed-conifer forests (Dove and others 2020) to decades in Ponderosa pine forests (Deluca and Sala 2006; Kurth and others 2014). Increased nitrification and NO_3^- concentrations can leave recently burned forests susceptible to additional N losses through leaching over time (Hanan and others 2017; Gustine and others 2022).

With repeated high-severity fire, the possibility of reduced ecosystem organic N after each successive fire could lead to progressively less mineralizable N and thus continuously smaller post-fire NH_4^+ pulses. This was observed in our case study, where soil NH_4^+ concentrations were higher in the 1-y single-burn site than the 1-y double-burn site. Net nitrification rates were also slightly higher in the 1-y single-burn site than the 1-y double-burn site (though not significantly), potentially reflecting differences in the abundance of NH_4^+ substrate. Soil NO_3^-, on the other hand, was higher after repeated burning (1-y post-fire). This discrepancy may just reflect different rates of leaching and runoff between sites (Murphy and others 2006; Johnson and others 2009), though soil NO_3^- was also higher 5 years after repeated short-interval fire than repeated long-interval fire in the greater Yellowstone ecosystem (Turner and others 2025). If N becomes less tightly cycled with repeated burning, the potential for N export off-site could further prevent soil total C and N recovery and reinforce the likelihood of soil C and N remaining on path B.

Nitrogen fixation can make up for fire-driven ecosystem N losses over the course of decades (Johnson and others 2012). This may not occur everywhere, however, as the deposition of NH_4^+-rich ash can prevent N-fixing bacteria or early successional plant symbionts (for example, *Lupinus* spp.) from fixing N during the immediate post-fire period where light and other resources are less limiting (Vitousek and Howarth 1991; Smithwick and others 2005; Turner and others 2007). For example, in a mixed broadleaf-conifer forest in Klamath National Forest, the presence of early successional N-fixing *Ceonothus spp.* did not drive a rapid recovery of ecosystem N stocks; instead, it was estimated that upwards of 100 years between fires would be needed for a return to pre-fire N conditions via asymbiotic N-fixer activity (Yelenik and others 2013).

Repeated high-severity fire may have broader impacts on soil microbial communities, extending beyond microbes associated with N cycling, as the microbial community adapts to ecosystem-level changes in fire activity and vegetation dynamics (Hamman and others 2008; Wang and others 2012; Lombao and others 2020; Barreiro and Díaz-Raviña 2021). The most direct impacts to microbes would occur through soil heating (Wang and others 2012; Fernández-García and others 2019a, 2019b), while changes in substrate availability (Barreiro and Díaz-Raviña 2021; Roth and others 2023) and the soil environment (for example, soil pH; Caiafa and others 2023) could act as indirect effects.

Severe fire has been shown to reduce microbial abundance, richness (Caiafa and others 2023), and diversity (Adkins and others 2020) in Western forests, with some recovery over the course of 10 years (Caiafa and others 2023). In the Sierra Nevada, it appears that two or more decades may be necessary for the altered microbial community to return to a pre-fire condition (Certini 2005; Dove and others 2022). With repeated high-severity fire, microbial communities could become increasingly dominated by fire-resistant organisms that are resistant to soil heating, grow rapidly, or can degrade pyrogenic C while recovery of less fire-resistant organisms may be continuously stunted by reburning C (Glassman and others 2016; Nelson and others 2022; Revillini and others 2025). As such, it is possible that the partial recovery of microbial communities between fire events may leave proxies of overall microbial activity (for example, microbial biomass and soil respiration) best represented by path A while microbial richness and functional diversity may be represented by path C. Reductions in microbial abundance could align with the somewhat lower mean cumulative R_h values observed in the double-burn site than the single-burn site of our case study.

The loss of ectomycorrhizal fungi (EMF) with repeated burning would have particularly important implications for forest recovery, as EMF are critical for water and nutrient transport in regenerating conifers (Franco and others 2014; Anthony and others 2022). Ectomycorrhizal fungi populations can be reduced by severe fire (Caiafa and others 2023), though certain taxa are better adapted to survive high temperatures than others (''fire fungi''; Glassman and others 2016). We posit that the persistence of EMF despite novel reburn activity could play an important role in the resilience of Sierra Nevada and southern Cascades forests into the future, but we lack sufficient information with which to hypothesize a potential path associated with EMF.

As research attempts to keep pace with accelerating fire regime change in the Sierra Nevada and southern Cascades, there is an emergent need to understand whether there are generalizable effects of repeated high-severity fire on soils. Research focused on soil heating effects between initial and subsequent severe fire events may be critical to address this uncertainty, as fire intensity (energy release; Keeley 2009) and soil heating effects may differ between initial and repeat fire events despite consistently high severity (biomass consumption; Keeley 2009). For example, while it is likely that less live fuel is present aboveground in a repeat fire event relative to an initial fire event in the Sierra Nevada, the burning of early successional chaparral vegetation (Marion and others 1991) and dead forest vegetation (for example, snags and downed logs; Monsanto and Agee 2008) can substantially raise surface soil temperatures. Similarly, while there could be less forest floor litter present in a repeat fire event relative to an initial fire event, the fuel structure of montane chaparral exists close to the soil surface. Soil moisture is also likely to differ between initial and repeat fire events based on differences in forest structure and evapotranspiration, with implications for heat transfer down the soil profile as wet soils take more energy to heat and are more conductive than dry soils (Smits and others 2016).

Conclusions

We demonstrate that the spatial extent of high-severity reburning has increased across the Sierra Nevada and southern Cascades in the last decades. Repeated high-severity fire may be impactful for forest resilience across the region as a result of both above and belowground effects. Our chronosequence-based measurements that included multiple severe fires demonstrated instances of reburn effects differing from initial fire effects for different soil properties. However, the variables we measured were not comprehensive, nor was the study area necessarily representative of the entire region in which severe reburns are occurring. We utilized extant literature on fire effects from single severe fires and, where possible, repeat severe fires to expand upon our own observations and hypothesize different biogeochemical trajectories if severe fires continue to occur at high frequencies in the region. While our results may provide an empirical and conceptual baseline, we intend this work to encourage more research on the effects of repeated high-severity fire in the Sierra Nevada and southern Cascades as well as other forested regions with intensifying reburn activity.

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DATA AVAILABILITY

The spatial and climate datasets used in this study are publicly available through the Monitoring Trends in Burn Severity program (mtbs.gov) and Climate Engine (climateengine.org). The spatial analyses used are publicly available on Zenodo (https://doi.org/10.5281/zenodo.13174285). The case study data are publicly available on Dryad (https://doi.org/10.5061/dryad.gmsbcc2xd).

REFERENCES

Abatzoglou JT, Battisti DS, Williams AP, Hansen WD, Harvey BJ, Kolden CA. 2021. Projected increases in western US forest fire despite growing fuel constraints. Commun Earth Environ 2.

Adkins J, Docherty KM, Gutknecht JLM, Miesel JR. 2020. How do soil microbial communities respond to fire in the intermediate term? Investigating direct and indirect effects associated with fire occurrence and burn severity. Sci Total Environ 745.

Agbeshie AA, Abugre S, Atta-Darkwa T, Awuah R. 2022. A review of the effects of forest fire on soil properties. J for Res (Harbin) 33:1419–1441.

Agee JK. 1998. The Landscape Ecology of Western Forest Fire Regimes. Northw Sci 72:24–34.

Airey Lauvaux C, Skinner CN, Taylor AH. 2016. High severity fire and mixed conifer forest-chaparral dynamics in the southern Cascade Range, USA. For Ecol Manage 363:74–85.

Anthony MA, Crowther TW, van der Linde S, Suz LM, Bidartondo MI, Cox F, Schaub M, Rautio P, Ferretti M, Vesterdal L, De Vos B, Dettwiler M, Eickenscheidt N, Schmitz A, Meesenburg H, Andreae H, Jacob F, Dietrich HP, Waldner P, Gessler A, Frey B, Schramm O, van den Bulk P, Hensen A, Averill C. 2022. Forest tree growth is linked to mycorrhizal fungal composition and function across Europe. ISME J 16:1327–1336.

Barreiro A, Díaz-Raviña M. 2021. Fire impacts on soil microorganisms: Mass, activity, and diversity. Curr Opin Environ Sci Health 22:100264. https://doi.org/10.1016/j.coesh.2021.100264.

Blanco-Canqui H, Shapiro CA, Wortmann CS, Drijber RA, Mamo M, Shaver TM, Ferguson RB. 2013. Soil organic carbon: The value to soil properties. J Soil Water Conserv 68.

Bossio DA, Cook-Patton SC, Ellis PW, Fargione J, Sanderman J, Smith P, Wood S, Zomer RJ, von Unger M, Emmer IM, Griscom BW. 2020. The role of soil carbon in natural climate solutions. Nat Sustain 3:391–398.

Brady MK, Hanan EJ, Dickinson MB, Miesel JR, Wade L, Greenberg J. 2022. How interactions between wildfire and seasonal soil moisture fluxes drive nitrogen cycling in Northern Sierra Nevada forests. Int J Wildl Fire 31:786–798.

Buma B, Weiss S, Hayes K, Lucash M. 2020. Wildland fire reburning trends across the US West suggest only short-term negative feedback and differing climatic effects. Environ Res Lett 15.

Caiafa M V., Nelson AR, Borch T, Roth HK, Fegel TS, Rhoades CC, Wilkins MJ, Glassman SI. 2023. Distinct fungal and bacterial responses to fire severity and soil depth across a ten-year wildfire chronosequence in beetle-killed lodgepole pine forests. For Ecol Manage 544.

Certini G. 2005. Effects of fire on properties of forest soils: A review. Oecologia 143:1–10.

Christensen BT. 2001. Physical fractionation of soil and structural and functional complexity in organic matter turnover. Eur J Soil Sci 52:345–353.

Collins BM, Kelly M, Van Wagtendonk JW, Stephens SL. 2007. Spatial patterns of large natural fires in Sierra Nevada wilderness areas. Landsc Ecol 22:545–557.

Coop JD, Parks SA, Stevens-Rumann CS, Crausbay SD, Higuera PE, Hurteau MD, Tepley A, Whitman E, Assal T, Collins BM, Davis KT, Dobrowski S, Falk DA, Fornwalt PJ, Fulé PZ, Harvey BJ, Kane VR, Littlefield CE, Margolis EQ, North M, Parisien MA, Prichard S, Rodman KC. 2020. Wildfire-driven forest conversion in western north american landscapes. Bioscience 70:659–673.

Coppoletta M, Merriam KE, Collins BM. 2015. Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecol Appl 26:686–699.

Covington WW, Sackett SS. 1986. Effect of periodic burning on soil nitrogen concentrations in ponderosa pine. Soil Sci Soc Am J 50:452–457.

Daly AB, Jilling A, Bowles TM, Buchkowski RW, Frey SD, Kallenbach CM, Keiluweit M, Mooshammer M, Schimel JP, Grandy AS. 2021. A holistic framework integrating plant-microbe-mineral regulation of soil bioavailable nitrogen. Biogeochemistry. https://doi.org/10.1007/s10533-021-00793-9.

Davis KT, Higuera PE, Dobrowski SZ, Parks SA, Abatzoglou JT, Rother MT, Veblen TT. 2020. Fire-catalyzed vegetation shifts in ponderosa pine and Douglas-fir forests of the western United States. Environ Res Lett 15.

Davis KT, Peeler J, Fargione J, Haugo RD, Metlen KL, Robles MD, Woolley T. 2024. Tamm review: A meta-analysis of thinning, prescribed fire, and wildfire effects on subsequent wildfire severity in conifer dominated forests of the Western US. For Ecol Manage 561.

Deluca TH, Sala A. 2006. Frequent Fire Alters Nitrogen Transformations in Ponderosa Pine Stands of the Inland Northwest. Ecology 87:2511–2522.

Dennison PE, Brewer SC, Arnold JD, Moritz MA. 2014. Large wildfire trends in the western United States, 1984–2011. Geophys Res Lett 41:2928–2933.

Donato DC, Fontaine JB, Campbell JL. 2016. Burning the legacy? Influence of wildfire reburn on dead wood dynamics in a temperate conifer forest. Ecosphere 7.

Donato DC, Fontaine JB, Robinson WD, Kauffman JB, Law BE. 2009. Vegetation response to a short interval between high-severity wildfires in a mixed-evergreen forest. Journal of Ecology 97:142–154.

Dove NC, Safford HD, Bohlman GN, Estes BL, Hart SC. 2020. High-severity wildfire leads to multi-decadal impacts on soil biogeochemistry in mixed-conifer forests. Ecol Appl 30.

Dove NC, Taş N, Hart SC. 2022. Ecological and genomic responses of soil microbiomes to high-severity wildfire: linking community assembly to functional potential. ISME J.

Drury CF, Hart SC, Yang XM. 2008. Nitrification techniques for soils. In: Carter MR, Gregorich EG, Eds. Soil sampling and methods of analysis, 2nd edn. Boca Raton, FL: Canadian Society of Soil Science. pp 495–513.

Eidenshink J, Schwind B, Brewer K, Zhu Z-L, Quayle B, Howard S. 2007. A project for monitoring trends in burn severity. http://www.fi

Environmental Systems Research Institute (ESRI). 2025. ArcGIS Pro.

Fernández-García V, Marcos E, Fernández-Guisuraga JM, Taboada A, Suárez-Seoane S, Calvo L. 2019a. Impact of burn severity on soil properties in a Pinus pinaster ecosystem immediately after fire. Int J Wildl Fire 28:354–364.

Fernández-García V, Miesel J, Baeza MJ, Marcos E, Calvo L. 2019b. Wildfire effects on soil properties in fire-prone pine ecosystems: Indicators of burn severity legacy over the medium term after fire. Appl Soil Ecol 135:147–156. https://doi.org/10.1016/j.apsoil.2018.12.002.

Franco AR, Sousa NR, Ramos MA, Oliveira RS, Castro PML. 2014. Diversity and Persistence of Ectomycorrhizal Fungi and Their Effect on Nursery-Inoculated Pinus pinaster in a Post-fire Plantation in Northern Portugal. Microb Ecol 68:761–772.

Gill NS, Turner MG, Brown CD, Glassman SI, Haire SL, Hansen WD, Pansing ER, Clair SBST, Tomback DF. 2022. Limitations to propagule dispersal will constrain postfire recovery of plants and fungi in Western Coniferous Forests. XX:1–18.

Glassman SI, Levine CR, Dirocco AM, Battles JJ, Bruns TD. 2016. Ectomycorrhizal fungal spore bank recovery after a severe forest fire: Some like it hot. ISME J 10:1228–1239.

Grabinski ZS, Sherriff RL, Kane JM. 2017. Controls of reburn severity vary with fire interval in the Klamath Mountains, California. USA: Ecosphere. p 8.

Gustine RN, Hanan EJ, Robichaud PR, Elliot WJ. 2022. From burned slopes to streams: how wildfire affects nitrogen cycling and retention in forests and fire-prone watersheds. Biogeochemistry 157:51–68.

Hagmann RK, Hessburg PF, Prichard SJ, Povak NA, Brown PM, Fulé PZ, Keane RE, Knapp EE, Lydersen JM, Metlen KL, Reilly MJ, Sánchez Meador AJ, Stephens SL, Stevens JT, Taylor AH, Yocom LL, Battaglia MA, Churchill DJ, Daniels LD, Falk DA, Henson P, Johnston JD, Krawchuk MA, Levine CR, Meigs GW, Merschel AG, North MP, Safford HD, Swetnam TW, Waltz AEM. 2021. Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests.

Hamman ST, Burke IC, Knapp EE. 2008. Soil nutrients and microbial activity after early and late season prescribed burns in a Sierra Nevada mixed conifer forest. For Ecol Manage 256:367–374.

Hanan EJ, Schimel JP, Dowdy K, D'Antonio CM. 2016. Effects of substrate supply, pH, and char on net nitrogen mineralization and nitrification along a wildfire-structured age gradient in chaparral. Soil Biol Biochem 95:87–99.

Hanan EJ, Tague CN, Schimel JP. 2017. Nitrogen cycling and export in California chaparral: The role of climate in shaping ecosystem responses to fire. Ecol Monogr 87:76–90.

Hart SC, Nason GE, Myrold DD, Perry DA. 1994. Dynamics of gross nitrogen transformations in an old-growth forest: the carbon connection. Ecology 75:880–891.

Hessburg PF, Spies TA, Perry DA, Skinner CN, Taylor AH, Brown PM, Stephens SL, Larson AJ, Churchill DJ, Povak NA, Singleton PH, McComb B, Zielinski WJ, Collins BM, Salter RB, Keane JJ, Franklin JF, Riegel G. 2016. Tamm review: management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. For Ecol Manage 366:221–250. https://doi.org/10.1016/j.foreco.2016.01.034.

Hoecker TJ, Turner MG. 2022. A short-interval reburn catalyzes departures from historical structure and composition in a mesic mixed-conifer forest. For Ecol Manage 504:119814. https://doi.org/10.1016/j.foreco.2021.119814.

Hurteau MD, Liang S, Westerling ALR, Wiedinmyer C. 2019. Vegetation-fire feedback reduces projected area burned under climate change. Sci Rep 9.

Islam MdR, Singh B, Dijkstra FA. 2022. Stabilisation of soil organic matter: interactions between clay and microbes. Biogeochemistry 160:145–158. https://doi.org/10.1007/s10533-022-00956-2.

Johnson DW, Miller WW, Susfalk RB, Murphy JD, Dahlgren RA, Glass DW. 2009. Biogeochemical cycling in forest soils of the eastern Sierra Nevada Mountains, USA. For Ecol Manage 258:2249–2260.

Johnson DW, Walker RF, McNulty M, Rau BM, Miller WW. 2012. The long-term effects of wildfire and post-fire vegetation on sierra nevada forest soils. Forests 3:398–416.

Johnstone JF, Allen CD, Franklin JF, Frelich LE, Harvey BJ, Higuera PE, Mack MC, Meentemeyer RK, Metz MR, Perry GLW, Schoennagel T, Turner MG. 2016. Changing disturbance regimes, ecological memory, and forest resilience. Front Ecol Environ 14:369–378.

Keeley JE. 2009. Fire intensity, fire severity and burn severity: A brief review and suggested usage. Int J Wildl Fire 18:116–126.

Keeley JE, Syphard AD. 2021. Large California wildfires: 2020 fires in historical context. Fire Ecol 17.

Knicker H. 2007. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. Biogeochemistry 85:91–118.

Kolden CA, Smith AMS, Abatzoglou JT. 2015. Limitations and utilisation of monitoring trends in burn severity products for assessing wildfire severity in the USA. Int J Wildland Fire 24:1023–1028.

Kurth VJ, Hart SC, Ross CS, Kaye JP, Fulé PZ. 2014. Stand-replacing wildfires increase nitrification for decades in southwestern ponderosa pine forests. Oecologia 175:395–407.

Lavallee JM, Soong JL, Cotrufo MF. 2020. Conceptualizing soil organic matter into particulate and mineral-associated forms to address global change in the 21st century. Glob Chang Biol 26:261–273.

Lombao A, Barreiro A, Fontúrbel MT, Martín A, Carballas T, Díaz-Raviña M. 2020. Key factors controlling microbial community responses after a fire: Importance of severity and recurrence. Sci Total Environ 741.

Lydersen JM, Collins BM, Brooks ML, Matchett JR, Shive KL, Povak NA, Kane VR, Smith DF. 2017. Evidence of fuels management and fire weather influencing fire severity in an extreme fire event. Ecol Appl 27:2013–2030.

Lydersen JM, Collins BM, Coppoletta M, Jaffe MR, Northrop H, Stephens SL. 2019. Fuel dynamics and reburn severity following high-severity fire in a Sierra Nevada, USA, mixed-conifer forest. Fire Ecol 15.

Marion GM, Moreno JM, Oechel WC. 1991. Fire severity, ash deposition, and clipping effects on soil nutrients in chaparral. Soil Sci Soc Am J 55:235–240.

Merino A, García-Oliva F, Fontúrbel MT, Vega JA. 2021. The high content of mineral-free organic matter in soils increases

their vulnerability to wildfire in humid-temperate zones. Geoderma 395.

Miller JD, Safford HD, Crimmins M, Thode AE. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. Ecosystems 12:16–32.

Monsanto PG, Agee JK. 2008. Long-term post-wildfire dynamics of coarse woody debris after salvage logging and implications for soil heating in dry forests of the eastern Cascades, Washington. For Ecol Manage 255:3952–3961.

Murphy JD, Johnson DW, Miller WW, Walker RF, Blank RR. 2006. Prescribed fire effects on forest floor and soil nutrients in a Sierra Nevada forest. Soil Sci 171:181–199.

Nagel TA, Taylor Alan H. 2005. Fire and persistence of montane chaparral in mixed conifer forest landscapes in the northern Sierra Nevada, Lake Tahoe Basin, California, USA 1. J Torrey Bot Soc 132:442–457. https://doi.org/10.3159/1095-5674(2005)132[442:FAPOMC]2.0.CO;2.

Nave LE, Vance ED, Swanston CW, Curtis PS. 2011. Fire effects on temperate forest soil C and N storage. Ecol Appl 21:1189–1201.

Nelson AR, Narrowe AB, Rhoades CC, Fegel TS, Daly RA, Roth HK, Chu RK, Amundson KK, Young RB, Steindorff AS, Mondo SJ, Grigoriev IV, Salamov A, Borch T, Wilkins MJ. 2022. Wildfire-dependent changes in soil microbiome diversity and function. Nat Microbiol 7:1419–1430.

Nolan RH, Collins L, Leigh A, Ooi MKJ, Curran TJ, Fairman TA, Resco de Dios V, Bradstock R. 2021. Limits to post-fire vegetation recovery under climate change. Plant Cell Environ 44:3471–3489.

Parks SA, Holsinger LM, Miller C, Nelson CR. 2015a. Wildland fire as a self-regulating mechanism: The role of previous burns and weather in limiting fire progression. Ecol Appl 25:1478–1492.

Parks SA, Miller C, Parisien MA, Holsinger LM, Dobrowski SZ, Abatzoglou J. 2015b. Wildland fire deficit and surplus in the western United States, 1984–2012. Ecosphere 6.

Paudel A, Coppoletta M, Merriam K, Markwith SH. 2022. Persistent composition legacy and rapid structural change following successive fires in Sierra Nevada mixed conifer forests. For Ecol Manage 509.

Pellegrini AFA, Caprio AC, Georgiou K, Finnegan C, Hobbie SE, Hatten JA, Jackson RB. 2021. Low-intensity frequent fires in coniferous forests transform soil organic matter in ways that may offset ecosystem carbon losses. Glob Chang Biol 27:3810–3823.

Pellegrini AFA, Harden J, Georgiou K, Hemes KS, Malhotra A, Nolan CJ, Jackson RB. 2022. Fire effects on the persistence of soil organic matter and long-term carbon storage. Nat Geosci 15:5–13.

Pellegrini AFA, Hobbie SE, Reich PB, Jumpponen A, Brookshire ENJ, Caprio AC, Coetsee C, Jackson RB. 2020. Repeated fire shifts carbon and nitrogen cycling by changing plant inputs and soil decomposition across ecosystems. Ecol Monogr 90:1–20.

Pellegrini AFA, Jackson RB. 2020. The long and short of it: a review of the timescales of how fire affects soils using the pulse-press framework. Advances in ecological research. Vol. 62. Academic Press Incpp 147–171.

Pickett STA. 1989. Space-for-time substitution as an alternative to long-term studies. In: Likens GE, Ed. Long-term studies in ecology, . New York: Springer-Verlag.

Poeplau C, Don A, Six J, Kaiser M, Benbi D, Chenu C, Cotrufo MF, Derrien D, Gioacchini P, Grand S, Gregorich E, Griepentrog M, Gunina A, Haddix M, Kuzyakov Y, Kühnel A, Macdonald LM, Soong J, Trigalet S, Vermeire ML, Rovira P, van Wesemael B, Wiesmeier M, Yeasmin S, Yevdokimov I, Nieder R. 2018. Isolating organic carbon fractions with varying turnover rates in temperate agricultural soils – A comprehensive method comparison. Soil Biol Biochem 125:10–26. https://doi.org/10.1016/j.soilbio.2018.06.025.

Prichard SJ, Hessburg PF, Hagmann RK, Povak NA, Dobrowski SZ, Hurteau MD, Kane VR, Keane RE, Kobziar LN, Kolden CA, North M, Parks SA, Safford HD, Stevens JT, Yocom LL, Churchill DJ, Gray RW, Huffman DW, Lake FK, Khatri-Chhetri P. 2021. Adapting western North American forests to climate change and wildfires: 10 common questions. Ecol Appl 31:1–30.

R Core Team. 2025. R: a language and environment for statistical computing.

Revillini D, Searcy CA, Afkhami ME. 2025. 50-year fire legacy regulates soil microbial carbon and nutrient cycling responses to new fire. Soil Biol Biochem 208.

Rodman KC, Veblen TT, Battaglia MA, Chambers ME, Fornwalt PJ, Holden ZA, Kolb TE, Ouzts JR, Rother MT. 2020. A changing climate is snuffing out post-fire recovery in montane forests. Glob Ecol Biogeogr 29:2039–2051.

Ross CS, Kaye JP, Kaye MW, Kurth VJ, Brimmer R, Hart SC, Fulé PZ. 2012. Ecosystem carbon remains low for three decades following fire and constrains soil CO_2 responses to precipitation in southwestern ponderosa pine forests. Ecosystems 15:725–740.

Roth HK, McKenna AM, Simpson MJ, Chen H, Srikanthan N, Fegel TS, Nelson AR, Rhoades CC, Wilkins MJ, Borch T. 2023. Effects of burn severity on organic nitrogen and carbon chemistry in high-elevation forest soils. Soil Environ Health 1.

Safford HD, Paulson AK, Steel ZL, Young DJN, Wayman RB. 2022. The 2020 California fire season: a year like no other, a return to the past or a harbinger of the future? Glob Ecol Biogeogr 31:2005–2025.

Safford HD, Stevens JT. 2017. Natural range of variation for yellow pine and mixed-conifer forests in the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests, California, USA. General Technical Report - Pacific Southwest Research Station, USDA Forest Service PSW-GTR-256:229.

Scharlemann JPW, Tanner EVJ, Hiederer R, Kapos V. 2014. Global soil carbon: Understanding and managing the largest terrestrial carbon pool. Carbon Manag 5:81–91.

Smith P, Ashmore MR, Black HIJ, Burgess PJ, Evans CD, Quine TA, Thomson AM, Hicks K, Orr HG. 2013. Review: The role of ecosystems and their management in regulating climate, and soil, water and air quality. J Appl Ecol 50:812–829.

Smithwick EAH, Turner MG, Mack MC, Chapin FS. 2005. Postfire soil N cycling in northern conifer forests affected by severe, stand-replacing wildfires. Ecosystems 8:163–181.

Smits KM, Kirby E, Massman WJ, Baggett LS. 2016. Experimental and modeling study of forest fire effect on soil thermal conductivity. Pedosphere 26:462–473.

Sokol NW, Sanderman J, Bradford MA. 2019. Pathways of mineral-associated soil organic matter formation: Integrating the role of plant carbon source, chemistry, and point of entry.

Sokol NW, Whalen ED, Jilling A, Kallenbach C, Pett-Ridge J, Georgiou K. 2022. Global distribution, formation and fate of mineral-associated soil organic matter under a changing climate: A trait-based perspective. Funct Ecol 36:1411–1429.

Steel ZL, Safford HD, Viers JH. 2015. The fire frequency-severity relationship and the legacy of fire suppression in California forests. Ecosphere 6.

Stevens-Rumann CS, Kemp KB, Higuera PE, Harvey BJ, Rother MT, Donato DC, Morgan P, Veblen TT. 2018. Evidence for declining forest resilience to wildfires under climate change. Ecol Lett 21:243–252.

Taylor AH, Harris LB, Skinner CN. 2022. Severity patterns of the 2021 Dixie Fire exemplify the need to increase low-severity fire treatments in California's forests. Environ Res Lett 17.

Turner MG. 2010. Disturbance and landscape dynamics in a changing world. Ecology 91.

Turner MG, Braziunas KH, Hansen WD, Harvey BJ. 2019. Short-interval severe fire erodes the resilience of subalpine lodgepole pine forests. Proc Natl Acad Sci U S A 166:11319–11328.

Turner MG, Heumann RE, Kiel NG, Warren JA, Cleveland CC. 2025. Reburning Before Recovery: Effects of Short-Interval Fire on Subalpine Forest Nitrogen Stocks and Fluxes. Ecosystems. https://doi.org/10.1007/s10021-024-0094.

Turner MG, Smithwick EAH, Metzger KL, Tinker DB, Romme WH. 2007. Inorganic nitrogen availability after severe stand-replacing fire in the Greater Yellowstone ecosystem. Proc Natl Acad Sci U S A 104:4782–4789.

Ulery AL, Graham RC, Bowen LH. 1996. Forest Fire Effects on Soil Phyllosilicates in California. Soil Sci Soc Am J 60:309–315.

Urza AK, Sibold JS. 2017. Climate and seed availability initiate alternate post-fire trajectories in a lower subalpine forest. J Veg Sci 28:43–56.

Vitousek PM, Howarth RW. 1991. Nitrogen limitation on land and in the sea: How can it occur?

Wan S, Hui D, Luo Y. 2001. Fire Effects on Nitrogen Pools and Dynamics in Terrestrial Ecosystems: A Meta-Analysis.

Wang Q, Zhong M, Wang S. 2012. A meta-analysis on the response of microbial biomass, dissolved organic matter, respiration, and N mineralization in mineral soil to fire in forest ecosystems. For Ecol Manage 271:91–97.

Whalen ED, Grandy AS, Sokol NW, Keiluweit M, Ernakovich J, Smith RG, Frey SD. 2022. Clarifying the evidence for microbial- and plant-derived soil organic matter, and the path toward a more quantitative understanding. Glob Chang Biol 28:7167–7185.

Yelenik S, Perakis S, Hibbs D. 2013. Regional constraints to biological nitrogen fixation in post-fire forest communities. Ecology 94:739–750.