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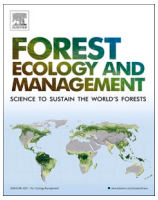

Stream and soil nitrogen response to overlapping forest disturbance: The impact of harvesting, wildfire, and post-fire management on a western Oregon, USA watershed

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ABSTRACT

To understand the magnitude of effects from forest disturbance in watersheds—and their impacts on ecosystems and communities—it is essential to integrate field measurements within aquatic and terrestrial environments. We leveraged 13 years of empirical data across a stream network in the Hinkle Creek Experimental Watershed (1941 ha) located on an intensively managed Douglas-fir (*Pseudotsuga menziesii)* plantation in western Oregon, USA, which was ~98 % burned by wildfire in 2020. We investigated the impact of overlapping disturbance (forest harvesting, wildfire, and post-fire land management) on stream water nitrate (NO_3^--N) and the potential linkage between post-fire riparian mineral soils and in-stream water quality. There was no impact of time since fire on soil NO_3^--N, a decrease in ammonium (NH_4^+-N), an increase in potentially mineralizable N, and no clear contribution of these post-fire soil concentrations to streams. We observed no change in stream water NO_3^--N concentrations after forest harvesting and average concentrations ranged from 0.05 to 0.26 mg L^{-1}. After wildfire and post-fire management, average concentrations ranged from 0.22 to 1.12 mg L^{-1}. Contrary to other studies, there was a negative relationship ($r^2 = 0.84$, $p < 0.001$) between concentrations and high soil burn severity. While we observed higher concentrations in the catchment that experienced more salvage logging, better spatial and temporal information on forest operations from land managers is needed to help disentangle disturbance periods. Additionally, simultaneous *in-situ* measurements of streams and soils could help illuminate biogeochemical connections to identify the source, magnitude, and legacy of disturbance impacts.

1. Introduction

Wildfires are a global phenomenon (Bowman et al., 2009) that play a crucial role in ecosystems by altering the biogeography of landscapes and regulating vegetation structure (Pausas and Keeley, 2009, 2019; Bond et al., 2005). However, increases in fire severity, area burned, and length of fire season in recent decades (Jones et al., 2022; Reilly et al., 2017) have raised concerns about the growing impacts to communities, drinking water (Emelko et al., 2011; Hohner et al., 2019), and the biophysical systems on which they depend (Shuman et al., 2022). Shifts in wildfire regimes are expected to continue due to global climate change, increased fuel loads in forests, and rapid expansion of development at the wildland-urban interface (Abatzoglou and Williams, 2016; Murphy et al., 2018). While the annual area burned and burn severity in recent decades are not necessarily outside the long-term range of fire regimes for some regions (Murphy et al., 2018), there are now greater pressures and demands on our forests and forest resources than ever before and, as such, there is an increasing need to understand the impacts of disturbance (i.e., wildfire) on watersheds.

Wildfires can have a broad range of effects on terrestrial and aquatic systems, including impacts on their physical and chemical properties, biotic communities, and the suite of biogeochemical processes that connect them. Examples of fire-induced changes to soil physical processes include reductions in soil infiltration rates (Ebel and Moody, 2017) and increases in soil erosion due to exposed mineral soil (Wagenbrenner and Robichaud, 2014). Examples of soil chemical changes include loss of total nitrogen (N) due to volatilization at relatively low temperatures (>120 °C) but short-term increases in bio-available N (i.e., NH_4^+-N and NO_3^--N) driven by microbial activity (Wan et al., 2001). Similarly, wildfires generally lead to an immediate

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loss of terrestrial total carbon (C) but a potential deposit of more recalcitrant C in the form of charcoal (DeLuca and Aplet, 2008). Wildfires can also impact hydrologic processes by resulting in increased net precipitation due to the loss of interception storage capacity (Williams et al., 2019). The loss of forest cover and evapotranspiration rates is often also accompanied by changes in soil structure, which may reduce soil infiltration capacity (Collar et al., 2021; Moody et al., 2019), leading to shifts in the timing and magnitude of streamflow and increased risk of flash floods and debris flows (Thomas et al., 2021). These post-disturbance hydrologic shifts can influence the transport of sediment, trace elements, and nutrients from burned hillslopes to streams, creating substantial and lasting impacts on source water quality (Moody and Martin, 2009; Rhoades et al., 2011; Silins et al., 2009; Smith et al., 2011) and aquatic communities (Jager et al., 2021; Silins et al., 2014).

Post-fire changes in stream water quality are highly variable. For example, in an analysis of data from the first five years after wildfires in the western U.S., particulate forms of N increased in ~25–38 % of watersheds while dissolved forms of N decreased in ~38 % of watersheds (Rust et al., 2018). Some dissolved forms of nitrogen, such as nitrate (NO_3^--N), have increased ~5- to 27-times post-fire when compared to unburned watersheds (Bladon et al., 2008; Rhoades et al., 2019; Stephan et al., 2012). Although the greatest rates of stream water NO_3^--N are generally observed in the first couple years after wildfire, increases in some systems have persisted 14 years after the initial wildfire disturbance (Rhoades et al., 2019). Elevated N concentrations in streams, along with increased light availability—due to a loss of the riparian canopy—and subsequent increases in stream temperatures can shift aquatic ecosystem structure due to accelerated growth of aquatic plants and benthic communities (Gustine et al., 2022; Silins et al., 2014).

While many studies have quantified the magnitude of changes to N concentrations after wildfire, we have a poor understanding of the mechanisms influencing different post-fire water quality responses across ecoregions and often poor model representations of post-fire biochemical responses (Stephan et al., 2015). Moreover, we often lack the long-term hydrologic observations necessary to define a departure from baseline variability to discern the true effects of wildfires (Ebel and Mirus, 2014). Similarly, we lack a systematic understanding of recovery processes necessary to predict the magnitude and duration of fire effects on hydrologic and water quality responses (Ebel et al., 2022; Wagenbrenner et al., 2021). Although many hydrological studies point to chemical and physical changes in the soil as the source of change in water quality, the time needed for soil structure redevelopment after wildfire remains largely unknown (Wagenbrenner et al., 2021) and few studies have simultaneously quantified post-fire conditions within terrestrial and aquatic environments, limiting foundational insight into the biogeochemical connections which could support crucial land management decisions (Grimm et al., 2003; Stephan et al., 2012, 2015).

Adding substantial complexity to our understanding of wildfire effects on post-fire nutrient dynamics is the uncertainty created by additional hillslope and watershed disturbance from pre-fire or post-fire forest management activities. These land management activities may include forest harvesting, forest thinning, post-fire salvage logging, and/or post-fire hillslope stabilization treatments, which are often poorly constrained within study designs (Girona-García et al., 2021). Moreover, some research has illustrated that salvage logging operations may have incremental effects over and above wildfire impacts (Emelko et al., 2011; Silins et al., 2014). Thus, the compound interactions of forest disturbance alters rates and trajectories of both forest recovery (Kleinman et al., 2019) and hydrologic recovery (Wagenbrenner et al., 2021). The exclusion of pre- and post-fire land management context from many post-fire studies is a major confounding variable when interpreting research results, highlighting the need for longer-term datasets that are inclusive of multiple disturbance phases.

Our study, located on the west slopes of the Cascade Mountains in Oregon, USA, provided a rare and unique opportunity to investigate and compare the effects of both wildfire and pre-fire forest management activities. Our primary objectives were to:

(a) Quantify the impact of overlapping forest disturbance on stream water quality by conducting a quantitative comparison of stream water NO_3^--N concentrations during different forest and post-fire management phases (pre-harvest, post-harvest, post-fire management);

(b) Assess the impact of pre-fire forest harvesting and time since fire on extractable and mineralizable N concentrations (NO_3^--N, NH_4^+-N, potentially mineralizable N) in riparian mineral soils (0–15 cm);

(c) Evaluate the potential contribution of post-fire soil N concentrations to stream water NO_3^--N by establishing a paired terrestrial-stream monitoring study across a stream network.

2. Methods

2.1. Site description

Our study occurred in the Hinkle Creek watershed, which is a 1940 ha catchment managed for timber production of 60-year-old harvest-regenerated Douglas-fir (*Pseudotsuga menziesii*). The watershed is located in the foothills of the western Cascades, approximately 40 km northeast of Roseburg, Oregon, USA (43° 25' 20" N, 123° 02' 10" W; Fig. 1). Hinkle Creek is a tributary to the Calapooya Creek in the Umpqua River basin.

Climate in the region is Mediterranean with warm, dry summers and cool, wet winters. Rainfall is the dominant form of precipitation in November through May with snowfall occurring intermittently throughout the winter. The 30-year normal (1991–2020) annual precipitation was approximately ~2270 mm (PRISM Climate Group, 2022). Over the duration of the study period (2002–2022), the average annual precipitation was ~1660 mm (Fig. S1) and average annual air temperature was ~11 °C (PRISM Climate Group, 2022).

The dominant tree species across our study area was Douglas-fir with riparian corridors vegetated by western hemlock (*Tsuga heterophylla*), western red cedar (*Thuja plicata*), red alder (*Alnus rubra*), bigleaf maple (*Acer macrophyllum*), vine maple (*Acer circinatum*), and shrub species include salmonberry (*Rubus spectabilis*). Soils are Typic Haplumbrepts, Typic Palehumults, and Typic Hapludands (National Resources Conservation Service). Bedrock geology is primarily basalt and rhyolite flows, with Brown Mountain basalts at the highest elevations, volcanolithic, sandstone, conglomerate, laharic breccia, rhyolite, and dacite flows at intermediate elevations, and Holocene and Pleistocene landslide deposits at lower elevations (Wells et al., 2001). Elevation across the study site ranges from 400 to 1250 m above sea level.

The Hinkle Creek Experimental Watershed (2002–2011) was established as a paired, nested watershed study to quantify the effects of forest harvest practices on streamflow, nutrients, suspended sediment, stream temperature, and fish (Bateman et al., 2016; Kibler et al., 2013; Surfleet and Skaugset, 2013). The study area included an unharvested, reference catchment (North Fork Hinkle; 858 ha) and a catchment that underwent multiple timber harvest treatments (South Fork Hinkle; 1083 ha). In 2001, just prior to the original study there was a small harvest entry (103 ha, 9.5 %) in the South Fork. In 2002, study sites were established to quantify water quantity and quality at the mainstem at the outlet of North Fork Hinkle (NFH) and South Fork Hinkle (SFH) and in several nested catchments, including Meyers (MYE, 86 ha), Fenton (FEN, 23 ha), Russell (RUS, 96 ha) and Beebe (BBE, 111 ha) (Fig. 1).

Streams were monitored from 2002–2005 to establish pre-harvest, baseline relationships across all stream quality sampling sites (Fig. 1). In 2005–2006, five clearcut units in South Fork Hinkle (150 ha or 14 % of the catchment area) were harvested adjacent to small, non-fish bearing streams $\leq$ 2 m (7 feet) in width and medium, non-fish bearing streams 2–3.5 m (7–12 feet) in width. In accordance with the Oregon Forest Practice Rules (Cloughesy and Woodward, 2018), no riparian buffer containing overstory merchantable trees were retained on small, non-fish bearing streams. The watershed was continuously monitored

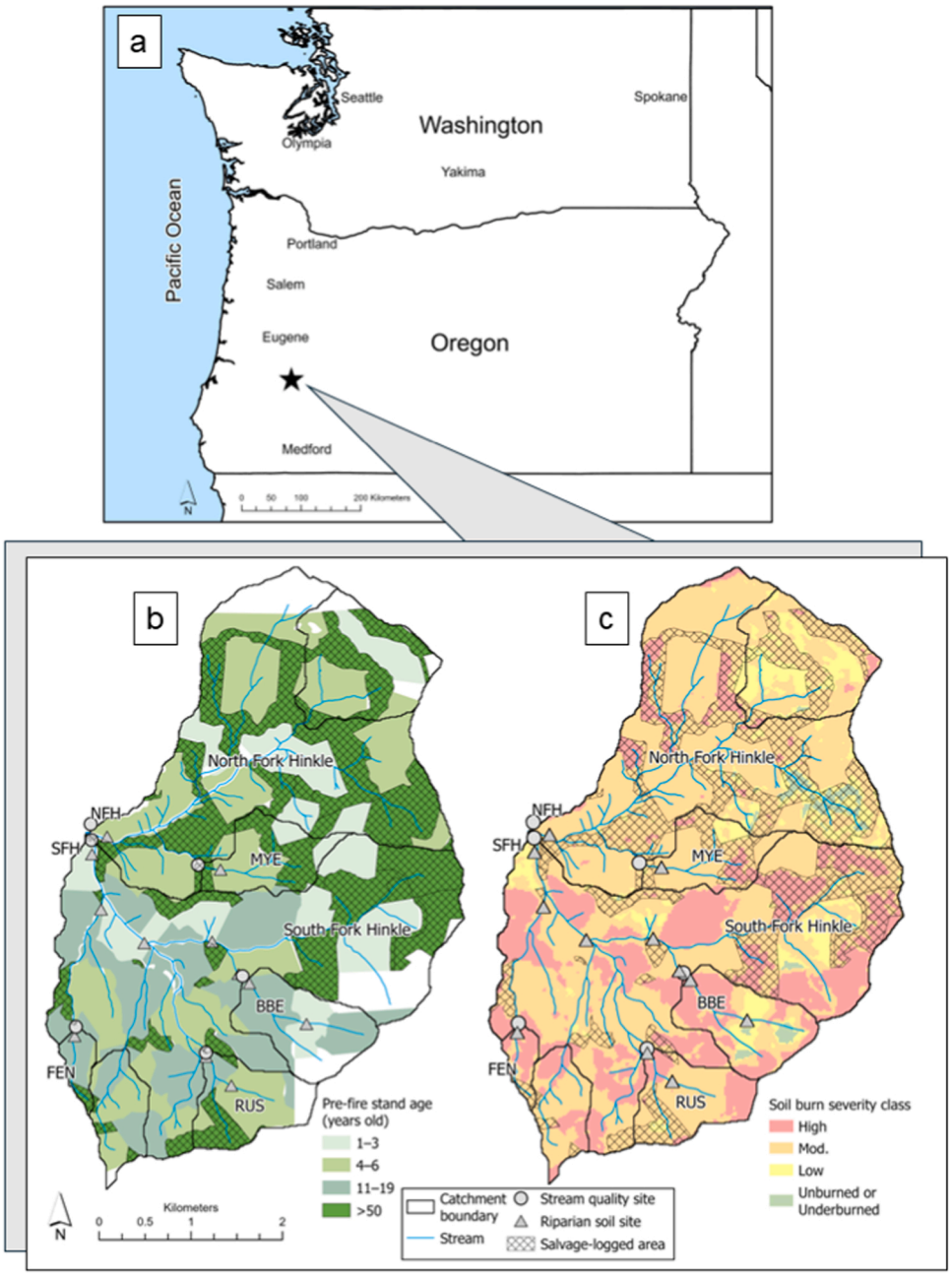

Fig. 1. Maps of (a) the location of the Hinkle Creek Experimental watershed in Oregon, USA, (b) the pre-fire forest harvesting history condensed into four, pre-fire stand age groups (1–3, 4–6, 11–19, and >50 years-old at time of fire), and (c) the soil burn severity, which also shows the areas of post-fire salvage logging, stream water sampling sites, and soil sampling sites.

for harvesting effects until 2008–2009 when an additional four clearcut units in South Fork Hinkle (131 ha, 12 %) were harvested downstream of the preceding harvest entry. A 15-meter (50-foot) fixed-width riparian buffers were retained adjacent to the small, fish-bearing streams and a 20-meter (70-foot) buffer was retained adjacent to medium fish-bearing streams.

All forest harvesting was completed by hand-felling the trees, which were then yarded to landings with a slackline, skyline cable system (Surfleet and Skaugset, 2013). Post-harvest site preparation for revegetation included burning of slash piles in the vicinity of landings and aerial application of herbicides. Harvested areas were then replanted with plug-1 Douglas-fir seedlings at a spacing of 176 trees/ha in the first winter after each harvest entry. Post-harvest monitoring of the watershed continued until 2011, which marked the end of the original Hinkle Creek study. While research activity had ended, forest harvesting activities continued in South Fork Hinkle (385 ha, 36 %) and North Fork Hinkle (446 ha, 51 %) between 2014–2019, which created a mosaic of management history across the watershed.

In September 2020, the Archie Creek fire burned more than 53,000 ha in southwestern Oregon, including ~98 % of the Hinkle Creek watershed on the western edge of the fire (Fig. 1c). Approximately 8.8 % of the watershed burned at low severity, 67.1 % at moderate severity, and 22.5 % at high severity (Table 1). Soil burn severity metrics were calculated from the Burn Area Emergency Response (BAER) program (https://burnseverity.cr.usgs.gov/baer/baer-imagery-support-data-download). In 2020, post-fire management operations began, including salvage logging, herbicide application, and replanting of Douglas-fir seedlings. Post-fire salvage logging operations began in December 2020 and continued until October 2021 and included those stands that had not been harvested during the original study period, so

Table 1
Area (ha), proportion (%) of catchment harvested, and proportion (%) of area burned by soil burn severity categories in the nested sub-catchments in the Hinkle Creek Experimental Watershed. Underlined values represent the proportion harvested during the original study phase (2014–2016), while bold values represent the proportion that was salvage-logged after fire (2020–2021).

Catchment			Years and proportion of harvest (%)					Proportion of burn (%)		
	Site	Area (ha)	2001–2009	2014–2016	2017–2019	2020–2021	Low	Mod.	High	Total
South Fork Hinkle	SFH	1083	35.0	20.3	15.4	**20.5**	5.5	55.6	38.0	99.1
	FEN	23	72.8	10.1	0.0	**15.7**	0.4	65.9	33.6	99.9
	RUS	96	16.4	50.5	0.3	**22.7**	2.4	69.8	27.8	99.9
	BBE	111	58.3	3.6	21.8	**0.1**	15.7	34.1	44.7	94.5
North Fork Hinkle	NFH	858	0.4	36.5	14.7	**38.3**	12.1	78.5	7.0	97.7
	MYE	86	0.0	40.8	16.3	**40.8**	8.0	79.9	12.1	99.9

that the average tree age was approximately > 50 years-old. The salvage logging operations included approximately 21 % of South Fork Hinkle and 38 % of North Fork Hinkle. Forest managers were granted approval to exceed the 48.5 ha (120-acre) clearcut area limit to salvage the burned but merchantable timber from these sites. Trees were again felled by hand with the logs yarded to landings with a slackline, skyline cable system. Regulations of the Oregon Forest Practice Rules required the retention of any remaining live trees along riparian corridors of fish bearing streams within 15 m (50 ft) along small streams, 21 m (70 ft) along medium streams, and 30 m (100 ft) along large streams (defined as greater than 3.5 m (12 feet) in width). Merchantable trees that were killed by the fire were harvested from the riparian areas up to 6 m (20 ft) from the edge of all fish-bearing streams and large, non-fish bearing streams. Following salvage logging, Douglas-fir seedlings were replanted throughout the watershed during the winter (approximately December–March) of 2021, 2022, and 2023. Across the entire watershed, herbicide application (weed control, site prep spray, and brush spray) was applied in the spring, summer, and fall (approximately March–November) in 2021, 2022, and 2023 to reduce competition for the regenerating Douglas-fir seedlings.

To facilitate a quantitative comparison of overlapping disturbance (i.e., timber harvesting, wildfire, and post-fire land management) on stream water nitrate (NO_3^--N) concentrations, we re-reestablished six stream sites from the original study, including four headwater sub-catchments (FEN, RUS, BBE, MYE) and two downstream sites at the outlets of mainstem South Fork (SFH) and North Fork Hinkle (NFH) (Fig. 1). To investigate the impact of pre-fire harvest history on riparian soils, we employed a chronosequence design by condensing all years of forest harvesting into three groups, including: stands that were 11–19 year-old at the time of the wildfire (harvested in 2001–2009), 4–6 year-old stands (harvested 2014–2016), and 1–3 year-old stands (harvested 2017–2019). Approximately 15 months after the Archie Creek Fire, we established 12 post-fire soil study sites across our forest stand age groups and co-located these with our stream water sites (Fig. 1). To investigate time since fire on riparian soils, we sampled 15 months, 25 months, and 32 months after fire.

2.2. Sample collection and laboratory methods

2.2.1. Soil nutrients and physical properties

During the pre-harvest study phase, in 2003, concentrations of total nitrogen (TN) and carbon (TC) were quantified from 27 soil pits throughout the Hinkle Creek watershed (George, 2006). At that time, averages were calculated for eight soil series. However, our post-fire study was designed to try to capture the effects of harvest history and, as such, our summary data was only representative of four soil series. Additionally, our study locations were located in the near-stream riparian zone, which limited our ability to employ a detailed pre- and post-fire soil nutrient comparison. Instead, we present an overall range of pre- and post-fire TN and TC values to give insight into the general soil nutrient pools (Table 2).

The 12 post-fire soil study sites were distributed as four soil sampling sites within each of our three, pre-fire stand age groups. At each site, we collected five subsamples for a total of 60 samples collected each period. We collected samples 15 months post-fire (December 2021), 25 months post-fire (October 2022) and 32 months post-fire (May 2023). Sampling dates were selected based on availability of personnel and accessibility of the field sites. At each site, we collected subsamples approximately one meter from the high-water mark to avoid collection in areas of potential stream inundation. At each site, we randomly selected one side of the stream to begin sample collection and marked with a pin flag and GPS point for return sampling. Mineral soil samples were collected from the top 0–15 cm with a hand trowel and approximately ~300 g was placed into a Whirl-pak bag. The trowel was wiped with a clean towel between each sampling point to ensure no cross-contamination. We collected each subsequent subsample on the opposite side of the stream, approximately three meters upstream from the previous subsample location to ensure samples were spatially independent. This was repeated until five subsamples were collected. At three sites, the harvest history (i.e., pre-fire stand age group) was different on either side of the stream—in these cases, all five subsamples were collected from the same side of the stream to be representative of the same type of treatment. All samples were transported to the lab in a cooler with icepacks and refrigerated (4 °C) until analysis for soil properties (SOM, pH, texture, gravimetric soil water content) and soil chemical characteristics, including nitrate (NO_3^--N), ammonium (NH_4^+-N), mineralizable N, total N, and total C.

Soil organic matter (SOM) content was determined by loss on ignition by placing 1–2 g of oven-dried and sieved (>2 mm) soil into a pre-heated muffle furnace set to 550 °C for 2 hours (Nelson and Sommers, 1996). Soil pH (1:2 soil:water ratio) was determined by measuring 10 g of dried, sieved soil into a plastic beaker, then adding ~20 ml of deionized water. The solution was thoroughly mixed with a glass stir rod then left to settle for 10 minutes. The solution was measured with a pH meter (FiveEasy Plus pH meter FP20, Mettler Toledo, LLC, Columbus, OH, USA) at standard buffer solutions of pH 7 and pH 4. Values of SOM and pH were averaged across the subsamples from each study site for each of the sampling periods (15, 25, and 32-months post-fire).

Composite samples from each site and post-fire sampling period were made by weighing 5 g of dried, sieved (>2 mm) soil from each subsample into a quart-size Ziploc bag, then gently mixed. Percent (%) clay content of each composite sample (Table 2) was determined by first removing organic matter by treating 1 g of each sample with 10–20 ml of sodium hypochlorite (NaOCl), adjusted to a pH of 9.5 with HCl. Then, the sample was heated in a boiling-water bath for 15 minutes and centrifuged at 800 rpm for 5 minutes prior to decanting (Moore, D.M & Reynolds, R.C., Jr., 1989). This procedure was repeated until all organic matter was removed and clay content was then determined by laser diffraction (Bettersizer S3 Plus, Bettersizer Inc. Costa Mesa, CA, USA). Composite samples were also analyzed for total nitrogen (TN) and carbon (TC) (Table 2) in the Soil Health Lab at Oregon State University, Corvallis, OR, USA using the Elementar Vario Macro Cube.

To quantify concentrations of extractable and mineralizable soil N, a frozen soil sample was removed from the freezer, coarse fragments and

Table 2
Riparian mineral soil (0–15 cm) properties of each pre-fire stand age group (1–3, 4–6, 11–19 years-old) during three post-fire sampling phases (15, 25, 32 months post-fire).

Pre-fire stand age (years)	Time after fire (months)	pH	SOM (%)	Total C (g kg^{-1})	Total N (g kg^{-1})	Clay (%)
1–3	15	5.6	19.5	47.7	3.4	15
	25	5.6	13.1	40.8	2.4	
	32	5.5	14.3	44.2	3.0	
4–6	15	5.9	18.6	43.2	2.7	21
	25	5.4	14.2	45.5	2.7	
	32	5.2	13.3	33.1	2.0	
11–19	15	5.8	17.7	44.5	2.8	21
	25	5.2	14.4	46.6	2.7	
	32	5.5	13	38.4	2.3	

plant matter were removed, and two subsamples of 10 g were measured into two, 50 ml polypropylene centrifuge tubes. The first tube was immediately extracted with 30 ml of 2 M KCl—this sample was then shaken (60 min, ~100 rev min^{-1}) and filtered through a Whatman #42 paper. Soil nitrate (NO_3^--N) concentration was determined on filtrate of the first tube by pipetting 50 µl of solution into a cuvette and pipetting 2 ml of solution made by mixing 400 ml DI water, 50 ml of 1 M HCl, 400 mg Vanadium (III) chloride (VCL_3), 200 mg sulfanilamide, and 10 mg NEDD (N-(1-naphthyl) ethylenediamine dihydrochloride). Cuvettes were covered and left at room temperature for 12 hours, then read on a spectrophotometer at 540 nm. Soil ammonium (NH_4^+-N) concentration was determined on filtrate of the first tube by using a Lachat colorimetric autoanalyzer (Hach Company, Loveland, CO, USA).

The second polypropylene tube, containing 10 g of soil only, was placed in a glove box that was degassed with N_2 by pulling a vacuum, inflating with N_2, and the degassing process was repeated three times. Then, 30 ml of degassed deionized water was added to the sample tube and left in the glove box to degas for 12–24 hours. Then, the tube's headspace was flushed with N_2, capped, and incubated at 25 °C for 14 days. After incubation of the second tube, contents were transferred into a 100 ml low-density polyethylene Nalgene bottle and extracted with 30 ml of 4 M KCl, shaken (60 min, ~100 rev min^{-1}) and filtered through a Whatman #42 paper. Then soil filtrate was analyzed for NH_4^+-N using a Lachat colorimetric autoanalyzer (Hach Company, Loveland, CO, USA). We then quantified 14-d potentially mineralizable N (PMN; Hart et al., 1994), by calculating the net mineralization as NH_4^+-N at day 14 minus NH_4^+-N at time zero (Choromanska and DeLuca, 2002). To determine the dry weight value of soil nitrogen concentrations, gravimetric soil water content was determined placing approximately 10 g of moist soil (M_{wet}) from each sub-sample into a pre-weighed tin boat. The soil samples were then dried in an oven at 110 °C for 24 hours to remove all moisture. After drying, the samples were weighed again to determine the dry mass (M_{dry}) and gravimetric water content was calculated using the following formula: $(M_{wet} - M_{dry})/M_{dry} \times 100$ to express as a percentage.

2.2.2. Stream water nitrate

In the original Hinkle Creek Experimental Watershed Study, water samples were collected approximately each month from 2002–2011 from the downstream end of Montana flume throats in 1,000-ml acid-washed Nalgene bottles. Sampling sites included four headwater sub-catchments (FEN, RUS, BBE, MYE) and two downstream locations at the outlets of mainstem South Fork (SFH) and North Fork Hinkle (NFH). During this study period, 112 samples were collected in the pre-harvest period and 196 samples were collected during the post-harvest period for a total of 308 samples collected in the pre-fire period. Water samples were refrigerated following collection and submitted within 24 hours to the Cooperative Chemical Analytical Laboratory (CCAL) at Oregon State University to quantify a suite of water quality analytes, including nitrate (NO_3^--N).

Approximately 14 months after the Archie Creek wildfire, we re-established collection of water samples from the original study sites. Depth-integrated water samples were collected into 500 ml high-density polyethylene bottles, which were acid-washed, triple rinsed with deionized water, air-dried, and triple-rinsed again in stream water at the site prior to sample collection. Stream water samples were collected on an approximately weekly basis during the wet season (October–May) and monthly basis during the dry season (June–September) for a total of 261 samples in the post-fire management period, collected ~14–32 months post-fire. Samples were stored in a cooler on ice while transported to the lab, then in the refrigerator (4 °C) before filtering. Samples were filtered within 24 hours of collection by using a plastic receiving flask, vacuum pump, and Whatman Grade GF/F (0.7 µm) filters. The filtered water was stored in a freezer at −5 °C until analysis. Samples were submitted to CCAL at Oregon State University and analyzed for NO_3^--N using the cadmium reduction method (APHA 4500-NO3 I; EPA 353.2.) and a LachatQuikChem 8500 Analyzer (Hach Company, Loveland, CO, USA).

2.3. Statistical analysis

In our analysis of soil N, we considered the mean from each sampling site to be a replicate of the pre-fire stand age group (1–3, 4–6, and 11–19 years-old) and tested for a treatment effect of pre-fire stand age and time since fire (at 15, 25, and 32 months). We used the *lme4* package in R (Bates et al., 2015) to build a linear mixed effect model with the sampling site as a random effect and an interactive effect between the pre-fire stand age and time since fire. Residuals from the fitted models were checked graphically (Pearson residual plot). We noted that the assumptions of constant variance and normality were met reasonably for values of NO_3^--N and 14-d PMN and data from our model are presented as marginal means. Data not conforming to these assumptions were transformed using a log transformation (NH_4^+-N) and back-transformed to be presented as median values and ratios. The variance term for the NO_3^--N data was estimated at zero. We then performed a global mean comparison using the *emmeans* package (Lenth et al., 2024) for pairwise comparisons between pre-fire stand age groups across time, within pre-fire stand age groups across time, and of average soil concentrations across time. We applied a Tukey adjustment to all pairwise comparisons to control for the family-wise error rate (Tukey, 1949).

We calculated mean stream water NO_3^--N concentrations for each study site during the pre-harvest (2002–2005), post-harvest (2006–2011), and post-fire management phase (2021–2023). Concentrations of stream water NO_3^--N in North Fork Hinkle were then calculated as an average of MYE and NFH. Concentrations of stream water NO_3^--N in South Fork Hinkle were calculated as an average of FEN, RUS, BBE and SFH. We used the *lme4* package in R (Bates et al., 2015) to build a linear mixed effect model, with study site as a random effect, an interactive effect between study phase and treatment designation from the original harvest study (i.e., "reference" or "treatment"), and proportion (%) of catchment burned at high soil burn severity within a 30-meter stream buffer as a fixed effect. We chose to consider burn severity nearest to the stream because conditions in the riparian environment are distinct from the hillslope/upland environment and act as a transitional area between terrestrial and aquatic ecosystems (Luke et al., 2007). Additionally, wildfire behavior can differ significantly between the riparian and hillslope/upland environments (Halofsky and Hibbs, 2008). As such, we wanted to only consider the conditions that would support our objective of investigating the biogeochemical connection between the near-stream terrestrial and aquatic zones. Residuals from the fitted models were checked graphically (Pearson residual plot). From our model, we noted that the assumptions of constant variance and normality were met reasonably. We then used a series of mean comparisons using the *emmeans* package (Lenth et al., 2024) and applied a Tukey adjustment to control for the family-wise error rate (Tukey, 1949) and estimates from the model are presented as marginal means.

The relationships between average stream water NO_3^--N

concentrations and extent of catchment burned upstream of the sampling location, within a 30-meter stream buffer, were further evaluated using least-squares linear regression by plotting the marginal effect of fire with partial residuals.

3. Results

3.1. Extractable and mineralizable post-fire N concentrations

3.1.1. Differences between pre-fire stand age groups across time

There were no differences in marginal mean NO_3^--N concentrations between any stand age groups and the only difference in NH_4^+-N and 14-d PMN concentrations occurred in our first post-fire sampling period, approximately 15 months after fire (Fig. 2). Specifically, marginal median NH_4^+-N concentrations were greater in the 1–3 year-old stands than in the 4–6 year-old stands ($t_{(25.2)} = 2.08$, $p \leq .05$) and marginal mean 14-d PMN concentrations were lower in the 1–3 year-old stands than in the 11–19 year-old stands ($t_{(25.2)} = -2.57$, $p \leq .05$).

3.1.2. Differences within pre-fire stand age groups across time

There were no differences in marginal mean NO_3^--N concentrations during any post-fire sampling period within any of our pre-fire stand age groups (Fig. 2). Marginal median concentrations of NH_4^+-N decreased over time in all stand age groups. Within the 1–3 year-old stands, soil NH_4^+-N concentrations decreased by 18.23 mg kg^{-1} from the 15-month to 25-month period ($t_{(18)} = 5.04$, $p < .0001$), with a further reduction by 4.96 mg kg^{-1} in the 32-month period ($t_{(18)} = 4.91$, $p < .001$). Within the 4–6 year-old stands, soil NH_4^+-N concentrations decreased by 5.42 mg kg^{-1} from the 15-month to 25-month period ($t_{(18)} = 4.32$, $p < .001$) with a further reduction by 4.43 mg kg^{-1} in the 32-month period ($t_{(18)} = 4.26$, $p < .001$). Finally, within the 11 – 19 year-old stands, soil NH_4^+-N concentrations decreased by 10.47 mg kg^{-1} from the 15-month to 25-month period ($t_{(18)} = 8.24$, $p < .0001$), with a further reduction of 3.32 mg kg^{-1} in the 32-month period ($t_{(18)} = 3.95$, $p = .003$).

Contrastingly, marginal mean 14-d PMN concentrations increased over time in the youngest and oldest stands when comparing the first and last sampling periods of the study (Figure 2). In the 1–3 year-old stands, 14-d PMN concentrations increased by 19.52 mg kg^{-1} from the 15-month to 32-month period ($t_{(18)} = 4.0$, $p = .002$) and increased by 13.33 mg kg^{-1} in the 11–19 year-old stands from the 15-month to 32-month period ($t_{(18)} = 2.73$, $p = .03$).

3.1.3. Time since fire

When averaging across all mineral soil sampling sites, we observed a decrease in median NH_4^+-N concentrations and a slight increase in mean 14-d PMN concentrations between our post-fire sampling periods (Fig. 3). There were no differences in marginal mean NO_3^--N concentrations between the sampling periods. Overall, marginal median NH_4^+-N concentrations declined from 16.84 ± 1.95 mg kg^{-1} (95 % confidence interval (CI): 13.27, 21.38) in the 15-month sampling period to 6.23 ± 0.72 mg kg^{-1} (CI: 4.91, 7.91) in the 25-month time period by a ratio of 2.70 ± 0.40 mg kg^{-1} ($t_{(18)} = 6.73$, $p < .0001$). There was a further reduction in NH_4^+-N concentrations to 2.04 ± 0.24 mg kg^{-1} (CI: 1.61, 2.59) by a ratio of 3.06 ± 0.45 in the 32-month period ($t_{(18)} = 7.58$, $p < .0001$). Mean 14-d PMN increased from 8.66 ± 2.99 (CI: 2.35, 15.00) in the 25-month sampling period to 17.57 ± 2.99 mg kg^{-1} (CI: 11.26, 23.90) in the 32-month sampling period by an estimate of 8.91 ± 2.82 mg kg^{-1} ($t_{(18)} = 3.17$, $p = .014$).

3.2. Stream water nitrate

3.2.1. Comparisons of North and South Fork Hinkle catchment averages

When averaging concentrations across sampling sites in the North

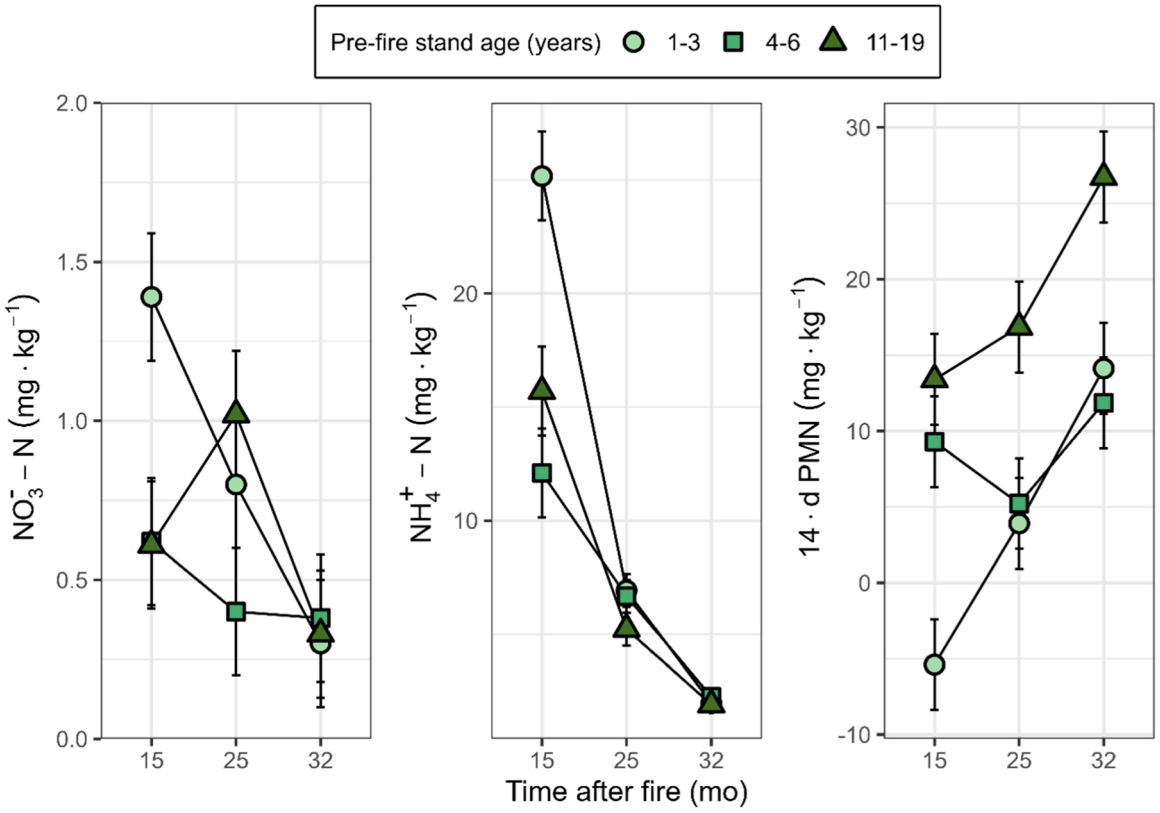

Fig. 2. Riparian mineral soil (0–15 cm) concentrations (mg kg^{-1}) of estimated marginal mean NO_3^--N, estimated marginal median NH_4^+-N, and estimated marginal mean 14-d PMN across each pre-fire stand age group (1–3, 4–6, and 11–19 years-old) across the post-fire sampling periods (15 months, 25 months, 32 months). Error bars denote one standard error from the mean. Note the different y-axis scales.

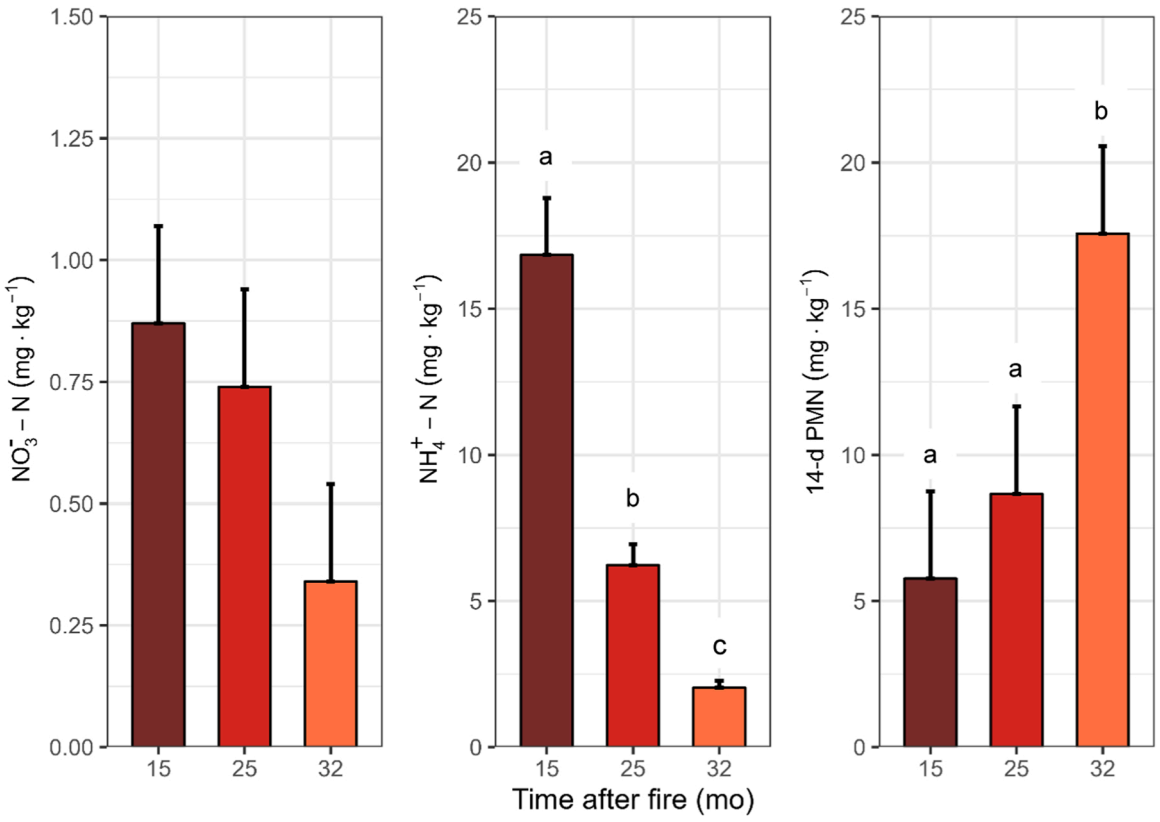

Fig. 3. Riparian mineral soil (0–15 cm) concentrations (mg kg^{-1}) of estimated marginal mean NO_3^--N, estimated marginal median NH_4^+-N, and estimated marginal mean 14-d PMN determined 15 months ($n = 58$), 25 months ($n = 55$), and 32 months ($n = 57$) post-fire. Error bars denote one standard error from the mean and letters above error bars denote statistical differences detected by the linear mixed effect model. Note the different y-axis scales.

Fork Hinkle (NFH, MYE) and South Fork Hinkle (SFH, FEN, RUS, BBE), we observed little variability in stream water NO_3^--N between the pre-harvest and post-harvest study periods (Fig. 4). Statistically, there was no evidence that mean NO_3^--N concentrations differed between the pre- and post-harvest study periods in the North Fork ($t_{(7)} = 0.30$, $p = .95$) or South Fork ($t_{(7)} = -0.09$, $p < .99$). Similarly, there was no difference between the North Fork (reference) and South Fork (harvested) catchments during the pre-harvest study period ($t_{(6)} = -0.80$, $p = .46$) or during the post-harvest study period ($t_{(6)} = -1.03$, $p = .35$). After wildfire and post-fire management, the estimated marginal mean of stream water NO_3^--N concentration in South Fork Hinkle was 0.49 mg L^{-1} (95 % CI: 0.33, 0.66); three-times greater when compared to the pre-harvest period ($t_{(8)} = -3.68$, $p = .015$) and post-harvest period ($t_{(8)} = -3.62$, $p = .017$). Comparatively, in the North Fork, NO_3^--N concentrations were 0.98 mg L^{-1} (CI: 0.74, 1.21), approximately 20-times greater than the pre-harvest period ($t_{(8)} = -8.22$, $p < .0001$) and 34-times greater than the post-harvest period ($t_{(8)} = -8.42$, $p < .0001$).

3.2.2. Variation among individual sampling sites

We observed substantial variation in stream NO_3^--N concentrations across individual sampling sites during the various study phases (Fig. 5). For example, at the FEN sampling site, concentrations remained low throughout all study phases; mean NO_3^--N concentrations were 0.02 ± 0.01 mg L^{-1} in the pre-harvest period, 0.20 ± 0.13 mg L^{-1} in the post-harvest period, and 0.22 ± 0.13 mg L^{-1} in the post-fire management period. Comparatively, at the BBE sampling site, mean NO_3^--N concentrations were 0.45 ± 0.27 mg L^{-1} in the pre-harvest period, 0.26 ± 0.15 mg L^{-1} in the post-harvest period, and 0.60 ± 0.26 mg L^{-1} in the post-fire management period. At the NFH sampling site, mean NO_3^--N concentrations were 0.05 ± 0.11 mg L^{-1} in the pre-harvest period, 0.02 ± 0.02 mg L^{-1} in the post-harvest period, and 1.12 ± 0.50 mg L^{-1} in the post-fire management period.

3.2.3. Negative correlation between high burn severity and stream water nitrate

We observed a negative correlation between the proportion of catchment burned at high soil burn severity and mean stream water NO_3^--N concentrations (Fig. 6). Specifically, for every 1 % increase in the proportion of a catchment burned at high soil burn severity within a 30-meter buffer upstream of our sampling sites, there was a 0.03 mg L^{-1} decrease in the mean concentration of stream water NO_3^--N. Statistically, there was strong evidence ($t_{(7.9)} = -3.24$, $p < .001$) supporting this relationship from the partial slope for burn severity from our linear mixed effect model when holding all other predictors fixed.

4. Discussion

4.1. Soil nitrogen dynamics

In our study—15 months after the Archie Creek Fire in Oregon, USA—we observed greater concentrations of NH_4^+-N in the soil of younger forest stands (1–3 years-old) when compared to the soil of the sapling stage stands (4–6 years-old) and the pre-mature stands (11–19 years-old). In part, this may have been influenced by lower nitrogen (N) uptake by seedlings in the younger stands prior to the fire, which is common immediately after forest harvesting (Gholz et al., 1984), with early successional forests taking up an increasing amount of nutrients and elements from mineral soils (Hume et al., 2018). It is also possible that the larger concentrations of 14-d PMN we observed in the older stands was due to greater organic matter accumulation prior to the fire event, leading to greater charcoal accumulation during the fire event. The deposition and incorporation of charcoal into mineral soils can impart a number of beneficial soil physical and chemical properties (DeLuca and Aplet, 2008), increasing microbial activity (Carter et al., 2018) and net nitrification in post-fire soils (DeLuca and Sala, 2006). We

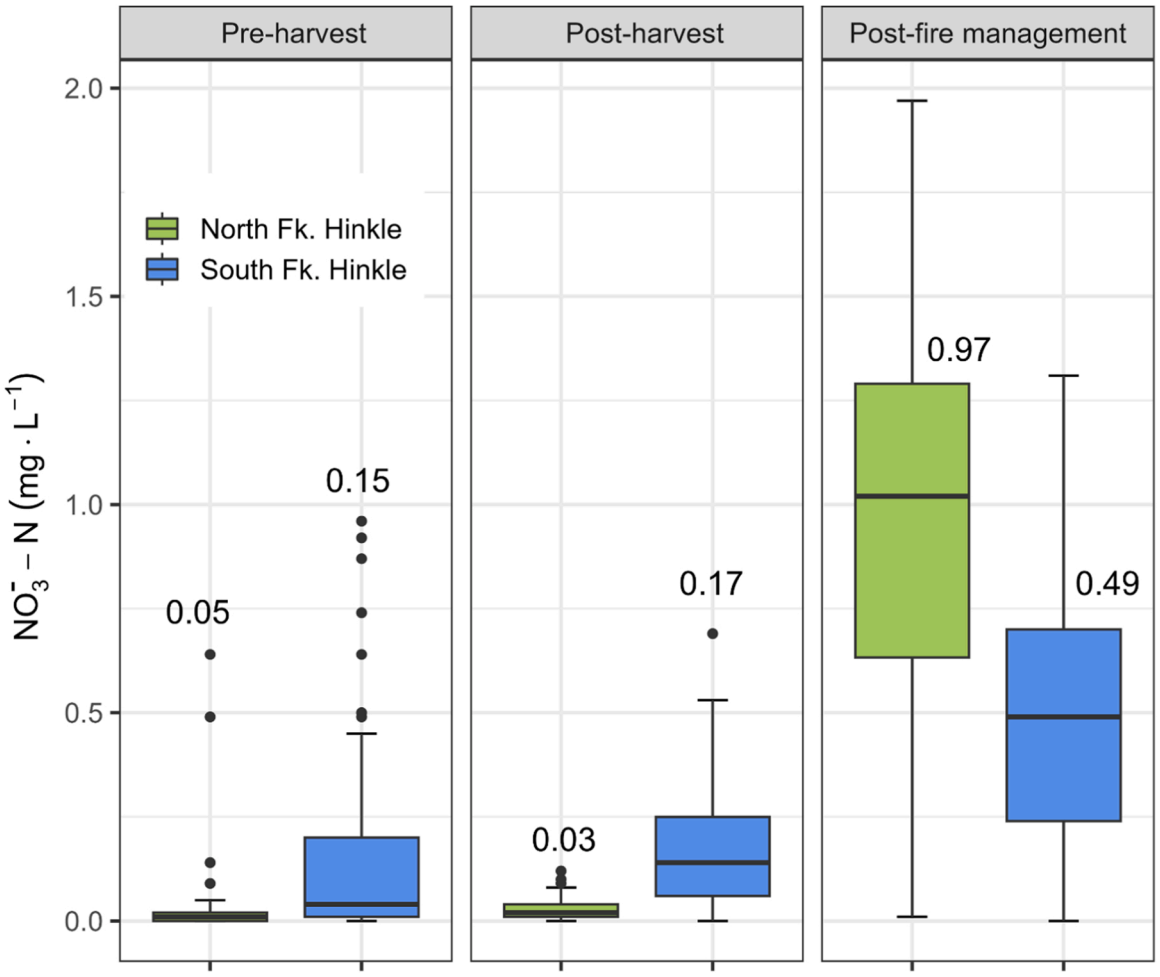

Fig. 4. Box-and-whisker plots of mean stream water NO_3^--N (mg L^{-1}) concentrations between North and South Fork Hinkle catchment during the pre-harvest phase (2002–2005), post-harvest phase (2006–2011), and during the post-fire management phase (2021–2023). During the original harvest study, North Fork Hinkle was held as a reference watershed and South Fork as the treatment watershed. During the 2020 Archie Creek wildfire, ~98 % of both catchments were burned. Average concentrations are listed as values above each figure. From the Hinkle Creek experimental watershed in western Oregon.

did not detect any differences in pH, SOM, total C, total N, or clay content among our pre-fire stand age groups, suggesting that commonly measured physical soil properties would not serve as suitable explanations or proxies of N dynamics in our sites. Measuring alternative soil properties that microbes use in the process of N mineralization, such as specific attributes of SOM (e.g., microbially available SOM, particulate organic matter), could improve predictions of soil N availability (Osterholz et al., 2017). Further, fluxes of nutrients using *in situ* methods such as buried ion exchange resins of buried cores can provide more accurate assessments of nutrient turnover following disturbance events (DeLuca and Sala, 2006; Schimel and Bennett, 2004). Alternatively, measuring potential C mineralization, which has a strong relationship with N indictors, could serve as a more feasible proxy for land managers (Liptzin et al., 2023), though further research is needed to evaluate these relationships within forest soils.

We observed an overall decrease in NH_4^+-N with time since fire and, while we lack pre-fire observations from our soil study sites necessary to make a comparison to any pre-fire conditions, an initial increase in soil NH_4^+-N followed by a decline in NH_4^+-N with time since fire is supported broadly in the literature (Wan et al., 2001). Wildfire generally creates an ephemeral increase of extractable soil NH_4^+-N and subsequent pulse of extractable NO_3^--N, which return to pre-fire concentrations within the first one to two years after fire (Choromanska and DeLuca, 2002; Grogan et al., 2000; Stephan et al., 2012; Wan et al., 2001). The post-fire increases of extractable N have been attributed to volatilization and subsequent condensation of N during soil heating (Knoepp and Swank, 1993), the presence of ash (Grogan et al., 2000), charcoal deposition (DeLuca et al., 2006), and oxidation of dead plant and microbial biomass (Díaz-Raviña et al., 1996). Coupled with reduced uptake from vegetation and limited transformation from microbes killed during fire, the pulse of NH_4^+-N often remains available in the soil until it is depleted by recovering vegetation, immobilized, and/or nitrified to NO_3^--N by recovered microbial communities.

Soil NO_3^--N also increases after fire, but only after NH_4^+-N has increased (Kaye and Hart, 1998; Neary et al., 1999; Wan et al., 2001). Accordingly, these increases in soil NO_3^--N are also depleted over time, but due to different pathways. Where NH_4^+-N adsorbs onto negatively charged colloidal surfaces, the loss of NO_3^--N is predominately attributed to leaching to deeper soil layers or transported off of hillslopes to adjacent streams (Stephan et al., 2012) and both forms are taken up by plants and microorganisms. While some studies have reported a temporal lag in increases of NO_3^--N after wildfire due to nitrification of increased NH_4^+-N (Wan et al., 2001), there are also observations of a weak-to-no effect from wildfire (Knoepp and Swank, 1993; J. Murphy et al., 2006). Interestingly, Choromanska and DeLuca (2002) observed lower NO_3^--N concentrations in mineral soils not previously exposed to fire (<80 years) and burned in a laboratory setting when compared to soils recently exposed to high severity wildfire (≤1 year) and then burned in a laboratory setting. This difference in NO_3^--N concentrations was possibly due to the low presence of nitrifiers from conditions

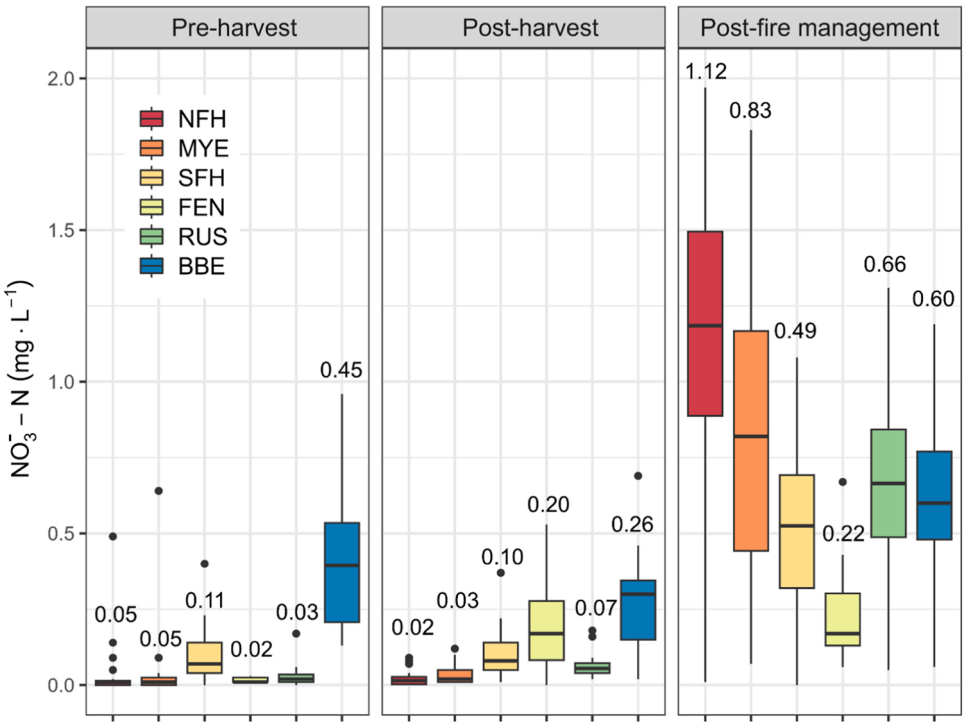

Fig. 5. Boxplot of stream water NO_3-N concentrations of each sampling site during the pre-harvest (2002–2005), post-harvest (2006–2011), and post-fire management phase (2021–2023). Average site concentrations are listed above each figure. From the Hinkle Creek experimental watershed in western Oregon.

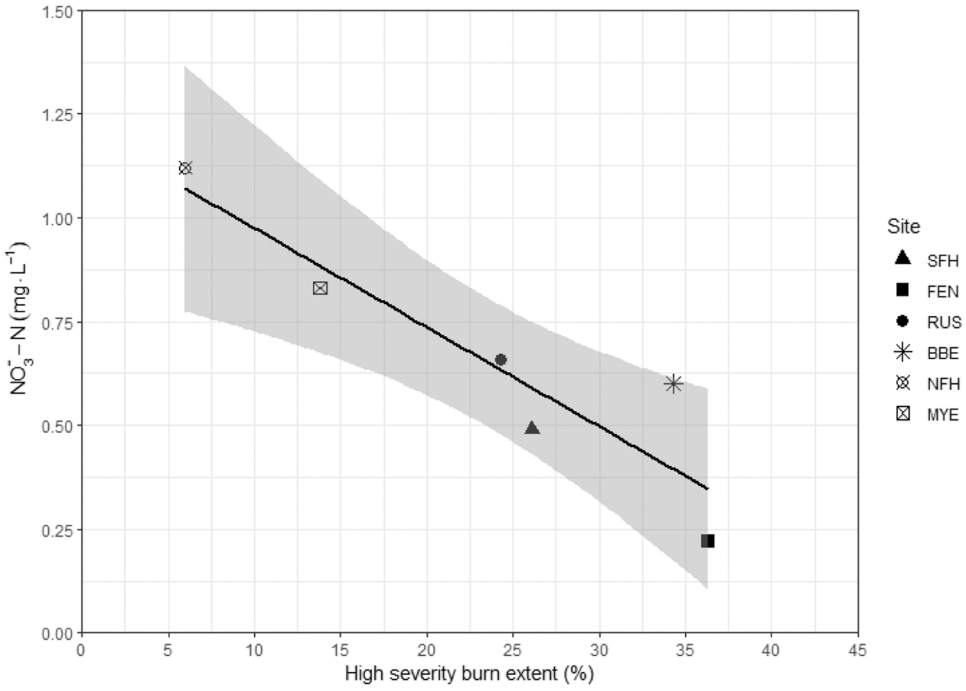

Fig. 6. Linear relationship between mean stream water NO_3-N concentrations at each sampling site through the three phases of our study (pre-harvest, post-harvest, and post-fire management) and the proportion (%) of a catchment burned at high severity within a 30 m stream buffer. Shaded areas represent 95 % confidence intervals. Statistical analysis was conducted using the marginal effect of fire with partial residuals from the full linear mixed effect model (β=-0.29, r^2 = 0.84, $p < 0.001$).

brought about by fire exclusion (Neary et al., 1999). Our study sites had extensive fire suppression efforts beginning in the mid-1800s limiting exposure to fire in the recent past, which could help to explain the overall low concentrations of soil NO_3-N we observed and lack of detectable difference among our treatments.

In contrast to many other studies that reported a decrease in mineralizable N with time since fire (Choromanska and DeLuca, 2002; DeLuca et al., 2002; DeLuca and Zouhar, 2000), we observed a 3-fold increase from 15 to 32 months post-fire. This trend is consistent with a prescribed fire study by Gundale et al. (2005), which demonstrated

higher rates of N mineralization after three years in burned study plots. Furthermore, there was a positive linear relationship with fire severity and N mineralization (Gundale et al., 2005), which would be consistent with our study as we qualitatively observed that most of our sampling sites experienced a moderate-high severity burn (Fig. 1).

In the immediate three months before the Archie Creek fire, from June–July 2020, there was a total of 148 mm of rain (Fig. S1). Thus, we assume that soil water conditions in the soil were relatively high and homogenous at the time of fire in early September. In the laboratory setting, greater soil moisture during heating resulted in lower mineralizable N due to the faster and deeper heat penetration in moist soils, which can kill even heat-tolerant microbes (Choromanska and DeLuca, 2002). However, it is unlikely that the temperatures within our mineral soils (0–15 cm) penetrated deep enough to sterilize the microbial communities at our sites, as heat transfer down to depths below a few centimeters is unlikely to occur in natural soils due to the insulating effect of porous soil medium and latent heat of vaporization associated with soil moisture (Choromanska and DeLuca, 2002). Moreover, is it possible that the moist conditions of our soils within the riparian zones resulted in higher heat capacities with lower maximum temperatures and lower charring intensities during the fire event when compared to drier soils located in the upland/hillslope environment (Badía et al., 2017). The consistent moisture conditions in our soils may have provided ideal conditions for soil microorganisms, lessening physiological stress and leading to more N mineralization (Tiemann and Billings, 2011). Further post-fire comparisons of the riparian and upland/hillslope zones are needed to investigate the impact of fire effects on microbial communities, differences in N mineralization rates, and subsequent forest recovery in these zones over time.

We theorize that the divergence from the typical post-fire temporal pattern observed in other soil N studies is, in part, confounded by our experimental study design, the relatively short duration of our post-fire study, and the lack of an unburned control. Furthermore, many of the previous studies noted here were initiated after burning in the laboratory setting or after prescribed burning, which is typically designed to burn at lower intensities. In contrast, our study was situated after an infrequent, very large high-severity fire, unprecedented in the contemporary data record for our region (Abatzoglou et al., 2021). The placement of our sampling sites near streams limits comparisons to other studies, as most previous research has been situated in the hillslope/upland environment with very different soil conditions than ours. To our knowledge, there are no publications at this time that have investigated time-since-fire on extractable and mineralizable N concentrations within riparian zones.

4.2. Stream water nitrate before and after forest harvesting

In the initial phase of our study (2002–2011), we did not observe shifts in average stream water NO_3^--N concentrations that could be attributed to forest harvesting. Additionally, during both the pre-harvest and post-harvest study periods, stream water NO_3^--N concentrations remained relatively low (Fig. 4), averaging 0.15–0.17 mg L^{-1}in South Fork and 0.03–0.05 mg L^{-1} in North Fork Hinkle Creek. This is typical for our region, where average NO_3^- concentrations in forested streams of the western United States are 0.20 mg L^{-1} and much lower when compared to the Northeast region of the United States, with a mean of 0.50 mg L^{-1} (Binkley et al., 2004). However, concentrations can vary greatly between streams depending on factors such as forest composition, geology, stream order, basin size, atmospheric deposition, and seasonal patterns (Binkley and Brown, 1993; Feller, 2005; Smith et al., 2003).

The lack of response in stream water NO_3^--N after forest harvesting was somewhat surprising as the majority of studies have documented significant increases. For example, stream water nitrate concentrations increased 3–100-times in the first year and half after clearcutting an old-growth coniferous forest watershed in the Oregon Cascade Mountains (Sollins and McCorison, 1981). Similarly, in the Bull Run Watershed in Oregon, NO_3^--N concentrations increased more than six-fold after road construction, clearcutting, and broadcast burning of timber residue (Harr and Fredriksen, 1988). In the Needle Branch watershed of the Oregon Coast Range, maximum nitrate nitrogen concentrations increased approximately three-fold after clearcutting and broadcast burning (Brown et al., 1973). However, the previous studies mentioned here were conducted on managed sites in old-growth forests (approximately 365–450 years old) that employed older forest harvesting practices lacking riparian buffers—best management practices (including riparian buffers) have since been shown to effectively reduce nitrate transport from harvested hillslopes into streams (Löfgren et al., 2009; Richardson and Béraud, 2014). Within study sites of similar forest composition and stand ages to ours, clearcut harvesting with a fixed riparian buffer resulted in no changes in stream NO_3^--N or nitrogen levels when compared to sites that were clearcut without buffer (Johnson et al., 2023; Stednick, 2008). However, understanding the ability of riparian vegetation to reduce nitrate transport to streams after disturbance may be complicated by the presence of N-fixing alder in riparian areas, which can significantly increase nitrate availability (Compton et al., 2003; Wigington et al., 1998). Indeed, N_2-fixing red alder (*Alnus rubra*) was a dominate, but spatially variable riparian species at our sites, which may have contributed to the substantial variability across our study sites (Fig. 5).

4.3. Stream water nitrate after wildfire and post-fire management

During the first few years following the Archie Creek Fire and post-fire management operations, average stream water NO_3^--N concentrations were 0.49 mg L^{-1} in South Fork Hinkle and 0.97 mg L^{-1} in North Fork Hinkle, with maximum concentrations of approximately 1.25 mg L^{-1} in South Fork and 2.0 mg L^{-1} in North Fork (Fig. 4). Our results align with several studies that have shown large exports of nitrate after fire. For example, in the first five years after the Hayman Fire in Colorado, Rhoades et al. (2011) observed mean NO_3^--N concentrations of 0.54 mg L^{-1} and seasonal peaks of 1.5–2.3 mg L^{-1} in catchments burned at high-severity extent (i.e., >45 % of catchment proportion burned at high-severity). In the eastern slopes of the Canadian Rocky Mountains, Bladon et al. (2008) observed mean stream water NO_3^- concentrations of 0.49 mg L^{-1} in burned sites compared to 0.08 mg L^{-1} in unburned sites in the first year after wildfire. After wildfire in central Idaho, USA, Stephan et al. (2012) measured mean NO_3^--N concentrations of 0.34 mg L^{-1} at burned sites compared to 0.04 mg L^{-1} at unburned sites, and concentrations in the burned sites remained elevated three years later. In part, the variation in reported concentrations and the duration of impact could be attributed to differences in wildfire behavior, regional precipitation patterns, and vegetation regrowth. However, our understanding is greatly limited by the relatively short duration (<3 years) of most post-fire hydrologic studies (Wagenbrenner et al., 2021). Thus, we need longer-term observations to follow post-fire patterns and trends, which could provide critical knowledge to further assess and help predict post-fire recovery.

Previous research has indicated a positive, linear relationship between stream NO_3^--N concentrations and catchment area burned and between NO_3^--N and increasing burn severity (Rhoades et al., 2011, 2019; Riggan et al., 1994; Stephan et al., 2012). Given that the Archie Creek Fire burned nearly the entire Hinkle Creek watershed (~98 %), we lack the variability necessary to conduct further comparisons of catchment area burned. However, we had enough variability in burn severity that we observed a negative relationship between stream NO_3^--N concentrations and area burned at high severity. Surprisingly, as the proportion of the watershed burned at high severity increased, NO_3^--N concentrations in the stream decreased (Fig. 6). We posit that the greater concentrations of stream water NO_3^--N in North Fork Hinkle Creek—despite burning at slightly lower severity (~86 % moderate-high) compared to the South Fork catchment (~94 % moderate-high)—may have been due, in part, to the greater proportion of the

drainage that underwent post-fire management. Approximately 38 % of North Fork was salvaged logged after the fire, compared to 21 % of the South Fork drainage. Thus, the inverse relationship we observed between high severity burning and stream water NO_3^--N concentrations could be due to the influence of post-fire management operations. While there are likely some differences in biogeochemistry between our catchments that could be driving differences in NO_3^--N response, soil compaction from post-fire logging operations may have contributed to increases in runoff due to reductions in soil porosity and infiltration capacities (Prats et al., 2019, 2021), which would influence the water quality response. We theorize that greater runoff and slower vegetation recovery, due to the intensity of post-fire salvage logging, could be creating greater delivery of NO_3^--N to streams in North Fork Hinkle.

Within a balanced system, the loss of ecosystem N through biological uptake by terrestrial and aquatic plants is balanced by the inputs of N (atmospheric inputs and fixation/mineralization) so that watershed-scale export of various forms of nitrogen are an integrated signature of landscape biogeochemical processes (Cairns and Lajtha, 2005; Goodale et al., 2000; Vitousek and Reiners, 1975), As such, post-fire land management activity has the potential to influence N availability and transport to streams. However, our understanding of the impacts of post-fire land management, including salvage logging, are limited due to a dearth of studies quantifying the effect of wildfire and post-fire management on water quality. Moreover, the limited research has occurred in substantially different forest types. For example, in subalpine forests of Norway spruce in south-eastern Germany, there were no changes in maximum nitrate concentrations after salvage logging (Georgiev et al., 2021). Comparatively, within a radiata pine plantation in south-eastern Australia, the impact of both wildfire and salvage logging had a minor impact of stream solute concentrations, including NO_3^--N (Smith et al., 2012). The limited research on the efficacy of post-fire land management strategies to recoup lost economic value while reducing negative ecological impacts has led to continued debate about the potential benefits and trade-offs of these activities (Leverkus et al., 2012, 2020; McIver and Starr, 2000). The importance of addressing these knowledge deficiencies is critical due to increased fire activity in many regions, leading to post-fire land management decisions with incomplete knowledge and substantial uncertainty regarding the efficacy of the management approach (Bladon, 2018; Lucas-Borja et al., 2020; Moya et al., 2020; Zema, 2021),

5. Conclusions

The lack of pattern in post-fire soil NO_3^--N, the negative relationship we observed between burn severity and stream water NO_3^--N concentrations, and the absence of a clear connection between post-fire soils and streams is confounded by the spatial and temporal variability in post-fire land management (e.g., savlage logging) in the Hinkle Creek watershed. The observations presented here from our post-fire management period represent a cumulative response to severe wildfire within an actively managed landscape. While we were able to leverage 10 years of pre-fire observations, the near-complete burn of the Hinkle Creek watershed and near-immediate post-fire salvage logging prevented us from establishing reference sites (either unburned or wildfire, only), limiting our ability to disentangle watershed response to discrete disturbance types. Thus, while observations from this study are representative of typical post-fire land management in private industrial forests of our region, more controlled study conditions and better information on forest operations from land managers could help to disentangle overlapping forest disturbance on post-fire soil and water quality. To identify the the source and legacy of watershed disturbance impacts, which could help inform land management decisions, future research should pair long-term empirical observations of available soil nutrients (e.g., ionic resin capsules), above-ground vegetation surveys, and measurements of stream conditions (e.g., temperature, canopy cover, primary productivity).

CRediT authorship contribution statement

DeLuca Thomas H.: Writing – review & editing, Supervision, Resources, Project administration, Methodology, Funding acquisition, Conceptualization. **McCredie Katherine Elizabeth:** Writing – review & editing, Writing – original draft, Visualization, Project administration, Methodology, Investigation, Formal analysis, Conceptualization. **Bladon Kevin D.:** Writing – review & editing, Supervision, Resources, Project administration, Methodology, Funding acquisition, Conceptualization.

Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Kevin D. Bladon reports financial support was provided by The National Council for Air and Stream Improvement. Kevin D. Bladon reports financial support was provided by Oregon Forest Industries Council. Kevin D. Bladon reports financial support was provided by OSU College of Forestry Fish and Wildlife Habitat in Managed Forests Research Program. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.foreco.2025.122787.

Data availability

Data will be made available on request.

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