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Influence of Mixed Conifer Forest Thinning and Prescribed Fire on Soil Temperature and Moisture Dynamics in Proximity to Forest Logs: A Case Study in New Mexico, USA

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Abstract: Forest management activities often include fuels reduction through mechanized thinning followed by prescribed fire to remove slash. Management prescriptions may include the retention of logs for wildlife habitat and microsites for enhanced tree regeneration. We examined aboveground microclimate and belowground soil temperature and volumetric water content (VWC) dynamics beneath and adjacent to logs at 10, 20, and 30 cm depths in a mixed conifer forest. We assessed the soil variables over 7 years during pre-treatment, post-thinning, and post-fire using a Before–After/Control–Impact experimental design. We found that thinning and burning caused large increases in solar radiation and mean and maximum wind speeds, but only small changes in air temperature and humidity. The treatments increased the soil temperatures beneath the logs by up to 2.7 °C during spring, summer, and fall; the soil VWC increased from 0.05 to 0.08 m^3/m^3 year-round at 20 and 30 cm depths. Microsites 1–2 m away from the logs also showed soil temperature increases of up to 3.6 °C in spring, summer, and fall, while the measurements of the soil VWC produced variable results (moderate increases and decreases). The increased VWC in late winter/spring likely resulted from reduced plant transpiration and greater snow amounts reaching the ground without being intercepted by the forest canopy. Log retention on thinned and burned sites provided microsites with increased soil temperature and moisture in the top 30 cm, which can enhance soil ecosystem processes and provide refugia for invertebrate and vertebrate wildlife.

Keywords: prescribed fire; forest restoration; amphibian conservation; climate change; forest logging

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1. Introduction

Forest restoration and management activities to prevent or reduce the impacts of high-severity wildfires to ecosystems and biodiversity are becoming more critical with the threat of climate warming, land use change, and increased wildfire frequency [1–5]. Climate change is leading to higher temperatures, lower humidity, and greater vapor pressure deficits [6]. The result is larger areas of wildland burns due to lower fuel moisture levels [7,8]. In addition, forest regeneration is being reduced under these conditions as a result of greater moisture deficits that inhibit post-fire seedling re-establishment, growth, and eventual survival to reproductive age [9]. In the extreme, such as in the southern Rocky Mountains, extensive high-severity stand-replacement fires are resulting in the conversion of forests to montane meadows and shrublands, with minimal or no conifer re-establishment [10–12].

To reduce the risk of high-severity wildfire, forest restoration actions in stands with high fuel loads and increased densities of trees often involve thinning operations followed by planned broadcast burns to remove slash and expose mineral soils to colonization by herbaceous grasses and forbs. As part of this strategy, some existing logs or freshly cut tree boles may be intentionally left behind to provide microsites for tree seedling recruitment (voluntary or re-planted) [13] and wildlife habitat [14]. Logs and other coarse woody debris have long been recognized as important forest ecosystem components, providing heterogeneity to soil environments [15,16]. Soil micro-climatological conditions (temperature and moisture) following forest management and fire drive rates of soil respiration and organic matter decomposition [17–25], diversity of fungi [26,27], abundance, metabolic rate, and reproduction of soil invertebrates [28–31], and environmental refugia for vertebrate wildlife [32–36].

Previous studies on soil temperature and moisture changes with forest thinning and burning have produced a range of results. In natural (untreated) forests, micro-environmental differences exhibit soil temperatures at a 5 cm depth that are approximately 1 °C warmer in gaps vs. understory and soil moisture 2–6% higher in shaded understory sites [18]. For thinned forests, thinning operations have resulted in soil temperature increases (compared to no-thinning controls) of 2.5–15 °C, while the soil moisture responses vary from no differences between treatments to somewhat drier (decrease of 10%) or wetter (increase of 15%) in post-thinned environments [37–39]. In forests that have been thinned and then burned, the soil temperatures have increased by 0–8 °C, and the soil moisture has shown increases between 2% and 40% [38,40].

Clear-cutting forests leads to soil temperature increases of 0.8–8 °C, but the effects on soil moisture range from decreases by up to 14%, to no effect, and to increases by up to 30% [25,41–45]. Within clear-cuts, the presence of slash reduces the soil temperatures by 0.5 to 2.5 °C and either has no effect on soil moisture or increases soil moisture by 3–5% and can reduce the rate of soil moisture depletion from spring to summer [46–49]. Clear-cuts that are subsequently treated with a slash burning also experience increased soil temperatures in the range of 1–6 °C, while soil moisture can be either decreased by 30% or increased by 12–20% compared to control sites [19,50,51].

Salvage logging following fires or wind-throw events have shown either no soil temperatures responses or 1–10 °C soil temperature increases compared to those recorded in areas without salvage operations. Soil moisture can be higher by <1% to 7% on both non-salvaged sites and salvaged sites if branches are left on site after logging [13,52–55].

The high variability across studies of the soil temperature and moisture responses to forest management operations can be attributed to differences in forest type (e.g., pine forest versus mixed conifer versus boreal spruce forests), regional climate (seasonally warm and dry versus consistently cool and moist), soil types, slope and aspect, depth of litter layers, presence and amounts of coarse woody debris, and amount of remaining forest canopy that can intercept/re-direct precipitation (especially snowfall). Large increases in soil temperature (~15 °C) can create inhospitable conditions for tree regeneration and wildlife; hence, site-specific assessments of forest management practices are necessary to evaluate their potential impacts to the resident flora and fauna.

In this case study, our goal was to understand the impacts of forest thinning and subsequent broadcast slash burning on the soil micro-environment below and adjacent to pre-existing logs left in the project area. We anticipated that removing some of the forest canopy would alter air temperatures, humidity, wind, and solar radiation, which in turn would affect the soil temperatures and moisture levels in the vicinity of the logs. We endeavored to evaluate these micro-environmental changes at different soil depths via a 7-year "Before–After/Control–Impact" (BACI) experimental design, testing the null hypothesis that forest thinning and burning would have no effect on any environmental variable. We anticipated that the soil temperatures would increase (due to greater solar radiation exposure) and that soil moisture would decrease (due to greater evaporation and transpiration of the colonizing herbaceous vegetation), though the a priori magnitudes of these changes would be difficult to predict, given the large range of responses observed in previous studies.

2. Materials and Methods

We conducted the study in the Jemez Mountains of northern New Mexico, within the National Park Service's Valles Caldera National Preserve (Preserve) (Figure 1). The Preserve is a volcanic caldera with numerous internal rhyolite domes [56] and comprises 36,000 ha of high-elevation (2380–3430 m) coniferous forests and meadows. A former sheep and cattle ranch that had been subjected to extensive forest clear-cutting in the 20th century, the Preserve was acquired by the Federal Government in 2000; extensive landscape restoration programs for forests, rangelands, and wetlands have been underway since 2003.

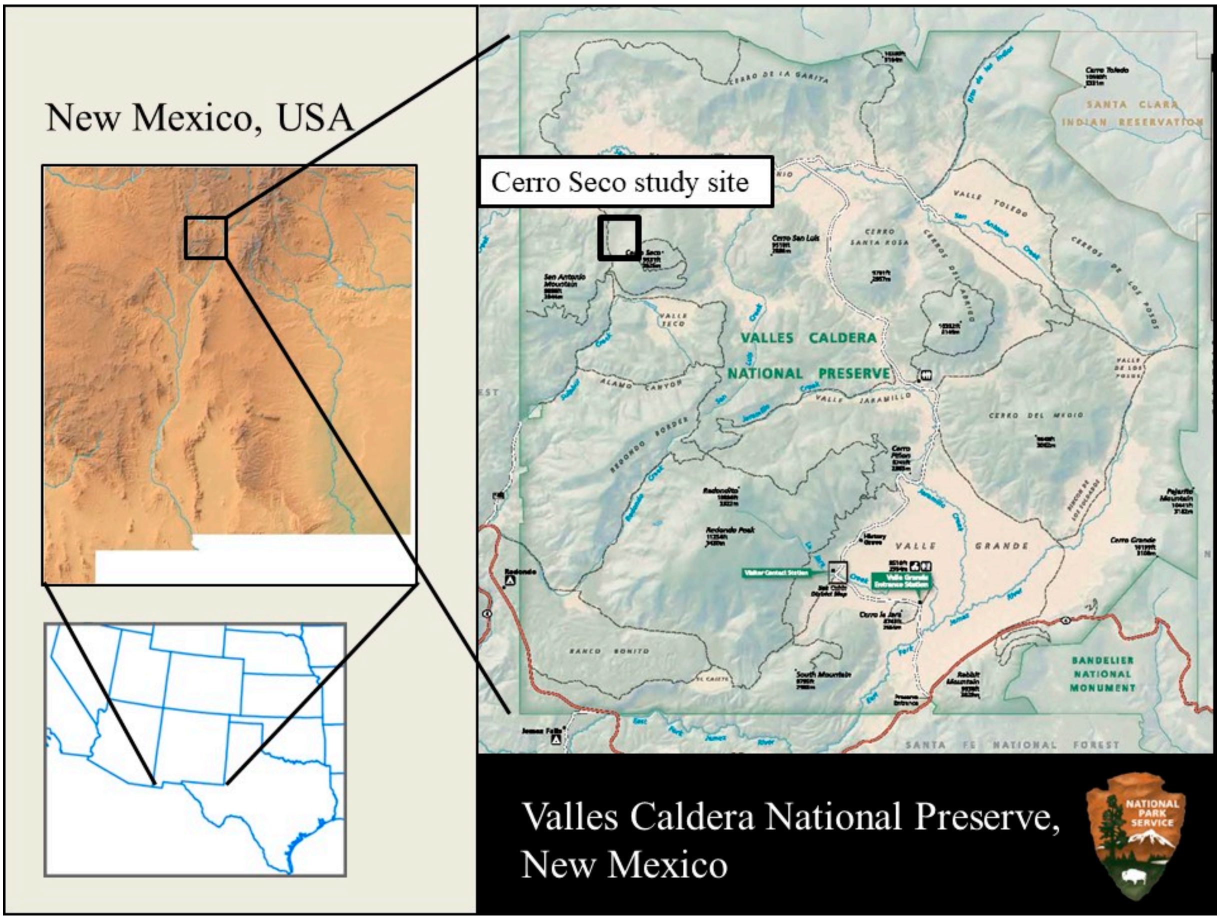

Figure 1. Map of the Cerro Seco study site in Valles Caldera National Preserve, NM.

The study site was on the western slope of Cerro Seco (Figure 1; Lat. 35°57′11″, Long. 106°35′30″, elevation 2665 m, and a slope of 18%). The site's mixed conifer forest consisted of ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), Colorado blue spruce (*Picea pungens* Engelm.), and Engelmann spruce (*Picea engelmannii* Parry ex Engelm.). Fire suppression/prevention efforts throughout the 20th century allowed the tree density on the site to exceed 1100 trees/ha at the time of the study; Cerro Seco had been selectively logged during 1980–1991 [57]. The soils are alluvium from ~800,000-year-old pyroclasts, classified as Vitrandic Agriudolls and Hapludalfs [58]; these soils, of volcanic rhyolite parent materials, comprise the majority of soil types in Cerro Seco and nearby volcanic domes of comparable age. The mean annual precipitation is 594 mm, with ~40% of it falling as rain during the summer monsoons, and ~50% coming as winter snows; the mean monthly temperatures range from −4.8 °C in January to 15.4 °C in July (Valle Grande Remote Access Weather Station (RAWS) data, 2003–2022 [59]).

We designed the study to assess the influence of forest thinning operations and slash burning on soil temperature and moisture beneath and adjacent to logs using a Before–After/Control–Impact (BACI) experimental design. We instrumented four logs, two of which served as controls (North and South Control Logs), while two were subjected to thinning and burning treatments (North and South Treatment Logs). The logs were selected after canvasing the thinning unit and adjacent control forest and appeared to have been cut during the logging period 25 years before. The logs had lost most of their branches and bark, but their sapwood was still relatively solid. The diameters at midpoint (instrument locations) ranged from 37 to 52 cm. The logs were oriented east–west or northeast–southwest. Micro-environmental measurements were recorded at each log with on-site weather stations and belowground sensors. Sensors for soil temperature and moisture (volumetric water content, VWC) were deployed at three soil depths (10, 20, and 30 cm) in five columns: one column directly beneath a log, and four columns at 1 and 2 m distances north and south of the logs (Figure 2); due to shading effects of the logs and accumulation of winter snows, the north sides of the logs were predicted to be cooler and wetter than the south sides, at least in sites closer to the log. Weather stations were installed next to each log; the weather station and soil sensor data loggers were enclosed in steel cattle panels to prevent elk from damaging the instruments.

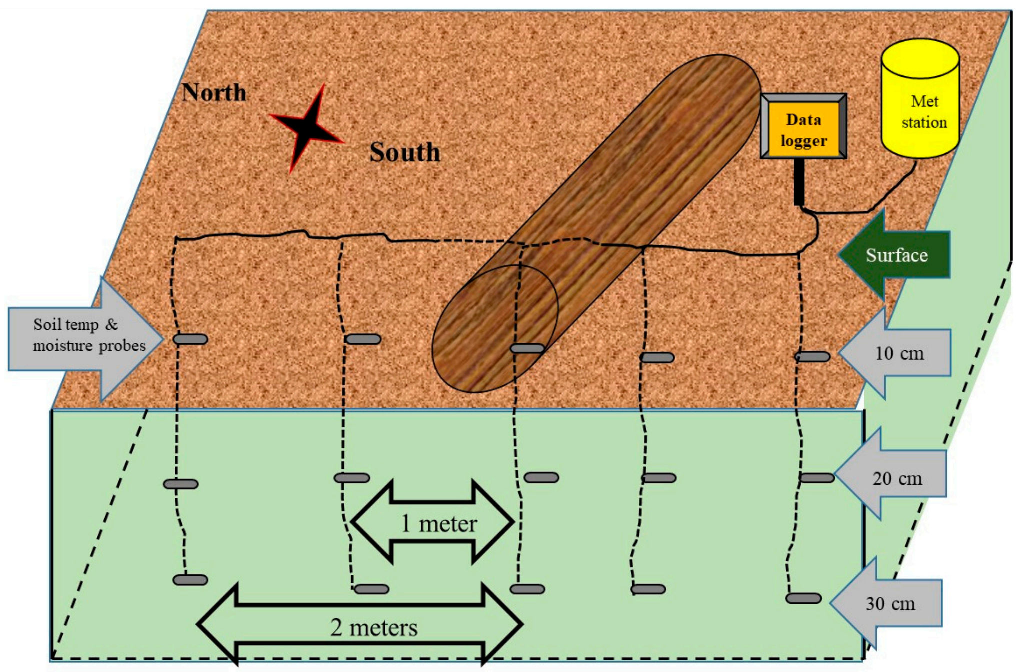

Figure 2. Schematic layout of the soil sensors relative to a log (not to scale).

We measured forest stand characteristics at each site using NPS Preserve standard protocols [60], including tree species composition, tree density, percentage canopy cover, and litter depth. The sample plots were 20 × 25 m, centered on each study log. The measurements were taken annually during summer/fall periods.

The weather stations included sensors for air temperature and relative humidity (Decagon Model VP-3; Meter Group, Pullman, WA, USA), wind speed and direction (Davis cup anemometer, 2 m high; Davis Instruments Corporation, Hayward, CA, USA), photosynthetically active radiation (PAR) (QSO_S PAR Photon Flux Sensor, 170 cm high; ICT International, Armidale, NSW, Australia), and precipitation (tipping-bucket ECRN-50; Meter Group), with a Decagon EM50 datalogger (Meter Group). The soil probes were Decagon 5TM sensors (Meter Group) for temperature and volumetric water content (VWC).

The instruments were deployed in September 2015 and became operational on 1 October 2015; the study ran for 7 water years, ending on 30 September 2022. The instruments measured all variables each minute, and the data loggers recorded 10 min average values (except for precipitation, for which the totals were recorded). The 10 min data values were averaged to produce mean daily values for graphical presentation, and the daily values were averaged for mean weekly values for statistical analyses. The precipitation collectors could not measure snowfall; so, winter precipitation data (November–March) were taken by a nearby Ameriflux ET Flux Tower [61] (4.5 km south of the study site and at a similar elevation of 2740 m; see https://ameriflux.lbl.gov/sites/siteinfo/US-Vcs#data-citation, accessed 31 January 2023).

The study included three periods of instrument measurements: (1) a two-year pre-treatment period (from October 2015 to December 2017); (2) a forest thinning treatment in December 2017, followed by two years of post-thinning measurements (December 2017–October 2019); and (3) a planned managed broadcast burn on 17–18 October 2019, followed by three years of post-fire measurements. We separated the data into seasonal periods to assess differences in warming versus cooling trends: Spring = April–May; Summer = June–August; Fall = September–October; Winter = November–March.

We used a $2 \times 3 \times 4 \times 5$ Repeated-Measures Analysis of Variance with 4-way interactions to test for differences in the mean weekly values among the treatments (control vs. thinning + burning), periods (pre-treatment, post-thinning, post-burning), seasons (spring, summer, fall, winter), and locations (beneath a log, 1 m north, 2 m north, 1 m south, 2 m south). The treatments and locations were analyzed as between-subject factors, and the periods and seasons were analyzed as within-subject factors. We identified significant differences among the means using the Least Significant Difference (LSD) means comparison tests. The absolute difference values for reporting were corrected for a priori microsite differences between control and treatment sites using Z-scores, in which the average values during the pre-treatment period (2015–2017) of the two control sites were subtracted from the average values of the treatment sites. Statistical analyses were conducted using the software Statistix© 10 (Analytical Software, Tallahassee, FL, USA). We defined statistical significance as $p < 0.05$.

3. Results

3.1. Forest Treatments

The site was mechanically thinned by a contractor in December 2017, using a feller-buncher and skidder; the log decks were removed via trucks during winter–spring in 2018 (Figure 3A). The thinning prescription was to reduce the canopy closure from 80–85% to 45% over the total unit area; the thinning was conducted also to create open gaps of meadow to function as fire breaks and increase the habitat heterogeneity. Spruce were to be cut, and Douglas-fir and ponderosa pine were to be left standing; patches of spruce were to be left as "islands" of forest in areas containing few Douglas-fir and pines. Prior to forest thinning, the treatment sites' forest stands had tree densities of 1319 trees/ha (North site) and 1539 trees/ha (South site), with a canopy cover of 80% and 85%, respectively, and a mean forest litter depth of 11.6 cm; the control sites had 1139 and 1519 trees/ha, with 79% and 85% canopy cover and a mean litter depth of 9.4 cm. [Note: the trees included canopy trees with a >15 cm diameter at breast height (dbh) and saplings (2.5–15 cm dbh), but not seedlings (<2.5 cm dbh)]. The thinning treatment reduced the tree densities to 40 trees/ha (North) and 360 trees/ha (South), with 11% and 40% canopy cover, respectively, and a mean litter depth of 6.5 cm.

NPS fire crews conducted a broadcast slash burn on 17–18 October 2019 (Figure 3B), which further reduced the tree density via burning of some saplings; wind-throw contributed to additional losses. The tree density on the south site was reduced to 300 trees/ha, while on the North site, only two dead saplings were left standing by the log as wind-throw had toppled the nearby spruce trees left after thinning; this site now more closely represented a forest gap as defined in the thinning prescription (Figure 4). The fire also removed much of the slash and forest litter and exposed the mineral soil; the mean litter depth at the treatment log sites was reduced to 0.9 cm (94% reduction). The fire personnel prevented the logs from burning, although some scorching took place on parts of the two treatment logs. Following the treatments, we observed no new immediate regeneration of seedlings during the 3 years of the post-fire study.

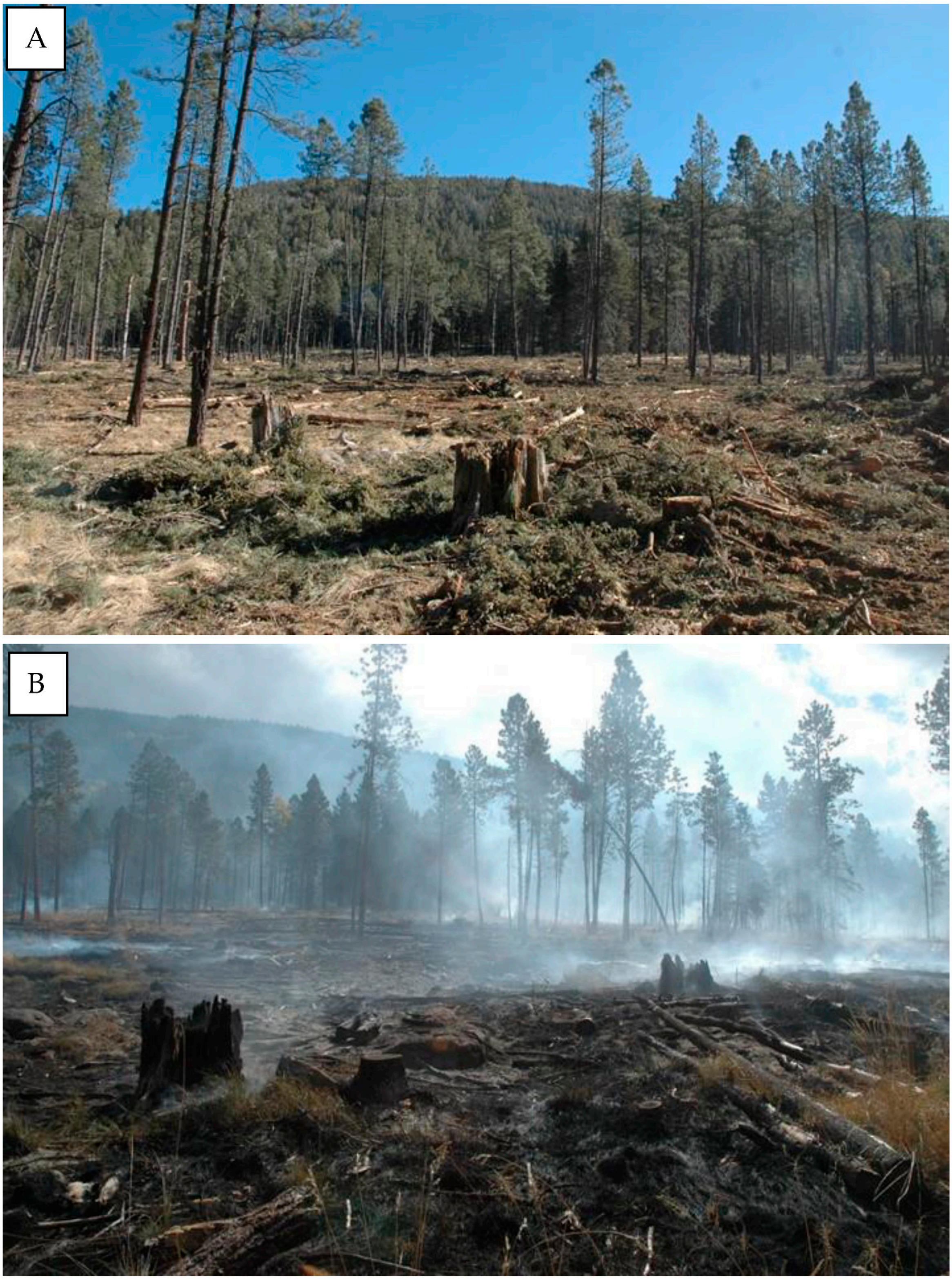

Figure 3. (**A**) Photograph of the study site during the thinning operations, December 2017. (**B**) Photograph of the planned, managed slash burn on the study site on 17–18 October 2019.

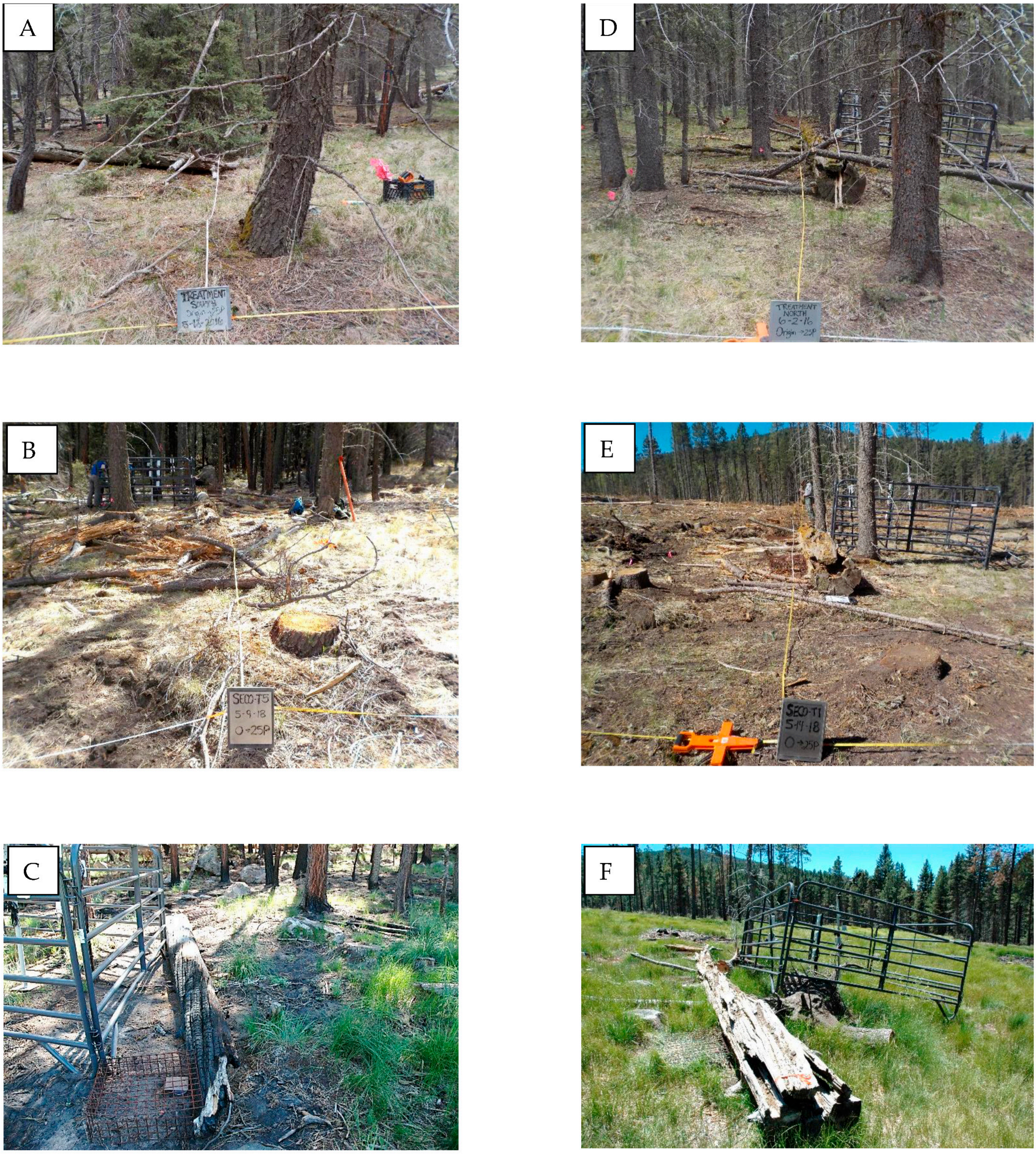

Figure 4. Photographs of logs on the treatment sites: South Site (**left** column) and North Site (**right** column). (**A**,**D**): Pre-thinning, undisturbed condition in 2016. (**B**,**E**): Post-thinning condition in 2018. (**C**,**F**): Post-fire condition in 2022.

3.2. Site Climate Dynamics

Precipitation on the study area over the seven water years varied from a low of 264 mm in the water year 2018 to a high of 830 mm in the water year 2017 (Supplemental Figure S1A). The winter precipitation peaked at 502 mm in the water year 2017 and totaled 379 mm and 355 mm in the water years 2019 and 2020, respectively; the remaining years produced winter totals between 77 and 217 mm.

The air temperatures on both Control and Treatment sites displayed consistent values and dynamics in all years (see example data set from the North Control site, Supplemental Figure S1B). The treatment effects (thinning and burning) resulted in small but significant changes in the air temperature (Table 1; Supplemental Table S1). The temperatures across all sites ranged from −16.8 °C to 22.7 °C.

Table 1. Summary of the mean weekly values of the meteorological variables at Valles Caldera National Preserve, NM. Values are means ± SE. Superscript letters indicate significant ($p < 0.05$) treatment differences among means based on Repeated-Measures ANOVA LSD means comparison tests.

Site	Period	Air Temperature (°C)	Relative Humidity (%)	PAR (μmol/m²s)	Mean Wind Speed (m/s)	Maximum Wind Speed (m/s)
Control	Pre-treatment	5.14 ± 0.99^{A}	0.69 ± 0.02^{A}	52 ± 5^{A}	0.17 ± 0.01^{A}	1.27 ± 0.03^{A}
	Post-thinning	5.28 ± 1.02^{A}	0.61 ± 0.02^{B}	63 ± 5^{AB}	0.27 ± 0.01^{B}	1.87 ± 0.05^{B}
	Post-fire	5.28 ± 1.04^{A}	0.61 ± 0.02^{B}	65 ± 5^{B}	0.25 ± 0.01^{B}	1.92 ± 0.04^{B}
Treatment	Pre-treatment	5.58 ± 0.98^{A}	0.67 ± 0.02^{A}	53 ± 5^{A}	0.24 ± 0.01^{A}	1.49 ± 0.03^{C}
	Post-thinning	6.24 ± 1.00^{B}	0.57 ± 0.01^{B}	258 ± 14^{C}	0.75 ± 0.03^{B}	3.53 ± 0.08^{D}
	Post-fire	6.34 ± 1.01^{B}	0.57 ± 0.02^{B}	287 ± 14^{D}	0.84 ± 0.04^{C}	3.79 ± 0.09^{E}

The air relative humidity exhibited considerable variability in daily and seasonal values, with low values of 12% and highs of 100% during precipitation events (Supplemental Figure S1C). The mean daily relative humidity was highest in winter (70%) and lowest in spring (49%), with intermediate values in fall (63%) and summer (60%). The humidity values were on average 9% higher in the two pre-treatment years than in the post-treatment years on both Control and Treatment sites (Table 1; Supplemental Table S2).

The wind speeds were low within the untreated forest canopy. The mean daily wind speeds were 0.12–0.22 m/s on the pre-treatment Control sites and showed a significant increase of 0.1 m/s after the pre-treatment period (Table 1; Supplemental Figure S2A); the mean maximum daily wind speed ranged from 1.15–1.39 m/s on the two Control sites and also displayed an increase after the first two years of the study (Table 1; Supplemental Figure S2B). In contrast, the wind speeds on the Treatment sites showed much larger increases following the thinning and burning treatments (Figure 5A,B; Table 1; Supplemental Tables S3 and S4).

Pre-treatment solar radiation (daily PAR) averaged 51 μmol/m^{2}s across the study sites (Supplemental Figure S2C) and increased by 15 μmol/m^{2}s after the pre-treatment period on the Control sites (Table 1). On the Treatment sites, PAR exhibited large increases (mean of 269 μmol/m^{2}s) following canopy reduction from the thinning treatment (Figure 5C; Table 1; Supplemental Table S5).

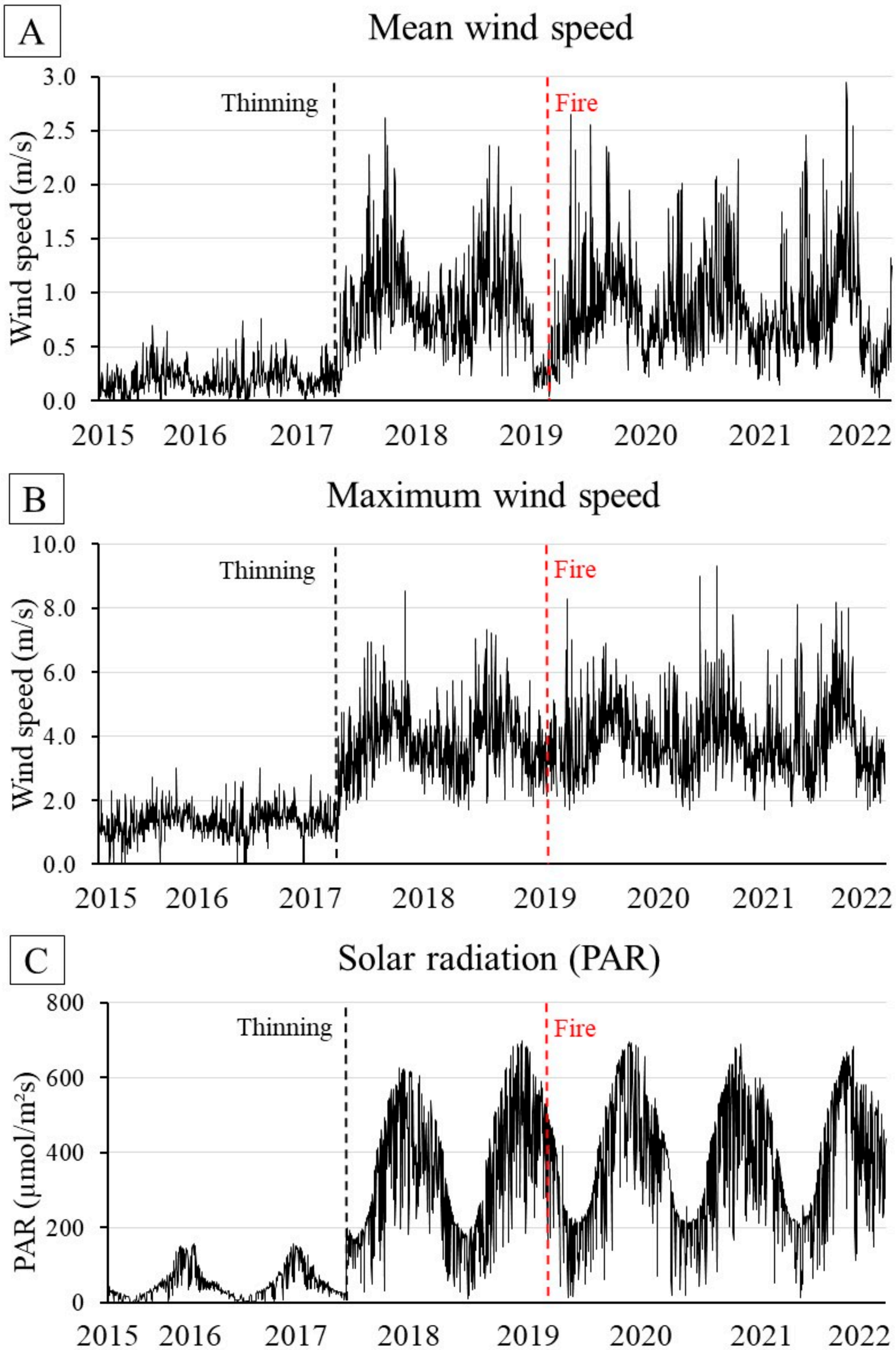

Figure 5. Example of the thinning and burning treatment effects from the North Treatment site on (**A**) mean wind speed, (**B**) maximum wind speed, and (**C**) Photosynthetically Active Radiation (PAR) during 2015–2022, Valles Caldera National Preserve, NM.

3.3. Soil Temperatue Dynamics

The soil temperature dynamics on the Control sites remained consistent throughout the seven years of the study. In soils directly beneath the control logs, the temperatures ranged from lows of −2 °C, to 0 °C, to highs near 12 °C at all three depths (10, 20, 30 cm), with the shallow 10 cm depth having slightly higher temperatures than other depths in summer, and slightly lower temperatures than deeper soils in winter (Supplemental Figure S3A). Sites 1–2 m away from the log showed similar patterns, except that the summer temperatures reached maximum values of 13 °C–16 °C, while the winter temperatures were comparable to those of sub-log sites.

The Repeated-Measures ANOVA results at each soil depth indicated significant differences in soil temperatures associated with treatments, periods, seasons, locations, and most interaction terms (Figure 6, Table 2; see Supplementary Tables S6–S8 for ANOVA soil temperature summaries); hence, we rejected our original null hypothesis of no effects from thinning and burning. The treatments of thinning and subsequent broadcast slash burning around the logs resulted in significant increases in the mean weekly soil temperatures beneath the logs during fall, spring, and summer, with changes in temperature ranging from +1.4 °C to +2.7 °C depending on the soil depth; the maximum temperature increases were observed during summer (Figure 6; Table 2). The thinning and burning treatments had minimal effect on the winter soil temperatures. We observed similar patterns in soil temperatures at 1–2 m from the logs in both directions (north and south), with minimal changes in winter and maximum effects in spring, summer, and fall (in the range from approximately +1 °C to +3.6 °C (Figure 7; Table 2).

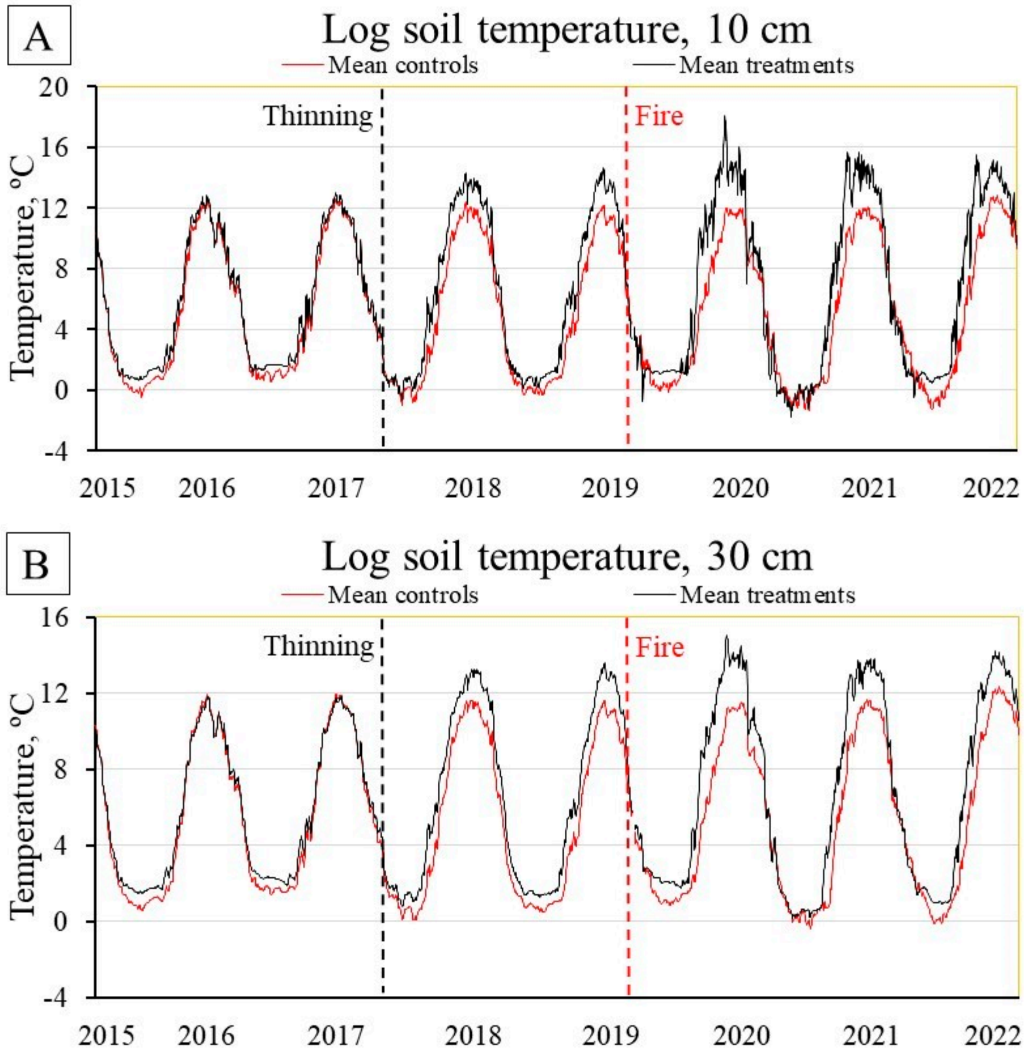

Figure 6. Mean soil temperature dynamics at 10 cm (**A**) and 30 cm (**B**) depths between Control and Treatment sites beneath the logs during 2015–2022 on the Cerro Seco study site, Valles Caldera National Preserve, NM.

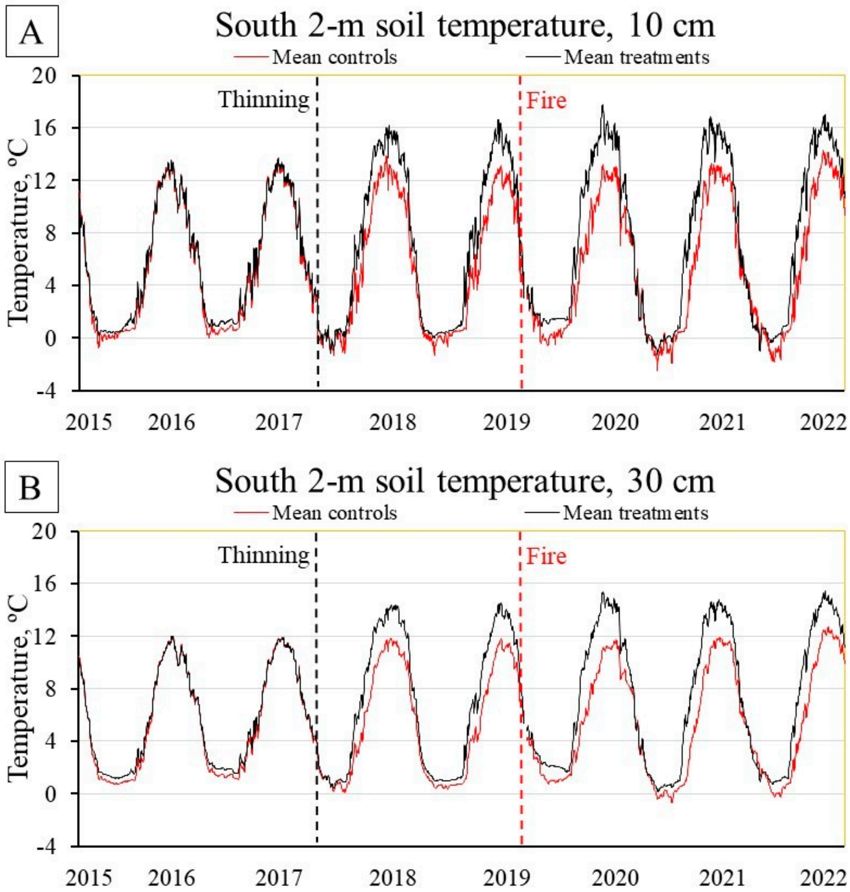

Figure 7. Example graphics for mean soil temperature dynamics at 10 cm (**A**) and 30 cm (**B**) depths between Control and Treatment sites at sites 2 m south of the logs during 2015–2022 on the Cerro Seco study site, Valles Caldera National Preserve, NM.

Table 2. Summary of the mean weekly soil temperature ($T_{D,}$ °C) differences between pre-treatment and post-treatment values at different depths (D = 10, 20, 30 cm) with thinning and burning treatments at Valles Caldera National Preserve, NM. Treatment values were corrected for site differences using Control sites' pre-treatment Z-scores during the water years 2016–2017. Boldface font indicates significant ($p < 0.05$) treatment differences from the pre-treatment values based on Repeated-Measures ANOVA LSD means comparison tests.

		Post-Thinning, Pre-Fire				**Post-Fire**			
		(Water Years 2018, 2019)				**(Water Years 2020, 2021, 2022)**			
Microsite	**Variable**	**Fall**	**Winter**	**Spring**	**Summer**	**Fall**	**Winter**	**Spring**	**Summer**
Log	T_{10}	**1.15**	0.02	**1.20**	**1.69**	0.42	−0.46	**2.17**	**2.54**
	T_{20}	**1.30**	0.01	**1.14**	**1.72**	0.62	−0.47	**2.03**	**2.68**
	T_{30}	**1.43**	0.18	**1.16**	**1.77**	**1.19**	−0.15	**1.98**	**2.75**
1-m north	T_{10}	**1.37**	−0.51	**2.14**	**2.06**	**1.20**	−0.27	**2.06**	**2.18**
	T_{20}	**1.45**	−0.41	**1.68**	**1.95**	**1.25**	0.02	**1.48**	**2.24**
	T_{30}	**1.49**	−0.43	**1.62**	**1.87**	**1.40**	−0.17	**1.33**	**2.25**
2-m north	T_{10}	**1.65**	0.01	**2.35**	**2.40**	**1.31**	**0.44**	**2.75**	**2.28**
	T_{20}	**1.64**	0.03	**2.17**	**2.25**	**1.37**	**0.39**	**2.47**	**2.30**
	T_{30}	**1.64**	0.05	**1.85**	**2.04**	**1.50**	**0.40**	**2.09**	**2.30**
1-m south	T_{10}	**1.93**	−0.04	**1.97**	**2.41**	**1.36**	0.18	**2.95**	**2.70**
	T_{20}	**2.01**	0.01	**1.82**	**2.36**	**1.65**	0.21	**2.82**	**2.86**
	T_{30}	**1.91**	0.01	**1.69**	**2.28**	**1.69**	0.24	**2.67**	**2.91**
2-m south	T_{10}	**1.86**	−0.01	**2.30**	**2.84**	**1.55**	0.14	**3.63**	**3.38**
	T_{20}	**1.83**	−0.02	**2.13**	**2.84**	**1.51**	0.14	**3.29**	**3.31**
	T_{30}	**1.85**	−0.02	**1.93**	**2.77**	**1.71**	0.19	**3.00**	**3.37**

In addition, the diel patterns of soil temperatures over time scales of several days differed following the thinning and burning treatments (Figure 8). The typical pre-treatment summer temperatures were regimented, with shallow soils consistently having warmer temperatures throughout diurnal and nocturnal periods compared to deeper soils, with relatively low amplitudes over 24 h periods. Post-fire 10 cm soils displayed increased diel amplitudes, with temperature dips indicating greater cooling at night than in deeper (20 cm) soils and higher peaks during late afternoon (Figure 8A,B). The winter soil temperatures were inversely regimented (the soils at deeper depths were warmer) and were relatively stable at 20 and 30 cm depths, exhibiting virtually no diel patterns; shallow soils in winter were generally colder, with some diel warming and cooling (Figure 8C,D).

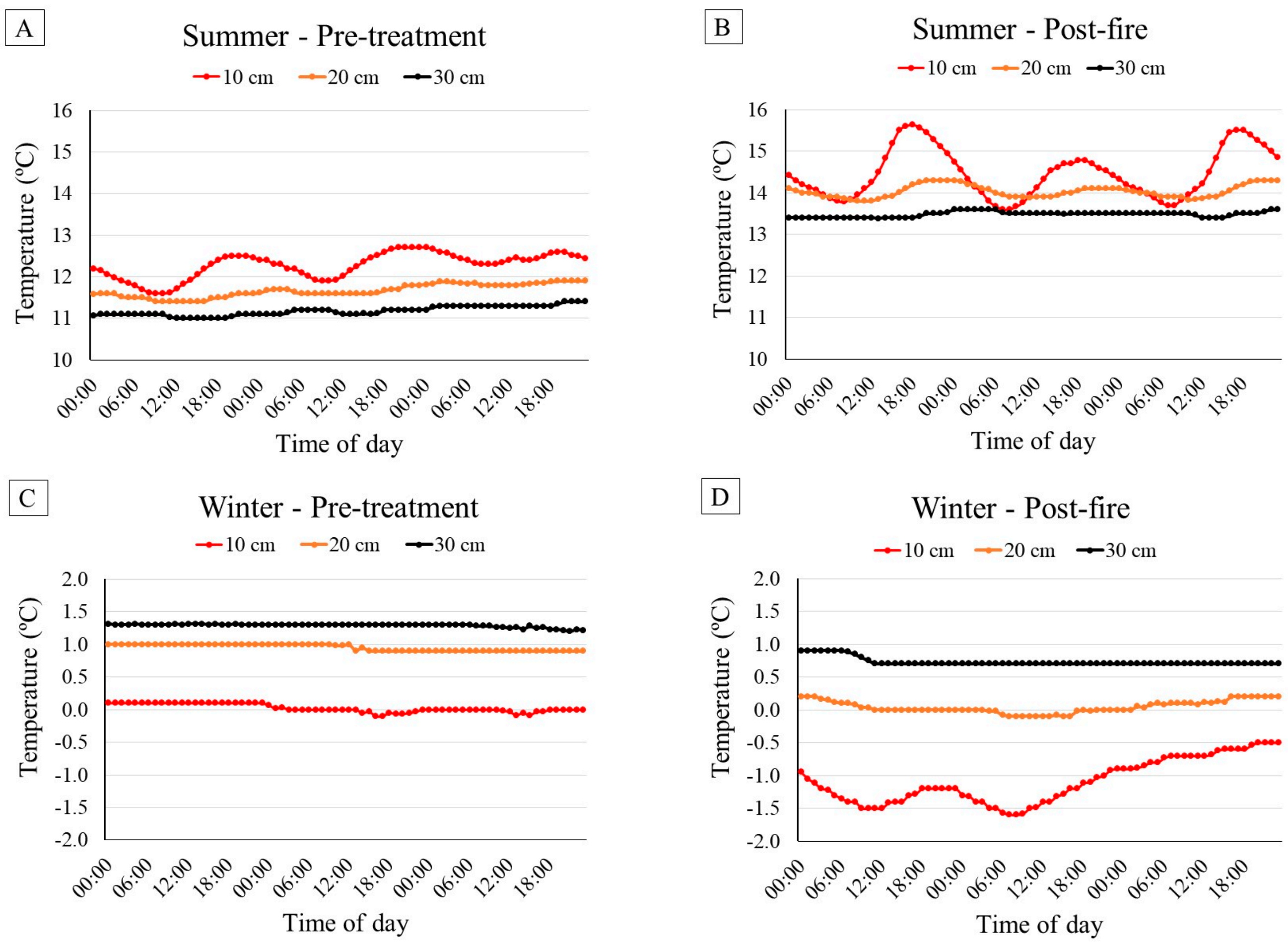

Figure 8. Examples of diel patterns of the soil temperature beneath the logs at different depths at the Cerro Seco study site, Valles Caldera National Preserve, NM. Data from the North Treatment Site. (**A**): 22–24 July 2016 (pre-treatment); (**B**): 14–16 July 2020 (post-fire); (**C**): 26–28 January 2016 (pre-treatment); (**D**): 12–14 February 2019 (post-fire).

3.4. Soil Moisture Dynamics

Soil moisture (VWC) also exhibited significant treatment, period, season, location, and most interaction effects in the Repeated-Measures ANOVA results (Table 3, Supplementary Tables S9–S11); again, we rejected the original null hypothesis of no effects for forest thinning and burning. Over the entire study, the mean daily soil moisture values on the Control sites ranged from 0.10 to 0.25 m^3/m^3 (Supplemental Figure S4). The soils beneath the logs at 10 cm depth were significantly drier than the soils at both 20 and 30 cm depths (Table 2). The soil moisture peaked in spring, concomitant with snowmelt, and exhibited smaller spikes during summer and fall associated with monsoon rain events (Supplemental

Figure S4). The thinning treatments resulted in periods of temporary increased moisture, with the North Treatment site (11% canopy cover) attaining field capacity (0.50 m^3/m^3) at all soil depths in spring 2019 and again in 2020 following the burn treatment in fall 2019 (Supplemental Figure S4). The South Treatment site (40% canopy cover) produced only modest increases in soil moisture at the 30 cm depth compared to the pre-treatment levels.

The mean weekly soil moisture at the 10 cm depth beneath the logs tended to either not respond to thinning and burning or show a decrease in the VWC values, although the magnitude of these decreases was small (<0.03 m^3/m^3, Figure 9, Table 3). Deeper soils (20 and 30 cm) beneath the logs following burning showed larger VWC increases in the range of 0.05–0.08 m^3/m^3 (Figure 9, Table 3).

The soil moisture values 1 m north of the logs exhibited significant drying at the 10 cm depth following thinning and burning, but the magnitude of the changes was <0.07 m^3/m^3 (Table 3). Deeper soils tended to increase in VWC by approximately 0.03 m^3/m^3. Other sites away from the logs (1 m south and 2 m north) displayed mixed responses but showed significant VWC increases in winter–spring with snowmelt periods of 0.03 to 0.08 m^3/m^3 (Table 3). The 2 m South Site displayed no significant changes in VWC (Figure 10), despite showing the largest increase in temperature (Figure 7).

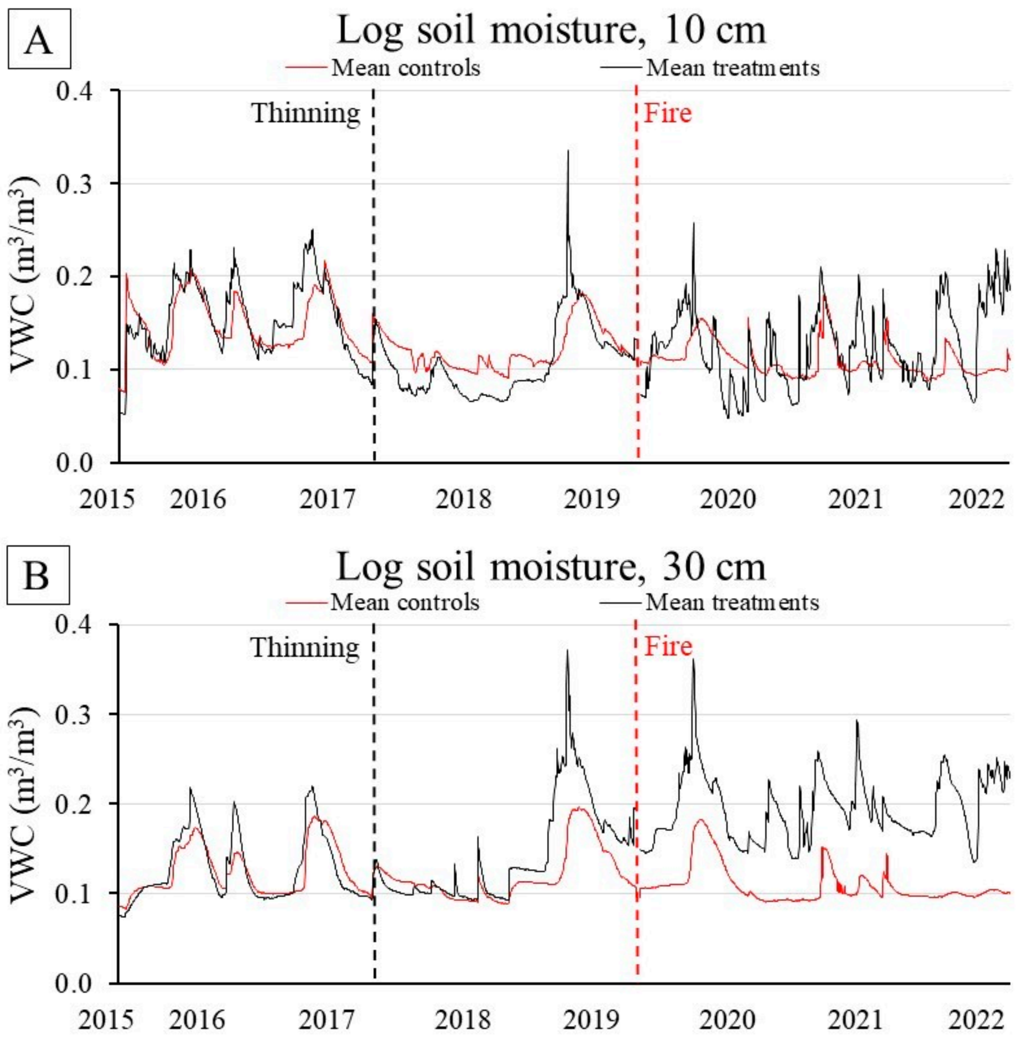

Figure 9. Mean soil moisture dynamics at 10 cm (**A**) and 30 cm (**B**) depths between Control and Treatment sites beneath the logs during 2015–2022 in the Cerro Seco study site, Valles Caldera National Preserve, NM.

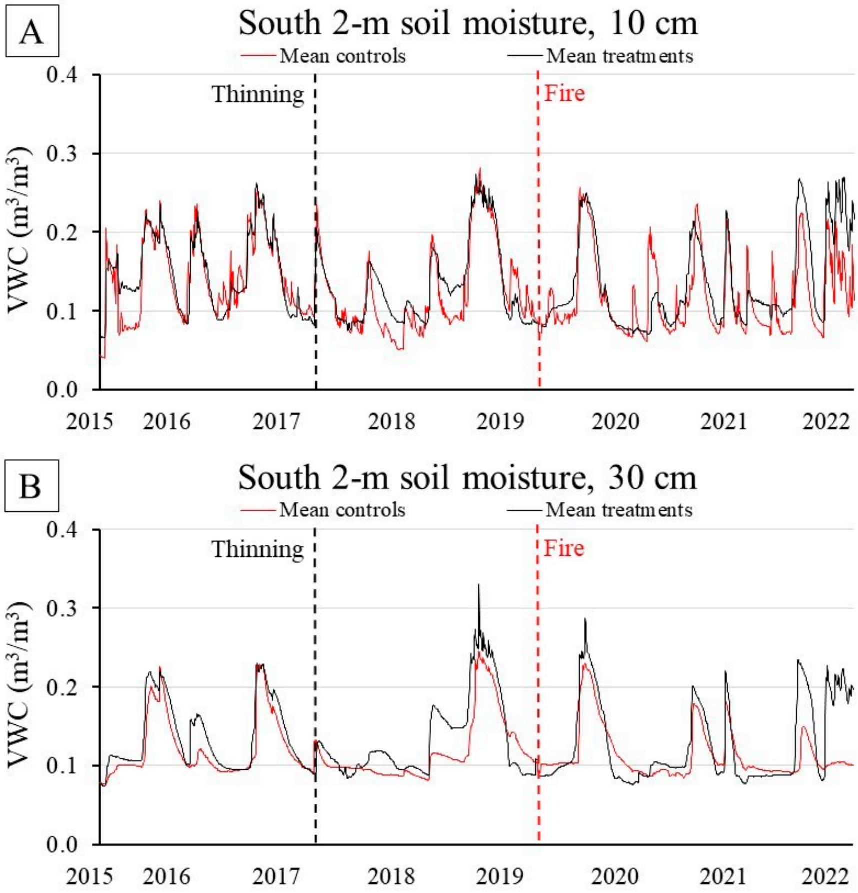

Figure 10. Mean soil moisture dynamics at 10 cm (**A**) and 30 cm (**B**) depths between Control and Treatment sites 2 m south of the logs during 2015–2022 in the Cerro Seco study site, Valles Caldera National Preserve, NM.

Table 3. Summary of the mean weekly soil volumetric water content ($VWC_{D,}$ m^3 water/m^3 soil) differences between pre-treatment and post-treatment values at different depths (*D* = 10, 20, 30 cm) with thinning and burning treatments at Valles Caldera National Preserve, NM. Treatment values were corrected for site differences using Control sites' pre-treatment Z-scores during the water years 2016–2017. Boldface font indicates significant ($p < 0.05$) treatment differences from the pre-treatment values based on Repeated-Measures ANOVA LSD means comparison tests.

		Post-Thinning, Pre-Fire				**Post-Fire**			
		(Water Years 2018, 2019)				**(Water Years 2020, 2021, 2022)**			
Microsite	**Variable**	**Fall**	**Winter**	**Spring**	**Summer**	**Fall**	**Winter**	**Spring**	**Summer**
Log	VWC_{10}	0.00	**−0.03**	0.00	**−0.02**	0.01	0.00	0.00	−0.01
	VWC_{20}	0.03	−0.01	0.03	0.01	**0.05**	**0.05**	**0.07**	**0.05**
	VWC_{30}	**0.03**	0.01	**0.03**	**0.02**	**0.07**	**0.08**	**0.08**	**0.08**
1-m north	VWC_{10}	**−0.03**	0.00	**−0.06**	**−0.05**	**−0.03**	**−0.03**	**−0.07**	**−0.04**
	VWC_{20}	−0.02	**0.04**	0.01	−0.01	0.00	0.01	**0.04**	−0.01
	VWC_{30}	−0.01	**0.03**	**0.03**	0.00	0.00	**0.04**	**0.04**	0.00
2-m north	VWC_{10}	0.01	0.01	0.01	−0.01	0.02	**0.05**	0.01	0.00
	VWC_{20}	0.02	0.00	0.03	0.00	0.04	**0.04**	0.02	0.01
	VWC_{30}	0.00	0.00	0.02	−0.01	0.02	**0.03**	0.01	−0.01
1-m south	VWC_{10}	0.03	0.05	**0.06**	0.04	0.04	0.06	**0.07**	0.04
	VWC_{20}	0.04	0.05	**0.08**	0.05	0.05	0.06	**0.08**	0.05
	VWC_{30}	0.04	0.02	**0.05**	0.03	**0.06**	**0.06**	**0.07**	0.05
2-m south	VWC_{10}	−0.01	0.00	0.02	0.01	0.00	0.00	0.01	0.01
	VWC_{20}	0.00	0.02	0.02	−0.01	−0.02	0.00	0.00	−0.02
	VWC_{30}	−0.02	0.02	0.01	−0.02	−0.03	0.00	0.01	−0.03

4. Discussion

Fire suppression in the Rocky Mountains during the 20th century created changes in forest stand structure that now require considerable management actions to be rectified; with respect to the soils, long-term fire suppression led to lower nutrient availability, increased pore spaces and water-holding capacity, generally lower soil temperatures, greater hydrophobicity, and increased seasonally dry conditions [62]. Forest restoration actions utilizing thinning operations and planned/managed fire are designed to reduce forest fuels and return natural fire regimes to the landscape [63]. Based on our results, the retention of logs in thinned and burned areas can provide microsites with warmer temperatures and enhanced soil moisture, which can contribute to wildlife habitat and biological diversity and promote ecosystem processes [17–36].

On our study site, the thinning operations and associated post-treatment wind-throw removed nearly 90% of the trees (from a mean of 1429 trees/ha to 150 trees/ha) and reduced the tree canopy closure from an average of 83% to 26% above the experimental log sites; the post-thinning fire reduced the litter layer depth by 94%, exposing patches of mineral soil. Due to the wind-throw event that felled the North site's dominant canopy trees, this site effectively changed from a forest-thinned site to a forest gap site. As such, our two instrumented sites spanned the range of open gap to thinned forest; however, the soils' microenvironments of both sites responded to the treatments in a similar fashion. During the 3 post-fire years, herbaceous plants began colonizing the sites, returning some grass and forb vegetation cover (Figure 3). The treatments opened the site to much greater levels of PAR and wind speeds, but only minimal increases in the average air temperatures occurred (Table 1).

As a result of the increased solar inputs, the soil temperatures beneath logs at all depths measured (10, 20, and 30 cm) increased by a value from 1.1 to 2.7 °C in spring, summer, and fall, while the winter temperatures remained essentially unchanged (Table 2). The temperature increase came with the exposure to increased solar radiation immediately after the thinning, and additional soil warming occurred after the fire removed most of the litter layer adjacent to the logs. In addition, the post-fire diel soil temperature cycles in summer nearly doubled in amplitude (to 2 °C) compared to the pre-treatment cycles (1 °C; Figure 8A,B), warming to nearly 16 °C in the afternoons and cooling to below 14 °C in the mornings. In winter, the reduced forest canopy coverage exposed the sites to increased heat loss and resulted in slightly colder temperatures, particularly at the 10 cm depth (Figure 8C,D). This was especially evident in the winter of 2017–2018, which had virtually no snowpack to insulate the ground from radiative heat losses (Figure S4A). The soil temperatures at 1 and 2 m away from the logs also increased following thinning and burning, with the largest increases (up to 3.6 °C) occurring in spring and summer (Table 2).

We had predicted that the soil moisture beneath logs in thinned and burned areas would decrease, and this was the observed result at the 10 cm depth; however, the soil moisture actually increased in all seasons after the burn treatment at the 20 and 30 cm depths (Table 3). The moisture increases were greatest at the 30 cm depth (up to 0.08 m^3/m^3), while the shallower depth (10 cm) showed no significant increases in any season. The increase in soil moisture likely resulted from reduced tree transpiration demands on soil water, offset by increased soil evaporation and herbaceous plant establishment; apparently, the gain from reduced tree water requirements was greater than the loss from increased evaporation and grass/forb transpiration. In addition, snowpack snow water equivalent (SWE) dynamics appeared to play a role. Spring moisture on the more open treatment plot (North Site) achieved field capacity (~50% VWC) in 2019 and 2020, the two post-thinning winters with >300 mm of winter precipitation (Supplemental Figure S4). This was likely due to larger quantities of snow reaching the ground after the site treatments, which made greater amounts of SWE available during spring snowmelt. Previous research on the Preserve found that high-density forest canopy conditions (>40% cover) intercept snowfall, with subsequent sublimation of snow water equivalent [64]. Open gaps in the forest canopy allow 25% more snow depth on the ground, versus closed canopy stands [64]. In addition, the logs accumulate wind-blown snow on both sides, increasing the local distribution of snow adjacent to them. Three of the four soil sites 1 and 2 m away from the logs also recorded higher moisture levels in winter–spring of <0.05 m^3/m^3 with occasional instances of increases to 0.08 m^3/m^3 (Table 3).

Our results for soil temperatures and moisture produced both similarities and differences compared to other studies reporting soil environmental values from coniferous forests. For example, in western Washington, thinning operations across a range of green tree retentions (0, 15, 40, 100%) yielded significantly higher soil temperatures (3 °C) between 0% and 100% retention treatments, results which are comparable to our observations using natural logs [37]; in addition, the soil moisture results showed no treatment differences by late summer, which is also consistent with our results. In another study on soils of post-thinned and burned sites in mixed conifer forests of the Sierra Nevada, the results showed no significant differences in temperature among the burn treatments (no burn, light burn, hot burn), while the soil moisture indicated a more rapid drying in spring following burning compared to not burning. The soil moisture difference in spring indicated 2% drier thinned/burned sites compared to controls [40]. Finally, microclimates in Sierra Nevada forests were monitored during and after thinning and burning, and sites that were thinned and burned exhibited increased air temperatures from 58% to 124%, decreased relative humidity, increased PAR by 50–255%, and increased wind speed by 15–194% [38]. The soil temperatures increased in the thinned and burned plots by up to 8° C, and the soil moisture increased by 8–40% [38].

These studies, combined with other results from clear-cutting and salvage-logging operations [13,19,25,41–55], indicate a wide variety of responses in soil temperatures and moisture depending on site-specific circumstances. In our mixed conifer forests of northern New Mexico, the retention of natural logs following thinning and burning provided microsites with generally warmer sub-log soil temperatures in spring, summer, and fall, while adjacent sites experienced comparable warming levels. In addition, the soil moisture increased in 20–30 cm deep soils in most microsites beneath and adjacent to logs. While the magnitude of the temperature differences was not great (<3.6 °C), warmer temperatures and increased moisture would enhance soil respiration and nutrient cycling processes [17–25] and increase the biological activities of soil invertebrates [28–31] and surface-active insects (e.g., ants [65] and grasshoppers [66]).

Logs in thinned areas that increase soil moisture, while not attaining critically high temperatures [67,68], may also continue to provide refugia for vertebrate wildlife, such as amphibians. The moisture within logs is known to track underlying soil moisture [69]; so, if soil moisture increases, then intra-log moisture increases will follow, and this should enhance favorable microenvironments for wildlife. For example, the endemic and Federalfy endangered Jemez Mountains salamander (*Plethodon neomexicanus* Stebbins & Riemer) has preferred nocturnal activity temperatures of 10.5° to 13.0 °C during wet (100% RH) conditions [70] and a critical thermal maximum of 33.5 °C [71]. The soil moisture range for surface-active salamanders is 9–61% (by water mass) [72]. The soil temperatures 4 cm beneath cover objects (logs and rocks) differed in areas impacted by high- and medium-severity forest fires and areas with low-severity or no fire; the summer soil temperatures ranged from 11 °C to 15 °C in low-severity and unburned sites and from 14 °C to 24 °C in high- and medium-severity fire sites [32]. The nighttime soil temperatures 10 cm under the logs in our thinning/burn site during summer ranged from 13.5 °C to 14.5 °C (Figure 9), very close to the preferred temperature of the salamander. Therefore, the retention of logs during thinning and making efforts to protect the logs during planned fires will likely enhance the post-treatment habitats for this and other small vertebrate wildlife species.

5. Conclusions

The results of our study demonstrated that the practice of retaining natural logs for providing microhabitat heterogeneity during forest management activities in northern New Mexico can lead to microenvironments with small growing-season soil temperature increases (2–3 °C) and moderate increases (0.09 m^3/m^3) in soil moisture beneath and adjacent to logs. While our case study was limited to Vitrandic Agriudoll and Hapludalf soils, these are typical soils derived from volcanic rhyolite parent materials and are widespread in the Jemez Mountains; as such, our results should apply to other areas in the region. The observed temperature and moisture changes provide a somewhat warmer and moister environment for ecosystem soil processes and activities of soil and surface-active invertebrates. The soil moisture also influences the moisture levels within logs lying on the soil surface and thus potentially improves the intra-log environmental conditions for fungi, invertebrates, and vertebrates such as salamanders. We conclude that forest management operations which focus on ecosystem restoration should include the retention of natural logs to provide increased soil microsite heterogeneity and potentially favorable conditions for vegetation establishment and wildlife refugia.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/f14061117/s1, Table S1. Repeated-Measures ANOVA Table for air temperature on the Cerro Seco study site, Valles Caldera National Preserve, New Mexico; Table S2. Repeated-Measures ANOVA Table for relative humidity on the Cerro Seco study site, Valles Caldera National Preserve, New Mexico; Table S3. Repeated-Measures ANOVA Table for PAR on the Cerro Seco study site, Valles Caldera National Preserve, New Mexico; Table S4. Repeated-Measures ANOVA Table for mean wind speed on the Cerro Seco study site, Valles Caldera National Preserve, New Mexico; Table S5. Repeated-Measures ANOVA Table for maximum wind speed on the Cerro Seco study site, Valles Caldera National Preserve, New Mexico; Table S6. Repeated-Measures ANOVA Table for soil temperature at 10 cm depth on the Cerro Seco study site, Valles Caldera National Preserve, New Mexico; Table S7. Repeated-Measures ANOVA Table for soil temperature at 20 cm depth on the Cerro Seco study site, Valles Caldera National Preserve, New Mexico; Table S8. Repeated-Measures ANOVA Table for soil temperature at 30 cm depth on the Cerro Seco study site, Valles Caldera National Preserve, New Mexico; Table S9. Repeated-Measures ANOVA Table for soil Volumetric Water Content (VWC) at 10 cm depth on the Cerro Seco study site, Valles Caldera National Preserve, New Mexico; Table S10. Repeated-Measures ANOVA Table for soil Volumetric Water Content (VWC) at 20 cm depth on the Cerro Seco study site, Valles Caldera National Preserve, New Mexico; Table S11. Repeated-Measures ANOVA Table for soil Volumetric Water Content (VWC) at 30 cm depth on the Cerro Seco study site, Valles Caldera National Preserve, New Mexico; Figure S1. Precipitation, air temperature, and relative humidity on the North Control site during 2015–2022, Valles Caldera National Preserve, NM; Figure S2. North Control site (A) mean wind speed, (B) maximum wind speed, and (C) Photosynthetically Active Radiation (PAR) during 2015–2022, Valles Caldera National Preserve, NM; Figure S3. Soil temperature dynamics beneath logs during 2015–2022 on the Cerro Seco study site, Valles Caldera National Preserve, NM.; Figure S4. Seasonal precipitation and soil moisture (Volumetric Water Content, VWC) dynamics beneath logs during 2015–2022 on the Cerro Seco study site, Valles Caldera National Preserve, NM.

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References

1. Thom, D.; Seidl, R. Natural disturbance impacts on ecosystem services and biodiversity in temperate and boreal forests. *Biol. Rev.* **2016**, *91*, 760–781. [CrossRef] [PubMed]
2. Kelly, L.T.; Giljohann, K.M.; Duane, A.; Aquilué, N.; Archibald, S.; Batllori, E.; Bennett, A.F.; Buckland, S.T.; Canelles, Q.; Clarke, M.F.; et al. Fire and biodiversity in the Anthropocene. *Science* **2020**, *370*, abb0355. [CrossRef] [PubMed]
3. McDowell, N.G.; Allen, C.D.; Anderson-Teixeira, K.; Aukema, B.H.; Bond-Lamberty, B.; Chini, L.; Clark, J.S.; Dietze, M.; Grossiord, C.; Hanbury-Brown, A.; et al. Pervasive shifts in forest dynamics in a changing world. *Science* **2020**, *368*, eaaz9463. [CrossRef]
4. McLauchlan, K.K.; Higuera, P.E.; Miesel, J.; Rogers, B.M.; Schweitzer, J.; Shuman, J.K.; Tepley, A.J.; Varner, J.M.; Veblen, T.T.; Adalsteinsson, S.A.; et al. Fire as a fundamental ecological process: Research advances and frontiers. *J. Ecol.* **2020**, *108*, 2047–2069. [CrossRef]
5. Williams, A.P.; Cook, E.R.; Smerdon, J.E.; Cook, B.I.; Abatzoglou, J.T.; Bolles, K.; Baek, S.H.; Badger, A.M.; Livneh, B. Large contribution from anthropogenic warming to an emerging North American megadrought. *Science* **2020**, *368*, 314–318. [CrossRef] [PubMed]
6. Jain, P.; Castellanos-Acuna, D.; Coogan, S.C.P.; Abatzoglou, J.T.; Flannigan, M.D. Observed increases in extreme fire weather driven by atmospheric humidity and temperature. *Nat. Clim. Change* **2022**, *12*, 63–70. [CrossRef]
7. Abatzoglou, J.T.; Williams, A.P. Impact of anthropogenic climate change on wildfire across western US forests. *Proc. Natl. Acad. Sci. USA* **2016**, *113*, 11770–11775. [CrossRef] [PubMed]
8. Kitzberger, T.; Falk, D.A.; Westerling, A.L.; Swetnam, T.W. Direct and indirect climate controls predict heterogeneous early-mid 21st century wildfire burned area across western and boreal North America. *PLoS ONE* **2017**, *12*, e0188486. [CrossRef]
9. Stevens-Rumann, C.S.; Kemp, K.B.; Higuera, P.E.; Harvey, B.J.; Rother, M.T.; Donato, D.C.; Morgan, P.; Veblen, T.T. Evidence for declining forest resilience to wildfires under climate change. *Ecol. Lett.* **2018**, *21*, 243–252. [CrossRef]
10. Coop, J.D.; Parks, S.A.; McClernan, S.R.; Holsinger, L.M. Influences of prior wildfires on vegetation response to subsequent fire in a reburned Southwestern landscape. *Ecol. Appl.* **2016**, *26*, 346–354. [CrossRef]
11. Coop, J.D.; Parks, S.A.; Stevens-Rumann, C.S.; Crausbay, S.D.; Higuera, P.E.; Hurteau, M.D.; Tepley, A.; Whitman, E.; Assal, T.; Collins, B.M.; et al. Wildfire-driven forest conversion in western North American landscapes. *BioScience* **2020**, *70*, 659–673. [CrossRef]
12. Guiterman, C.H.; Gregg, R.M.; Marshall, L.A.; Beckmann, J.J.; van Mantgem, P.J.; Falk, D.A.; Keeley, J.E.; Caprio, A.C.; Coop, J.D.; Fornwalt, P.J.; et al. Vegetation type conversion in the US Southwest: Frontline observations and management responses. *Fire Ecol.* **2022**, *18*, 6. [CrossRef]
13. Castro, J.; Allen, C.D.; Molina-Morales, M.; Marañón-Jiménez, S.; Sánchez-Miranda, A.; Zamora, R. Salvage logging versus the use of burnt wood as a nurse object to promote post-fire tree seedling establishment. *Restor. Ecol.* **2011**, *219*, 537–544. [CrossRef]
14. deMaynadier, P.G.; Hunter, M.L., Jr. The relationship between forest management and amphibian ecology: A review of the North American literature. *Environ. Rev.* **1995**, *3*, 230–261. [CrossRef]
15. Graham, S.A. The felled tree trunk as an ecological unit. *Ecology* **1925**, *6*, 397–411. [CrossRef]
16. Thorn, S.; Seibold, S.; Leverkus, A.B.; Michler, T.; Müller, J.; Noss, R.F.; Stork, N.; Vogel, S.; Lindenmayer, D.B. The living dead: Acknowledging life after tree death to stop forest degradation. *Front. Ecol. Environ.* **2020**, *18*, 505–512. [CrossRef]
17. Rey, A.; Pegoraro, E.; Tedeschi, V.; de Parri, I.; Jarvis, P.G.; Valentini, R. Annual variation in soil respiration and its components in a coppice oak forest in Central Italy. *Glob. Change Biol.* **2002**, *8*, 851–866. [CrossRef]
18. Peng, Y.; Thomas, S.C. Soil CO_2 efflux in uneven-aged managed forests: Temporal patterns following harvest and effects of edaphic heterogeneity. *Plant Soil* **2006**, *289*, 253–264. [CrossRef]
19. Guo, J.; Yang, Y.; Chen, G.; Xie, J.; Gao, R.; Qian, W. Effects of clear-cutting and slash burning on soil respiration in Chinese fir and evergreen broadleaved forests in mid-subtropical China. *Plant Soil* **2010**, *333*, 249–261. [CrossRef]

20. Hagemann, U.; Martin, T.; Moroni, M.T.; Gleißner, J.; Makeschin, F. Disturbance history influences downed woody debris and soil respiration. *For. Ecol. Manag.* **2010**, *260*, 1762–1772. [CrossRef]
21. Ruehr, N.K.; Knohl, A. Environmental variables controlling soil respiration on diurnal, seasonal and annual time-scales in a mixed mountain forest in Switzerland. *Biogeochemistry* **2010**, *98*, 153–170. [CrossRef]
22. Marañón-Jiménez, S.; Castro, J.; Kowalski, A.S.; Serrano-Ortiz, P.; Reverter, B.R.; Sánchez-Cañete, E.P.; Zamora, R. Post-fire soil respiration in relation to burnt wood management in a Mediterranean mountain ecosystem. *For. Ecol. Manag.* **2011**, *261*, 1436–1447. [CrossRef]
23. Xu, J.; Chen, J.; Brosofske, K.; Li, Q.; Weintraub, M.; Henderson, R.; Wilske, B.; John, R.; Jensen, R.; Li, H.; et al. Influence of timber harvesting alternatives on forest soil respiration and its biophysical regulatory factors over a 5-year period in the Missouri Ozarks. *Ecosystems* **2011**, *14*, 1310–1327. [CrossRef]
24. Lepistö, A.; Futter, M.N.; Kortelainen, P. Almost 50 years of monitoring shows that climate, not forestry, controls long-term organic carbon fluxes in a large boreal watershed. *Glob. Change Biol.* **2014**, *20*, 1225–1237. [CrossRef] [PubMed]
25. Kellman, L.; Myette, A.; Beltrami, H. Depth-dependent mineral soil CO_2 production processes: Sensitivity to harvesting-induced changes in soil climate. *PLoS ONE* **2015**, *10*, e0134171. [CrossRef]
26. Heilmann-Clausen, J.; Christensen, M. Fungal diversity on decaying beech logs—Implications for sustainable forestry. *Biodivers. Conserv.* **2003**, *12*, 953–973. [CrossRef]
27. Ylisirniö, A.-L.; Mönkkönen, M.; Hallikainen, V.; Ranta-Maunus, T.; Kouki, J. Woodland key habitats in preserving polypore diversity in boreal forests: Effects of patch size, stand structure and microclimate. *For. Ecol. Manag.* **2016**, *373*, 138–148. [CrossRef]
28. Wikars, L.-O.; Schimmel, J. Immediate effects of fire-severity on soil invertebrates in cut and uncut pine forests. *For. Ecol. Manag.* **2001**, *141*, 189–200. [CrossRef]
29. Meehan, T.D. Mass and temperature dependence of metabolic rate in litter and soil invertebrates. *Physiol. Biochem. Zool.* **2006**, *79*, 878–884. [CrossRef]
30. Bezkorovainaya, I.N.; Krasnoshchekova, E.N.; Ivanova, G.A. Transformation of soil invertebrate complex after surface fires of different intensity. *Biol. Bull.* **2007**, *34*, 517–522. [CrossRef]
31. Koivula, M.; Vanha-Majamaa, I. Experimental evidence on biodiversity impacts of variable retention forestry, prescribed burning, and deadwood manipulation in Fennoscandia. *Ecol. Process.* **2020**, *9*, 11. [CrossRef]
32. Cummer, M.R.; Painter, C.W. Three case studies of the effect of wildfire on the Jemez Mountains salamander (*Plethodon neomexicanus*): Microhabitat temperatures, size distributions, and a historical locality perspective. *Southwest. Nat.* **2007**, *52*, 26–37. [CrossRef]
33. Haan, S.S.; Desmond, M.J.; Gould, W.R.; Ward, J.P., Jr. Influence of habitat characteristics on detected site occupancy of the New Mexico endemic Sacramento Mountains Salamander, *Aneides Hardii*. *J. Herpetol.* **2007**, *41*, 1–8. [CrossRef]
34. Semlitsch, R.D.; Todd, B.D.; Blomquist, S.M.; Calhoun, A.J.K.; Gibbons, J.W.; Gibbs, J.P.; Graeter, G.J.; Harper, E.B.; Hocking, D.J.; Hunter, M.L., Jr.; et al. Effects of timber harvest on amphibian populations: Understanding mechanisms from forest experiments. *BioScience* **2009**, *59*, 853–862. [CrossRef]
35. Kluber, M.R.; Olson, D.H.; Puettmann, K.J. Amphibian distributions in riparian and upslope areas and their habitat associations on managed forest landscapes in the Oregon Coast Range. *For. Ecol. Manag.* **2008**, *256*, 529–535. [CrossRef]
36. Margenau, E.L.; Wood, P.B.; Brown, D.J.; Ryan, C.W. Evaluating mechanisms of short-term woodland salamander response to forest management. *Environ. Manag.* **2023**, *71*, 321–333. [CrossRef]
37. Heithecker, T.D.; Halpern, C.B. Variation in microclimate associated with dispersed-retention harvests in coniferous forests of western Washington. *For. Ecol. Manag.* **2006**, *226*, 60–71. [CrossRef]
38. Ma, S.; Concilio, A.; Oakley, B.; North, M.; Chen, J. Spatial variability in microclimate in a mixed conifer forest before and after thinning and burning treatments. *For. Ecol. Manag.* **2010**, *259*, 904–915. [CrossRef]
39. Kovács, B.; Tinya, F.; Guba, E.; Németh, C.; Sass, V.; Bidló, A.; Ódor, P. The short-term effects of experimental forestry treatments on site conditions in an oak–hornbeam forest. *Forests* **2018**, *9*, 406. [CrossRef]
40. Gray, A.N.; Zald, H.S.J.; Kern, R.A.; North, M. Stand conditions associated with tree regeneration in Sierran mixed conifer forests. *For. Sci.* **2005**, *51*, 198–210.
41. Gordon, A.M.; Schlentner, R.E.; van Cleve, K. Seasonal patterns of soil respiration and CO_2 evolution following harvesting in the white spruce of interior Alaska. *Can. J. For. Res.* **1987**, *17*, 304–310. [CrossRef]
42. Bhatti, J.S.; Fleming, R.L.; Foster, N.W.; Meng, F.-R.; Bourque, C.P.A.; Arp, P.A. Simulations of pre- and post-harvest soil temperature, soil moisture, and snowpack for jack pine: Comparison with field observations. *For. Ecol. Manag.* **2000**, *138*, 413–426. [CrossRef]
43. Slesak, R.A. Soil temperature following logging-debris manipulation and aspen regrowth in Minnesota: Implications for sampling depth and alteration of soil processes. *Soil Sci. Soc. Am. J.* **2013**, *77*, 1818–1824. [CrossRef]
44. Webster, K.L.; Hazlett, P.W.; Brand, G.; Nelson, S.A.; Primavera, M.J.; Weldon, T.P. The effect of boreal jack pine harvest residue retention on soil environment and processes. *For. Ecol. Manag.* **2021**, *497*, 119517. [CrossRef]
45. Yamulki, S.; Forster, J.; Xenakis, G.; Ash, A.; Brunt, J.; Perks, M.; Morison, J.I.L. Effects of clear-fell harvesting on soil CO_2, CH_4, and N_2O fluxes in an upland Sitka spruce stand in England. *Biogeosciences* **2021**, *18*, 4227–4241. [CrossRef]
46. Harrington, T.B.; Slesak, R.A.; Schoenholtz, S.H. Variation in logging debris cover influences competitor abundance, resource availability, and early growth of planted Douglas-fir. *For. Ecol. Manag.* **2013**, *296*, 41–52. [CrossRef]

47. Harrington, T.B.; Peter, D.H.; Slesak, R.A. Logging debris and herbicide treatments improve growing conditions for planted Douglas-fir on a droughty forest site invaded by Scotch broom. *For. Ecol. Manag.* **2018**, *417*, 31–39. [CrossRef]
48. Trottier-Picard, A.; Thiffault, E.; DesRochers, A.; Paré, D.; Thiffault, N.; Messier, C. Amounts of logging residues affect planting microsites: A manipulative study across northern forest ecosystems. *For. Ecol. Manag.* **2014**, *312*, 203–215. [CrossRef]
49. Cirelli, D.; Vinge, T.; Lieffers, V.J. Assisted lodgepole pine regeneration on reclamation sites using logging slash as both a mulch and natural seed source. *Can. J. For. Res.* **2016**, *46*, 1132–1137. [CrossRef]
50. Weber, M.G. Forest soil respiration after cutting and burning in immature aspen ecosystems. *For. Ecol. Manag.* **1990**, *31*, 1–14. [CrossRef]
51. Kulmalaa, L.; Aaltonen, H.; Berninger, F.; Kieloaho, A.-J.; Levula, J.; Bäck, J.; Hari, P.; Kolari, P.; Korhonen, J.F.J.; Kulmala, M.; et al. Changes in biogeochemistry and carbon fluxes in a boreal forest after the clear-cutting and partial burning of slash. *Agric. For. Meteorol.* **2014**, *188*, 33–44. [CrossRef]
52. Peterson, C.J.; Leach, A.D. Salvage logging after windthrow alters microsite diversity, abundance and environment, but not vegetation. *Forestry* **2008**, *81*, 361–376. [CrossRef]
53. Marcolin, E.; Marzano, R.; Vitali, A.; Garbarino, M.; Lingua, E. Post-fire management impact on natural forest regeneration through altered microsite conditions. *Forests* **2019**, *10*, 1014. [CrossRef]
54. Urretavizcaya, M.F.; Defossé, G.E. Restoration of burned and post-fire logged *Austrocedrus chilensis* stands in Patagonia: Effects of competition and environmental conditions on seedling survival and growth. *Int. J. Wildland Fire* **2019**, *28*, 365–376. [CrossRef]
55. Wooten, J.T.; Stevens-Rumann, C.S.; Schapira, Z.H.; Rocca, M.E. Microenvironment characteristics and early regeneration after the 2018 Spring Creek Wildfire and post-fire logging in Colorado, USA. *Fire Ecol.* **2022**, *18*, 10. [CrossRef]
56. Goff, F. *Valles Caldera: A Geologic History*; University of New Mexico Press: Albuquerque, NM, USA, 2009.
57. Balmat, J.; Kupfer, J. *Assessment of Timber Resources and Logging History of the Valles Caldera National Preserve*; Technical Report VCT04011 for Valles Caldera Trust; University of Arizona: Tucson, AZ, USA, 2004; p. 69.
58. Hibner, C.D.; Strenger, S.; Miller, A.; Sebring, S.; Olson, D.; Nemecek, J.; Schmit, S.; Bishop, C.; Ferguson, C.; Robbie, W.; et al. *Terrestrial Ecological Unit Inventory of the Valles Caldera National Preserve, Sandoval County, New Mexico*; Unpublished Report; USDA Forest Service, Region 3, Albuquerque, New Mexico and Natural Resources Conservation Service; USDA: Santa Fe, NM, USA, 2010.
59. Western Regional Climate Center (WRCC). Valles Caldera National Preserve Climate Stations. 2022. Available online: https://wrcc.dri.edu/vallescaldera/ (accessed on 15 October 2022).
60. Suazo, M.M.; Oertel, R.; Trader, L. Valles Caldera National Preserve: Forest monitoring objectives and methods. In *Methods Manual*; National Park Service: Valles Caldera National Preserve: Jemez Springs, NM, USA, 2013. Available online: https://irma.nps.gov/DataStore/Reference/Profile/2298941 (accessed on 15 January 2023).
61. Litvak, M. AmeriFlux BASE US-Vcs Valles Caldera Sulphur Springs Mixed Conifer. version 12-5; AmeriFlux AMP, (Dataset); Lawrence Berkeley National Laboratory: Berkeley, CA, USA, 2022. [CrossRef]
62. Keane, R.E.; Ryan, K.C.; Veblen, T.T.; Allen, C.D.; Logan, J.; Hawkes, B. *Cascading Effects of Fire Exclusion in the Rocky Mountain Ecosystems: A Literature Review*; General Technical Report, RMRSGTR-91; U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: Denver, CO, USA, 2002; p. 24.
63. Buford, M.; Chan, C.; Crockett, J.; Reinhardt, E.; Sloan, J. *From Accelerating Restoration to Creating and Maintaining Resilient Landscapes and Communities across the Nation*; USDA Forest Service Report FS-1069; USDA Forest Service: Washington, DC, USA, 2015. Available online: https://www.fs.usda.gov/sites/default/files/accelerating-restoration-update-2015-508-compliant.pdf (accessed on 15 January 2023).
64. Veatch, W.; Brooks, P.D.; Gustafson, J.R.; Molotch, N.P. Quantifying the effects of forest canopy cover on net snow accumulation at a continental, mid-latitude site. *Ecohydrology* **2009**, *2*, 115–128. Available online: https://soi.org/10.1002/eco.45 (accessed on 15 January 2023). [CrossRef]
65. Hölldobler, B.; Wilson, E.O. *The Ants*; Belknap Press: Cambridge, MA, USA, 1990.
66. Chapman, R.F.; Joern, A. *Biology of Grasshoppers*; John Wiley & Sons: New York, NY, USA, 1990.
67. Hossack, B.R.; Eby, L.A.; Guscio, C.G.; Corn, P.S. Thermal characteristics of amphibian microhabitats in a fire-disturbed landscape. *For. Ecol. Manag.* **2009**, *258*, 1414–1421. [CrossRef]
68. Kluber, M.R.; Olson, D.H.; Puettmann, K.J. Downed wood microclimates and their potential impact on plethodontid salamander habitat in the Oregon Coast Range. *Northwest Sci.* **2009**, *83*, 25–34. [CrossRef]
69. Green, M.B.; Fraver, S.; Lutz, D.A.; Woodall, C.W.; D'Amato, A.W.; Evans, D.M. Does deadwood moisture vary jointly with surface soil water content? *Soil Sci. Soc. Am. J.* **2022**, *86*, 1113–1121. [CrossRef]
70. Reagan, D.P. Ecology and distribution of the Jemez Mountains salamander, *Plethodon neomexicanus*. *Copeia* **1972**, *1972*, 486–492. [CrossRef]

71. Whitford, W.G. Physiological responses to temperature and desiccation in the endemic New Mexico plethodontids, *Plethodon neomexicanus* and *Aneides Hardii*. *Copeia* **1968**, *1968*, 247–251. Available online: http://www.jstor.org/stable/1441750?origin=JSTOR-pdf (accessed on 15 January 2023). [CrossRef]
72. Everette, E.M. Habitat Characterization and Environmental Influences of the Jemez Mountains Salamander (*Plethodon neomexicanus*). Master's Thesis, New Mexico State University, Las Cruces, NM, USA, 2003.