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Wildfire loss of forest soil C and N: Do pre-fire treatments make a difference?

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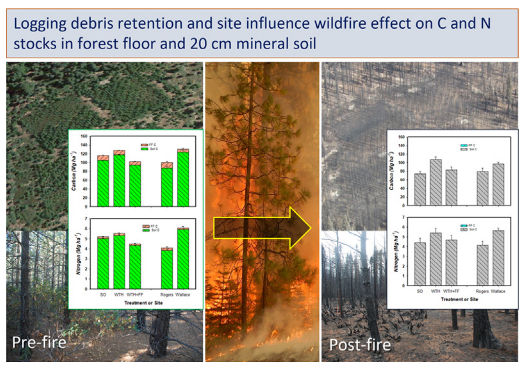
HIGHLIGHTS

- Silvicultural treatments may influence wildfire effects on soil C and N loss.
- Pre- and post-fire forest floor and mineral soil were used to quantify C and N loss.
- Severe wildfires consumed entire forest floor and subsequently all C and N in it.
- Stem only harvesting that left harvest residue on site had the largest C and N loss.
- Magnitude of soil C and N loss by fire is proportional to initial C and N contents.

GRAPHICAL ABSTRACT



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ABSTRACT

Losses of C and N from the forest floor and top 20-cm of soil were estimated following separate severe wildfires at two Long-Term Soil Productivity sites in the Sierra Nevada of California, USA. Experimental treatments applied 20 years prior to the wildfires included factorial combinations of 1) organic matter (OM) removal following clear-cut harvesting (SO, stem only harvest, WTH, whole-tree harvest, and WTH + FF, WTH plus the forest floor removal), 2) soil compaction (three levels of intensity), and 3) with and without understory vegetation control. Wildfires caused complete losses of the forest floor in all treatments and also oxidized varying portions of OM in the topsoil. As such, pre-fire forest floor measures were used as an estimate of forest floor C and N loss, and post-fire soil measures of C and N were compared to pre-fire soil data to estimate of mineral soil losses. Averaged over all treatments, the less-productive site that also had lesser accumulations of detritus (Wallace) lost 35.1 Mg C ha^{-1}, or 25 % of its original C stores, while the more-productive site with greater detritus (Rogers) lost 18.4 Mg C ha^{-1}, or 20 % of its original. The SO treatments that left harvest residue on site ended up with much greater losses of C: 36 % versus 15 and 17 % for WTH and WTH + FF, respectively. The SO also yielded the largest losses (25–30 %) of C in the top 10-cm of soil. The other treatments had smaller or inconsistent effects (understory vegetation control) or no effect (soil compaction). Our results suggest that potential benefits from SO by leaving residue on site to soil C and N accumulation can also be readily eliminated by wildfire which commonly occurs at these fire-prone forest ecosystems.

1. Introduction

Wildfires in the US West have intensified in recent decades (Westerling et al., 2006; Abatzoglou and Williams, 2016). Over the past 10 years, about 5.2 million ha of forest and grasslands have burned in California, which is

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double the 2.6 million ha burned during the previous decade. Today's fires are of unprecedented severity and have either decimated or threatened many rural communities and associated natural resources in the Sierra Nevada, including the loss of our long-term research study sites. For example, the 2014 King Fire burned over 39,000 ha and consumed all forest vegetation within the Wallace Long-Term Soil Productivity (LTSP) study site. The North Complex Fire burned 129,000 ha including the Rogers LTSP site (Fig. 1A). These two fires killed all trees and understory vegetation in the plots and consumed the entire forest floor, and burned into the top soil layers, leaving only tree stems with some upper-level branches and skeletons of manzanita (*Arctostaphylos* sp.) and remains of a few hardwood species (Fig. 1B-C). Because the LTSP study includes contrasting treatments of logging debris retention, soil compaction disturbance, and natural vegetation control, we were offered a unique opportunity to study the resilience of soil C and N following cumulative effects from silviculture manipulation and wildfire.

A considerable amount of literature exists on fire effects on soil C and N, as summarized in literature reviews and meta-analyses (Certini, 2005; Johnson et al., 2008; Nave et al., 2011; Caon et al., 2014; Pellegrini et al., 2018; Mayer et al., 2020). Some studies show that heating during fires alters soil properties and, consequently, may influence ecosystem productivity. For example, Pellegrini et al. (2018) found a mean loss of 36 % C and 38 % N due to burning in multiple grassland and broadleaf forest studies, yet minimal effects to other soil properties. Using a meta-analysis of 468 soil C and N ratios from 57 publications, Nave et al. (2011) showed that combined prescribed and wildfires reduced forest floor C mass by an average of 59 % and N mass by 50 % in temperate forests. However, fire's effect on C and N pools in the mineral soil (11 and 12 % reduction, respectively) was not statistically significant. They also found that fire severity was an important factor, as no changes were noted following low- and moderate-severity prescribed fire, whereas significant reductions were found for wildfire sites.

Because it is very difficult to design an experiment incorporating wildfire as a treatment, most studies compare post-fire soil C and N with adjacent unburned forests as "pseudo" controls (but see Johnson et al., 2007; Bormann et al., 2008). Whether such controls are appropriate given possible differences in stand structure, understory and fuel composition, microclimate, topography, and soil properties (i.e., why did a particular area not burn?) is typically unknown. Given the challenges in identifying true controls, it is perhaps not surprising that significant differences in post-fire soil C and N among fires are commonly observed (Certini, 2005; Nave et al., 2011). Johnson et al. (2007) compared pre- and post-fire nutrient pools and fluxes after the 2002 Gondola Fire burned through previously established research plots in a natural stand (Murphy et al., 2006). They concluded that loss of ecosystem C and N was primarily from combustion of the forest floor and vegetation layers, and not the mineral soil. In contrast, using a retrospective and chronosequence study with pseudo-controls, Dove et al. (2020) found 45–69 % lower mineral soil C to 100 cm depth on burned versus unburned sites across multiple wildfires (including the King Fire) in the Central Sierra Nevada of California. Bormann et al. (2008) and Homann et al. (2011), using pre- and post-fire measurements, calculated that 23 Mg C ha^{-1} and 0.7 Mg N ha^{-1} were lost from the O horizon plus top 6-cm mineral soil during the Biscuit Fire in southwestern Oregon, of which about 60 % of the losses was from mineral soil. They also reported that wildfire resulted in twice the amount of soil C and N lost compared to prescribed fire. A comprehensive review by Brown et al. (2003) concluded that loss of soil organic matter was the most serious long-term concern from wildfire and that the soil heating was mostly affected by both a large loading of surface fuels and by fuels that were widespread such as large forest floor mass or fine woody debris. These studies also demonstrated that the magnitude of fire effects depends somewhat on surface soil horizon depth, where C and N concentrations are usually highest in the soil profile. Mayor et al. (2016) found soil fertility loss

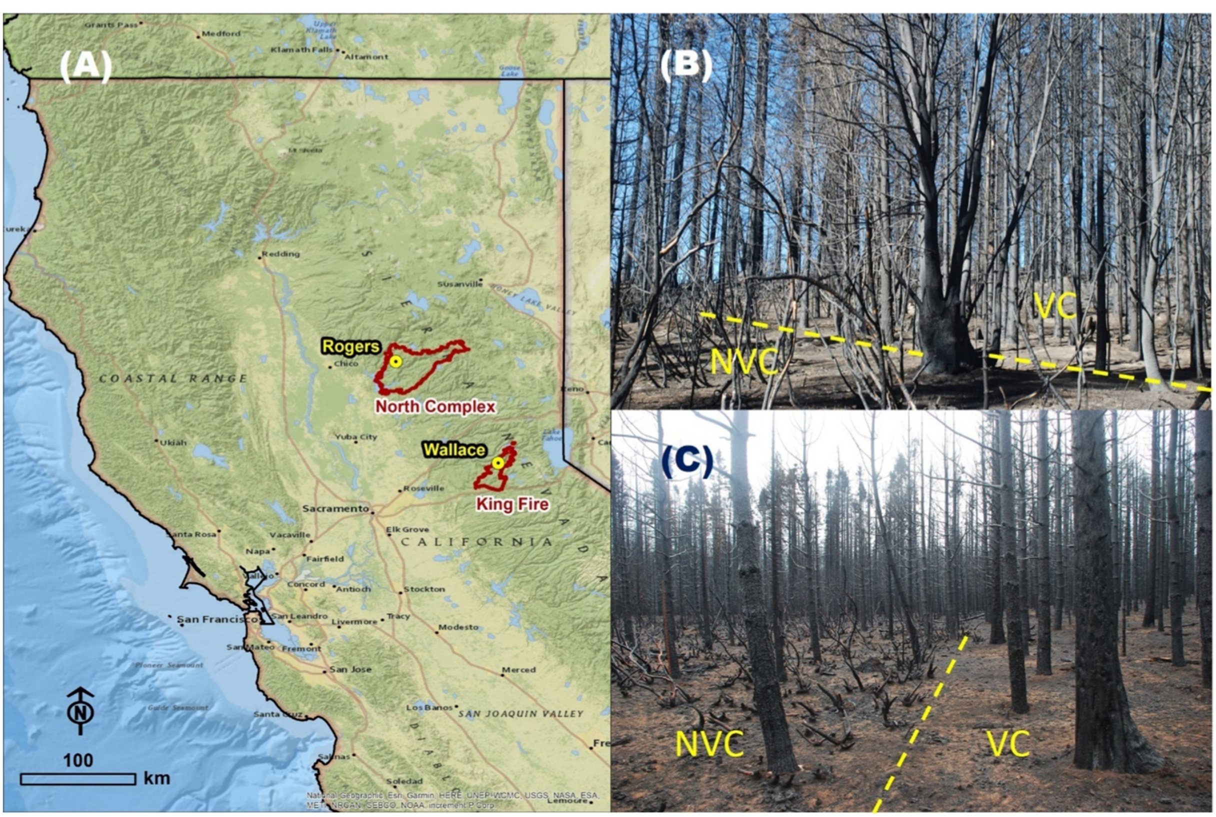

Fig. 1. (A) Locations of Rogers Long-Term Soil Productivity (LTSP) site that was burned by the 2020 North Complex Fire and Wallace LSTP site that was burned by the 2014 King Fire in Northern California, USA. The post-fire plantation shows a plot with understory present (NVC) on front and absent (NC) on the back at Rogers (B) and understory treatments at Wallace (C).

with increasing fire recurrence, including a positive effect of wildfire on soil C and N at the top 0–5 cm mineral soil eight months after the fire, but longer-term, with repeated wildfires, they detected significant drops in soil organic matter quantity and quality (Mayor et al., 2016).

The amounts of standing vegetation biomass and logging slash at the time of the fire are also important factors that influence soil responses to fire (per Brown et al., 2003). However, contrasting results have been reported, ranging from a positive relationship between fuel loading and post-fire soil C or N loss (Tomkins et al., 1991; Vose and Swank, 1993), to a neutral relationship in prescribed fire studies (Macadam, 1987; Campo et al., 2008), and to a negative relationship in a wildfire study (Homann et al., 2011). Indeed, fire effects can be light as results from the Fire and Fire Surrogates study, including three installations in California mixed conifer forests. Boerner et al. (2009) and Stephens et al. (2012) found that fuel reduction treatments including combinations of thinning and prescribed fire had short-term effects on forest floor C and N standing stocks as forest floor C and N recovered to pre-burn levels within ten years, and no change to mineral soil C or N was observed.

Rogers and Wallace are two of >60 LTSP installations located across the US and Canada (Powers, 2006). The study is the world's largest coordinated effort to understand how organic matter (logging debris) removal and soil compaction associated with tree harvesting affect soil and forest productivity (Powers et al., 1989; Powers, 2006). Tree and understory vegetation growth, forest floor accumulation, and soil chemical and physical properties have been repeatedly measured on a 5- or 10-year basis following replanting of harvested sites (Powers et al., 2005; Ponder et al., 2012; Zhang et al., 2017; Busse et al., 2021). 20-Year remeasurements were conducted in 2016 at Rogers (five years before the North Complex Fire) and in 2013 at Wallace (one year before the King Fire). Therefore, live biomass pools and fuel masses were well quantified prior to the wildfires. Our main purpose here is to assess whether the wildfire impacts on soil C and N were related to pre-wildfire treatments of organic matter removals, soil compaction and competing vegetation control. We hypothesized that (i) high levels of combustible organic material left from harvest slash retention will lead to the largest accumulations of organic matter and thus greater fire severity and the largest net loss of soil C and N, (ii) compaction will promote soil C and N loss during fire as soil bulk density and, consequently, thermal conductivity and heating increase (Abu-Hamdeh and Reeder, 2000; Ochsner, 2019), and (iii) removing understory vegetation will moderately reduce total fuel loading and, consequently, reduce fire severity, leading to lower losses of soil C and N.

2. Materials and methods

2.1. Study sites and experimental treatments

Rogers and Wallace LTSP installations were established on the western slope of the Sierra Nevada in 1996 and 1993, respectively (Fig. 1). Rogers is located on the Feather River Ranger District, Plumas National Forest (Lat. 39.7725 N; Long. 121.3205 W; Elev. 1250 m). Soils are deep (175–195 cm) sandy loam, mapped as Chaix family. The original forest was a 112-year-old, even-aged, mixed-conifer forest with standing biomass of 493 Mg ha^{-1} and forest floor mass of 77 Mg ha^{-1} (Powers, 2006). Wallace is located on the Georgetown Ranger District, Eldorado National Forest (Lat. 38.9707 N; Long. 120.5194 W; Elev. 1567 m). The soil is moderately deep (90–150 cm) sandy loams, mapped as McCarthy-Ledmount and Crozier-McCarthy Associations. The original forest was a 230-year-old, even-aged, mixed-conifer forest with standing biomass of about 450 Mg ha^{-1} and forest floor mass of 116 Mg ha^{-1} (Powers, 2006).

Both sites are characterized by a Mediterranean climate, with hot, dry summers and mild or cold, wet winters. Annual precipitation is about 2100 mm at Rogers and 1500 mm at Wallace. About 600 mm of precipitation fell between the wildfire and soil collection dates at Rogers (Aug 19, 2020 to April 8, 2021) base on a nearby RAWS weather station (https://raws.dri.edu/cgi-bin/rawMAIN.pl?caCJAR). At Wallace, about 108 mm of precipitation was recorded (Sept 17 to Nov 12, 2014) at the closest weather station (https://raws.dri.edu/cgi-bin/rawMAIN.pl?caCHEL).

Two main factors common to the LTSP network were applied: organic matter removal and soil compaction during harvesting of mature forests. Organic matter (OM) removal treatments included three levels: stem only (SO) where only tree stems were removed and harvest slash was retained on site; whole-tree harvest (WTH) where the entire tree was removed; and whole-tree harvest plus forest floor removed (WTH + FF), where the forest floor was removed along with the entire harvested trees. The OM treatments were combined in a factorial design with three levels of soil compaction: none, intermediate, and severe where the forest floor was temporarily removed and vibrating plate compactors were manually run over the intermediate and severe compacted plots and forest floor material then replaced in a somewhat mixed state (see Busse et al., 2021 for details). The three-by-three factorial design resulted in nine treatment plots (0.4-ha) at each site. Each plot was then divided into two, 0.2-ha subplots, with and without vegetation control. If uniform ground was not suitable for two contiguous splits, two subplots were separately located. In the no-vegetation control (NVC) subplot, all vegetation, including both planted trees and naturally regenerated vegetation, were allowed to develop. On the other subplot, the treatment was vegetation control (VC), where only planted trees were kept, which was achieved with repeated removal of non-planted vegetation as necessary for the first five years.

Three tree species: ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), and sugar pine (*Pinus lambertiana* Douglas) were planted at both sites. In addition, a fourth Douglas-fir (*Pseudotsuga menziesii* var. *menziesii* (Mirb.) Franco) was planted at Rogers and giant sequoia (*Sequoiadendron giganteum* (Lindl.)) was planted at Wallace.

2.2. Vegetation and forest floor measurements

Detailed protocols for vegetation measurements can be found in Zhang et al. (2017). Briefly, an inner 0.1-ha measurement plot was established in each of the 0.2-ha subplots, and all planted trees were tagged and measured at age 5, 10, and 20 years. Tree height, diameter at 1.37 m height (dbh), height to live crowns, and crown width in two directions were also measured for all planted trees. Using these measurements, we estimated individual-tree biomass using allometric equations developed for northern California conifer forests or elsewhere if allometric equations were not available for a given species (Powers et al., 2013). Aboveground biomass of competing vegetation was also measured. With both planted trees and competing vegetation measurements, we were able to estimate aboveground tree biomass (TreeAGB) and total aboveground vegetation biomass (TAGB).

Forest floor mass (FF_M) including woody debris and slash was also quantified at year 20 and added to TAGB to determine total combustible fuels (Fuel_M). FF_M (combined litter + duff) was measured as the depth of the O horizon at 30 systematically located points per split plot (Busse et al., 2021). Average depth per split plot was converted to mass by multiplying by plot-specific bulk density values, determined by collecting, drying (65 °C for at least 72 h), and weighing all O horizon organics within 30 × 23 cm frames (8 samples per split plot). Organic matter concentration was determined using an average loss-on-ignition (LOI) for forest floor materials collected from a different study at a similar site quality (Zhang et al., 2020). Then, we estimated forest floor carbon stock by multiplying forest floor mass by 0.58 (Van Bemmelen's estimate of C to OM ratio; Minasny et al., 2020).

2.3. Soil sampling and analysis

Soil samples were collected in year 20 from three depths (0–10, 10–20, and 20–30 cm) for soil physical and chemical analyses using a volumetric core sampler. This corresponded to five years before the North Complex Fire at Rogers and two years before the King Fire at Wallace. Detailed sampling methods can be found in Busse et al. (2021). Briefly, composite samples (9 systematic locations per split plot) were dried to constant weight at 105 °C, sieved (2 mm), and weighed to determine fine soil bulk density. Organic C and N concentrations were determined on fine fraction samples by dry combustion using a CNS analyzer. Concentrations were converted to total content (mass) by multiplying the concentrations by the fine fraction bulk density of each sample.

Post-fire sampling procedures and measurements were the same as the pre-fire protocols sampling except, (1) samples were only collected from 0–10 cm and 10–20 cm depths given the lack of heat penetration typically found below 20 cm during wildfire (Certini, 2005; Caon et al., 2014; Ochsner, 2019), (2) bulk density was not collected at the post-fire Wallace samples, thus bulk density values were estimated using the pre-fire and post-fire regression relation for the Rogers site where individual post-fire bulk density samples averaged 97 % of the paired pre-fire bulk density samples and 90 % of the pairs were within 10 % of each other (Fig. S1); and (3) soil C and N at Wallace were determined by potassium dichromate oxidation and ferrous ammonium sulfate titration, respectively (Nelson and Sommers, 1996). We corrected C and N values using relations established between the two methods from our laboratory measurements that indicated the chemical analysis averaged 2 % less C and 95 % of paired samples were within 10 % of each other (chemical analyses v. dry CNS combustion, Fig. S2).

Since the forest floor was completely consumed in both fires, and there were no post-fire collections of forest floor. As such, C and N losses (FF_C_{loss} and FF_N_{loss}) were assumed equivalent to the pre-fire forest floor C and N stocks.

2.4. Statistical analysis

All variables were analyzed based on a split-plot, randomized, complete-block design with treatments as the fixed effect and site (regarded as a block) as a random effect using SAS PROC MIXED (SAS Institute Inc., 2012). OM and compaction (Comp) were regarded as the main plot effect and vegetation treatment (VC and NVC) was as subplot effect. We used α = 0.10 as the critical value due to only 2 sites. The statistical model is:

$$y_{ijkl} = \mu + \alpha_i + \beta_j + \alpha\beta_{ij} + \gamma_k + \varepsilon_{1ijk} + \phi_l + \alpha\phi_{il} + \beta\phi_{jl} + \alpha\beta\phi_{ijl} + \varepsilon_{2ijkl} \quad (1)$$

where y_{ijkl} is the dependent variable summarized for the ith OM treatment, jth Comp level, and the kth site, and lth understory vegetation control split, μ is the overall mean, α_i and β_j are the fixed effect of the ith OM or jth Comp ($i = 1, 2,$ and 3), ϕ_l is the fixed effect of the lth understory vegetation ($l = 1$ and 2), γ_k is the random effect of the kth site ($j = 1, 2, \ldots,$ and 12), $\gamma_k \sim N(0, \sigma_B^2)$, and ε_{1ijk} is an experimental error to test main plot effect and ε_{2ijkl} is for the rest of the terms, $\varepsilon_{1ijk} \sim iid\ N(0, \sigma_{e1}^2)$ and $\varepsilon_{2ijkl} \sim iid\ N(0, \sigma_{e2}^2)$.

To test treatment effects on wildfire-caused changes in soil C and N concentrations, C/N, and pre-fire fine bulk density (BD_f), we used relative changes ($X_{post\text{-}fire}/X_{pre\text{-}fire}$) by adding depth to the model, where X was the variable. We did so because we assumed that these variables reacted similarly to the fires between sites and among treatments.

For each variable analysis, residuals were examined to ensure that statistical assumptions of normality and homoscedasticity were met. If not, a natural log transformation was applied. During the model selection process, we selected the model with the minimum Akaike information criterion (AIC). Multiple comparisons among treatments were conducted for least squares means by the Tukey-Kramer test by controlling for the overall α = 0.10.

3. Results

3.1. Pre-fire plantation features

3.1.1. Standing vegetation biomass

After 20-years of plantation growth, the vegetation control (VC) treatment was particularly effective in removing competing vegetation and thus had a particularly large effect on growth of planted trees (TreeAGB). With the nearly complete removal of competing vegetation, the planted trees in VC had approximately twice biomass ($P < 0.01$) as the planted trees in the no-vegetation control (Fig. 2A; Table 1). Due to prevalent shrubs at both sites and some hardwood species at Rogers, total vegetation aboveground biomass (TAGB) did not statistically differ between vegetation treatments. In addition, no other effects were significant for any variables (Table 1) although stem only harvest (SO) showed more total biomass than the WTH and WTH + FF (Fig. 2B).

3.1.2. Forest floor mass

At 20 years of treatments, the forest floor mass (FF_M) was 20 % greater in VC than in NVC ($P = 0.09$) (Fig. 2A; Table 1). This result was unexpected

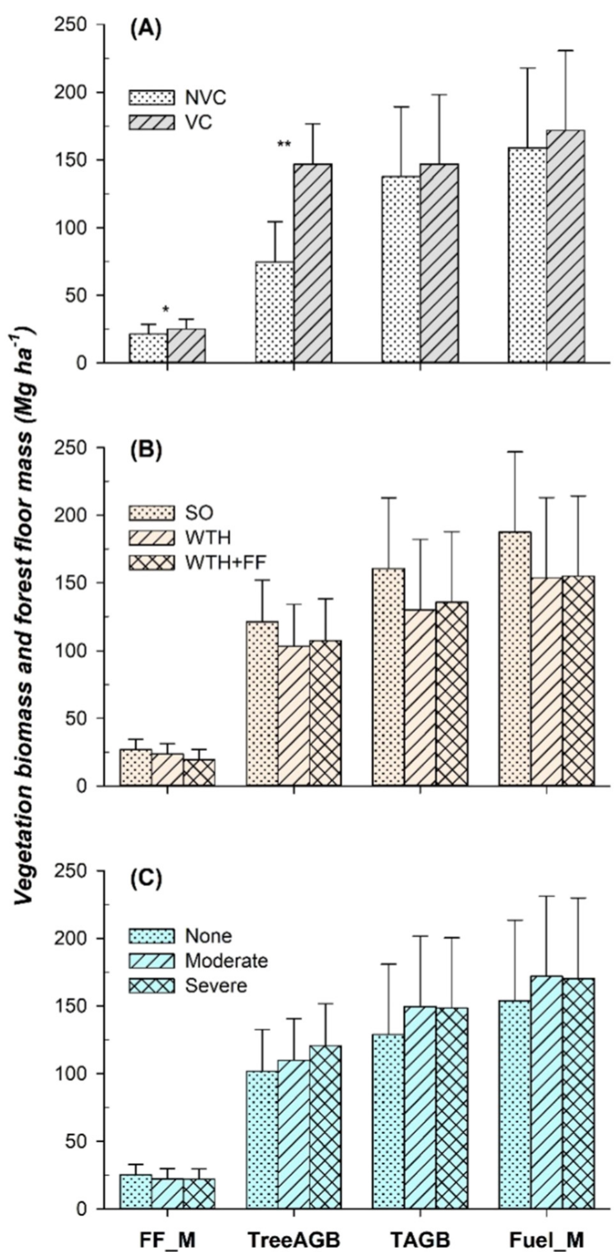

Fig. 2. Least square means and standard errors from two replicated plantations of organic matter accumulation trends at year 20 of growth (pre-fire). Shown are the three main treatments. A: competing vegetation control (VC) or no control (NVC); B: OM removals following harvesting (stem only, whole-tree, whole-tree with forest floor removed), and C: three levels of soil compaction. Responses are FF_M forest floor mass, TreeAGB planted trees aboveground mass, TAGB total vegetation aboveground mass, and Fuel_M sum of all fuels or FF_M plus TAGB mass. * indicates difference at $P < 0.10$ and ** indicates difference at $P < 0.01$.

Table 1
Probabilities of analyses of variance from PROC MIXED model for planted tree aboveground biomass (TreeAGB), total AGB (TAGB: planted trees + competing vegetation), forest floor mass (FF_M), combustible fuel mass (Fuel_M: total AGB + FF_M), soil C loss, soil N loss, total C and N loss (FF + soil). Also shown are relative changes of C, N, and C/N concentration (post-fire/pre-fire). Treatments are soil compactions (Comp), organic matter removals (OM), vegetation control (Veg). Bold numbers indicate significant difference at P < 0.10.

Effect	Num DF	Den DF	TreeAGB	TAGB	FF_M	Fuel_M	Soil C loss	Soil N loss	Total C loss	Total N loss	Rel [C]	Rel [N]	Rel C/N ratio
Compaction	2	8	0.51	0.49	0.75	0.63	0.60	0.84	0.54	0.81	0.19	0.80	0.38
OM	2	8	0.52	0.27	0.31	0.22	**0.09**	0.20	**0.08**	0.17	0.15	**0.03**	0.22
Comp * OM	4	8	0.81	0.98	0.50	0.95	0.73	0.56	0.69	0.52	0.83	0.86	0.77
Veg	1	9	**<0.01**	0.56	**0.09**	0.45	0.91	0.18	0.75	0.16	**0.08**	0.67	**<0.01**
Veg * Comp	2	9	0.69	0.96	0.33	0.92	0.84	0.98	0.90	0.97	0.96	0.71	0.45
Veg * OM	2	9	0.88	0.90	0.25	0.95	0.55	0.85	0.62	0.88	0.36	0.14	0.90
Veg * Comp * OM	4	9	0.88	0.97	0.23	0.96	0.56	0.37	0.58	0.38	0.65	0.63	0.99

given the lack of difference in TABG growth and likely reflected the higher percentage of decay-resistant conifer litter for VC compared to NVC. Conversely, neither OM removal nor soil compaction had a statistically detectable effect on forest floor mass. However, the trends in forest floor mass among OM removals were the same as the growth responses of TreeAGB and TAGB, with WTH and WTH + FF showing 10 % and 20 % lower FF_M than SO (Fig. 2B). Compaction showed a trend of reduced FF_M (Fig. 2C).

3.1.3. Fuel mass

Collectively, total fuel mass (Fuel_M) followed the same patterns as the vegetation. VC accumulated more fuel mass than NVC (171.9 versus 158.8 Mg ha^{-1}, respectively), but the differences were not statistically significant ($P = 0.45$). Fuel masses for the OM removal treatments were 187.5 Mg ha^{-1} for SO, 153.7 Mg ha^{-1} for WTH, and 155.0 Mg ha^{-1} for WTH + FF, respectively ($P = 0.22$). No significant compaction effect was found either ($P = 0.63$) with 154.0 Mg ha^{-1} for no compaction, 171.8 Mg ha^{-1} for moderate, and 170.4 Mg ha^{-1} for severe compaction levels, respectively.

Although the random site effect is not outputted as the fixed effect terms in the mixed statistical model, it is worth noting that two sites differed substantially in productivity regardless of vegetation and forest floor. For example, combustion fuels (Fuel_M: trees + understory + FF_M) at Rogers (223.6 Mg ha^{-1}) was more than doubled at Wallace (107.2 Mg ha^{-1}). Yet, the trends among treatments within a site were not significantly different between the two sites based on preliminary analyses (data not shown).

3.2. Forest floor C and N losses

Because there were essentially no remaining forest floors at both sites following the fires, the pre-fire estimates of forest floor C and N were essentially the estimates of loss. Therefore, both trends and statistical analyses of pre-fire standing stocks and losses post fire would yield the same results for FF_M (Table 1; Fig. 2A–C). C and N loss was significantly greater in VC than in NVC (Fig. 3). In addition, the trends of more SO loss in C and N than other two removals and in no compaction than in moderate and severe compaction levels were also observed because they had more C and N in the larger organic matter accumulations to start with.

3.3. Mineral soil C and N

3.3.1. C and N concentration changes ($X_{post\text{-}fire}/X_{pre\text{-}fire}$)

Because of a lack of a soil depth effect and a lack of depth-associated interactions for soil variables, we used the model [Eq. (1)] to test the significance of the treatment effect. Relative C concentration change showed a significant effect of vegetation treatment with an increase C concentration ratio for NVC and a decrease C ratio for VC ($P = 0.08$; Table 1; Fig. 4). No other treatment effects were found to be statistically significant. As for N concentration ratio, we only found a significant OM effect ($P = 0.03$) with SO being reduced by the fires. Relative C/N ratio changes differed between NVC (1.09) and VC (0.93) reflecting the proportionally greater loss of C versus N in the VC treatment.

While the primary goal of the study was to determine the treatment effect on responses of soil C and N to the wildfires, site variation and other treatment means for these variables are provided in Table S1. Overall, soil C and N concentration changes were minor at Rogers where carbon was reduced by fire from 5.91 (±0.47 SE) % to 5.08 ± 0.35 % at 0–10 cm; and increased from 3.70 ± 0.29 % to 4.11 ± 0.39 % at 10–20 cm (Table S1). N increased at both levels from pre-fire 0.22 ± 0.02 % to post-fire 0.23 ± 0.02 % at 0–10 cm and 0.14 ± 0.01 % to 0.18 ± 0.02 % at 10–20 cm, respectively. Larger changes occurred at Wallace, where soil C concentrations were reduced by fire from 11.77 ± 0.85 % to 8.74 ± 0.43 % at 0–10 cm and from 9.31 ± 0.63 % to 7.09 ± 0.37 % at 10–20 cm. Similarly, both depths

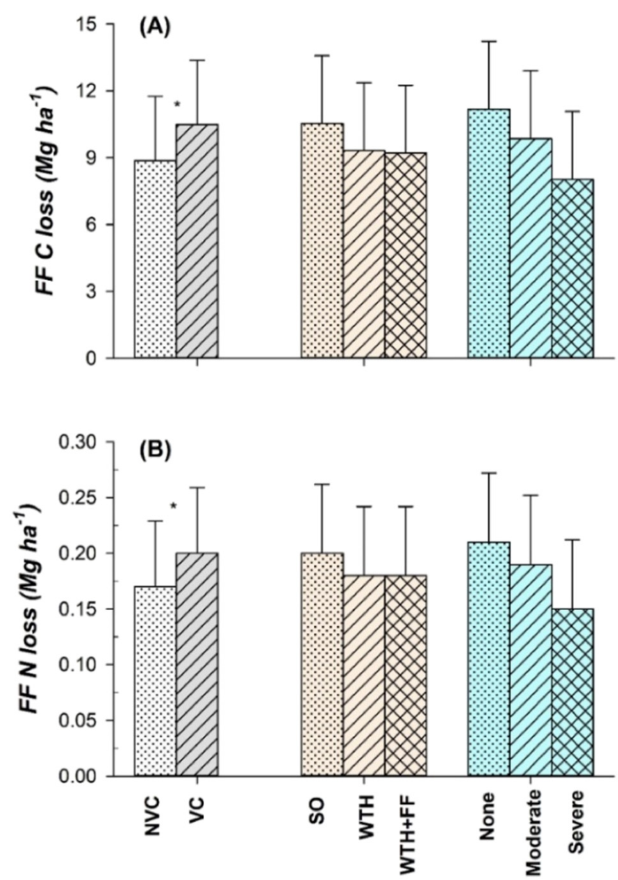

Fig. 3. Least square means with standard errors for forest floor carbon (A) and nitrogen (B) loss from the fires between competing vegetation treatments, among organic matter removals, and soil compaction levels. Abbreviations are the same as those in Fig. 2. * indicates difference at $P < 0.10$.

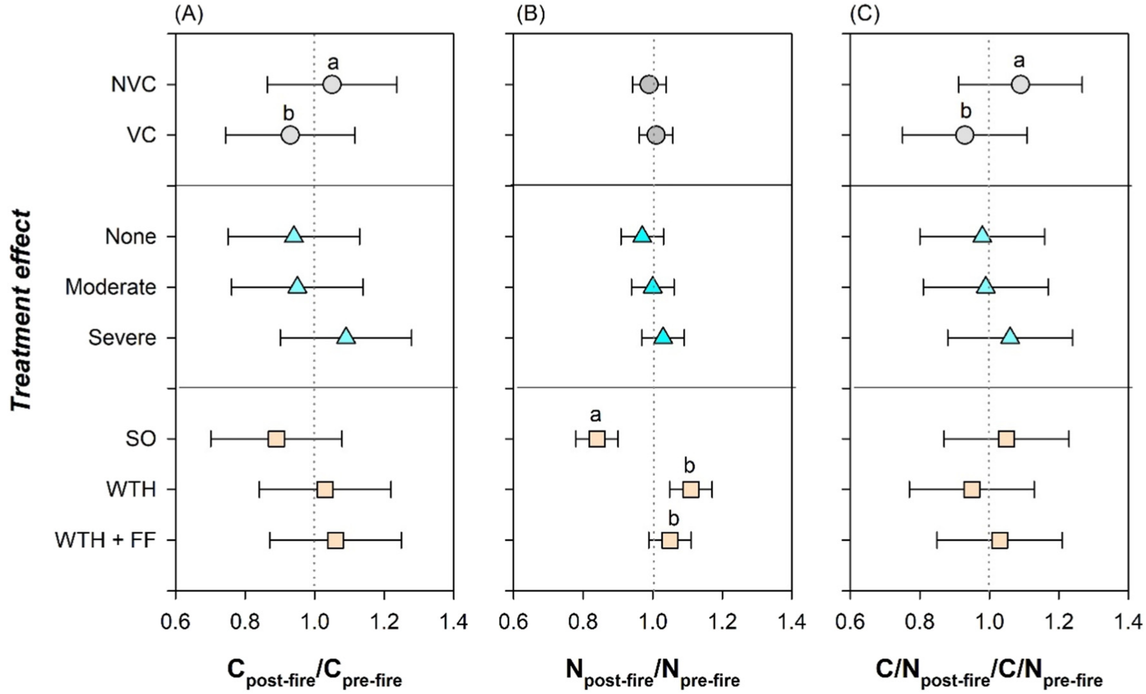

Fig. 4. Relative changes (least square means and standard errors) in top 20 cm soil C and N concentrations and C/N between post-fire and pre-fire for treatment effects. The significant effects are showed with different letters ($P < 0.10$).

showed proportionally smaller reduction of N concentrations with 0.56 ± 0.04 % to 0.43 ± 0.03 % at 0–10 cm and 0.46 ± 0.04 % to 0.35 ± 0.00 % at 10–20 cm. Other treatment means are also presented in Table S1.

3.3.2. Changes in bulk density

Prior to the fires, fine soil bulk density (BD_f) differed significantly between NVC and VC, among compaction levels (severe > intermediate > none), and among OM removals (WTH + FF > WTH > SO) in addition to depth differences (top 10 cm < 10–20 cm) (Table S2). The only significant interaction occurred between compaction and OM. At Rogers, the fire had minimal effect on BD_f at 0–10 cm (pre-fire 0.95 vs post-fire 0.92 Mg m^{-3}) and at 10–20 cm (1.08 to 1.05 Mg m^{-3}) (Table S2). On average, pre-fire BD_f was over 62–67 % higher at Rogers than at Wallace. Although post-fire bulk density was not measured at Wallace, there were no reasons to expect that fire effect on BD_f among treatments differed from Rogers. Therefore, post-fire bulk density changes at Wallace would be estimated based on fire effect on BD_f at Rogers.

3.3.3. Changes in soil C and N content

Soil carbon loss from top 20 cm differed significantly only among the OM removals ($P < 0.10$, Table 1) with SO showing greater losses than WTH or WTH + FF: (30.3, 11.0, and 10.0 Mg ha^{-1}, respectively, Table 2). Wallace showed a greater loss than Rogers. Greater C loss was found at 0–10 cm (12.4 Mg ha^{-1}) than at 10–20 cm (5.0 Mg ha^{-1}) as expected (data not shown).

No treatment effect was significant for N losses (Table 1). Site differences were observed with a gain of 0.29 Mg ha^{-1} at Rogers versus a loss of 0.39 Mg ha^{-1} at Wallace in the surface 20 cm profile (Table 2). Although not statistically significant, OM treatment trends included a loss of 0.55 Mg ha^{-1} for SO, a gain of 0.03 Mg ha^{-1} and 0.37 Mg ha^{-1} for WTH and WTH + FF.

3.4. Total C and N loss or gain

Total carbon loss due to wildfire in the top 20 cm of mineral soil + forest floor differed significantly among OM removal treatments (SO > WTH = WTH + FF (Tables 1 and 2). When expressed as percentage C

Table 2
Means (SE) of C and N loss from 20 cm depth of mineral soil, forest floor, and the total losses as well as their percentages of total losses of C and N stocks from wildfires at Rogers and Wallace. Random site effect was not statistically tested. Veg control nor soil compaction had no statistically detected effect.

Variable/effect		Mineral soil at 20 cm	Forest floor	Total	Loss %
C loss (Mg ha^{-1})					
Site	Rogers	5.89 (3.95)	12.54 (1.08)	18.4 (4.19)	20.2 (4.56)
	Wallace	28.30 (6.14)	6.84 (0.64)	35.1 (6.37)	25.3 (5.11)
OM effect	SO	30.30 (5.28)a	11.18 (1.49)a	42.1 (5.02)a	35.8 (3.62)a
	WTH	11.03 (6.95)b	9.86 (1.25)ab	20.4 (6.59)b	15.4 (4.79)b
	WTH + FF	9.96 (7.60)b	8.03 (1.29)b	18.9 (6.69)b	16.9 (7.13)b
N loss (Mg ha^{-1})					
Site	Rogers	−0.29 (0.21)	0.24 (0.02)	−0.04 (0.21)	−1.70 (5.57)
	Wallace	0.39 (0.31)	0.13 (0.01)	0.52 (0.31)	7.03 (5.59)
OM effect	SO	0.55 (0.34)a	0.21 (0.03)a	0.77 (0.34)a	14.46 (6.81)a
	WTH	−0.03 (0.26)a	0.19 (0.02)a	0.16 (0.26)a	3.16 (5.27)a
	WTH + FF	−0.37 (0.36)a	0.15 (0.03)a	−0.22 (0.36)a	−4.53 (7.45)a

Note: Numbers with different letters within each effect indicate difference at P < 0.10.

loss relative to pre-fire C, SO had twice the relative C loss compared to the other OM treatments. On average, just under one-quarter of the C was lost due to wildfire, with more than one-third lost from SO treatment (35.8 %), or about twice the percentage of WTH (15.4 %) or WTH + FF (16.9 %). It is worth to note that Wallace showed a substantial more C loss (35.1 Mg ha^{-1}) than Rogers (18.4 Mg ha^{-1}). Similar trends were found for total N loss (-0.04 Mg ha^{-1} at Rogers vs. $+0.55$ Mg ha^{-1} at Wallace). No other effects were significant (Table 1), although the trend among OM treatments was similar to total C losses (Table 2).

Forest floor C and N losses were positively related to the combined biomass variables ($r > 0.60$, $P < 0.01$) (Fig. S3). Essentially, more fuel (particularly forest floor organics) led to larger C and N losses. However, no individual variable was significantly related to soil and total C or N loss ($-0.29 \le r \le 0.12$, $P > 0.10$) (Fig. S3).

4. Discussion

Weather, topography, and fuels are main factors controlling wildfire behavior. Due to increased fuel loads, global warming, and drying weather in the western United States, the entire West, and California in particular, has experienced many so-called megafires in the last decade, that burn extensive landscapes at high intensity and severity (Millar and Stephenson, 2015; Parks and Abatzoglou, 2020). Because weather and topography are not easily altered, actively managing fuels is the only option of these three to reduce fire hazard. Treatments applied in the LTSP study (OM removal, compaction, and understory vegetation) provide a rare opportunity to determine how such treatments influence wildfire effects. Results in our study demonstrate that severe wildfires on these sites effectively consumed the entire forest floor and killed all standing vegetation regardless of prior treatment; but did not uniformly affect soil C and N across sites or among OM removal treatments.

Our results support a portion of the first hypothesis that high levels of combustible organic material from harvest slash retention in the OM treatment (SO and WTH) will lead to the largest net loss of soil C and N in the forest floor (Table 1, Fig. 3). Therefore, soil C and N increases from SO and WTH treatment (Busse et al., 2021) were eliminated by high-severity wildfires. However, although there was substantially more aboveground biomass at Rogers than at Wallace, we found less total C and N losses at Rogers because of greater mineral soil C and N losses at Wallace (Table 2). There are several potential explanations. First, there were larger C and N pools in the mineral soil at Wallace than at Rogers; the higher C content and perhaps more labile C at Wallace were more susceptible to burning (Busse et al., 2014; Muqaddas et al., 2019). Second, C and N pools in mineral soil are much larger than the pools in the forest floor, especially in young plantations (McFarlane et al., 2009; Mattson and Zhang, 2019; Busse et al., 2021). Therefore, large C and N losses from mineral soil as opposed to losses from forest floor may better reflect fire intensity and severity at Wallace. Third, although soil texture was a sandy loam at both sites (Zhang et al., 2017; Busse et al., 2021), different elevation, slope, aspect, vegetation structure complexity, soil moisture content, and weather and fuel conditions could have resulted in fire severity varying between sites (Rothermel, 1972; Holden et al., 2009; Busse et al., 2010). In fact, by reconstructing the King Fire with airborne observation and microscale simulations, Coen et al. (2018) found that fire-induced winds exceeded an order of magnitude of the ambient winds when it burned through the Wallace plots, which would add greater levels of heat. Finally, site differences in time duration and precipitation amount between the fires and sample collection dates might have resulted in differential erosion or leaching losses of C and N (Busse et al., 2014).

Among OM treatment, the SO treatment showed consistently more C and N losses regardless of understory vegetation treatment or compaction (Table 2), suggesting that more logging-debris retention during harvest over 20 years ago but still remaining resulted in more heat generation and more total C and N losses. The SO treatment plots are thought to have had more forest floor and total combustible biomass, from debris left from the previous harvests. Essentially, with higher OM content, C and N losses are expected to be higher.

The results pose a dilemma for foresters where keeping more slash residues for the benefits of increased soil organic matter and nutrients may be also creating the added fuels for increased wildfire severity that not only consumes the entire forest floor, but also causes additional C and N losses in the mineral soil beyond that added by the residue. Surprisingly, the WTH + FF treatment showed similar C loss and less N loss compared with WTH, where it was expected that reducing the fuels of the forest floor by removal at the start of the experiment may have reduced fire severity and therefore C loss. Because no logging debris or forest floor were kept on WTH + FF from the previous stand, soil organic matter and nitrogen pools were exclusively influenced by the current tree stand.

The large differences we observed in mineral C and N losses between sites demonstrates the challenges that can occur when comparing results from different studies, especially using absolute values, with their own unique soil type and stand history and condition. Nonetheless, our forest floor results are consistent with the previous comparable studies that measured C and N before and after fire in a high elevational Sierra Nevada pine forest (Murphy et al., 2006; Carroll et al., 2007; Johnson et al., 2007). They found fire consumed an average of 94 % C and 92 % N from the forest floor, which is similar to our estimates of 100 % for C and N. Their estimate of 10 Mg ha^{-1} forest floor C loss falls within our range for Wallace and Rogers (6.8 to 12.5 Mg ha^{-1}) largely because their pre-fire forest floor masses were similar to ours. The numbers are also comparable to another wildfire study with pre- and post-fire soil sampling in southwestern Oregon. Bormann et al. (2008) and Homann et al. (2011) estimated the C loss from the O-horizon to be 9 Mg ha^{-1}. The losses estimated from these high intensity wildfires are much higher than what has been summarized in meta-analyses or syntheses (Wan et al., 2001; Certini, 2005; Nave et al., 2011; Pellegrini et al., 2018). Although Nave et al. (2011) included all earlier mentioned wildfires, they also included some studies with estimates obtained in retrospective studies or extrapolated from the laboratory. Compared with results from lower-intensity prescribed fires on soil C and N (Boerner et al., 2009; Stephens et al., 2012), we found that wildfires burning under severe conditions would pose significantly greater soil C and N loss, which raises caution in predicting wildfire effects using data and trends from prescribed fire studies.

Our second hypothesis postulated that pre-existing soil compaction increased mineral soil C or N loss during wildfire. It is well established that heat transfer is most effective through solid compared to porous material (Hopmans and Dane, 1986), suggesting higher soil temperatures, and, consequently, carbon losses in compacted soil. In our study, both sites had about 20 % greater bulk density in compacted compared to non-compacted soils (Table S2; Busse et al., 2021). Despite this, we found limited evidence of differential C or N loss due to compaction. This finding can be attributed to a few factors. Foremost, soil heat transfer during fire is controlled predominantly by soil moisture content and fuel mass consumption (Busse et al., 2010). In our case, compacted and non-compacted soils had similar, extremely-low (late summer) moisture content and comparable fuel loads (by experimental design). In addition, a 20 % increase in bulk density in these sandy loam soils corresponds to a relatively minor, 8 % reduction in total porosity (Busse et al., 2021).

Fire oxidizes soil organic matter and releases CO_2, and other gases, and creates charcoal and ash. Charcoal, in particular, can be mixed into the soil and increase soil C and N (Rau et al., 2009; Busse et al., 2014). In their meta-analysis of fire effects on soil C, including 48 study sites from 13 publications, Johnson and Curtis (2001) found the effect of prescribed fire was 6 % less carbon than controls, whereas wildfire generally resulted in higher soil C content due to the incorporation of unburned residues and deposition of charcoal. In this study, we have also seen that some treatment plots gained C and N. Dyrness et al. (1989) speculated that different fuels in the forest floor caused both increases and decreases in soil C and N in their study in Alaska. If that is true, our forest floor fuels should only differ between NVC and VC treatments. Yet, both treatments showed variable loss and gain of C and N at both sites. One possible explanation is that between the times of fire and soil sampling, there were about 600 mm precipitation at Rogers and about 100 mm at Wallace according to the closest RAWS weather stations. Some C and N in charcoal or unburned wood or bark might have leached into the mineral soil (Busse et al., 2014; Mattson and Swank, 2014). This possibility was supported by the C and N concentration changes between the two sites. Compared to pre-fire concentrations, post-fire C and N concentrations increased at Rogers in the 10–20 cm depth but decreased at the 0–10 cm depth and decreased at both depths at Wallace (Table S1). In addition, there was a possibility of topsoil erosion at both sites (Busse et al., 2014). Indeed, small-scale rilling and re-deposition was observed on some of the plots, even though the slope is gentle at both sites (<15 %). The rills were infrequently distributed across plots, however, and thus likely had a minimal effect on our 0–20 cm sampling profile.

Long-term research projects, like the LTSP Study, provide invaluable information for forest practitioners and scientists by offering conclusive evidence of forest resilience, cumulative impacts from sequential disturbances, and the role of climate change as a driver of forest structure, composition, and function. Clearly, there is value in maintaining long-term research studies that can serve as an important benchmark for understanding natural resource sustainability. However, maintaining long-term plots is not an easy proposition, as external pressures from budget constraints, unforeseen disturbances (e.g., wildfire), changing research priorities, and retirement of scientific staff can be limiting. For example, changing drought and wildfire regimes have contributed to the loss of 5 out of the original 12 LTSP research installations in California alone via either wildfire or insect infestations. Given the increases of wildfire frequency and intensity as well as stresses across the world, protecting these long-term research installations becomes a serious challenge. These disturbances pose a huge challenge to every forest researcher and are an undeniable reason to treasure long-term data from these plots.

5. Conclusions

With pre- and post-wildfire sampling, we demonstrated that the cumulative effects of harvest disturbance and wildfire on soil C and N varied by site. Both the North Complex and King fires consumed entire forest floors which contained about 12.5 Mg C ha^{-1} and 0.24 Mg N ha^{-1} at Rogers and 6.8 Mg C ha^{-1} and 0.13 Mg N ha^{-1} at Wallace. The wildfire produced unexpected patterns of loss of detrital C and N as a result of interactions between the growth rates of the planted conifers and the competing vegetation and the treatments. On average, 25 % of detrital C (forest floor and mineral soil C) to 20 cm depth was lost to fire at poorer-growth Wallace site (35.1 Mg ha^{-1}) versus 20 % loss at the better- growth Rogers site (18.4 Mg ha^{-1}). The Wallace site, despite poorer growth, had more pre-fire soil C and therefore lost more soil C post fire. Soil compaction had slight trend of increasing detrital C or N loss from fires that was not statistically significant. In the stem-only removals during clearcut harvests (SO), which left the most residue on site and which also was associated with a non-significant trend of greater forest growth, loss of C and N after fires was over twice as great as whole tree harvest treatments that removed more organic matter. But the additional removal of the forest floor with whole tree harvests treatment showed no different loss compared to the whole-tree harvests which left the forest floor on site. Our sites show that stem-only harvesting with branches and foliage left on site yielded greater C and N losses to wildfires. Understory vegetation control (removal) increased pre-fire forest floor C and N mass, but had no effect on overall loss C and N to wildfires.

CRediT authorship contribution statement

Jianwei Zhang: Conceptualization, Investigation, Methodology, Data analysis; Writing – original draft preparation. **Matt Busse:** Methodology, Data interpretation, Writing – review and editing; **Silong Wang:** Methodology, Sample analysis, Writing – review and editing; **Dave Young:** Investigation, Sample collection, Writing – review and editing; **Kim Mattson:** Investigation, Sample collection, Data interpretation, Writing – review and editing.

Data availability

Data will be made available on request.

Declaration of competing interest

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

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Appendix A. Supplementary data

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