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Fire-Induced Carbon Loss and Tree Mortality in Siberian Larch Forests

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Key Points:

- Siberian larch forest fires cause high tree mortality despite species traits and stand structure hypothesized to promote low severity fires
- Compared with boreal North America, fires in Siberian larch forests result in greater aboveground C loss but lower belowground C loss

Supporting Information:

Supporting Information may be found in the online version of this article.

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Abstract Climate change is intensifying the fire regime across Siberia, with the potential to alter carbon combustion and post-fire carbon re-accumulation trajectories. Few field-based estimates of fire severity (e.g., carbon combustion and tree mortality) exist in Siberian larch forests (*Larix* spp.), which limits our ability to project how an intensified fire regime will affect regional and global climate feedbacks. Here, we present field-based estimates of fire-induced tree mortality and carbon loss in eastern Siberian larch forests. Our results suggest that fires in this region result in high tree mortality (means of 83% and 76% at Arctic and subarctic sites, respectively). In both absolute and relative terms, aboveground carbon loss following fire is higher in Siberian larch forests than in North America, but belowground carbon loss is considerably lower. This suggests fundamental differences in wildfire behavior and carbon dynamics between dominant vegetation types across the boreal biome.

Plain Language Summary With climate change, forest fires in Siberia are expected to become more severe and more frequent, which could amplify climate change by transferring carbon from the ecosystem to the atmosphere. Although Siberian larch forests make up 20% of the boreal forest biome by area, scientific understanding of the Siberian larch fire regime is limited because the region is remote and mostly roadless. We collected data from burned and nearby unburned Siberian larch forests to understand the effects of fire on the ecosystem. We found that fires in Siberian larch forests kill, on average, about 75% of trees and result in large carbon losses to the atmosphere. These observations of tree mortality and carbon loss are higher than reported in most satellite-based studies and demonstrate that fires in Siberian larch forests could contribute to ongoing climate change.

1. Introduction

Boreal forests play an important role in Earth's climate and terrestrial systems, comprising ~50% of global terrestrial carbon (C) stocks while occupying only 20% of the vegetated land, and exhibiting the largest seasonal albedo variation of any ecosystem worldwide (ACIA, 2005; Chapin et al., 2000; DeLuca & Boisvenue, 2012). Fire is the dominant disturbance in boreal forests (Payette, 1992; Stocks et al., 2001), and with climate change, fires are expected to increase in frequency, size, and severity (Flannigan et al., 2009; Soja et al., 2007; Wotton et al., 2017; Young et al., 2017). These expected changes in boreal fire regimes could increase the amount of C combusted and change post-fire C accumulation by altering post-fire tree recruitment (Alexander & Mack, 2016; Alexander, Mack, Goetz, Loranty, et al., 2012; Alexander et al., 2018; Johnstone & Chapin, 2006; Johnstone et al., 2010; Thornley & Cannell, 2004; Turetsky et al., 2011; Walker et al., 2019). Additionally, larger burn areas will increase the proportion of land recovering from fire and can have long-lasting effects on the regional energy balance through vegetation-driven impacts on surface albedo, evapotranspiration, and ground insulation (Chambers & Chapin, 2003; Liu et al., 2005; Randerson et al., 2006; Rocha et al., 2012; Webb et al., 2021; Yoshikawa et al., 2002).

Eurasian larch (*Larix* spp.) forests occupy 20% of the boreal forest biome (Abaimov, 2010), and their annual burned area is an order of magnitude greater than that of any other vegetation type in the permafrost zone (Loranty et al., 2016). Recently, increasing temperatures and related atmospheric drivers (e.g., polar jet dynamics,

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vapor pressure deficit) have led to unprecedentedly large fire years in Siberia (Descals et al., 2022; Scholten et al., 2022; Talucci et al., 2022) and record-high fire emissions across Eurasia (Zheng et al., 2023). Understanding the effects of fire in Eurasian larch forests is therefore critical to projecting the effect of intensified fire regimes on global and regional climate and ecosystem processes. Yet relatively little is known about forest dynamics and fire regimes in this critical ecosystem (Mccarty et al., 2021; Veraverbeke et al., 2021).

In contrast to the mostly stand-replacing crown fires in boreal North America, fires in Eurasian larch forests are thought to be typically less severe (Rogers et al., 2015; Wirth, 2005). This paradigm is based on continental-scale satellite-based remote sensing analyses combined with a theoretical framework of how the traits of the regionally dominant tree species influence fire regimes (Rogers et al., 2015; Wirth, 2005). Many North American boreal species have flammable foliage and maintain lower branches, which act as a fuel ladder, and encourage crown fires (De Groot et al., 2013; Rogers et al., 2015; Wirth, 2005). In contrast, Eurasian larch species have traits associated with fire resistance, such as self-pruning of lower branches, high leaf moisture, and thick bark, which, in addition to low canopy closure, should promote low-intensity surface fires (De Groot et al., 2013; Rogers et al., 2015; Wirth, 2005).

There is considerable heterogeneity in fire characteristics across the larch ecosystem, however, with remote sensing and field observations documenting higher mortality in the northern reaches of the Siberian larch range (Krylov et al., 2014; Matveev & Usoltzev, 1996; Shuman et al., 2017; Tsvetkov, 2006a). Here, marginal growing conditions lead to smaller trees whose root systems are more susceptible to fire damage. In contrast, the southern Siberian larch range is characterized by larger diameter larch and pine trees that can withstand surface fires (Matveev & Usoltzev, 1996; Tsvetkov, 2006a).

Recent estimates indicate that C combustion in larch forests is similar to boreal North America (Veraverbeke et al., 2021), which challenges the conventional wisdom that fires in Siberian larch forests are less severe than in North America. There are few published field observations of fire severity in Siberian larch forests, however, and the methods and locations of existing studies are not well documented. Resolving the apparent disconnect between some previous work indicating a prevalence of low severity fires (De Groot et al., 2013; Rogers et al., 2015; Wirth, 2005) and others documenting predominantly high severity fires (Matveev & Usoltzev, 1996; Tsvetkov, 2006a; Veraverbeke et al., 2021) is critical to understanding future fire-driven C emissions and albedo trajectories.

Here we present an analysis of the effect of fire on C stocks and tree mortality at Arctic and subarctic Siberian larch sites. We collected field data in recently burned and adjacent unburned forest stands to estimate the impacts of wildfire on tree mortality, aboveground C stocks (fine and coarse woody debris, live shrubs, standing dead trees, and live trees) and belowground C stocks (soil organic layer). We combined our field-based estimates with a synthesis of published literature to better understand Siberian fire regimes and to compare fire impacts in Siberian larch forests to the more extensively studied North American boreal forests.

2. Materials and Methods

2.1. Overview

We evaluated wildfire impacts on tree mortality and C storage at 13 sites dominated by Siberian larch (*Larix cajanderi*) (Figure 1). We quantified live tree density and above- and belowground C stocks in burned and adjacent unburned stands. Aboveground C stocks included fine and coarse woody debris, live shrubs, standing dead trees, and live trees. Belowground C stocks included all components of the soil organic layer (SOL): brown moss, leaf litter within the brown moss, fibric material, coarse and fine roots, and organic soil. Green moss, along with any leaf litter within it, was excluded from analysis. To determine if fire severity at our field sites was representative of eastern and northeastern Siberian wildfires, we used Landsat satellite data to compare fire severity at our field sites to the surrounding region using the difference Normalized Burn Ratio (dNBR) (Eidenshink et al., 2007). Our field-based sampling and analyses are described below, and the Landsat analysis is described in the Supporting Information S1. All analyses were implemented in R (R Core Development Team, 2023) unless stated otherwise.

2.2. Study Regions

We studied larch forests in two regions of Siberia spanning subarctic and Arctic zones: near Yakutsk (subarctic; 62.0621°N, 129.7773°E) and near Cherskiy (Arctic; 68.7471°N, 161.3356°E) (Figure 1). Hereafter we refer to the

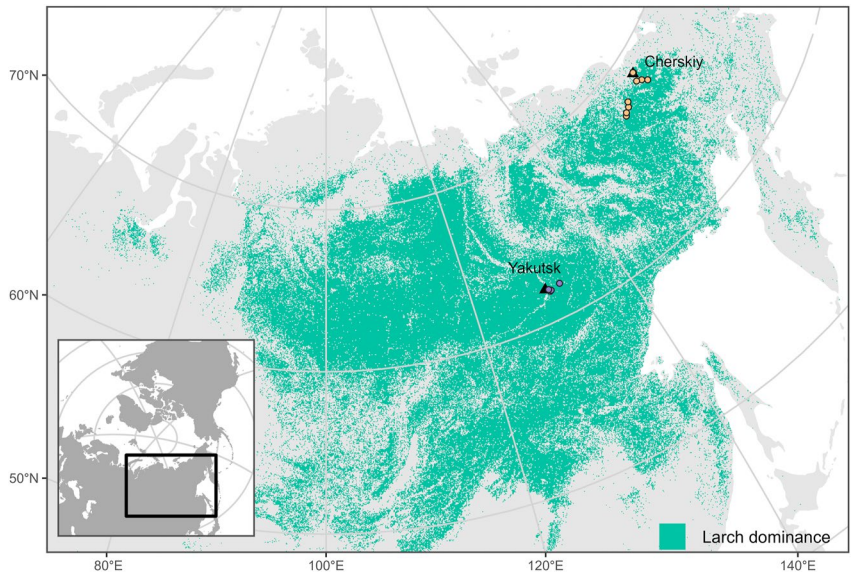

Figure 1. Location of Arctic (yellow) and subarctic (purple) study sites and nearby cities (Cherskiy and Yakutsk, black triangles).

Cherskiy and Yakutsk regions as the Arctic and subarctic study regions, respectively Larch (*L. cajanderi*) is the only tree species at our Arctic sites and the dominant tree species at our subarctic sites, where other tree species include *Betula platyphylla*, *Pinus sylvestris*, and *Populus tremula* (Litkina, 2002). In both regions, understory vegetation consists of deciduous shrubs (e.g., *Betula nana exilis* and *Salix* spp.), evergreen shrubs (e.g., *Vaccinium vitis-idaea*, *Empetrum nigrum*, and *Ledum decumbens*), forbs and grasses (e.g., *Carex appendiculata*, *Artemisia tilesii*, *Calamagrostis neglecta*), mosses (e.g., *Aulacomnium turgidum*, *Dicranum* spp., *Polytrichum* spp.), and lichens (e.g., *Cetraria cuculata*, *Cladonia rangiferina* (Paulson et al., 2021; Petrovsky & Koroleva, 1979).

2.3. Field Sampling

In 2018 and 2019, we sampled recently burned (3–36 years post-fire at the Arctic sites and 17 years at the subarctic) and adjacent unburned forests across 13 fire scars (9 Arctic and 4 subarctic). We identified fire scars using Landsat-derived fire perimeters (Berner et al., 2012), Google Earth Pro version 7.3, and field observations. From this data set, we chose fire scars to sample based on our ability to access them from a boat (Arctic study region) or road (both study regions). None of the fire scars re-burned between the identified fire year and the time of field sampling.

For each fire scar, we established one to three transects located at least 100 m apart. Each transect extended from the unburned forest surrounding the burn into the burned forest. On most transects, sampling plots were located in the unburned forest at 25 and 50 m from the burn edge and in the fire scar at 25, 50, 75, 100, 150, and 200 m from the burn edge. When necessary, we adjusted the locations of unburned plots to avoid ambiguous burn edges and riparian areas, or due to the size or shape of the accessible unburned area. Similarly, the burned portion of some transects was shorter than 200 m due to logistical constraints or irregularities in the shape of the fire scar. In total, we sampled 147 burned plots (122 Arctic and 25 subarctic) and 51 unburned plots (42 Arctic and 9 subarctic) (Table 1).

We sampled trees in circular plots, with radii ranging from 2 to 10 m depending on tree density. We tallied all live and dead trees originally rooted within each plot and we measured their diameter at breast height (1.37 m) or, for trees <1.37 m tall, at their base. We excluded from our analysis individuals that were judged to be post-fire recruits based on their size or age (estimated in the field for conifers by counting stem nodes). We noted whether trees were alive or dead, and if dead trees were standing or fallen. Fallen dead trees were identified based on the presence of a maintained tree structure, with a clear bole and canopy with at least some branches. For dead trees within the unburned plots, we visually estimated the percent canopy remaining. For dead trees within the burned plots, we recorded a crown combustion index modified from Walker et al. (2018), where each tree is ranked from

Table 1
Carbon Stocks and Fire Impacts

	Arctic									
	ANS	Alnus	BP	CN	FOC	FU	Gonzo	HR	Shark	Regional mean
Site characteristics										
Year of fire	2003	1984	1983	2001	2015	2015	2001	1990	2010	
Year sampled	2019	2019	2019	2018	2018	2018	2018	2019	2018	
Landscape position	Floodplain	Floodplain	Upland	Upland	Floodplain	Floodplain	Upland	Upland	Upland	
Latitude	67.86110097	68.10890976	68.32109625	68.78283243	67.76913989	67.65294962	67.52918101	68.75177017	67.99446666	
Longitude	161.6834865	161.3378388	160.6455729	161.4749875	156.3084611	155.4007094	154.9274558	161.5370772	156.9732333	
# of burned plots	18	6	12	18	12	12	18	12	14	
# of unburned plots	6	2	4	6	4	4	6	4	6	
Unburned forest structure										
Tree density (stems m^{-2})	0.34 ± 0.08	0.85 ± 0.80	0.16 ± 0.06	0.55 ± 0.19	0.61 ± 0.27	0.66 ± 0.37	1.28 ± 0.49	0.05 ± 0.01	0.23 ± 0.13	0.53 ± 0.13
Basal area (m^2 ha^{-1})	18.8 ± 4.1	9.7 ± 0.9	8.2 ± 2.7	10.9 ± 3.3	33.8 ± 8.2	17.9 ± 5.9	14.3 ± 3.4	2.1 ± 0.4	9.5 ± 2.1	13.9 ± 3.0
SOL depth (cm)	9.5 ± 0.8	7.5 ± 0.5	10.8 ± 2.1	6.0 ± 0.6	6.8 ± 0.2	9.5 ± 0.6	4.9 ± 0.3	11.0 ± 1.0	5.8 ± 0.9	8.0 ± 1.0
Belowground C pool (g C m^{-2})	3,114 ± 705	1,854 ± 161	2,472 ± 248	3,038 ± 625	4,643 ± 977	3,363 ± 515	1,450 ± 180	2,580 ± 544	4,416 ± 1,610	2,992 ± 354
Aboveground C pool (g C m^{-2})	3,493 ± 765	2,369 ± 44	1,616 ± 439	1,682 ± 515	5,229 ± 1,163	3,099 ± 769	3,164 ± 424	551 ± 68	2,432 ± 654	2,626 ± 447
Total C pool (g C m^{-2})	6,088 ± 871	4,222 ± 116	4,088 ± 477	4,721 ± 1,078	9,872 ± 750	6,462 ± 718	4,614 ± 465	2,486 ± 693	6,848 ± 2,110	5,489 ± 710
Percentage of C pool from aboveground	57.0 ± 10.5	56.2 ± 2.6	37.9 ± 7.2	33.0 ± 6.7	52.4 ± 10.8	46.4 ± 8.3	67.7 ± 3.6	38.6 ± 20.9	40.2 ± 5.2	48.0 ± 4.0
Burned forest structure										
Tree density (stems m^{-2})	0.02 ± 0.01	0.93 ± 0.21	0.00 ± 0.00	0.00 ± 0.00	0.10 ± 0.06	0.07 ± 0.03	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.12 ± 0.10
Basal area (m^2 ha^{-1})	1.2 ± 0.7	4.8 ± 0.9	0.0 ± 0.0	0.1 ± 0.1	5.1 ± 2.9	4.8 ± 1.9	0.0 ± 0.0	0.1 ± 0.1	0.0 ± 0.0	1.8 ± 0.8
SOL depth (cm)	8.7 ± 0.5	7.4 ± 0.4	9.4 ± 0.8	4.5 ± 0.3	7.8 ± 1.3	9.5 ± 0.7	3.9 ± 0.4	7.0 ± 0.2	4.1 ± 0.5	7.0 ± 1.0
Belowground C pool (g C m^{-2})	3,162 ± 272	1,246 ± 327	2,819 ± 248	2,379 ± 175	4,765 ± 1,068	4,554 ± 467	1,685 ± 151	2,149 ± 388	2,082 ± 382	2,760 ± 405
Aboveground C pool (g C m^{-2})	896 ± 128	4,397 ± 609	705 ± 79	629 ± 86	2,803 ± 596	2,153 ± 292	1,582 ± 166	155 ± 12	2,225 ± 252	1,727 ± 443
Total C pool (g C m^{-2})	3,531 ± 377	5,436 ± 722	3,524 ± 261	3,008 ± 169	7,568 ± 806	6,707 ± 589	3,266 ± 264	2,305 ± 387	4,159 ± 397	4,389 ± 597
Percentage of C pool from aboveground	35.8 ± 7.5	81.8 ± 5.0	20.8 ± 2.6	21.7 ± 3.0	40.3 ± 8.7	32.0 ± 3.5	48.3 ± 2.2	8.7 ± 1.2	56.8 ± 5.8	38.0 ± 7.0
Fire severity and C loss										
dNBR (severity)	187 ± 28 (low)	-	-	-	379 (high)	285 ± 24 (moderate)	220 ± 30 (low)	-	435 ± 27 (high)	301 ± 47
Crown severity index	1.8 ± 0.2	0.6 ± 0.2	2.6 ± 0.1	2.3 ± 0.1	1.4 ± 0.2	1.9 ± 0.2	2.5 ± 0.1	1.7 ± 0.2	1.8 ± 0.2	2.0 ± 0.0
Tree mortality (%)	94.3	−8.9	98.6	99.2	83.9	89.1	100	92.9	100	83.2 ± 11.7

Table 1
Continued

	Arctic									
	ANS	Alnus	BP	CN	FOC	FU	Gonzo	HR	Shark	Regional mean
Aboveground C loss (g C m^{-2})	2,597	−2,029	911	1,053	2,426	946	1,582	396	207	899 ± 457
Percentage of aboveground C pool lost	74.3	−85.6	56.4	62.6	46.4	30.5	50	71.9	8.5	35.0 ± 16.6
Belowground C loss (g C m^{-2})	−48	607	−347	659	−122	−1,191	−235	430	2,334	232 ± 324
Percentage of belowground C pool lost	−1.5	32.7	−14	21.7	−2.6	−35.4	−16.2	16.7	52.9	6.0 ± 9.1
Total C lost (g C m^{-2})	2,549	−1,422	564	1,712	2,304	−245	1,347	826	2,541	1,131 ± 450
Percentage of total C pool lost	41.9	−33.7	13.8	36.3	23.3	−3.8	29.2	33.2	37.1	19.7 ± 8.2
Percentage of C loss from belowground	−1.9	−42.7	−61.5	38.5	−5.3	486.1	−17.4	52.1	91.9	60.0 ± 55.6

	Subarctic				
	FRK	Korova	Maya	ROB	Regional mean
Site characteristics					
Year of fire	2002	2002	2002	2002	
Year sampled	2019	2019	2019	2019	
Landscape position	Upland	Upland	Upland	Upland	
Latitude	61.820498	61.997064	61.955284	61.873475	
Longitude	130.23569	131.82096	130.16887	130.47849	
# of burned plots	9	3	4	9	
# of unburned plots	4	1	1	3	
Unburned forest structure					
Tree density (stems m^{-2})	0.61 ± 0.20	0.88	0.47	1.60 ± 0.45	0.89 ± 0.25
Basal area (m^2 ha^{-1})	31.9 ± 7.3	16.8	26.3	24.2 ± 3.3	24.8 ± 3.1
SOL depth (cm)	4.3 ± 2.2	7	1.4	7.2 ± 1.8	5.0 ± 1.0
Belowground C pool (g C m^{-2})	1,187 ± 330	2,403	773	2,613 ± 772	1,744 ± 451
Aboveground C pool (g C m^{-2})	7,273 ± 2,441	3,481	5,356	4,108 ± 487	5,054 ± 836
Total C pool (g C m^{-2})	8,460 ± 2,120	5,884	6,128	6,721 ± 456	6,798 ± 581
Percentage of C pool from aboveground	81.5 ± 6.2	59.2	87.4	62.1 ± 9.5	73.0 ± 7.0
Burned forest structure					
Tree density (stems m^{-2})	0.26 ± 0.03	0.05 ± 0.01	0.15 ± 0.05	0.24 ± 0.07	0.17 ± 0.05
Basal area (m^2 ha^{-1})	2.7 ± 0.6	0.3 ± 0.1	1.8 ± 0.6	4.6 ± 1.4	2.4 ± 0.9

Table 1
Continued

	Subarctic				
	FRK	Korova	Maya	ROB	Regional mean
SOL depth (cm)	1.3 ± 0.2	0.7 ± 0.2	2.2 ± 0.4	2.5 ± 0.7	2.0 ± 0.0
Belowground C pool (g C m^{-2})	980 ± 154	822 ± 347	1,250 ± 210	1,956 ± 452	1,252 ± 251
Aboveground C pool (g C m^{-2})	2,491 ± 185	1,234 ± 355	2,618 ± 442	2,416 ± 402	2,190 ± 321
Total C pool (g C m^{-2})	3,471 ± 183	2,055 ± 271	3,868 ± 354	4,372 ± 682	3,442 ± 498
Percentage of C pool from aboveground	71.8 ± 4.0	60.0 ± 14.2	66.3 ± 7.5	57.2 ± 5.3	64.0 ± 3.0
Fire severity and C loss					
dNBR (severity)	446 ± 40 (high)	236 ± 40 (moderate)	421 ± 50 (high)	402 ± 57 (high)	376 ± 48
Crown severity index	1.7 ± 0.1	2.7 ± 0.1	2.0 ± 0.3	2.0 ± 0.2	2.0 ± 0.0
Tree mortality (%)	57.1	94.7	67.6	84.8	76.0 ± 8.4
Aboveground C loss (g C m^{-2})	4,782	2,247	2,737	1,692	2,864 ± 674
Percentage of aboveground C pool lost	65.8	64.6	51.1	41.2	55.7 ± 5.9
Belowground C lost (g C m^{-2})	207	1,581	−477	657	492 ± 431
Percentage of belowground C pool lost	17.4	65.8	−61.7	25.1	11.6 ± 26.7
Total C loss (g C m^{-2})	4,989	3,828	2,260	2,349	3,356 ± 652
Percentage of total C pool lost	59	65.1	36.9	35	49.0 ± 7.6
Percentage of C loss from belowground	4.1	41.3	−21.1	28	13.1 ± 13.7

Note. Belowground C refers to the soil organic layer (see Methods). Aboveground C is the sum of C in FWD, CWD, live shrubs, standing dead trees, and live trees. For each site, we report means and standard errors across plots within burned and unburned stands. For each region, we report means and standard errors across site-level means. Sites burned before 2001 do not have dNBR values due to sparse Landsat coverage before 2000. Tree mortality is based on comparisons of burned and unburned stands.

0 to 3 according to burn severity: 0 = unburned; 1 = low severity (fine branches intact); 2 = moderate severity (majority of fine branches combusted); 3 = high severity (most of the canopy combusted). We measured the basal diameter of tall shrubs (*Salix*, *Alnus*, and *Betula* spp.) along a 5-m long belt transect within each plot. The standard width of the shrub transect was 1 m but was reduced to 0.5 m when shrub density appeared high.

We measured woody debris according to Brown (1974) as follows: we counted the number of times a 20 m transect through the center of each plot was intercepted by class I fine woody debris (FWD; 0.0–0.49 cm diameter) and class II FWD (0.5–0.99 cm) along the first 5 m; class III FWD (1.0–2.99 cm) along the first 10 m; and classes IV FWD (3.0–4.99 cm) and V FWD (5.0–6.99 cm) and coarse woody debris (CWD; >7 cm diameter) along the entire 20 m length.

We measured SOL depth in four locations at each plot: 2 m from plot center in each cardinal direction. We harvested one of these SOL profiles (~4 cm × 4 cm × SOL depth) from each plot for lab processing (see Supporting Information S1 for SOL processing methods). Despite the small size of the SOL samples, standard errors of belowground C among plots within sites were always substantially smaller than the stand-level mean belowground C (Table 1).

2.4. Tree Mortality Estimates

We estimated fire-induced percent tree mortality (%) at each of the 13 sites as $100 \times (L_{pre}–L_{post})/L_{pre}$, where L_{pre} and L_{post} are live tree densities before and after the fire, respectively. At each site, we estimated L_{post} as the mean density of live trees in the burned plots. Because we could not directly estimate L_{pre}, we considered two different indirect estimates. First, we estimated L_{pre} as the live tree density of the adjacent unburned forest; we refer to this as the "unburned" method. Second, we estimated L_{pre} as the reconstructed pre-fire live tree density of the burned plots; we refer to this as the "pre-burn" method. We reconstructed L_{pre} by multiplying the total tree density (including alive and standing and fallen dead trees, but excluding post-fire recruits) in the burned plots by a correction factor. The correction factor was estimated at each site as the live fraction of total pre-fire trees (live/total), averaged across the unburned plots. We aggregated the site-level live fractions to regional means (Arctic sites: 0.78; subarctic sites: 0.91) because ANOVA indicated that the live fractions varied by region ($p < 0.02$) but not by site ($p > 0.2$).

2.5. Carbon Stock Estimates

We calculated aboveground biomass stocks using published allometric equations for trees and shrubs (Alexander et al., 2012a, 2012b; Berner et al., 2015; Delcourt & Veraverbeke, 2022; Mäkelä & Vanninen, 1998), multipliers for fine woody debris (Delcourt & Veraverbeke, 2022; Nalder et al., 1997), and decay class density values for coarse woody debris (Ter-Mikaelian et al., 2008). For alive and standing dead trees where the canopy was partially or wholly missing, we subtracted a portion of canopy biomass from the total tree biomass according to field observations of percent canopy missing or burn severity. Biomass values were converted to C based on component-specific C densities. Detailed above- and belowground C stock assessment methods, including pre-burn tree C reconstructions, are provided in the Supporting Information S1.

At each site, we averaged the plot-level above- and belowground C stock estimates within the burned and unburned stands to get stand-level values. We estimated above- and belowground C loss as the difference between the unburned and burned stand values; percent C loss was calculated relative to the unburned C stock at each site.

3. Results and Discussion

3.1. Tree Mortality

We found high fire-induced tree mortality (i.e., reductions in live tree density) in both Siberian larch study regions. Mean ± standard error stand-level estimates across sites were 83 ± 12% and 76 ± 8% for the Arctic and subarctic regions, respectively, when comparing burned to unburned, and 81 ± 11% and 72 ± 7% when comparing burned to pre-burn reconstructions (Figure 2). Our sites represented a range of fire severities, as measured by field-based observations of crown severity and the satellite-based dNBR metric, with nearly half classified as low and moderate burns when compared to the dNBR of all fires in eastern and northeastern Siberia

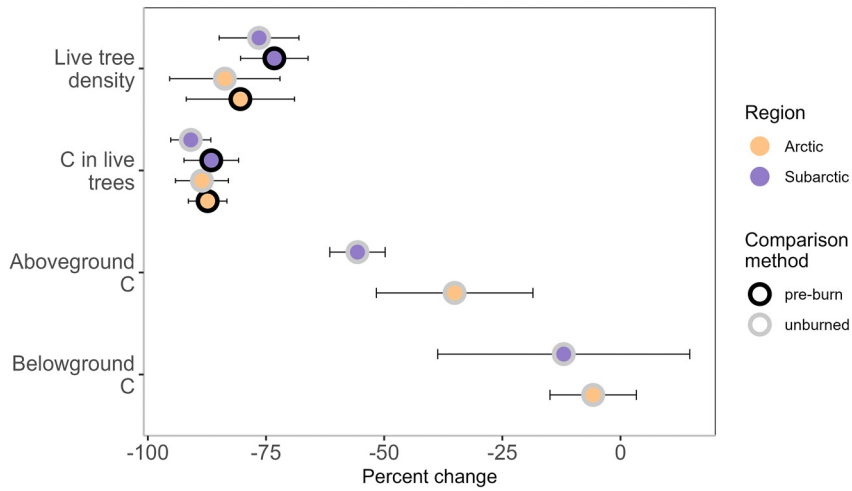

Figure 2. Percent change in tree density (i.e., inferred tree mortality) and C stocks between burned/unburned plots (gray circles) and burned/pre-burn reconstructions (black circles). Aboveground C is the sum of C in all alive and dead aboveground vegetation. Belowground C is C in the soil organic layer (see Methods). Error bars are ±1 standard error.

(Table 1). Thus, tree mortality was high at our sites even though the burns we sampled do not appear to be unusually severe, suggesting that stand-replacing fires are common in eastern and northeastern Siberia.

Our field-based estimates of fire-induced tree mortality were higher than remote sensing-based estimates of mortality across all northeastern Eurasian boreal forests (~62%), Eurasian larch forests (~50%) (Rogers et al., 2015), and the eastern and northeastern Siberian taiga ecoregions (67%) (Shuman et al., 2017). These results support previous analyses that show that fires are more lethal in larch forests than in other Siberian forest types (Bartalev & Stytsenko, 2021; Lupyan et al., 2022), and that fire-induced tree mortality is higher in northern regions of Siberia (Krylov et al., 2014), including both our subarctic and Arctic sites. Our results indicate that fires in Siberian larch forests result in high mortality, despite larch having traits associated with fire resistance (e.g., high leaf moisture and self-pruning of lower branches) and low canopy closure (Kharuk et al., 2011; Rogers et al., 2015; Sofronov & Volokitina, 2010; Wirth, 2005).

Our finding of highly lethal, stand replacing fires in Siberian larch forests is consistent with some previous observations that have not been widely cited in the international literature (Tsvetkov, 2006a, 2006b). Siberian larch trees growing on continuous permafrost have shallow root systems, which often do not exceed the depth of the SOL (Hewitt et al., 2022; Kajimoto, 2010; Kropp et al., 2019). Surface fires can therefore cause extensive root damage that ultimately results in tree mortality (Fang et al., 2018; Tsvetkov, 2006a). At the same time, the substantial organic layer fuel load characteristic of these forests promotes slow-burning fires with long residence times that can damage trees at their base, even if the flames do not reach the upper canopy (Tsvetkov, 2006a, 2006b).

3.2. Carbon Loss

Our measurements were taken between 3- and 36-year post-fire (Table 1), and therefore include post-fire vegetation regrowth and re-accumulation of the SOL in the burned plots, as well as vegetation growth and SOL accumulation in the unburned plots. Given that C accumulation occurs more rapidly in early successional forests (i.e., our burned plots) than in late-stage forests (i.e., our unburned plots) (Gao et al., 2018), estimating C loss by comparing burned and unburned plots could result in an underestimate of C loss, and the magnitude of the underestimate should increase with time-since-fire. Nonetheless, we found that time-since-fire did not significantly impact above- ($p = 0.2$) or belowground ($p = 0.9$) C loss estimates. These non-significant relationships likely reflect several factors, including the limited number of study sites, uncertainty in site-level estimates, and the small magnitude of C accumulation rates relative to the large variance among sites in burn severity and C loss.

We estimate that fire resulted in an aboveground C loss of 899 ± 457 (mean ± standard error) and 2,864 ± 674 g C m^{-2} at our Arctic and subarctic sites, respectively (Table 1), which is considerably higher than aboveground C losses reported for fires in northern boreal North America (185–565 g C m^{-2}) (Walker

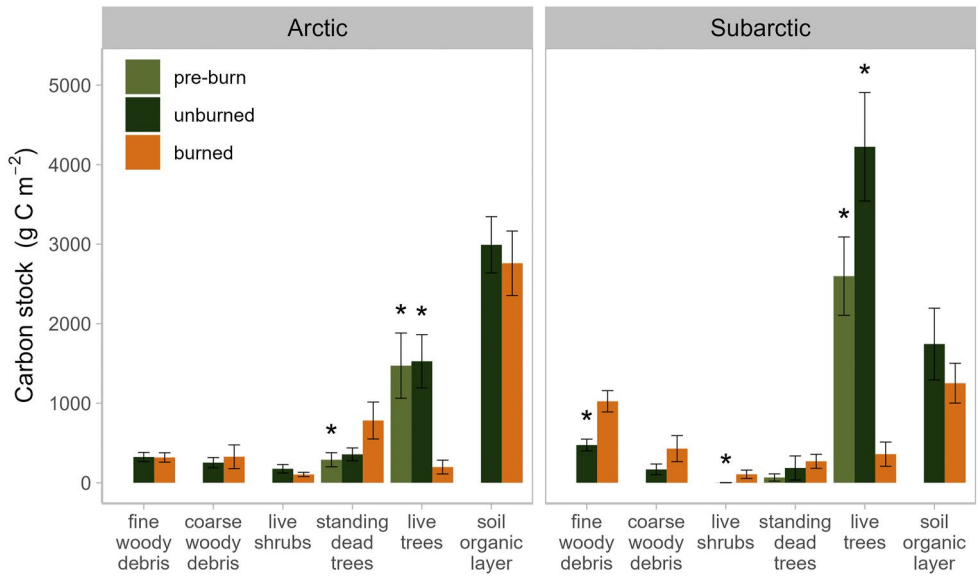

Figure 3. Belowground (soil organic layer) and aboveground C stocks. Error bars denote one standard error of the mean across sites. Stars indicate significant ($p < 0.05$) differences between burned plots and unburned measurements or pre-burn reconstructions based on Bonferroni-corrected paired *t*-tests.

et al., 2020). Most of the aboveground loss in our study reflects a decrease in live tree C that was only partially offset by an increase in standing dead tree C at our Arctic sites and downed woody debris C at our subarctic sites (Figure 3). As a percentage of unburned C stocks, fire caused a 35 ± 17% decrease in aboveground C stocks at our Arctic sites and 56 ± 6% at our subarctic sites (Figure 2). These estimates are at the high end of or greater than the northern boreal North American range (21%–39%) (Walker et al., 2020), which suggests more severe aboveground disturbance severity in Eurasian larch forests compared with northern North America. This finding of comparatively high C loss from aboveground sources in Siberian larch forests is similar to an independent study near our subarctic field sites (Veraverbeke et al., 2021), but differs from remote sensing-based estimates suggesting less severe fire-driven vegetation destruction in northeastern Eurasia than in North America (Rogers et al., 2015; Van Der Werf et al., 2017).

The discrepancy between field and remote sensing-based estimates of vegetation destruction could be due to limitations of field-based research that only allow us to sample a small proportion of all fires, whereas remote sensing analyses can be employed at the biome-scale. Alternatively, differences between field and remote sensing-based estimates could reflect the poor performance of standard remote sensing products in larch ecosystems (Bendavid et al., 2023; Loranty et al., 2018; Montesano et al., 2016), particularly in regions with low canopy cover (Montesano et al., 2009). Accordingly, we found no relationship between the site mean dNBR and the percentage of the aboveground C pool combusted ($p = 0.2$). It is difficult to know if the poor correspondence between dNBR and our field data reflects limitations of dNBR in our study regions or the appropriateness of our field data for evaluating the satellite-derived metric. A field study that was designed to evaluate satellite-based dNBR in two Siberian larch fire scars, for example, found relatively good agreement between dNBR and field-based estimates of burn severity (Delcourt et al., 2021).

Belowground C loss from fire was 232 ± 324 (mean ± standard error) and 492 ± 431 g C m^{-2} at our Arctic and subarctic sites, respectively (Table 1), which represents a 6 ± 9% (Arctic sites) and 12 ± 27% (subarctic sites) decrease compared to belowground C stocks in the unburned plots (Figure 2). In both relative and absolute terms, these estimates are considerably smaller than belowground combustion estimates reported for northern North America (2,553–3,100 g C m^{-2}; 20%–51% of pre-fire C) (Walker et al., 2020). Thus, while 81%–94% of total C lost to fire is from belowground pools in northern North America (Walker et al., 2020), belowground C contributed a smaller percentage of total C lost at our sites (60 ± 56% and 13 ± 14% for Arctic and subarctic sites, respectively) (Table 1). A separate study conducted near our subarctic sites also found that the belowground combustion fraction in larch forests is lower than in North America, although their estimate (~75%) was higher than ours (Veraverbeke et al., 2021), highlighting the spatial heterogeneity of burn severity.

Total C loss from fire, including above- and belowground C, was 1,131 ± 450 and 3,356 ± 652 g C m^{-2} at our Arctic and subarctic sites, respectively (Table 1). Our subarctic loss estimate is similar to previous estimates of ~3,000 g C m^{-2} in *L. gmelinii* forests (Tsvetkov, 2006b) and 3,360 ± 930 g C m^{-2} in *L. cajanderi* forests near our subarctic sites (Veraverbeke et al., 2021). Our estimates of total C loss from larch forests were ~65% lower (Arctic sites) and ~2% higher (subarctic sites) than the average estimates from northern North American boreal forests (3,118–3,514 g C m^{-2}) (Walker et al., 2020). These results challenge the idea that Eurasian boreal forests uniformly combust less C per unit area than North American forests (Van Der Werf et al., 2017; Wirth, 2005), and highlight the importance of differentiating loss by latitude and forest type (i.e., C combustion is higher in larch forests than in other Eurasian boreal forest types (Veraverbeke et al., 2021)).

4. Conclusions

We provide evidence that fires in the widespread larch-dominated forests of Siberia result in high tree mortality, despite species traits and stand structure hypothesized to promote low severity fires (Rogers et al., 2015; Wirth, 2005). Given that larch trees tend to form open canopy forests that do not promote crown fires (Kharuk et al., 2011; Sofronov & Volokitina, 2010), our results suggest that fires in this region are high severity surface fires. Additionally, we show that total C loss following fire in Siberian larch forests can be as high as in northern North America, but that the above- and belowground components of loss are partitioned differently, with a higher fraction coming from aboveground pools in larch forests. As fires become more frequent and severe with climate change, high fire-induced tree mortality and aboveground C losses in larch dominated forests of Siberia may have broad-reaching impacts on global climate through high C emissions and tree-cover mediated albedo changes.

Data Availability Statement

The data used in this study are freely and publicly available through the Arctic Data Center. Field measurements: Alexander et al., 2020; Carbon stock estimates: Webb et al., 2023. Code used to derive dNBR, perform analyses, and create figures is available through at E. Webb 2023. Larch extent (Figure 1) was based on the ESA CCI land cover 2015 product (Defourny, 2017).

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