# Final Monitoring Report, 2010-2024

## Southwest Jemez Mountains
## Collaborative Forest Landscape Restoration Program
## (SWJM CFLRP)







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Cover photos: Series of photographs showing SWJM CFLRP forest restoration on Banco Bonito, Valles Caldera National Preserve, NM. Top left: Pre-treatment stand of second-growth ponderosa pine. Top right: Feller-buncher thinning operation. Bottom left: Post-thinning stand, with masticated slash. Bottom right: Stand conditions after prescribed fire with developing forest floor herbaceous layer. Photos by NPS staff.

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# Final Monitoring Report, 2010-2024

# Southwest Jemez Mountains Collaborative Forest Landscape Restoration Program (SWJM CFLRP)

## Introduction

The Southwest Jemez Mountains Collaborative Forest Landscape Restoration Program (SWJM CFLRP) on the Santa Fe National Forest and Valles Caldera National Preserve is a long-term collaborative effort to restore forest ecosystems at the landscape scale and improve resilience to major disturbances, including fire, insects, disease, and climate change. Lead partners include the Santa Fe National Forest (U.S. Forest Service), Valles Caldera National Preserve (National Park Service), US Geological Survey, Forest Stewards Guild, New Mexico Forest and Watershed Restoration Institute, Pueblo of Jemez, Los Amigos de Valles Caldera, Rio Grande Return and The Nature Conservancy. The entire SWJM Collaborative includes 40 organizations, agencies, tribes and universities (Table 1).

This summary report represents the results of the SWJM CFLRP implementation and monitoring programs. The report is structured around the objectives listed in the original Forest Landscape Restoration Act passed by Congress in 2009. The results presented have been contributed by the collaborating groups of the SWJM CFLRP based on their monitoring work through 2024.

***CFLRP Direct Objectives*:** As defined in the Forest Landscape Restoration Act (PL 111-11, Sec. 4003(c)), the natural resources monitoring program objectives are to evaluate the degree to which restoration actions:

(1) contribute toward the restoration of the structure and composition of pre-fire-suppression old growth stands,

(2) reduce the risk of uncharacteristic wildfire, and/or maintain or re-establish natural fire regimes,

(3) improve fish and wildlife habitat, including endangered, threatened and sensitive species,

(4) maintain or improve water quality and watershed function,

(5) prevent, remediate, or control invasions of exotic species, and

(6) create local economic development opportunities.

**Table 1. List of collaborating organizations for the Southwest Jemez Mountains Collaborative Forest Landscape Restoration Program (in alphabetical order)**

1. Bandelier National Monument, National Park Service
2. Forest Stewards Guild
3. Great Old Broads for Wilderness
4. Hawks Aloft, Inc.
5. Keystone Restoration Ecology
6. Los Alamos National Laboratory
7. Los Amigos de Valles Caldera
8. New Mexico Department of Game and Fish
9. New Mexico Environment Department
10. New Mexico Forest & Watershed Restoration Institute, Highlands University
11. New Mexico Forest Industry Association
12. New Mexico State Forestry Division (Energy, Minerals and Natural Resources Department)
13. New Mexico State University, Department of Fish, Wildlife, and Conservation Ecology
14. New Mexico Trout
15. Northern Arizona University
16. Pueblo of Jemez
17. Pueblo of Santa Clara
18. Rio Grande Return
19. Rio Grande Water Fund
20. Rio Puerco Alliance
21. Rocky Mountain Elk Foundation
22. Santa Fe National Forest, U.S. Forest Service
23. Texas Tech University, Department of Natural Resources Management
24. The Nature Conservancy
25. Trout Unlimited
26. U.S. Bureau of Indian Education, Southwestern Indian Polytechnic Institute, Department of Natural Resources Management
27. U.S. Department of Agriculture, Systematic Entomology Laboratory/Smithsonian Institution
28. U.S. Fish & Wildlife Service
29. University of Arizona, Department of Soil, Water and Environmental Science
30. University of Maryland, Center for Environmental Science
31. University of Nevada, Desert Research Institute
32. University of New Mexico, Department of Biology
33. University of New Mexico, Museum of Southwestern Biology
34. USGS, New Mexico Landscapes Field Station
35. USGS, New Mexico Cooperative Fish and Wildlife Research Unit
36. Valles Caldera National Preserve, National Park Service
37. Village of Jemez Springs
38. Walatowa Timber Industries & TC Company
39. Wild Turkey Federation
40. WildEarth Guardians

The SWJM CFLRP Collaborative was formed in 2009 to develop the project proposal to submit to the USDA Forest Service. The two major land units included in the project area were the majority of the Jemez Ranger District on the Santa Fe National Forest (SFNF, ~109,000 acres), and Valles Caldera National Preserve (Preserve, ~89,000 acres), with 12,000 acres of interspersed private and Jemez Pueblo tribal lands, for a total project area of 210,000 acres. At the time, Valles Caldera was part of USDA and the National Forest System lands and was eligible for CFLRP funding. The project proposal was selected for funding by USFS in FY2010, with some initial funding arriving in summer 2010. The main CFLRP funding ramped up to $4 million per year in the next few fiscal years, split between the SFNF and the Preserve. Extra funding of 8.3% of each annual budget was added to the Preserve budget from the SFNF for monitoring programs on both the Forest and Preserve lands, with additional funds going to the Forest Stewards Guild for socioeconomic monitoring. In 2015, Congress moved the Preserve to the National Park Service, which changed the land unit from National Forest System to Department of Interior land and, hence, made the Preserve ineligible to receive CFLRP funds from USDA. Fortunately, with support of the New Mexico Congressional Delegation, Senators Tom Udall and Martin Heinrich permanently increased the base budget for the Preserve within the National Park Service by $1.5 million/yr to compensate for the loss of CFLRP funds and to ensure that ecosystem restoration and management would continue in perpetuity. With NPS funds, the Preserve was able to continue participation in the SWJM CFLRP as a fully-active member, and continued its role as the leader of the Collaborative's monitoring program.

The SWJM CFLRP has fostered partnerships with land management entities within the project area (Pueblo of Jemez and Valles Caldera National Preserve) as well as a number of NGOs to greatly increase the amount of on-the-ground work and monitoring that we have accomplished. Of particular note is the robust monitoring program coordinated by the Preserve, supported by a combination of Forest Service and NPS funding. In addition to forest restoration and fire/fuels management, the Collaborative has greatly expanded restoration of streams and wetlands – partnerships with Rio Grande Return, Rio Puerco Alliance, Los Amigos de Valles Caldera, Keystone Restoration Ecology, the Pueblo of Jemez, Trout Unlimited, New Mexico Trout, The Nature Conservancy, the Forest Stewards Guild and the New Mexico Environment Department have in particular increased our capacity to conduct riparian and water quality improvements along streams and degraded wetlands throughout the project area.

The partnership between the Pueblo of Jemez and TC Company, the operator on the Stewardship Contract to feed and operate the Walatowa Timber Industries mill, ensures that our restoration efforts produce forest products that support the local economy and provide jobs for residents of the Pueblo.

A unique facet of the SWJM project area is the prevalence of cultural sites, this being the ancestral homeland of the Pueblo of Jemez and a sacred landscape for 37 other tribes. We have treated vegetation/fuels on a vast majority of the known cultural sites (primarily ruins of field houses and pueblos) on the SFNF and the Preserve to protect them from damage due to wildfires. These efforts have additionally strengthened our relationship with the Jemez Pueblo.

While the authorized 10-year funding under the CFLRP program lapsed following the 2019 federal fiscal year, restoration and monitoring work continued with previously obligated funding, appropriated funds in 2020-2024, and cooperator contributions, both financial and volunteer.

# Executive Summary

The following section provides a brief summary of the SWJM CFLRP accomplishments and monitoring results over the 15 years of the program. Details of project implementation and monitoring data follow in the subsequent chapters.

**Implementation summary:** The Southwest Jemez Mountains Collaborative Forest Landscape Restoration Program (SWJM CFLRP), led by the Santa Fe National Forest and Valles Caldera National Preserve, and partnering with 38 other collaborating agencies, tribes, universities, non-governmental organizations, and contractors, succeeded in restoring large tracts of forested watersheds during the project period of 2010-2024. Within the project area of 210,000 acres, a total of 22,495 acres of forest was thinned to reduce fuel loads and restore stand structure, along with another ~30,000 of fuels treatments. Over 30,000 acres of project land was burned in prescribed (planned) fires; however, the 2011 Las Conchas Fire and 2013 Thompson Ridge Fire burned nearly 55,000 acres of Valles Caldera before the SWJM CFLRP could be implemented. Range restoration projects covered over 41,000 acres, and streams and wetlands throughout the project area were stabilized and expanded. Invasive non-native plant species were identified and treated both manually and with herbicide. In total, 98 miles of old roads were closed and reclaimed in the project area, and 77 miles of old un-used livestock fence were removed to improve wildlife habitat and increase accessibility to recreationalists.

**Socioeconomic Monitoring:** The SWJM CFLRP sustained a strong collaboration across the life of the project centered on multiparty monitoring. During the program, an important workforce and wood processing capacity was built in collaboration with a private company and the Pueblo of Jemez. In total, 1,403 full time equivalent jobs were supported across the life of the project. $49,617,550 in labor wages were injected into the economy over the life of the project. For timber production, 206,843 tons of biomass were converted into over 13 different value-added products between 2013 and 2024. Job training programs consistently engaged local youth across the life of the project employing over 115 youth. Separately, 44 undergraduate and graduate students were also engaged with the project monitoring.

**Climate Monitoring:** The climate in the SWJM CFLRP project area reflected the warming trend found world-wide, with temperatures climbing at a rate of 1.8° F per century (exactly the rate of increase found by the Inter-Governmental Panel on Climate Change (IGPCC). The rate of temperature increase was even faster at higher elevations in the project area. Temperature increases were most pronounced during the spring and summer months, with winter monthly temperatures remaining relatively constant. The project area during the CFLRP (2010-2024) was consistently below average in precipitation, with the exception of 2015, and extreme dry years led to high-severity wildfires within the project area (e.g., the 2011 Las Conchas Fire and the 2013 Thompson Ridge Fire). Relative humidity during the project years decreased, and maximum wind speeds increased. Thus, the SWJM CFLRP occurred during a period of time that was warmer and drier than at any time during the last 110 years.

**Forest Vegetation Monitoring and Fire Risk Analysis:** The SWJM CFLRP activities were successful in reducing potential crown fire by ~50% and lowering estimated flame lengths as measured with FlamMap fire behavior model analyses. These results were consistent using canopy data derived from both LANDFIRE and LiDAR data and showed enhanced reductions within treatment areas compared to the overall landscape. From the field monitoring of forest stand structure and forest-floor herbaceous vegetation, the overall analysis showed that

treatments resulted in (1) a reduction in forest canopy closure, but well within the 50-80% goal range; (2) a reduction in tree density attributed to small diameter reductions; (3) a small reduction in basal area attributed to removing small diameter trees and retaining large trees; (4) increases in crown base height that will help prevent fire from traveling from the forest surface to tree canopies via ladder fuels; (5) reductions in total surface fuel load; and (6) increases in live native herbaceous vegetation after prescribed fire (increases in light, nutrients, and water). Finally, project implementation areas were surveyed for non-native invasive plants, and over 500 acres of project area were treated with herbicides or manual removal. Invasive "weeds" were dominated by bull thistle, musk thistle, Canada thistle, cheatgrass, oxe-eye daisy, Siberian elm, mullein and one small population of Russian knapweed.

**Watershed Restoration of Wetlands and Streams:** Stream and wetland restoration activities were conducted across the SWJM CFLRP project area by collaborating organizations and contractors using funding support acquired through grant proposals to state and federal agencies. In total for wetland restoration, 27 different types of hydrologic structures were constructed at 1,010 locations to raise water tables, spread water out into former wetland and fen environments, and slow water currents to re-establish wetland vegetation. For streambank stabilization, 191,725 willows, alders, cottonwoods and other riparian woody plant species were planted along streams, and nearly 1,000 Beaver Dam Analogs and Post-Assisted Log Structures were installed at 148 stream locations. Monitoring results found that riparian vegetation quickly responded to hydrologic treatments, expanding wetland sedge and rush communities into formerly dry meadows and grasslands. As of 2024, one stream has been removed from the New Mexico Environment Department's list of impaired streams – Jaramillo Creek's water temperatures have now dropped to recommended levels due to shading by willows planted along its banks. Beavers have begun to voluntarily re-colonize another stream (San Antonio Creek) with planning underway for beaver reintroductions to other streams in the project area.

**Wildlife Monitoring – Fish:** One of the objectives of the CFLRP was to "improve fish habitat" through landscape restoration of forested watersheds. While the streams of the SWJM CFLRP had been listed by the New Mexico Environment Department as "impaired" for high temperatures and high turbidity levels, the fish communities in the Jemez River watershed were nearly complete in their native species composition; only the Rio Grande cutthroat trout was missing, having been extirpated during the 20$^{th}$ century through the introduction of non-native trout species (German brown trout, and rainbow trout). The native non-game fish assemblage included the longnose dace, the Rio Grande sucker, the Rio Grande chub, and the fathead minnow, and these species populated their appropriate stream habitats throughout the CFLRP project area. In 2003, it was discovered that the Rio Grande sucker and Rio Grande chub were missing from the upper Rio San Antonio region, but these species were ultimately reintroduced on Valles Caldera National Preserve in 2008 and successfully established breeding populations in the following years. As such, forest restoration on the watershed was not expected to significantly affect fish communities, but monitoring activities were instituted to ensure that potential negative impacts from thinning (e.g., possible soil erosion from heavy equipment use) and increased recreation fishing activities did not cause detrimental changes in the fish populations. In 2011, right at the beginning of the SWJM CFLRP, the Las Conchas Fire burned extensive areas of the upper Jemez River watershed on the Preserve, and post-fire monsoon floods caused massive fish kills in the upper watershed – primarily of the brown and rainbow trout populations. Surprisingly, the native non-game fish species did not appear to have suffered high mortality from the floods, and their populations expanded following the temporary

elimination of the predaceous brown trout population. This "predator release" response continued for several years, but as brown trout recolonized the flooded reaches from downstream populations, the native non-game fish species declined to their pre-fire levels. Overall, stand-replacement fire at a landscape level had a much greater impact on stream fisheries than did smaller-scale forest thinning activities.

**Wildlife Monitoring – Aquatic Invertebrates:** As with the fish monitoring described above, the main emphasis for monitoring aquatic invertebrates was directed to post-Las Conchas Fire effects. Monitoring results indicated that water temperature, dissolved oxygen, and water turbidity were the important predictors of Shannon diversity and feeding group richness of aquatic invertebrates in project area streams. We learned the aquatic macroinvertebrate communities across un-burned as well as burned streams of the Preserve exhibited large inter-annual variation before and after the wildfire, leading us to conclude that in these streams, fire impacts were not substantially influencing the invertebrate communities. These temporal responses appear unrelated to fire effects and may simply reflect the overarching influence of the larger diel or daily range in stream temperature and seasonal pulses of materials that occur in Jemez Mountains streams.

**Wildlife Monitoring – Mammals**: One of the primary objectives of the SWJM CFLRP was to "improve wildlife habitat" for resident species of large and small mammals. The landscape scale forest thinning and prescribed burning were anticipated to alter vast areas of forest stand structure and increasing understory herbaceous vegetation. These actions would have changed habitat characteristics and food/shelter resources; as such, the collaborating organizations decided to undertake detailed monitoring of large mammals throughout the CFLRP project area and in the surrounding countryside. In addition, following the Las Conchas Fire in 2011, we conducted a 3-year monitoring program to assess the response of small mammals to wildfire in both ponderosa pine forests and mixed conifer forests.

The responses of large mammals to forest restoration treatments and wildfires were variable and depended on the type of disturbance, post-disturbance recovery period, and general diet and habitat use characteristics of each species. Elk, with their grass dominated diets, responded most strongly to wildfires, with selection for high severity burns, as well as prescribed fire and thinning treatments shortly after treatment. Mule deer selected recent prescribed fire and wildfire-burned areas in spring, but avoided more recently thinned areas. Mule deer strongly selected for thinned areas > 5 years old, after recovery of woody browse. Black bears selected areas with more structural complexity, providing both foraging resources and security cover. Bears generally avoided thinned areas, but did select for more recent prescribed fire areas. Bears selected for areas that were more recently burned with high severity wildfires, as well as areas experiencing high severity fires >10 years prior. Mountain lion habitat use was strongly related to the distribution and vulnerability of mule deer and elk, their primary prey in the study area. Day beds and den sites were typically found in rugged terrain that offered high levels of concealment. Overall, initial results suggest complex responses to restoration treatments and wildfire, driven mostly by species ecology related to security cover and food resources. Further analyses will account for independent and interactive effects of restoration treatments and wildfires in combination with other habitat characteristics.

For the small mammal monitoring, we sampled small mammals for three years (2012-2014) on replicated burned and unburned (control) stands in both ponderosa and mixed conifer forest types. We recorded a total of 15 small mammal species (11 rodents, one rabbit, and three shrews). Small mammals exhibited lower species richness on burned mixed conifer sites than

control sites, but species richness was only slightly lower on burned ponderosa pine sites compared to control sites. Deer mouse abundances were comparable between burned and control sites, but two species of chipmunks, woodrats, red squirrels, three species of shrews, and mountain cottontail rabbits were rare or absent in burned forests. Golden-mantled ground squirrels were more abundant on burned mixed conifer sites than control sites, and voles (*Microtus*) colonized the burned sites within three years of the fire (montane voles in 2013, and long-tailed voles in 2014). Rodent demographic data indicated mixed sex and age classes were present for the more abundant species, including reproductively active adult females; the exception was the long-tailed vole in burned ponderosa pine forests, which was represented only by male adults and subadults. Vole colonization occurred after burned sites attained 80 – 90% herbaceous cover. Given the lack of conifer regeneration, we anticipate that the small mammal community will retain its composition of meadow-grassland species for the foreseeable future.

**Wildlife Monitoring – Birds:** The SWJM CFLRP partner, Hawks Aloft, Inc., conducted avian point count surveys from 2012-2024 to evaluate avian use within eight habitat types located in Valles Caldera National Preserve and the Santa Fe National Forest. Cumulatively, over 13 years of surveys, avian density (birds/hectare) was highest in mixed conifer forest (2.51 birds/ha) and burned mixed conifer forest (2.28 birds/ha). Avian species richness was highest in riparian habitat (115 species) and ponderosa pine forest (89 species). We documented a total of 141 bird species. In a comparison of unthinned and thinned ponderosa pine forest plots, cumulative (2012-2024) avian density was significantly higher at treated points (2.27 birds/ha) than untreated (1.68 birds/ha); however, the difference was largely driven by data collected after the thinned points were subjected to prescribed fire. Density was significantly higher at thinned and burned points (2.41 birds/ha) than points that were only thinned (1.97 birds/ha). Avian richness was higher at thinned points (59 species) than unthinned points (58 species). We documented 57 avian species of conservation concern, including two species listed as threatened by the State of New Mexico: Peregrine Falcon (*Falco peregrinus*) and Gray Vireo (*Vireo vicinior*). The response of relatively common species of concern was variable among unburned and burned habitats of like type and between untreated and treated ponderosa pine forest. Among 15 relatively common species of concern, nine were more abundant in unburned forest and six were more abundant in burned forest, while eight were more abundant in unthinned forest and seven were more abundant in thinned forest. Our results suggest that the use of a combination of treatments, including prescribed fire, thinning, and areas of no treatment, conducted in patchy arrangements over time and area could improve overall avian diversity and benefit the greatest number of species.

**Wildlife Monitoring – Terrestrial Invertebrates:** Ecologists have long understood that "invertebrates rule the world" due to their incredible diversity, abundance and world-wide distribution, along with the importance of their ecosystem roles in plant herbivory, decomposition, nutrient cycling, predator-prey interactions, parasitism, and disease. As part of the SWJM CFLRP monitoring, we undertook monitoring of common groups of invertebrate taxa that typically include "pest and beneficial" insects, along with other important taxa such as spiders. We focused on three phenomena: (1) effects of the high-severity wildfire on common species of beetles, grasshoppers, ants, spiders and moths; (2) forest thinning and prescribed fire impacts on ground-dwelling arthropods; and (3) post-fire impact of the Douglas fir beetle on old-growth trees in the History Grove on Valles Caldera. Results showed that wildfire and restoration actions shifted arthropod species composition from forest obligates to grassland meadow species, although considerable overlap among communities continued to persist.

Douglas fir beetle populations expanded following wildfires, and resulted in widespread mortalities of old-growth Douglas firs; young Douglas firs were not significantly affected.

**Soils and Soil Biota Monitoring:** 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.

Soil biotic communities are responsible for a disproportionate number of ecosystem functions, including carbon and nutrient cycling, soil fertility, soil stability and water retention. Understanding how these communities change based on land management activities is a first step in understanding how ecosystem function might change. Forest treatments (thinning and use of logging machinery) can strongly impact soil abiotic parameters (soil compaction, bulk density, hydrology, organic matter, carbon, and nutrients) and important groups of soil organisms (nematodes, microarthropods, bacteria, and fungi). We addressed the effects of forest thinning and prescribed fire on soil biotic communities and abiotic parameters. Responses of soil invertebrates to forest management vary across taxa, treatment type, and disturbance intensity. Although our findings suggest that many soil invertebrates are relatively resilient or resistant to forest treatments, certain faunal groups may be negatively impacted by some management activities, especially at higher levels of disturbance.

**Threatened and Endangered Species:** The SWJM CFLRP project area supported three species listed by the US Fish and Wildlife Service as Threatened or Endangered under the Endangered Species Act. These included the Jemez Mountains Salamander (*Plethodon neomexicanus*) [Endangered], the Mexican Spotted Owl (*Strix occidentalis lucida*) [Threatened], and the New Mexican Meadow Jumping Mouse (*Zapus hudsonius luteus*) [Endangered]. Two other species have recently been added to the potential list of protected species within the SWJM CFLRP project area, but neither has as yet been observed to have permanent populations on our sites. These include the Endangered subspecies of Mexican wolf (*Canis lupus baileyi*) and the Candidate species monarch butterfly (*Danaus plexippus*). Monitoring surveys have thus far indicated that restoration activities have had little direct effect on any protected species, and that habitat improvements should promote continued occupancy in existing habitats and provide additional suitable habitat for population expansions.

**Cultural Resources:** The SWJM CFLRP included considerable activities by Cultural Resource specialists in both the Forest Service and the National Park Service. These activities

included pre-restoration survey work to meet compliance requirements of the National Historic Preservation Act (NHPA), as well as direct fuels assessments and reductions projects on cultural resource sites throughout the project area. These activities were funded by USFS and NPS as part of the cost-match requirements, as well as outside grants to evaluate restoration techniques in reducing fire risk on cultural resource sites. In total, pre-thinning and pre-burning cultural resource surveys were conducted on 26,000 acres, and fuels assessments and/or treatments were completed on 3,100 cultural resource sites (mostly ancient field houses and pueblo ruins). In addition, with funding from the Joint Fire Science Program and the USFS Rocky Mountain Research Station, archaeologists and fire specialists from the NPS and USFS evaluated drought as a key variable causing forest fires that result in a wide range of effects to archeological sites, historic structures, cultural landscapes, and traditional cultural places. They assessed impacts to archeological resources seen in the Jemez Mountains over the past forty years. This interdisciplinary effort provided the SWJM CFLRP with insights on archeological fire effects and the fuels and fire environments that cause them, leading to the implementation of cultural resource site fuels management during the 15 years of the project period.

*Acknowledgements:* The SWJM CFLRP was successful due to the efforts of all the partners in the Collaborative (Table 1). The original CFLRP proposal was led by Susan Bruin, Jen Cramer, Joe Norrell, Julie Bain, Chris Napp and Jon Williams of the Santa Fe National Forest, and Marie Rodriguez and Bob Parmenter from Valles Caldera National Preserve. Craig Allen from the US Geological Survey provided extensive scientific background on fire history and forest fuel loading in the Jemez Mountains. Anne Bradley from The Nature Conservancy assisted with Collaborative development and communications. Marie Rodriguez led the writing of the 2014 Environmental Impact Statement (EIS) for CFLRP activities on the Preserve, and Chris Napp led the 2016 EIS for the Santa Fe National Forest. Jeremy Marshall and Karl Buermeyer were the implementation coordinators for the SFNF, while Marie Rodriguez and Bob Parmenter planned and implemented projects on the Preserve; Bob Parmenter also coordinated the monitoring program. We thank all the Collaborative's participants over the years who worked in the field to implement the forest thinning activities, prescribed fires, stream and wetland restorations, livestock fence removals, road closures, invasive weed controls, and the wide scope of monitoring activities. We also thank all the authors of this report (listed in each Chapter) who contributed their findings to our collective knowledge of landscape restoration and the overall success of the SWJM CFLRP.

# Chapter 1. Implementation Summary

**Summary.** The Southwest Jemez Mountains Collaborative Forest Landscape Restoration Program (SWJM CFLRP), led by the Santa Fe National Forest and Valles Caldera National Preserve, and partnering with 38 collaborating agencies, tribes, universities, non-governmental organizations, and contractors, succeeded in restoring large tracts of forested watersheds during the project period of 2010-2024. Within the project area of 210,000 acres, a total of 22,495 acres of forest was thinned to reduce fuel loads and restore stand structure, along with another ~30,000 of fuels treatments. Over 30,000 acres of project land was burned in prescribed (planned) fires; however, the 2011 Las Conchas Fire and 2013 Thompson Ridge Fire burned nearly 55,000 acres of Valles Caldera before the SWJM CFLRP could be implemented. Range restoration projects covered over 41,000 acres, and streams and wetlands throughout the project area were stabilized and expanded. Invasive non-native plant species were identified and treated both manually and with herbicide. 98 miles of old roads were closed and reclaimed in the project area, and 77 miles of old un-used livestock fence was removed to improve wildlife habitat and increase accessibility to recreationalists.

## Section 1.1. Forest Service Implementation Accomplishments 2010-2024

Project implementation on the USFS side has continued throughout the project period and up until present day 2024, although the Covid-19 pandemic, the Mexican Spotted Owl injunction, and periodic fire restrictions played havoc with work schedules on the landscape.

Overall, restoration work in the project area on the Santa Fe National Forest (Table 1.1.1, Fig. 1.1.2) has accomplished **12,981** acres of thinning treatments with an additional **1,750** acres under contract, **29,199** acres of fuels work, **14,797** acres of planned (prescribed) burns, and **9,093** acres of managed natural fires. In addition, **143** acres of stream and wetlands have been treated, along with **486** acres of invasive plants being treated. Regarding the Stewardship Contract, there are still multiple timber task orders (Redondo, Cebollita) ahead in FY25 and FY26 and the current District Ranger and partners want to continue this valuable restoration work. There are also **41,013** acres of range restoration work accomplished, which includes fuels management, riparian exclosures, water source development, and managed/natural fires.

Table 1.1. List of Santa Fe National Forest Implementation Acres 2010-present

| Resource | Area Treated |
|---|---|
| Forestry | 12,981 Acres |
| Fuels | 29,199 Acres |
| Invasives | 468 Acres |
| Range | 41,013 Acres |
| Watershed | 143 Acres |
| Wildfire | 6,992 Acres |
| **Grand Total** | **90,797 Acres** |

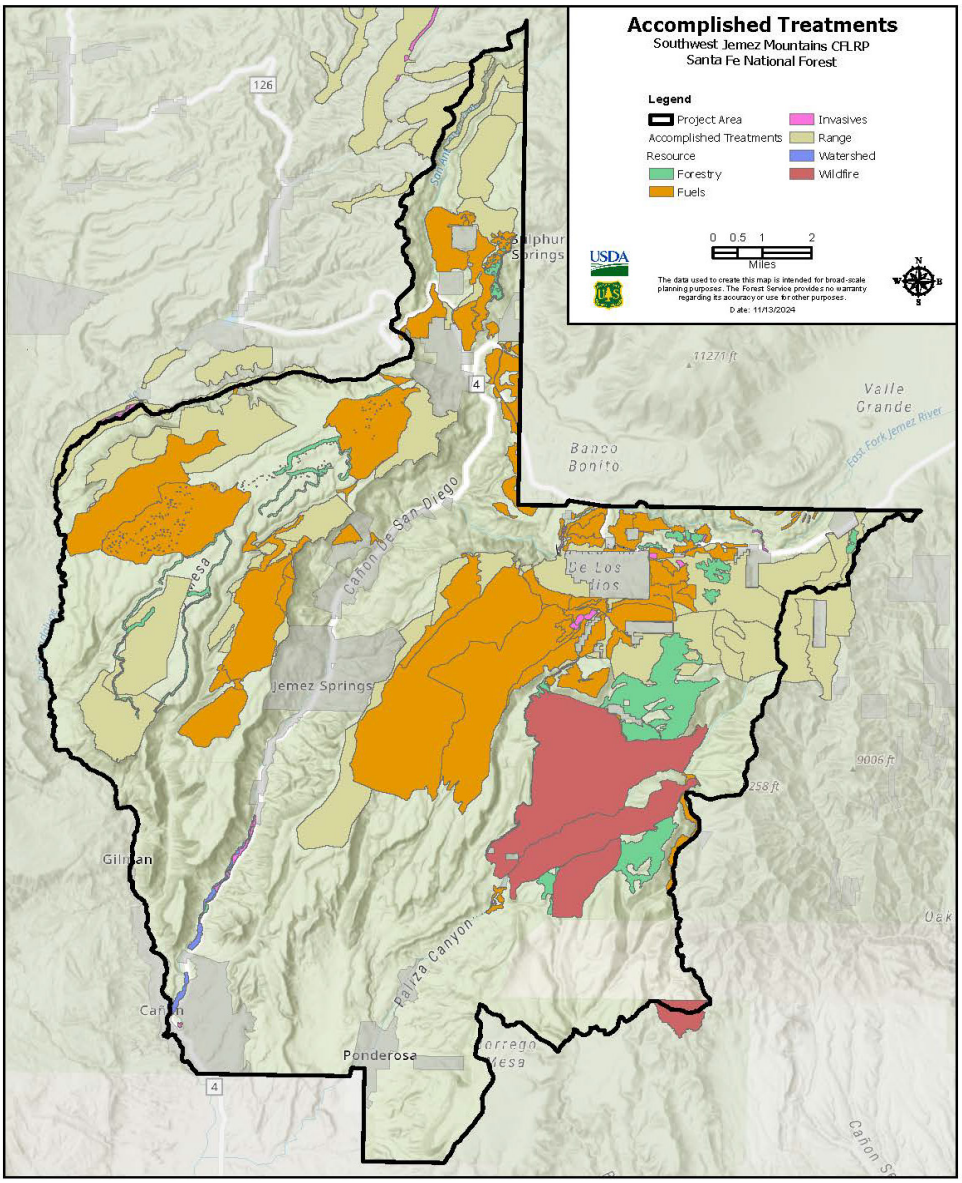



Figure 1.1.2. Map of treatment areas on the Santa Fe National Forest portion of the SWJM CFLRP project area, 2010-2024.

The East Fork and Falls Task Orders were completed in 2020, and the Pino West Task Order was completed in the fall of 2021. These treatments border on the residential areas of Bennet Lane and Sierra los Pinos, greatly reducing fire risk in this Wildland-Urban Interface (WUI) from Forest Service lands. In addition, more than 11 miles of road were decommissioned in and around the East Fork Task Order, and 3.5 miles in Pino West. Thinning in the Cerro Pelado Task Order had just wrapped up when the Cerro Pelado wildfire burned through almost the entire treated area, starting on April 22nd, 2022. Preliminary observation of a limited number of the units documented that fire behavior was affected by the treatments, causing the canopy fire to drop to the ground; however, the extremely high temperatures fueled by 50 mph+ winds scorched the bases of the trees and will likely result in future heavy mortality of a large percentage of the residual trees (Figure 1.1.3). The FTEM report showed that these treatments kept the fire out of the community of Sierra De Los Pinos during extreme fire conditions during the first day.



Figure 1.1.3. Cerro Pelado treatment unit following the Cerro Pelado wildfire. Note the untreated stand in the background with dead trees, and surviving trees in the thinned foreground. The consumed log deck indicates extreme burn conditions.

In two locations on the West Mesas, we employed a novel technique using a masticator to fell small trees and reduce slash without completely chipping them (known as “mechanical lop and scatter”) to thin and create holding lines for prescribed burns (Figure 1.1.4). Also note that skinning the bark off of the tree boles allows for rapid drying of the boles, minimizing the risk of bark beetle outbreaks from slash.




Figure 1.1.4. Left: Post-treatment of "mechanical lop and scatter" on Holiday Mesa; Right: Masticator.

A thinning project, San Juan East, has just been completed along Forest Road 10 on the eastern side of the Jemez Ranger District, where smaller trees have been thinned out and decked to provide a source of easily-accessible personal-use firewood (Figure 1.1.5). The 1,067-acre Paliza Task order was just completed in 2024.



Figure 1.1.5. Before and after photo-point, San Juan East cut/skid/deck

The use of fire was held up somewhat during the owl injunction, as well as during the height of the pandemic due to concerns around smoke complicating the effects of COVID. As conditions allowed, slash piles in the three previously mentioned task order areas were burned. Prior to the pandemic in 2019. The Tent Rocks broadcast burn obtained some excellent results on 1,300 acres of Virgin Mesa and adjacent to the WUI community of La Cueva. This burn utilized the aforementioned "mechanical lop and scatter" for establishing holding lines to great effect.

Also in 2019, a lightning strike on Canejos Peak off of Forest Road 266 provided an opportunity to manage the fire for resource benefits. Unique to this project was the fact that the planned perimeter also included the previously awarded Paliza South Task Order. T.C. Company, the contractor for all of the Task Orders, was supportive in allowing us to achieve our fuels management objectives prior to operating there.

A 2019 project in cooperation with NM State Forestry, the Natural Resources Conservation Service (NRCS), the Pueblo of Jemez, and homeowners in the Area 3 residential area north of Jemez Springs, thinned and piled fuels both on Forest Service and private land to reduce fire risk (Figure 1.1.6).



Figure 1.1.6. Thinned and piled fuels and holding line for pile burning along Forest/private boundary.

More recently, in September 2021, a 1,595-acre broadcast burn was conducted on Stable Canyon, on the west side of the West Mesas (Figure 1.1.7). This is the first prescribed burn in a Mexican spotted owl protected Activity Center (PAC) and achieved excellent effects.



Figure 1.1.7. Stable Canyon Broadcast Burn (credit A.Silva, USFS)

*Stream restoration*

Rio Grande Return, a local non-profit, committed to restoring riverscapes and ecosystems unique to the Southwest, has made excellent progress on the San Antonio Creek Restoration Project (see details in Chapter 5 below). Riparian restoration treatments have been completed, including construction of dozens of beaver dam analogs (BDAs) and instream log structures, planting of thousands of willows, the extension of an existing exclosure, four new exclosures, and control of sedimentation from the abandoned section of FR 376 (Figures 1.1.8 and 1.1.9). This work was accomplished in part with funding from the National Forest Foundation and the New Mexico Department of Game and Fish.

Following installation of the BDAs, the water table continued to rise in the surrounding terraces due to sealing of the structures by fine sediment and debris accumulation as well as through active maintenance by beaver. A significant expansion of pooling behind the structures has occurred, and the water table has risen by as much as three feet in some locations. The beavers are utilizing willow that have been planted pursuant to the current project as well as past, separate projects. Significant harvest of willow by beaver has continued, with evidence of food caches being stored within the channel during multiple site visits in 2020 and 2021. Additional plantings installed in spring 2022 helped to alleviate this pressure and over 30 acres have been planted with willow. With high levels of regeneration rates of cut willow and with the additional plantings, it is anticipated that the area will be able to support several resident beaver families within five to ten years.

Log structures installed in 2019 are functioning to achieve the intended purpose. The installations have promoted deposition of fine sediments, facilitating channel narrowing, aggradation, and the increasing of meander length. Similar designs were used in 2020 with the addition of the utilization of whole trees including the root balls. The 2020 structures provide increased vertical engagement during flood events and are designed to have high habitat value to aquatic and riparian dependent species. 2022 installations utilized similar designs as before, with the addition of Post-Assisted Log Structures (PALS) and Assisted Log Structures (ALS).



Fig. 1.1.8. Before (2020) and after (2023) images of stream restoration (BDA) in San Antonio Creek (Photos: R. Whittlesey, RGR)



Figure 1.1.9 Preconstruction photo (2020) and post construction photo (2022) of BDA in San Antonio Creek

A cooperative effort between Trout Unlimited, The Nature Conservancy, the San Diego Cattlemen's Association, and the Forest Service installed water wells on the uplands either side of San Antonio Creek (Figure 1.1.10). The objective is to provide a source of water on the uplands to reduce the tendency of cattle to migrate into the canyon and the riparian area. Pipe and drinkers will be installed this season.



Figure 1.1.10. Well site on the west upland of San Antonio Creek (Photo: A. Bishop, USFS)

*Protected Species*

Mexican spotted owl monitoring by Forest Service staff confirmed 10 (!) fledglings in 2020 and 3 in 2021 within the SWJM CFLRP footprint (Fig. 1.1.11).



Fig. 1.1.11. Adult Mexican spotted owl.

*Road Closures*

Over the SWJM CFLRP period, a total of 37 miles of old roads were closed. Many of these included treatments to prevent erosion (berms and plugs) and were allowed to naturally re-vegetate with nearby native grasses and forbs. Through an agreement with the Bureau of Land Management roads crew, storm-proofing and pipe barrier installation were conducted in Mushroom Basin in November of 2020 to reduce erosion and prevent unauthorized use of closed roads (Figure 1.1.12).



Figure 1.1.12. Pipe rail (in progress) and erosion control structure in Mushroom Basin

## Section 1.2. Valles Caldera National Preserve

*Forest thinning*

During 2010-2024, Valles Caldera National Preserve (Preserve) staff conducted 50 forest thinning projects covering 9,514 acres (Figure 1.2.1, Table 1.2.1). The overall strategy of the thinning actions on the Preserve has been to concentrate on forested areas that had not been partially burned during the 2011 Las Conchas Fire or the 2013 Thompson Ridge Fire (the yellow shaded areas in Fig. 1.2.1). In addition, the Preserve has focused on thinning the south-facing slopes of the forested domes within the caldera (as these are the warmer, drier aspects that tend to burn hotter during wildfires), as well as the Banco Bonito lava flow in the extreme southwest corner of the Preserve. Steeper slopes throughout the Preserve were thinned by hand crews using cut-and-pile or lop-and-scatter techniques (Figure 1.2.2), whereas flatter areas were thinned using mechanical equipment with removal of the logs to commercial forest products vendors (e.g., Walatowa Timber Industries at the Pueblo of Jemez, or Pete's Landscaping for firewood in Albuquerque)(Figure 1.2.3).

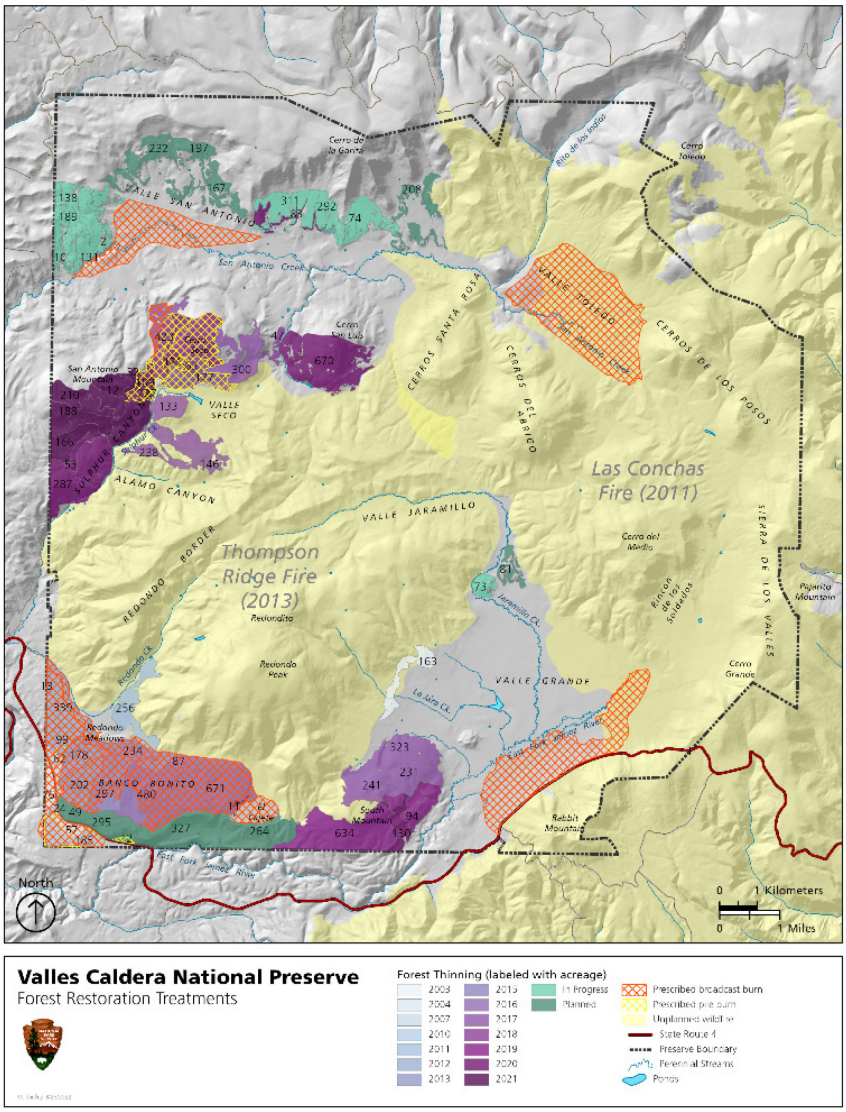



Figure 1.2.1. Map of thinning and planned fire treatments on Valles Caldera National Preserve, 2010-2024; thinning units listed as "In Progress" were finished in 2024. Areas in yellow show the burned acres of the 2011 Las Conchas Fire and the 2013 Thompson Ridge Fires.

Table 1.2.1. List of forest thinning projects on Valles Caldera National Preserve, 2010-2024.

| Thinning unit name | Year | Acres | Thinning treatment | Fuel disposition |
|---|---|---|---|---|
| Redondo Block 2 | 2010 | 339 | Hand | Cut and Pile |
| Redondo Block 3 South | 2010 | 185 | Mechanical | Haul Out |
| Redondo Block 3 North | 2010 | 12 | Hand | Cut and Pile |
| Redondo Block 1 | 2011 | 256 | Mechanical | Haul Out |
| Redondo 1B | 2012 | 99 | Mechanical | Haul Out |
| Redondo 1A | 2012 | 339 | Mechanical | Haul Out |
| Cajete 2 | 2013 | 202 | Mechanical | Haul Out |
| Cajete 1 | 2013 | 178 | Mechanical | Haul Out |
| Cajete 4 | 2015 | 234 | Mechanical | Haul Out |
| Cajete 3 | 2015 | 297 | Mechanical | Haul Out |
| Cajete Unit 3-4 Prep | 2016 | 13 | Hand | Lop and Scatter |
| Cajete 5 | 2016 | 480 | Mechanical | Haul Out |
| Cajete 7 | 2016 | 87 | Mechanical | Haul Out |
| Cajete 6 | 2016 | 671 | Mechanical | Haul Out |
| Seco 8 | 2017 | 177 | Hand/Mechanical | Haul Out/Cut and Pile |
| Seco 5 | 2017 | 112 | Hand/Mechanical | Haul Out/Mechanical Scatter |
| South Mountain | 2017 | 231 | Mechanical | Haul Out |
| Seco 4 | 2017 | 300 | Mechanical | Mechanical Scatter |
| Seco 3 | 2017 | 238 | Mechanical | Mechanical Scatter |
| South Mountain | 2017 | 323 | Hand | Cut and Pile |
| Seco 1 | 2017 | 133 | Mechanical | Mechanical Scatter |
| Seco 2 | 2017 | 146 | Mechanical | Mechanical Scatter |
| South Mountain A | 2017 | 241 | Hand | Cut and Pile |
| Seco 6 | 2018 | 63 | Hand/Mechanical | Haul Out/Mechanical Scatter |
| Seco 7 | 2018 | 47 | Hand/Mechanical | Haul Out/Cut and Pile |
| Valle San Antonio Prep | 2018 | 2 | Mechanical | Haul Out |
| South Mountain A | 2019 | 94 | Hand | Cut and Pile |
| South Mountain B | 2019 | 634 | Hand | Cut and Pile |
| Seco 9 | 2019 | 423 | Hand | Cut and Pile |
| NW Corner Vet Crew FY20 | 2019 | 10 | Hand | Cut and Pile |
| South Mountain A | 2019 | 121 | Hand | Cut and Pile |
| Hwy 4 Blowdown | 2020 | 1 | Hand | Cut and Pile |
| Banco RTRL | 2020 | 11 | Hand | Cut and Pile |
| Seco 9 | 2020 | 121 | Hand | Cut and Pile |
| South Mountain A | 2020 | 130 | Hand | Cut and Pile |
| San Antonio Mtn South 8 | 2020 | 109 | Hand | Cut and Pile |
| San Antonio Mtn South 2 | 2020 | 234 | Hand | Cut and Pile |
| San Antonio Mtn South 1 | 2020 | 287 | Hand | Cut and Pile |
| San Antonio Mtn South 3 | 2020 | 53 | Hand | Cut and Pile |
| Cerro San Luis | 2020 | 670 | Hand | Cut and Pile |

| | | | | |
|---|---|---|---|---|
| North Rim B | 2020 | 11 | Hand | Lop and Scatter |
| North Rim A | 2020 | 83 | Hand | Lop and Scatter |
| San Antonio Mtn South 5 | 2021 | 227 | Hand | Cut and Pile |
| San Antonio Mtn South 4 | 2021 | 112 | Hand | Cut and Pile |
| San Antonio Mtn South 7 | 2021 | 32 | Hand | Cut and Pile |
| San Antonio Mtn South 4 | 2021 | 188 | Hand | Cut and Pile |
| San Antonio Mtn South 7 | 2021 | 112 | Hand | Cut and Pile |
| San Antonio Mtn South 6 | 2021 | 210 | Hand | Cut and Pile |
| San Antonio Mtn South 3 | 2021 | 166 | Hand | Cut and Pile |
| Cerro Piñon | 2022 | 70 | Hand | Cut and Pile |
| | TOTAL: | 9,514 | | |



Figure 1.2.2. Hand-crew thinning of Valles Caldera forest units with cut-and-pile prescriptions.




Figure 1.2.3. Mechanical thinning on Valles Caldera forest restoration units. Feller-buncher w/processor creates decks, that are transported to Walatowa Timber Industries mill for commercial wood products.

*Fire management*

Fire crews (NPS Pueblo Parks Fire Team and collaborating agencies) implemented 23 planned (prescribed) fires during the CFLRP period, composed of a mix of post-thinning broadcast burns and pile burns; area of burns totaled 6,804 acres (Table 1.2.2, Figure 1.2.4). The Preserve has updated its Fire Management Plan in 2023-2024, with an Environmental Assessment having been written and opened for public comments in 2024. The final decision is being reviewed by the NPS Regional Office this winter, and a Finding of No Significant Impact (FONSI) is anticipated in early 2025. The updated fire management plan will emphasize the use of future managed fires from lightning strikes for resource benefit to facilitate the necessary maintenance of restored forested watersheds and return the landscape to its former fire regime and fire return intervals.

Table 1.2.2. List of planned (prescribed) burns on Valles Caldera National Preserve, 2010-2024.

| **Fire name** | **Year** | **Acres** | **Fire type** |
|---|---|---|---|
| Redondo Piles | 2011 | 14 | Rx piles |
| Redondo Piles | 2011 | 12 | Rx piles |
| Redondo Meadow | 2011 | 179 | Rx broadcast |
| SW Banco | 2012 | 339 | Rx broadcast |
| North Banco Cajete 2 | 2015 | 202 | Rx broadcast |
| North Banco Cajete 1 | 2016 | 178 | Rx broadcast |
| North Banco Cajete 3 | 2016 | 244 | Rx broadcast |
| North Banco Cajete Hwy South | 2016 | 34 | Rx broadcast |
| Valle Grande 2016 | 2016 | 1,251 | Rx broadcast |
| North Banco Cajete 5-6 | 2018 | 1,164 | Rx broadcast |
| Valle San Antonio | 2018 | 845 | Rx broadcast |
| Seco Piles | 2018 | 240 | Rx piles |
| Seco 5 | 2019 | 90 | Rx broadcast |
| North Banco Redondo A+B | 2019 | 379 | Rx broadcast |
| North Banco Redondo A escarpment | 2019 | 108 | Rx broadcast |
| North Banco Cajete 4 | 2019 | 175 | Rx broadcast |
| Seco 9 Piles | 2020 | 422 | Rx piles |
| Cajete Meadow Rx | 2021 | 151 | Rx broadcast |
| San Antonio Mtn 8 Piles | 2022 | 109 | Rx piles |
| Seco 9 Piles | 2022 | 121 | Rx piles |
| San Antonio Mtn 5 Piles | 2022 | 16 | Rx piles |
| San Antonio Mtn 1-7 Piles | 2023 | 235 | Rx piles |
| South Mountain | 2023 | 296 | Rx piles |
| | | | |
| TOTAL: | | 6,804 | |



Figure 1.2.4. Fire crews conducting prescribed fires on Valles Caldera National Preserve.

The objectives for the thinning projects were to not only reduce the risk of high-severity stand-replacing fires by reducing the fuel loads, but also to increase the water-balance in the remaining trees by reducing inter-tree competition for water, nutrients and sunlight. In addition, we wanted to encourage an extensive herbaceous sub-canopy layer to provide forage for wildlife (elk, deer, and small mammals) along with increased insect populations for other wildlife (birds, small mammals, reptiles and amphibians). In nearly all cases of thinning and prescribed fires, these objectives were successfully realized; an example is provided below of repeat photos from one of the ponderosa pine stands (Redondo meadow) that began as a dog-hair thicket of young pines, and progressed through mechanical thinning, prescribed fire, and herbaceous plant community development (Figure 1.2.5).




Figure 1.2.5. Time-series photographs of a ponderosa pine thinning site near Redondo Meadow on Valles Caldera National Preserve. Site was mechanically thinned with a feller-buncher, and logs were trucked to the Walatowa Timber Industries mill at the Pueblo of Jemez. Slash was then masticated (2010) and burned (2011). Herbaceous regrowth achieved 100% cover by 2017.

*Streams and Wetlands Restoration*

Using outside funding from grants procured by collaborating partner organizations, a total of 27 different types of hydrologic structures were constructed at 1,010 locations to raise water tables, spread water out into former wetland and fen environments, and slow water currents to re-establish wetland vegetation; over 500 acres of wetlands were restored on Valles Caldera National Preserve. For streambank stabilization, 116,725 willows, alders, cottonwoods and other riparian woody plant species were planted along Preserve streams, and Beaver Dam Analogs and Post-Assisted Log Structures were installed at 20 stream locations. See Chapter 5 for details.

*Improving Wildlife Habitat – Removing Old Livestock Fences*

Valles Caldera National Preserve has had a long history of livestock grazing, both under private ownership in the 19th and 20th centuries, and under the Valles Caldera Trust and National Park Service. During the 1960s and 1970s, the owners of the Baca Ranch installed many miles of livestock fences; when the land was purchased by Congress in 2000, the livestock stocking rate was reduced to ensure "light grazing" on the Preserve's grasslands and forests. By 2009, the management program had developed sufficiently that many of the old livestock fences were no longer needed, and the remaining fences would be converted to "wildlife friendly" fences (with smooth wire on the top and bottom strands instead of barbed wire – this would reduce injuries by wildlife attempting to pass under or over the fences). Numerous wildlife injuries and deaths had been documented on both Preserve and National Forest lands, so a campaign was begun on Valles Caldera to remove un-used fences. Between 2009 and 2020, a total of 77 miles of internal fences were removed and recycled at metal-recycling centers, greatly reducing wildlife fatalities and removing hazardous impediments to hikers, anglers, and equestrian recreationalists (Figure 1.2.6). Collaborating volunteer organizations, university and high school students, Boy Scouts, church groups, YCC crews and contractors all participated in the fence-removal program.



Figure 1.2.6. Fence removal reduces wildlife mortality and injuries (above); volunteers remove fences (top right) while an elk herd with calves watches from the distance. Once the fence is gone (right), wildlife and recreationalists have unimpeded movements on the landscape.

*Road Closures*

During the period of the SWJM CFLRP, Valles Caldera closed a total of 47 miles of old administrative and logging roads. Most of these roads were no longer necessary for Preserve management activities, but some were closed to due post-Las Conchas and post-Thompson Ridge fires damages, particularly from fallen trees burned in the fires. Fallen logs were left in place to slow soil erosion. Other roads that were closed received erosion control treatments (berms or plugs).

*Gas Pipeline Retirement and Reclamation*

The New Mexico Gas Company (NMGC) had operated a natural gas pipeline across the Preserve since the late 1940s, and in 2021, the company replaced the pipeline with a new pipeline running well outside the Preserve boundaries. As a result, the NMGC worked with the Preserve to shut down the pipeline, plug sections under streams with concrete, and reclaim the access road across the Preserve. A total 14.5 miles of pipeline access roads were reclaimed in 2023-2024, with various structures (berms, plugs, run-out ditches) to control erosion and reseeding and harrowing with native grass and wildflower seeds collected from adjacent grasslands. Particular attention was paid to stream crossings, where rock rip-rap was installed to prevent downcutting erosion and allow stable wetland soils to be colonized by wetland sedges. Initial responses of local vegetation succession on the reclaimed roads appears to be developing well.

# Chapter 2. Socioeconomic Monitoring

**Collaborators:** Gabe Kohler, Eytan Krasilovsky – *Forest Stewards Guild*.

**Summary: Key Socioeconomic Monitoring Highlights**

- Strong collaboration across the life of the project centered on multiparty monitoring.
- Important workforce and wood processing capacity built.
- 1,416 full time equivalent jobs supported across the life of the project.
- $51,040,128 in labor wages injected into the economy over the life of the project.
- 206,843 tons of biomass were converted into over 13 different value-added products between 2013 and 2024.
- Job training programs consistently engaged local youth across the life of the project employing over 115 youth. Separately 44 students were also engaged with the project.

**Introduction**

This report builds upon the Southwest Jemez Collaborative Forest Restoration Program 10-Year Socioeconomic monitoring report, written and released in 2020 (Kohler and Piccarello 2020). The purpose of this report is to document the socio-economic impacts of the SW Jemez CFLR over the last fifteen years to local communities and stakeholders. Socio-economic monitoring efforts to date have focused on evaluating the project's progress towards the CFLR goals of "identifying the cost of forest restoration, stabilize local harvesting and wood processing businesses, increase wood utilization, track investments in local economies with jobs and other project activities, and track perceptions of collaboration" (PL 111-11, Sec. 4003(c)).

The SW Jemez CFLR was funded in 2010 and the socioeconomic monitoring agreement between the Forest Stewards Guild and Santa Fe National Forest was put in place in 2013. This lag between the project's start and the start of socioeconomic monitoring creates a gap in the data from 2010-2013. This gap limits the amount of pre-implementation data for the project and the socioeconomic monitoring in general. Additionally, although the monitoring was mandated to continue through 2024, the Guild's original agreement for socioeconomic monitoring ended in 2019. A modification was completed in 2021 that ended in 2023. Due to the uncertainty around funding and years where there was not adequate funding, the Guild took a more streamlined approach to socioeconomic monitoring in 2019-2024 after the project sunset. This report includes data that was collected throughout the entire monitoring period 2013-2024 as well as data that was collected until the project sunset, between 2013-2019.

**Socioeconomic monitoring methods**

A multiparty monitoring and adaptive management plan was collaboratively developed and implemented by the numerous stakeholders of the SW Jemez Landscape Collaborative. The plan contains comprehensive monitoring and evaluation guidelines to support the SWJM landscape strategy.

Monitoring social and economic change in response to restoration activities is a required component of all multiparty monitoring plans. In the SW Jemez CFLR, socioeconomic monitoring topics were identified from three main sources, listed in order of priority of completion:

1. Reporting on jobs and economic indicators as specified by the CFLR National Outcomes and Indicators (Aplet et al. 2011). These indicators include the following requirements:
    a. Use of the **Treatments for Restoration Economic Analysis Tool (TREAT)** to calculate economic contribution and economic impact analyses for restoration as a result of the CFLR activities.
    b. Direct monitoring of project partners to supplement TREAT through surveys and interviews covering various socioeconomic indicators.

2. Monitoring the socioeconomic goals from the Act
    a. Reduction in costs including: restoration thinning, wildfire risk reduction costs savings, prescribed fire, and wildfire suppression.
    b. Collaboration monitoring of the diversity of stakeholders, strength of collaborative group, and tracking the collaborative process over time.
    c. Woody biomass utilization.
    d. Direct and indirect benefits to local economies.
    e. Positive and negative social and economic effects of the project.

3. Social and economic monitoring as identified in the SWJM monitoring proposal.
    a. Effects on recreation use
    b. Changes in ecosystem services
    c. Training and education efforts

TREAT is a standardized method that was specifically developed by the USFS for comparison across all CFLR projects for capturing economic job data (USDA, 2015). TREAT provides CFLR projects with a standard interface to estimate employment and labor income impacts from proposed restoration activities, and its use is required for work plans and annual reports.

TREAT estimates regional economic impacts of monitoring services based on inputs of dollars invested and volume of wood removed. While a useful tool for comparing CFLR projects across different regions, the multiparty team felt it important to also track socioeconomic indicators directly through surveys and interviews with project partners. The ancillary impact of restoration activities, such as additional contracts awarded outside the landscape and number of individuals employed (in addition to full time equivalent) is an important part of the story of the benefit that the CFLR has for local communities and economies. Many of these ancillary benefits are outlined in the annual reports prepared by the Forest Stewards Guild and the Santa Fe National Forest.

Socioeconomic monitoring was conducted annually through surveys and follow-up interviews with project partners. Surveys were administered via email to a member contact list developed provided by USFS and VCNP staff. The VCNP (managed from 2002-2015 through the board of the Valles Caldera Trust under USDA) coordinated the majority of the project monitoring and in October 2015 had its management transferred by Congress from the USDA Trust to the National Park Service. This shift ended the ability of the VCNP to receive CFLR funds for restoration projects, as the Preserve lands were no longer National Forest System lands. However, due to the longstanding partnerships in the SWJM landscape, VCNP has continued their on-the-ground

restoration work and monitoring activities through additional funding from the Department of Interior and National Park Service, and the Collaborative continued to see the project at the landscape scale across jurisdictions. A Service First inter-agency agreement between the Santa Fe National Forest and VCNP provided additional monitoring support to VCNP to continue conducting field monitoring work on the SFNF portion of the project area. As a result, the work in the VCNP is included in the socioeconomic monitoring.

Partners were asked to report the location and type of work they completed, number of employees, and the amount spent on labor and non-labor costs. Annual reports combined data from partner surveys with qualitative descriptions of project outcomes derived from follow-up interviews.

Project partners are engaged in the following kinds of activities: harvesting and wood processing, prescribed fire (implementation and preparation) and wildfire suppression, forest restoration, cultural resource protection, ecological monitoring, youth engagement, and watershed and stream restoration. To maintain the ability to compare data over time, a similar set of questions were asked of project partners. However, through an adaptive management process, Forest Stewards Guild staff experimented with different survey methods to try and improve the survey response rates and quality of data. Ultimately, the most effective though time consuming method was direct contact through phone or in-person interviews with project partners. In previous years, a Google form was used to survey partners about their project activities.

Wood utilization is an integral component of the CFLR program so a specific set of questions was asked of the sawmill operator, Walatowa Timber Industries (WTI), to track how woody biomass is being utilized. One of the most common challenges with collecting data from WTI was that units of measurement and periods of performance did not always match up with how TREAT and federal fiscal years are measured. While a standard set of indicators were used to inform questions asked of WTI, often the survey and interview method devolved into a more freeform note taking and transcription session. A common response to a question asked was “I don’t track things that way.” Ultimately, accurate data was reported but made the use of a consistent survey method difficult. Despite these challenges, we collected annual data on these key metrics from WTI:

- Amount of material harvested,
- Volume of material sold,
- Location of purchasers,
- Products created, and
- Market values of wood products

In order to assess the impact of restoration activities on the local communities, we collected annual data from restoration and monitoring partners, including WTI, on the following indicators:

- Type of restoration or monitoring work conducted,
- Contracts and sub-contracts,
- Acres or miles restored/monitored,
- Labor hours invested,
- Labor versus non-labor costs,
- Travel costs and percentage spent locally, and
- Materials purchased and percentage spent locally.

Partner surveys and interview questions were designed based on recommendations from the National Forest Foundation early CFRL work, information from the Colorado Forest Restoration

Institute, and from the Forest Stewards Guild's programmatic experience in New Mexico (Barret et. al., 2018). Two surveys were used – one for harvesting and wood processing and the other for all other restoration activities including monitoring. Not all respondents were able to answer all questions and many respondents estimated to the best of their ability.

As previously mentioned, in FY15 the Valles Caldera National Preserve (VCNP) was transferred by Congress to the National Park Service. VCNP continued to participate as an active member of the Collaborative, continuing its lead role in the monitoring program and data sharing among participants. Following this transition, VCNP received annual funding from the National Park System that replaced USDA CFLR funding. This funding was tracked as leveraged partner funding.

TREAT and partner surveys (including the interview findings) are complimentary data sets. The data collected directly from partners should not be considered in addition to TREAT. For example, labor costs and wages from TREAT and partner surveys and interviews should not be added together to arrive at a total wages number generated from the SW Jemez CFLR for a given year. Rather TREAT and direct monitoring of partners helps creates a fuller picture of the socioeconomic impact of the CFLR. Partner surveys capture qualitative and quantitative data needed to contextualize the socioeconomic impacts modelled by TREAT and helped to characterize the structure of the local economy for the TREAT model. These complementary data sets provide two different methods and types of data for understanding the same socioeconomic impacts – TREAT providing a general estimate of economic impacts across counties, and partner surveys providing reported economic impacts in the context of specific businesses that were involved in the SW Jemez CFLR.

## Section 2.1: Employment and Wages

*Modeled Employment and Wages - TREAT*

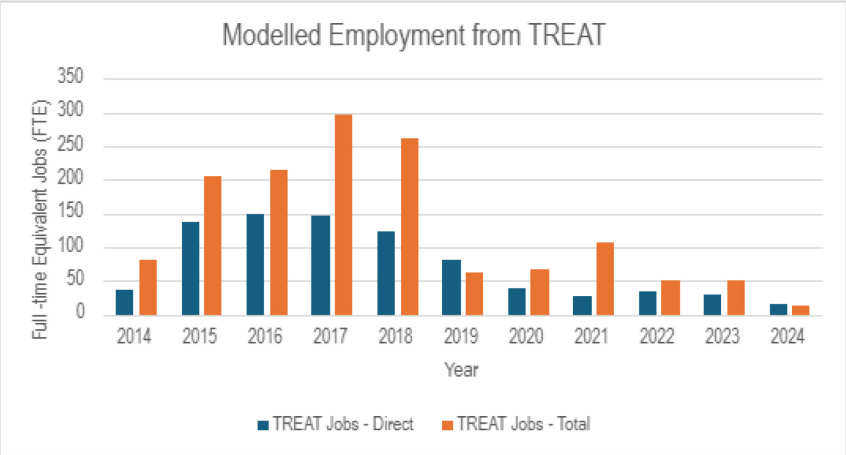



Fig. 2.1. Modelled employment from 2014-2024 from TREAT.

Over the life of the CFLRP, total modeled employment increased steadily between 2014 and 2017 and direct employment increased steadily between 2014 and 2018. Between 2017 and 2018, investment of CFLN funds decreased by 34%, which correlates with the decrease in modelled employment. After the project sunset, between 2019 and 2024, modeled employment remains consistent with slight peak in total and direct employment in 2021and 2022, respectively. The modeled employment levels for project sunset years, 2019 -2024, are similar to 2014, before much of the investment in project implementation occurred. This pattern in the modelled employment data shows a strong signal of CFLN investment and the timber volumes associated with implementation (Figure 2.1).

The major peak in total employment in 2017 and 2018 correlates closely with the establishment of the stewardship contract with Walatowa Timber Industries in 2016. The establishment of the stewardship contract generated greater volume of timber into the local sawmill and likely contributed to a higher amount of modelled employment through TREAT. According to the TREAT model, over the life of the project, the SW Jemez directly contributed to 822 full-time equivalent jobs in the projects area of interest (Rio Arriba, Sandoval, and Bernalillo Counties). When all matching funds and additional timber volumes are considered, the SW Jemez CFLRP contributed to a total of 1,416 full-time equivalent jobs in the area of interest, according to the TREAT model.

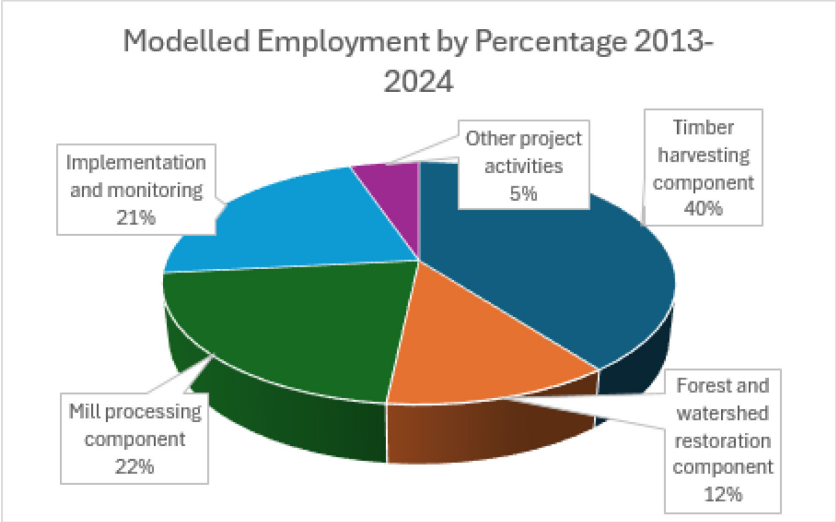



Fig. 2.2: Modelled employment percentages by sector 2014-2024 from TREAT

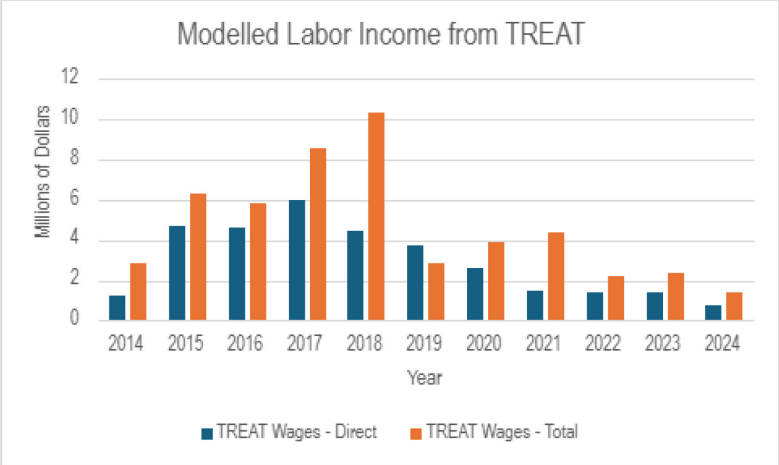



Fig. 2.3: Modelled labor income from 2014-2024 from TREAT

Modeled labor income from TREAT shows a similar pattern to modeled employment with peaks in 2017 and 2018. According to the TREAT model, the SW Jemez CFLRP directly contributed to $32,474,468 in labor wages in the area of interest. When all matching funds and additional timber volumes were input into the TREAT model, the SW Jemez CFLRP is estimated to support a total of $51,040,128 in labor wages in the area of interest over the life of the project (Figure 2.3).

*Observed Employment*

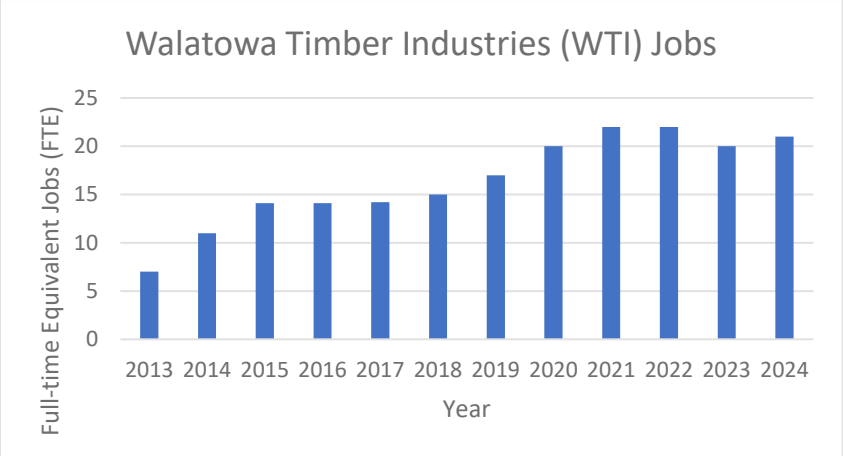



Fig. 2.4: Reported employment at WTI from 2013-2024

The decrease in modelled labor income and employment around 2018 (34% decrease in CFLN funding) is not reflected in observed data of employment from Walatowa Timber Industries (WTI) (Figure 2.4). This indicates that although CFLRP investments decreased in 2018, this did not correspond with a decrease in sawmill employment as the TREAT model would expect. This finding indicates that the increase in sawmilling employment associated with CFLRP funding was sustained longer than expected. This may be due to a lag effect between investment and its effects on employment, or it may signal that Walatowa Timber Industries sustained the initial increase in employment by accessing additional timber volume that was not associated with the CFLRP.

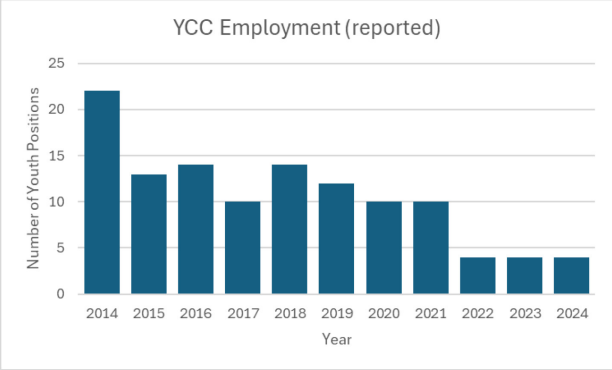



Fig. 2.5: Reported youth employment from 2014-2024



Fig. 2.7: Forest Stewards Guild Youth Corps in SW Jemez CFLRP project area

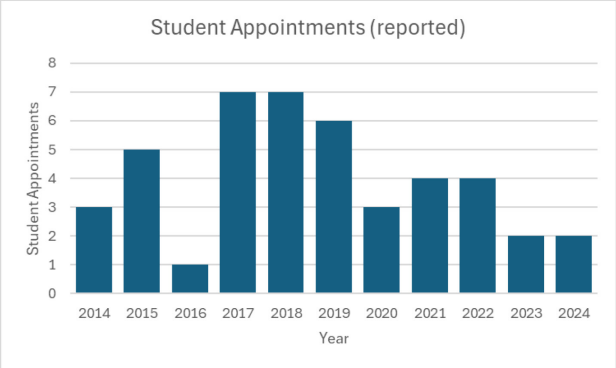



Fig. 2.6: Number of graduate and undergraduate researchers employed by project partners for monitoring from 2014-2024

In addition to monitoring the amount of employment that the SW Jemez CFLRP generated across the region, annual surveys of contractors working in the project landscape provided information about the quality of employment supported by the CFLRP. Employment at Walatowa Timber Industries and with Conley Enterprises is conveniently located near Jemez Pueblo and Jemez Springs, where the majority of employees live. This proximate location reduces commute times, allowing employees to spend additional time with their families, have decreased fuel and transportation costs, and experience reduced risk associated with highway travel. CFLRP monitoring and implementation contracts supported a high proportion of youth employment and employment of graduate-level researchers from various academic institutions. The youth and academic employment generated by the SW Jemez CFLRP support workforce development objectives. By investing in the training and onboarding of youth and researchers through the SW Jemez CFLRP, New Mexico businesses may benefit from reduced training costs for future project work, if workers are able to find future opportunities.

## Section 2.2: Restoration Costs

The cost of forest restoration through mechanical treatments and wild or planned ignitions is a key monitoring indicator in the CFLR (Table 2.1). For the SW Jemez CFLR, we did not collect data on restoration costs in the SW Jemez project landscape before project implementation to compare restoration costs against. However, because the EIS for the project was not signed until 2015, we use an average of 2010 and 2011 to serve as a baseline of restoration costs before implementation began. This average was $460/acre for mechanical treatments. There was limited pre-2013 data to establish a baseline average of restoration costs through planned and unplanned ignitions, so we did not compare costs before and after project implementation. Additionally,

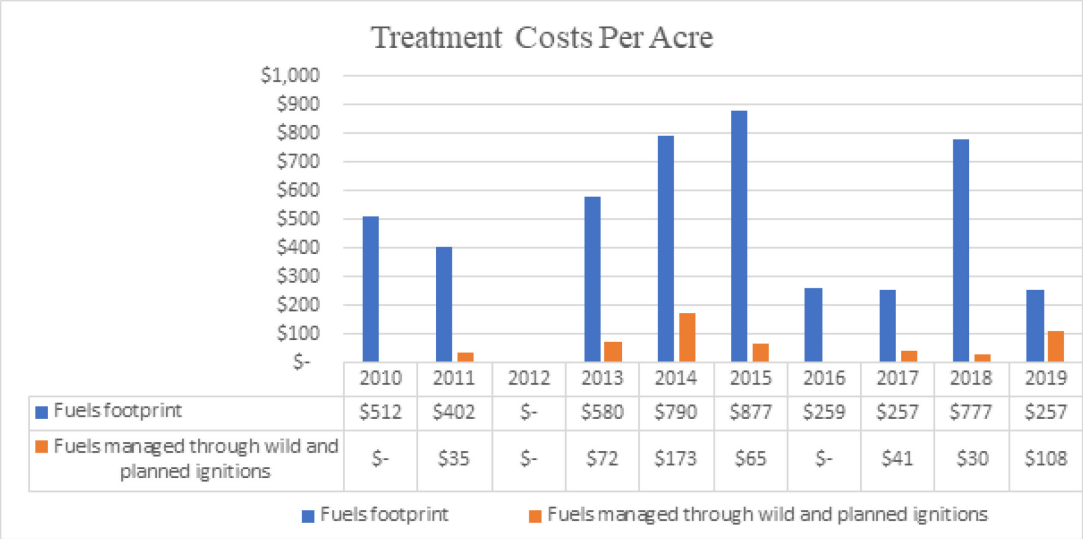
annual reports were not completed for years after the project sunset, 2019-2024, so this analysis does not include information from those years.

Table 2.1: Range of treatment costs from 2013-2019 data

| | Average | High | Low | Median |
|---|---|---|---|---|
| Mechanical Treatments (per acre) | $ 523 | $ 877 | $ 257 | $ 512 |
| Fuels managed through wild and planned ignitions (per acre) | $ 75 | $ 173 | $ 30 | $ 65 |
| Stream restoration (per mile) | $ 9,912 | $ 33,587 | $ 1,083 | $ 5,757 |
| Road decommissioned (per mile) | $ 6,364 | $ 3,700 | $ 15,000 | $ 5,102 |

Compared to pre-2013 restoration costs, the SW Jemez CFLR experienced at least a 44% decrease in restoration costs through mechanical treatments in 2016, 2017, and 2019. However, in 2014, 2015, and 2018, the cost of restoration through mechanical treatments increased by at least 68% each year. Although the 10-year average of restoration costs through mechanical treatments did not decrease substantially from pre-implementation years (2010-2012), we learned that forest restoration in dry forests should be evaluated in terms of the wildfire and climate outcomes of the treatments (do we still have forests persisting) and secondarily based on the tradeoffs of investing in mechanical, prescribed, and managed fire treatments. However, this does not indicate that program dollars were not well invested – metrics like cost per unit treated do not account for the longer-term value of having successful local operators and infrastructure. Rather, it highlights that several factors can influence restoration costs such as available NEPA cleared acres, harvest design, prescription factors, weather, and travel and haul distances. Furthermore, site conditions can significantly impact the cost per acre for mechanical treatments. Steeper, more rugged acres will take longer to accomplish and result in greater wear and tear of equipment. An average of $523.44 per acre cost for mechanical treatments is very reasonable for the analysis time period compared with similar programs such as the Forest Health Initiative (FHI) program funded by New Mexico State Forestry for treatments on private land. Depending on site conditions, FHI treatments did cost between $550 and $1,050 per acre in 2018. That comparative cost included some slash management but did not include the development of a forest management plan or recognize mobilization costs.

For other restoration activities like stream and watershed restoration, site conditions are too variable to make comparisons from year to year. For example, riparian restoration includes a wide range of activities such as planting vegetation, use of heavy equipment to restore flood plains and induce meandering, and labor intensive – but using material sourced from the site – erosion control methods.




| | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 |
|---|---|---|---|---|---|---|---|---|---|---|
| Fuels footprint | $512 | $402 | $- | $580 | $790 | $877 | $259 | $257 | $777 | $257 |
| Fuels managed through wild and planned ignitions | $- | $35 | $- | $72 | $173 | $65 | $- | $41 | $30 | $108 |

Fig. 2.8: Restoration costs through mechanical treatments and wild and planned ignitions per acre for each project year of the SW Jemez CFLR.

Figure 2.8 shows the fluctuation in restoration costs throughout the SW Jemez CFLR and demonstrates the cost effectiveness of restoration using wild and planned ignitions. The cost of fuels treatments ranged from a low of $257 an acre in 2017 to a high of $877 dollars an acre in 2015. However, it should be noted that due to funding for some treatments being paid for over multiple years and using multiple agreements, it was difficult in some years to pin down an exact cost per acre of treatments for some years. Furthermore, Forest Service reporting can account for the same acre more than once using different metrics. For example, an “acre of wildland urban interface” treated might also be counted as an “acre mitigated” for fuels reductions. Every effort was made to avoid double counting of acres and missed costs of treatments. Due to the potential miscalculation in annual cost per acre figures, the average figure for treatments between 2010 and 2019 of $523 is the more useful figure to consider. The cost of treatment through wild and planned ignitions ranged from a low of $35 an acre to a high of $173 an acre. Although 1,702 acres were mechanically treated by VCNP and SFNF in 2012, we were not able to identify how much these treatments cost in order to determine a cost per acre for 2012. Unlike other project years, the annual report in 2012 did not specify the total treatment costs for annual accomplishments.

## Section 2.3: Wood Processing and Utilization

From 2013-2019 the Forest Service sold 93,353 CCF of merchantable timber from the Santa Fe National Forest through the SW Jemez CFLRP. Volume output spiked in 2016, which correlates with the completion of the stewardship contract with Walatowa Timber Industries. The largest years of volume output took place in 2016 and 2017 (Figure 2.9).

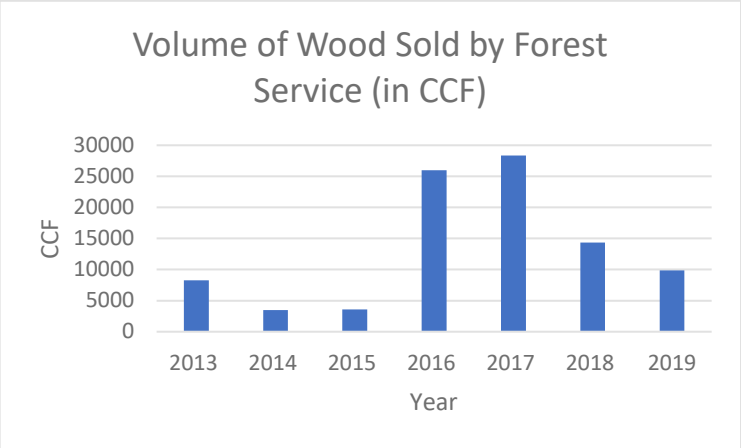



Fig. 2.9: Volume of wood sold from SW Jemez CFLRP acres from 2013-2019

In addition to merchantable timber, the SW Jemez generated over 100,000 green tons of woody byproduct from small diameter trees and other material that was available for value-added use. Fluctuations in biomass show some correlation with the volume of timber sold. For both biomass and volume of timber sold, 2016 and 2017 were peak years (Figures 2.9 and 2.10). Walatowa Timber Industries (WTI) invested heavily in a pellet mill in 2016 to capture some of this biomass for a value-added wood product (Figure 2.11). The pellet mill that WTI installed has a capacity of 32 tons per day. At this rate of processing, the SW Jemez Project generated enough biomass to supply the pellet mill with feedstock for over 8 years of constant operation.

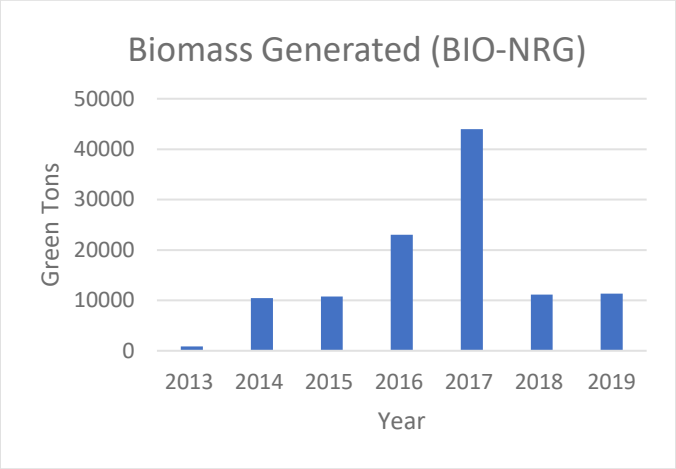



Fig. 2.11: Biomass generated by restoration treatments within the SW Jemez CFLRP. Measured in green tons.

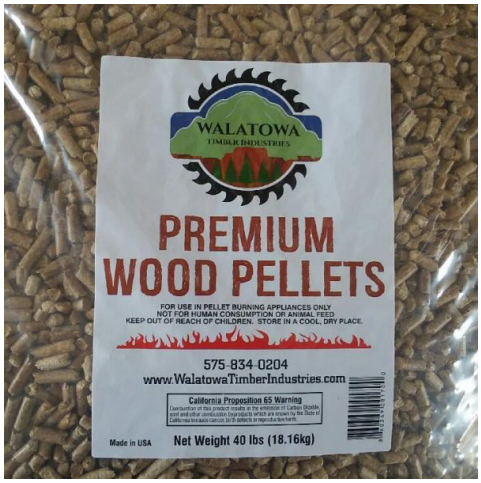



Fig. 2.10: Wood pellets manufactured and sold by Walatowa Timber Industries

WTI, a partnership between TC Company and Jemez Pueblo, completed an average of 794 acres of mechanical thinning a year for an approximate total of 9,537 acres of mechanical treatments between 2013 and 2024. Walatowa Timber Industries remains the primary sawmill to process woody biomass from the CFLR landscape. Between 2013 and 2019, an average of 95% of material was claimed for value added use, while an average of 5% was left for wildlife habitat.

These mechanical treatments generated an average of 18,804 tons of sawlogs and biomass per year for an approximate total of 206,843 tons of wood that were converted into various value-added products between 2013 and 2023 (Figures 2.12, 2.13; Table 2.2).

To remain competitive, WTI invested heavily in their infrastructure throughout the life of the CFLRP and owner Terry Conley attributed these investments to being awarded the stewardship contract from the SW Jemez CFLRP. In a 2024 interview with Conley, he described the long-term stabilizing effect that the SW Jemez CFLRP had on WTI, allowing him to invest and innovate in his wood processing infrastructure in ways that will benefit the restoration economy of Northern New Mexico long after the CFLRP has sunset.

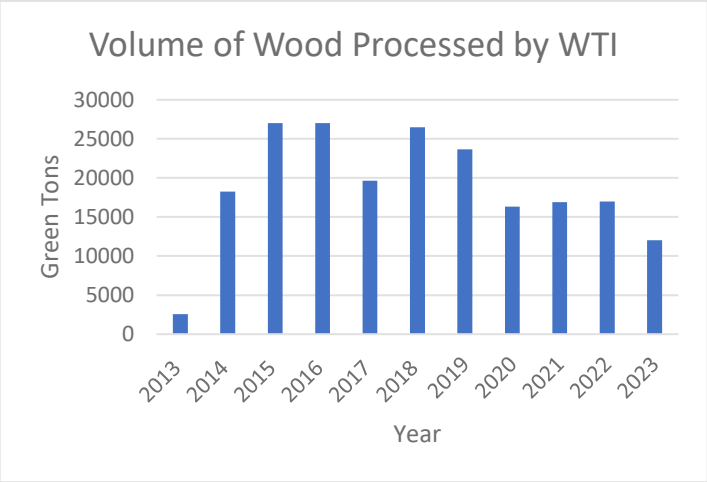



Fig. 2.12: Volume of value-added wood products processed by Walatowa Timber Industries (WTI)

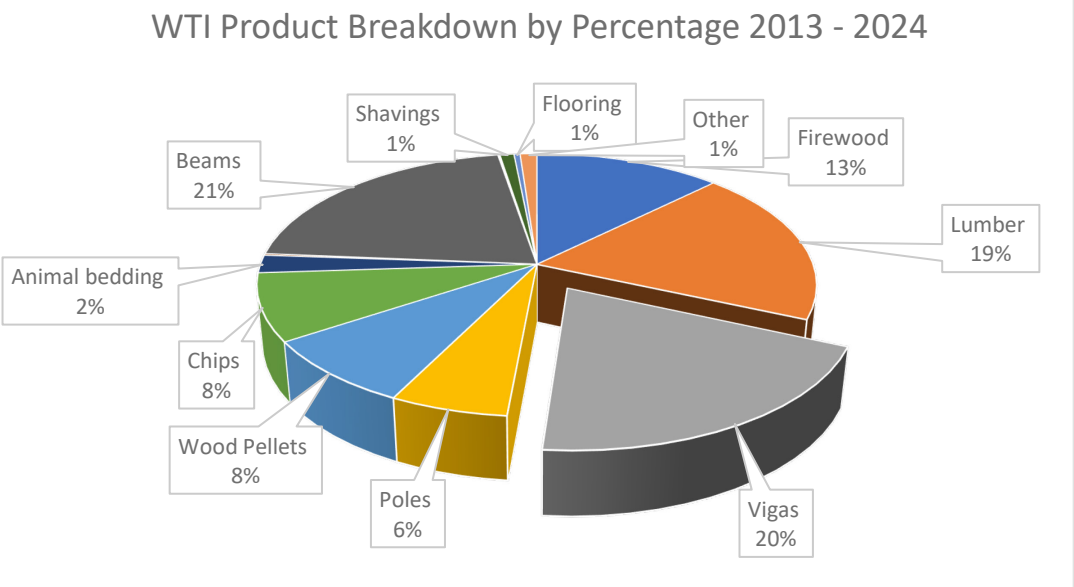



Fig. 2.13: Product breakdown by percentage of Walatowa Timber Industries (WTI).

Table 2.2: Value-added wood product volumes by percentage for Walatowa Timber Industries from 2013-2024. Measured in green tons.

| Value Added Wood Products Produced by WTI (in tons) | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | Total |
| Firewood | 103 | 915 | 4,050 | 14,850 | 785 | 1,059 | 710 | 816 | 1,183 | 895 | 576 | 25,942 |
| Lumber | 798 | 1,262 | 5,670 | 9,990 | 3,294 | 4,444 | 3,078 | 1,959 | 3,515 | 2,264 | 3,213 | 39,487 |
| | | | | | | | | | | | | |
| Vigas | 489 | 5,307 | 4,860 | | 5,851 | 7,892 | 5,210 | 3,265 | 3,328 | 2,940 | 1,861 | 41,003 |
| Poles | 283 | 1,098 | 1620 | | 1,525 | 2,057 | 1,184 | 1,959 | 676 | 1,183 | 600 | 12,185 |
| Wood Pellets | | | | | 422 | 569 | 7,105 | 2,612 | 2,704 | 2,230 | 1,525 | 17,167 |
| Chips | 746 | 137 | 10,800 | 1,350 | 734 | 982 | 710 | 489 | 500 | 709 | | 17,157 |
| Animal bedding | 154 | 137 | | 810 | 549 | 741 | 473 | | 500 | 507 | 660 | 4,531 |
| Pallets and crates | | 420 | | | | | | | | | | 420 |
| Beams | | 5,856 | | | 6,020 | 8,120 | 5,210 | 4,734 | 4,309 | 6,084 | 3,213 | 43,546 |
| Mulch | | 137 | | | | | | | | | | 137 |
| Landscaping Ties | | 137 | | | | | | | | | | 137 |
| Shavings | | | | | 457 | 617 | | 163 | 185 | 169 | 360 | 1,951 |
| Flooring | | 841 | | | | | | | | | | 841 |
| Other | | 2,013 | | | | | | 326 | | | | 2,339 |
| Total | 2,573 | 18,260 | 27,000 | 27,000 | 19,637 | 2,6481 | 2,3680 | 16,323 | 16,900 | 16,981 | 12,008 | 206,843 |

## Section 2.4: Collaboration

Collaboration is a process and an approach to managing diverse interests in land management in complex and high value landscapes. Collaboration styles and approaches vary widely across the country, often context and community driven. The collaboration style in New Mexico is informal without charters or many other documents in place. This informal style was typical of the Southwest Jemez Collaborative. In 2013 and 2014 there was an effort to develop a charter for the collaborative with the hopes that a charter would strengthen the collaborative particularly during times of disagreement. Ultimately there wasn't enough interest in a more formal collaborative style and by 2015 the effort was shelved.

The collaboration in the Southwest Jemez CFLR was strongest when it centered itself on landscape and community values and the extent to which the robust monitoring program could maintain and increase awareness of the project, its activities, and its impacts with people who deeply cared about those values. The development of the CFLR proposal was very collaborative

in nature with many people investing a week in workshop sessions identifying values and concerns and often co-writing proposal sections together.

A critical collaboration was with local tribes. While many tribes are connected to the culturally rich SW Jemez Mountains landscape, the Pueblo of Jemez and the Pueblo of Santa Clara were two tribes who were very active in the project. Jemez Pueblo and their Natural Resource Department were actively involved in work on the VCNP and SFNF as well as the small portion of tribal trust lands that were in the CFLR boundary. This involvement included in fire management, fuels management, and other important aspects. Santa Clara Pueblo was very involved with their Forestry Department as their tribal trust lands abut the CFLR boundary and their forestry crews contributed to fuels and fire management efforts. Both tribes were also engaged in post-fire efforts in the landscape.

Another important collaborative aspect of the project was the intention to support training and career pathways for local youth interested in land management. One example is the Forest Stewards Guild's Forest Stewards Youth Corps who had longstanding crew of high school aged youth stationed at the Jemez Ranger District. Almost 100% of their project work occurred in the project and they typically helped with trail clearing, fence management, invasive species control, cultural site protection, timber marking, and habitat protection projects. In 2018 the Guild started a fall fire and fuels based program with a crew based nearby at the Espanola Ranger District. In 2020, due to the pandemic, the Guild stood up a fall fire and fuels crew stationed at Jemez Pueblo and for the past 5 seasons have implemented projects and supported prescribed burns in the landscape. Both the Espanola and Jemez Pueblo crews (18 – 25 year olds) supported prescribed fires in the project area and continue to do so through fall 2024. Some alumni of the program now work for the SFNF.

Once funded the collaborative efforts intensified as the project gained attention and notoriety of being in an elite group of projects nationally. In the early years of the project the collaborative effort hit a roadblock as the project initiated an EIS process to NEPA "clear" priority areas on USFS lands not already analyzed. The roadblock was centered around the USFS and its challenges with clear direction for line officers to implement environmental planning in a collaborative context while also complying with the Federal Advisory Committee Act (FACA). This roadblock caused many partners and community groups to feel excluded from collaboration. Interestingly, the VCNP completed an EA for across a large area in record time, received accolades from the Society of Environmental Planners for the EA, all while engaging with the Collaborative and others in an open and clear process. These two different experiences are likely due to organizational culture and history and the fact that the VCNP was a small, young, and nimble organization. Due to this roadblock the Collaborative focused on the monitoring program as the main space for the Collaborative to engage and contribute.

As the Collaborative focused on the monitoring program and its many facets, leadership at the VCNP and the SFNF initiated an annual "All-Hands" project meeting that largely focused on annual accomplishments and monitoring efforts. This wildly successful event grew from having 40 – 60 attendees from 2012 – 2015 to having 80 – 120 attendees in 2016 – 2019. These large meetings filled up conference spaces and lecture halls and around 2016 a second day was added to interpret and discuss the implications of all the management accomplishments and monitoring

findings that were shared. 2019 was the last "All-Hands" meeting of the Southwest Jemez Collaborative due to the funding sunset overlapping with the Covid-19 pandemic. There was a final field meeting in late 2021 to reconvene partners for the last year where new funding was obligated.

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# Chapter 3. Climate Monitoring

**Monitoring project title: Monitoring climate dynamics during landscape restoration**

**Collaborators:** Robert R. Parmenter, Scott Compton, David Pittenger – *National Park Service, Valles Caldera National Preserve;* Douglas I. Moore – *University of New Mexico;* Greg McCurdy – *Desert Research Institute RAWS Network; National Weather Service; NOAA Climate Reference Network; NRCS SNOTEL Network.*

**Summary:** The climate in the SWJM CFLRP project area reflected the warming trend found world-wide, with temperatures climbing at a rate of 1.8° F per century (exactly the rate of increase found by the Inter-Governmental Panel on Climate Change (IGPCC). The rate of temperature increase was even faster at higher elevations in the project area. Temperature increases were most pronounced during the spring and summer months, with winter monthly temperatures remaining relatively constant. The project area during the CFLRP (2010-2024) was consistently below average in precipitation, with the exception of 2015, and extreme dry years led to high-severity wildfires within the project area (e.g., the 2011 Las Conchas Fire and the 2013 Thompson Ridge Fire). Relative humidity during the project years decreased, and maximum wind speeds increased. Thus, the SWJM CFLRP occurred during a period of time that was warmer and drier than at any time during the last 110 years.

## Climate Monitoring

Climate in the SWJM CFLRP/LRMP project area is continually monitored using Remote Access Weather Stations (RAWS). Data from this network of stations are collected and managed under an agreement with the Western Regional Climate Center at the Desert Research Institute at the University of Nevada, Reno; weather data can be found on the internet at the following link (Fig. 3.1):

https://wrcc.dri.edu/vallescaldera/

Weather data are uploaded hourly via the GOES satellite system and displayed with an approximately 1-2 hour delay. The WRCC website includes a series of analytical software tools that can quickly provide summary statistics of the various parameters being measured at each station (Fig. 3.2).

Weather stations were established throughout the project area in years prior to the SWJM CFLRP, and the network of stations was expanded in 2011 with the start of the monitoring program (Fig. 3.1). In addition, a Climate Reference Network (CRN) meteorological station was established in 2004 by the National Oceanic and Atmospheric Administration (NOAA) (see https://www.ncei.noaa.gov/access/crn/sensors.htm?stationId=1138); the station is located in the Valle Grande next to the NPS RAWS station. Finally, the National Resource Conservation Service (NRCS) operates two SNOTEL sites on and adjacent to Valles Caldera National Preserve: the Garita Peak site (https://wcc.sc.egov.usda.gov/nwcc/site?sitenum=1173) and the Quemazon site (https://wcc.sc.egov.usda.gov/nwcc/site?sitenum=708&state=NM).

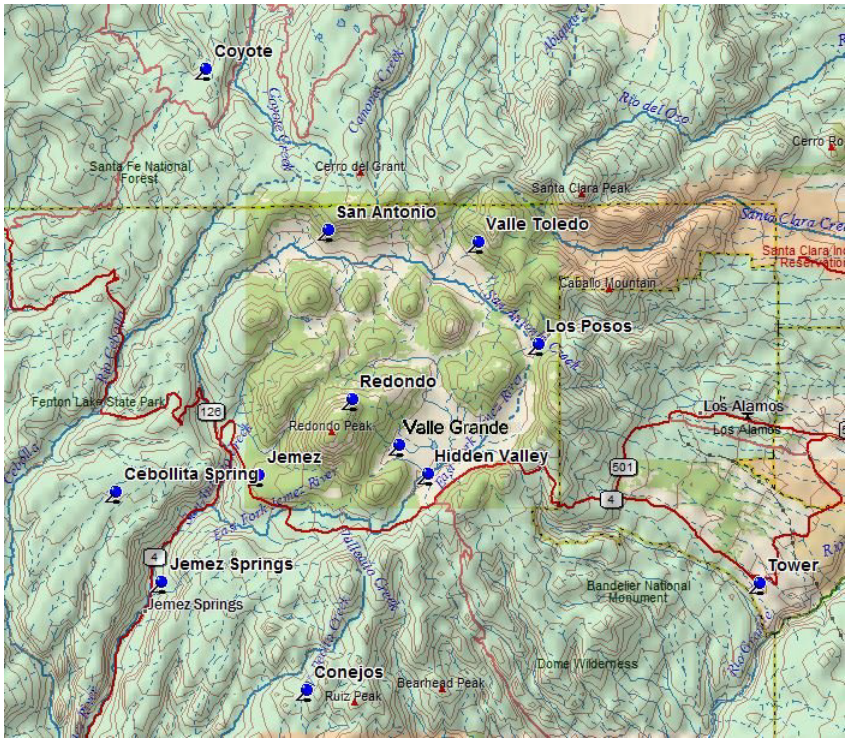
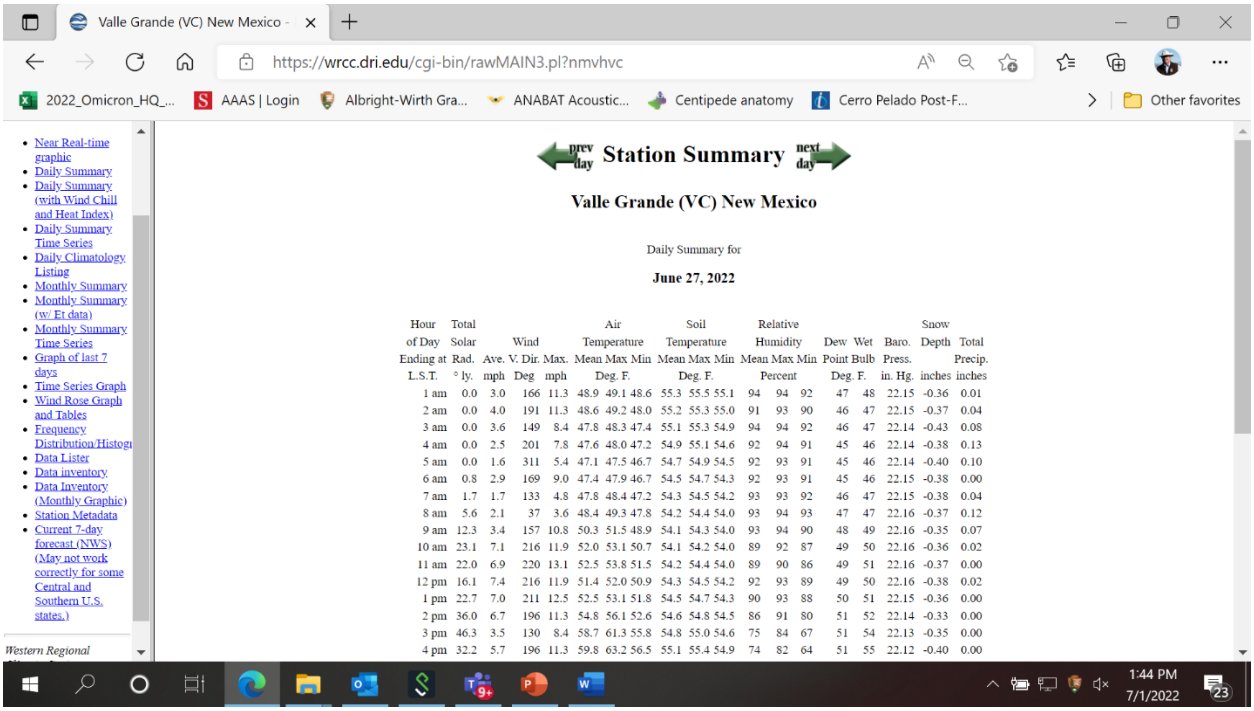



Figure 3.1. Map of RAWS station locations in the Jemez Mountains.




| Hour of Day Ending at L.S.T. | Total Solar Rad. ° ly. | Wind Ave. mph | Wind V. Dir. Deg | Wind Max. mph | Air Temperature Mean Deg. F. | Air Temperature Max Deg. F. | Air Temperature Min Deg. F. | Soil Temperature Mean Deg. F. | Soil Temperature Max Deg. F. | Soil Temperature Min Deg. F. | Relative Humidity Mean Percent | Relative Humidity Max Percent | Relative Humidity Min Percent | Dew Point Deg. F. | Wet Bulb Deg. F. | Baro. Press. in. Hg. | Snow Depth inches | Total Precip. inches |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 am | 0.0 | 3.0 | 166 | 11.3 | 48.9 | 49.1 | 48.6 | 55.3 | 55.5 | 55.1 | 94 | 94 | 92 | 47 | 48 | 22.15 | -0.36 | 0.01 |
| 2 am | 0.0 | 4.0 | 191 | 11.3 | 48.6 | 49.2 | 48.0 | 55.2 | 55.3 | 55.0 | 91 | 93 | 90 | 46 | 47 | 22.15 | -0.37 | 0.04 |
| 3 am | 0.0 | 3.6 | 149 | 8.4 | 47.8 | 48.3 | 47.4 | 55.1 | 55.3 | 54.9 | 94 | 94 | 92 | 46 | 47 | 22.14 | -0.43 | 0.08 |
| 4 am | 0.0 | 2.5 | 201 | 7.8 | 47.6 | 48.0 | 47.2 | 54.9 | 55.1 | 54.6 | 92 | 94 | 91 | 45 | 46 | 22.14 | -0.38 | 0.13 |
| 5 am | 0.0 | 1.6 | 311 | 5.4 | 47.1 | 47.5 | 46.7 | 54.7 | 54.9 | 54.5 | 92 | 93 | 91 | 45 | 46 | 22.14 | -0.40 | 0.10 |
| 6 am | 0.8 | 2.9 | 169 | 9.0 | 47.4 | 47.9 | 46.7 | 54.5 | 54.7 | 54.3 | 92 | 93 | 91 | 45 | 46 | 22.15 | -0.38 | 0.00 |
| 7 am | 1.7 | 1.7 | 133 | 4.8 | 47.8 | 48.4 | 47.2 | 54.3 | 54.5 | 54.2 | 93 | 93 | 92 | 46 | 47 | 22.15 | -0.38 | 0.04 |
| 8 am | 5.6 | 2.1 | 37 | 3.6 | 48.4 | 49.3 | 47.8 | 54.2 | 54.4 | 54.0 | 93 | 94 | 93 | 47 | 47 | 22.16 | -0.37 | 0.12 |
| 9 am | 12.3 | 3.4 | 157 | 10.8 | 50.3 | 51.5 | 48.9 | 54.1 | 54.3 | 54.0 | 93 | 94 | 90 | 48 | 49 | 22.16 | -0.35 | 0.07 |
| 10 am | 23.1 | 7.1 | 216 | 11.9 | 52.0 | 53.1 | 50.7 | 54.1 | 54.2 | 54.0 | 89 | 92 | 87 | 49 | 50 | 22.16 | -0.36 | 0.02 |
| 11 am | 22.0 | 6.9 | 220 | 13.1 | 52.5 | 53.8 | 51.5 | 54.2 | 54.4 | 54.0 | 89 | 90 | 86 | 49 | 51 | 22.16 | -0.37 | 0.00 |
| 12 pm | 16.1 | 7.4 | 216 | 11.9 | 51.4 | 52.0 | 50.9 | 54.3 | 54.5 | 54.2 | 92 | 93 | 89 | 49 | 50 | 22.16 | -0.38 | 0.02 |
| 1 pm | 22.7 | 7.0 | 211 | 12.5 | 52.5 | 53.1 | 51.8 | 54.5 | 54.7 | 54.3 | 90 | 93 | 88 | 50 | 51 | 22.15 | -0.36 | 0.00 |
| 2 pm | 36.0 | 6.7 | 196 | 11.3 | 54.8 | 56.1 | 52.6 | 54.6 | 54.8 | 54.5 | 86 | 91 | 80 | 51 | 52 | 22.14 | -0.33 | 0.00 |
| 3 pm | 46.3 | 3.5 | 130 | 8.4 | 58.7 | 61.3 | 55.8 | 54.8 | 55.0 | 54.6 | 75 | 84 | 67 | 51 | 54 | 22.13 | -0.35 | 0.00 |
| 4 pm | 32.2 | 5.7 | 196 | 11.3 | 59.8 | 63.2 | 56.5 | 55.1 | 55.4 | 54.9 | 74 | 82 | 64 | 51 | 55 | 22.12 | -0.40 | 0.00 |

Fig. 3.2. Example screen view of RAWS weather data, showing list of variables in main hourly data table, and list of software analyses available (left column in blue font).

In addition to the RAWS network, CRN and SNOTEL sites, a National Weather Service (NWS) station has been operating in Jemez Springs since 1914. The data from this station have been obtained from NWS and summarized for the period 1914-2024 (inclusive) to assess the degree of climate warming and precipitation patterns over the last 110 years.

**Climate trend results**

Atmospheric air temperatures from the Jemez Springs NWS station exhibited a statistically significant warming during the 20th Century and through 2023 (Fig. 3.3). The slope (rate of temperature increase) of the regression line indicated a 1.8º F increase in temperature over 100 years. This value is very close to the predicted temperature increase for our region of North America that was estimated by the Inter-Governmental Panel on Climate Change (IGPCC); as such, these data support the contention that the southwestern portion of the Jemez Mountains are exhibiting climate warming at a rate comparable to predictions by global climate models (GCMs) and that the forests in our project area are being subjected to climate trends that are similar to those of the American Southwest.

A more detailed analysis of these temperature data revealed that climate warming is occurring unevenly throughout the year. A month-by-month analysis of the 110-yr data set produced differences in rates of warming among spring, summer and winter months. The maximum rate of warming was observed during the month of June (4.4º F per 100 yr), whereas the months of October, November and December exhibited no temperature increase at all (Fig. 3.4).

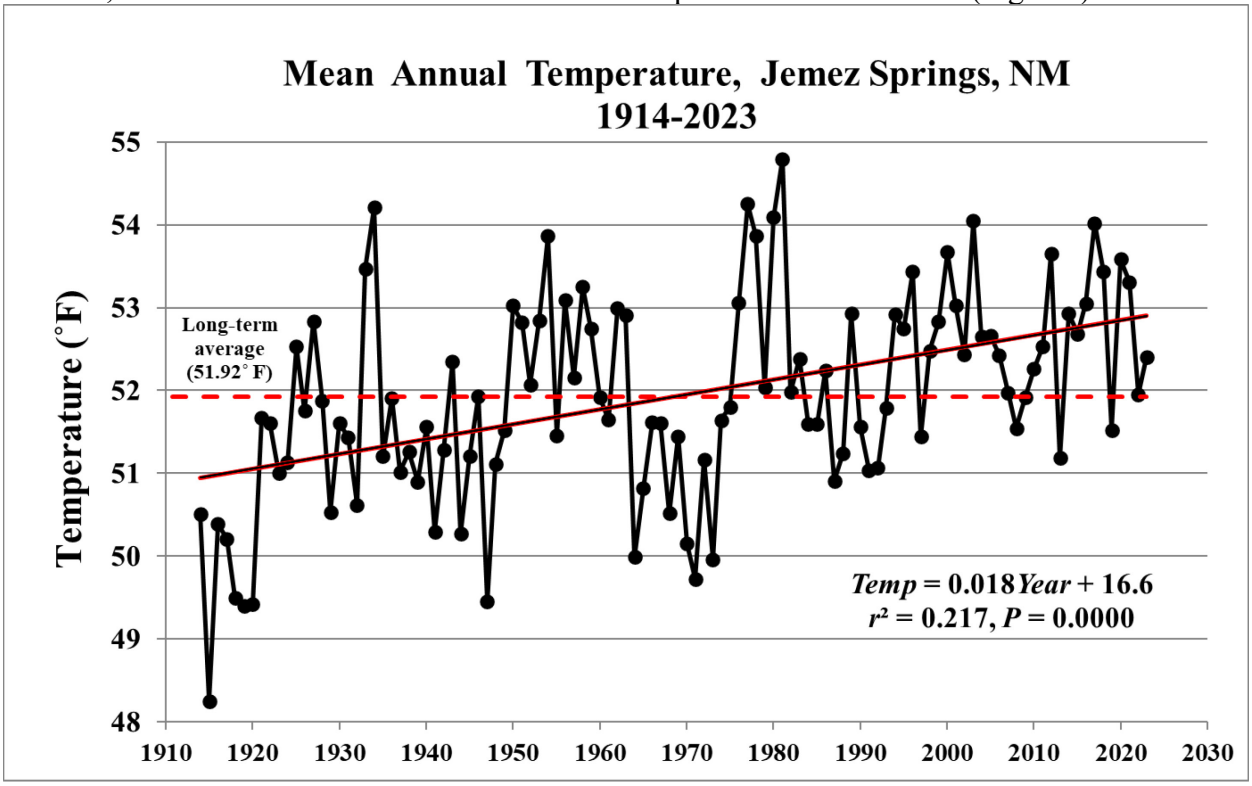



Fig. 3.3. Mean annual atmospheric temperatures recorded in Jemez Springs, 1914-2023.

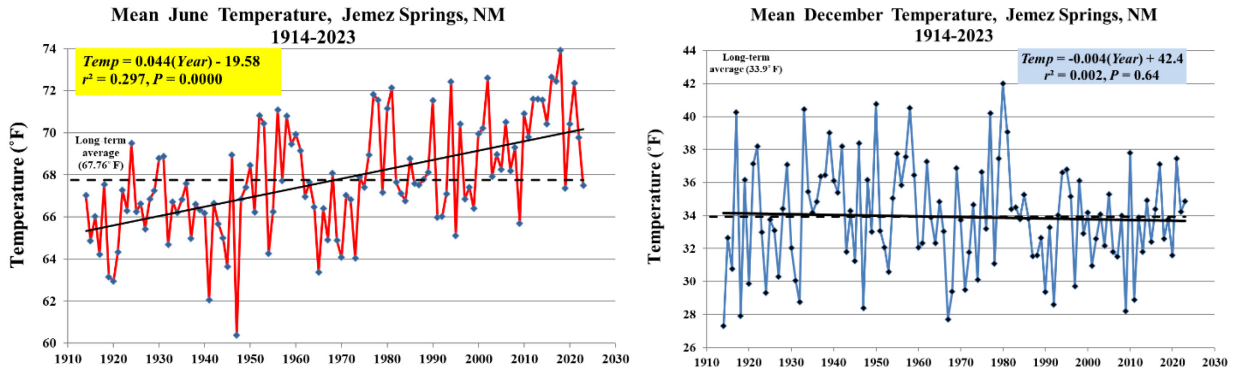



Fig. 3.4. Mean monthly atmospheric temperatures in June (left) and December (right) recorded in Jemez Springs, 1914-2023.

Precipitation patterns in Jemez Springs during this time period (Fig. 3.5) show considerable variation from year to year, but a statistical trend is evident – precipitation in Jemez Springs was above average during the 1920s and 1930s, and then entered the historic extended drought during the 1940s and 1950s. A particularly wet period began in the late-1970s and continued through the 1980s, with drought again returning in the 1990s through present-day. This pattern is aligned with the Pacific Decadal Oscillation (PDO), which influences long-term patterns of the jet stream that affects winter storm tracks in northern New Mexico. The periodicity for this oscillation is 20-30 years of above average precipitation followed by 20-30 years of below-average precipitation. If this pattern holds up, northern New Mexico should be emerging from the drought portion of this cycle sometime in the coming 5-10 years. All of the CFLRP/LRMP activities (2010-2024) have occurred during the drought phase of the PDO cycle, with only the year 2015 having substantially above-average precipitation.

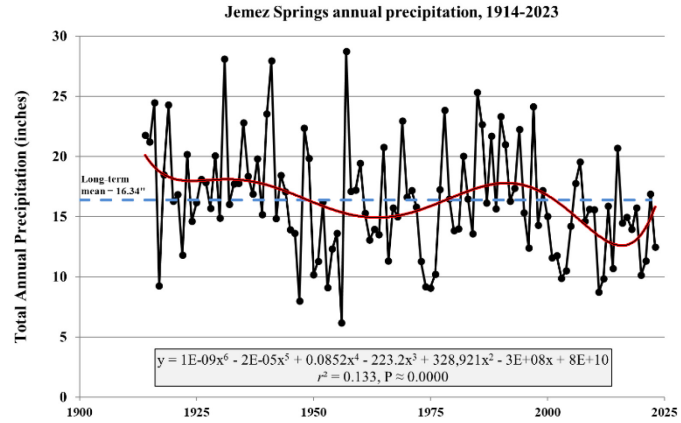



Fig. 3.5. Total annual precipitation data from Jemez Springs during 1914-2023.

Climate data from the CFLRP/LRMP RAWS network expand on the patterns observed from the Jemez Springs NWS station. The RAWS with the longest record is the **Valle Grande station**, and these data are reported below.

***Temperature.*** Mean annual temperatures in the Valle Grande over the 20 years of record (2004-2023, inclusive) show a rapid increase in annual temperatures of 0.093º C per year (Fig. 3.6); this is equivalent to 0.167º F per year, or 16.7º F per century, a rate that is nearly an order of magnitude greater than the temperature increase (1.8º F per century) observed in the Jemez Springs data (compare with Fig. 3.3 above). Also different from the Jemez Springs data, the warming trend is most apparent during April and late summer/fall (July-October), but with similar non-significant trends during winter (Fig. 3.7). Maximum temperatures also are increasing, but minimum temperatures do not appear to be changing significantly (Fig. 3.8). Climate data sets of short duration (less than 30 years) need to be interpreted cautiously, so the exceptionally rapid temperature increase may not represent the trend magnitude for the rest of this century; however, it is clear that atmospheric temperatures have been rising rapidly in the Valle Grande over the last 20 years.

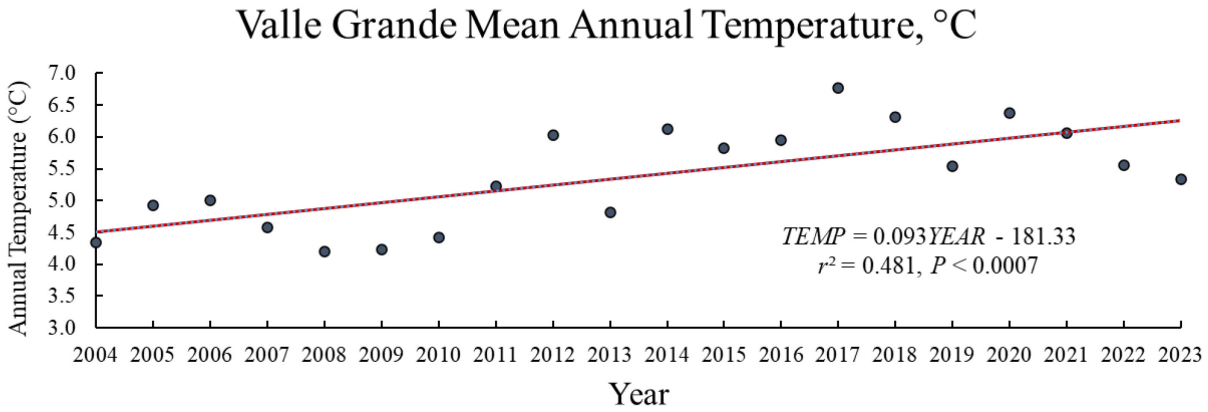



Fig. 3.6. Annual atmospheric temperature data from the Valle Grande RAWS, 2004-2023.

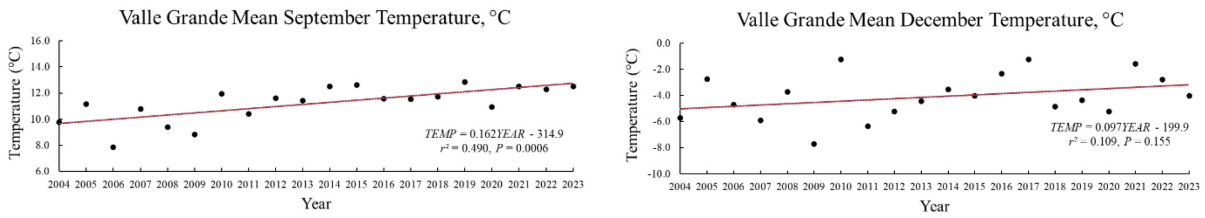



Fig. 3.7. Left: Monthly mean temperatures for September. Right: Monthly mean temperatures for December.

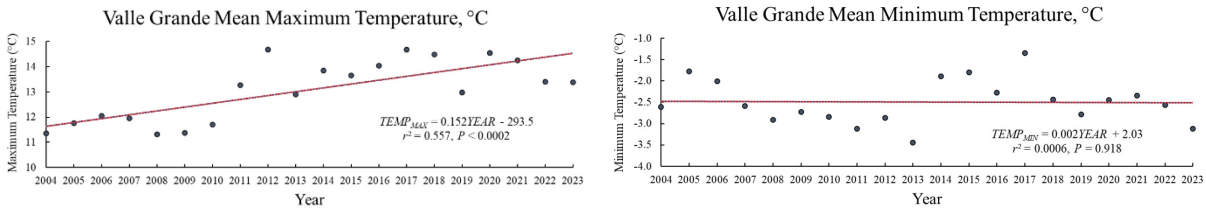



Fig. 3.8. Left: Mean annual maximum temperatures. Right: Mean annual minimum temperatures.

***Precipitation***. The total annual precipitation recorded in the Valle Grande averaged 596 mm/yr, with a low of 361 mm in 2020 and a high of 889 mm in 2015 (Fig. 3.9). Overall, the CFLRP project area exhibited no significant trend in annual precipitation during the period of record.

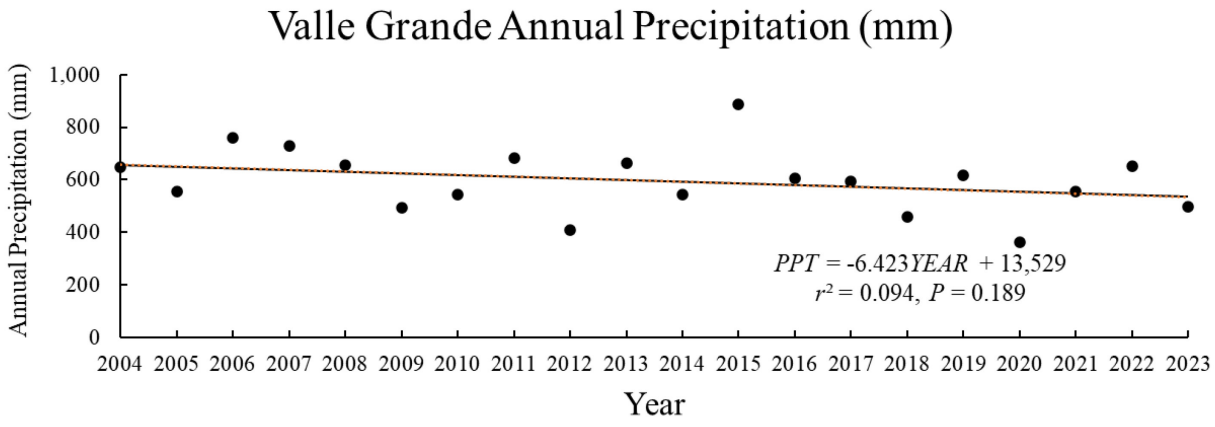



Fig. 3.9. Total annual precipitation (mm) in Valle Grande, 2004-2023.

***Relative humidity.*** The mean annual relative humidity in the Valle Grande displayed a statistically significant downward trend, becoming generally drier (Fig. 3.10). This would be a result of warming atmospheric temperatures (which have been shown to be increasing), rather than a decrease in precipitation (which shows no significant trend). Mean annual relative humidity has dropped from ~60% to ~52% over the last two decades.

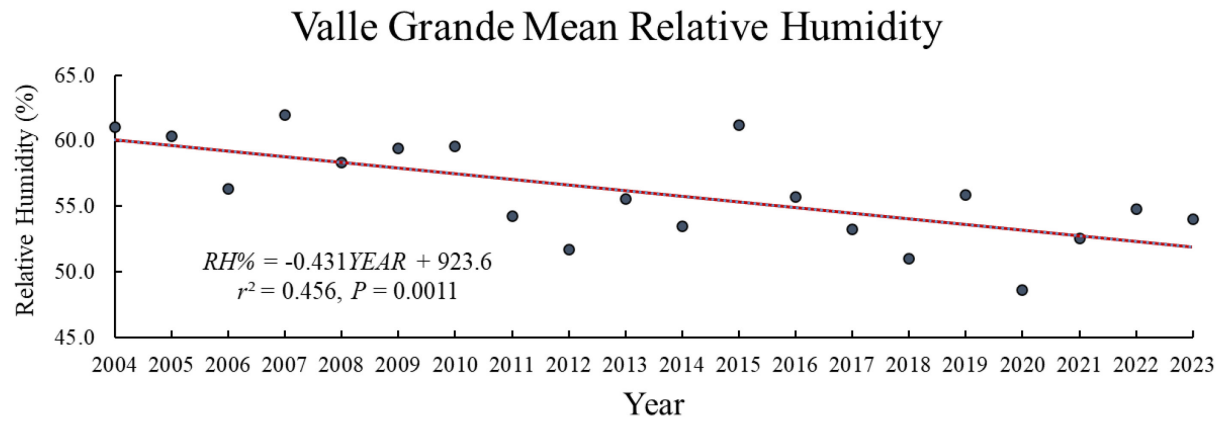



Fig. 3.10. Mean annual relative humidity (%) in Valle Grande, 2004-2023.

***Wind speed***. The mean annual wind speeds during 2004-2023 averaged 3.3 meters/sec, and ranged from 1.9 to 2.5 meters/second. Overall, there was no statistically significant trend in mean wind speeds over the period of record (Fig. 3.11). However, the mean maximum wind gust speeds did show a significant increase in speeds; the average 10-second maximum gust speed recorded was 17.5 meters/sec, and over the 20 years of monitoring, wind gusts increased from ~16.2 meters/sec during 2004-2008, to ~18.2 meter/sec during 2019-2023, or an increase of ~2 meters/sec (Fig. 3.12). Wind gust speeds are important in forest restoration, as these gusts can topple trees in exposed locations – e.g., newly thinned stands with fewer neighboring trees to shield isolated trees from high wind speeds. As with humidity, these higher wind gust speeds may be related to higher temperatures, as warmer atmospheric temperatures provide higher energy levels that drive wind dynamics.

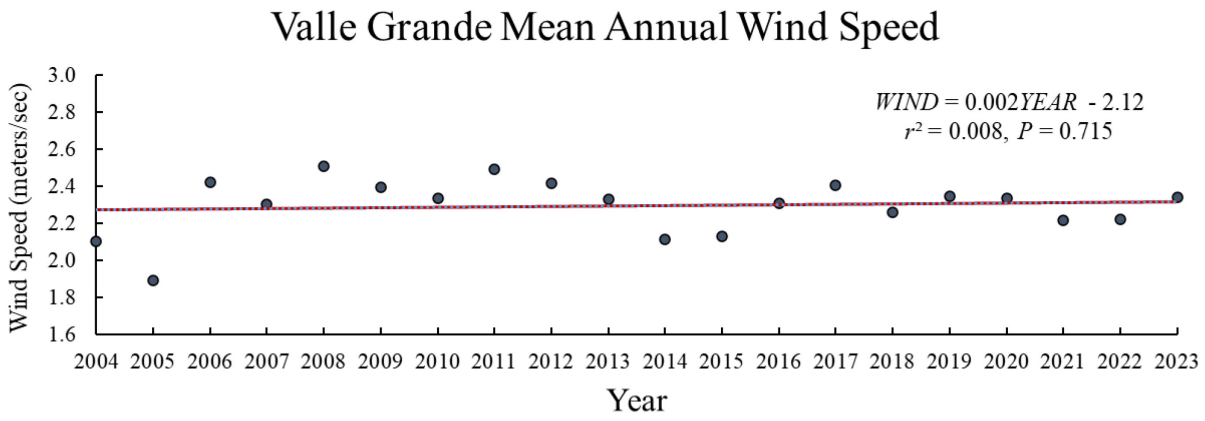



Fig. 3.11. Mean annual wind speed in the Valle Grande, 2004-2023

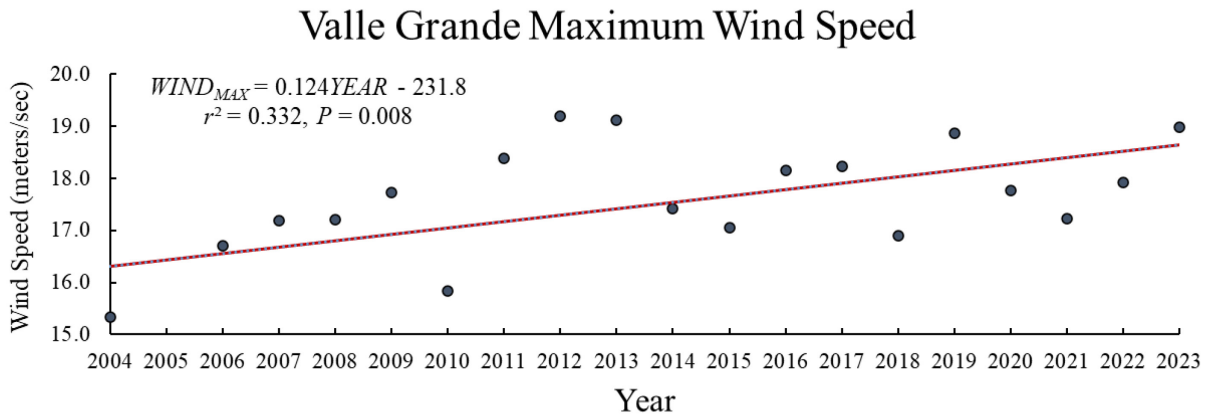



Fig. 3.12. Mean annual maximum 10-second wind gust speed in the Valle Grande, 2004-2023.

# Chapter 4. Forest Vegetation Monitoring and Fire Risk Analysis

**Collaborators:** Steven del Favero, Benjamin Bright – *US Forest Service.* Laura Trader, Kathy Brinkman, Brianna Jol, Benjamin Stout, Ana Casillas-Brownson, Sarah Hall, Martina Suazo, Benjamin Bleiman, Alec McKeand, Chloe Pryor – *National Park Service, Pueblo and Four Winds Fire Ecology Program and Valles Caldera National Preserve.*

**Summary:** The SWJM CFLRP activities were successful in reducing potential crown fire and lowering estimated flame lengths as measured with FlamMap fire behavior model analyses. These results were consistent using canopy data derived from both LANDFIRE and LiDAR data and showed enhanced reductions within treatment areas compared to the overall landscape. From the field monitoring of forest stand structure and forest-floor herbaceous vegetation, the overall analysis showed that treatments resulted in (1) a reduction in forest canopy closure, but well within the 50-80% goal range; (2) a reduction in tree density attributed to small diameter reductions; (3) a small reduction in basal area attributed to removing small diameter trees and retaining large trees; (4) increases in crown base height that will help prevent fire from traveling from the forest surface to tree canopies via ladder fuels; (5) reductions in total surface fuel load; and (6) increases in live native herbaceous vegetation after prescribed fire (increases in light, nutrients, and water). Finally, project implementation areas were surveyed for non-native invasive plants, and over 500 acres of project area were treated with herbicides or manual removal. Invasive "weeds" were dominated by bull thistle, musk thistle, Canada thistle, cheatgrass, oxe-eye daisy, Siberian elm, mullein and one small population of Russian knapweed.

## Section 4.1. SWJM CFLRP Fire Risk Analysis

Fire risk estimates in the SWJM CFLRP project area were derived from FlamMap Fire Behavior simulation models using two variables: (1) Areas of potential stand-replacing crown fire; and (2) estimated flame lengths. We conducted fire risk analyses for the SWJM CFLRP project area at the time of the CFLRP proposal development (2009), and then again at the end of the project (2024); the changes in fire risk over those 15 years would provide an assessment of the success of the project in reducing overall fire risk. FlamMap fire behavior models were run based on landscape fuels and forest stand estimates generated from two different datasets, first, using 2009 and 2024 LANDFIRE data (satellite derived national products); and then using Light Distance And Ranging (*LiDAR*) data collected in the early years of the project (2010/2012) and again at the end of the project (2024). While the two different approaches would likely produce different magnitudes of modelled fire risk, we anticipated that the patterns of change would be similar.

### Methods

FlamMap Fire Behavior models were run using the Interagency Fuel Treatment Decision Support System (IFTDSS). Fire behavior was modeled under 97$^{th}$ percentile extreme weather conditions, determined from the nearest Remote Automatic Weather Station (RAWS – see Chapter 3). FlamMap requires an input "Landscape File" which is a multiband raster file composed of terrain data: (elevation, slope, aspect), canopy data: (canopy height, canopy cover, canopy base height, canopy bulk density), and Fuel Model data: Scott and Burgan Fire Behavior

Fuel Model (FBFM40). For the various model runs, these custom landscape files were assembled using ArcGIS software. Pre- and post-treatment landscape files were assembled using (1) LANDFIRE canopy data from 2009 and 2024 and (2) LiDAR modeled canopy data from 2010/2012 and 2024. LANDFIRE Fuel Model data (FBM40) was used for all model runs. Fire behavior model results were summarized for the entire SW Jemez project area, areas of the project burned in full suppression wildfires, and also for treatment areas that were not burned in full suppression wildfires. Treatment areas include mechanical thinning, RX burns, and wildfires managed for resource benefit.

Pre-project LiDAR data were obtained in 2010 on Valles Caldera National Preserve, and 2012 on the Santa Fe National Forest portion of the SWJM CFLRP project area; data were acquired at a resolution of 10 pulses/$m^2$ for these flight missions. In 2024, a post-project LiDAR mission was conducted, with a resolution of 20 pulses/$m^2$. These LiDAR data and predictive modeling were then used to construct raster layers describing forest fuel: stand height, canopy bulk density, and canopy base height. First, we created random forest (RF) models predicting the above fuel attributes, as measured on 65 fixed-radius field observation plots, from coincident LiDAR metrics. RF models explained 72, 53, and 27% of the variation in stand height, canopy bulk density, and canopy base height, respectively. We then produced LiDAR metric grids across 2010, 2012, and 2024 LiDAR project extents, and applied the above RF models to these grids to produce fuel attribute maps. The fire model LANDFIRE was then run on these fuels maps as described above to assess changes between 2010/2012 and 2024.

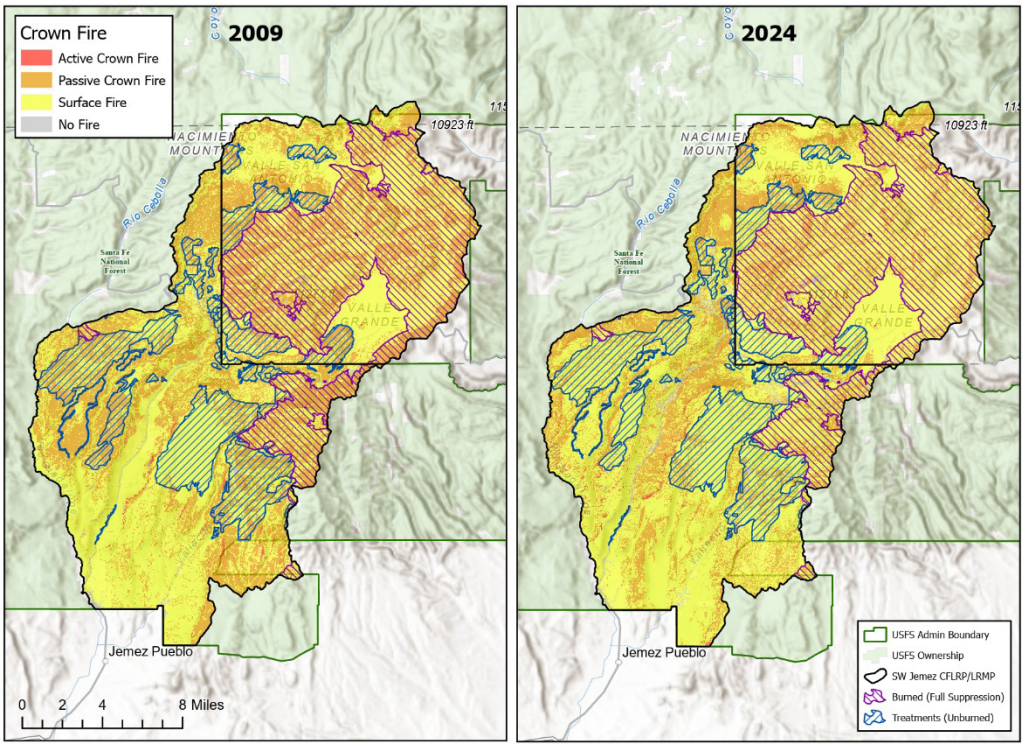
**Results**

The FlamMap fire behavior model results demonstrated that the SWJM CFLRP restoration activities reduced fire risk across the project area through reduced potential canopy fire and through shorter estimated flame lengths. Both modeled datasets (Landsat and LiDAR) provided similar reductions in fire risk, although the absolute magnitudes of the risk values were different.

*Fire Behavior Models based on LANDFIRE data*: Using the LANDFIRE canopy metrics, the FLAMMAP fire behavior model across the entire project area (210,748 acres) produced a reduction of potential crown fire from 87,769 acres in 2009 to 58,637 acres in 2024 (a 33% decline in acreage; see Figure 4.1.1). In 2009, potential crown fire acreage represented 42% of the landscape, and in 2024, that value was reduced to 28% (Figure 4.1.2, left histogram). The actual treatment sites within the project area, which had been strategically selected due to their high fuel loads and locations on the landscape (e.g., in areas of Wildland-Urban Interface (WUI) or need for establishing fire breaks), exhibited significantly greater reductions in potential crown fire, dropping from 18,096 acres in 2009 to 8,519 acres in 2024, or a reduction of 53% in acreage (Figure 4.1.2, right histogram). In contrast, acreage that burned in uncontrolled wildfires during the project period (2010-2024) showed a reduction in potential crown fire of 48% (Figure 4.1.2, middle histogram), indicating that (1) CFLRP treatments were more effective in reducing potential crown fire than actual wildfires, and (2) the treated areas still had live forests left standing after treatment, compared to areas with stand-replacement wildfire that did not.

The results of the fire behavior modeling also yielded reduced flame length estimates following the CFLRP treatments (Figures 4.1.3, 4.1.4). The percentages of high flame lengths

(> 11 ft.) declined by more than half, while the percentage of very low flame lengths (< 1 ft.) increased. These differences were even greater in actual treated areas (Figure 4.1.3, bottom) than the average flame lengths across the entire CFLRP landscape (Figure 4.1.3, top). The dominance of low flame lengths in the treatment areas are indicative of greatly reduced fuel loads and replacement of dead-and-down woody debris with herbaceous vegetation and needle-fall litter, reducing ladder fuels and the risk of stand-replacement fire in the project area.



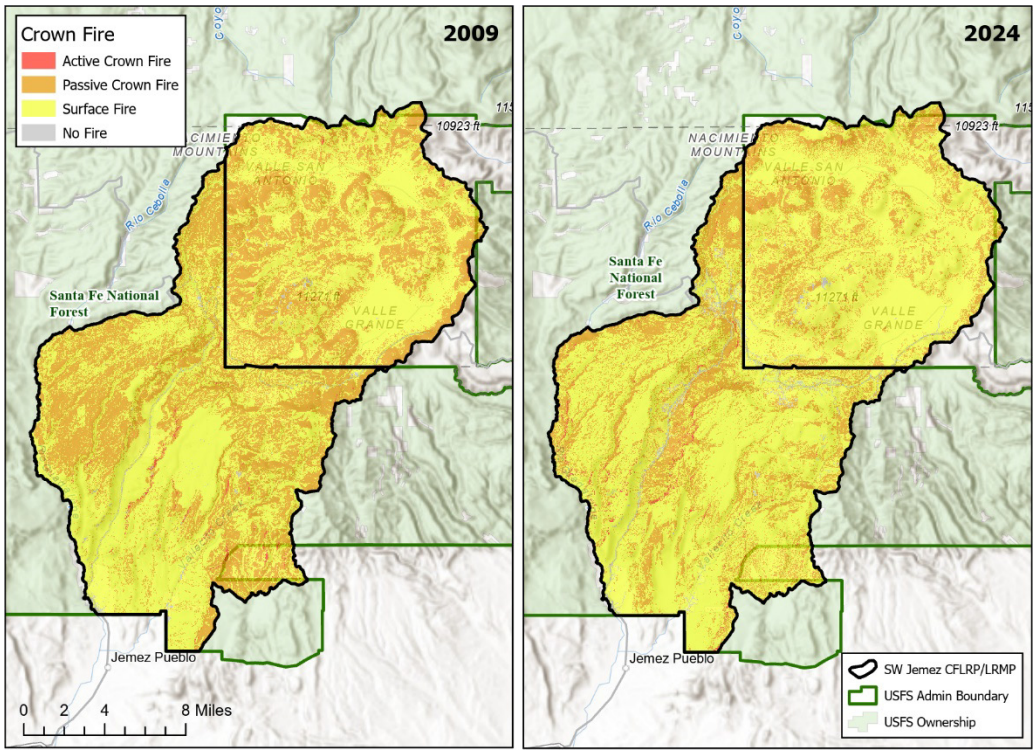



Figure 4.1.1. Maps of potential crown fire activity in the SWJM CFLRP project area between 2009 and 2024 based on FlamMap simulation models. Top: Crown fire areas with CFLRP treatment areas and wildfire burned areas superimposed; Bottom: Maps of crown fire potential within the project area.

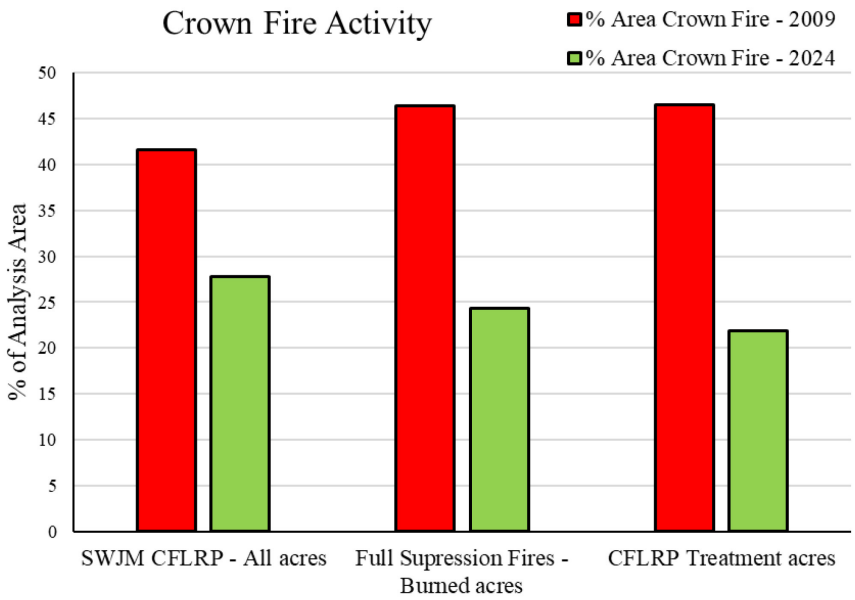



Figure 4.1.2. Reduction of potential crown fire activity resulting from SWJM CFLRP restoration treatments between 2009 and 2024 based on FlamMap fire behavior models. Results include analyses for the entire SWJM CFLRP project area (210,748 acres), the untreated areas within the project area that

burned in full-suppression wildfires (e.g., Las Conchas, Thompson Ridge, Guacamalla, Cajete, and Cerro Pelado fires), and in all the treated areas restored during the SWJM CFLRP activities.

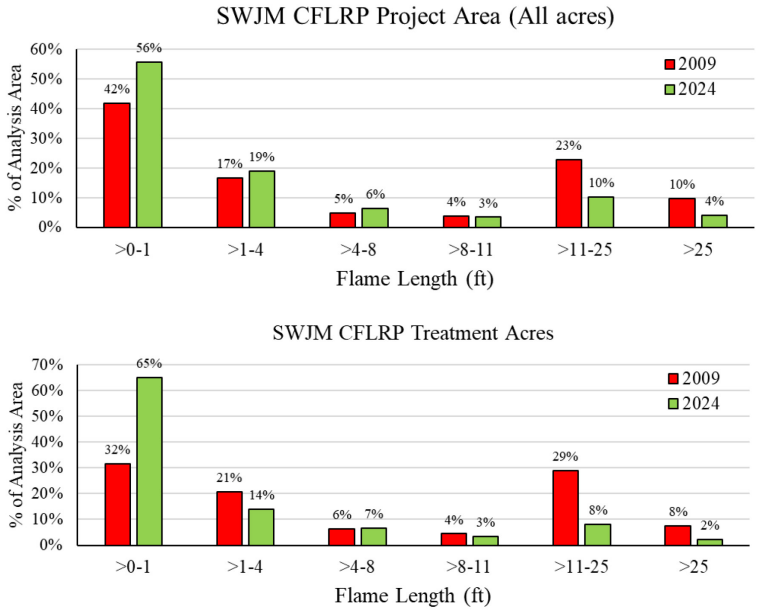



Figure 4.1.3. Changes in modeled wildfire flame lengths resulting from SWJM CFLRP restoration treatments between 2009 and 2024 based on FlamMap fire behavior models. Top: Average flame lengths for the entire SWJM CFLRP project area; Bottom: Flame lengths for treated areas only. Note shifts from higher flames lengths in 2009 to lower flame lengths in 2024 due to reduced fuel loads in treated areas.

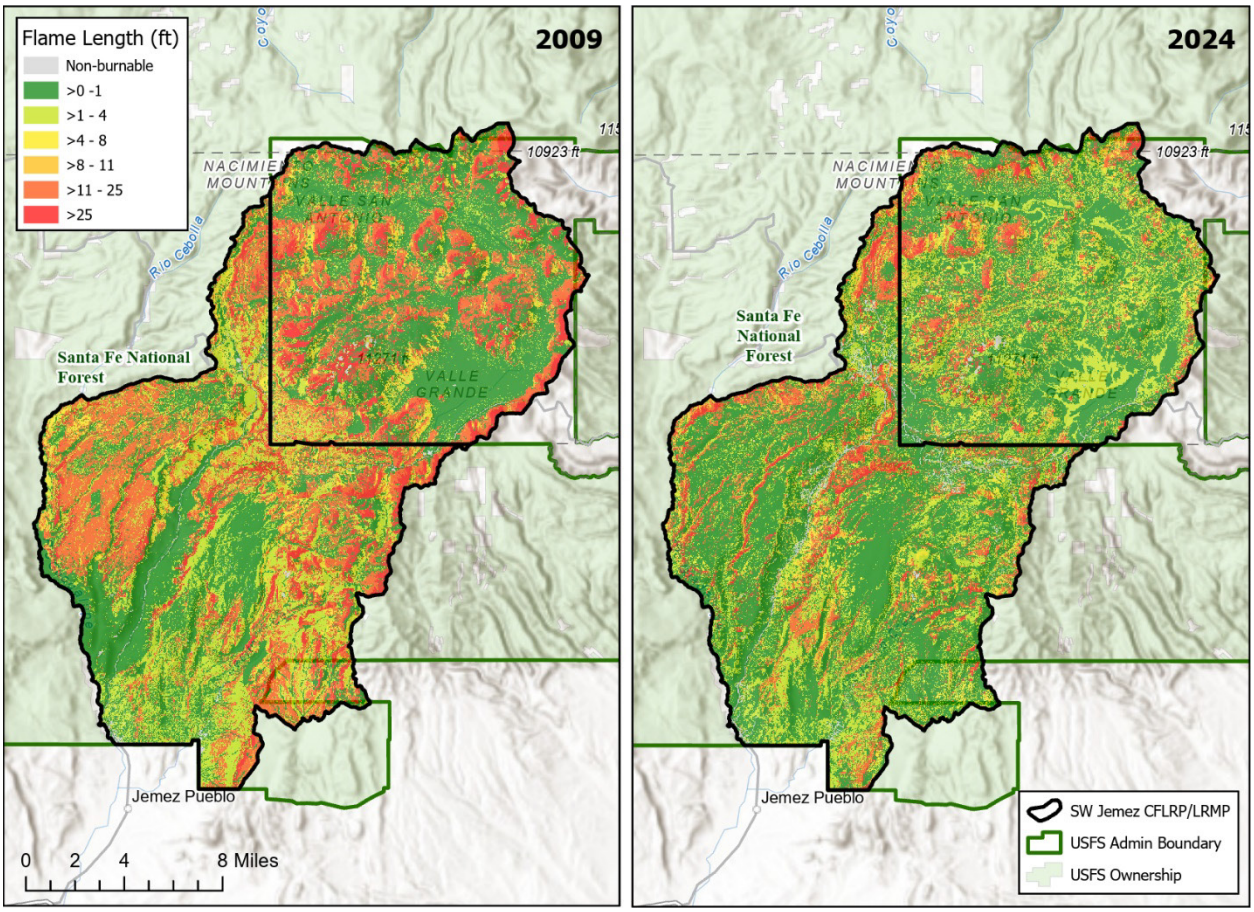



Figure 4.1.4. Maps of modeled wildfire flame lengths within the SWJM CFLRP project area between 2009 and 2024 based on FlamMap simulation models.

*Fire Behavior Models based on LiDAR Data*: Using the LiDAR derived canopy metrics, the FlamMap model produced much smaller total acres of high fire risk than did the models using LANDFIRE data (Figures 4.1.5, 4.1.6). However, the models showed similar declines in potential crown fire risk between 2010 and 2024 at the project area landscape level, and even greater declines within actual treatment areas. Interestingly, the LiDAR data produced an increase of 2.5% potential crown fire in areas that burned in uncontrolled wildfires between 2009 and 2024; this was likely due to the LiDAR picking up the "canopy" of regenerating young aspens and patches of shrubby vegetation (Gambel oak and New Mexico locust shrubs).

Flame length estimates using LiDAR derived canopy metrics were comparable to those of the LANDFIRE data, with clear reductions in the longer flame length categories and compensatory increases in low flame lengths (< 1 ft.); decreases were greater again in actual project treatment areas than across the entire CFLRP area (Figure 4.1.7).

In total, both models produced clear reductions in fire risk via estimates of potential crown fire and in reduced flame lengths. While only time will tell how the SWJM CFLRP activities actually influence fire risk and landscape burn patterns and intensities, the FlamMap results would indicate that the program was successful in reducing high-severity fire risk across our landscape's project area.

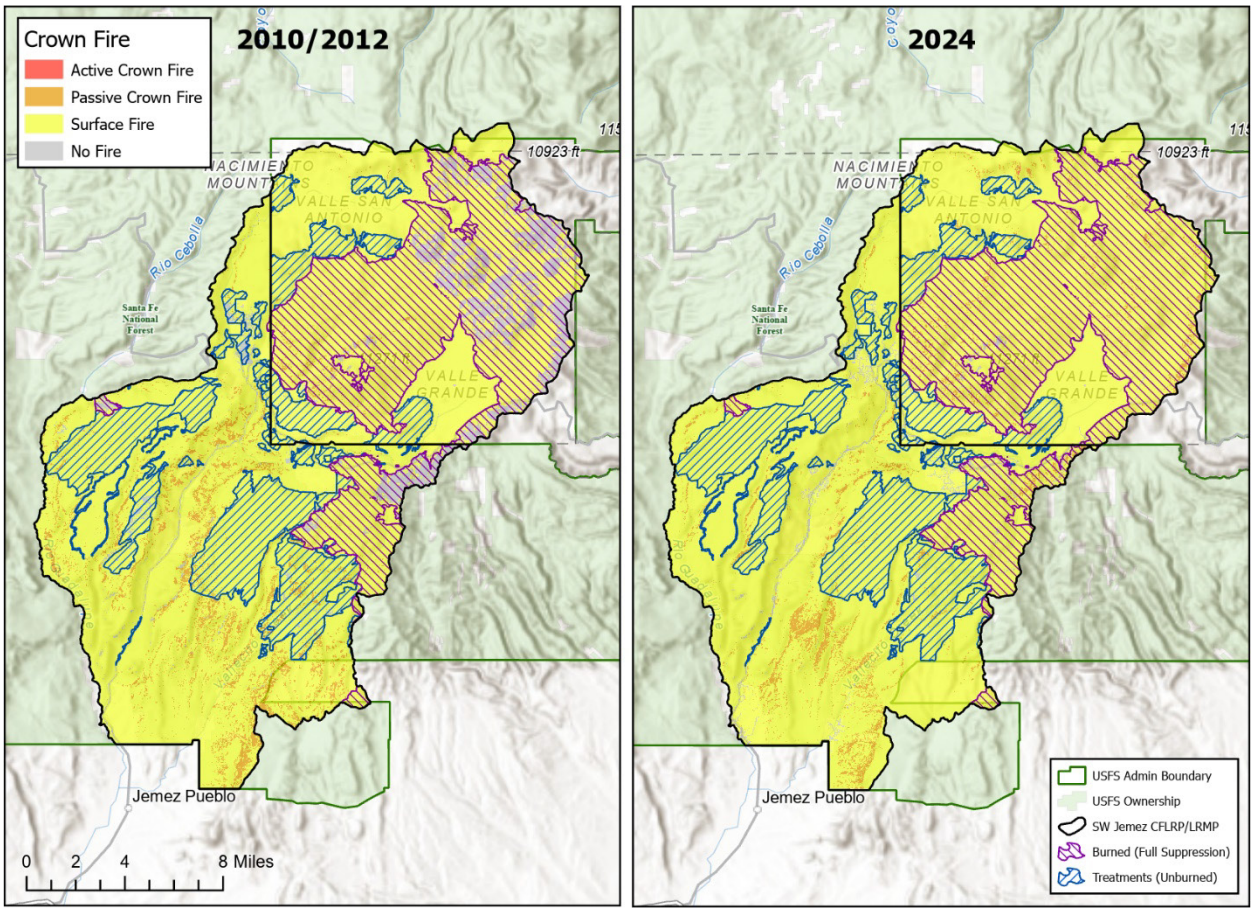



Figure 4.1.5. Maps of potential crown fire activity within the SWJM CFLRP project area between 2009 and 2024 based on LiDAR-based models of fuel loads in FlamMap.

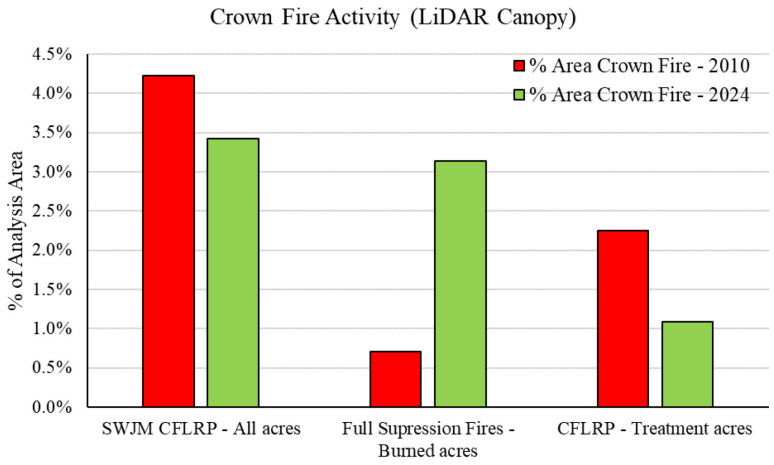



Figure 4.1.6. Reduction of potential crown fire activity resulting from SWJM CFLRP restoration treatments between 2009 and 2024 based on LiDAR derived fuel models input into FlamMap simulation models. Note that while absolute values of crown fire risk are lower than those using LANDFIRE inputs, the patterns of treatment effects are similar (compare to Figure 4.1.2 above).

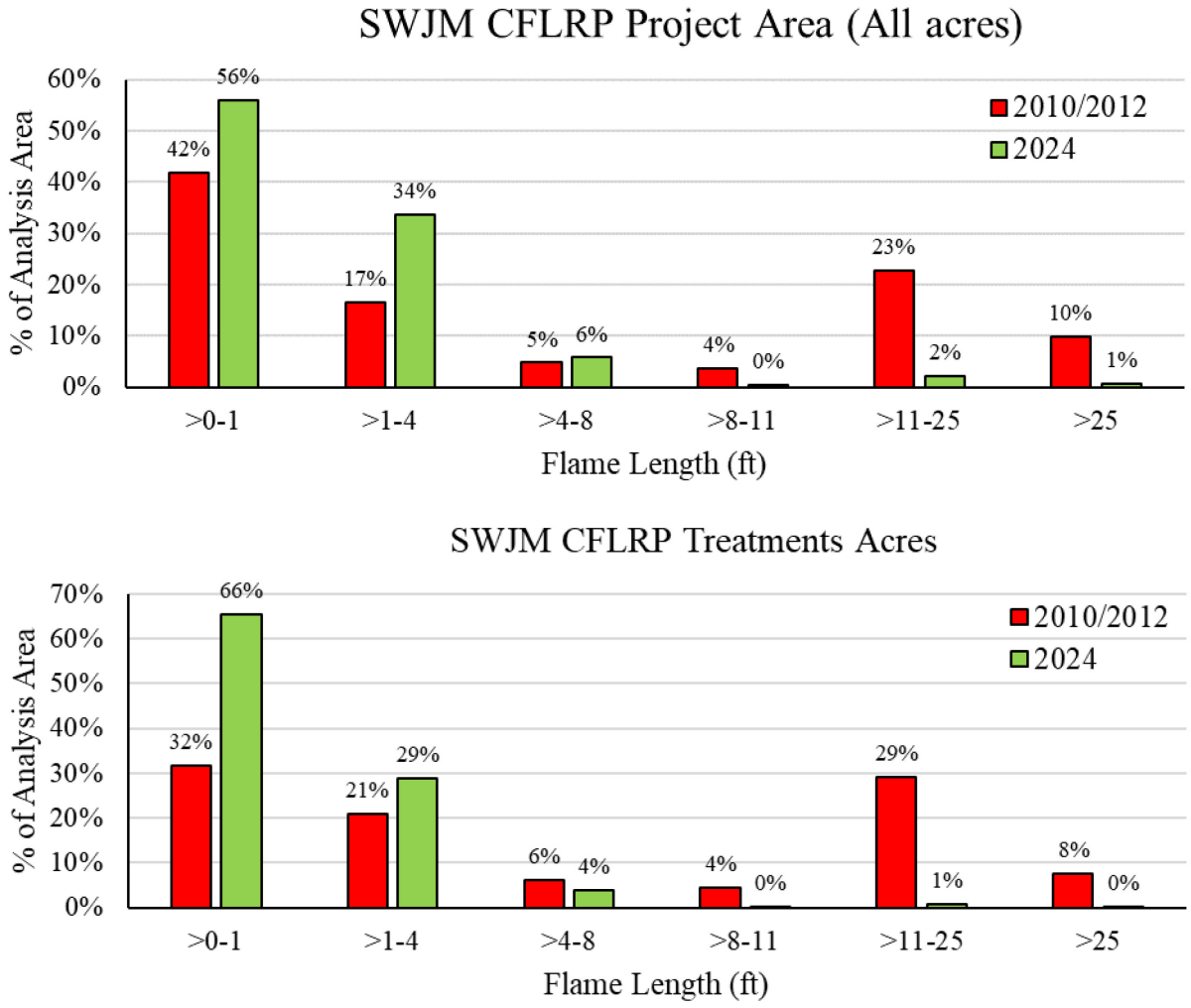



Figure 4.1.7. Changes in modeled wildfire flame lengths resulting from SWJM CFLRP restoration treatments between 2009 and 2024 based on LiDAR-derived fuel loads with FlamMap simulation models. Top: Average flame lengths for the entire SWJM CFLRP project area; Bottom: Flame lengths for treated areas only. Note shifts from higher flames lengths in 2009 to lower flame lengths in 2024 due to reduced fuel loads in treated areas.

## Section 4.2. Vegetation Monitoring Results

Monitoring of the CFLRP/LRMP projects is conducted by the NPS Pueblo and Four Winds Fire Ecology Program and Valles Caldera's Vegetation Program. Sites are generally sampled prior to treatment implementation, with permanent plots established and sampled again at various time intervals after treatment(s). In many cases, sites were thinned and then prescribed burned (pile burn and/or broadcast burn) and monitoring may occur before and after each treatment type.

Data analysis of monitoring studies in this report include one or more of the following: live herbaceous vegetation percent cover; native and non-native herbaceous vegetation percent cover; live tree canopy closure percent; live tree basal area; live tree density; live tree crown base height; and total surface fuel loading.

Data analysis was conducted in R statistical software. A mixed model for a repeated measures design was fit for each dataset and pre- and post-treatment values were compared using Tukey-adjusted pairwise comparisons. All tests were conducted at the alpha 0.05 level. Graphics were created in Microsoft Excel.

Herbaceous vegetation includes non-woody annual and perennial forbs, grasses, and grass-like (sedges and rushes) plants. Live trees include all species with a diameter at breast height of at least one inch. Surface fuel loading includes materials on the forest surface such as logs, branches, twigs, pinecones, needles, dead plant material, and duff.

Repeat photography is taken at each site during each visit, contributing additional understanding of post-treatment responses of the vegetation and surface fuels.

A summary of the monitoring sites and data collection is presented in Table 4.2.1. Pueblo and Four Winds Fire Ecology plot locations are shown in Figure 4.2.1 (VCNP) and Figure 4.2.2 (VCNP).

Table 4.2.1. Summary of CFLRP/LRMP monitoring sites and data collection. Data analyses from Project Areas highlighted in green are included in this report.

| Land Mgmt. Agency | Project Area | Monitoring Group | Vegetation Community | Treatment Year | # of Plots | Data Collection Years | Protocols |
|---|---|---|---|---|---|---|---|
| USFS | Cat Mesa | Pueblo and Four Winds Fire Ecology Program | Ponderosa pine | | 18 | 2018, 2022 | Trees, canopy closure shrubs, veg, fuels, photos |
| USFS | Monument Canyon | Pueblo and Four Winds Fire Ecology Program | Ponderosa pine | 2012 | 25 | 2009, 2012, 2013, 2014, 2017, 2022 | Trees, shrubs, veg, fuels, photos |
| USFS | Paliza North | Pueblo and Four Winds Fire Ecology Program | Ponderosa pine, Mixed Conifer | 2014 | 7 | 2012, 2013, 2014, 2015, 2016, 2019 | Trees, shrubs, veg, fuels, photos |

| Land Mgmt. Agency | Project Area | Monitoring Group | Vegetation Community | Treatment Year | # of Plots | Data Collection Years | Protocols |
|---|---|---|---|---|---|---|---|
| USFS | Paliza South | Pueblo and Four Winds Fire Ecology Program | Ponderosa pine, Mixed Conifer | | 7 | 2012, 2013 | Trees, shrubs, veg, fuels, photos |
| USFS | San Juan Canyon | Pueblo and Four Winds Fire Ecology Program | Ponderosa pine | | 10 | 2021 | Trees, shrubs, veg, fuels, photos |
| USFS | San Juan Mesa | Pueblo and Four Winds Fire Ecology Program | Ponderosa pine | 2012 | 20 | 2010, 2011, 2012, 2016, 2021 | Trees, shrubs, veg, fuels, photos |
| USFS | West Pino (San Juan Mesa) | Pueblo and Four Winds Fire Ecology Program | Ponderosa pine | 2017 | 10 | 2015 | Trees, canopy closure, shrubs, veg, fuels, photos |
| VALL | Banco North | Pueblo and Four Winds Fire Ecology Program | Ponderosa pine, Mixed Conifer | 2015, 2016, 2017, 2019 | 17 | 2015, 2016, 2017, 2019 | Trees, canopy closure, shrubs, veg, fuels, photos |
| VALL | Banco South | Pueblo and Four Winds Fire Ecology Program | Ponderosa pine | 2012 | 10 | 2020 | Canopy Closure, fuels |
| VALL | Cerro Pinyon | Pueblo and Four Winds Fire Ecology Program | Mixed Conifer | 2022, 2023 | 6 | 2021, 2023 | Trees, canopy closure, shrubs, veg, fuels, photos |
| VALL | Garita North and South | Pueblo and Four Winds Fire Ecology Program | Ponderosa Pine | | 5 | 2019 | Trees, canopy closure, shrubs, veg, fuels, photos |
| VALL | San Antonio Mountain | Pueblo and Four Winds Fire Ecology Program | Mixed Conifer | 2021 | 10 | 2020, 2023 | Trees, canopy closure, shrubs, veg, fuels, photos |
| VALL | Seco North and South | Pueblo and Four Winds Fire Ecology Program | Mixed Conifer | 2017 | 11 | 2016, 2017, 2019, 2020, 2021 | Trees, shrubs, veg, fuels, photos |
| VALL | South Mountain | Pueblo and Four Winds Fire Ecology Program | Mixed Conifer | 2017, 2018 | 9 | 2016, 2017, 2019, 2023 | Trees, canopy closure, shrubs, veg, fuels, photos |
| VALL | Triangle | Pueblo and Four Winds Fire Ecology Program | Ponderosa Pine | | 5 | 2020 | Trees, canopy closure, shrubs, veg, fuels, photos |

| Land Mgmt. Agency | Project Area | Monitoring Group | Vegetation Community | Treatment Year | # of Plots | Data Collection Years | Protocols |
|---|---|---|---|---|---|---|---|
| VALL | Garita, Twin Cabins South | Pueblo and Four Winds Fire Ecology Program | Mixed Conifer | 2019-2022 | 12 | 2018, 2024 | Trees, canopy closure, shrubs, veg, fuels, photos |
| USFS | Conejos | Valles Caldera Vegetation Program | Ponderosa pine | | 3 | 2013, 2020 | Trees, shrubs, veg, fuels, photos |
| USFS | East Fork | Valles Caldera Vegetation Program | Ponderosa pine | | 5 | 2018, 2019 | Trees, shrubs, veg, fuels, photos |
| USFS | Paliza North | Valles Caldera Vegetation Program | Ponderosa pine | 2014 | 10 | 2014, 2015, 2016, 2017, 2018 | Trees, shrubs, veg, fuels, photos |
| USFS | Paliza South | Valles Caldera Vegetation Program | Ponderosa pine | | 20 | 2013, 2021 | Trees, shrubs, veg, fuels, photos |
| USFS | San Juan Mesa | Valles Caldera Vegetation Program | Ponderosa pine | 2012 | 27 | 2012, 2013, 2014, 2017, 2019 | Veg, Photos |
| USFS | South Holiday | Valles Caldera Vegetation Program | Ponderosa pine | | 10 | 2018 | Trees, shrubs, veg, fuels, photos |
| USFS | West Pino (San Juan Mesa) | Valles Caldera Vegetation Program | Ponderosa pine | 2017 | 10 | 2015, 2019 | Veg, Photos |
| VALL | Banco North | Valles Caldera Vegetation Program | Ponderosa pine | 2015, 2016, 2017, 2019 | 24 | 2012, 2015, 2016, 2017, 2018, 2019 | Veg, Photos |
| VALL | Banco North Weeds | Valles Caldera Vegetation Program | Ponderosa pine | 2015, 2016, 2017, 2019 | 11 | 2015, 2016, 2017, 2018, 2019, 2021 | Veg, Photos |
| VALL | Banco South | Valles Caldera Vegetation Program | Ponderosa pine | 2012 | 9 | 2004, 2006, 2007, 2008, 2009, 2010, 2011, 2012, 2013, 2014, 2015, 2016, 2018, 2020, 2021 | Veg, Photos |
| VALL | Seco North Unit 5 | Valles Caldera Vegetation Program | Mixed Conifer | 2017, 2019 | 10 | 2016, 2018, 2019, 2020, 2021 | Trees, shrubs, veg, fuels, photos |
| VALL | Redondo Meadow | Valles Caldera Vegetation Program | montane Grasslands | 2011, 2012 | 3 | 2010, 2011, 2014, 2015, 2016, 2017, 2018 | |

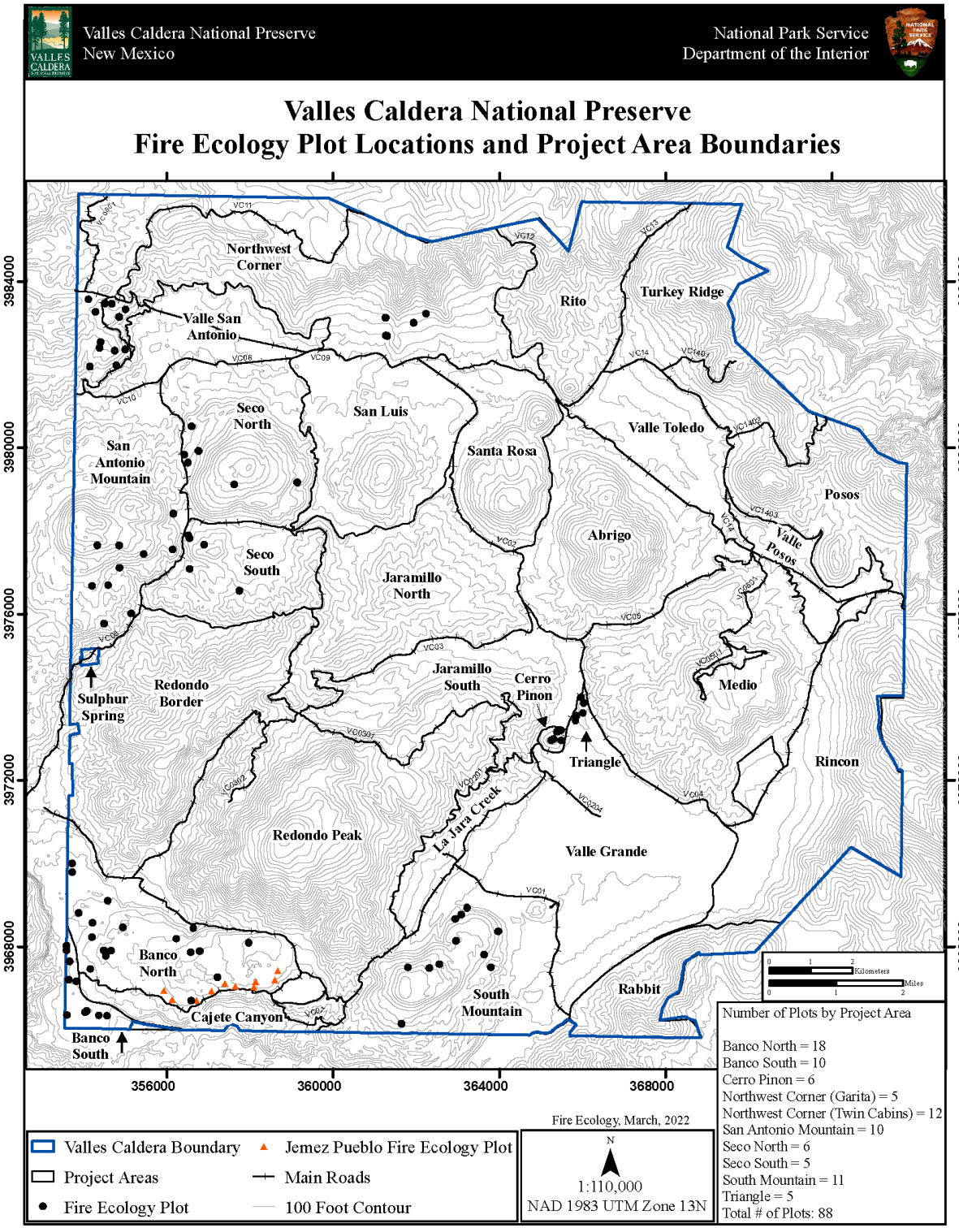



Figure 4.2.1. Pueblo and Four Winds Fire Ecology Plot Locations, Valles Caldera National Preserve

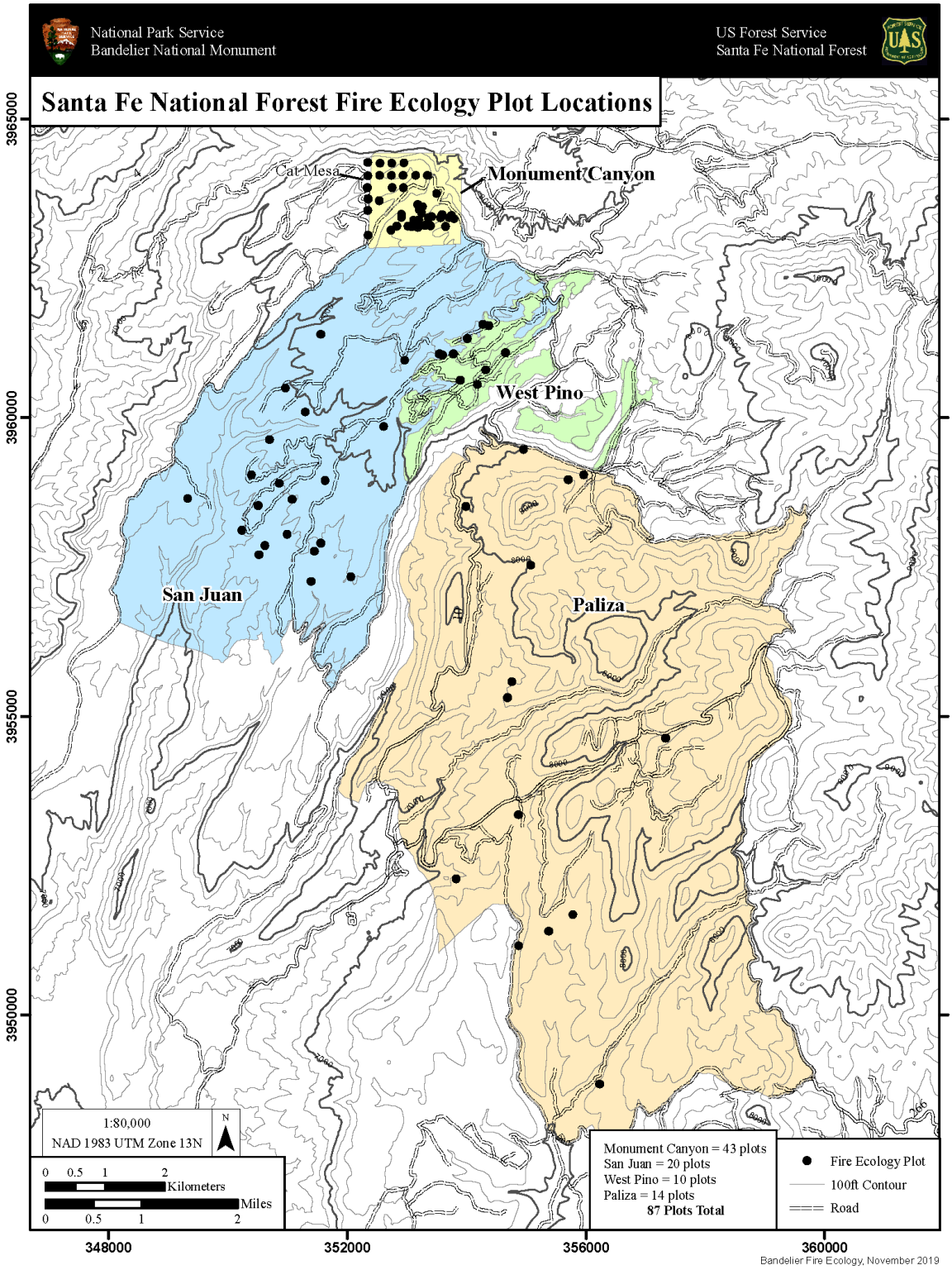



Figure 4.2.2. Pueblo and Four Winds Fire Ecology Plot Locations, Santa Fe National Forest

**Valles Caldera National Preserve, Banco North Project Area, Mixed-conifer Vegetation Community**

All vegetation and surface fuel plots (n=16) in Banco North were installed between 2015-2019, after forest thinning was completed between 2010-2014. The preburn data captures forest conditions after thinning operations but prior to prescribed fire.

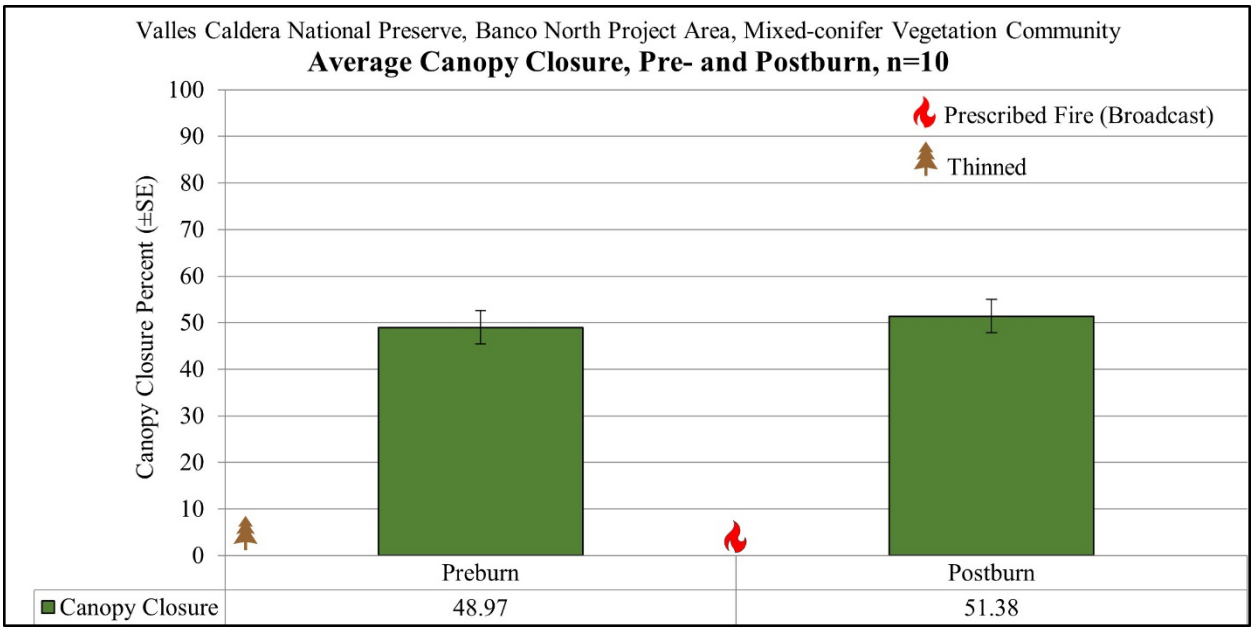



Figure 4.2.3. Valles Caldera National Preserve, Banco North Project Area, Average Canopy Closure. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a preburn average canopy closure percent of 48.97% (SE±3.55) and a postburn closure of 51.38% (SE±3.55). Canopy closure appears to have been maintained at or near the same level after the prescribed fire. Statistical analysis shows no evidence of a difference between preburn and postburn values (estimated difference 2.42, 95% CI: [-1.21, 6.04], p=0.167).

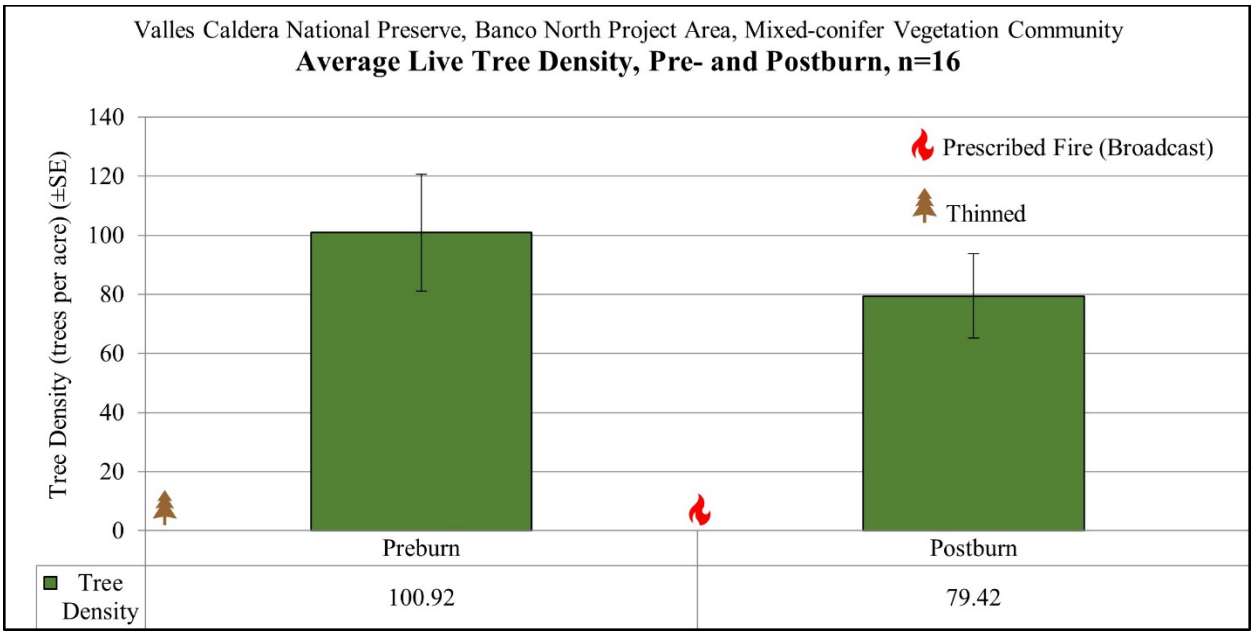



Figure 4.2.4. Valles Caldera National Preserve, Banco North Project Area, Average Live Tree Density. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a preburn average live tree density of 100.92 (SE±19.74) trees per acre and a postburn density of 79.42 (SE±14.26). Although there appears to be a reduction in live tree density between preburn and postburn values, there is no statistical evidence to support that assumption (estimated difference -21.50, 95% CI: [-47.85, 4.85], p=0.102).

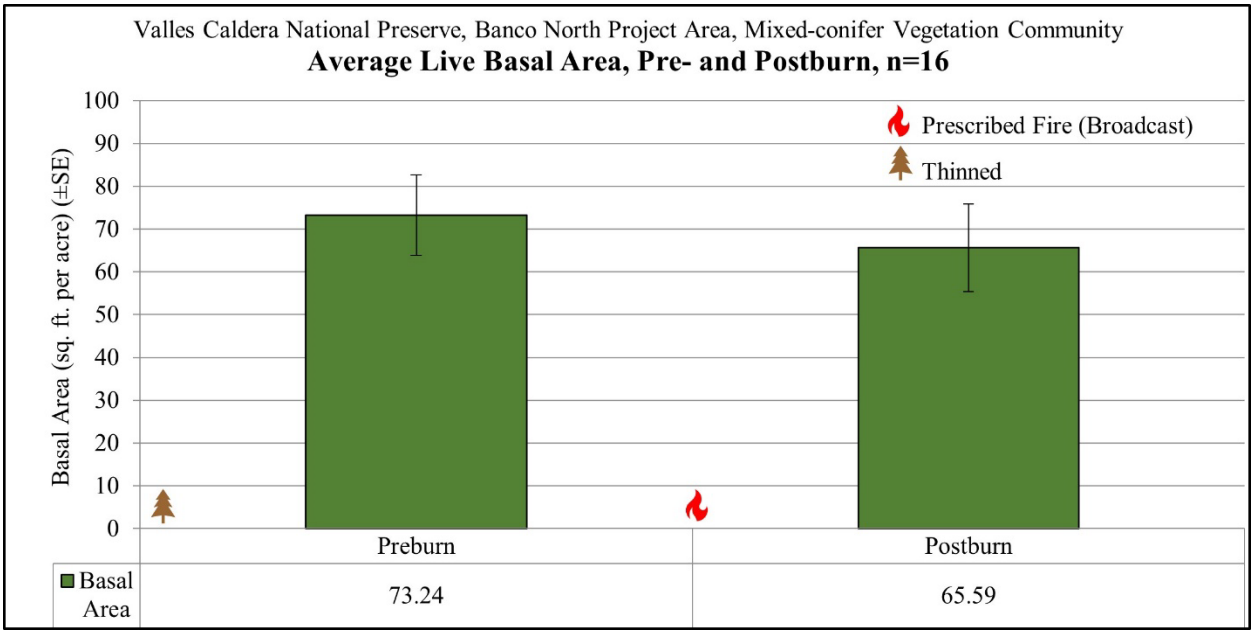



Figure 4.2.5. Valles Caldera National Preserve, Banco North Project Area, Average Live Basal Area. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a preburn average live basal area of 73.24 (SE±9.37) sq. ft. per acre and a postburn basal area of 65.59 (SE±10.27). Although there appears to be a small reduction in basal area between preburn and postburn values, there is no statistical evidence to support that assumption (estimated difference -7.65, 95% CI: [-20.11, 4.80], p=0.210).

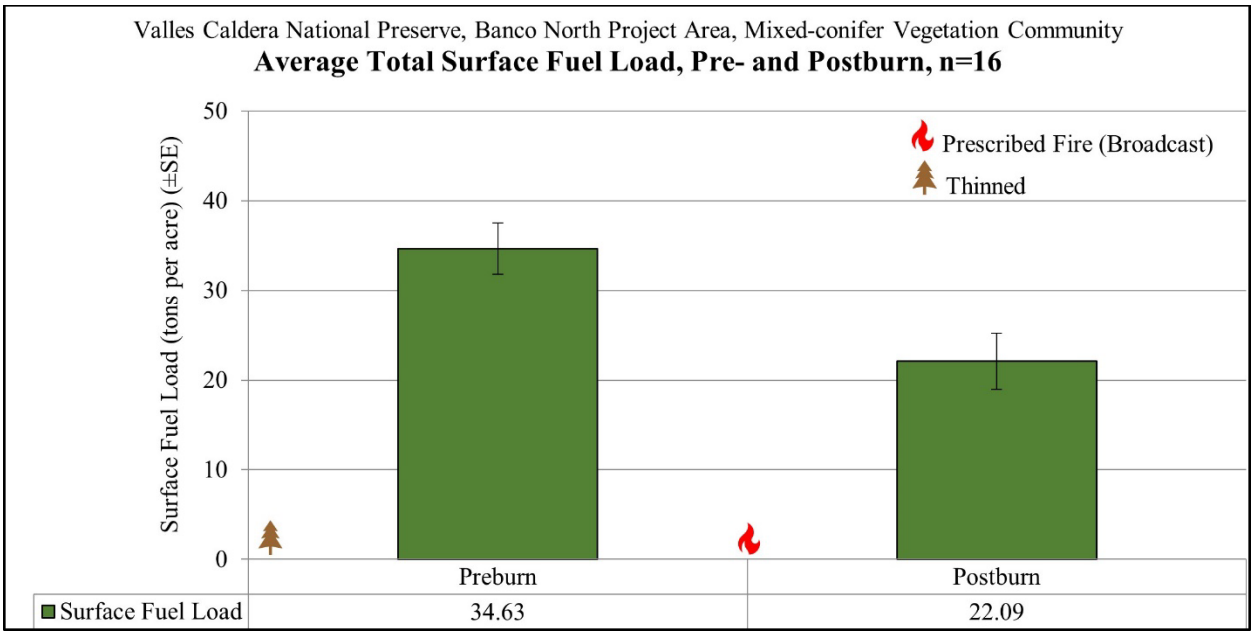



Figure 4.2.6. Valles Caldera National Preserve, Banco North Project Area, Average Total Surface Fuel Load. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a preburn average total surface fuel load of 34.63 (SE±2.86) tons per acre, higher than historical estimations for fuel loading in this vegetation community. Postburn average total surface fuel load is within the range of historical conditions at 22.09 (SE±3.14) tons per acre. Total surface fuel load appears to be reduced by 12.54 tons per acre after the prescribed fire, supported by statistical evidence of a difference between preburn and postburn values (95% CI: [-19.54, -5.55], p=0.002).




Figure 4.2.7. Preburn (left) and postburn (right) photos from Banco North Project Area, demonstrating reductions in surface fuel loading and smaller diameter trees. Photos: Pueblo and Four Winds Fire Ecology Program.

**Valles Caldera National Preserve, Cerro Pinyon Project Area, Mixed-conifer Vegetation Community**

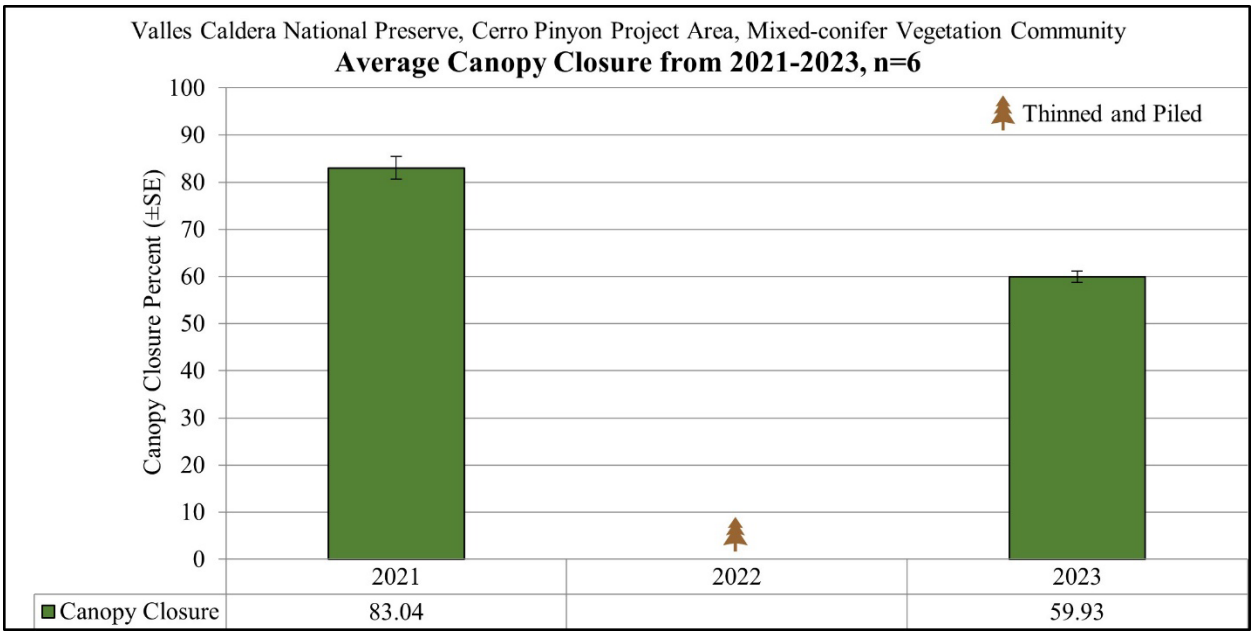



Figure 4.2.8. Valles Caldera National Preserve, Cerro Pinyon Project Area, Average Canopy Closure. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average canopy closure of 83.04% (SE±2.43) and a post-thinning closure of 59.93% (SE±1.18), within the desired management goal range of 50 to 85%. Canopy closure appears to be reduced by 27.83% after thinning operations, supported by analysis that shows evidence of a difference between pre-thinning and post-thinning values (estimated difference -23.11, 95% CI: [-30.05, 16.18], p=0.0004).

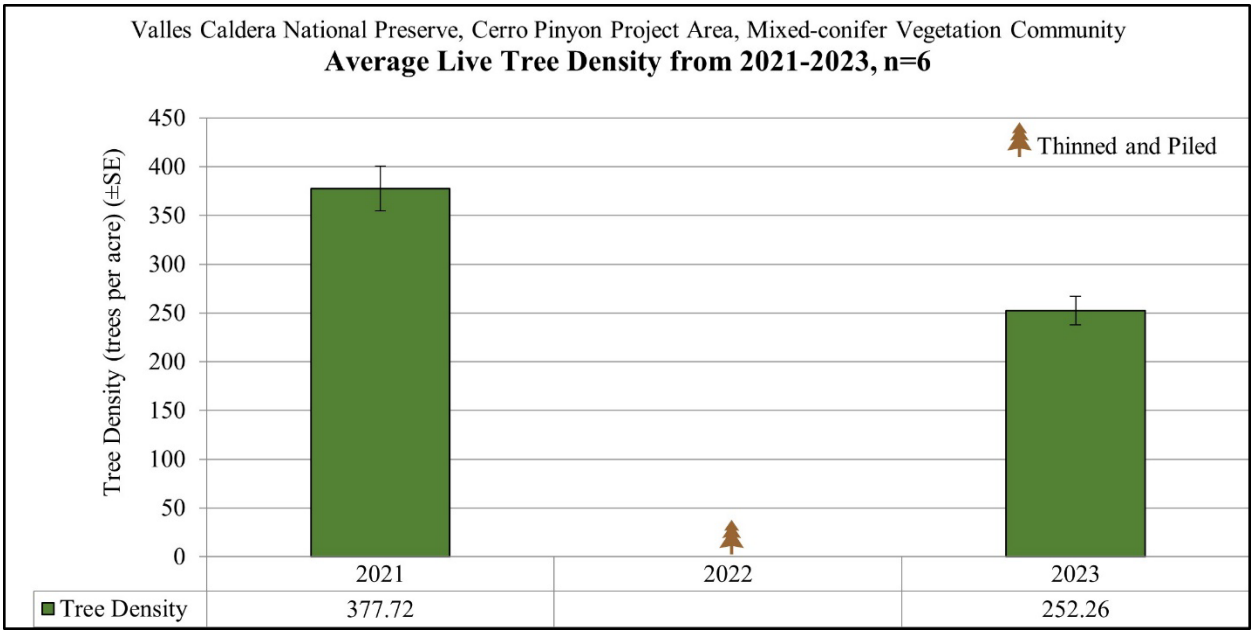



Figure 4.2.9. Valles Caldera National Preserve, Cerro Pinyon Project Area, Average Live Tree Density. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live tree density of 377.72 (SE±22.96) trees per acre, higher than the estimated range of historical tree densities in this vegetation community. Post-thinning tree density is recorded at 252.26 (SE±14.69) trees per acre, within the range of estimated historical tree densities. There appears to be a 125.46 trees per acre reduction in live tree density after thinning operations. Statistical analysis shows evidence of a difference between pre-thinning and post-thinning values (95% CI: [-182.50, -68.41], p=0.002).




Figure 4.2.10. Pre- (left) and post-thinning (right) photos from Cerro Pinyon Project Area, showing reductions in small diameter trees (ladder fuels). Photos: Pueblo and Four Winds Fire Ecology Program.

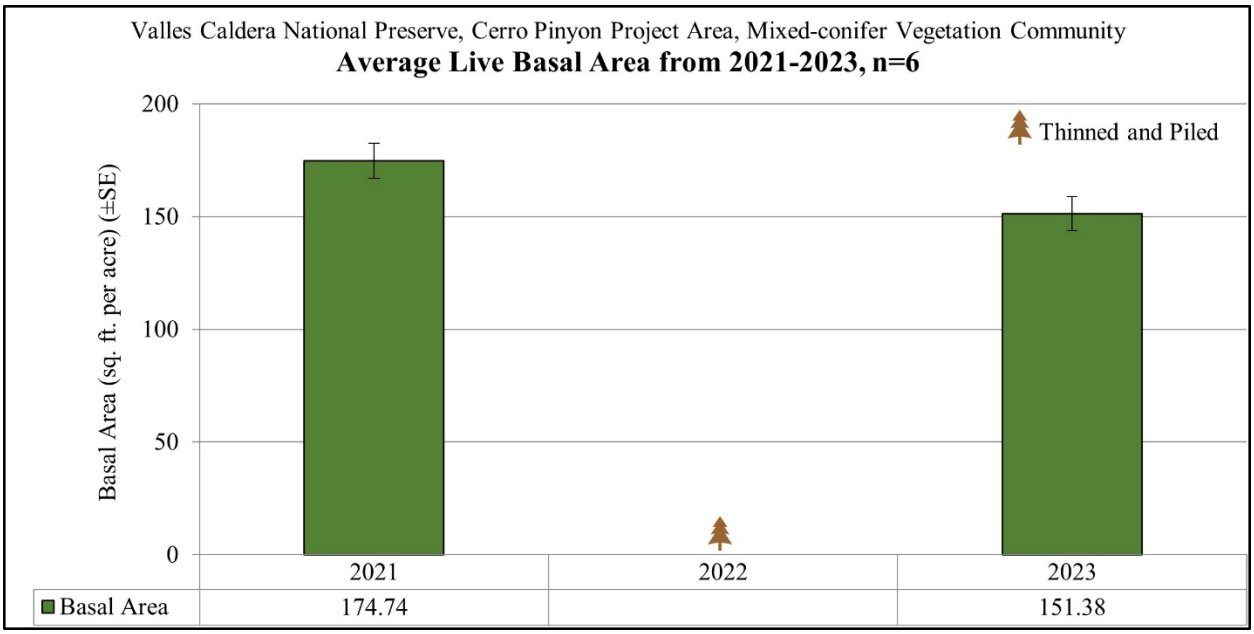



Figure 4.2.11. Valles Caldera National Preserve, Cerro Pinyon Project Area, Average Live Basal Area. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live basal area of 174.74 (SE±7.87) sq. ft. per acre and a post-thinning basal area of 151.38 (SE±7.51). There appears to be a 23.37 sq. ft. per acre reduction in basal area after thinning operations, supported by statistical evidence of a difference between pre-thinning and post-thinning values (95% CI: [-37.82, -8.91], p=0.009).

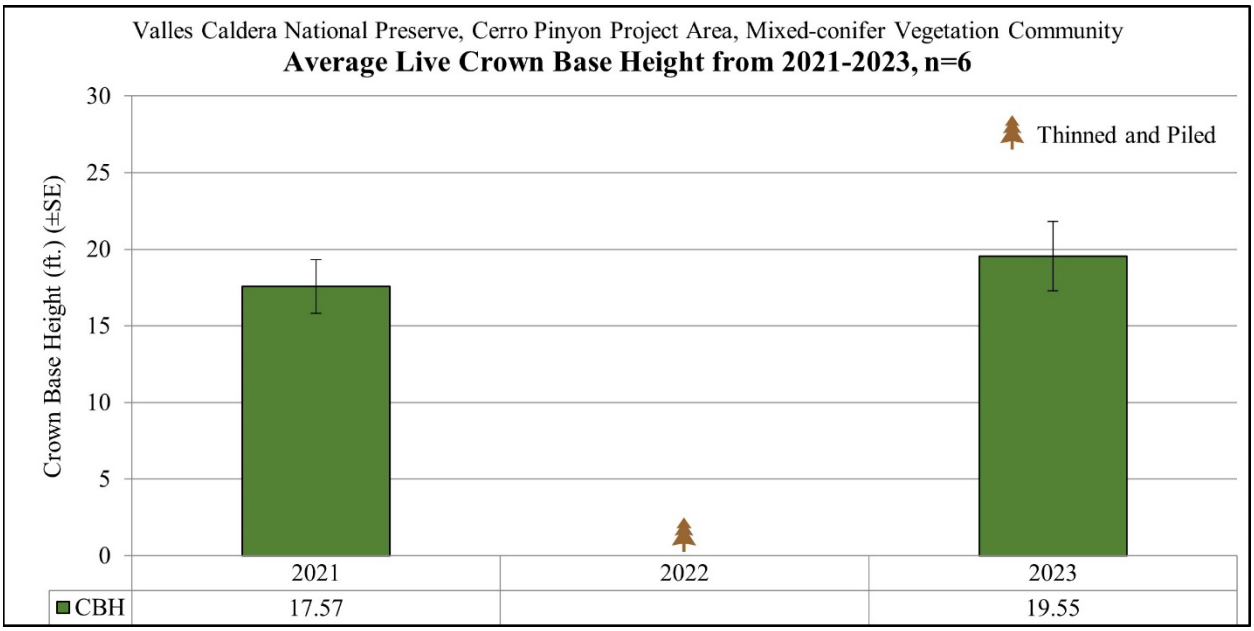



Figure 4.2.12. Valles Caldera National Preserve, Cerro Pinyon Project Area, Average Live Crown Base Height. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live crown base height of 17.57 (SE±1.75) ft. and a post-thinning crown base height of 19.55 (SE±2.25) ft.. There appears to be a 1.98 ft. increase in live crown base height after thinning operations, as desired to reduce potential for fire spreading from the forest surface to the tree canopy. There is statistical evidence to support a difference between pre- and post-thinning values (95% CI: [0.06, 3.90], p=0.0452).

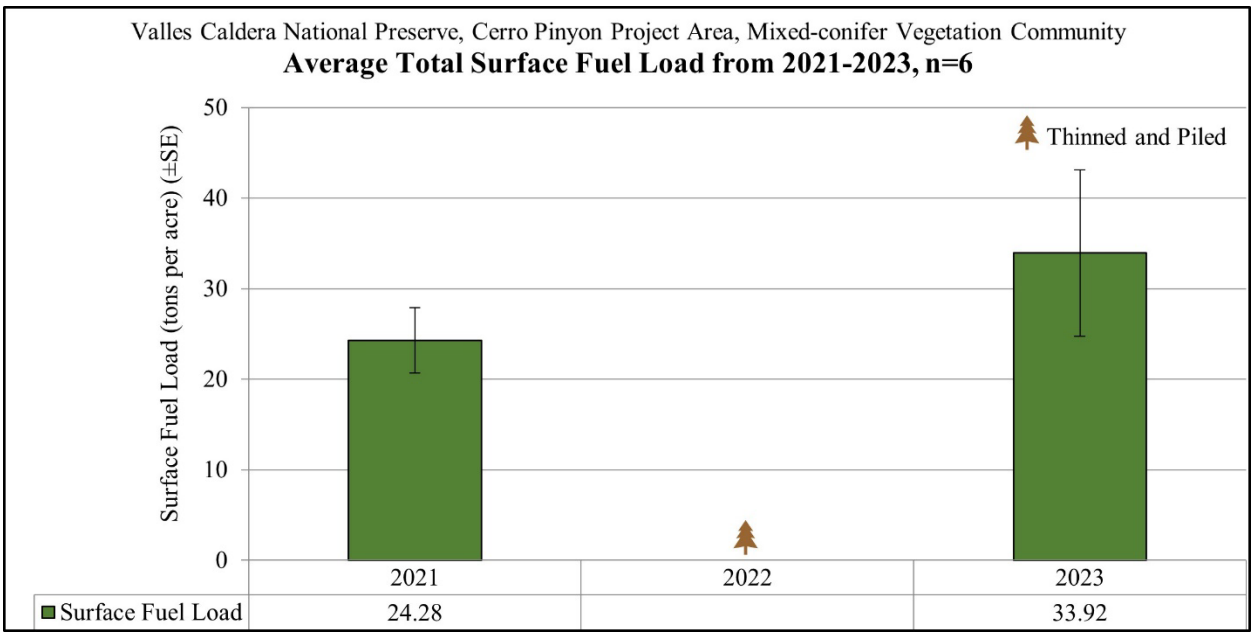



Figure 4.2.13. Valles Caldera National Preserve, Cerro Pinyon Project Area, Average Total Surface Fuel Load. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average total surface fuel load of 24.28 (SE±3.62) tons per acre, slightly higher than historical estimations for fuel loading in this vegetation community. Post-thinning average total surface fuel load is reported at 33.92 (SE±9.20) tons per acre. Total surface fuel load appears to have increased after thinning, although there is no statistical evidence to support that assumption (estimated difference 9.64, 95% CI: [6.12, 25.40], p=0.177). Thinned materials were placed in piles and haven't burned yet, creating the possibility of increased surface fuel loads. Piles are scheduled to burn in 2025 and 2026 and additional fuel loading data will be collected and analyzed at that time.

**Valles Caldera National Preserve, Garita Project Area, Twin Cabins South Unit, Mixed-conifer Vegetation Community**

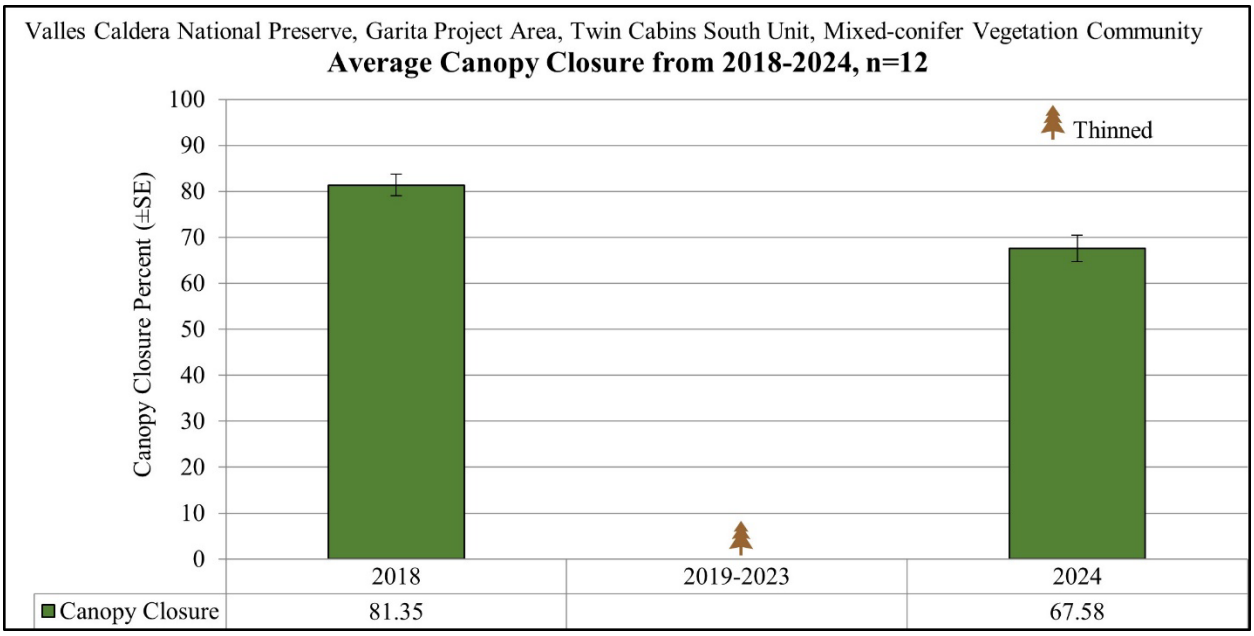



Figure 4.2.14. Valles Caldera National Preserve, Garita Project Area, Twin Cabins South Unit, Average Canopy Closure. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average canopy closure of 81.35% (SE±2.33) and a post-thinning closure of 67.58% (SE±2.90), within the desired management goal range of 50 to 85%. Canopy closure appears to be reduced by 16.93% after thinning operations. Statistical analysis shows evidence of a difference between values (estimated difference -13.77, 95% CI: [-19.64, -7.90], p=0.0003).

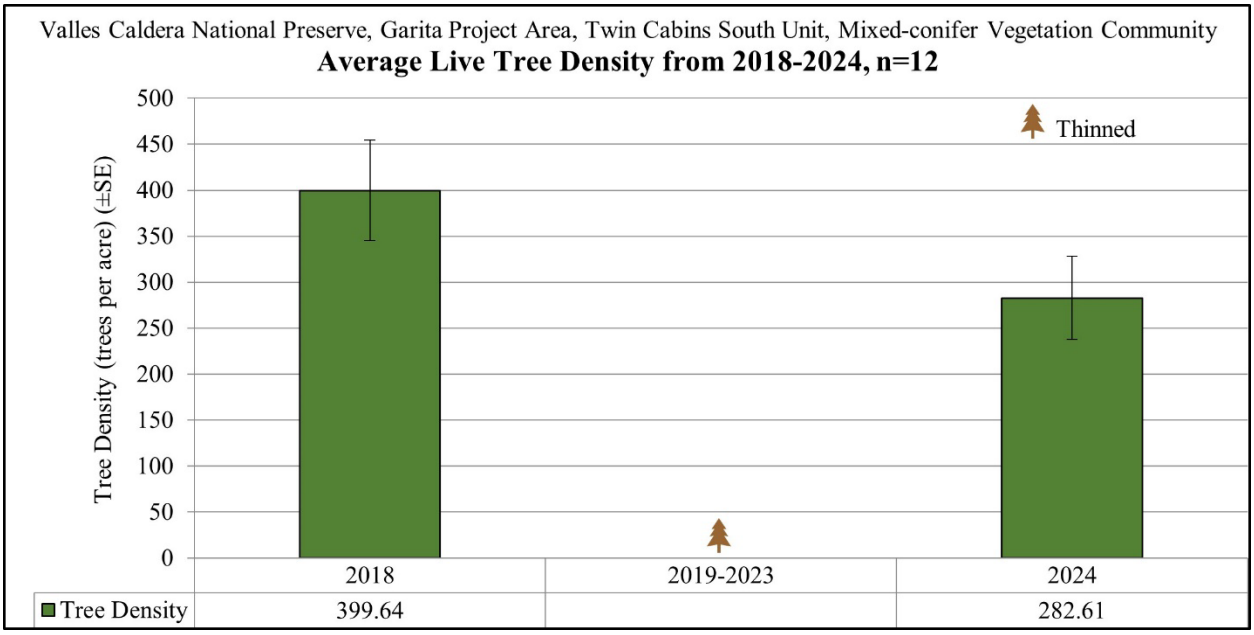



Figure 4.2.15. Valles Caldera National Preserve, Garita Project Area, Twin Cabins South Unit, Average Live Tree Density. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live tree density of 399.64 (SE±54.66) trees per acre. Post-thinning tree density is 282.61 (SE±45.21) trees per acre. Both are higher than the estimated range of historical tree densities in this vegetation community. There appears to be a 117.02 trees per acre reduction in live tree density after thinning operations, supported by statistical evidence of a difference between values (95% CI: [-194.29, -39.76], p=0.007).




Figure 4.2.16. Pre- (left) and post-thinning (right) photos from Garita Project Area, Twin Cabins South Unit showing reductions in small diameter trees (ladder fuels). Photos: Pueblo and Four Winds Fire Ecology Program.

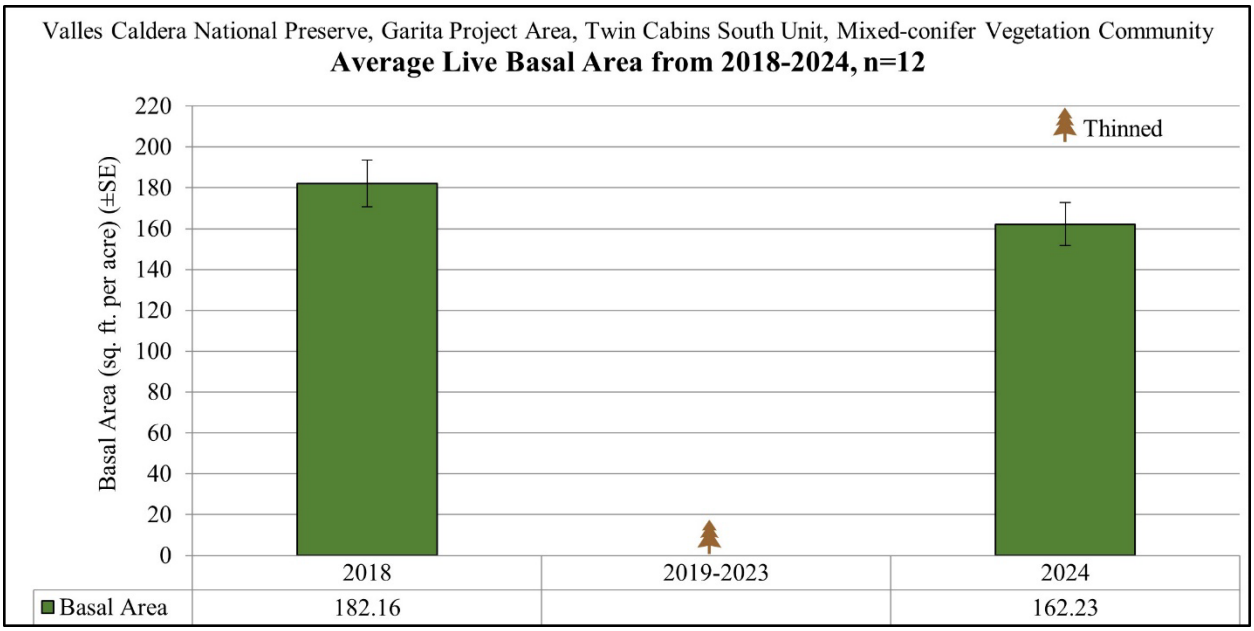



Figure 4.2.17. Valles Caldera National Preserve, Garita Project Area, Twin Cabins South Unit, Average Live Basal Area. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live basal area of 182.16 (SE±11.46) sq. ft. per acre and a post-thinning basal area of 162.23 (SE±10.40). There appears to be a 19.93 sq. ft. per acre reduction in basal area between pre-thinning and post-thinning values, supported by statistical evidence (95% CI: [-31.05, 8.81], p=0.002).

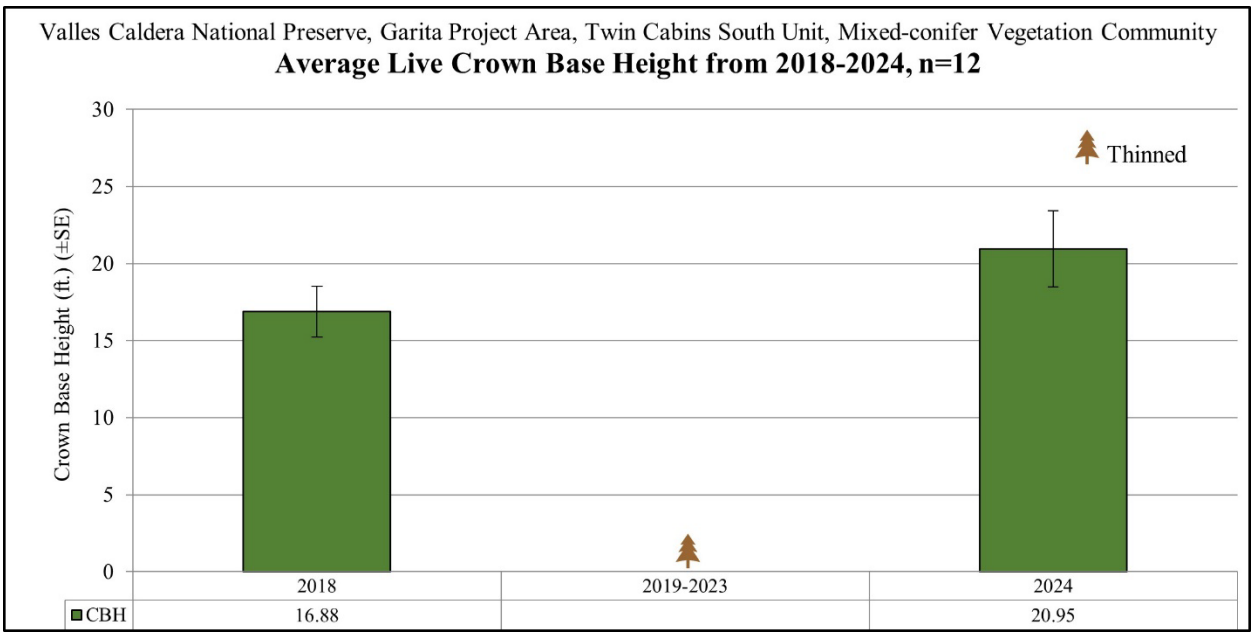



Figure 4.2.18. Valles Caldera National Preserve, Garita-Twin Cabins South Project Area, Average Live Crown Base Height. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live crown base height of 16.88 (SE±1.65) ft. and a post-thinning crown base height of 20.95 (SE±2.46). There appears to be a 4.06 ft. increase in live crown base height after thinning operations, as desired to reduce potential for fire spreading from the forest surface to the tree canopy. There is statistical evidence to support a difference between pre- and post-thinning values (95% CI: [-6.42, -1.70], p=0.003).

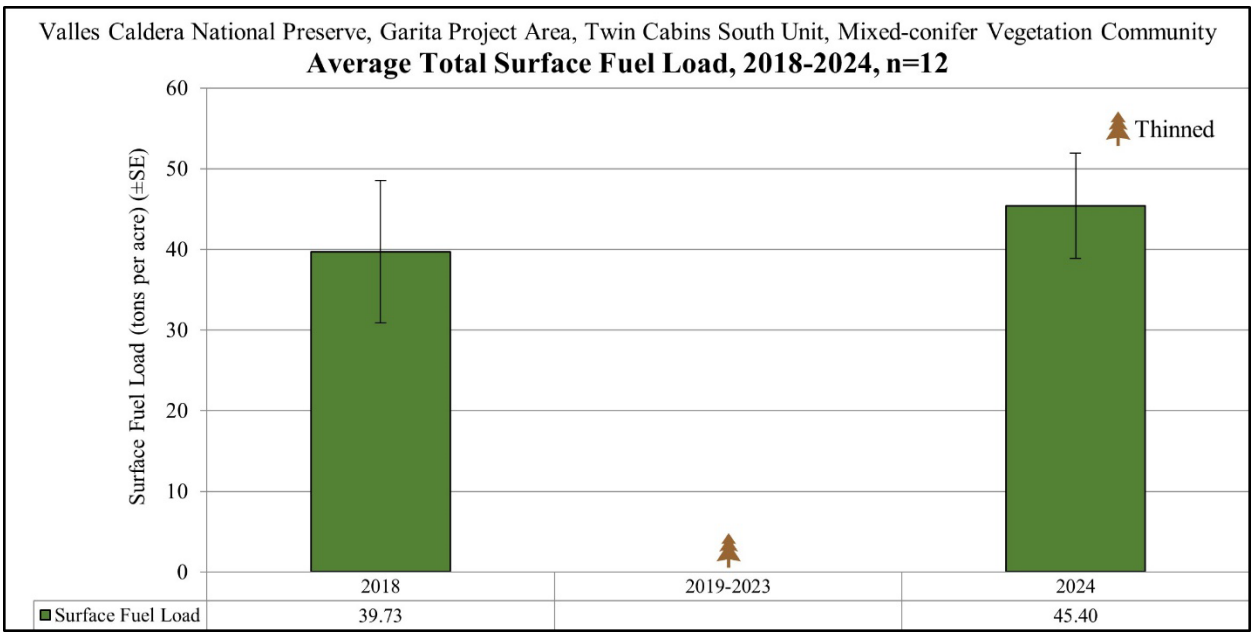



Figure 4.2.19. Valles Caldera National Preserve, Garita-Twin Cabins South Project Area, Average Total Surface Fuel Load. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average total surface fuel load of 39.73 (SE±8.82) tons per acre. Post-thinning average total surface fuel load is reported at 45.40 (SE±6.56) tons per acre. Both are higher than historical estimations for fuel loading in this vegetation community. There appears to be a 5.67 tons per acre increase in total surface fuel load after thinning operations. However, statistical analysis shows no evidence of a difference between pre- and post-thinning values (95% CI: [-6.26, 17.61], p=0.318).

**Valles Caldera National Preserve, Redondo Meadow, Montane Grassland Vegetation Community**

All plots (n=3) in Redondo Meadow were installed in 2010. Forest thinning was completed in 2010 and a prescribed fire was completed in ~2012. Data were collected until 2018.

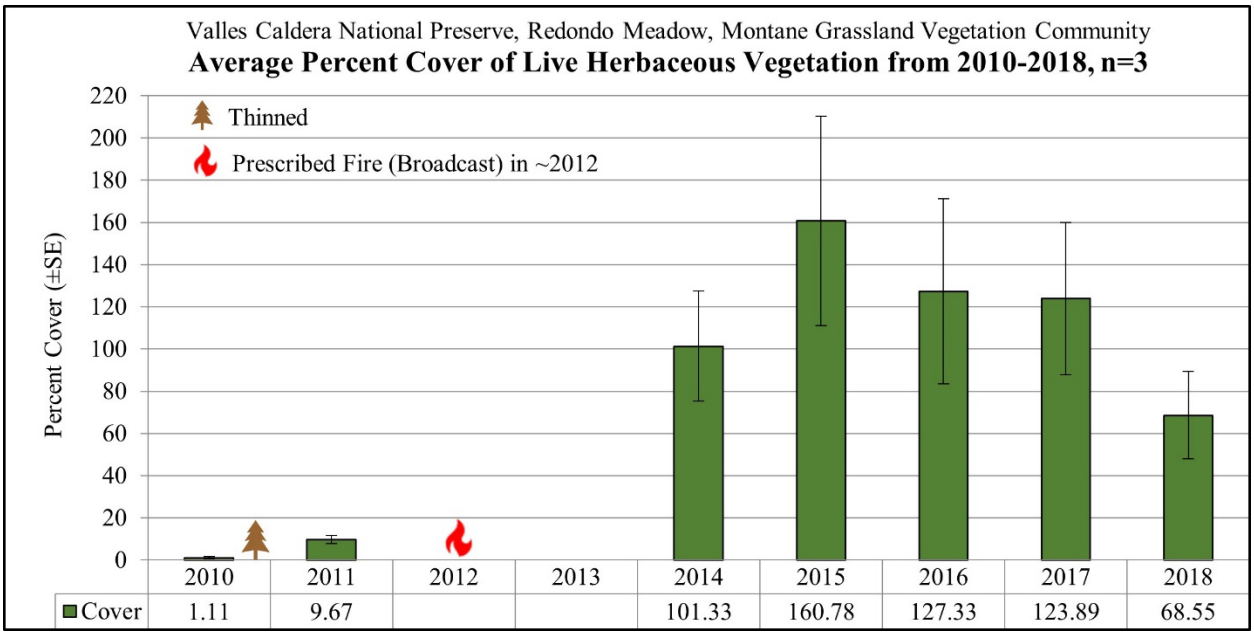
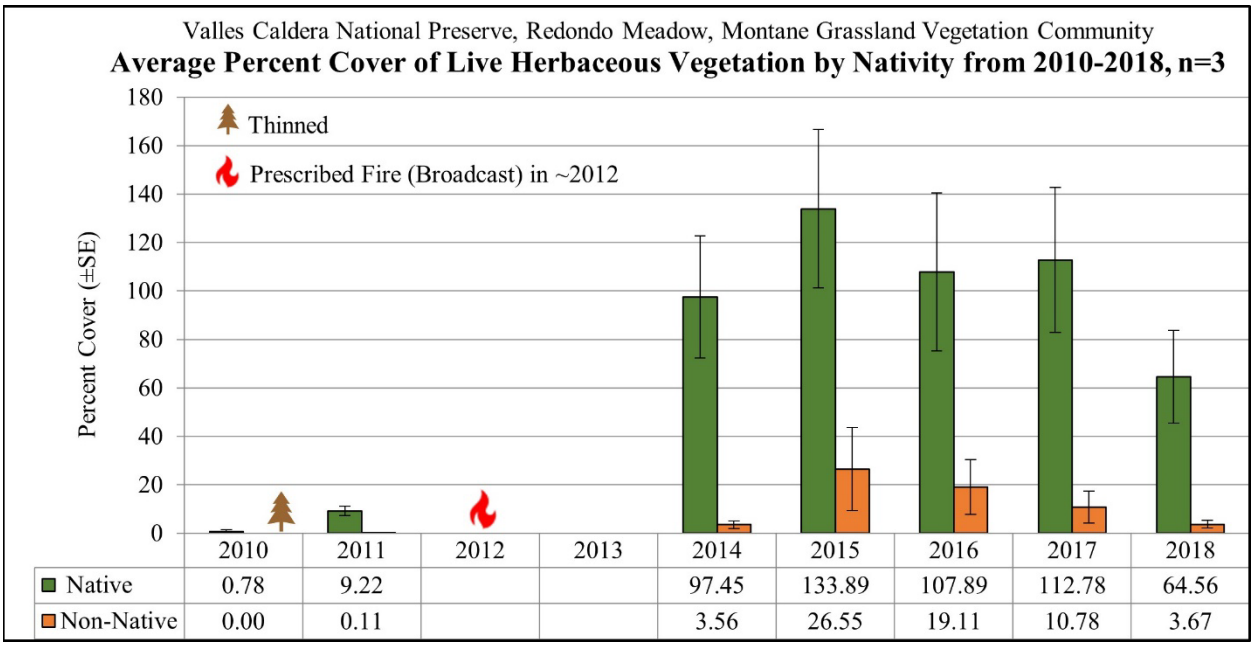



Figure 4.2.20. Valles Caldera National Preserve, Redondo Meadow, Average Percent Cover of Live Herbaceous Vegetation. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average live herbaceous cover of 1.11% (SE±0.62) and a post-thinning average cover of 9.67% (SE±1.89). Although live herbaceous cover appears to have increased after thinning operations, statistical analysis does not show evidence of a difference between pre-thinning and post-thinning values (estimated difference -8.56, 95% CI: [0111.75, 94.64], p=0.99993). Postburn average herbaceous vegetation increased to 101.33% (SE±26.17) in 2014 and peaked in 2015 with average cover of 160.78% (SE±49.58). Statistical analysis shows evidence of differences between preburn (2011) herbaceous cover and postburn years 2015 (estimate -151.11, CI: [-254.31, -47.92], p=0.003), 2016 (estimate -117.67, CI: [-220.86, -14.47], p=0.022) and 2017 (estimate -114.22, CI: [-217.42, -11.02], p=0.027). The postburn increase in herbaceous cover is likely due to nutrient cycling initiated by prescribed fire, with the greatest effects occurring in the first few years after treatment. It is of note that herbaceous vegetation cover is greatly influenced by precipitation.




| | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 |
|---|---|---|---|---|---|---|---|---|---|
| Native | 0.78 | 9.22 | | | 97.45 | 133.89 | 107.89 | 112.78 | 64.56 |
| Non-Native | 0.00 | 0.11 | | | 3.56 | 26.55 | 19.11 | 10.78 | 3.67 |

Figure 4.2.21. Valles Caldera National Preserve, Redondo Meadow, Average Percent Cover of Live Herbaceous Vegetation by Nativity. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average live herbaceous native cover of 0.78% (SE±0.62) and no non-native live herbaceous vegetation detected. Post-thinning native cover increased to 9.22% (SE±1.97) and non-native cover increased to 0.11% (SE±0.11), however statistical analysis shows no evidence of a difference between pre- and post-thinning live herbaceous percent cover for both native vegetation (95% CI: [-84.35, 67.46], p=0.9996) and non-native vegetation (95% CI: [-33.98, 33.67], p=1.000).

In 2014, summary statistics show a postburn average live herbaceous native cover of 97.45% (SE±25.23) and live herbaceous non-native cover of 3.56% (SE±1.55). Both native and non-native average live herbaceous cover peaked in 2015, about three years postburn, with average native cover at 133.89% (SE±32.74) and average non-native cover at 26.55% (SE±17.14). This increase is likely due to nutrient cycling initiated by prescribed fire. By 2018, average native cover was 64.56% (SE±19.21) and non-native cover was 3.67% (SE±1.57). Statistical analysis shows evidence of a difference between preburn (2011) and postburn native herbaceous vegetation cover for 2014 (estimate -88.23, CI: [-164.13, -12.33], p=0.019), 2015 (estimate -124.64, CI: [-200.57, -48.77], p=0.001), 2016 (estimate -98.67, CI: [-174.57, -22.76], p=0.009) and 2017 (estimate -103.56, CI: [-179.46, -27.65], p=0.006). Statistical analysis shows no evidence of a difference between 2011 preburn and 2018 postburn native cover (estimated difference -55.34, 95% CI: [-131.2, 20.57], p=0.223) although it appears to have remained elevated from preburn values.

While non-native vegetation cover appears to have increased after both thinning and prescribed burning treatments, there is no evidence of a difference between any of the years (Table 4.2.2.).

Table 4.2.2. Contrast tables for native and non-native live herbaceous vegetation in Redondo Meadow. Comparisons between relevant years are denoted in the table below, for example, year 2010-year 2011 compares the averages between the two years.

| *Native Live Herbaceous Vegetation* | | | | |
|---|---|---|---|---|
| **Contrast** | **Estimated Difference** | **CI-low** | **CI-high** | **P-value** |
| year 2010 - year 2011 | -8.44 | -84.35 | 67.46 | 0.9996 |
| year 2011 - year 2014 | -88.23 | -164.13 | -12.33 | 0.019424 |
| year 2011 - year 2015 | -124.67 | -200.57 | -48.77 | 0.00131 |
| year 2011 - year 2016 | -98.67 | -174.57 | -22.76 | 0.008762 |
| year 2011 - year 2017 | -103.56 | -179.46 | -27.65 | 0.006065 |
| year 2011 - year 2018 | -55.34 | -131.20 | 20.57 | 0.223238 |
| *Non-Native Live Herbaceous Vegetation* | | | | |
| **Contrast** | **Estimated Difference** | **CI-low** | **CI-high** | **P-value** |
| year 2010 - year 2011 | -0.11 | -33.89 | 33.67 | 1.00000 |
| year 2010 - year 2014 | -3.56 | -37.33 | 30.22 | 0.999708 |
| year 2010 - year 2015 | -26.55 | -60.33 | 7.22 | 0.165852 |
| year 2010 - year 2016 | -19.11 | -52.89 | 14.67 | 0.471396 |
| year 2010 - year 2017 | -10.78 | -44.56 | 23.00 | 0.910987 |
| year 2010 - year 2018 | -3.67 | -37.44 | 30.11 | 0.999652 |
| year 2011 - year 2014 | -3.45 | -37.22 | 30.33 | 0.999756 |
| year 2011 - year 2015 | -26.44 | -60.22 | 7.33 | 0.168749 |
| year 2011 - year 2016 | -19.00 | -52.78 | 14.78 | 0.47751 |
| year 2011 - year 2017 | -10.67 | -44.45 | 23.11 | 0.914677 |
| year 2011 - year 2018 | -3.56 | -37.33 | 30.22 | 0.999708 |

**Valles Caldera National Preserve, San Antonio Mountain Project Area, Mixed-conifer Vegetation Community**

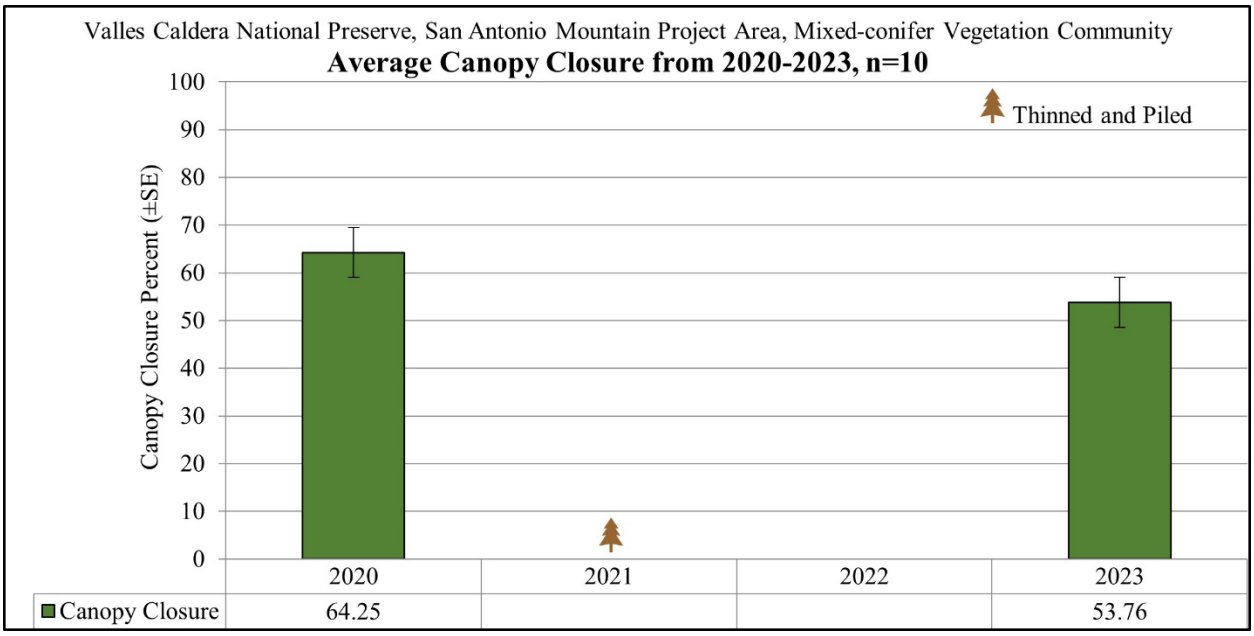



Figure 4.2.22. Valles Caldera National Preserve, San Antonio Mountain Project Area, Average Canopy Closure. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average canopy closure of 64.25% (SE±3.04) and a post-thinning closure of 53.76% (SE±5.64), within the desired management goal range of 50 to 85%. Canopy closure appears to be reduced by 16.35% after thinning operations. Statistical analysis shows evidence of a difference between values (estimated difference -10.49, 95% CI: [-18.60, -2.38], p=0.017).

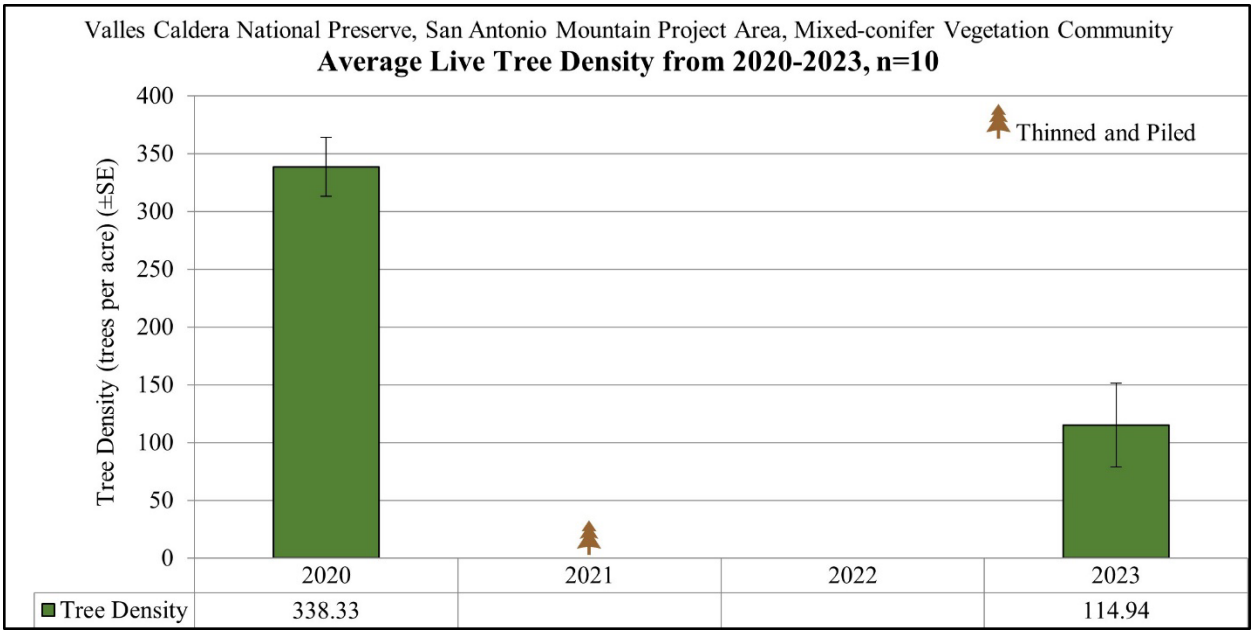



Figure 4.2.23. Valles Caldera National Preserve, San Antonio Mountain Project Area, Average Live Tree Density. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live tree density of 338.33 (SE±25.56) trees per acre, higher than the estimated range of historical tree densities in this vegetation community. Post-thinning tree density is 114.94 (SE±36.33) trees per acre, within the estimated range of historical tree densities. There appears to be a 223.39 trees per acre reduction in live tree density after thinning operations, supported by statistical evidence of a difference between values (95% CI: [-309.37, -137.41], p=0.0002).



Figure 4.2.24. Pre- (left) and post-thinning (right) photos from San Antonio Mountain Project Area, showing reductions in small diameter trees (ladder fuels) and seedling trees. Photos: Pueblo and Four Winds Fire Ecology Program.

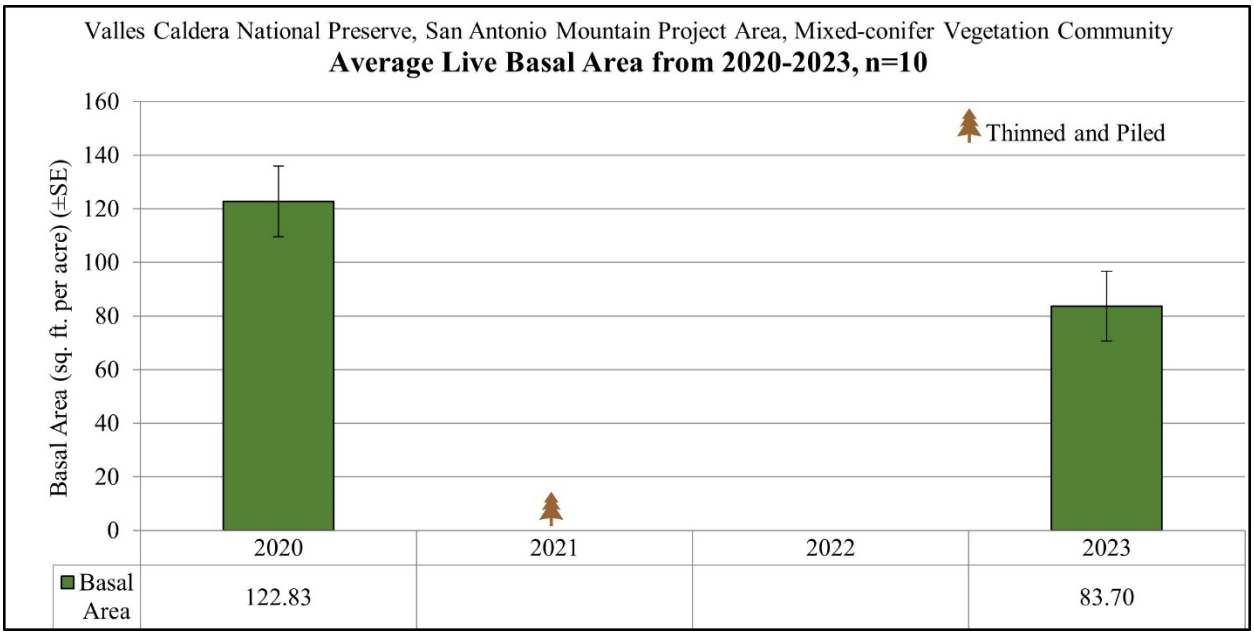



Figure 4.2.25. Valles Caldera National Preserve, San Antonio Mountain Project Area, Average Live Basal Area. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live basal area of 122.83 (SE±13.23) sq. ft. per acre and a post-thinning basal area of 83.70 (SE±13.00). There appears to be a 39.13 sq. ft. per acre reduction in basal area between pre-thinning and post-thinning values, supported by statistical evidence (95% CI: [-58.83, -19.43], p=0.002).

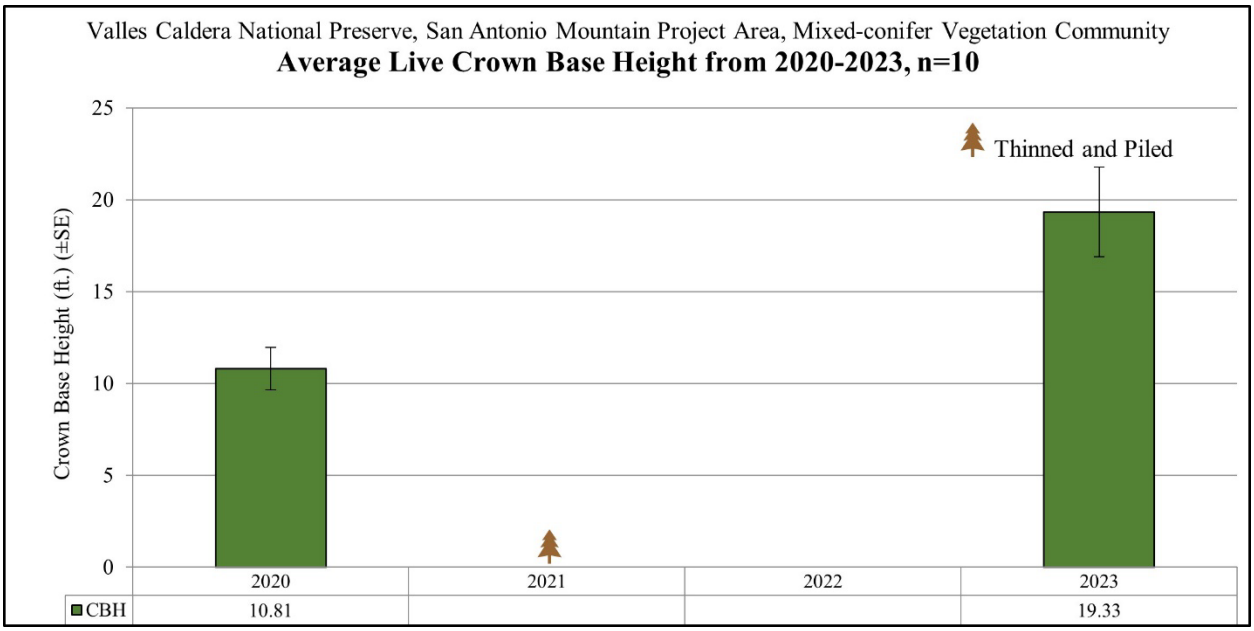



Figure 4.2.26. Valles Caldera National Preserve, San Antonio Mountain Project Area, Average Live Crown Base Height. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live crown base height of 10.81 (SE±1.16) ft. and a post-thinning crown base height of 19.33 (SE±2.43). There appears to be an 8.52 ft. increase in live crown base height after thinning operations, as desired to reduce potential for fire spreading from the forest surface to the tree canopy. There is statistical evidence to support a difference between pre- and post-thinning values (95% CI: [4.70, 12.35], p=0.0007).

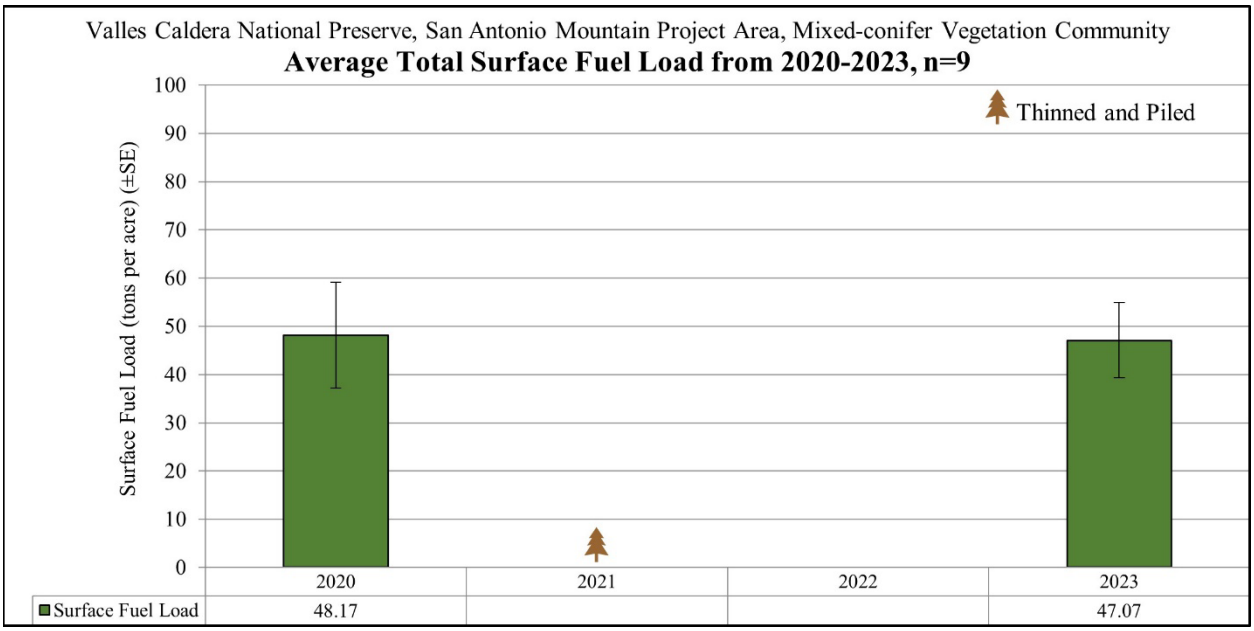



Figure 4.2.27. Valles Caldera National Preserve, San Antonio Mountain Project Area, Average Total Surface Fuel Load. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average total surface fuel load of 48.17 (SE±10.94) tons per acre, higher than historical estimations for fuel loading in this vegetation community. Post-thinning average total surface fuel load is reported at 47.07 (SE±7.78) tons per acre. There appears to be no change in total surface fuel load after thinning operations. Statistical analysis shows no evidence of a difference between values (estimated difference -1.11, 95% CI: [-23.01, 20.79], p=0.911). Thinned materials were placed in piles and pile burning was initiated in the project area. Additional fuel loading data will be collected after pile burning is completed.

**Valles Caldera National Preserve, Seco North and South Project Areas, Mixed-conifer Vegetation Community**

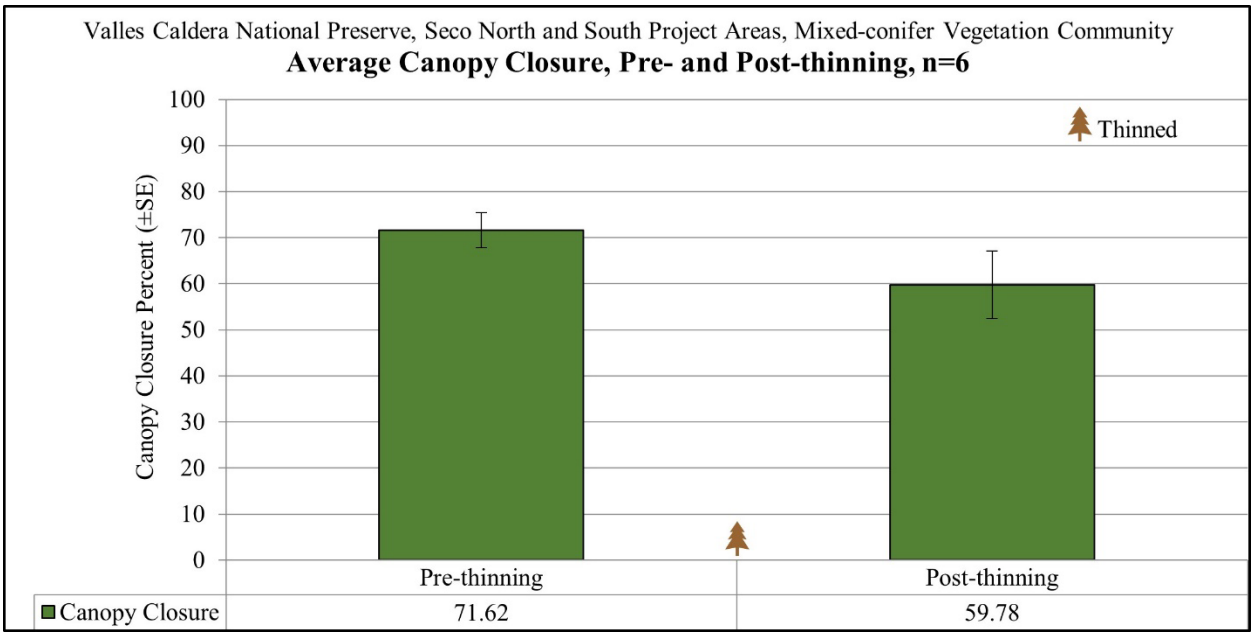



Figure 4.2.28. Valles Caldera National Preserve, Seco North and South Project Area, Average Canopy Closure. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average canopy closure of 71.62% (SE±3.84) and a post-thinning closure of 59.78% (SE±7.36), within the desired management goal range of 50 to 85%. Canopy closure appears to be reduced after thinning operations; however, statistical analysis does not show evidence of a difference between pre- and post-thinning values (estimated difference -11.84, 95% CI: [-32.84, 9.17], p=0.207).

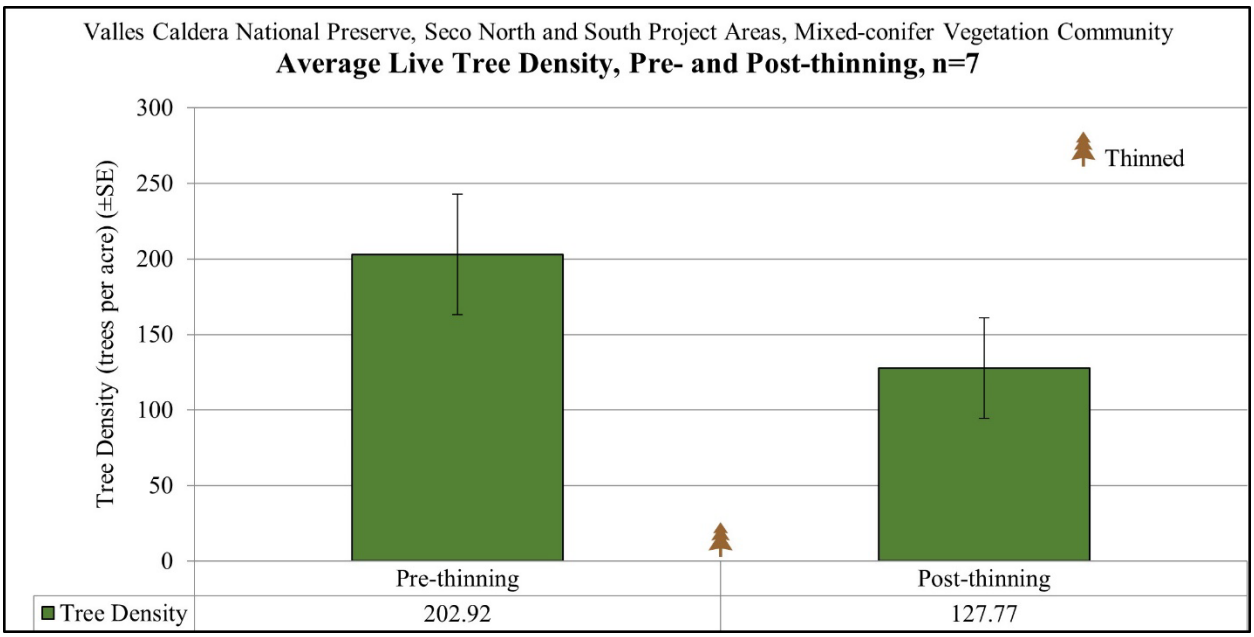



Figure 4.2.29. Valles Caldera National Preserve, Seco North and South Project Area, Average Live Tree Density. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live tree density of 202.92 (SE±39.91) trees per acre. Post-thinning tree density is 127.77 (SE±33.28) trees per acre. Both are within the estimated range of historical tree densities. There appears to be a 75.15 trees per acre reduction in live tree density after thinning operations, supported by statistical evidence of a difference between values (95% CI: [-150.94, 0.63], p=0.051).

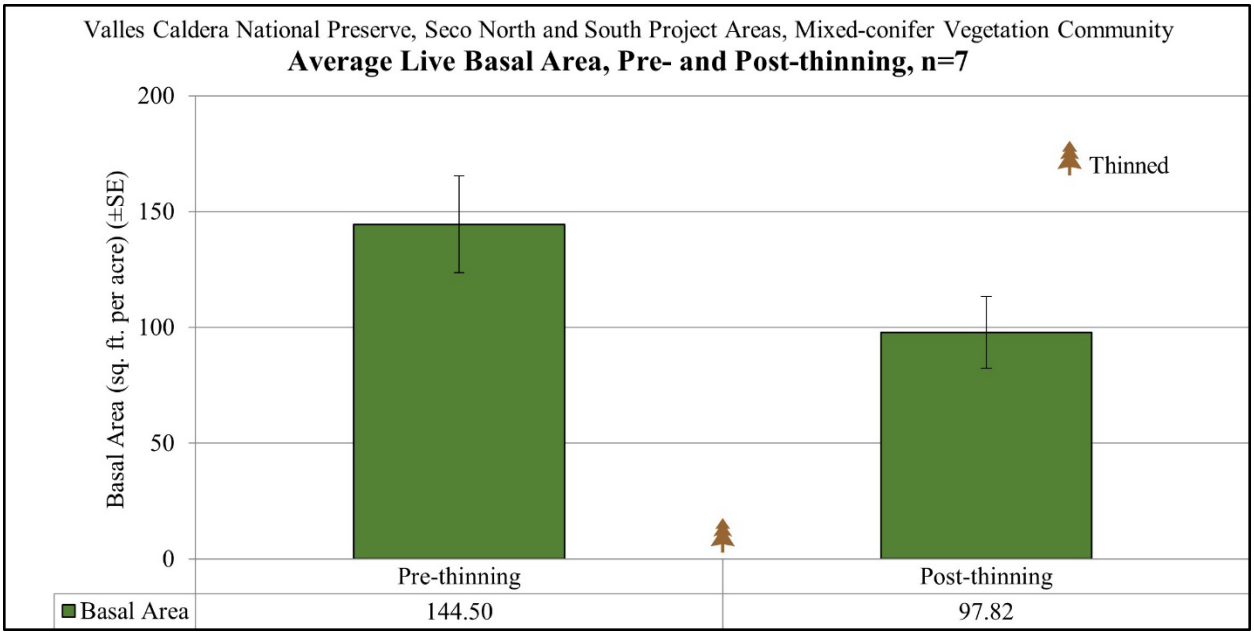



Figure 4.2.30. Valles Caldera National Preserve, Seco North and South Project Area, Average Live Basal Area. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live basal area of 144.50 (SE±20.99) sq. ft. per acre and a post-thinning basal area of 97.82 (SE±15.61). There appears to be a 46.68 sq. ft. per acre reduction in basal area between pre-thinning and post-thinning values, marginally supported by statistical evidence (95% CI: [-99.62, 6.27], p=0.074).

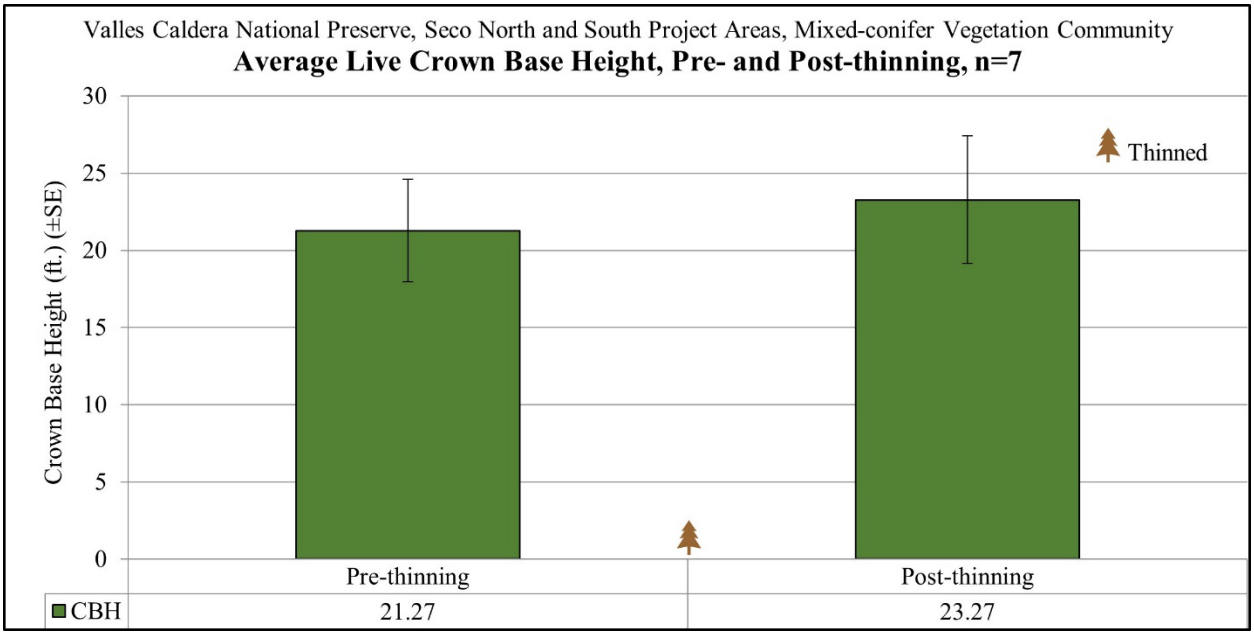



Figure 4.2.31. Valles Caldera National Preserve, Seco North and South Project Area, Average Live Crown Base Height. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average live crown base height of 21.27 (SE±3.33) ft. and a post-thinning crown base height of 23.27 (SE±4.13). There appears to be a slight increase in live crown base height after thinning operations, as desired to reduce potential for fire spreading from the forest surface to the tree canopy. However, there is no statistical evidence to support a difference between pre- and post-thinning values (Estimated difference 2.00, 95% CI: [-2.67, 6.68], p=0.335).

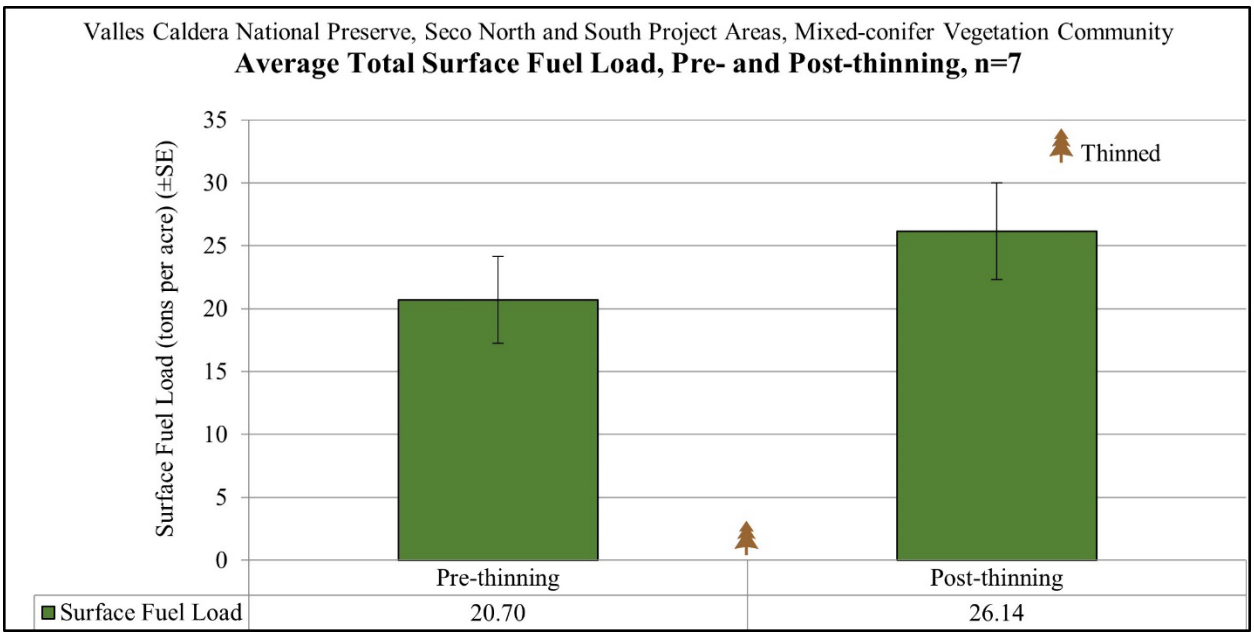
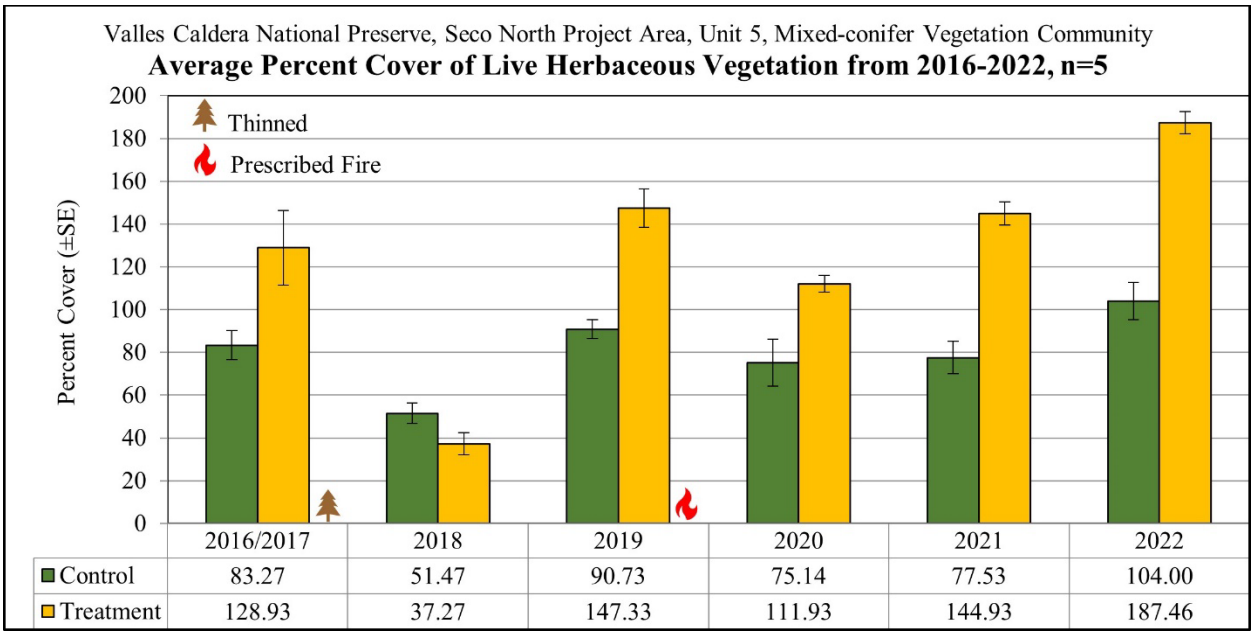



Figure 4.2.32. Valles Caldera National Preserve, Seco North and South Project Area, Average Total Surface Fuel Load. Pueblo and Four Winds Fire Ecology Program.

Summary statistics show a pre-thinning average total surface fuel load of 20.70 (SE±3.47) tons per acre. Post-thinning average total surface fuel load is reported at 26.14 (SE±3.85) tons per acre. Both are within historical estimations for fuel loading in this vegetation community. There appears to be a 5.44 tons per acre increase in total surface fuel load after thinning operations. However, statistical analysis shows no evidence of a difference between pre- and post-thinning values (95% CI: [-3.93, 14.81], p=0.205).

***Unit 5***

Ten plots in Seco North Project Area, Unit 5 were installed in 2016 and 2017, half of which were placed within a control area and half within the treatment unit (n=5). Forest thinning was completed in 2017 and a prescribed fire was completed in 2019. Data were collected until 2022.




| | 2016/2017 | 2018 | 2019 | 2020 | 2021 | 2022 |
|---|---|---|---|---|---|---|
| Control | 83.27 | 51.47 | 90.73 | 75.14 | 77.53 | 104.00 |
| Treatment | 128.93 | 37.27 | 147.33 | 111.93 | 144.93 | 187.46 |

Figure 4.2.33. Valles Caldera National Preserve, Seco North Project Area, Unit 5, Average Percent Cover of Live Herbaceous Vegetation. Valles Caldera Vegetation Program.

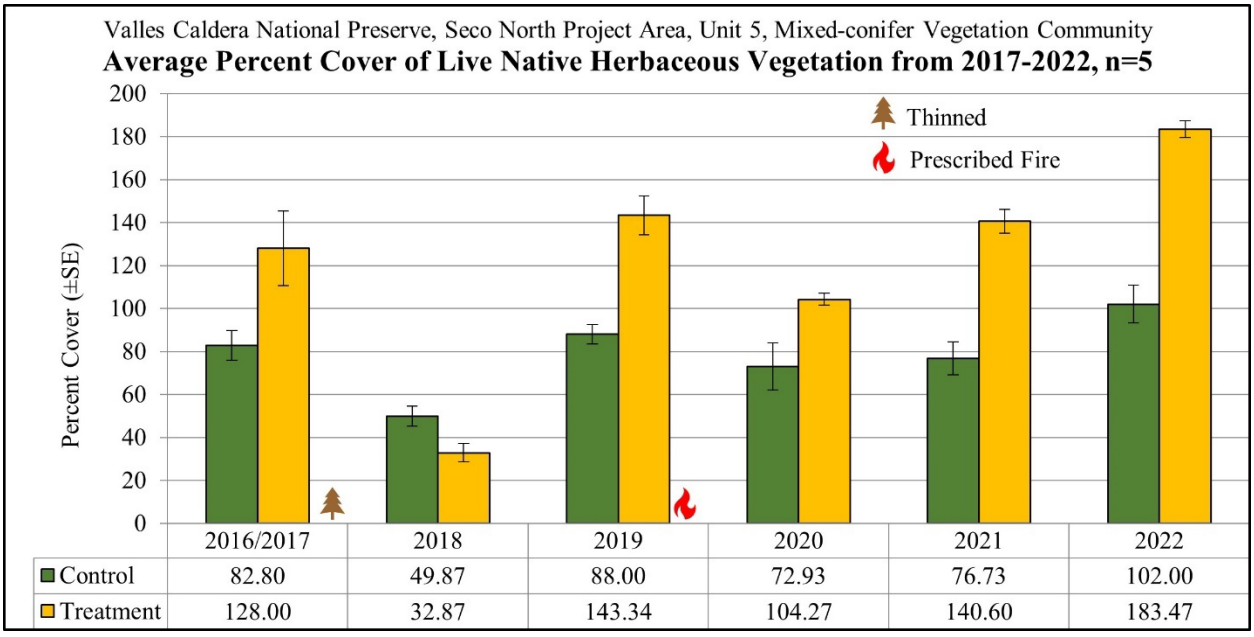
Summary statistics for the treatment plots show a pre-thinning average live herbaceous cover of 128.93% (SE±17.47) and a post-thinning average cover of 37.27% (SE±5.18). A 71.09% reduction in cover was observed, supported by statistical evidence of a difference between values (estimated difference 91.67, 95% CI: [60.67, 122.57], p=0.0000001), followed by a 295.30% increase in cover one-year post-thinning/preburn in 2019. This increase in average percent cover from 37.37% (SE±5.18) in 2018 to 147.33% (SE±8.95) in 2019 is also supported by statistical evidence of a difference between values (estimated difference -110.07, 95% CI: [-140.97, -79.16], p=0.000000006). The initial decrease and subsequent increase in live herbaceous cover may be due to disturbance impacts of thinning operations followed by recovery. Postburn average live herbaceous cover decreased to 111.93% (SE±3.91) in 2020, increased to 144.93% (SE±5.44) in 2021, and peaked at 187.46% (SE±5.20) in 2022. Statistical analysis shows evidence of differences between post-thinning/preburn (2019) herbaceous cover and postburn years 2020 (estimated difference 35.4, 95% CI: [4.49, 66.31], p=0.019) and 2022 (estimated difference -40.13, 95% CI: [-71.04, -9.23], p=0.007). The postburn fluctuations in herbaceous cover are likely due to nutrient cycling initiated by prescribed fire, with the greatest effects occurring in the first few years after treatment. It is of note that herbaceous vegetation cover is greatly influenced by precipitation.

Control plots did not receive thinning or burning treatments, therefore changes in live herbaceous cover were not expected. Observed changes between years may be attributed to precipitation patterns. Summary statistics show fluctuations in average live herbaceous cover for the control

plots over the sample years, with the lowest average cover of 51.47% (SE±4.76) reported in 2018 and the highest average cover of 104.00% (SE±8.67) in 2022. Statistical analysis shows evidence of a difference between percent cover in 2018 and the previous 2016/2017 collection years (estimated difference 31.80, 95% CI: [8.96, 54.64], p=0.003), as well as subsequent years 2019 (estimated difference -39.27, 95% CI: [-62.11,-16.24], p=0.0003), 2020 (estimated difference -23.67, 95% CI: [-46.51, -0.83], p=0.04), 2021 (estimated difference -26.07, 95% CI: [-48.91, -3.23], p=0.02), and 2022 (estimated difference -52.53, 95% CI: [-75.38, -29.69], p=0.000007). Additional differences can be observed in the accompanying table (Table 4.2.3.).

Table 4.2.3. Contrast tables for live herbaceous vegetation control and treatment plots in Seco Project Area, Unit 5. Comparisons between relevant years are denoted in the table below, for example, year2017-year2018 compares the averages between the two years.

| *Live Herbaceous Vegetation - Control* | | | | |
|---|---|---|---|---|
| **Contrast** | **Estimated Difference** | **CI-low** | **CI-high** | **P-value** |
| year2017 - year2018 | 31.80 | 8.96 | 54.64 | 0.0034317702 |
| year2017 - year2019 | -7.47 | -30.31 | 15.38 | 0.9031539985 |
| year2017 - year2020 | 8.13 | -14.71 | 30.97 | 0.8679145797 |
| year2017 - year2021 | 5.73 | -17.11 | 28.57 | 0.9662644788 |
| year2017 - year2022 | -20.73 | -43.58 | 2.11 | 0.0890927287 |
| year2018 - year2019 | -39.27 | -62.11 | -16.42 | 0.0003449353 |
| year2018 - year2020 | -23.67 | -46.51 | -0.83 | 0.0395255810 |
| year2018 - year2021 | -26.07 | -48.91 | -3.23 | 0.0196235869 |
| year2018 - year2022 | -52.53 | -75.38 | -29.69 | 0.0000071618 |
| year2019 - year2020 | 15.60 | -7.25 | 38.44 | 0.3047619505 |
| year2019 - year2021 | 13.20 | -9.64 | 36.04 | 0.4782166182 |
| year2019 - year2022 | -13.27 | -36.11 | 9.57 | 0.4726248084 |
| year2020 - year2021 | -2.40 | -25.24 | 20.44 | 0.9993938444 |
| year2020 - year2022 | -28.86 | -51.71 | -6.02 | 0.0084506864 |
| year2021 - year2022 | -26.47 | -49.31 | -3.62 | 0.0174304392 |
| *Live Herbaceous Vegetation - Treatment* | | | | |
| **Contrast** | **Estimated Difference** | **CI-low** | **CI-high** | **P-value** |
| year2017 - year2018 | 91.67 | 60.76 | 122.57 | 0.0000001389 |
| year2017 - year2019 | -18.40 | -49.3 | 12.51 | 0.4466589000 |
| year2017 - year2020 | 17.00 | -13.9 | 47.91 | 0.5296702000 |
| year2017 - year2021 | -16.00 | -46.91 | 14.91 | 0.5912803000 |
| year2017 - year2022 | -58.53 | -89.44 | -27.62 | 0.0001035828 |
| year2018 - year2019 | -110.07 | -140.97 | -79.16 | 0.0000000064 |
| year2018 - year2020 | -74.67 | -105.57 | -43.76 | 0.0000034642 |
| year2018 - year2021 | -107.67 | -138.57 | -76.76 | 0.0000000093 |
| year2018 - year2022 | -150.20 | -181.1 | -119.29 | 0.0000000000 |
| year2019 - year2020 | 35.40 | 4.49 | 66.31 | 0.0190782300 |
| year2019 - year2021 | 2.40 | -28.51 | 33.3 | 0.9998617000 |
| year2019 - year2022 | -40.13 | -71.04 | -9.23 | 0.0066263120 |
| year2020 - year2021 | -33.00 | -63.91 | -2.1 | 0.0321161400 |
| year2020 - year2022 | -75.53 | -106.44 | -44.63 | 0.0000029140 |
| year2021 - year2022 | -42.53 | -73.44 | -11.62 | 0.0038436960 |




| | 2016/2017 | 2018 | 2019 | 2020 | 2021 | 2022 |
|---|---|---|---|---|---|---|
| Control | 82.80 | 49.87 | 88.00 | 72.93 | 76.73 | 102.00 |
| Treatment | 128.00 | 32.87 | 143.34 | 104.27 | 140.60 | 183.47 |

Figure 4.2.34. Valles Caldera National Preserve, Seco North Project Area, Unit 5, Average Percent Cover of Live Native Herbaceous Vegetation. Valles Caldera Vegetation Program.

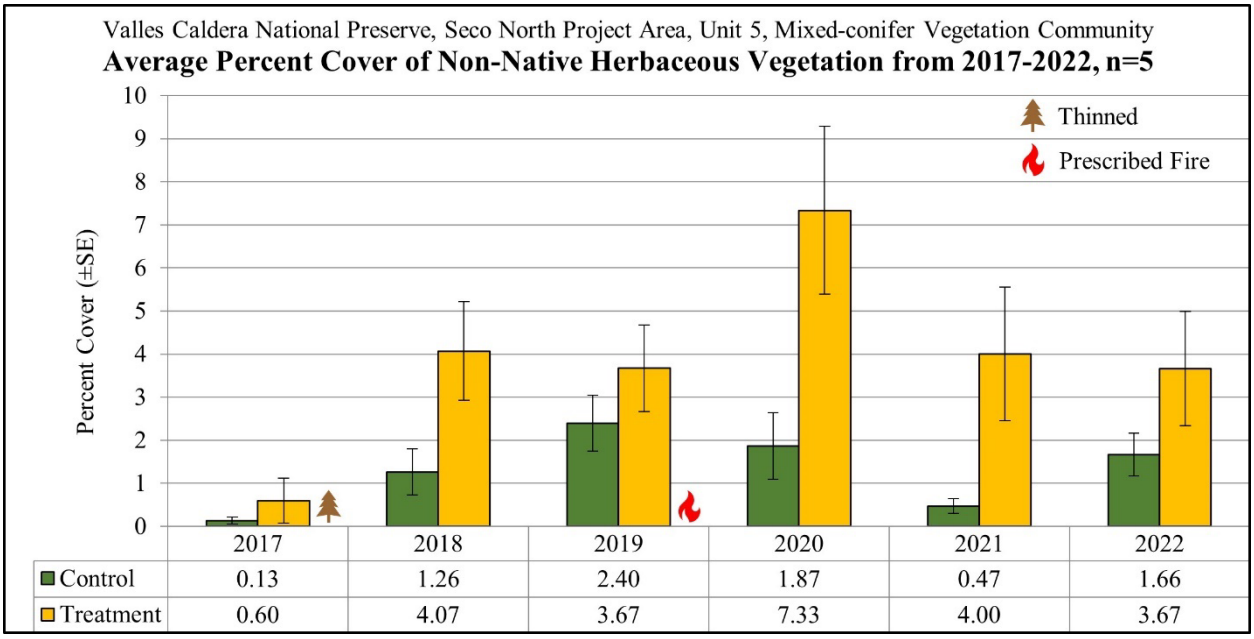
Summary statistics for the treatment plots show a pre-thinning average live native herbaceous cover of 128.00% (SE±17.31) and post-thinning native cover of 32.87% (SE±4.26). A 74.32% reduction in native cover was observed, supported by statistical evidence of a difference between pre-thinning and post-thinning values (estimated difference 95.13, 95% CI: [62.48, 127.78], p=0.0000001), followed by a 336.08% increase in native cover one-year post-thinning/preburn in 2019. This increase in average percent cover from 32.87% (SE±4.26) in 2018 to 143.34% (SE±9.04) in 2019 is also supported by statistical evidence of a difference between values (estimated difference -110.47, 95% CI: [-143.12, -77.82], p=0.00000002). The initial decrease and subsequent increase in live native herbaceous cover may be due to disturbance impacts of thinning operations followed by recovery. Postburn average live native herbaceous cover decreased to 104.27% (SE±2.73) in 2020, increased to 140.60% (SE±5.53) in 2021, and peaked at 183.47% (SE±3.96) in 2022. Statistical analysis shows evidence of differences between post-thinning/preburn (2019) native cover and postburn years 2020 (estimated difference 39.07, 95% CI: [6.42, 71.72], p=0.013) and 2022 (estimated difference -40.13, 95% CI: [-72.78, -7.48], p=0.012). The postburn fluctuations in native herbaceous cover are likely due to nutrient cycling initiated by prescribed fire, with the greatest effects occurring in the first few years after treatment. It is of note that herbaceous vegetation cover is greatly influenced by precipitation.

Control plots did not receive thinning or burning treatments, therefore changes in live native herbaceous cover were not expected. Observed changes between years may be attributed to precipitation patterns. Summary statistics show fluctuations in average live native herbaceous cover for the control plots over the sample years, with the lowest average cover of 49.87% (SE±4.56) reported in 2018 and the highest average cover of 102.00% (SE±8.82) in 2022. Statistical analysis shows evidence of a difference between native percent cover in 2018 and the previous 2016/2017 collection years (estimated difference 32.93, 95% CI: [10.31, 55.55], p=0.002), as well as subsequent years 2019 (estimated difference -38.13, 95% CI: [-60.75, -

15.52], p=0.0004), 2020 (estimated difference -23.06, 95% CI: [-45.68, -0.45], p=0.044), 2021 (estimated difference -26.86, 95% CI: [-49.48, -4.25], p=0.014), and 2022 (estimated difference -52.14, 95% CI: [-74.75, -29.52], p=0.000007). Additional differences can be observed in the accompanying table (Table 4.2.4.).

Table 4.2.4. Contrast tables for native live herbaceous vegetation control and treatment plots in Seco Project Area, Unit 5. Comparisons between relevant years are denoted in the table below, for example, year2017-year2018 compares the averages between the two years.

| *Native Live Herbaceous Vegetation - Control* | | | | |
|---|---|---|---|---|
| **Contrast** | **Estimated Difference** | **CI-low** | **CI-high** | **P-value** |
| year2017 - year2018 | 32.93 | 10.31 | 55.55 | 0.0021878226 |
| year2017 - year2019 | -5.20 | -27.82 | 17.41 | 0.9767049622 |
| year2017 - year2020 | 9.87 | -12.75 | 32.48 | 0.7428911209 |
| year2017 - year2021 | 6.07 | -16.55 | 28.68 | 0.9555029158 |
| year2017 - year2022 | -19.21 | -41.82 | 3.41 | 0.1261398909 |
| year2018 - year2019 | -38.13 | -60.75 | -15.52 | 0.0004338957 |
| year2018 - year2020 | -23.06 | -45.68 | -0.45 | 0.0440020223 |
| year2018 - year2021 | -26.86 | -49.48 | -4.25 | 0.0142798510 |
| year2018 - year2022 | -52.14 | -74.75 | -29.52 | 0.0000069249 |
| year2019 - year2020 | 15.07 | -7.55 | 37.69 | 0.3290792121 |
| year2019 - year2021 | 11.27 | -11.35 | 33.89 | 0.6282603222 |
| year2019 - year2022 | -14.00 | -36.62 | 8.61 | 0.4051428971 |
| year2020 - year2021 | -3.80 | -26.42 | 18.82 | 0.9943252449 |
| year2020 - year2022 | -29.07 | -51.69 | -6.46 | 0.0072598870 |
| year2021 - year2022 | -25.27 | -47.89 | -2.66 | 0.0230544518 |
| *Native Live Herbaceous Vegetation - Treatment* | | | | |
| **Contrast** | **Estimated Difference** | **CI-low** | **CI-high** | **P-value** |
| year2017 - year2018 | 95.13 | 62.48 | 127.78 | 0.0000001855 |
| year2017 - year2019 | -15.34 | -47.99 | 17.31 | 0.6817784349 |
| year2017 - year2020 | 23.73 | -8.92 | 56.38 | 0.2456401070 |
| year2017 - year2021 | -12.61 | -45.25 | 20.04 | 0.8252186527 |
| year2017 - year2022 | -55.47 | -88.12 | -22.82 | 0.0003963756 |
| year2018 - year2019 | -110.47 | -143.12 | -77.82 | 0.0000000153 |
| year2018 - year2020 | -71.40 | -104.05 | -38.75 | 0.0000147658 |
| year2018 - year2021 | -107.74 | -140.38 | -75.09 | 0.0000000235 |
| year2018 - year2022 | -150.60 | -183.25 | -117.95 | 0.0000000001 |
| year2019 - year2020 | 39.07 | 6.42 | 71.72 | 0.0134278769 |
| year2019 - year2021 | 2.74 | -29.91 | 35.38 | 0.9997981504 |
| year2019 - year2022 | -40.13 | -72.78 | -7.48 | 0.0107368744 |
| year2020 - year2021 | -36.34 | -68.98 | -3.69 | 0.0237572420 |
| year2020 - year2022 | -79.20 | -111.85 | -46.55 | 0.0000032555 |
| year2021 - year2022 | -42.87 | -75.51 | -10.22 | 0.0059883095 |




| | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 |
|---|---|---|---|---|---|---|
| Control | 0.13 | 1.26 | 2.40 | 1.87 | 0.47 | 1.66 |
| Treatment | 0.60 | 4.07 | 3.67 | 7.33 | 4.00 | 3.67 |

Figure 4.2.35. Valles Caldera National Preserve, Seco North Project Area, Unit 5, Average Percent Cover of Live Non-Native Herbaceous Vegetation. Valles Caldera Vegetation Program.

Summary statistics for the treatment plots show a pre-thinning average live non-native herbaceous cover of 0.60% (SE±0.52) and post-thinning non-native cover of 4.07% (SE±1.14). There appears to be an increase in percent cover of non-native herbaceous vegetation after thinning operations, however statistical analysis shows no evidence of a difference between pre-thinning and post-thinning values (estimated difference -3.47, 95% CI: [-8.45, 1.51], p=0.286). Similarly, the reported preburn non-native herbaceous cover of 3.67% (SE±1.01) increased to 7.33% (SE±1.95) postburn, but there is no statistical evidence of a difference in values (estimated difference -3.67, 95% CI: [-8.65, 1.31], p=0.234). The changes in percent cover of live non-native herbaceous vegetation may be due to increased disturbances from thinning and burning operations. It is of note that herbaceous vegetation cover is greatly influenced by precipitation.

Control plots did not receive thinning or burning treatments, therefore changes in live non-native herbaceous cover were not expected. Observed changes between years may be attributed to precipitation patterns or other disturbances that may have occurred in the area. Summary statistics show fluctuations in average live non-native herbaceous cover for the control plots over the sample years, with the lowest average cover of 0.13% (SE±0.08) reported in 2017 and the highest average cover of 2.40% (SE±0.65) in 2019. Statistical analysis shows evidence of a difference between non-native percent cover in 2017 and 2019 (estimated difference -2.27, 95% CI: [-4.50, -0.03], p=0.046), however those are the only years for which evidence of a difference exists.

**Santa Fe National Forest, East Fork Project Area, Ponderosa Pine Forest Vegetation Community**

All plots (n=5) in East Fork were installed and pre-thinning data were collected in 2018 before the project area was thinned and piled later that year. Post-thinning data were collected in 2019. The piles were burned in 2021 and postburn data were collected in 2023.

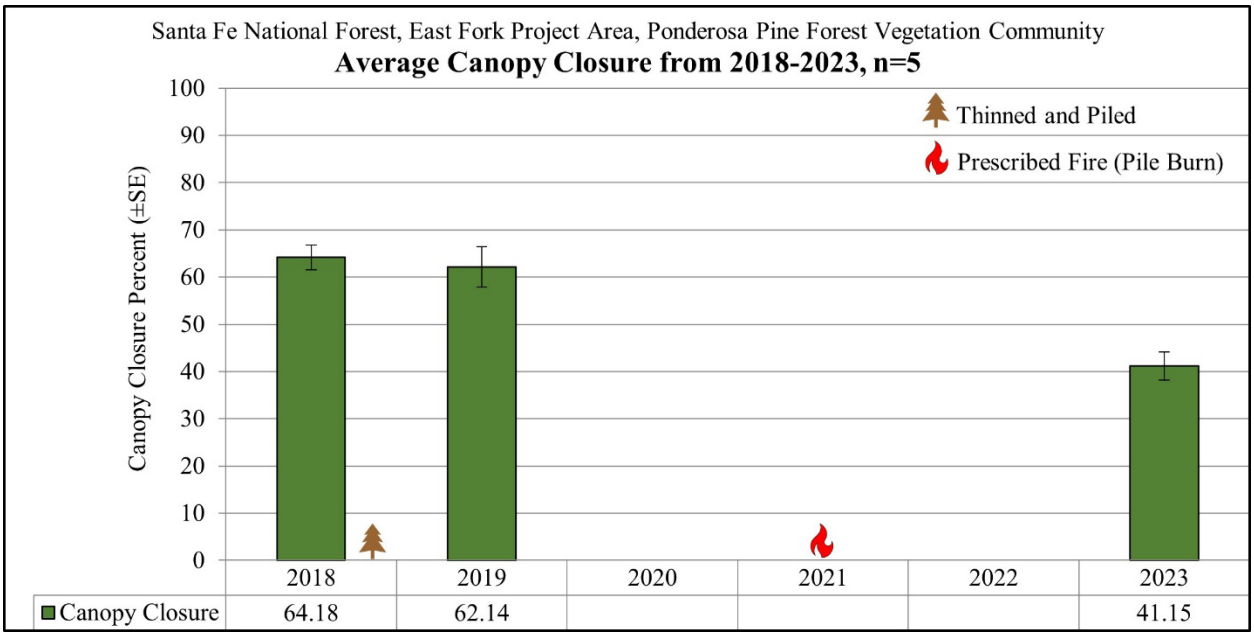



Figure 4.2.36. Santa Fe National Forest, East Fork Project Area, Average Canopy Closure. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average canopy closure of 64.18% (SE±2.64) and a post-thinning closure of 62.14% (SE±4.25). Statistical analysis shows no evidence of a difference between pre- and post-thinning values (estimated difference 2.04, 95% CI: [-11.49, 15.58], p=0.904). However, the postburn average canopy closure of 41.15% (SE±2.94) in 2023 shows a reduction of 33.78% from the post-thin/preburn closure of 62.14% (SE±4.25) in 2019, supported by statistical evidence of a difference between values (estimated difference 20.99, 95% CI: [7.46, 34.52], p=0.005).

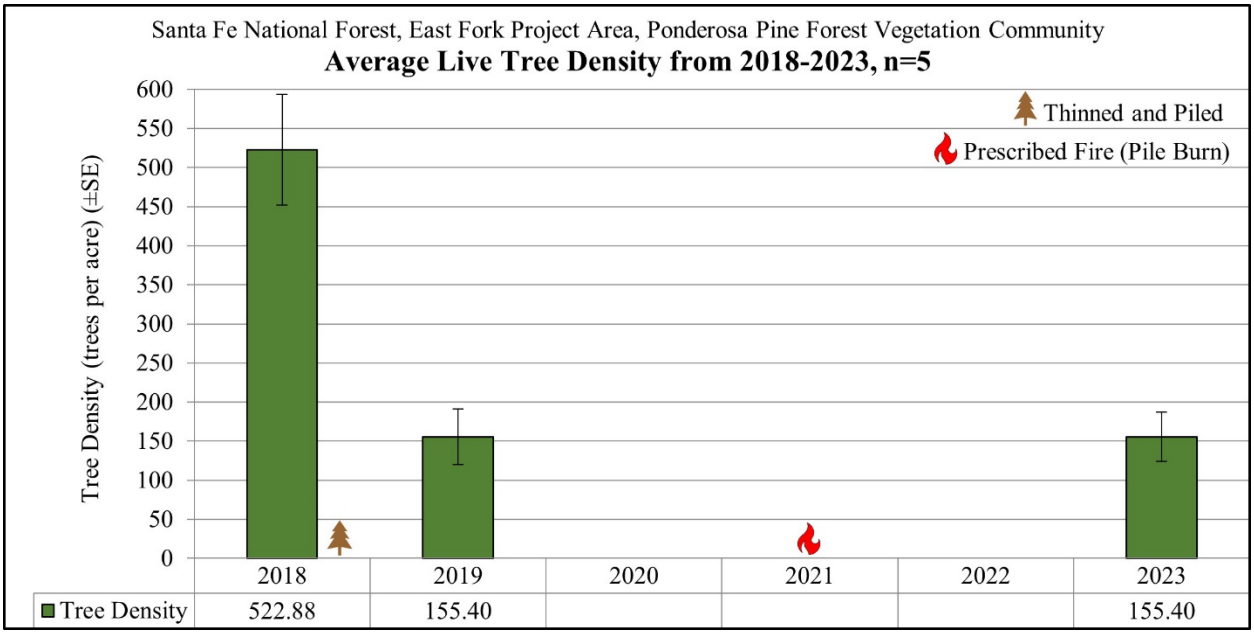



Figure 4.2.37. Santa Fe National Forest, East Fork Project Area, Average Live Tree Density. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average live tree density of 522.88 (SE±70.72) trees per acre, higher than the estimated range of historical tree densities in this vegetation community. Post-thinning tree density is 155.40 (SE±35.51) trees per acre. There appears to be a 367.48 trees per acre reduction in live tree density after thinning operations, supported by statistical evidence of a difference between values (95% CI: [168.84, 566.12], p=0.002). The postburn average live tree density was reported as 155.40 (SE±31.40). There appears to be no change in live tree density after pile burning, indicating minimal impacts (torching) to live trees during burning operations. Statistical analysis shows no evidence of difference between the values (estimated difference 0.00, 95% CI: [-198.64, 198.64], p=1.000).

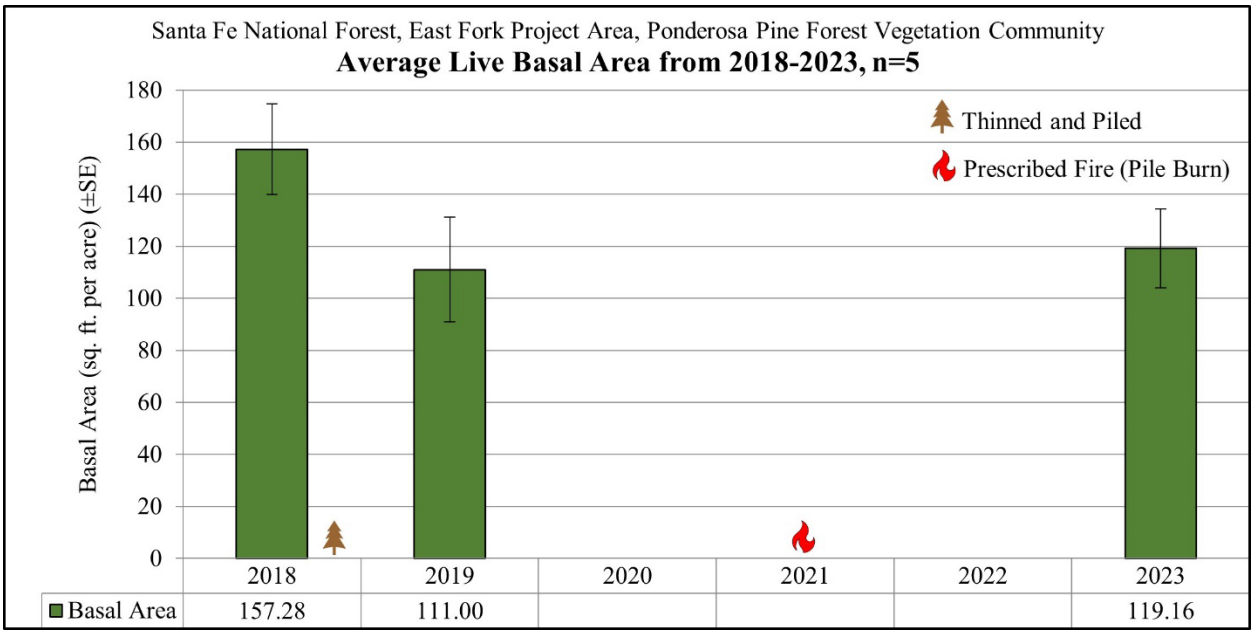



Figure 4.2.38. Santa Fe National Forest, East Fork Project Area, Average Live Basal Area. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average live basal area of 157.28 (SE±17.37) sq. ft. per acre and a post-thinning basal area of 111.00 (SE±20.10). There appears to be a 46.28 sq. ft. per acre reduction in basal area between pre-thinning and post-thinning values, supported by statistical evidence (95% CI: [32.73, 59.83], p=0.00003). After pile burning operations, there appears to be a slight increase in basal area, however statistical analysis shows no evidence of a difference (estimated difference -8.16, 95% CI: [-21.71, 5.39], p=0.256).

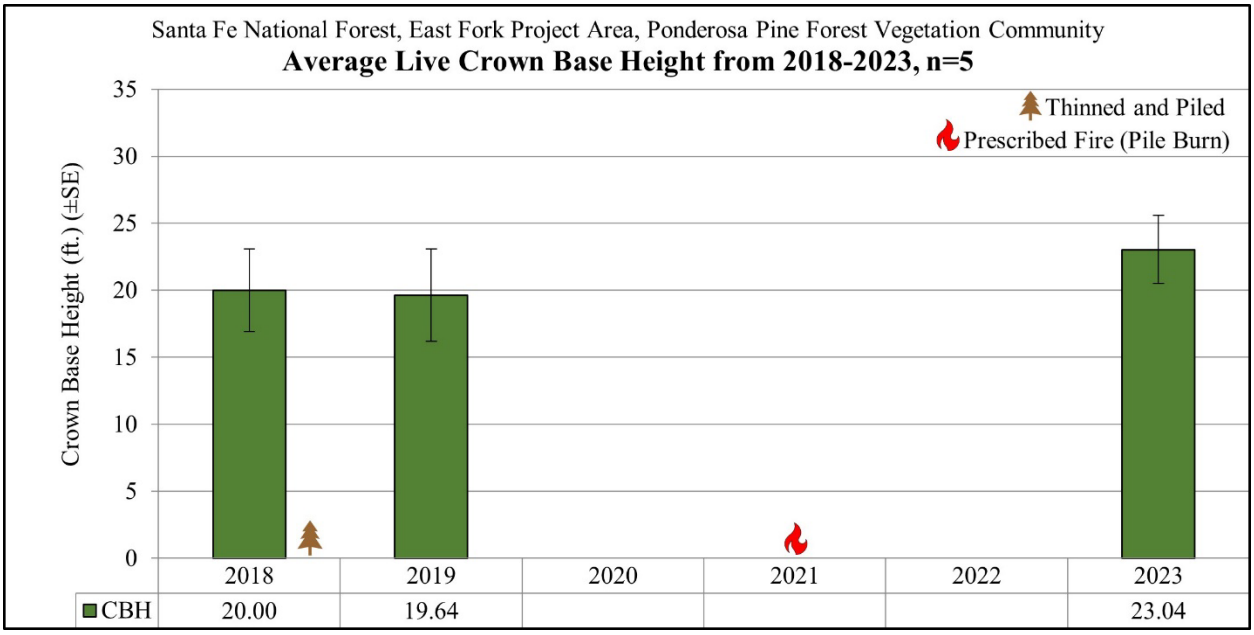



Figure 4.2.39. Santa Fe National Forest, East Fork Project Area, Average Live Crown Base Height. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average live crown base height of 20.00 (SE±3.07) ft. and a post-thinning crown base height of 19.64 (SE±3.45) ft. The slight decrease observed in live crown base height after thinning operations may be a result of differences or inconsistencies in field data collection methods or the potential for increased canopy growth with decreasing tree density and less competition for resources. However, statistical analysis shows no evidence of a difference between pre- and post-thinning values (estimated difference 0.36, 95% CI: [-2.33, 3.05], p=0.923). An average live crown base height of 23.04 (SE±2.55) ft. was reported postburn, with an observed increase of 3.40 ft. from the post-thin/preburn live crown base height of 19.64 (SE±3.45); statistical analysis supports evidence of a difference between values (95% CI: [-6.09, -0.71], p=0.017).

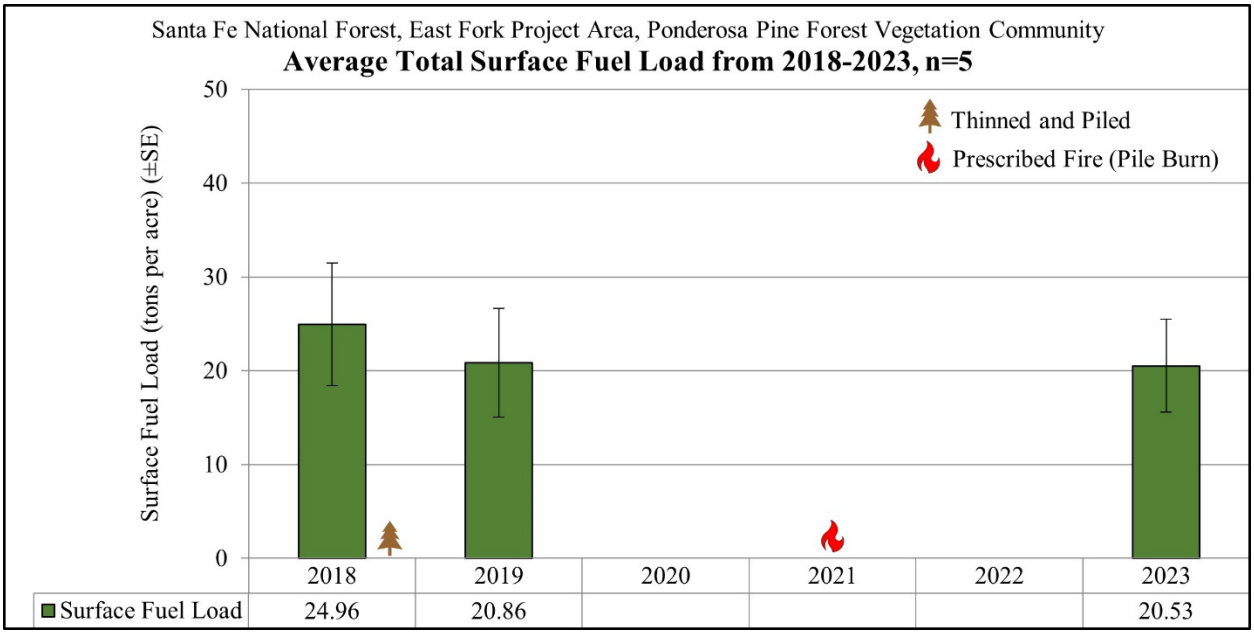



Figure 4.2.40. Santa Fe National Forest, East Fork Project Area, Average Total Surface Fuel Load. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average total surface fuel load of 24.96 (SE±6.54) tons per acre. Post-thinning average total surface fuel load is reported at 20.86 (SE±5.80) tons per acre. There appears to be a slight decrease in total surface fuel load after thinning operations, however statistical analysis shows no evidence of a difference between values (estimated difference 4.10, 95% CI: [-4.51, 12.70], p=0.404). Thinned materials were placed in piles and pile burning was initiated in the project area. There appears to be another slight decrease in total surface fuel load postburn, however there is no statistical evidence to support a difference between post-thin/preburn values in 2019 and postburn values in 2023 (estimated difference 0.33, 95% CI: [-8.27, 8.94], p=0.993).

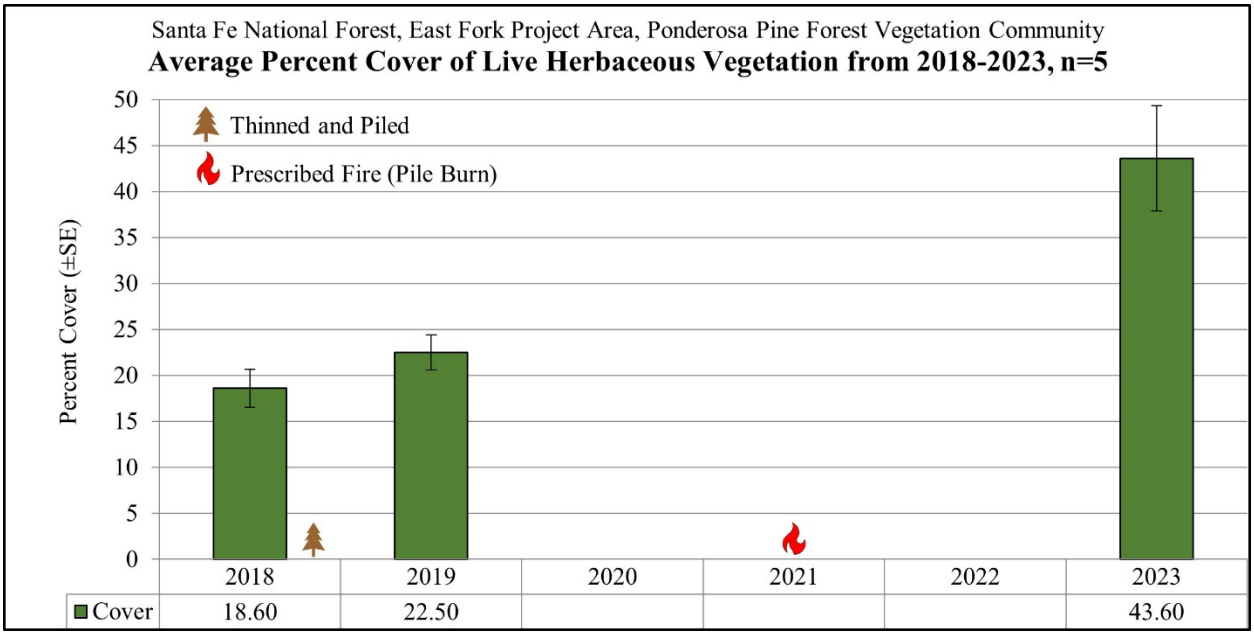



Figure 4.2.41. Santa Fe National Forest, East Fork Project Area, Average Percent Cover of Live Herbaceous Vegetation. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average live herbaceous cover of 18.60% (SE±2.06). Post-thinning cover increased to 22.50% (SE±1.89), however statistical analysis shows no evidence of a difference between pre- and post-thinning live herbaceous vegetation percent cover (estimated difference -3.90, 95% CI: [-13.93, 6.13], p=0.534). Live herbaceous cover was reported at 43.60% (SE±5.75) postburn in 2023, a 93.78% increase from post-thin/preburn values in 2019; statistical analysis supports evidence of a difference between values (estimated difference -21.10, 95% CI: [-31.13, -11.07], p=0.0008).

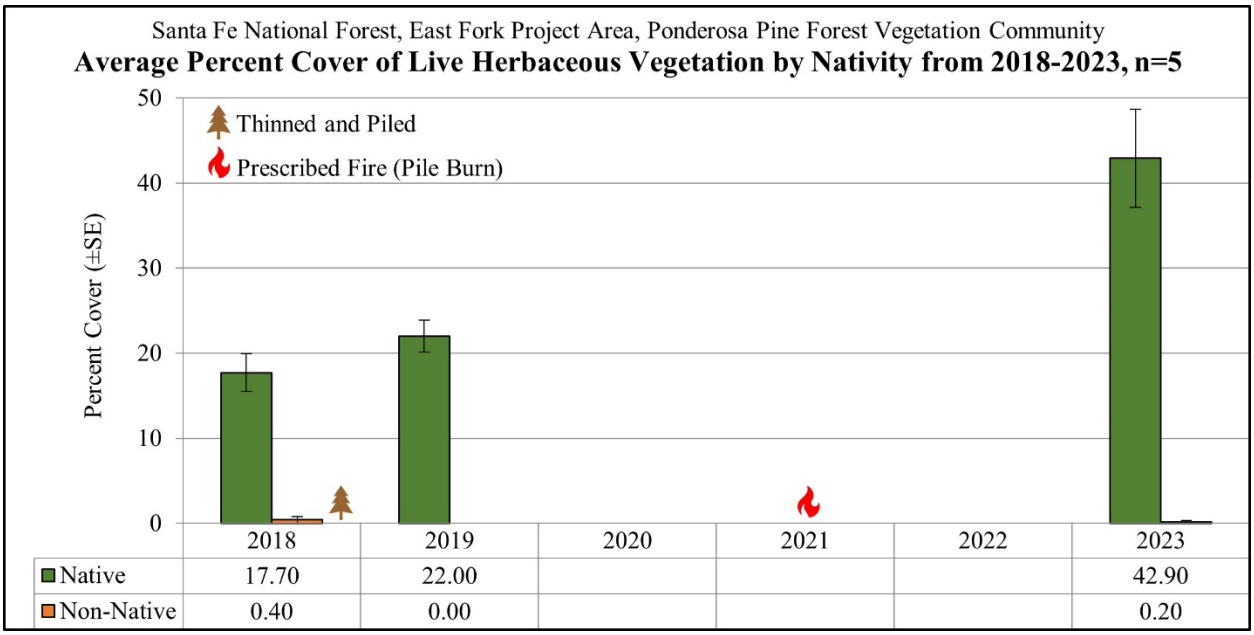



Figure 4.2.42. Santa Fe National Forest, East Fork Project Area, Average Percent Cover of Live Herbaceous Vegetation by Nativity. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average live herbaceous native cover of 17.70% (SE±2.22) and non-native cover of 0.40% (SE±0.40). Post-thinning native cover increased to 22.00% (SE±1.89) and no non-native cover was detected, however statistical analysis shows no evidence of a difference between pre- and post-thinning live herbaceous percent cover for both native vegetation (estimated difference -4.3, 95% CI: [-14.25, 5.65], p=0.467) and non-native vegetation (estimated difference 0.4, 95% CI: [-0.58, 1.38], p=0.502). Native cover was reported at 42.90% (SE±5.77) postburn in 2023, a 95.00% increase from post-thin/preburn values in 2019; statistical analysis supports evidence of a difference between values (estimated difference -20.9, 95% CI: [-30.85, -10.95], p=0.0008). Non-native cover was reported at 0.20% (SE±0.12) postburn, a slight increase from post-thin/preburn values, however there is no statistical evidence to support a difference (estimated difference -0.2, 95% CI [-1.18, 0.28], p=0.831).

**Santa Fe National Forest, Monument Canyon Project Area, Ponderosa Pine Forest Vegetation Community**

The United States Forest Service (USFS) designated Monument Canyon Research Natural Area (MCRNA) in 1932 to protect the old growth ponderosa pine forests from logging, grazing, and other human influences. Over time, this designation has resulted in some of the least human-disturbed ponderosa pine forests in Northern New Mexico.

MCRNA covers approximately 640 acres of a relatively flat plateau and consists mainly of mid-elevation (7,800 – 8,400 ft.) old growth ponderosa pine forests with smaller areas of mixed-conifer forest. This Biological Description is specific to a 235-acre portion of MCRNA that was mechanically thinned and masticated in 2007. The area was thinned to reduce the density of smaller diameter trees that had increased due to long-term fire exclusion. The next phase of treatment included a broadcast prescribed fire in 2012, with the objective to further reduce smaller diameter trees and forest fuels, maintain larger diameter trees, and increase native herbaceous vegetation.

The combined overstory and midstory tree density is approximately 123 trees per acre. The overstory, approximately 81 trees per acre, is comprised mostly of Ponderosa pine (*Pinus ponderosa*), at 70 trees per acre. White fir (*Abies concolor*), limber pine (*Pinus flexilis*), and Douglas fir (*Psuedotsuga menziesii*) are a smaller component. The midstory contains approximately 42 trees per acre, with 37 trees per acre consisting of ponderosa pine. White fir and limber pine are common midstory trees. Rocky Mountain juniper (*Juniperus scopulorum*) and Douglas fir are present but less common. Oak (*Quercus spp*.) are often present, occasionally in dense thickets.

Reproduction is present but highly spatially variable at approximately 78 trees per acre, consisting primarily of ponderosa pine (62 trees per acre) with a smaller component of aspen (*Populus tremuloides*), white fir, and Douglas fir. A notable increase in aspen is observed five years (2017) after the prescribed fire, from 1 stem per acre at pre-treatment to 9 at post-treatment.

Surface fuels are approximately 18 tons per acre, with 1000-hour fuels as the greatest contributor at 9 tons per acre. Litter and duff fuel load is approximately 8 tons per acre. 1, 10, and 100-hour fuels combined are approximately 1.5 tons per acre. Litter and duff depths are approximately 1.1 inches.

Shrub density is approximately 62 stems per acre. Common shrubs include oak, New Mexico locust (*Robinia neomexicana*), American red raspberry (*Rubus idaeus*), and wax current (*Ribes cereum*).

The surface layer cover is approximately 57% non-vegetated substrate, commonly long needle litter (44% cover), bare ground (7% cover), and wood (4% cover) and includes an herbaceous component at approximately 55% cover. The lifeform with the highest percent cover (23%) is forb, followed by graminoid at 18%, grass-like at 9%, tree at 2%, and shrub at 2%. Forb percent cover consists of 13% native species and 10% non-native; graminoid percent cover is comprised of 17.98% native species and 0.22% non-native; grass-like, tree, and shrub species are entirely

native. Common forb species include common mullein (*Verbascum thapsus*), a non-native biennial, and two native perennial forbs, Wooton's ragwort (*Senecio wootonii*) and woodland strawberry (*Fragaria vesca*). Predominant grass species include Arizona fescue (*Festuca arizonica*), rough bentgrass (*Agrostis scabra*), and squirreltail (*Elymus elymoides*), all native perennial grasses. Grass-like species are all native perennial sedges (*Carex spp*.).

Species richness is recorded at 85 species; 73 species were detected on vegetation transects and an additional 12 were observed within 5 meters on either side of the vegetation transects. Of the 85 species, 79 are native and 6 are non-native. Non-native forbs include a biennial forb, common mullein, two perennial forbs, bull thistle (*Cirsium vulgare*) and common dandelion (*Taraxacum officinale*), and one annual, Canadian horseweed (*Conyza canadensis*). Non-native grasses include Kentucky bluegrass (*Poa pratensis*), a perennial, and cheatgrass (*Bromus tectorum*), an annual. (Note: This biological description is based on Fire Ecology data collected on 25 plots in 2017.)

The Santa Fe National Forest implemented a broadcast prescribed fire in Monument Canyon Research Natural Area in 2012. Fire Ecology data was collected on 25 plots in 2009/2010, prior to the prescribed fire, and after the fire in 2012, 2013, 2014, and 2017. Ten-year postburn data will be collected in 2022.

**Surface Fuels**

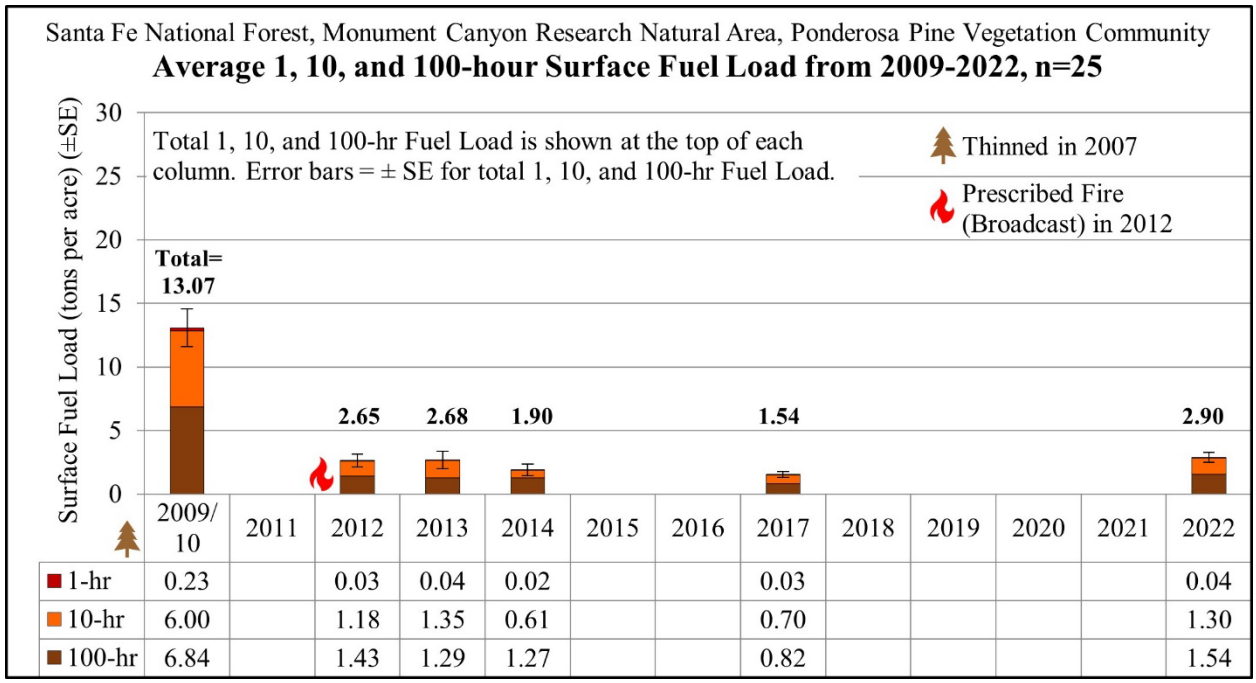



Figure 4.2.43. USFS, Monument Canyon, 1-100-hour surface fuel load results.

Due to thinning and mastication in 2007, average 1, 10, and 100-hour fuel load was recorded at extremely high levels (13.07 tons per acre) in 2009/2010. After the broadcast prescribed fire in 2012, it was recorded at 2.65 tons per acre, a 79.73 percent decrease from pre-fire levels. 1, 10, and 100-hour fuels remained low in 2013, 2014, 2017, and 2022.

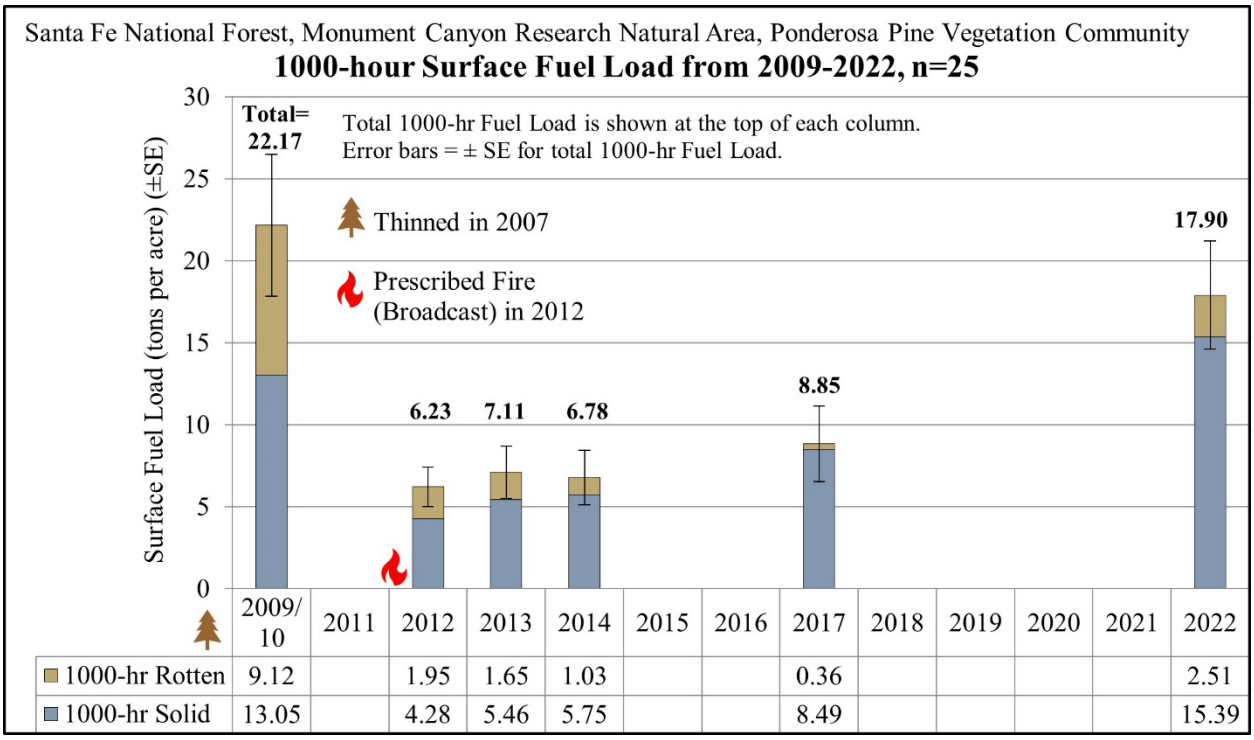



Figure 4.2.44. USFS, Monument Canyon, 1000-hour surface fuel load results.

Average 1000-hour fuel load was recorded at 22.17 tons per acre in 2009/2010, with solid 1000-hour fuels as the largest contributor at 13.05 tons per acre and rotten 1000-hour fuels at 9.12. After the broadcast prescribed fire in 2012, 1000-hour fuels were recorded at 6.23 tons per acre, a 72.00 percent decrease from pre-fire levels. 1000-hour fuel load remained fairly consistent from 2012-2014, with a slight increase in 2017 and a significant increase in 2022, particularly in the 1000-hr solid category and likely due to tree-fall between 2018 and 2022.

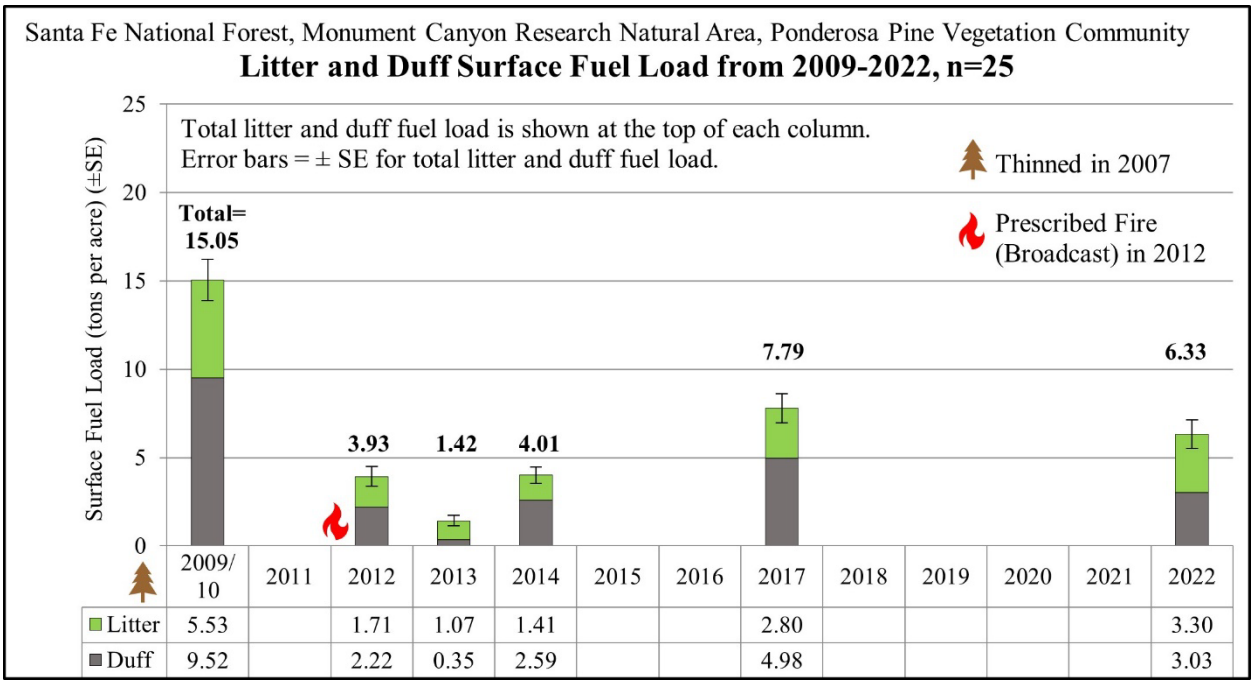



Figure 4.2.45. USFS, Monument Canyon, litter and duff surface fuel load results.

Due to mastication in 2007, average litter and duff fuel load was recorded at 15.05 tons per acre in 2009/2010, with duff contributing 9.52 tons per acre and litter at 5.53. After the broadcast prescribed fire in 2012, litter and duff fuel load was recorded at 3.93 tons per acre, a 73.89 percent decrease from pre-fire levels. Litter and duff fuel load remained low in 2013 and 2014. In 2017, litter and duff was recorded at 7.79 tons per acre, and at 6.33 tons per acre in 2022.

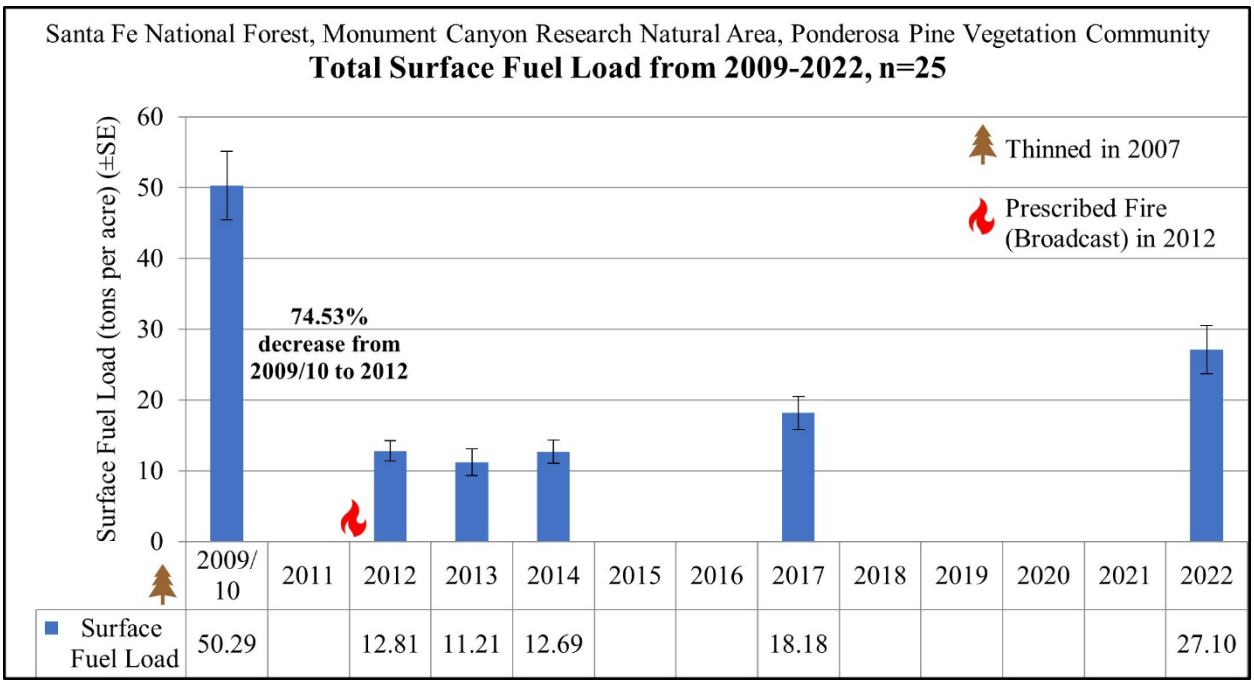



Figure 4.2.46. USFS, Monument Canyon total surface fuel load results.

Average total fuel load was recorded at 50.29 tons per acre in 2009/2010. After the broadcast prescribed fire in 2012, total fuel load was recorded at 12.81 tons per acre, a 74.53 percent decrease from pre-fire levels. Between 2012, 2013, and 2014, little to no change was recorded in total fuel load. In 2017, total fuel load was recorded at 18.18 tons per acre. In 2022, total fuel load was recorded at 27.10 tons per acre, an increasing trend.

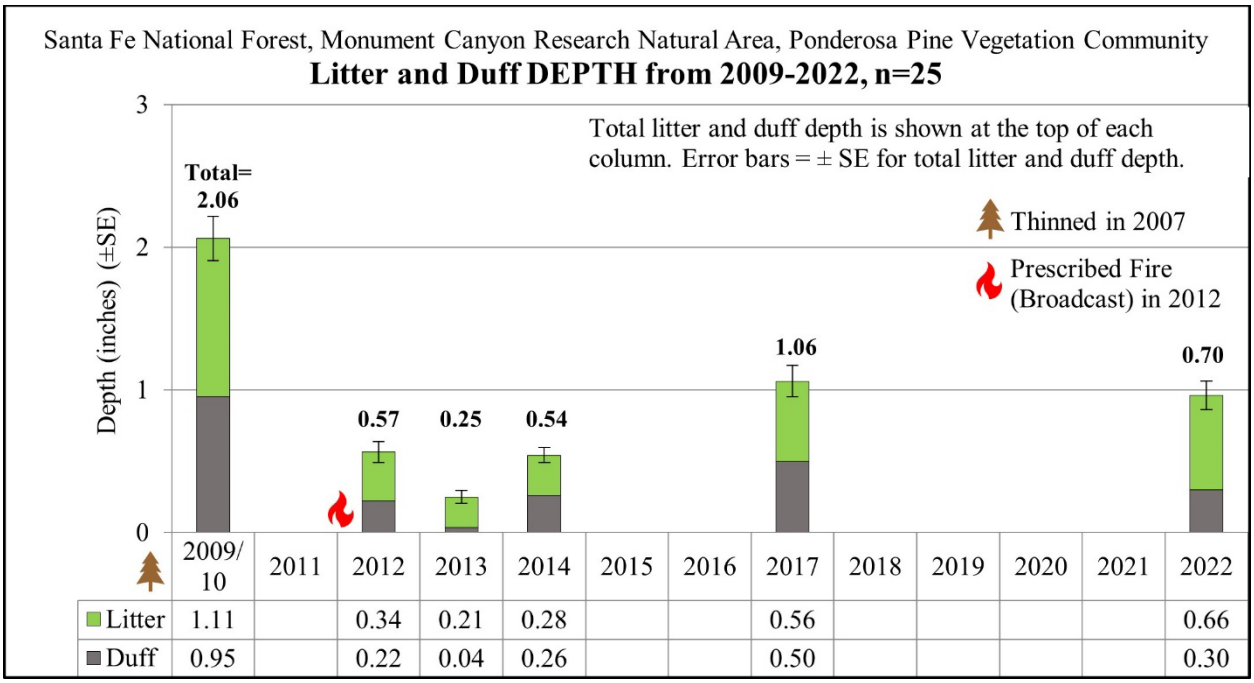



Figure 4.2.47. USFS, Monument Canyon litter and duff depth results.

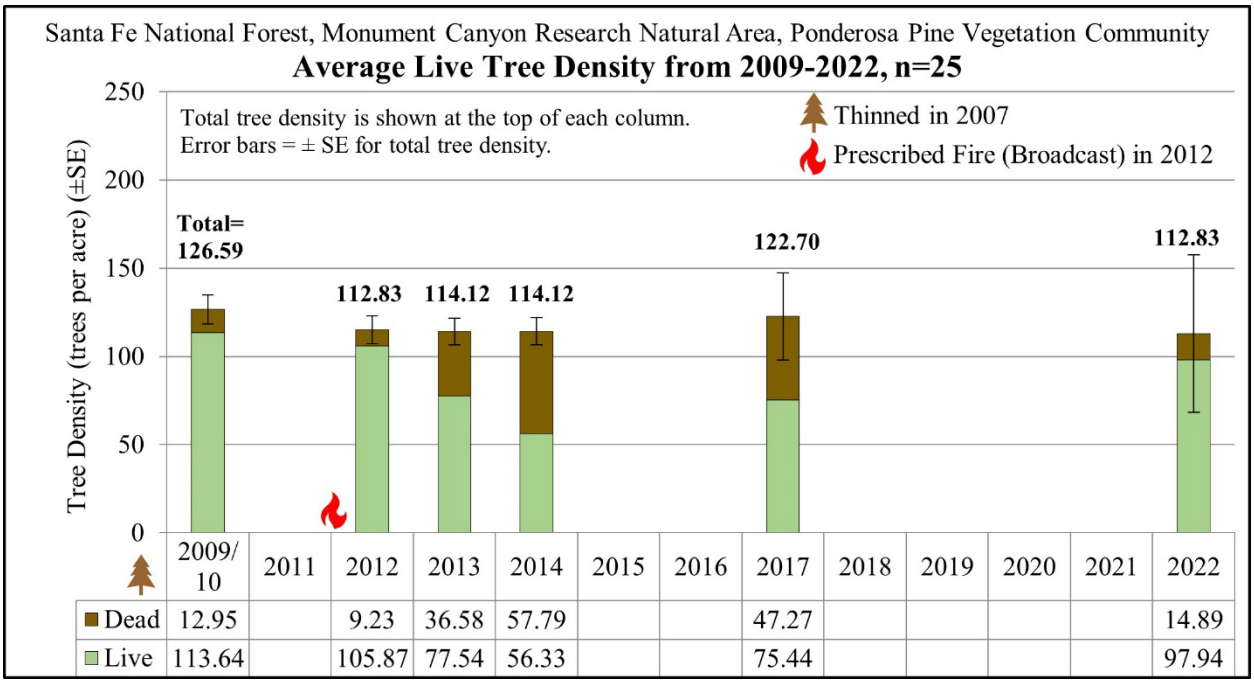
Average litter and duff depth was recorded at 2.06 inches in 2009/2010, with litter as the largest contributor at 1.11 inches and duff at 0.95. After the broadcast prescribed fire in 2012 (Figs. 3.18 and 3.19), litter and duff depth was recorded at 0.57 inches, a 72.33 percent decrease from pre-fire levels. In 2013 and 2014, litter and duff depths remained low. In 2017, litter and duff depth was recorded at 1.06 inches. In 2022, it was recorded at 0.70 inches.




Figure 4.2.48. Pre-treatment (2009) and post-treatment (2014) surface fuel loading in Monument Canyon Research Natural Area. Photos: Pueblo and Four Winds Fire Ecology Program, 2009 and 2014



Figure 4.2.49. Prescribed fire conducted in Monument Canyon Research Natural Area, 2012. Photos: Pueblo and Four Winds Fire Ecology Program, 2012.




| | 2009/10 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Dead | 12.95 | | 9.23 | 36.58 | 57.79 | | | 47.27 | | | | | 14.89 |
| Live | 113.64 | | 105.87 | 77.54 | 56.33 | | | 75.44 | | | | | 97.94 |

Figure 4.2.50. USFS, Monument Canyon tree density results.

Average tree density was recorded at 126.59 trees per acre in 2009/2010, with live trees as the largest contributor at 113.64 trees per acre and dead trees at 12.95. After the broadcast prescribed fire in 2012, tree density was recorded at 112.83 trees per acre. Total tree density remained at 114.12 trees per acre in 2013 and 2014, although the ratio between live and dead trees fluctuated, with live tree density reducing and dead tree density increasing. In 2017, total tree density increased to 122.70 trees per acre, with live tree density recorded at 75.44 trees per acre due to tree growth from the seedling category (less than 2.5 inches in diameter). In 2022, total tree density appears to decrease slightly to 112.83 trees per acre, with a reduction in dead standing trees and an increase in live trees due to recruitment from the tree seedling category.

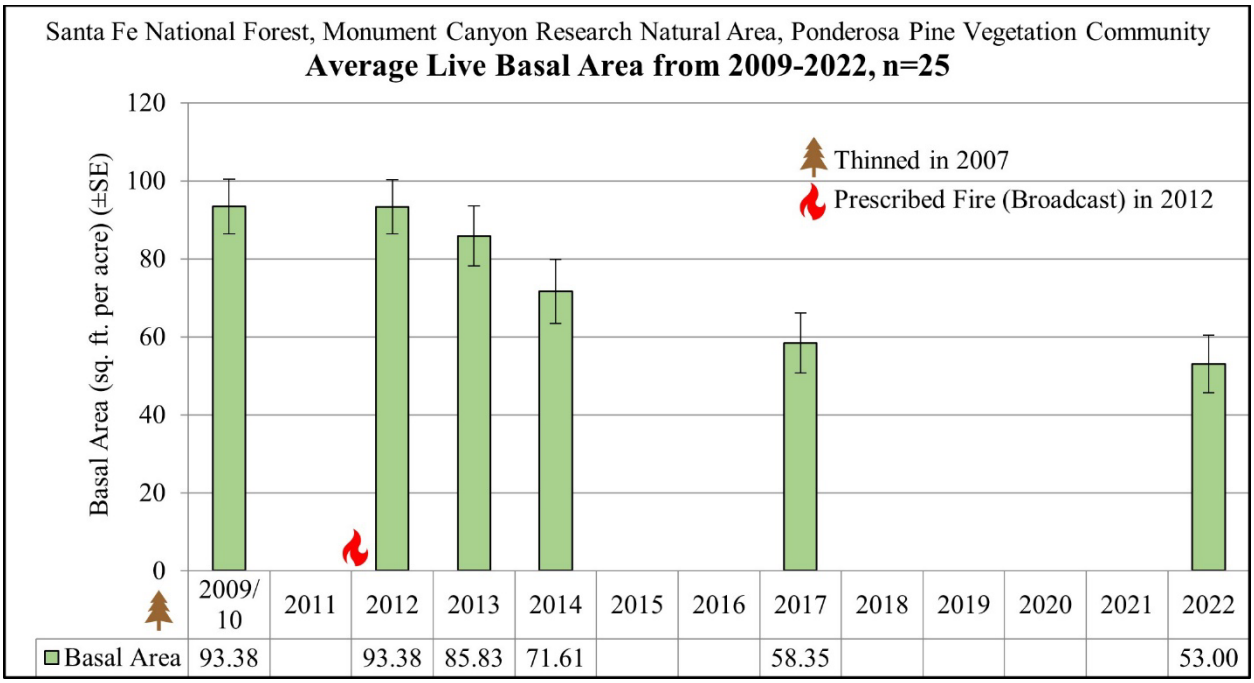



Figure 4.2.51. USFS, Monument Canyon live tree basal area results.

Average live tree basal area was recorded at 93.38 sq. ft. per acre in 2009/2010. After the broadcast prescribed fire in 2012, it was recorded at the same level, but decreased slightly in 2013 and 2014. In 2017, although live tree density increased, live tree basal area decreased to 58.35 sq. ft. per acre because the live tree density increase was due to small diameter trees. In 2022, live basal area appears to have decreased slightly to 53.00 sq. ft. per acre.

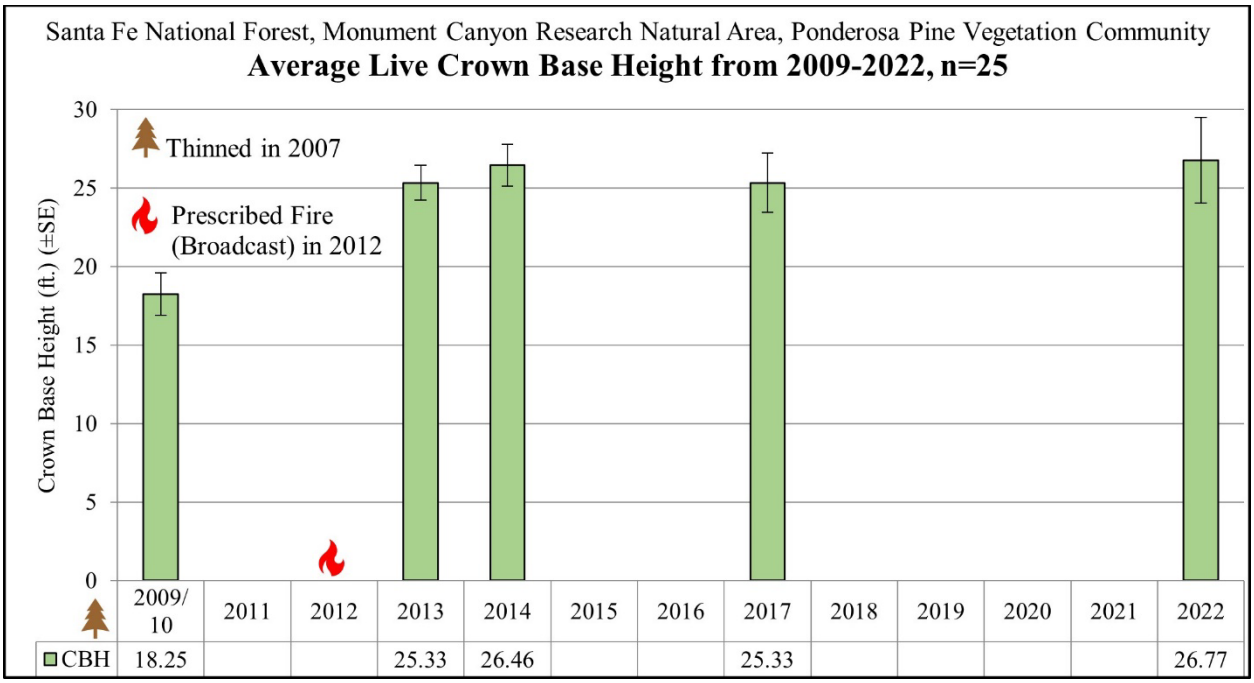



Figure 4.2.52. USFS, Monument Canyon live tree crown base height results.

Average live tree crown base height was recorded at 18.25 ft. in 2009/2010. In 2013, it was recorded at 25.33 ft., a 38.80 percent increase from 2009/2011, prior to the broadcast prescribed fire (2012). Live tree crown base height remained at similar levels in 2014, 2017, and 2022.

**Santa Fe National Forest, North Paliza Project Area, Ponderosa Pine Forest Vegetation Community**

Plots 11-15 (n=5) were installed in 2014, before the naturally ignited 2014 Pino Fire. Postburn data were collected between 2015 and 2023.

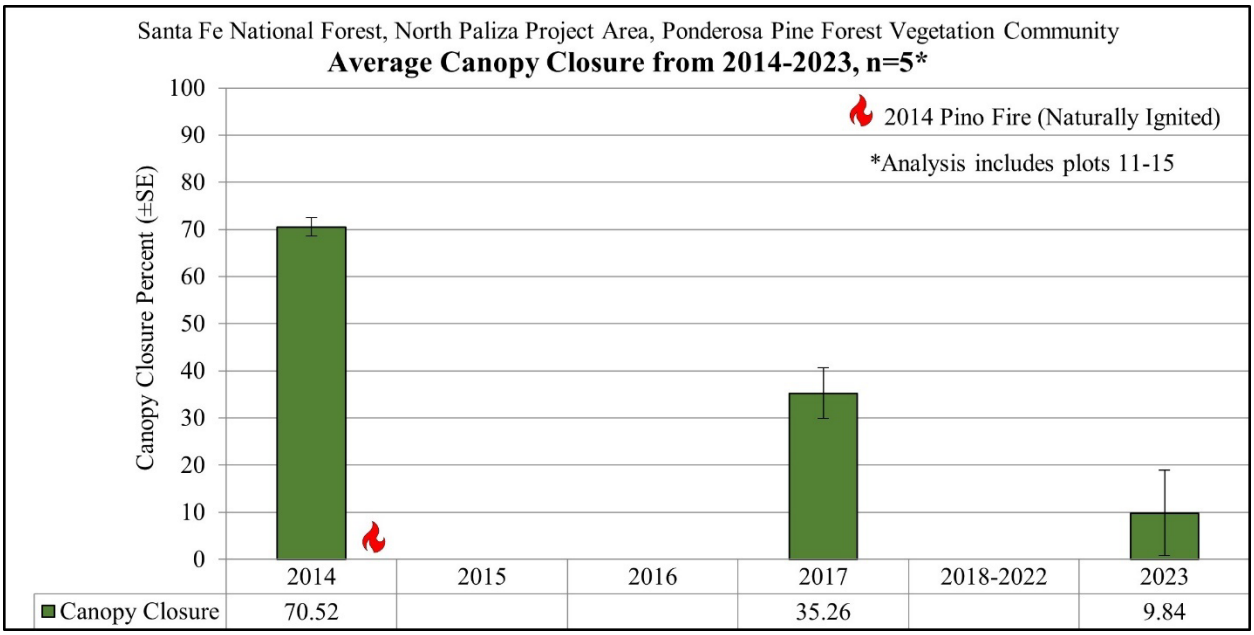



Figure 4.2.53. Santa Fe National Forest, North Paliza Project Area, Average Canopy Closure for plots 11-15. Valles Caldera Vegetation Program.

Summary statistics show a preburn average canopy closure of 70.52% (SE±1.99) and a postburn closure of 35.26% (SE±5.33) in 2017, a 50% decrease, supported by statistical evidence (estimated difference 35.26, 95% CI: [15.90, 54.62], p=0.002). By 2023, canopy closure was reduced to 9.84% cover, an 86% decrease from preburn conditions, and supported by statistical evidence (estimated difference 60.68, 95% CI: [41.32, 80.04], p=0.00005). This reduction of canopy closure indicates the 2014 Pino Fire burned at high-severity in this location of the forest.

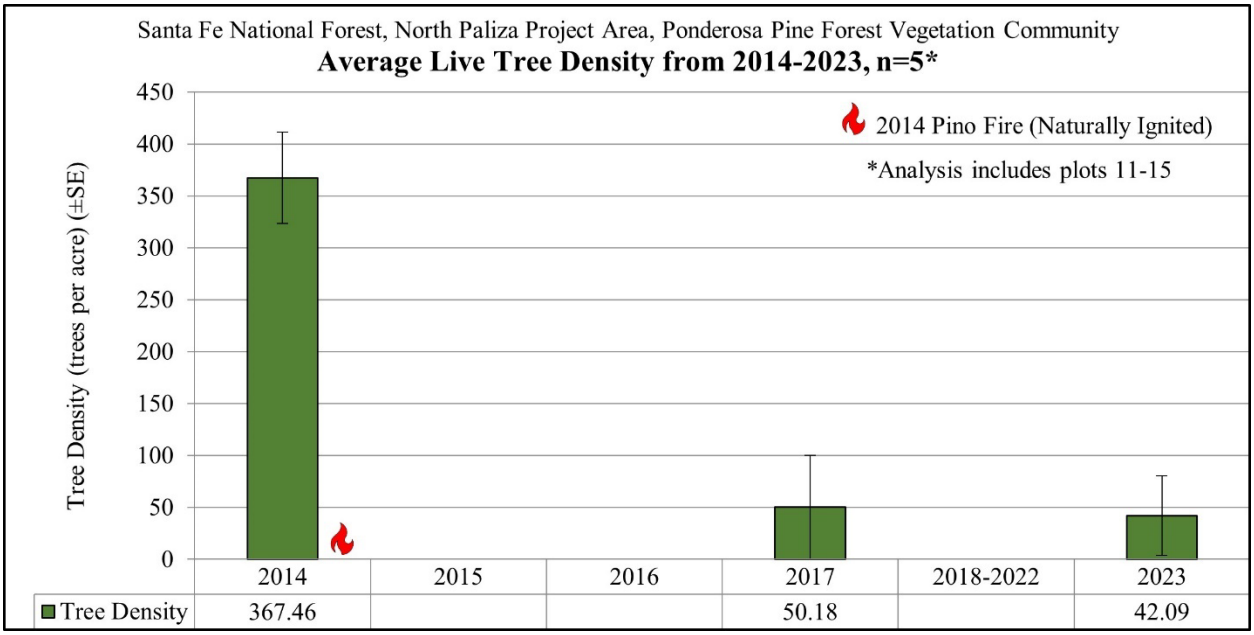



Figure 4.2.54. Santa Fe National Forest, North Paliza Project Area, Average Live Tree Density for plots 11-15. Valles Caldera Vegetation Program.

Summary statistics show a preburn average live tree density of 367.46 (SE±44.15) trees per acre, which is greater than estimated range of historical tree density of ponderosa pine forests. Postburn tree density is 50.18 (SE±50.18) trees per acre. There appears to be an 86.3% reduction in live tree density after the Pino Fire, supported by statistical evidence of a difference between values (95% CI: [149.29, 485.27], p=0.002). By 2023, live tree density was further reduced to 42.09 trees per acre, an 88.5% decrease from preburn density. Statistical analysis shows evidence of a difference between preburn and 2023 live tree densities (estimated difference 325.37, 95% CI: [157.38, 493.37], p=0.001). High preburn tree density likely contributed to postburn tree mortality due to potential for crown fire.

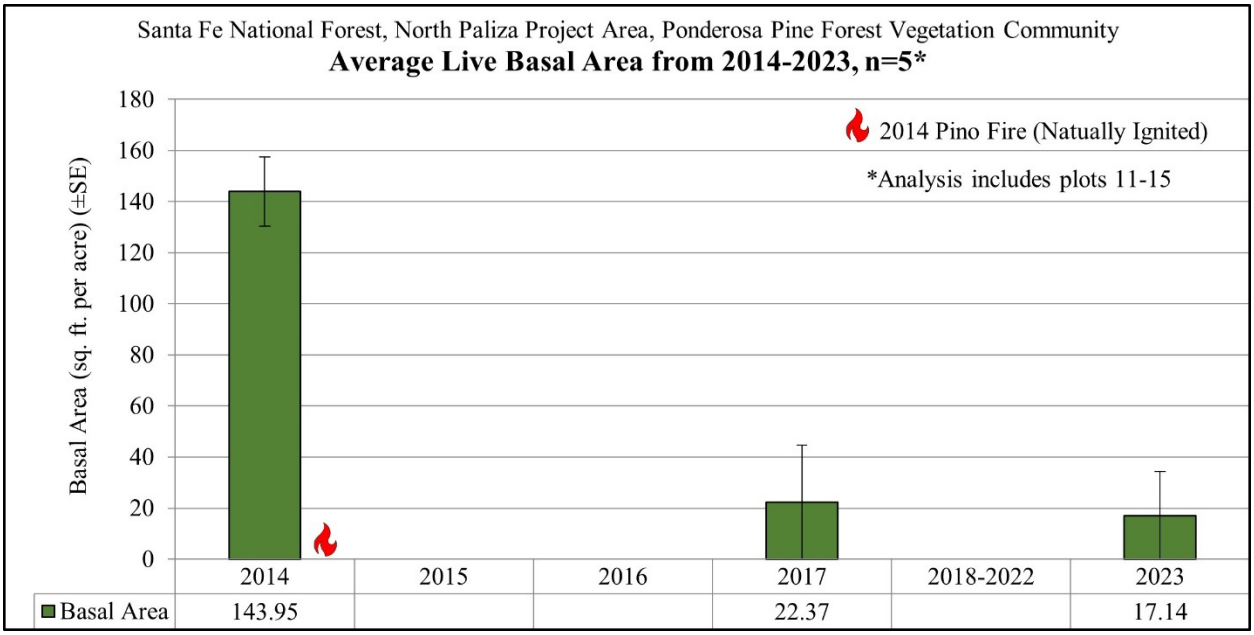



Figure 4.2.55. Santa Fe National Forest, North Paliza Project Area, Average Live Basal Area for plots 11-15. Valles Caldera Vegetation Program.

Summary statistics show a preburn average live basal area of 143.95 (SE±13.51) sq. ft. per acre and a postburn live basal area of 22.37 (SE±22.37) sq. ft. per acre in 2017 (an 84.5% decrease) and 17.41 (SE±17.10) sq. ft. per acre in 2023 (an 88.1% decrease). Preburn and postburn differences in values are supported by statistical evidence (estimated difference: 121.58, 95% CI: [54.84, 188.33], p=0.0021). Evidence of a difference between preburn live basal area and nine-years postburn (2023) is also supported by statistical evidence (estimated difference 126.81, 95% CI: [60.07, 193.55], p=0.0016).

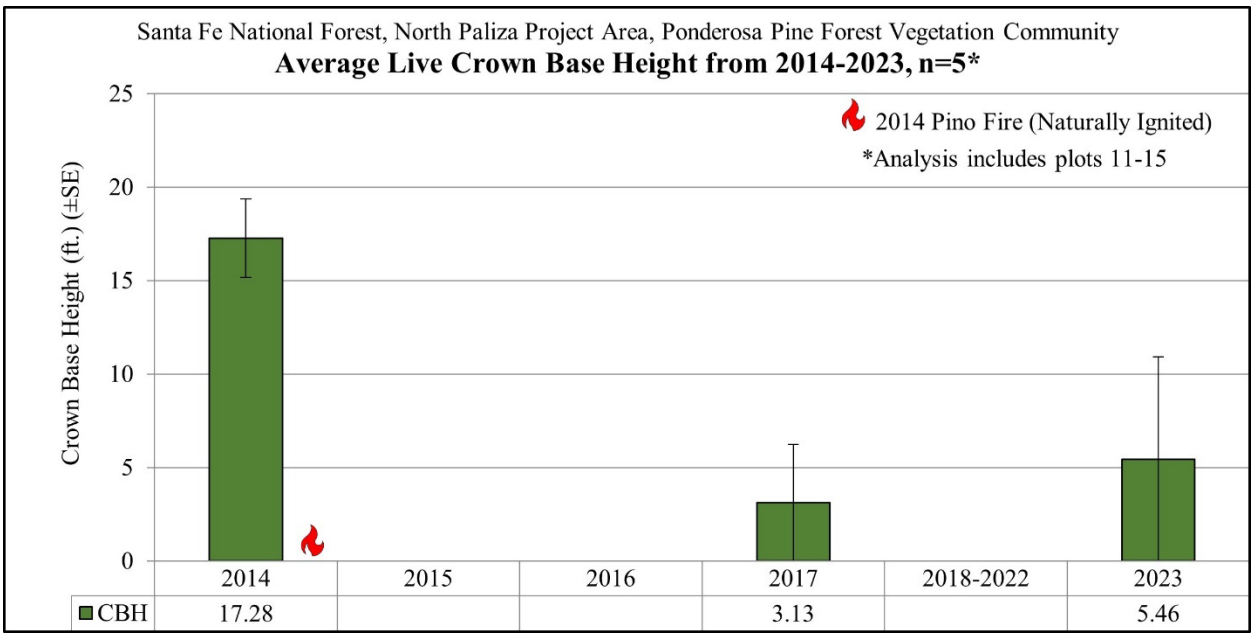



Figure 4.2.56. Santa Fe National Forest, North Paliza Project Area, Average Live Crown Base Height for plots 11-15. Valles Caldera Vegetation Program.

Summary statistics show a preburn average live crown base height of 17.28 (SE±2.10) ft. and a postburn crown base height of 3.13 (SE±3.13). This reduction of live crown base height is supported by statistical evidence (estimated difference 14.16, 95% CI: [1.27, 27.04], p=0.034). It appears that live crown base height increased in 2023 to 5.46 (SE±5.46) ft., however this difference from height in 2017 is not supported by statistical evidence (estimated difference -2.33, 95% CI: [-15.22, 10.55], p=0.865).

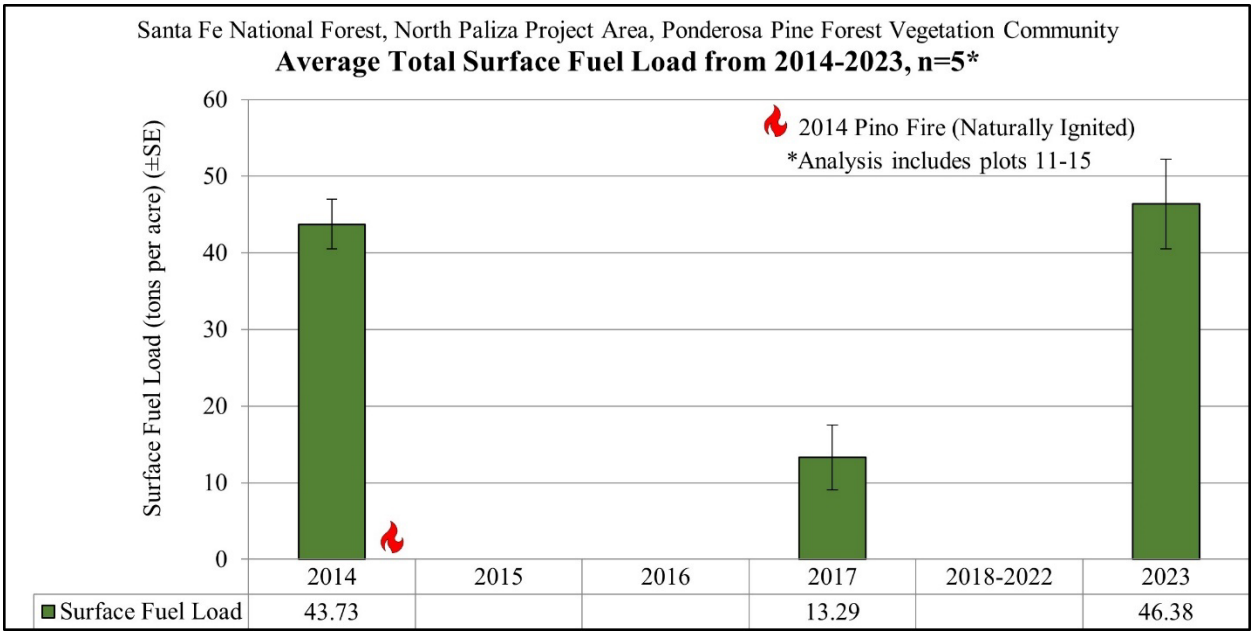



Figure 4.2.57. Santa Fe National Forest, North Paliza Project Area, Average Total Surface Fuel Load for plots 11-15. Valles Caldera Vegetation Program.

Summary statistics show a preburn average total surface fuel load of 43.73 (SE±3.23) tons per acre. Postburn average total surface fuel load is reported at 13.29 (SE±4.22) tons per acre. There appears to be a 30.45 tons per acre decrease in total surface fuel load after the 2014 Pino Fire and statistical analysis shows evidence of a difference between pre- and post-burn values (95% CI: [14.06, 46.83], p=0.002). Summary statistics from 2023 show total surface fuel load of 46.38 (SE±5.85) tons per acre, a 249% increase from the 2017 value. This increase is supported by statistical evidence (estimated difference -33.09, 95% CI: [-49.48, -16.70], p=0.001) and is likely due snags falling and adding to surface fuel loading between the years of 2017 and 2023.

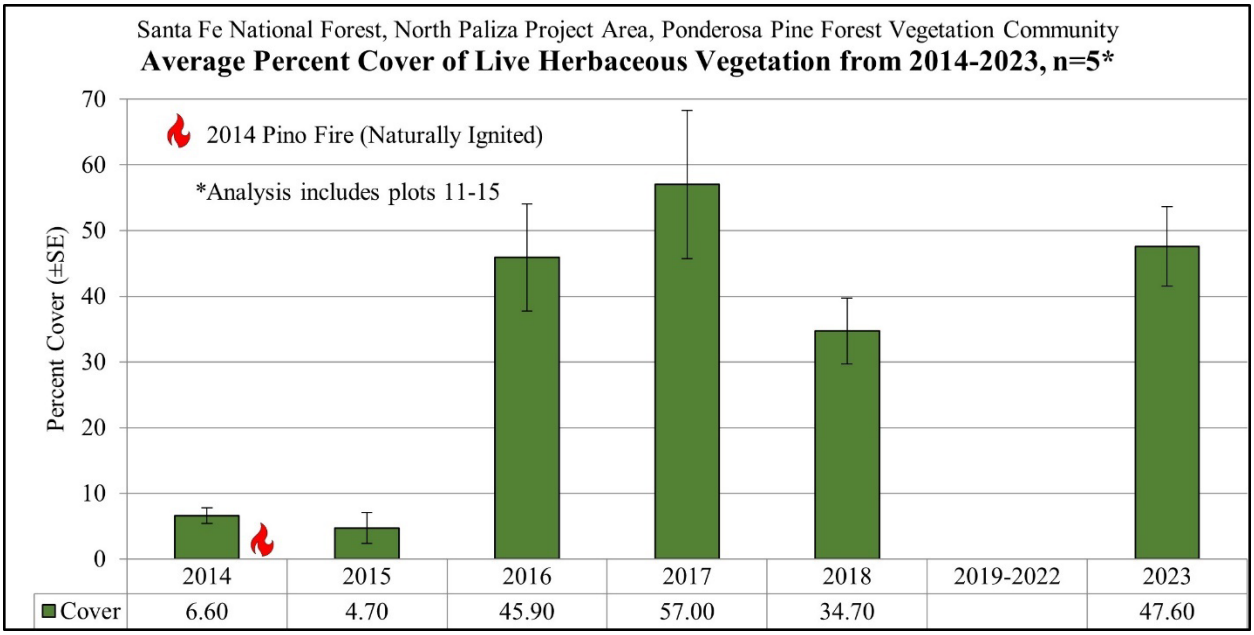
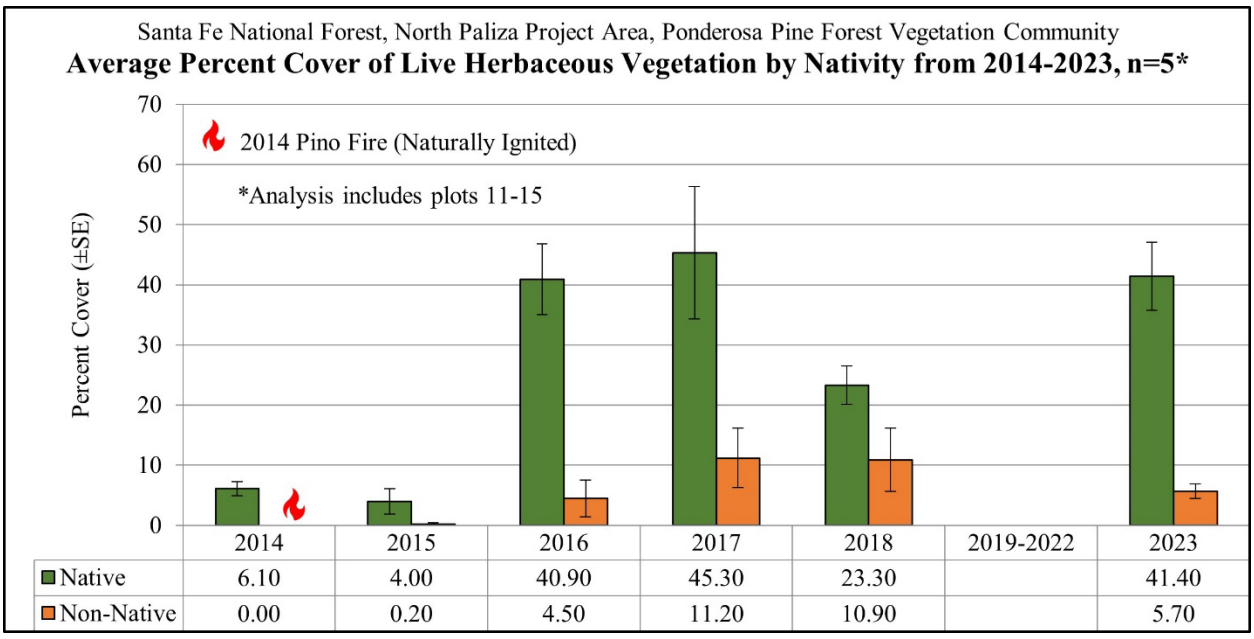



Figure 4.2.58. Santa Fe National Forest, North Paliza Project Area, Average Percent Cover of Live Herbaceous Vegetation for plots 11-15. Valles Caldera Vegetation Program.

Summary statistics show a preburn average live herbaceous cover of 6.60% (SE±1.17) and a postburn average cover of 4.70% (SE±2.33) in 2015. Statistical analysis does not show evidence of a difference between preburn and postburn values (estimated difference 1.90, 95% CI: [-26.02, 29.82], p=0.99993). Two years postburn, in 2016, percent cover increased to 45.90% (SE±8.16) and peaked in 2017 at 57.00% (SE±11.30). Statistical analysis shows evidence of a difference between preburn values and postburn years between 2016 and 2023 (Table 4.2.5.) and between postburn percent cover in 2015 and values across all subsequent years (Table 4.2.5.). The postburn increase in herbaceous cover is likely due to nutrient cycling initiated by the 2014 Pino Fire, however the one-year lag indicates an initial postburn recovery period. It is of note that herbaceous vegetation cover is greatly influenced by precipitation.

Table 4.2.5. Contrast tables for live herbaceous vegetation in North Paliza project area, plots 11-15. Comparisons between relevant years are denoted in the table below, for example, year2014-year2015 compares the averages between the two years.

| *Plots 11-15* | | | | |
|---|---|---|---|---|
| **Contrast** | **Estimated Difference** | **CI-low** | **CI-high** | **P-value** |
| year 2014 - year 2015 | 1.90 | -26.02 | 29.82 | 0.999928 |
| year 2014 - year 2016 | -39.30 | -67.22 | -11.38 | 0.003075 |
| year 2014 - year 2017 | -50.40 | -78.32 | -22.48 | 0.000190 |
| year 2014 - year 2018 | -28.10 | -56.02 | -0.18 | 0.047925 |
| year 2014 - year 2023 | -41.00 | -68.92 | -13.08 | 0.002001 |
| year 2015 - year 2016 | -41.20 | -69.12 | -13.28 | 0.001902 |
| year 2015 - year 2017 | -52.30 | -80.22 | -24.38 | 0.000119 |
| year 2015 - year 2018 | -30.00 | -57.92 | -2.08 | 0.030692 |
| year 2015 - year 2023 | -42.90 | -70.82 | -14.98 | 0.001238 |




| | 2014 | 2015 | 2016 | 2017 | 2018 | 2019-2022 | 2023 |
|---|---|---|---|---|---|---|---|
| Native | 6.10 | 4.00 | 40.90 | 45.30 | 23.30 | | 41.40 |
| Non-Native | 0.00 | 0.20 | 4.50 | 11.20 | 10.90 | | 5.70 |

Figure 4.2.59. Santa Fe National Forest, North Paliza Project Area, Average Percent Cover of Live Herbaceous Vegetation by Nativity for plots 11-15. Valles Caldera Vegetation Program.

Summary statistics show a preburn average live herbaceous native cover of 6.10% (SE±1.17) and no non-native vegetation detected. Postburn native cover decreased to 4.00% (SE±2.13) and non-native cover increased to 0.20% (SE±0.20). However, statistical analysis shows no evidence of a difference between pre- and postburn live herbaceous percent cover for both native vegetation (estimated difference 2.10, 95% CI: [-22.11, 26.31], p=0.9998) and non-native vegetation (estimated difference -0.2, 95% CI: [-12.54, 12.14], p=1.000). In 2016, one-year postburn, cover of native vegetation increased to 40.90% (SE±5.89) and non-native cover increased to 4.50% (SE±5.89) and both increases are supported by statistical evidence (see Table 4.2.6.). Percent cover of both native and non-native vegetation peaked in 2017, with average native cover reporting at 45.30% (SE±10.98) and average non-native cover at 11.20% (SE±4.96). The postburn increase in herbaceous cover is likely due to nutrient cycling initiated by the 2014 Pino Fire, however the one-year lag indicates an initial postburn recovery period. It is of note that herbaceous vegetation cover is greatly influenced by precipitation and that non-native cover typically increases following high-severity wildfire.

Table 4.2.6. Contrast tables for native and non-native live herbaceous vegetation in North Paliza project area, plots 11-15. Comparisons between relevant years are denoted in the table below, for example, year2014-year2015 compares the averages between the two years.

| *Native Live Herbaceous Vegetation* | | | | |
|---|---|---|---|---|
| **Contrast** | **Estimated Difference** | **CI-low** | **CI-high** | **P-value** |
| year 2014 - year 2015 | 2.10 | -22.11 | 26.31 | 0.999761 |
| year 2014 - year 2016 | -34.80 | -59.01 | -10.59 | 0.002497 |
| year 2014 - year 2017 | -39.20 | -63.41 | -14.99 | 0.000694 |
| year 2014 - year 2018 | -17.20 | -41.41 | 7.01 | 0.266699 |
| year 2014 - year 2023 | -35.30 | -59.51 | -11.09 | 0.002159 |
| year 2015 - year 2016 | -36.90 | -61.11 | -12.69 | 0.001354 |
| year 2015 - year 2017 | -41.30 | -65.51 | -17.09 | 0.000378 |
| year 2015 - year 2018 | -19.30 | -43.51 | 4.91 | 0.169379 |
| year 2015 - year 2023 | -37.40 | -61.61 | -13.19 | 0.001171 |
| year 2016 - year 2017 | -4.40 | -28.61 | 19.81 | 0.991861 |
| year 2016 - year 2018 | 17.60 | -6.61 | 41.81 | 0.245565 |
| year 2016 - year 2023 | -0.50 | -24.71 | 23.71 | 1.00000 |
| year 2017 - year 2018 | 22.00 | -2.21 | 46.21 | 0.088606 |
| year 2017 - year 2023 | 3.90 | -20.31 | 28.11 | 0.995335 |
| year 2018 - year 2023 | -18.10 | -42.31 | 6.11 | 0.220892 |
| *Non-Native Live Herbaceous Vegetation* | | | | |
| **Contrast** | **Estimated Difference** | **CI-low** | **CI-high** | **P-value** |
| year 2014 - year 2015 | -0.20 | -12.54 | 12.14 | 1.000000 |
| year 2014 - year 2016 | -4.50 | -16.84 | 7.84 | 0.855968 |
| year 2014 - year 2017 | -11.20 | -23.54 | 1.14 | 0.088958 |
| year 2014 - year 2018 | -10.90 | -23.24 | 1.44 | 0.102993 |
| year 2014 - year 2023 | -5.70 | -18.04 | 6.64 | 0.696266 |
| year 2015 - year 2016 | -4.30 | -16.64 | 8.04 | 0.877366 |
| year 2015 - year 2017 | -11.00 | -23.34 | 1.34 | 0.098114 |
| year 2015 - year 2018 | -10.70 | -23.04 | 1.64 | 0.113387 |
| year 2015 - year 2023 | -5.50 | -17.84 | 6.84 | 0.725753 |
| year 2016 - year 2017 | -6.70 | -19.04 | 5.64 | 0.542795 |
| year 2016 - year 2018 | -6.40 | -18.74 | 5.94 | 0.589086 |
| year 2016 - year 2023 | -1.20 | -13.54 | 11.14 | 0.999581 |
| year 2017 - year 2018 | 0.30 | -12.04 | 12.64 | 1.000000 |
| year 2017 - year 2023 | 5.50 | -6.84 | 17.84 | 0.725753 |
| year 2018 - year 2023 | 5.20 | -7.14 | 17.54 | 0.76825 |

Plots 16-20 (n=5) were installed in 2014. Thinning and piling was completed in 2016 and the piles were burned in 2021 and 2022. This portion of the project area burned in the 2022 Cerro Pelado Wildfire. Postburn data were collected in 2023.

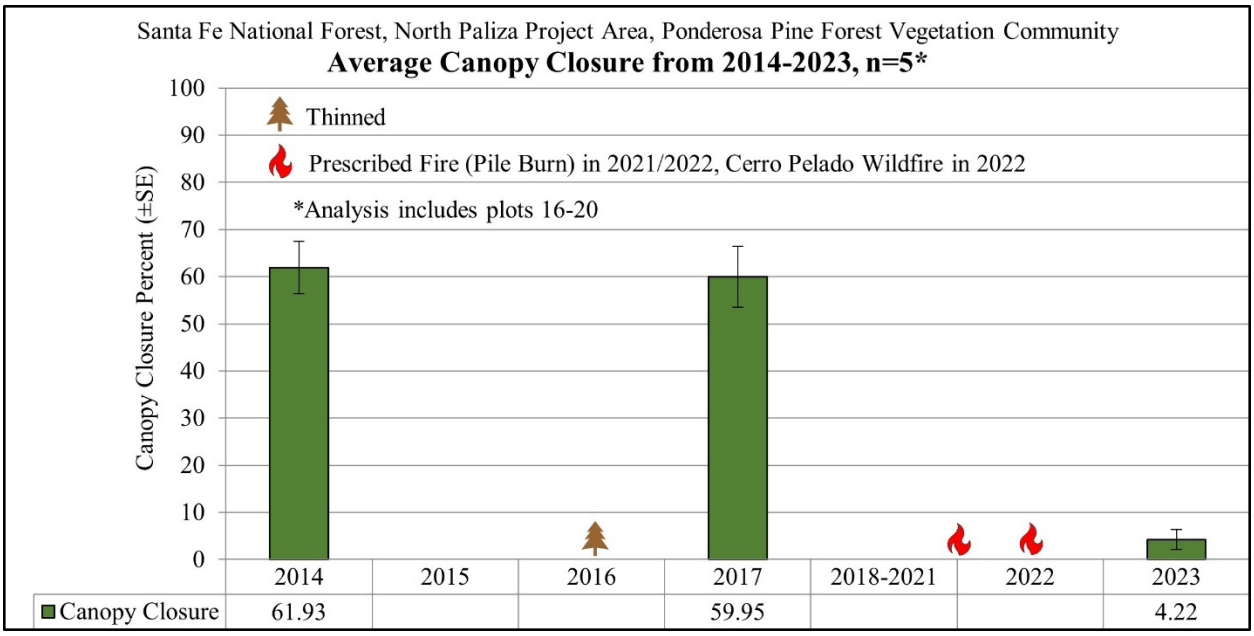



Figure 4.2.60. Santa Fe National Forest, North Paliza Project Area, Average Canopy Closure for plots 16-20. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average canopy closure of 61.93% (SE±5.55) and a post-thinning closure of 59.95% (SE±6.44). Canopy closure appears to be slightly reduced after thinning operations; however, statistical analysis does not show evidence of a difference between pre- and post-thinning values (estimated difference 1.98, 95% CI: [-10.42, 14.38], p=0.893). Postburn canopy closure was reduced by 93%, with an average closure of 4.22%. This decrease is supported by statistical evidence (estimated difference 55.73, 95% CI: [43.33, 68.13], p=0.000003) and is indicative of high tree mortality from the Cerro Pelado Fire.

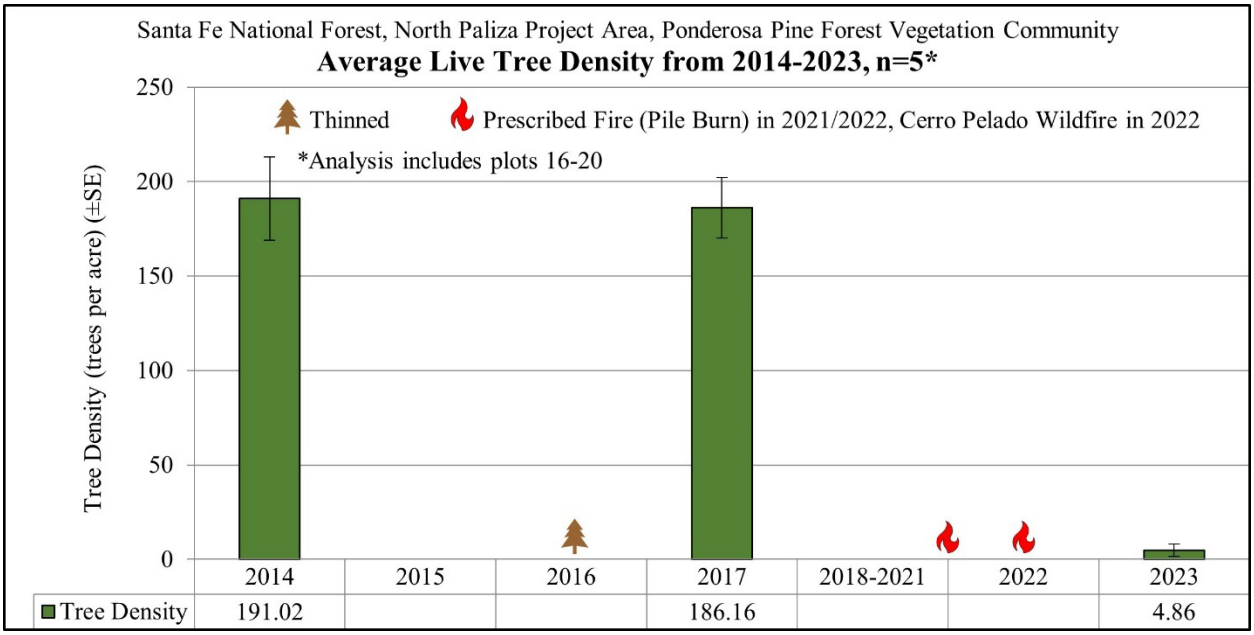



Figure 4.2.61. Santa Fe National Forest, North Paliza Project Area, Average Live Tree Density for plots 16-20. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average live tree density of 191.02 (SE±22.11) trees per acre. Post-thinning tree density is 186.16 (SE±15.98) trees per acre. There appears to be a 4.86 trees per acre reduction in live tree density after thinning operations, however this difference is not supported by statistical evidence (95% CI: [-38.04, 47.75], p=0.944). Postburn live tree density was greatly reduced, with an average of 4.86 (SE±3.24) trees per acre. Statistical analysis supports the difference between preburn and postburn densities (estimated difference 181.30, 95% CI: [138.41, 224.20], p=0.000005).

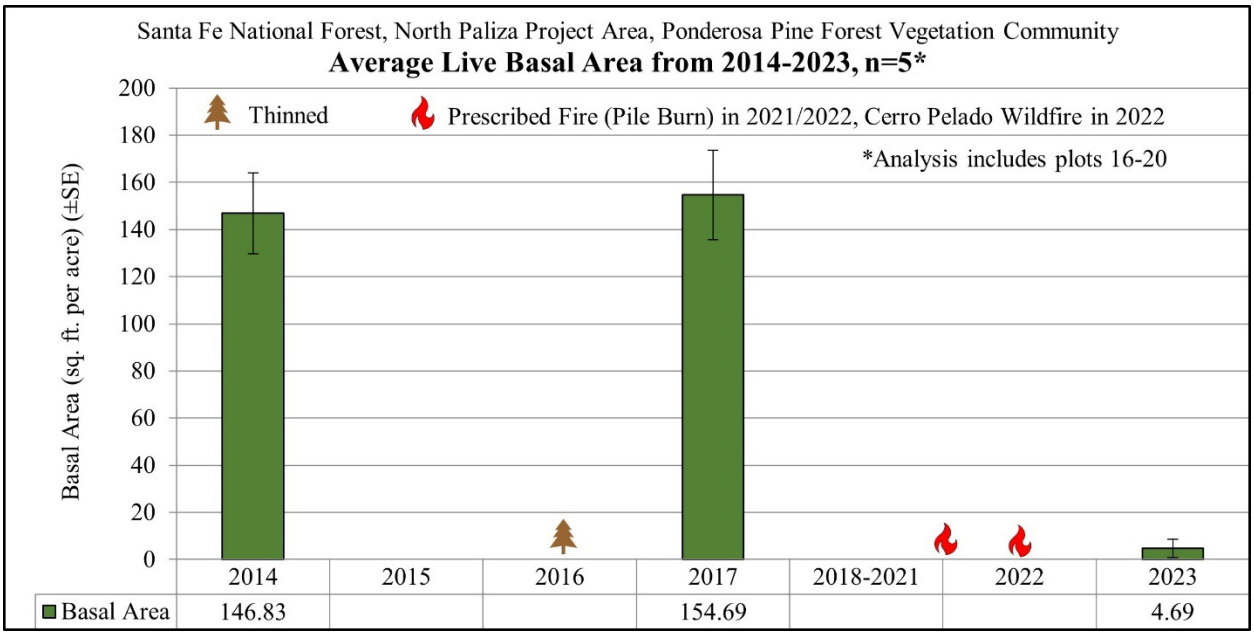



Figure 4.2.62. Santa Fe National Forest, North Paliza Project Area, Average Live Basal Area for plots 16-20. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average live basal area of 146.83 (SE±17.18) sq. ft. per acre and a post-thinning live basal area of 154.69 (SE±19.00) sq. ft. per acre, an unexpected slight increase live basal area after treatment. However, statistical analysis shows no evidence of a difference between pre-thinning and post-thinning values (estimate -7.85, 95% CI [-43.08, 27.37], p=0.804). Postburn live basal area was reduced by 96.97% from 2017, with an average live basal area of 4.69 sq. ft. per acre in 2023. This reduction is supported by statistical evidence (estimated difference: 150.00, 95% CI: [114.78, 185.23], p=0.000005) and is indicative of high-severity fire in this area of the Cerro Pelado Fire.

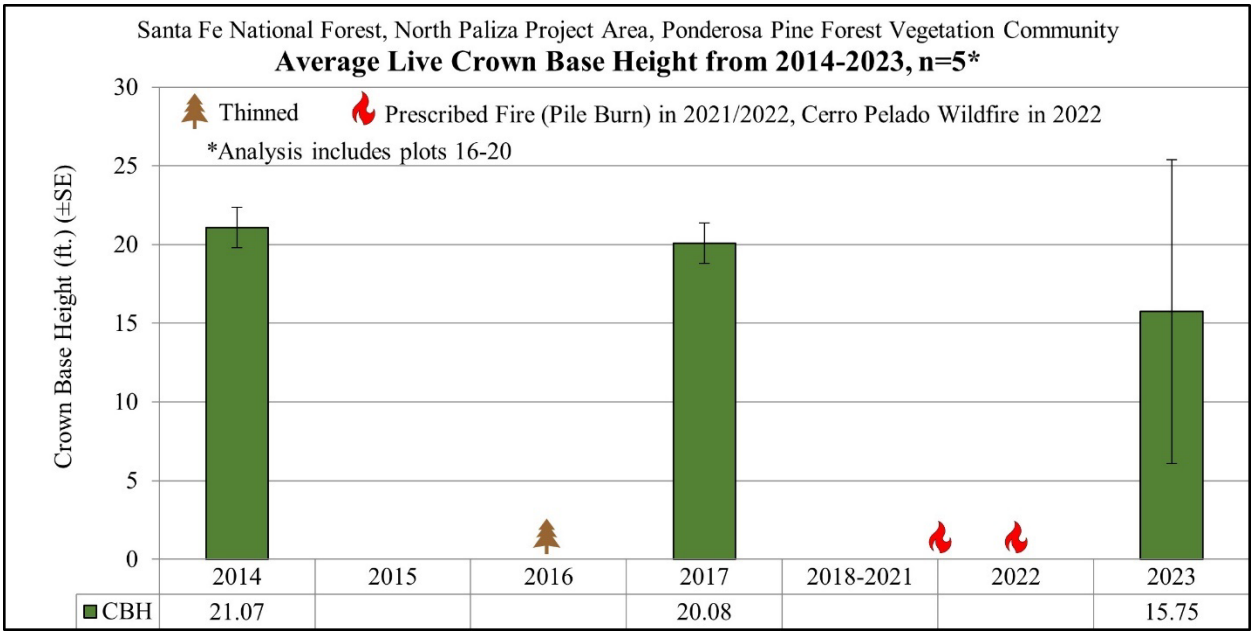



Figure 4.2.63. Santa Fe National Forest, North Paliza Project Area, Average Live Crown Base Height for plots 16-20. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average live crown base height of 21.07 (SE±1.29) ft. and a post-thinning crown base height of 20.08 (SE±4.13) ft. There appears to be a slight increase in live crown base height after thinning operations, as desired to reduce potential for fire spreading from the forest surface to the tree canopy. However, there is no statistical evidence to support a difference between pre- and post-thinning values (estimated difference 4.86, 95% CI: [-19.89, 21.85], p=0.990). Postburn live crown base height was further reduced to 15.75 (SE±9.65) ft. on average, though statistical analysis shows no evidence of a difference pre- and postburn (estimated difference 4.34, 95% CI: [-16.53, 25.20], p=0.827).

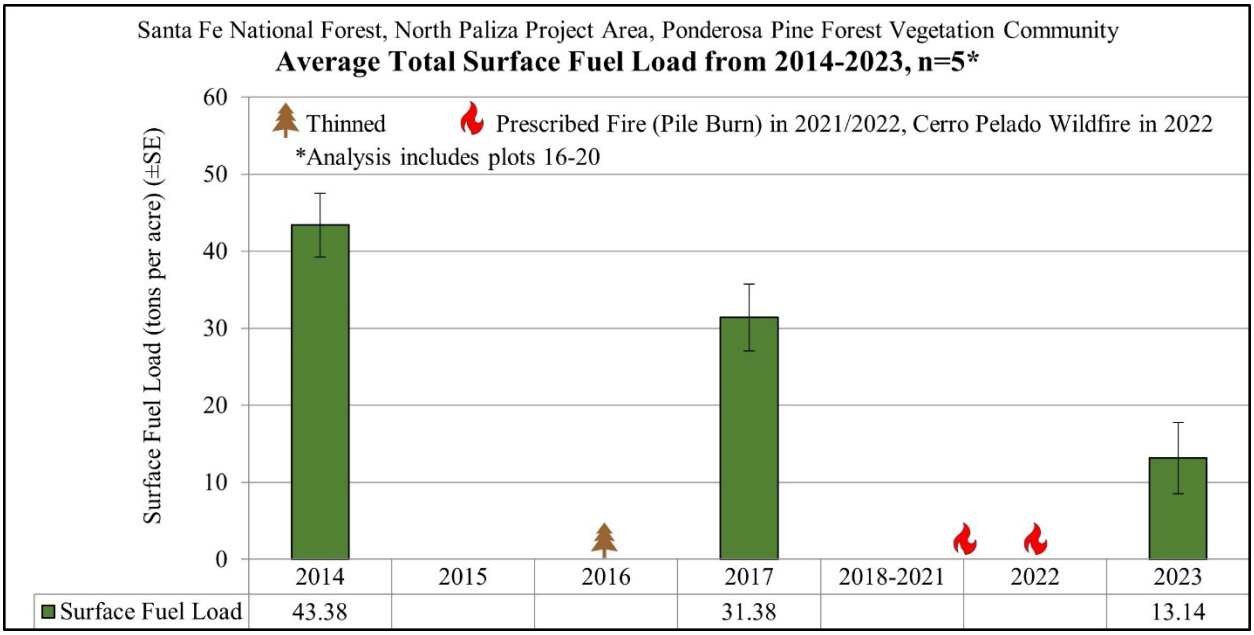



Figure 4.2.64. Santa Fe National Forest, North Paliza Project Area, Average Total Surface Fuel Load for plots 16-20. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning average total surface fuel load of 43.38 (SE±4.13) tons per acre. Post-thinning average total surface fuel load is reported at 31.38 (SE±4.33) tons per acre. There appears to be a 12.00 tons per acre decrease in total surface fuel load after thinning operations and statistical analysis shows evidence of a difference between pre- and post-thinning values (95% CI: [1.13, 22.87], p=0.032). Postburn average total surface fuel load is reported at 13.14 (SE±4.61) tons per acre and there is evidence of a difference between postburn and 2014 surface fuel load (estimated difference 30.24, 95% CI: [19.37, 41.11], p=0.0001) and postburn and 2017 surface fuel load (estimated difference 18.24, 95% CI: [7.37, 29.11], p=0.003).

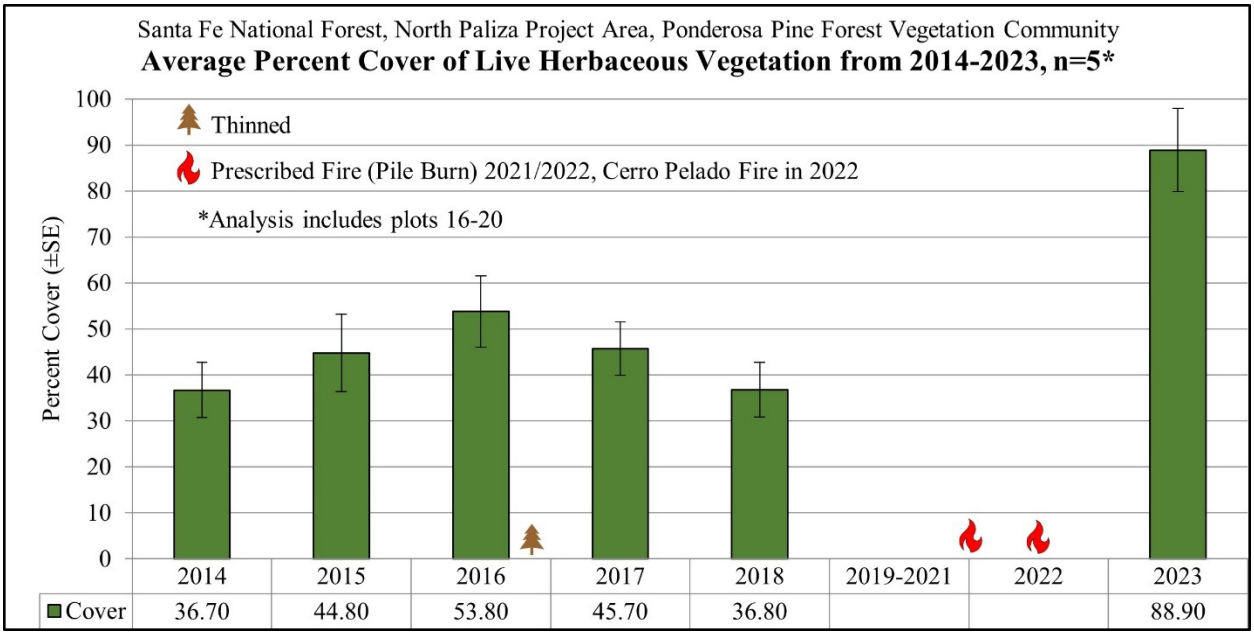
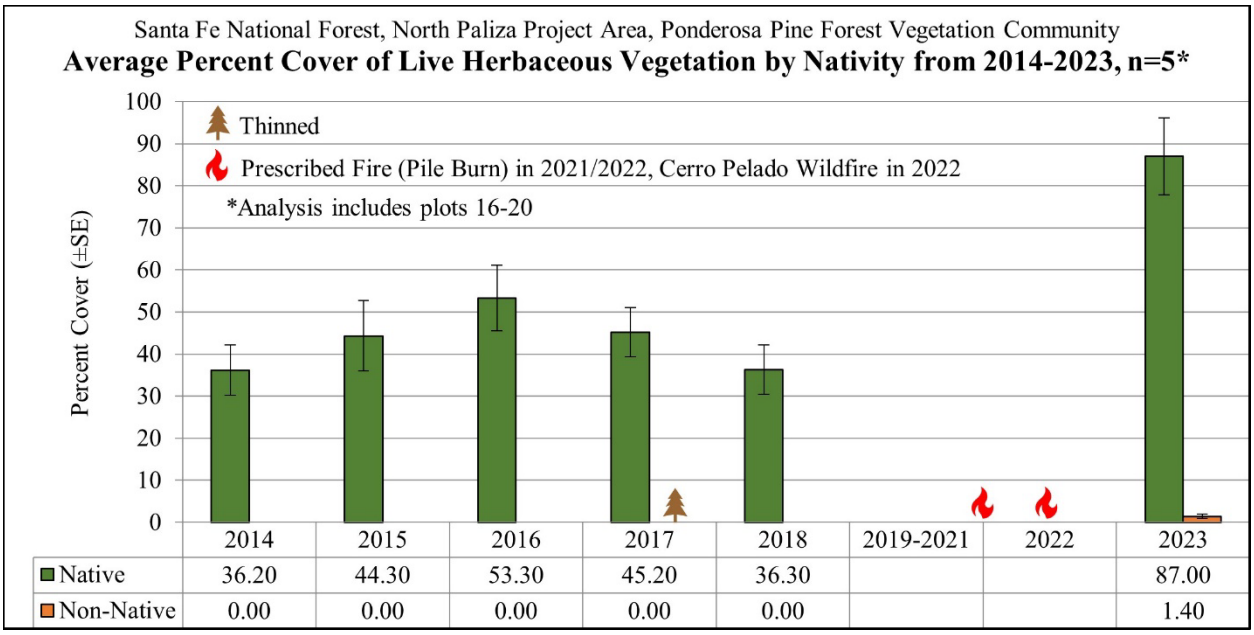



Figure 4.2.65. Santa Fe National Forest, North Paliza Project Area, Average Percent Cover of Live Herbaceous Vegetation for plots 16-20. Valles Caldera Vegetation Program.

Summary statistics show a pre-thinning (2016) average live herbaceous cover of 53.80% (SE±7.79) and a post-thinning (2017) average cover of 45.70% (SE±5.85). Statistical analysis does not show evidence of a difference between pre-thinning and post-thinning values (estimated difference 8.1, 95% CI: [-10.80, 27.00], p=0.756). Prior to thinning, there are fluctuations between cover from 2014 to 2016, but the changes can likely be attributed to precipitation. Following the 2022 Cerro Pelado Fire, average live herbaceous cover increased to 88.90% (SE±9.04) in 2023, and statistical analysis shows evidence of a difference between this value and all other years previously sampled, regardless of thinning treatment status (Table 4.2.7.). It is of note that herbaceous vegetation cover is greatly influenced by precipitation.

Table 4.2.7. Contrast tables for live herbaceous vegetation in North Paliza project area, plots 16-20. Comparisons between relevant years are denoted in the table below, for example, year2014-year2015 compares the averages between the two years.

| *Plots 16-20* | | | | |
|---|---|---|---|---|
| **Contrast** | **Estimated Difference** | **CI-low** | **CI-high** | **P-value** |
| year 2014 - year 2023 | -52.20 | -71.10 | -33.30 | 0.0000004399 |
| year 2015 - year 2023 | -44.10 | -63.00 | -25.20 | 0.0000058149 |
| year 2016 - year 2023 | -35.10 | -54.00 | -16.20 | 0.0001333142 |
| year 2017 - year 2023 | -43.20 | -62.10 | -24.30 | 0.0000078579 |
| year 2018 - year 2023 | -52.10 | -71.00 | -33.20 | 0.0000004535 |




| | 2014 | 2015 | 2016 | 2017 | 2018 | 2019-2021 | 2022 | 2023 |
|---|---|---|---|---|---|---|---|---|
| Native | 36.20 | 44.30 | 53.30 | 45.20 | 36.30 | | | 87.00 |
| Non-Native | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | | | 1.40 |

Figure 4.2.66. Santa Fe National Forest, North Paliza Project Area, Average Percent Cover of Live Herbaceous Vegetation by Nativity for plots 16-20. Valles Caldera Vegetation Program.

Summary statistics show a pre-thin (2017) average live herbaceous native cover of 45.20% (SE±5.85) and no non-native vegetation detected. Post-thin native cover decreased to 36.30% (SE±5.90) and non-native cover remained undetected. However, statistical analysis shows no evidence of a difference between pre- and post-thin live herbaceous percent cover for both native vegetation (estimated difference 8.90, 95% CI: [-9.65, 27.45], p=0.663) and non-native vegetation (estimated difference 0.0, 95% CI: [-0.97, 0.97], p=1.000). Summary statistics show a postburn average live herbaceous native cover of 87.00% (SE±9.13), a 140% increase from 2018. This increase is supported by statistical evidence (estimated difference -50.70, 95% CI: [-69.25, -32.15], p=0.0000005). Postburn average live herbaceous non-native cover was detected, with the average reported at 1.40% (SE±0.53). Statistical analysis shows evidence of a difference between preburn and postburn non-native cover (estimated difference -1.40, 95% CI: [-2.37, -0.43], p=0.002). The postburn increase in both native and non-native herbaceous cover is likely due to nutrient cycling initiated by the 2022 Cerro Pelado Fire. It is of note that herbaceous vegetation cover is greatly influenced by precipitation and that non-native cover typically increases following high-severity wildfire.

**Santa Fe National Forest, San Juan Mesa Project Area, Ponderosa Pine Forest Vegetation Community**

The San Juan Project Area is a mid-elevation ponderosa pine forest and savanna with a combined overstory and midstory tree density of approximately 209 trees per acre. The overstory, approximately 157 trees per acre, is comprised mostly of Ponderosa pine (*Pinus ponderosa*). Douglas fir (*Psuedotsuga menziesii*) and white fir (*Abies concolor*) are a small component and limber pine (*Pinus flexilis*) is occasionally present. The midstory contains approximately 133 trees per acre. Two-needle pinyon (*Pinus edulis*) and one-seed juniper (*Juniperus monosperma*) are common midstory trees. Alligator juniper (*Juniperus deppeana*) is present but less common. Oak (*Quercus spp*.) is often present, occasionally as dense thickets, due to a history of fire exclusion. (Note: The biological description is based on Fire Ecology data collected on 20 plots between 2010 and 2016.)

Reproduction is present at approximately 20 trees per acre, consisting primarily of ponderosa pine (16 trees per acre) with a smaller component of two-needle pinyon, one-seed juniper, and alligator juniper. Douglas fir is occasionally present. Surface fuels are low, at approximately 8 tons per acre, with litter and duff load as the greatest contributor at 4 tons per acre. 1000-hour fuels are approximately 2 tons per acre. 1, 10, and 100-hour fuels are approximately 2 tons per acre. Litter and duff depths are approximately 0.6 inches. Shrub density is approximately 106 stems per acre. Common shrubs include oak, New Mexico locust (*Robinia neomexicana*), Fendeler's ceanothus (*Ceanothus fendleri*), and Apache plume (*Fallugia paradoxa*). The surface layer is commonly long needle litter (73% cover), bare ground (3.5% cover), and wood (3.6% cover) but also includes a sparse herbaceous component at approximately 20% cover. The lifeform with the highest percent cover (7.5%) is graminoid (grass), followed by tree at 5%, shrub at 4%, forbs (wildflowers) at 2.9%, and grass-like at 1%. Graminoid percent cover is comprised of 7.48% native species and 0.02% non-native; tree, shrub, and grass-like species are entirely native; forbs consist of 2.8% native species and 0.1% non-native.

Common grass species include mountain muhly (*Muhlenbergia montana*), blue grama (*Bouteloua gracilis*), and prairie Junegrass (*Koeleria macrantha*), all native perennial grasses. Predominant forb species include Carruth's sagewort (*Artemisia carruthii*), Colorado rubberweed (*Hymenoxys richardsonii*), and Fendler's sandmat (Chamaesyce fendleri), all native and perennial. Species richness is recorded at 137 species, including 133 native species and 4 non-native species. Non-native species include two perennial grasses, Kentucky bluegrass (*Poa pratensis*) and smooth brome (*Bromus inermis*), a biennial forb, common mullein (*Verbascum Thapsus*), and a perennial forb, common dandelion (*Taraxacum officinale*).

The Santa Fe National Forest implemented a broadcast prescribed fire in the San Juan Project Area in 2012. Fire Ecology data were collected on 20 plots in 2010/2011, prior to the prescribed fire, and after the fire in 2012, 2016, and 2021.

**Surface Fuels Monitoring**

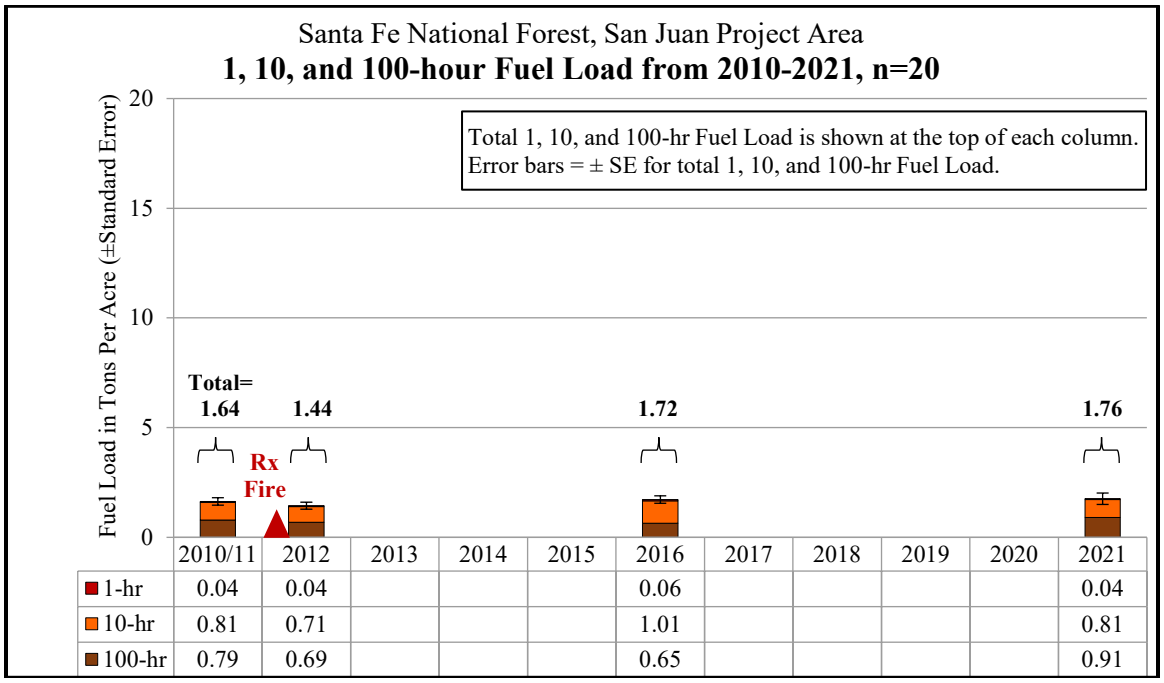



Figure 4.2.67. San Juan Mesa fuel load results.

Average 1, 10, and 100-hour fuel load was recorded at 1.64 tons per acre in 2010/2011. After the broadcast prescribed fire in 2012, it was recorded at 1.44 tons per acre, a 12.2 percent decrease from pre-fire levels. In 2016, 1, 10, and 100-hour fuels were recorded at 1.72 tons per acre and remained low at 1.76 tons per acre in 2021.



Figure 4.2.68. Prescribed fire in San Juan Project Area, 2012.
Photo: Pueblo and Four Winds Fire Ecology Program

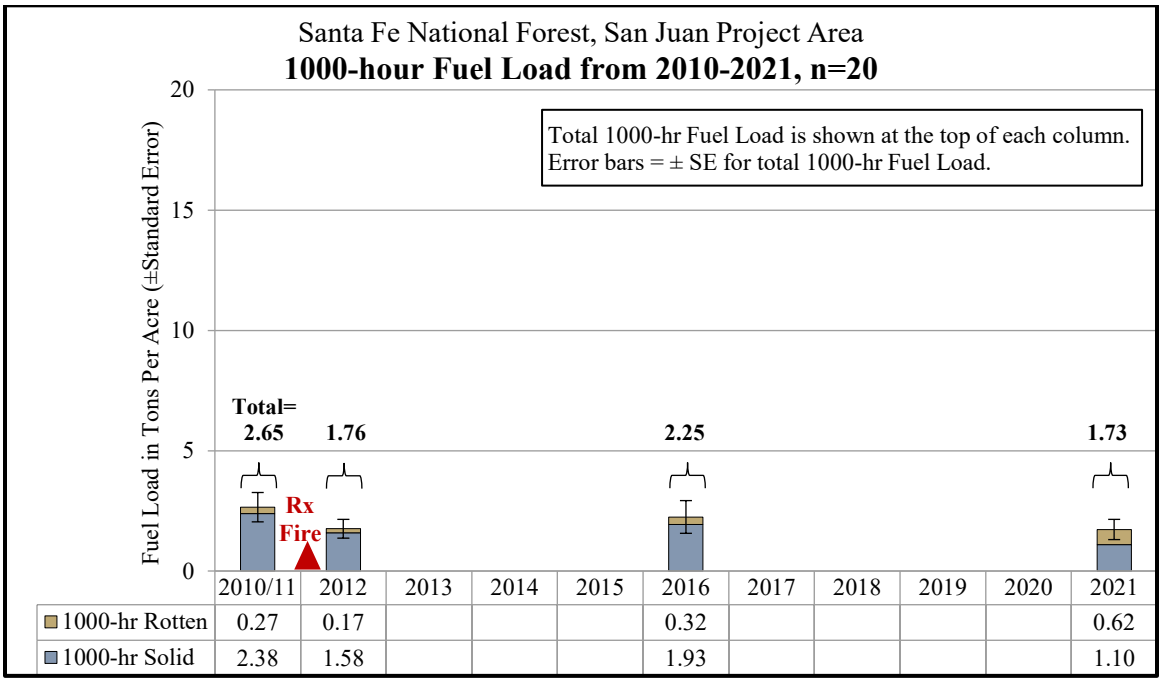



Figure 4.2.69. San Juan Mesa 1000-hour fuel load results.

Average 1000-hour fuel load was recorded at 2.65 tons per acre in 2010/2011, with solid 1000-hour fuels as the largest contributor at 2.38 tons per acre and rotten 1000-hour fuels at 0.27. After the broadcast prescribed fire in 2012, 1000-hour fuels were recorded at 1.76 tons per acre, a 33.59 percent decrease from pre-fire levels. In 2016, 1000-hour fuels were recorded at 2.25 tons per acre. In 2021, 1000-hour fuel load was recorded at 1.73 tons per acre.

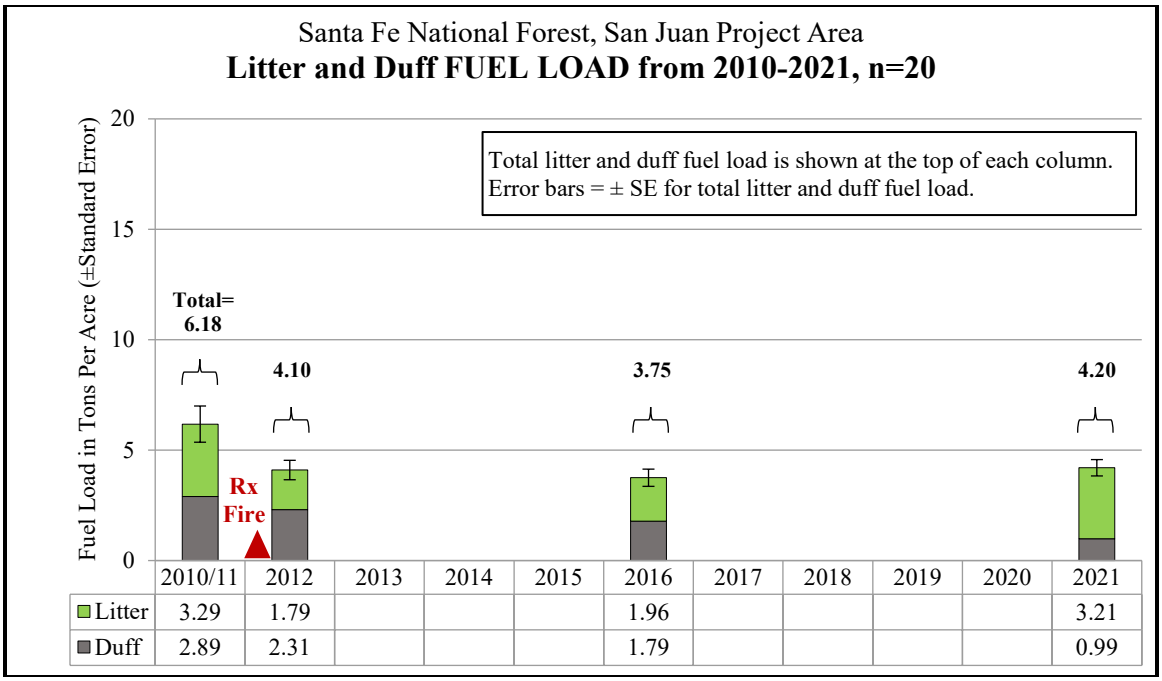



Figure 4.2.70. San Juan Mesa litter and duff fuel load results.

Average litter and duff fuel load was recorded at 6.18 tons per acre in 2010/2011, with litter contributing 3.29 tons per acre and duff at 2.89. After the broadcast prescribed fire in 2012, litter and duff fuel load was recorded at 4.10 tons per acre, a 33.66 percent decrease from pre-fire levels. In 2016 and 2021, there was little recorded change in litter and duff fuel load, at 3.75 tons per acre and 4.20, respectively.

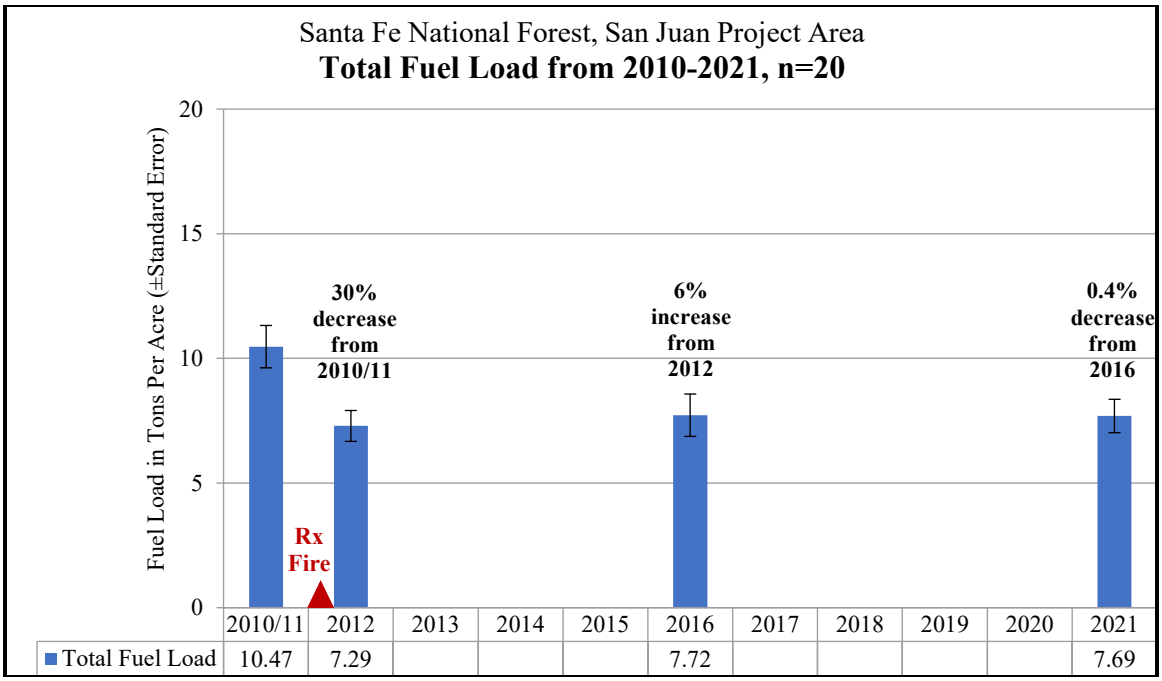



Figure 4.2.71. San Juan Mesa total fuel load results.

Average total fuel load was recorded at 10.47 tons per acre in 2010/2011. After the broadcast prescribed fire in 2012, total fuel load was recorded at 7.29 tons per acre, a 30.37 percent decrease from pre-fire levels. In 2016 and 2021, no change was recorded in total fuel load, 7.72 and 7.69 tons per acre, respectively.

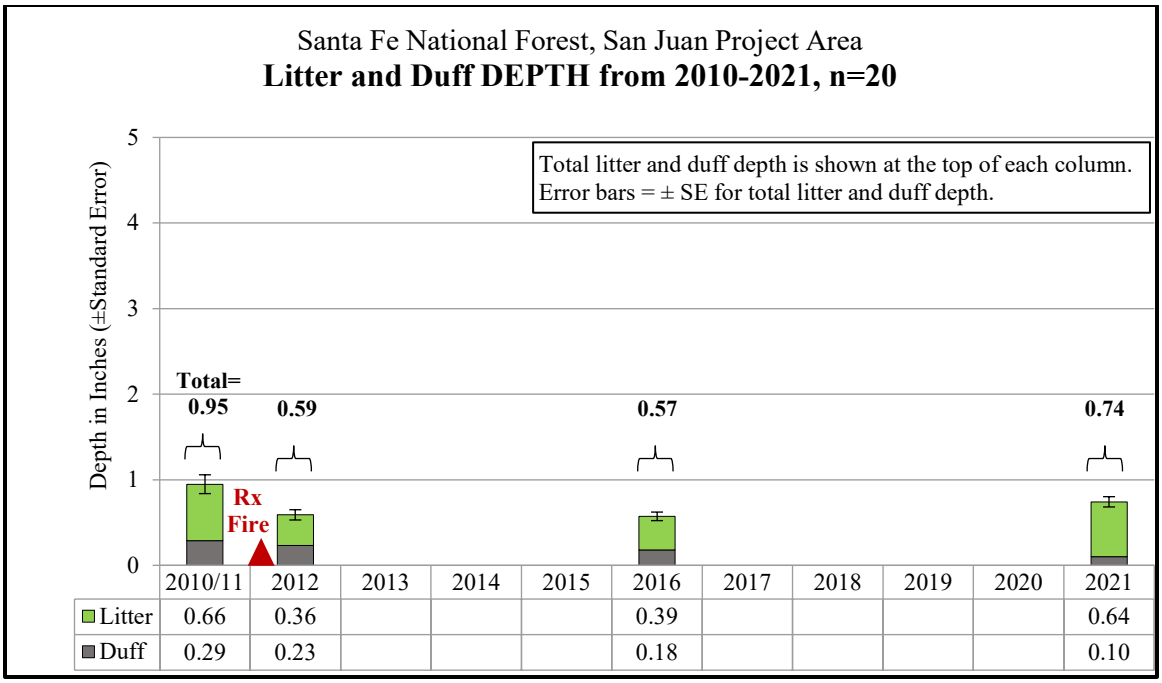



Figure 4.2.72. San Juan Mesa litter and duff depth results.

Average litter and duff depth was recorded at 0.95 inches in 2010/2011, with litter as the largest contributor at 0.66 inches and duff at 0.29. After the broadcast prescribed fire in 2012, litter and duff depth was recorded at 0.59 inches, a 37.90 percent decrease from pre-fire levels. In 2016, litter and duff depths were recorded at 0.57 inches. A small increase in litter and duff depths was recorded in 2021, at 0.74 inches.

## Tree Density

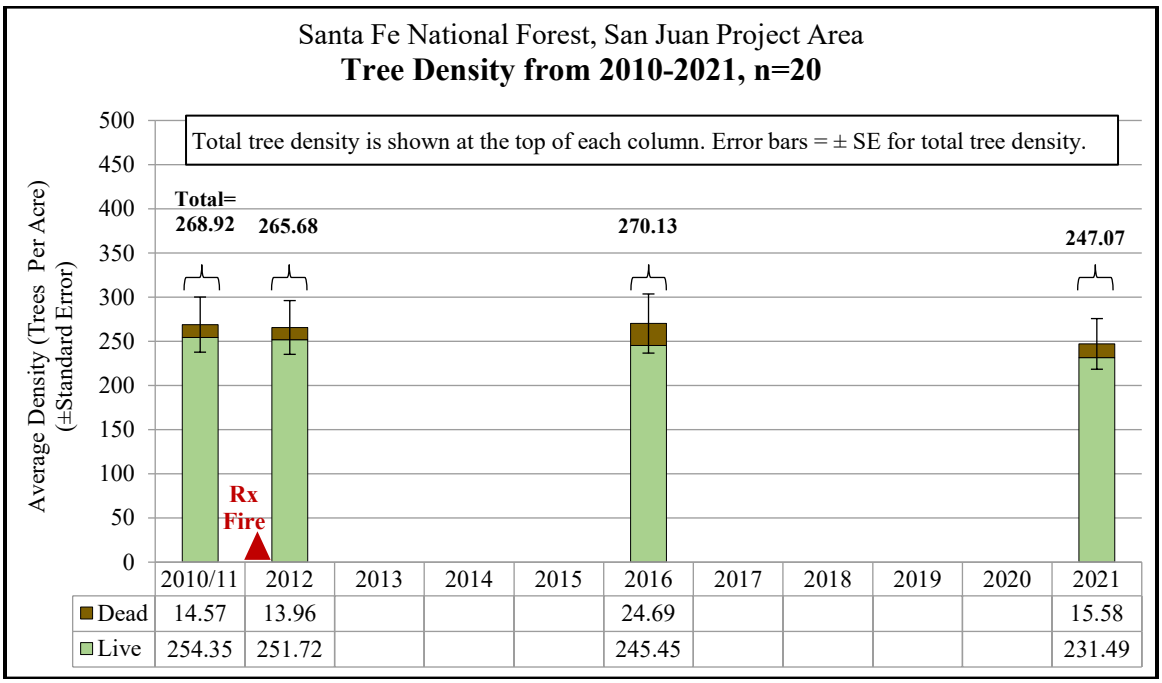



Figure 4.2.73. San Juan Mesa tree density results.

Average tree density was recorded at 268.92 trees per acre in 2010/2011, with live trees as the largest contributor at 254.35 trees per acre and dead trees at 14.57. After the broadcast prescribed fire in 2012, tree density was similar to pre-fire levels, at 265.68 trees per acre. In 2016, tree density was recorded at 270.13 trees per acre, with an increase in dead trees and a decrease in live trees (a shift in the ratio of live to dead trees). Tree density was recorded at 247.07 in 2021, an 8.54 percent reduction from 2016, with both live and dead tree categories showing a reduction in density.

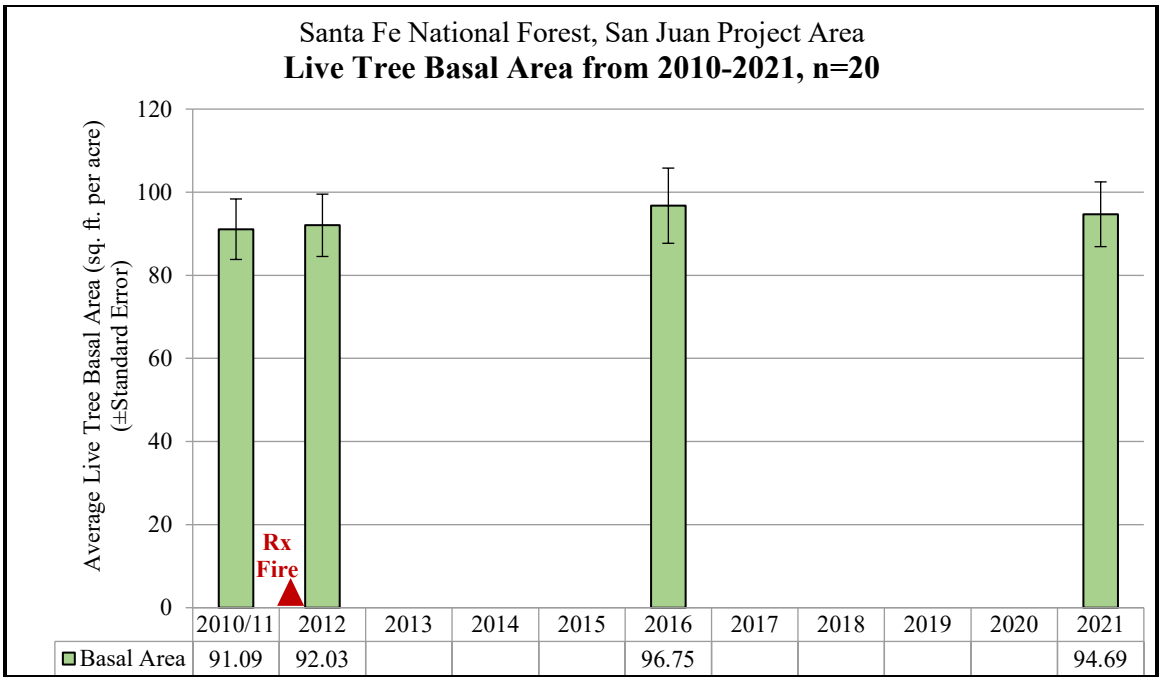



Figure 4.2.74. San Juan Mesa live tree basal area results.

Average live tree basal area was recorded at 91.09 sq. ft. per acre in 2010/2011. After the broadcast prescribed fire in 2012, it was nearly the same at 92.03 sq. ft. per acre. Live tree basal area increased slightly to (96.75 sq. ft. per acre) in 2016 and remained similar in 2021 at 94.69 sq. ft. per acre.

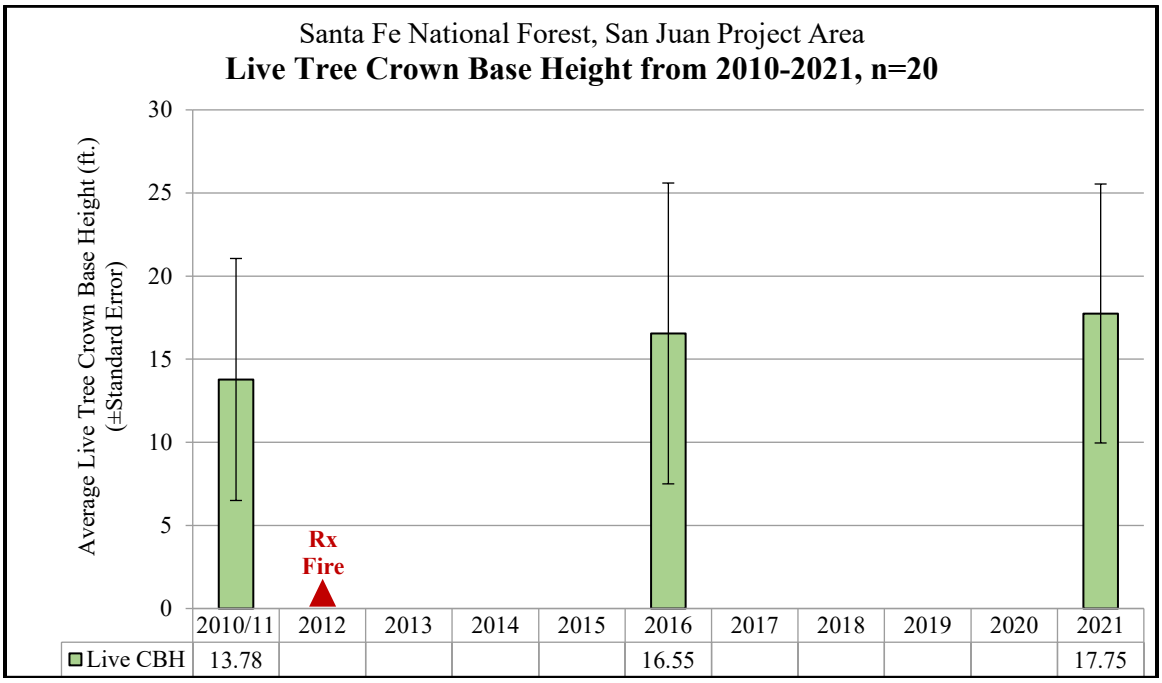
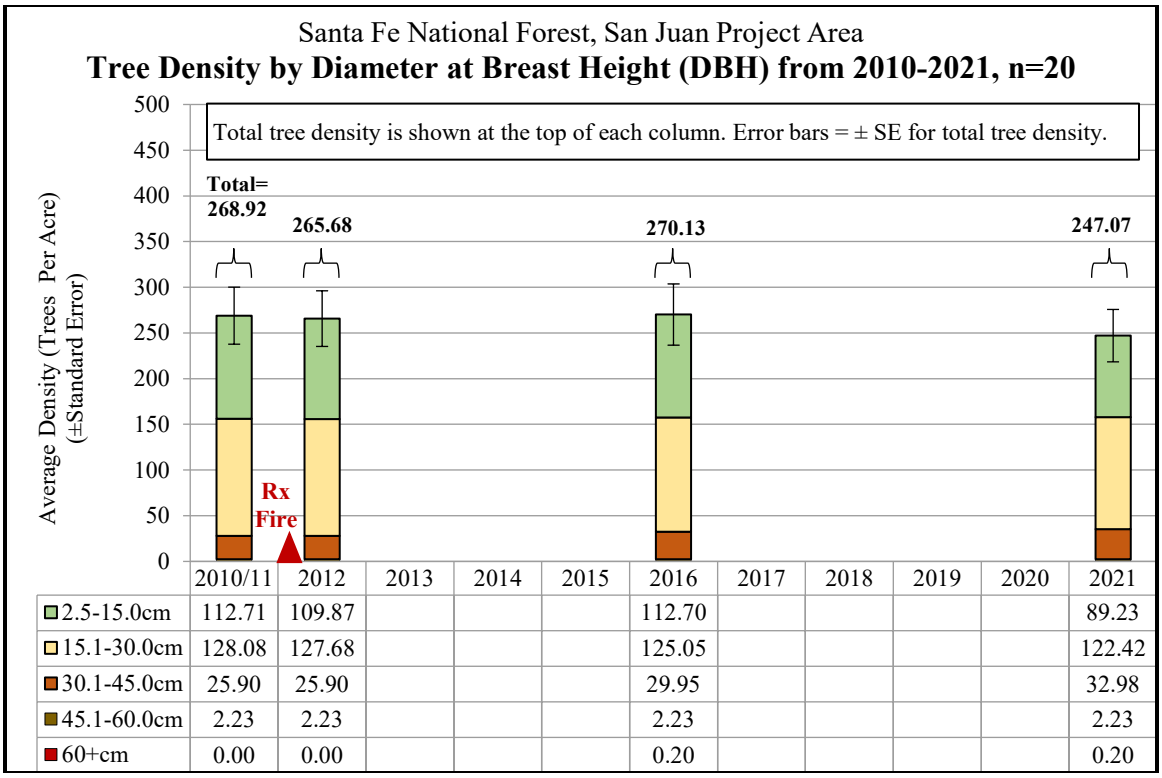



Figure 4.2.75. San Juan Mesa live tree crown base height results.

Average live tree crown base height was recorded at 13.78 ft. in 2010/2011. In 2016, it was recorded at 16.55 ft., a 20.10 percent increase from 2010/2011, prior to the broadcast prescribed fire (2012). Live tree crown base height was recorded at 17.75 in 2021.




| | 2010/11 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 2.5-15.0cm | 112.71 | 109.87 | | | | 112.70 | | | | | 89.23 |
| 15.1-30.0cm | 128.08 | 127.68 | | | | 125.05 | | | | | 122.42 |
| 30.1-45.0cm | 25.90 | 25.90 | | | | 29.95 | | | | | 32.98 |
| 45.1-60.0cm | 2.23 | 2.23 | | | | 2.23 | | | | | 2.23 |
| 60+cm | 0.00 | 0.00 | | | | 0.20 | | | | | 0.20 |

Figure 4.2.76. San Juan Mesa tree density and DBH results.

Average tree density in all size classes appears to remain fairly consistent from 2010 to 2016. In 2021, there is a noticeable reduction in the 2.5-15.0 cm size class, from 112.70 trees per acre in 2016 to 89.23 in 2021. A small reduction in the 15.1-30.0 cm size class in 2021 and increase in the 30.1-45.0 cm size class can be explained by tree growth into the 30.1-45.0 cm size class. Tree density in the 45.1-60.0 cm size class is low and remains consistent at 2.23 trees per acre throughout the monitoring period.

Photo points established prior to treatments allowed further understanding of the prescribed fire effects. An example photo set is shown in Figure 4.2.77. Note the successful consumption of dead and down woody fuels, and a substantial reduction of needle-cast litter and duff layer. However, as this was the first surface fire in decades, the heat of the fire killed the lower branch needles, which fell during the next 2 years; hence, the apparently heavy needle layer on the ground in the 2016 photo. A second planned fire will likely remove much of this needle layer and should open up the mineral soil to increased grass and forb colonization. This issue has been observed at numerous treatment sites in the Santa Fe National Forest and Valles Caldera National Preserve.




Figure 4.2.77. Top left. Pre-treatment Fire Ecology plot photo 2010/2011. Top right: Post-treatment photo 2012. Bottom left: Post-treatment photo 2016; Bottom right: Post-treatment photo 2021.

# Chapter 5. Watershed Restoration of Wetlands and Streams

**Collaborators:** Steve Vrooman, *Keystone Restoration Ecology*; Reid Whittlesey, Karen Menetrey, Cecil Rich, Todd Lopez, *Rio Grande Return*; Barbara Johnson, *Rio Puerco Alliance*; Nina Wells, *Los Amigos de Valles Caldera*; Robert R. Parmenter, Scott Compton, David Pittenger, *Valles Caldera National Preserve, National Park Service.*

**Summary**. Stream and wetland restoration activities were conducted across the SWJM CFLRP project area by collaborating organizations and contractors using funding support acquired through grant proposals to state and federal agencies. In total for wetland restoration, 27 different types of hydrologic structures were constructed at 1,010 locations to raise water tables, spread water out into former wetland and fen environments, and slow water currents to re-establish wetland vegetation. For streambank stabilization, 191,725 willows, alders, cottonwoods and other riparian woody plant species were planted along streams, and nearly 1,000 Beaver Dam Analogs and Post-Assisted Log Structures were installed at 148 stream locations. Monitoring results found that riparian vegetation quickly responded to hydrologic treatments, expanding wetland sedge and rush communities into formerly dry meadows and grasslands. As of 2024, one stream has been removed from the New Mexico Environment Department's list of impaired streams – Jaramillo Creek's water temperatures have now dropped to recommended levels due to shading by willows planted along its banks. Beavers have begun to voluntarily re-colonize another stream (San Antonio Creek) with planning underway for beaver reintroductions to other streams in the project area.

## Introduction

Watershed restoration in the SWJM CFLRP project area consisted of four major approaches.

*First*, restoration of former wetlands was undertaken through the installation of various engineered structures comprised of natural nearby materials (soil, rock, sod) to alter hydrologic channels to return water to historic flowpaths and spread out water onto adjacent riparian meadows and raise water tables; these actions have led to rapid successional changes of dryland bunchgrass and meadow vegetation towards wetland vegetation dominated by sedges and rushes. Over 500 acres of wetland has been restored in the Valles Caldera alone in 6 separate projects across the Preserve.

*Second*, stream restoration was achieved by implementation of treatments to address major impairments and restore the channel to a historic form and function. Stream restoration designs were created by a careful survey of channel form with longitudinal and cross-sectional profiles. Over 40 miles of channels on the Valles Caldera National Preserve were treated with a large assortment of treatments. Restoration work was performed on Santa Rosa Creek, Sulfur Creek, San Antonio Creek, East Fork Jemez River, Jaramillo Creek, and La Jara Creek.

*Third*, streambanks that had been incised and eroded during historic livestock grazing operations were stabilized with the establishment of woody plant species, principally willows, which allowed riparian soils to be held in place by the woody plant roots and the streams to be shaded by the overhanging willow branches. The presence of the restored woody riparian vegetation also

improved the stream habitat for beaver re-colonization – beavers had been extirpated during the latter 20th- century by the private landowners at the time.

*Fourth*, to assist beaver re-introduction, special structures known as "Beaver Dam Analogs" (BDAs), were constructed across streams to create initial pond habitat. BDAs would provide areas with deeper water to enhance survival of re-introduced beavers via escape refugia from predators.

As part of the watershed restoration program, a wetland vegetation monitoring plan was undertaken, using permanent plant transects cross through riparian areas and repeat photographs to assess vegetation changes following restoration actions; in addition, a water quality monitoring system using automated instruments (Sondes) was established, to assess long-term changes in stream water temperatures, dissolved oxygen, pH, conductivity and turbidity.

This Chapter is organized into the following sections:

Section 5.1. Wetland and Stream Restoration
Section 5.2. Riparian woody plant re-establishment
Section 5.3. Beaver Dam Analog installations
Section 5.4. Stream water quality monitoring

Following this Chapter, two additional Chapters related to aquatic ecosystems provide details on the monitoring programs for stream fisheries (Chapter 6) and stream aquatic macroinvertebrates (Chapter 7).

Funding for the watershed restoration activities was provided by outside grants from state and federal agencies as part of the SWMF CFLRP Collaborative cost-matching for the project. Table 5.1 lists the salient details for these grants.

Table 5.1. List of **28 fund sources** dedicated to watershed restoration in the SWJM CFLRP. Total funding during 2010-2024 was **$5,213,590**. *Abbreviations:* NMED = New Mexico Environment Department; TNC = The Nature Conservancy; NMDGF = New Mexico Department of Game and Fish; WWF = World Wildlife Fund; NFWF = National Fish and Wildlife Foundation; USFS = US Forest Service; NMED RERI = New Mexico Environment Department, River Ecosystem Restoration Initiative; NMED 319 = Environmental Protection Agency 319 Fund; NFF = National Forest Foundation.

| Funder | Grant Title | Recipient Organization | Fund Amount | Years | Names of leaders |
|---|---|---|---|---|---|
| NMED | Restoring La Jara Creek from Damage from the Thompson Ridge Fire | Los Amigos de Valles Caldera | $ 160,000 | 2017-2019 | Steve Vrooman, Barbara Johnson |
| Wildlife Conservation Society | Restoration of Slope Wetlands from Wildfire | Los Amigos de Valles Caldera | $ 170,000 | 2016-2018 | Steve Vrooman, Barbara Johnson, Nina Wells |
| NMED | Continuing Innovative Restoration of Historic Wetlands along Sulfur Creek | Los Amigos de Valles Caldera | $ 167,000 | 2012-2017 | Steve Vrooman, Barbara Johnson |

| | | | | | |
|---|---|---|---|---|---|
| NMED | Restoring Hydrologic Functioning to the Rito de los Indios | Los Amigos de Valles Caldera | $ 172,000 | 2015-2018 | Steve Vrooman, Barbara Johnson |
| Secure Rural Schools, SRS USFS | Restoring Roads and Streams in the San Antonio Watershed Damaged by the Las Conchas Fire | Los Amigos de Valles Caldera | $ 49,884 | 2014-2018 | Steve Vrooman, Barbara Johnson |
| NMED | Innovative Restoration of Historic Wetlands along Sulfur Creek | Los Amigos de Valles Caldera | $ 196,203 | 2012-2017 | Steve Vrooman, Barbara Johnson |
| NMED | Continuing Restoration Work on San Antonio Creek | Los Amigos de Valles Caldera | $ 167,498 | 2009-2012 | Steve Vrooman, Barbara Johnson |
| NMED | Reducing Temperature and Turbidity on San Antonio Creek by Restoring Slope Wetlands on Six Tributaries | Los Amigos de Valles Caldera | $ 165,400 | 2011-2014 | Steve Vrooman, Nina Wells, Barbara Johnson |
| NMED | East Fork Jemez River Innovative Wetland Restoration Project (East Fork Headwaters) | Rio Puerco Alliance | $ 191,490 | 2018-2021 | Steve Vrooman, Barbara Johnson |
| NMED | Restoring Jaramillo Creek | Los Amigos de Valles Caldera | $ 171,897 | 2011-2014 | Steve Vrooman, Barbara Johnson, Van Clothier |
| NMED | Lower Jaramillo Creek and Wetland Restoration Project | Keystone Restoration Ecology | $ 227,493 | 2020-2023 | Steve Vrooman, Keystone Restoration Ecology |
| NMED | Restoring Wetlands & Wet Meadows on VCNP | Los Amigos de Valles Caldera | $ 143,130 | 2007-2011 | Steve Vrooman, Barbara Johnson |
| NMED | Restoring San Antonio Creek | Los Amigos de Valles Caldera | $ 116,000 | 2009-2011 | Steve Vrooman, Barbara Johnson |
| NMED | San Antonio Creek Headwaters and Erosion Control Project | Rio Puerco Alliance | $ 259,214 | 2022-2025 | Steve Vrooman, Barbara Johnson |
| TNC, NMDGF | Redondo Creek Restoration Project | Rio Grande Return | $ 124,824 | 2021-2022 | Reid Whittlesey, Todd Lopez |
| WWF | Rio Cebolla Wetland and Watershed Natural Process Based Restoration Initiative | Rio Grande Return | $ 197,055 | 2023-2026 | Reid Whittlesey, Todd Lopez |
| NFWF | Rio Cebolla Wetland and Watershed Natural Process Based Restoration Initiative | Rio Grande Return | $ 189,481 | 2022-2024 | Reid Whittlesey, Todd Lopez |
| USFS | Upper Rio Cebolla Riparian Restoration | Rio Grande Return | $ 500,000 | 2024-2029 | Reid Whittlesey, Todd Lopez |

| | | | | | |
|---|---|---|---|---|---|
| NMED RERI | Middle Jaramillo Water Quality Improvement and Restoration Treatment | Wildearth Guardians | $ 139,750 | 2011-2014 | Jim Matison |
| NMED 319 | Upper Jaramillo | Wildearth Guardians | $ 223,130 | 2011-2015 | Jim Matison |
| NMED RERI | Valles Caldera Riparian Restoration and Beaver Habitat Restoration Project | Wildearth Guardians | $ 137,700 | 2010-2013 | Jim Matison |
| NFWF | Restoring Beaver Habitat in the Valles Caldera National Preserve | Rio Grande Return | $ 287,693 | 2024-2028 | Reid Whittlesey, Todd Lopez |
| NFF | Upper Rio Guadalupe and San Antonio Creek Watershed Restoration Initiative | Rio Grande Return | $ 120,000 | 2023-2028 | Reid Whittlesey, Todd Lopez |
| NMDGF | San Antonio Creek Fish Habitat Stamp Project | Rio Grande Return | $ 173,925 | 2022 | Reid Whittlesey |
| NFF | San Antonio Creek Restoration Project | Rio Grande Return | $ 64,706 | 2021 | Reid Whittlesey |
| USFS | Upper San Antonio Canyon Water Quality Improvement Project | Rio Grande Return | $ 350,000 | 2016-2020 | Jim Matison, Reid Whittlesey |
| NMED River Stewards | Upper San Antonio Canyon Water Quality Improvement Project | WildEarth Guardians | $ 137,100 | 2017-2019 | Jim Matison |
| NMED River Stewards | San Antonio Creek Riparian and beaver Habitat Restoration Project | Rio Grande Return | $ 211,377 | 2023-2025 | Reid Whittlesey |

## Section 5.1. Wetland and Stream Restoration.

When Congress purchased the Valles Caldera National Preserve in 2000, much of the potential wetland areas of the Preserve were dry due to significant downcutting of streams from overgrazing by sheep and cattle during the 19th and 20th centuries (Figure 5.1.1). Efforts by collaborating organizations and agencies were aimed at restoring hydrologic functioning of the watersheds, raising water tables, slowing stream discharge rates, and expanding wetlands and fens back to proper functioning condition. During the SWJM CFLRP project period (2010-2024), a total of 1,010 restoration structures were installed across all perennial watersheds in the Preserve (Figure 5.1.2). This restoration effort entailed the installation of 27 different structure types (Table 5.1.1) that were custom-engineered for each location in the watershed (see Zeedyk and Clothier 2009, Zeedyk et al. 2014, Zeedyk and Vrooman 2017). Several handbooks on wetland restoration were written on the techniques used and created for restoration on the Valles Caldera for the NMED Wetlands Program and the USA EPA. These can be found here: https://www.env.nm.gov/surface-water-quality/wetlands-technical-guides/

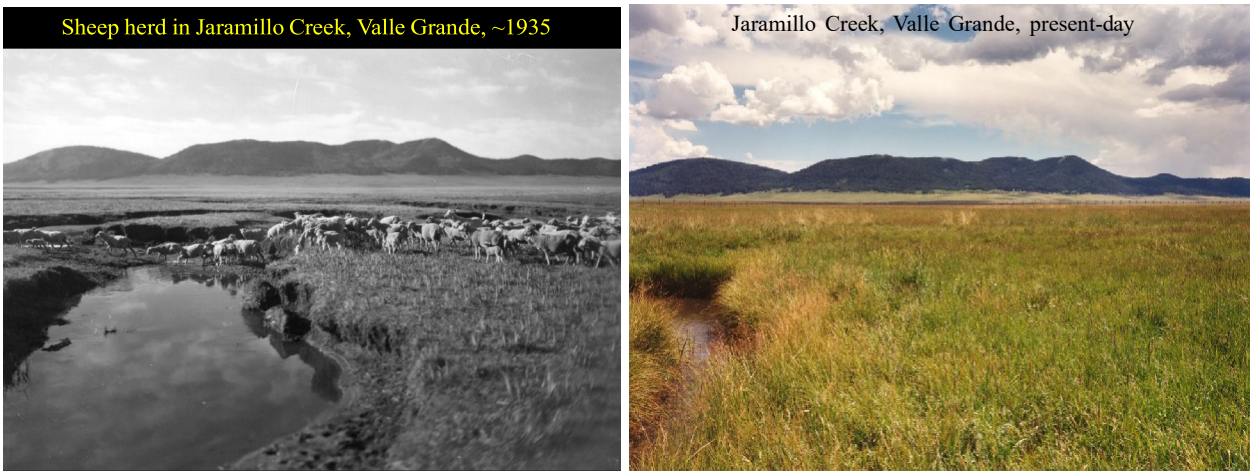



Figure 5.1.1. Repeat photographs of Jaramillo Creek in the Valle Grande of Valles Caldera National Preserve, showing severe erosion and vegetation impacts from early 20th-century sheep grazing. After removal of high-density livestock grazing, streams started to recover; restoration activities during 2010-2024 have further improved hydrologic functioning of area watersheds. Photo credits: Left: T. Harmon Parkhurst. Photo from the Museum of New Mexico, Image # 51457. Right: Steve Tharnstrom and John Hogan; photo from USGS Jemez Mountains Field Station, Image # JMFS 306 BAC retake color MNM51457.

The overall restoration objectives for watershed wetland restoration were to (1) increase water storage in the upper watersheds of the SWJM CFLRP project area, so as to create sustainable water supplies at high elevation for supplying continuous summer stream-flows downstream to water users (farmers, ranchers, residents, tribes), (2) improve wildlife and fisheries habitats, and (3) provide wetland and stream riparian fire breaks to landscape movements of wildfires.

*Wetland Creation and Restoration*

A large number of treatments were applied or even created to restore wetlands on the Valles Caldera National Preserve. Over time, treatments were refined to be smaller and more numerous, and were built primarily out of native grass sod and soil with rubber tracked machinery. Many

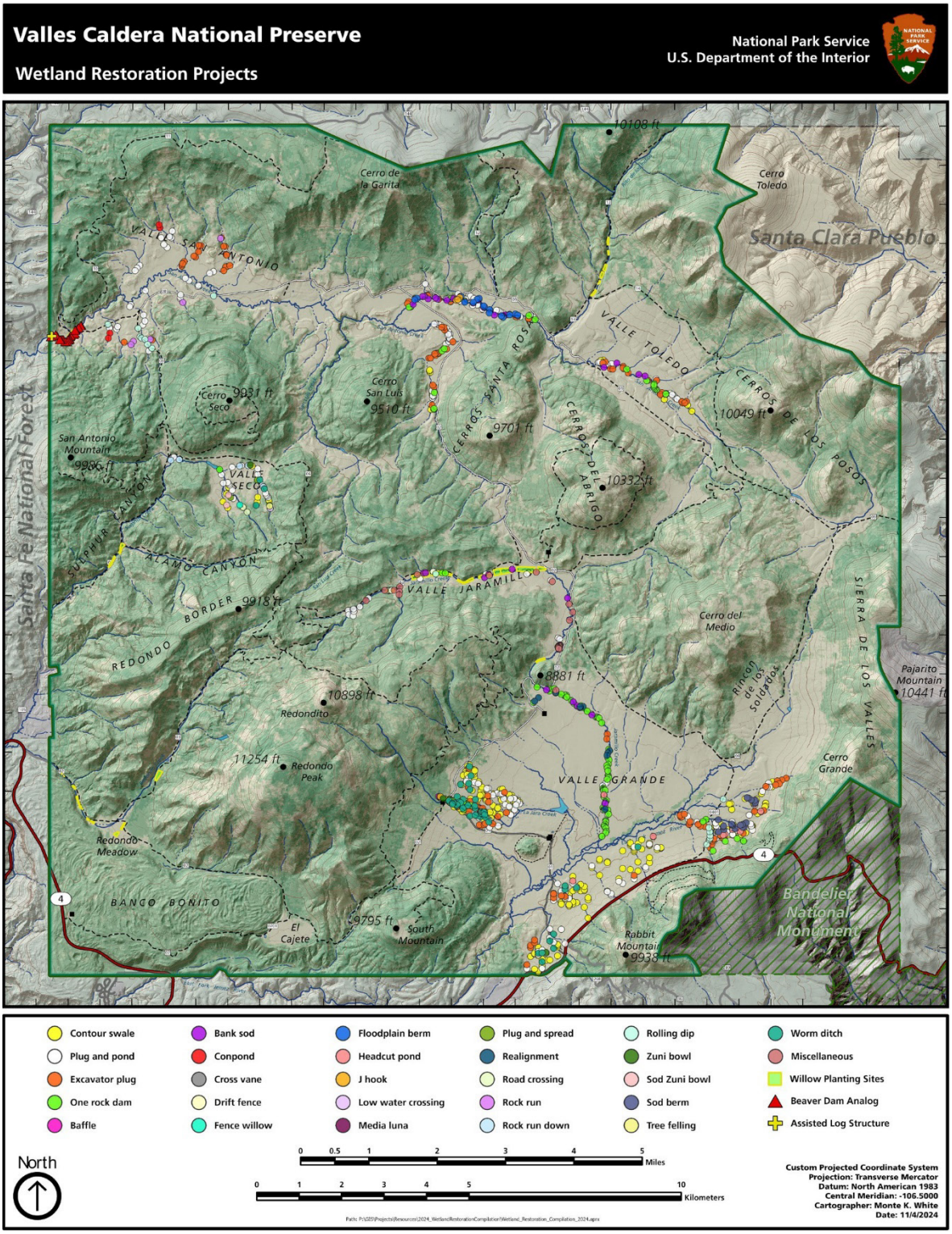



Figure 5.1.2. Map showing stream and wetland projects, many treated by hand with volunteers from the Albuquerque Wildlife Federation, NM Volunteers for the Outdoors and Los Amigos de Valles Caldera.

Over 525 acres of wetlands were restored as measured by wetland delineation monitoring prior and post implementation. Many earlier projects were not monitored for wetland creation and restoration, so the actual total of wetland restoration may be even higher.

*Example: La Jara Creek and Wetland Restoration*

The project on La Jara Creek was performed from 2017-2019 and involved the implementation of over 210 small restoration structures using native sod and soil. This project was monitored prior to restoration (Figure 5.1.3) and used wetland mapping from the NM Natural Heritage program for the over 1 square mile wetland. Post monitoring was performed in 2019 and 2020, showing a large increase in wetland acreage due to the restoration (Figure 5.1.4). A unique feature to the wetland and stream restoration work is the use of both LIDAR topography for landscape analysis as well as 1935 aerial photos from UNM RGIS. Many of the old photos show larger areas of wetland than at present, indicating that erosion damage was still occurring through the 20th century.

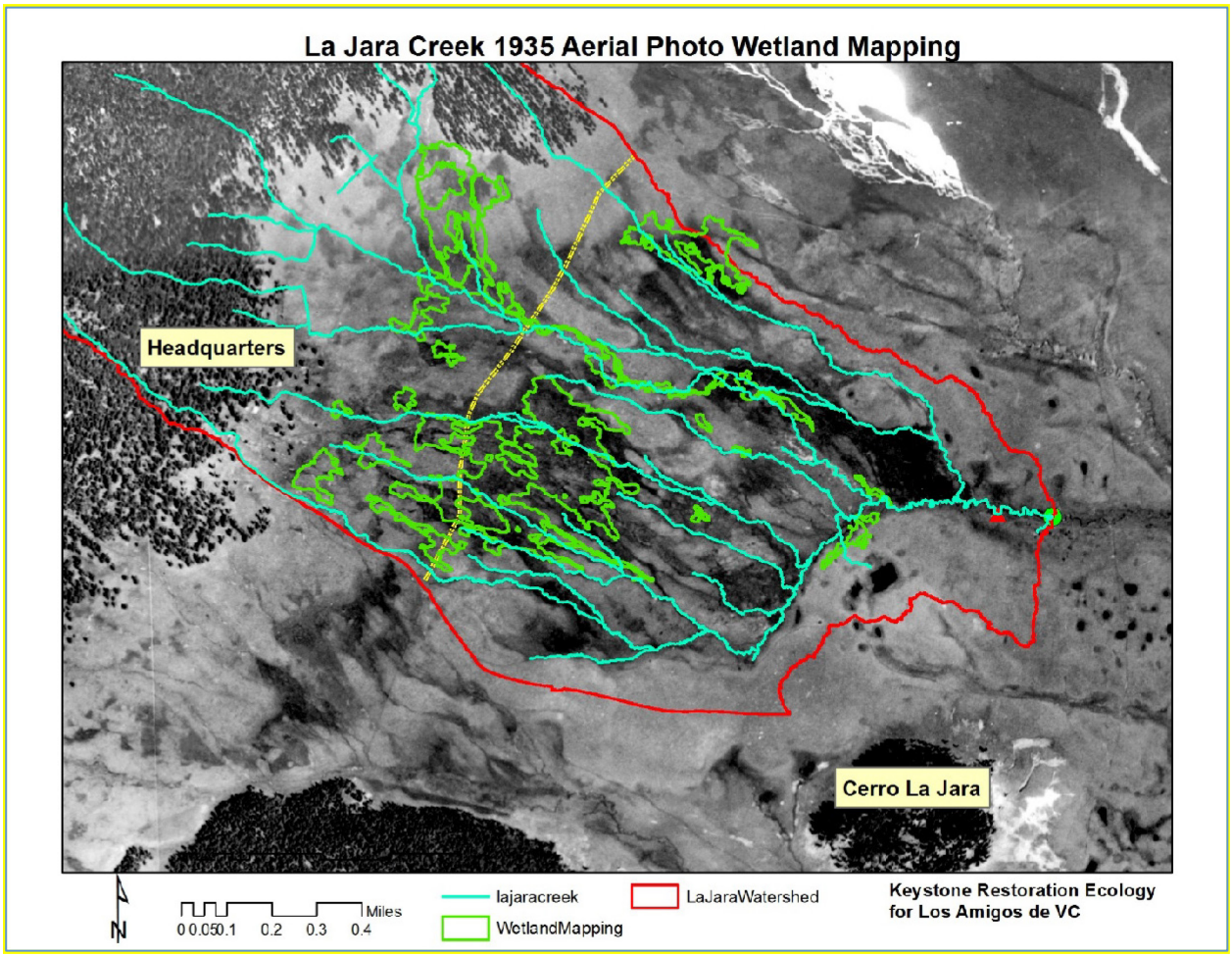



Figure 5.1.3. Map of La Jara Creek on Valles Caldera National Preserve. Green shows wetland mapped by NM Natural Heritage Program; dark areas are wetlands in 1935 that were not wetland in 1997 due to erosion.

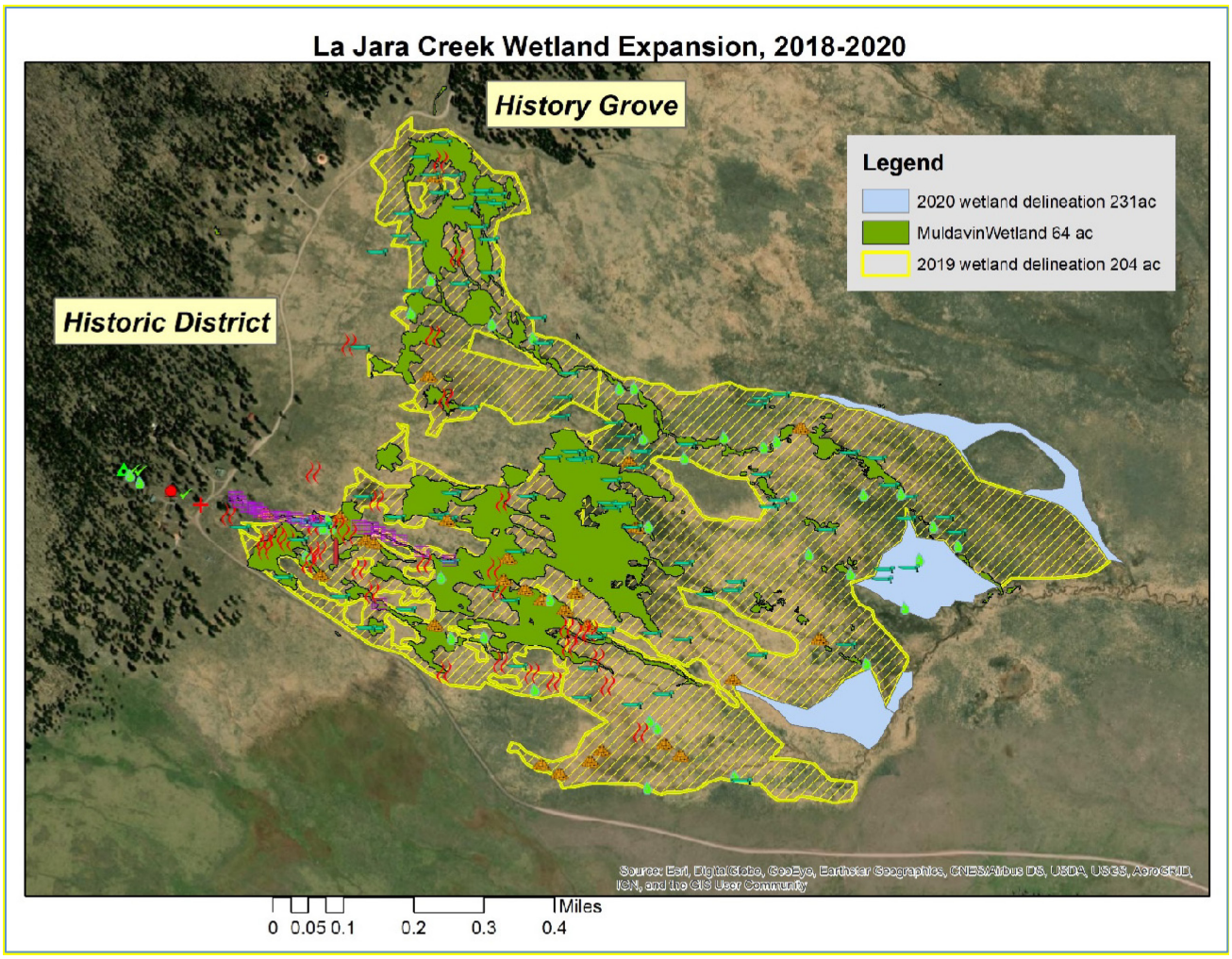



Figure 5.1.4. Wetland areas of La Jara Creek were mapped in 2018 with the Valles Caldera NPS vegetation crew over a smaller area and was found to match closely to a digitized layer created by NMNHP. Ground based monitoring was performed in 2019 and 2020. In 2020, 167 additional acres of wetland were delineated, showing the effects of restoration.

*Stream restoration with Induced Meandering and Natural Channel Design*

Stream channels across the project area have been impacted by historic overgrazing and lack of fire; many channels have degraded and lost access to their historic floodplain from the impacts of over 80,000 sheep in the early 20th century and thousands of cattle beginning after World War II. Many stream channels are shallow, choked with sediment and over-wide with little shade, contributing to these streams being listed for temperature and turbidity by the New Mexico Environment Department Surface Water Quality Bureau.

From the initial disturbance from livestock at the beginning of the 20th century, stream channels have adjusted and re-meandered over time, but have incised in the landscape at a lower channel elevation. This has lowered the water table and dried out the adjacent wetlands and riparian areas along the streams, leading to a huge loss in wetland vegetation, water storage and wetland function, in addition to the loss in quality habitat and low water temperatures in the streams themselves.

*Fire Effects on Channel Function:* The degraded channels were further affected by post-fire runoff events from the Las Conchas fire in 2011 and the Thompson Ridge Fire in 2013. The post-fire flood events on the fire scorched landscape were over 100 to even 1000 times the usual flood magnitude. These floods poured into channels that were suffering from long-term degradation from overgrazing, road building and other impacts.

The effect of the fires in the upper most tributaries was severe, with huge amounts of gullying contributing sediment to the channels downhill. As the stream channels flattened, the sediment added to the streams caused additional bank erosion and widening of the stream channel, raising temperatures and degrading habitat quality. Treatments were applied as soon as 1 year after the Las Conchas fire, and the source of sediment was utilized in many cases to fill in historic gullies with native soil by diverting flood flows to areas where they could be captured.

Many of the wetlands on the Preserve benefited from the sources of ash and soil from uphill, any many lowland areas filled in naturally with the large flood events. This provided an example for further restoration efforts to capture and utilize this material to restore degraded Creeks. Santa Rosa Creek, for example, a tributary of San Antonio Creek along the VC02 Road, gained about 4 feet in elevation from sediment capture after a series of workshops built one-rock dams and has returned to a small perennial creek from its previously gullied, overgrazed condition.

*Treatment Types*

Structures created from native wetland sod and rock materials sourced from Santa Clara Pueblo or Los Alamos County were constructed to create deeper pools for habitat for fish and invertebrates. Structures such as one-rock dams, Cross Vanes, and J-Hook Vanes (see below for structure descriptions) reconnected the channel with its floodplain and deepened pool habitats. The raising of the water table fosters the growth of streambank riparian vegetation, which in turn shades the channel and reduces temperature.

Where channels were gullied by old roads and trails, streams were realigned to the historic channel location, which raised the water table and reconnected drained wetlands with floodwaters, restoring many acres of degraded wetlands. Restoration of these “channel cut-offs” proved to be one of the easiest and most effective technique for spreading water across vast areas and was very effective in Lower Jaramillo Creek.

Lastly, long term grazing impacts had created thousands of feet of eroding channel banks, some 4 feet high. These were a major source of sediment to the channel; and as sediment is deposited, more erosion occurs on the downstream bank, and the process continues. Many banks on San Antonio Creek and Jaramillo Creek were treated by the creation of floodplain on the outside of meander bends using native wetland sod. This treatment narrows the channel and provides shade, as well as eliminating the major source of fine sediment.

Table 5.1.1. List of 27 wetland and stream restoration project structure types and numbers of structures installed during restoration wetlands in the CFLRP project area. Descriptions of project types are provided in the text below.

| Structure type | Number installed |
|---|---|
| Baffle | 1 |
| Bank sod | 45 |
| Contour pond | 4 |
| Contour swale | 261 |
| Cross vane | 2 |
| Drift fence | 1 |
| Excavator plug | 131 |
| Fence willow | 1 |
| Floodplain berm | 17 |
| Headcut pond | 11 |
| J hook | 6 |
| Low water crossing | 1 |
| Media luna | 1 |
| Miscellaneous | 26 |
| One rock dam | 118 |
| Plug and pond | 269 |
| Plug and spread | 2 |
| Realignment | 10 |
| Road crossing | 1 |
| Rock run down | 10 |
| Rolling dip | 16 |
| Sod berm | 20 |
| Sod Zuni bowl | 4 |
| Tree felling | 1 |
| Worm ditch | 48 |
| Zuni bowl | 3 |
| **Grand Total** | **1,010** |

Detailed descriptions of the various wetland and stream restoration structures are as follows:

**Baffle**: A baffle is a structure built to mimic the upper half of a floodplain point bar and is shaped like a right triangle. It is constructed out of juniper posts hammered into the ground with willows stems woven between them. Baffles can also be constructed out of boulders if materials are accessible and the project is using machinery. Water flowing around a bend will flow around the baffle and create a meander bend, and pool, opposite the baffle. In flood events, the baffle will be submerged under water and will capture sediment such as cobble, gravel, sand and clay. The lower half of the floodplain below the baffle will be naturally formed by deposited sediment in the eddy below the baffle structure.

**Bank Sod:** Use of Bank Sod is indicated where a shelf is excavated along the upper streambank on the outside of an eroding bend and the excavated sod is transplanted at the toe of the bank up to bank full elevation (Figure 5.1.5). This creates a floodplain bench at bankfull, increasing the floodplain width and decreasing the erosion pressure against the bank. A small backhoe or a skid steer loader is used to cut, transport, and carefully place the sod tiles. The vegetation quickly roots, helping to stabilize the bank soil, decreasing erosion and subsequent turbidity in the stream. This method was successfully used last year on San Antonio creek with the Valles Caldera Preserve (Figure 5.1.6).

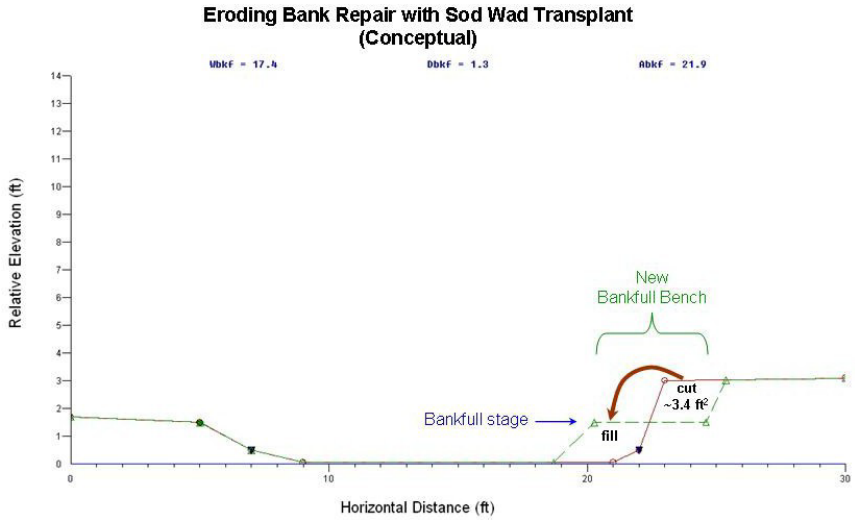



Figure 5.1.5. Typical Bank Sod erosion control cross-section



Figure 5.1.6. Photo of Bank Sod erosion control on San Antonio Creek

**Contour Pond:** A contour pond is an early description of a contour swale. It means a contour swale built entirely on contour, with water flowing out of the swale along the downstream edge. This is unlike most contour swales that are built slightly off contour, and water leaves the downstream end of the swale.

**Contour Swale:** A Contour Swale is placed slightly off contour and is dug by an excavator bucket (Figure 5.1.7). The swale is dug 8-12 inches deep (as deep as the sod) and the sod is used to line the downstream edge of the swale, creating a berm. Contour swales are placed slightly off contour and used to divert water away from headcuts and gullies and back onto the wetland surface.



Figure 5.1.7. Contour Swale diverting water towards viewer around shallow headcut in background, water flowing left to right, downhill towards East Fork of Jemez River.

**Cross Vane:** The Cross Vane structure was invented by Dave Rosgen. It is an inverted "V" structure with the point of the V pointed upstream. Each Vane Arm begins a the Bankfull Floodplain elevation and points upstream at a 5% angle. The "invert" of the structure is placed at the desired channel elevation, either the original elevation or slightly higher if the channel is degraded and a gully. A Cross Vane creates a pool below the structure and the Vane arms provide bank protection, as water rolls off both Vane Arms into the pool in the middle of the stream or river (Figure 5.1.8).



Figure 5.1.8. Cross Vane on San Antonio Creek at VC02 road crossing

**Drift Fence**: A drift fence is a short length of fence that "T"s off of a fenceline that suffers from cattle or elk trailing. The drift fence causes the ungulate to walk around the fence, which can carry water out of the fenceline trail into a less erosive location.

**Excavator Plug:** An excavator plug is a small plug constructed by excavator in order to divert water off of a trail, small gully, or other shallow feature (Figure 5.1.9). This plug holds little water, and spills water at a 2-4% slope downhill back onto the wetland surface nearby.



Figure 5.1.9. Plug to left of photo takes water right and re-irrigates about 2 acres of sloped wetland.

**Fence Willow**: Old groves of Bebb's willow were being grazed heavily by elk and cattle. A small exclosure was constructed in Tributary 2 of Valle Seco (West Side, uphill) to protect the area from over grazing and allow for natural recruitment to occur.

**Floodplain Berm**: This structure is built in conjunction with a Bank Sod structure. We learned that previous Bank Sod structures on Lower San Antonio (Six Tribs, below the lower road crossing) had been heavily damaged by a snowmelt flood event in winter 2022. To address this, we widened the floodplain and removed a large amount of soil to do this. The widened floodplain was planted with wetland sod at Bankfull Elevation. The excess soil was removed and placed on the terrace nearby to capture sheet flow from snowmelt and prevent this water from being captured in an abandoned stream channel and causing erosion as it poured back into San Antonio Creek. This creek appears to have dropped about 18 inches in 1935 (GIS analysis of historic aerials) and former channels collect snowmelt and cause headcutting as they pour into the existing channel.

**Headcut Pond Structure:** A plug and pond structure built to treat a headcut is called a "Headcut Pond". A Headcut Pond is built above a headcut, on stable ground, and the "plug" is built around and above the headcut to prevent water flow over the headcut. Figure 5.1.10 below shows headcut pond at "Nina's Spring" in the Valle San Antonio, where three separate headcuts at the end of a wetland were circumvented by the installation of one Headcut Pond above them.



Figure 5.1.10. Headcut Pond in East Fork Headwaters, water flowing right to left. Large Headcut to right is protected and irrigated from seepage from pond.

**J-Hook Vane**: This structure is two-thirds of a Cross Vane and is used much more commonly. A J-hook Vane is constructed at an eroding bend to protect against excessive meander scrolling. The Vane Arm points upstream to the invert of the structure and can be built of Rock or Logs. The invert is a cluster of boulders which creates a pool at the upper end of the Meander bend by causing scour. This also moves the current of the Creek into the pool and away from the bank, preventing further erosion.

**Low Water Crossing**: An effective way to cross a creek is to create a low water crossing at the riffle section of the creek, between the meander bends at the cross-over (in the middle of the "S" of a meander). This portion of a creek or river is where cobbles or gravel naturally accumulates and is a stable crossing point. This allows for infrequent crossing to occur, maintenance of this is cheap and easy and no culverts or box culverts are needed. Often a Cross Vane is used at the bottom of this structure to hold the grade of the crossing and prevent erosion downstream.

**Media Luna**: A Media Luna is a half-moon-shaped structure of 12 inch rocks used to spread water in the shape of an alluvial fan. Larger boulders can be used in areas of increased sheer stress. This structure is placed to spread water naturally across the landscape and induce sheet

flow. Where an alluvial fan is cut through by a gully, a Media Luna can repair the fan and allow for flow across the fan surface.

**Miscellaneous:** Various minor earthwork modifications of streambanks and wetlands to stabilize erosion and retain water.

**One Rock Dam:** A low-grade control structure built with a single layer of parallel rock rows on the bed of a channel (Figure 5.1.11). The structure prevents further down cutting of the channel, traps sediment behind the structure which helps rebuild the channel elevation and creates a moister environment favorable to colonization by native riparian vegetation. The one rock dam is a constructed riffle structure using small rocks, 12-24 inches in diameter. These constructed riffles are placed at the "cross-over" between two meander bends, and raise the grade of the channel to spread water across the floodplains on either side of the channel. Over time, these structures will raise the grade of the channel be capturing sediment and promoting the growth of wetland vegetation in the channel and on the floodplain. Follow-up maintenance and modification can involve the placement of another layer of rock on top to raise the grade of the channel another foot (Figure 5.1.11, left).

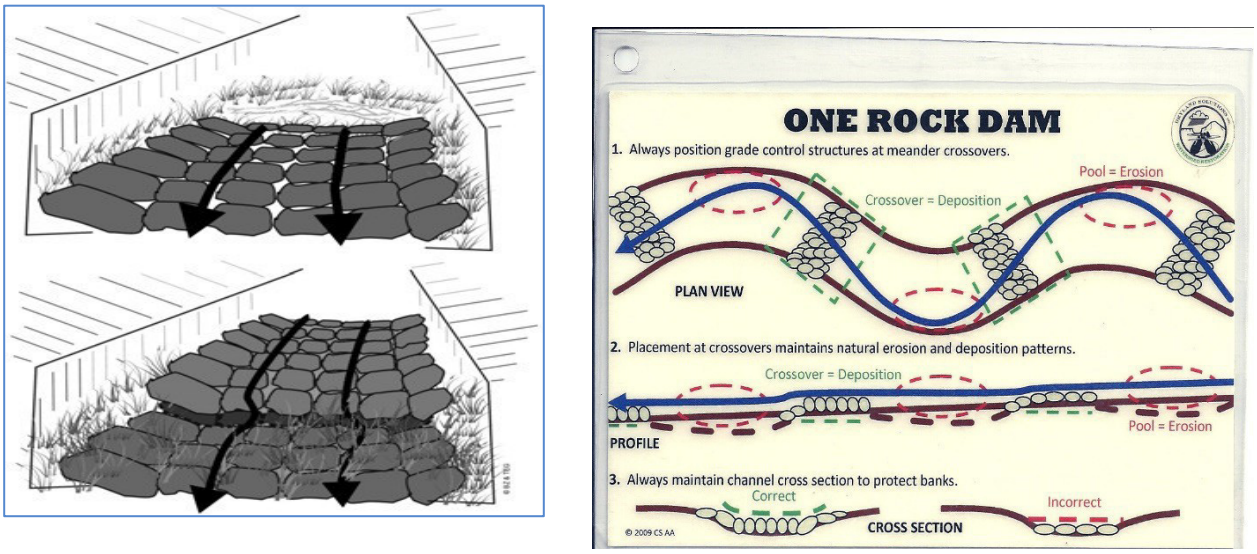



Figure 5.1.11. Left: One Rock Dams with arrows showing water flows; Right: Typical One Rock Dam plan view and cross-section

**Plug and Pond Structure:** A plug and pond structure is constructed in a gully to raise the water table to an abandoned stream channel or wetland surface (Figure 5.1.12). The sod is salvaged from the work area and placed aside for further use. Then, a pond is dug in the side of the gully to provide fill material to construct a plug. Once the plug is completed, it is faced with the salvaged sod so that it revegetates very quickly. Water fills the pond then spills out onto the former wetland or channel (Figure 5.1.12).



Figure 5.1.12. Plug and Pond structure, water spills to right looking downhill.

**Plug and Spread:** The plug and spread is constructed in dryland situations where a pond would rob water from the landscape. A gully is filled with soil and a dam or berm created to move water out of the gully back onto the landscape, usually used in wet meadows or wetland situations. Plug and Spreads are usually much larger than Plug and Ponds and created by bulldozer.

**Realignment**: A channel realignment was performed on Lower Jaramillo Creek to return the Creek to an abandoned channel in several locations. Channel cut-offs (avulsion) had occurred due to erosion, road building and culvert installation. This technique returned the Creek to an abandoned meander, which raised the channel grade and created at least 40 acres of re-wetted floodplain wetland downstream, as flood events were re-connected with an abandoned floodplain.

**Road Crossing** (see **Low Water Crossing**)

**Rock Run, Rock Rundown:** A Rock Rundown is a headcut repair structure used in a upland situation with little flow. The headcut is widened by 2X, and a gentle slope created with the soil. Rocks are placed on the slope, with the top rock at the elevation of the meadow above. This widened headcut will re-grass itself naturally over time, as the rocks act to prevent evaporation at the soil face of the headcut.

**Rolling Dip:** Rolling Dips are smooth, angled depressions constructed in a roadbed where the road grade reverses for a short distance and surface runoff is directed in the dip to the outside or inside of the road (Figure 5.1.13).



Figure 5.1.13. Rolling dip takes water to the right, away from viewer, and cars drive over without having to slow down.

**Sod Berm:** Many of the intermittent wetland tributaries in the headwaters of the East Fork Jemez River have degraded 2-4 feet below their historic elevation. They are still heavily grazed by elk and used for elk wallowing during the rutting season. Water is trapped in a narrow 1 foot wide and 6 inch-deep channel and wetland vegetation is limited to the sides of this “inner berm”. A sod berm structure is a contour swale placed in the bottom of these swales to promote meandering, spread water across the swale, capture sediment, and promote the growth of wetland vegetation (Figure 5.1.14).



Figure 5.1.14. Sod berm in bottom of the wide gully re-wets entire gully bottom

**Sod Zuni Bowl:** Many locations exist where a headcut is the lowest part of the landscape, and water cannot be diverted around the headcut to a former channel or sheetflow wetland surface by a Plug and Pond, Headcut Pond or Contour Swale. One treatment to prevent further head-cutting uphill is to build a Sod Zuni Bowl in the headcut (Figure 5.1.15). During periods of active flow, spash erosion is prevented and the area remains saturated. Wetland vegetation can be planted on both the headcut and the berm and kept saturated by water in the "bowl".



Figure 5.1.15. Sod Zuni Bowl at top of small gully,

**Tree felling:** This was used in Alamo Bog to address tree encroachment into the Bog, and as a way to capture sediment and raise the water table. Trees were felled at a 90-degree angle to the Bog (and flow), and this also was very effective in addressing trailing from trespass cattle and elk, acting as a drift fence.

**Worm Ditch:** A worm ditch is a headcut treatment structure constructed by laying out a string 2x as long as the "run" of the channel. A meandering channel 2x the present length is cut with shovels or and excavator to lead the water from upstream around the headcut and to a stable area below the headcut.

**Zuni Bowl:** Zuni Bowls are in-channel head cut repair structures composed of rock-lined step falls and plunge pools that prevent head cuts from continuing to migrate upstream (Figure 5.1.15). Zuni Bowls stabilize actively eroding head cuts by dissipating the energy of falling water at the head cut knick point. They create pool habitat and increase infiltration into the channel bed to promote the growth of native riparian vegetation in an area that was previously drying out. These structures can be built by hand or by machine, using 12- 36” diameter rock.



Figure 5.1.15. Zuni Bowl at headcut at top of Santa Rosa Creek

*Wetland Post-Restoration Monitoring*.

Following the installation of the restoration structures, monitoring crews assessed the response of wetland vegetation via plant measurement transects (placed perpendicular to stream channels) and repeat photographs. In all the monitoring sites, we observed a shift from wet meadow grasses (e.g., *Phleum pratense*) and forbs (e.g., *Achillea millefolium, Agoseris glauca,* etc.) to wetland obligate sedges (*Carex* spp.) and even some semiaquatic forbs like *Persicaria amphibia*. In areas that included pre-treatment seasonal flooding (from spring snowmelt), the vegetation shifted from facultative wetland sedges (e.g., *Carex praegracilis*) to more wetland obligate sedges such as *Carex aquatilis* and *Carex utriculata*. Vegetation responses were usually rapid, occurring within several years of structure construction (Figures 5.1.16, 5.1.17).

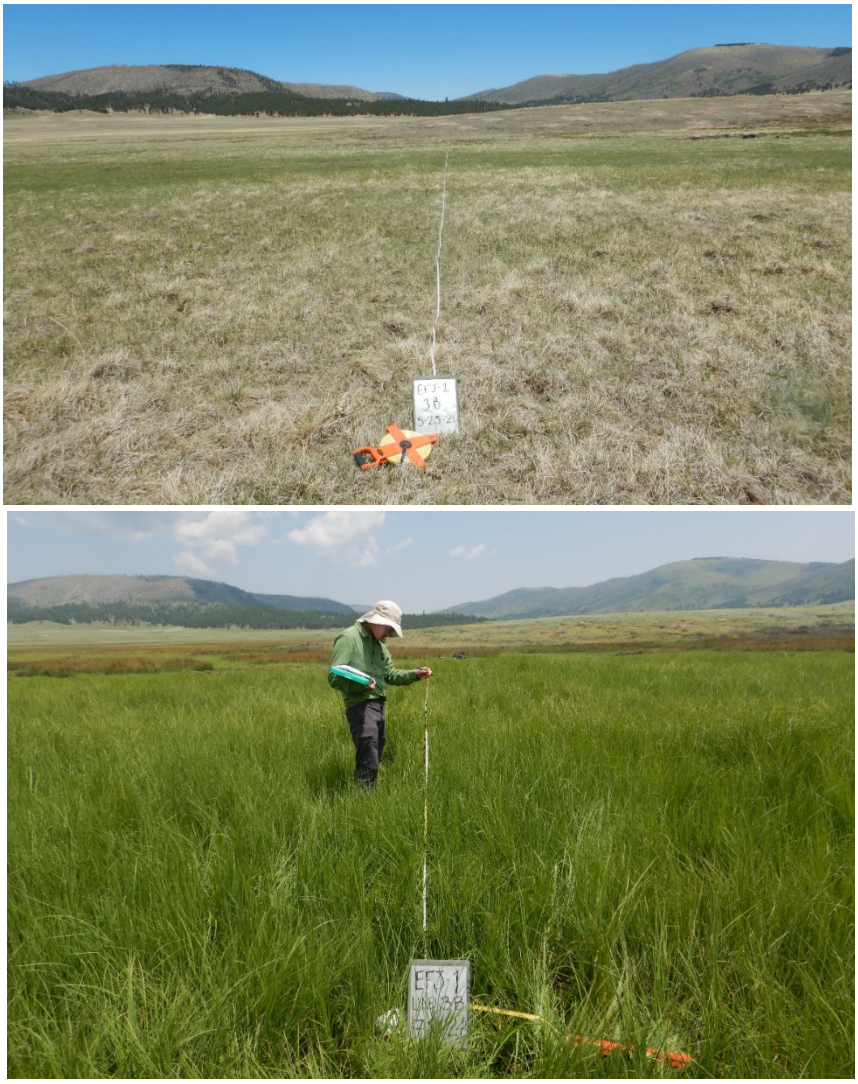


Figure 5.1.16. Repeat photos of wetland plant monitoring transect following restoration crossing the East Fork Jemez River, Valle Grande, Valles Caldera National Preserve. Top: May 2021. Bottom: July 2024. Note the large increases in wetland sedges, expanding from the original wetland in the 2021 photo, replacing upland bunchgrasses.



Figure 5.1.17. Repeat photographs of wetland monitoring transect crossing Jaramillo Creek (seen in middle of transect in distance). Top: 2021, prior to treatment. Bottom: 2024, 3 years post-restoration. Note the elk exclosure fence that was constructed to keep elk from disturbing the site during post-restoration recovery. 2021 vegetation composed of meadow species; 2024 vegetation dominated by sedges, with standing water from elevation of water table.

## Section 5.2. Riparian woody plant re-establishment

In 2009, during the preparation of the SWJM CFLRP proposal, nearly all the streams in the project area were impaired as designated by the New Mexico Environment Department (NMED), Surface Water Quality Bureau, with Total Maximum Daily Load (TMDL) exceedances for temperature and turbidity. These impairments were often due to a lack of woody riparian vegetation that exposed the streams to direct solar heating, and eroded, unstable stream banks that continually shed soil into the water columns. Previous populations of willows and alders had been destroyed by livestock in the 19th and 20th centuries, leaving only isolated individuals or clumps of riparian woody vegetation. In many locations, the solution was to re-establish woody plant species along the streambanks to provide shade and bank soil stability. These woody plants would also provide food and structural materials to beavers, which would enhance the development of riparian and wetland habitats along the streams (see Section 5.3 below).

To accomplish this task, two CFLRP non-profit collaborating organizations, the WildEarth Guardians and Rio Grande Return, successfully applied for grants to support the stream revegetation projects (Table 5.1 above). Field crews re-planted willows and other shrub species along streambanks, using mechanized drilling equipment to drill deep holes for willow cuttings and containerized shrubs and trees (Table 5.2.1, Figure 5.2.1). The crews also had to construct elk exclosure fences to prevent elk and trespass cattle from feeding on the newly-replanted woody plants. Over several years of post-planting recovery, the willows and other species grew and proliferated, forming dense thickets of shrubs along the streambanks. Such treatments were applied to streams on the Santa Fe National Forest (Rio San Antonio and lower Redondo Creek) and the Valles Caldera National Preserve (Jaramillo Creek, Rio San Antonio, Indios Creek, upper Redondo Creek, and Sulphur Creek). In total, 191,725 willows, shrubs and trees were planted along the streams of the CFLRP project area (Table 5.2.2).




Figure 5.2.1. Planting willow cuttings in drill holes along Redondo Creek, 2011.

Table 5.2.1. List of woody plant species used in restoring riparian woody vegetation along streams in the SWJM CFLRP project area.

| Common name | Species |
|---|---|
| Aspen | *Populus tremuloides* |
| Bebb's willow | *Salix bebbiana* |
| Narrowleaf cottonwood | *Populus angustifolia* |
| Narrowleaf willow, or coyote willow | *Salix exigua* |
| Snowberry | *Symphoricarpos oreophilus* |
| Strapleaf willow | *Salix ligulifolia* |
| Thinleaf Alder | *Alnus incana* |
| Wax Currant | *Ribes cereum* |
| Wood's rose | *Rosa woodsii* |

Table 5.2.2. Numbers of individual plants installed in restoring riparian woody vegetation along streams in the SWJM CFLRP project area.

| Species | No. planted |
|---|---|
| Willow | 186,500 |
| Cottonwood | 2,075 |
| Aspen | 1,750 |
| Alder | 1,300 |
| Snowberry | 50 |
| Wax currant | 50 |
| Total | 191,725 |

Monitoring of the planted vegetation revealed that survivorship varied depending on the location and the year in which the plants were installed. Survival was very good on the upper reaches of streams with slightly higher gradients, but the main factor influencing survival appeared to be drought conditions. Willows planted on Redondo Creek in 2011, during a very dry winter-spring period, experienced >90% mortality in some reaches. Similarly, the lower reaches of Jaramillo Creek saw slow establishment following drought years, with aboveground portions of plants dying during May and June, only to stump-sprout later in the year when the summer monsoons arrived. In spite of local mortality events, vegetation monitoring conducted in 2023 showed a tremendous increase across all planted streams in willow population, height and density as compared to monitoring that occurred a decade ago, shortly after planting. Redondo Creek, Jaramillo Creek and Rito de Los Indios clearly have sufficient healthy woody vegetation to support beaver. Sulphur Creek has small but healthy patches of willow. San Antonio Creek has some beaver habitat. Overall, willow populations have become well-established in all areas that were planted (Figures 5.2.2 and 5.2.3), providing extensive stands along miles of treated streams (Figure 5.2.4). Some volunteer beaver colonization into replanted reaches has taken place on San Antonio Creek, and beaver introductions in other creeks are now being planned for 2025 and future years.



Figure 5.2.2. Indios Creek riparian area before (left) and after (right) planting of willows and narrowleaf cottonwoods; note elk exclosure fences in the photo on right.



Figure 5.2.3. Jaramillo Creek willow populations.




Figure 5.2.4. GIS map of Jaramillo Creek showing stands of restored willow populations (light green); *Source:* Karen Menetrey, Rio Grande Return.

## Section 5.3. Beaver Dam Analog installations

As part of the stream restoration effort in the SWJM CFLRP stream restoration effort, the reintroduction of beavers was identified as a long-term priority to assist with stabilization of streambanks and enhancing wetland development and raising water tables. To assist with streambank stabilization and beaver reintroductions, the restoration plan included the installation of structures known as Beaver Dam Analogs (BDAs) and Post-Assisted Log Structures (PALS; Figure 5.3.1); see Wheaton et al. (2019) for details on these structures. Nearly a thousand BDAs and PALs were installed on a total of 148 sites along San Antonio Creek on both the Santa Fe National Forest and Valles Caldera National Preserve (Fig. 5.3.2) as well as Cebolla Creek on the SFNF (Figure 5.3.3).

Post-assisted log structures (PALS) and beaver dam analogues (BDAs) are hand-built structures. PALS mimic and promote the processes of wood accumulation whereas BDAs mimic and promote beaver dam activity. PALS and BDAs are permeable, temporary structures (lasting up to 10 years pending maintenance by hand crews or occupation by beaver), built using natural materials. BDAs differ from PALS in that BDAs create ponds and are built using a variety of fill materials, including logs, branches, gravel, and cobbles. PALS are built with only woody material, which tends to generally be larger diameter than the woody material used for BDAs (see Figures 5.3.4 and 5.3.5 for schematic construction plans).



Figure 5.3.1. Photos of Beaver Dam Analogs (BDAs) on San Antonio Creek, Santa Fe National Forest.

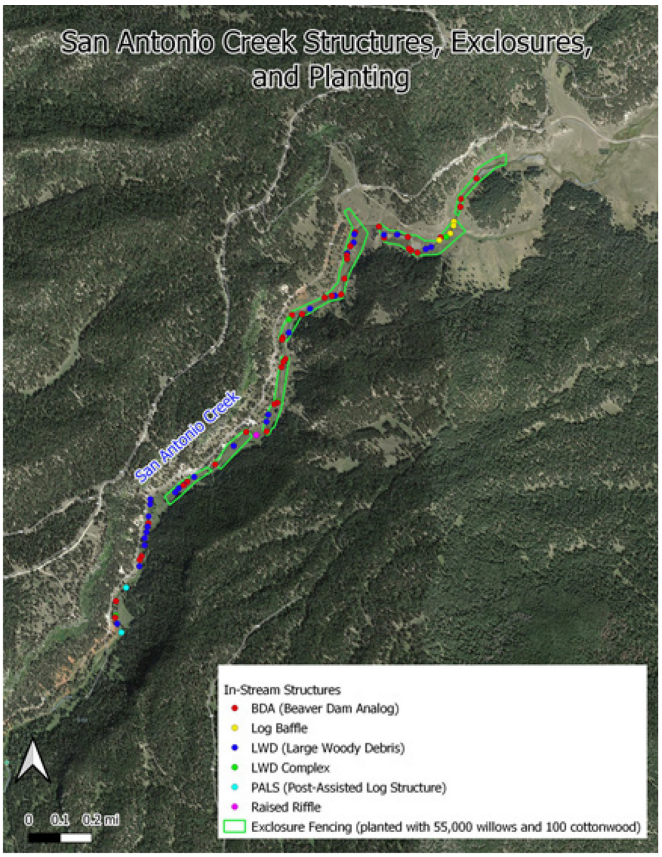



Figure 5.3.2. Map of the western reach of San Antonio Creek in the Santa Fe National Forest (lower left) and Valles Caldera National Preserve (upper right), showing locations of Beaver Dam Analog (BDA) and Post-Assisted Log Structures (PALS) installations on San Antonio Creek. Light green polygons show areas of willow plantings (see Section 5.2 above).

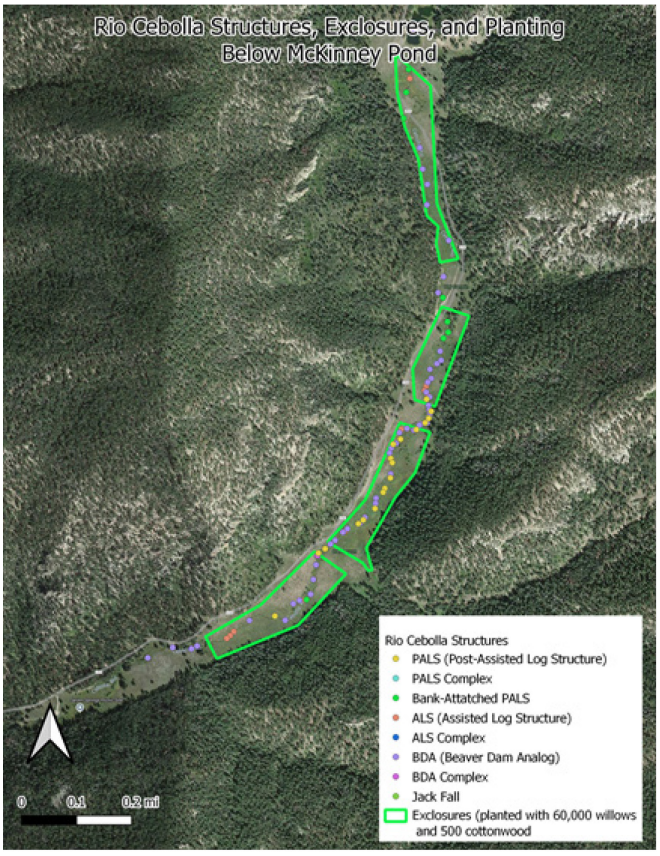



Figure 5.3.3. Map of Cebolla Creek in the Santa Fe National Forest, showing locations of Beaver Dam Analog (BDA) and Post-Assisted Log Structures (PALS) installations. Light green polygons show areas of willow plantings (see Section 5.2 above).

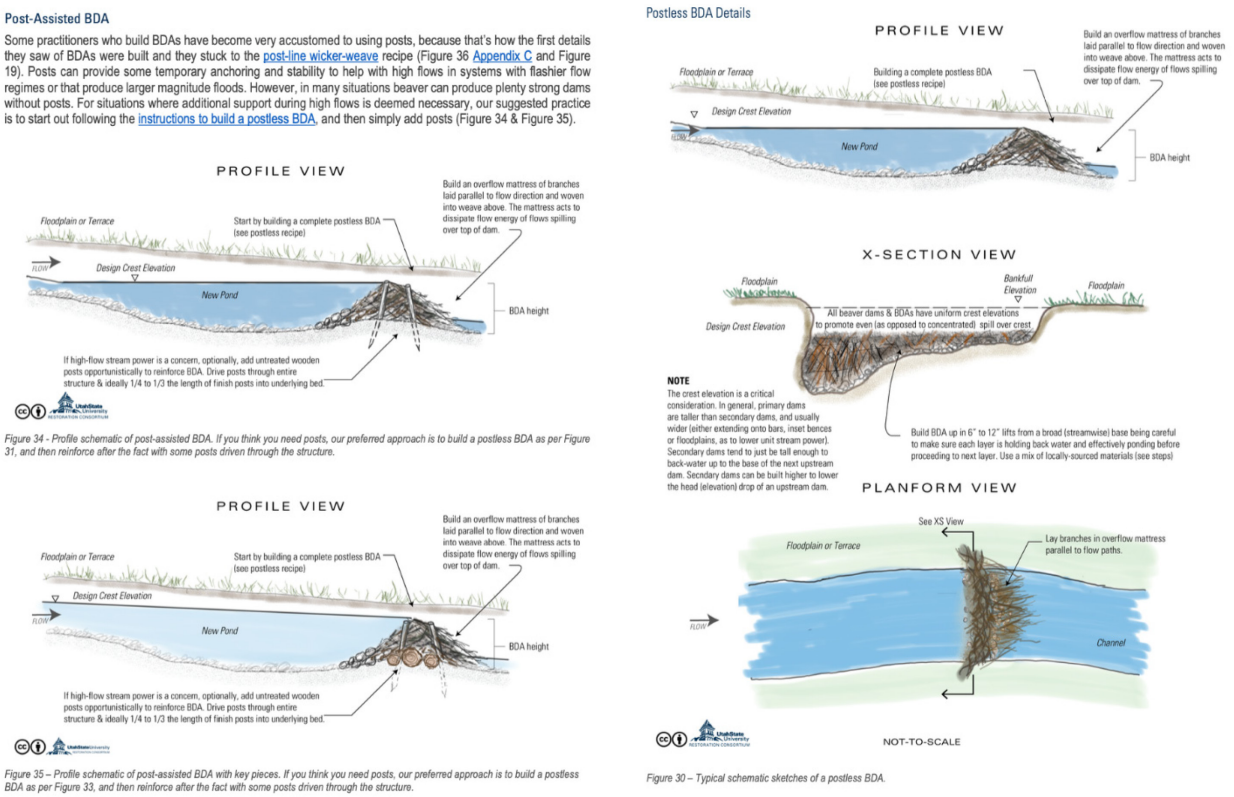



Figure 5.3.4. Above: Schematic plans for construction of Beaver Dam Analogs (BDAs). *Source:* Wheaton et al. 2019; https://lowtechpbr.restoration.usu.edu/manual/
Below: Left: Photo of heavy equipment installing a BDA on San Antonio Creek, Santa Fe National Forest; Right: Same site after construction of two BDAs, willow plantings, and a 10 acre elk exclosure.





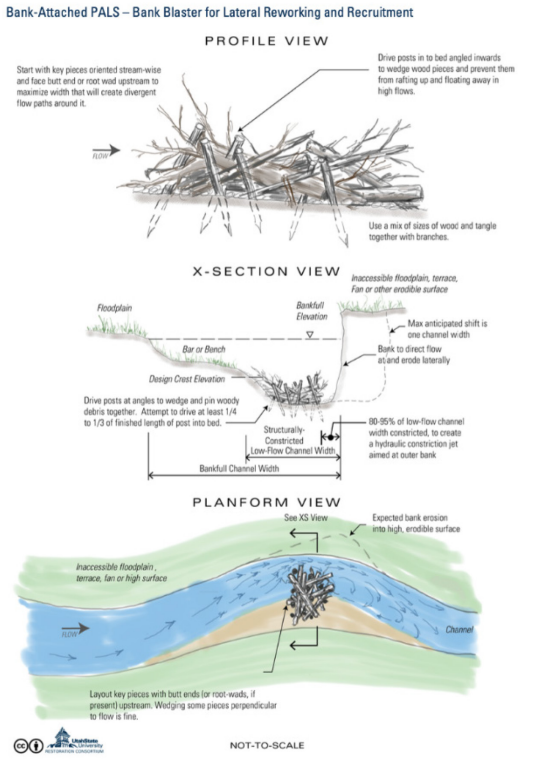



Figure 24 – Typical schematic sketches of a bank-attached PALS intended to blast and erode a bank to recruit sediment and/or wood.

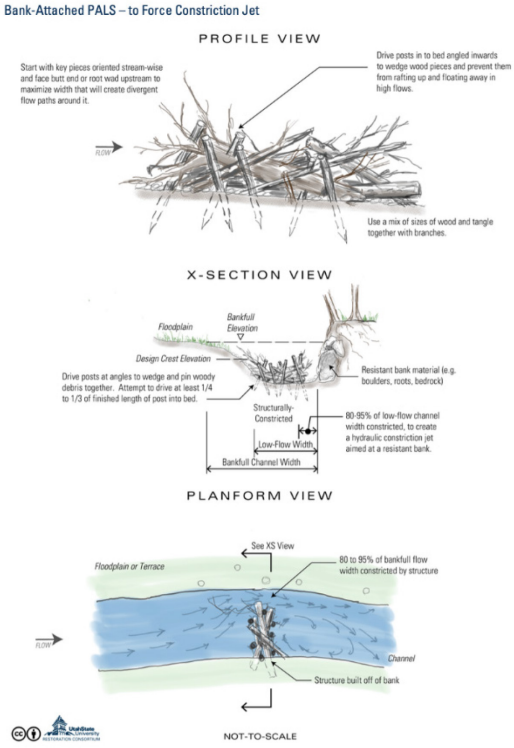



Figure 23 – Typical schematic of a bank-attached PALS directed at a resistant bank intended to force a constriction jet.

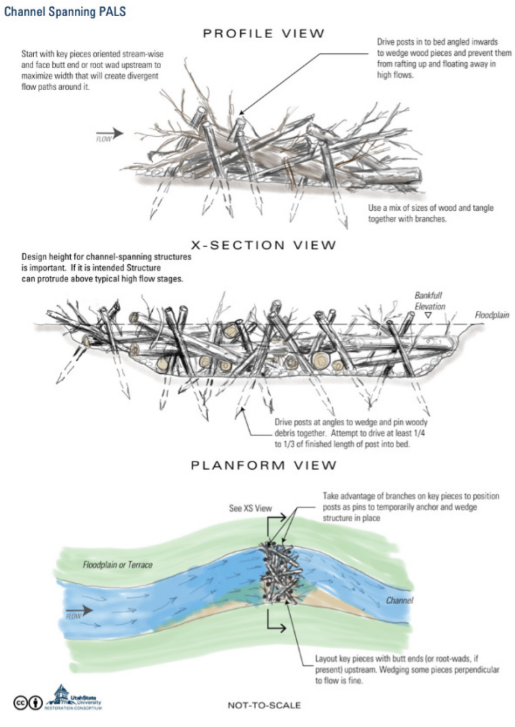



Figure 26 - Typical schematics of a channel-spanning PALS.

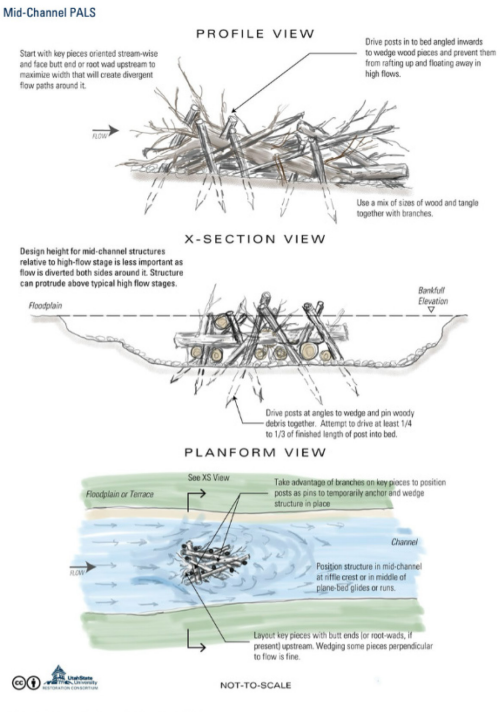



Figure 25 – Typical schematics of a mid-channel PALS.

Figure 5.3.5. Schematic plans for construction of Post-Assisted Log Structures (PALS). *Source:* Wheaton et al. 2019; https://lowtechpbr.restoration.usu.edu/manual/

Installation of BDAs and PALS produced dramatic effects on the hydrology of the restored streams. Monitoring of these changes through repeat photography confirmed that the structures were performing as expected, creating new wetland areas (e.g., Figure 5.3.6) and ponded areas suitable for beaver recolonization (Figure 5.3.7), and reconnecting the floodplain (Figure 5.3.8).



Figure 5.3.6. San Antonio Creek on the Santa Fe National Forest. Top: Pre-treatment reach of stream in 2011. Bottom: Same reach of stream in summer, 2024. Note higher water table and extensive wetland area created on right bank.



Figure 5.3.7. San Antonio Creek on the Santa Fe National Forest. Top: Stream reach prior to installation of Beaver Dam Analog (BDA) in 2019. Bottom: Stream site in 2024 with BDA installed.



Figure 1.3.8. San Antonio Creek on the Santa Fe National Forest. Top: Preconstruction photo taken in 2019 following planting and exclosure construction that took place between 2011 and 2013. Bottom: Post construction photo taken in spring 2023 (prior to vegetation green-up) illustrating floodplain connection facilitated by the installation of BDAs.

## Section 5.4. Stream water-quality monitoring

Streams in the SWJM CFLRP project area were selected for water quality monitoring using Sonde instruments (Yellow Springs Instruments, Inc., Yellow Springs, OH), equipped with sensors for temperature, dissolved oxygen, pH, conductivity and turbidity (Figure 5.4.1). Sondes were deployed during the ice-free season, usually late May to mid-October). We set the Sondes to record measurements every 15 minutes, and data were down-loaded and probes recalibrated every 2-3 weeks. Sondes were placed in the Rio San Antonio on the western edge of Valles Caldera National Preserve, and in the East Fork Jemez River in Hidden Valley (where the East Fork leaves the Valle Grande). Additional Sondes were placed in the Rio Cebolla on the Santa Fe National Forest, upstream and downstream of the stream restoration work area using Beaver Dam Analog structures (BDAs), and at the confluence of the Rio San Antonio and East Fork Jemez River at Battleship Rock.



Figure 5.4.1. A Sonde instrument used to monitor stream water quality.

*Monitoring Results*

The major impact factors on stream water quality were the incidents of large-scale wildfires in the watershed, specifically the 2011 Las Conchas Fire and the 2013 Thompson Ridge Fire. The post-fire floods that occurred during the summer monsoons created large increases in turbidity that lasted for up to five years after the fires. These turbidity spikes during the summers were evident on both the Rio San Antonio and East Fork Jemez River in Valles Caldera National Preserve (Figures 5.4.2 and 5.4.3).

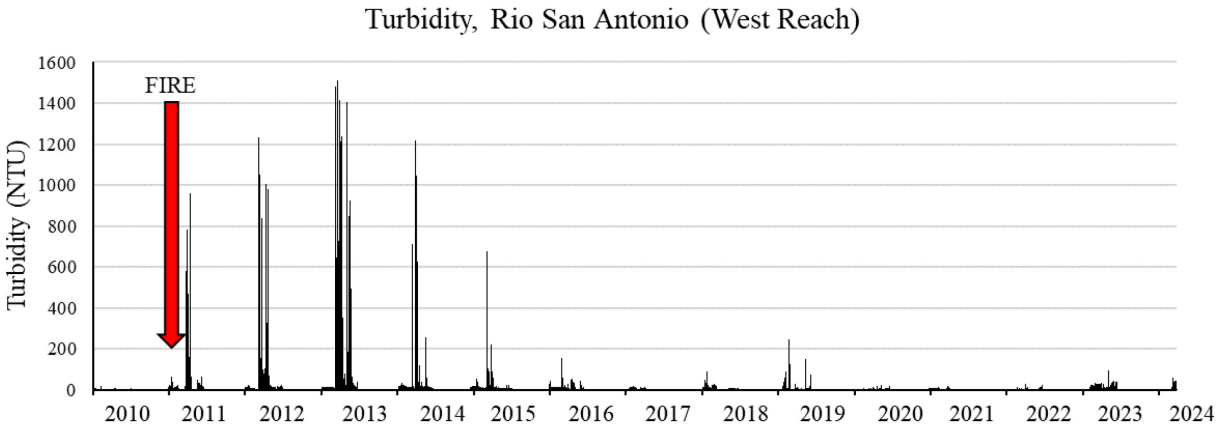



Figure 5.4.2. Stream water turbidity dynamics on the Rio San Antonio, western reach, on Valles Caldera National Preserve. Arrow indicates the occurrence of the Las Conchas Fire in the upper watershed in June-August 2011. The watershed hydrology returned to normal in 2016, five years after the fire.

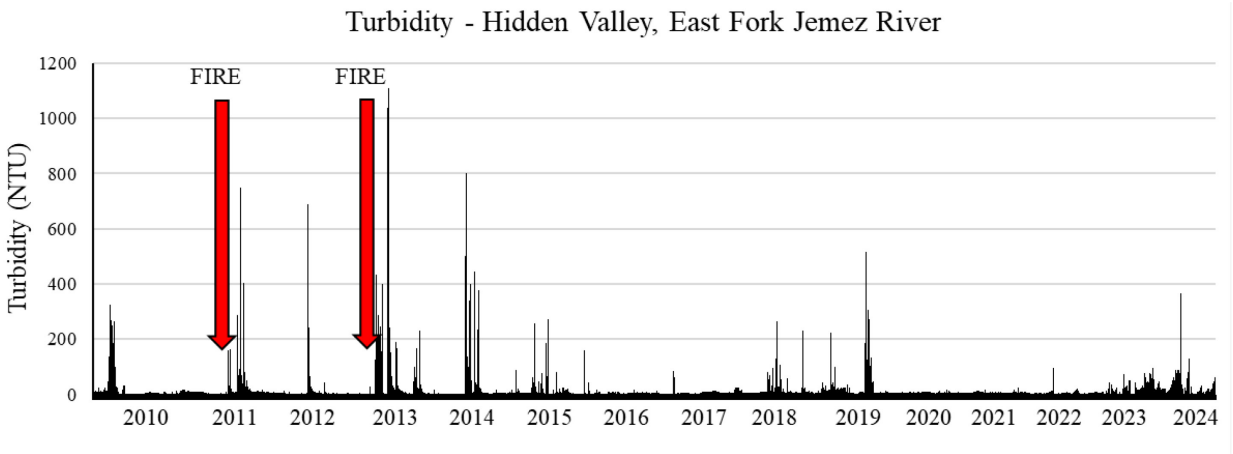



Figure 5.4.3. Stream water turbidity dynamics in the East Fork Jemez River at Hidden Valley on Valles Caldera National Preserve. Arrows indicate the occurrence of the Las Conchas Fire in the upper watershed in June-August 2011, and the Thompson Ridge Fire during May-June in 2013.

Long-term changes in temperature were recorded in the stream water during the time period of the SWJM CFLRP. Like the atmospheric (air) temperatures (see Chapter 3), the warming trend in the project area's streams was clear and statistically significant (Figures 5.4.4 and 5.4.5), even with the stream and wetland restoration activities undertaken in the project watersheds. Thus, while the stream and wetland restoration projects were successful in increasing acreages of wetlands and stabilizing miles of streambanks, water temperatures continued to slowly creep upward.

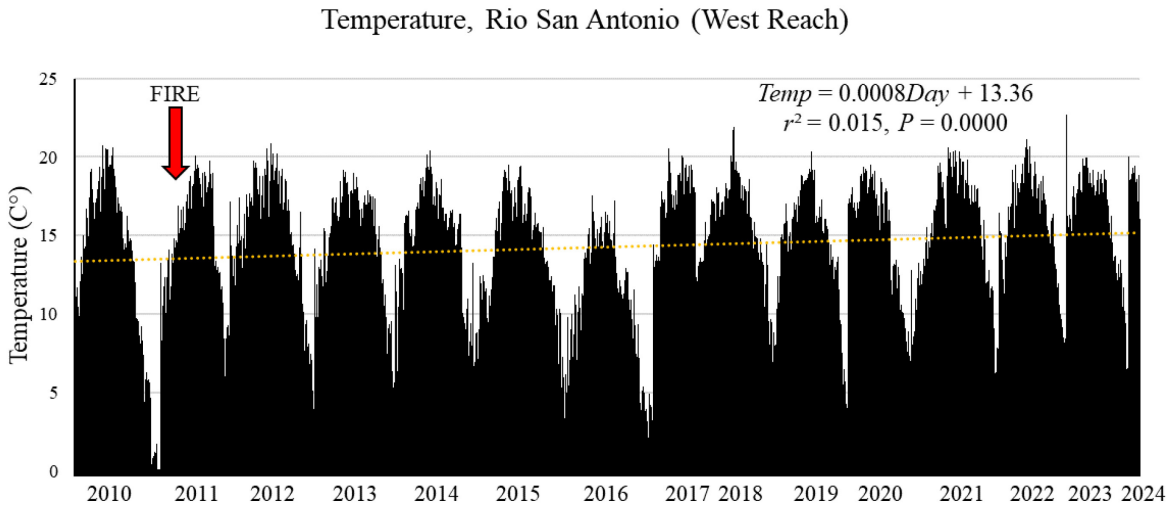



Figure 5.4.4. Stream water temperature dynamics on the Rio San Antonio, western reach, on Valles Caldera National Preserve. Arrow indicates the occurrence of the Las Conchas Fire in the upper watershed in June-August 2011. Note the warming trend (orange dotted line) with regression equation.

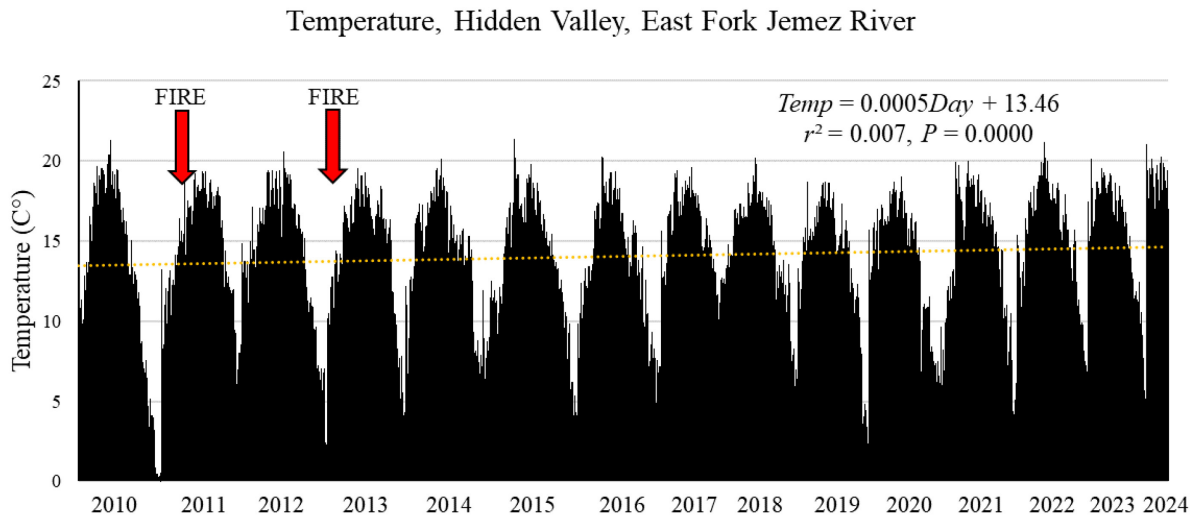



Figure 5.4.5. Stream water temperatures in the East Fork Jemez River in Hidden Valley on Valles Caldera National Preserve. Arrows indicate the Las Conchas Fire in June-August 2011 and the Thompson Ridge Fire in May-June 2013. Note the warming trend (orange dotted line) and regression equation.

An examination of the stream water quality data on the Rio Cebolla below the reach where BDAs were constructed shows the pattern of decreasing turbidity over several years after BDA installation (Figure 5.4.6). However, stream temperatures continued to warm during this period, along with the warming atmospheric temperatures (Figure 5.4.7).

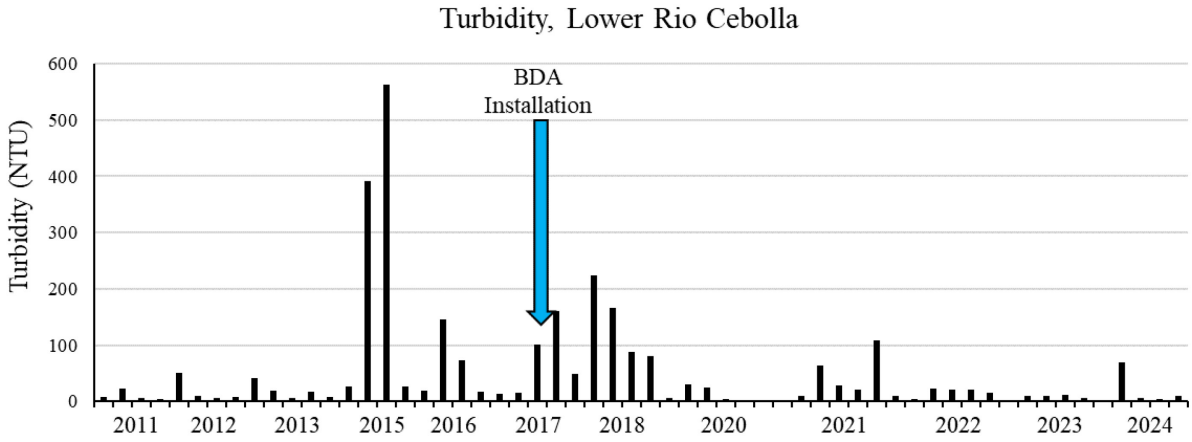



Figure 5.4.6. Turbidity dynamics in stream water (summer monthly means) in the Rio Cebolla below the reach with Beaver Dam Analogs (constructed in 2017). Increased turbidity from the construction was apparent initially, but stream turbidity returned to normal within a few years after deployment.

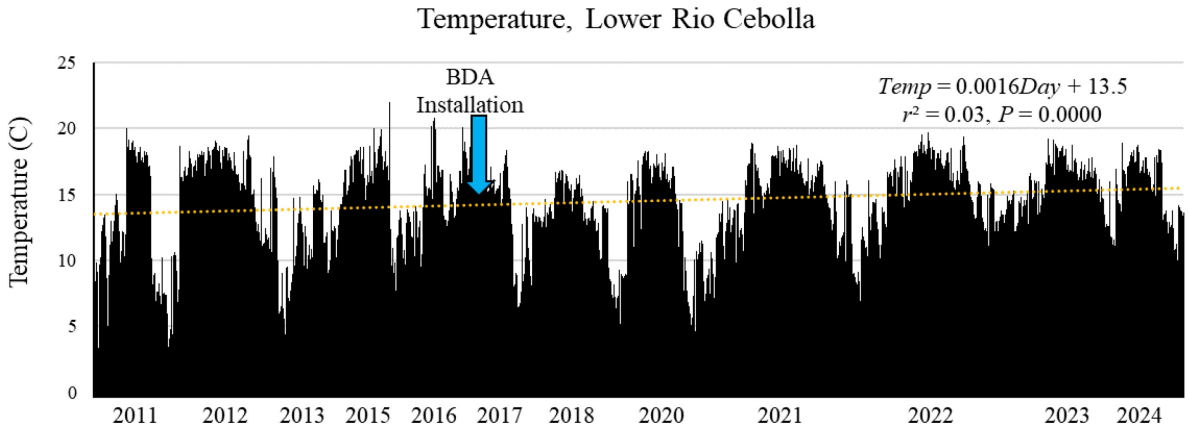



Figure 5.4.7. Stream water temperatures (daily averages in summer months, May-October) in the lower Rio Cebolla reach, below the area with Beaver Dam Analogs (BDAs). While the BDAs were successful in stabilizing the stream banks and expanding wetlands, stream temperatures continued to slowly rise in concert with atmospheric temperature warming.

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# Chapter 6. Wildlife Monitoring – Fish

## Monitoring project title: Monitoring stream fisheries for fire and restoration effects in the SWJM CFLRP project area.

**Collaborators:** Robert R. Parmenter, Mark Peyton, Tyler Obermeit – *National Park Service*; Colleen A. Caldwell, Michael Anderson – *U.S. Geological Survey, New Mexico Cooperative Fish and Wildlife Research Unit, Department of Fish, Wildlife and Conservation Ecology, New Mexico State University*; Numerous citizen volunteers from *New Mexico Trout, Trout Unlimited, Sierra Club*, and *New Mexico Gas Company*.

**Summary.** One of the objectives of the CFLRP was to "improve fish and wildlife habitat" through landscape restoration of forested watersheds. While the streams of the SWJM CFLRP had been listed by the New Mexico Environment Department as "impaired" for high temperatures and high turbidity levels, the fish communities in the Jemez River watershed were nearly complete in their native species composition; only the Rio Grande cutthroat trout was missing, having been extirpated during the 20th century through the introduction of non-native trout species (German brown trout, and rainbow trout). The native non-game fish assemblage included the longnose dace, the Rio Grande sucker, the Rio Grande chub, and the fathead minnow, and these species populated their appropriate stream habitats throughout the CFLRP project area. In 2003, it was discovered that the Rio Grande sucker and Rio Grande chub were missing from the upper Rio San Antonio region, but these species were ultimately reintroduced on Valles Caldera National Preserve in 2008 and successfully established breeding populations in the following years. As such, forest restoration on the watershed was not expected to significantly affect fish communities, but monitoring activities were instituted to ensure that potential negative impacts from thinning (e.g., possible soil erosion from heavy equipment use) and increased recreation fishing activities did not cause detrimental changes in the fish populations. In 2011, right at the beginning of the SWJM CFLRP, the Las Conchas Fire burned extensive areas of the upper Jemez River watershed on the Preserve, and post-fire monsoon floods caused massive fish kills in the upper watershed – primarily of the brown and rainbow trout populations. Surprisingly, the native non-game fish species did not appear to have suffered high mortality from the floods, and their populations expanded following the temporary elimination of the predaceous brown trout population. This "predator release" response continued for several years, but as brown trout recolonized the flooded reaches from downstream populations, the native non-game fish species declined to their pre-fire levels. Overall, stand-replacement fire at a landscape level had a much greater impact on stream fisheries than did smaller-scale forest thinning activities.

### Fisheries monitoring

Monitoring of fish communities was conducted throughout the Jemez River watershed on the Rio San Antonio and East Fork Jemez River, as well as the Jemez River in Jemez Springs. Fish were sampled on permanent 100-meter long reaches at 13 locations (see map in Figure 6.1), and most sites were eventually affected by the Las Conchas Fire and Thompson Ridge fires (Figure 6.2). Sites were sampled each spring and autumn using electro-shockers (Fig. 6.3). Each sample consisted of a minimum of 3 passes from downstream to upstream through the 100-m reach;

blocking nets were installed at the top and bottom of each reach to prevent fish from leaving or entering the sample area. Fish that were stunned by the electro-shocker were netted out of the stream and transported in buckets to a streambank processing site, where they were identified to species, measured and weighed. Fish were then placed in aerated buckets and until they were moved into a temporary holding pen in the stream; when the final sampling pass was complete, all fish were released back along the 100-meter reach. Data reported in this chapter are shown as the number of total fish by species within the 100-meter reach for each sample period.

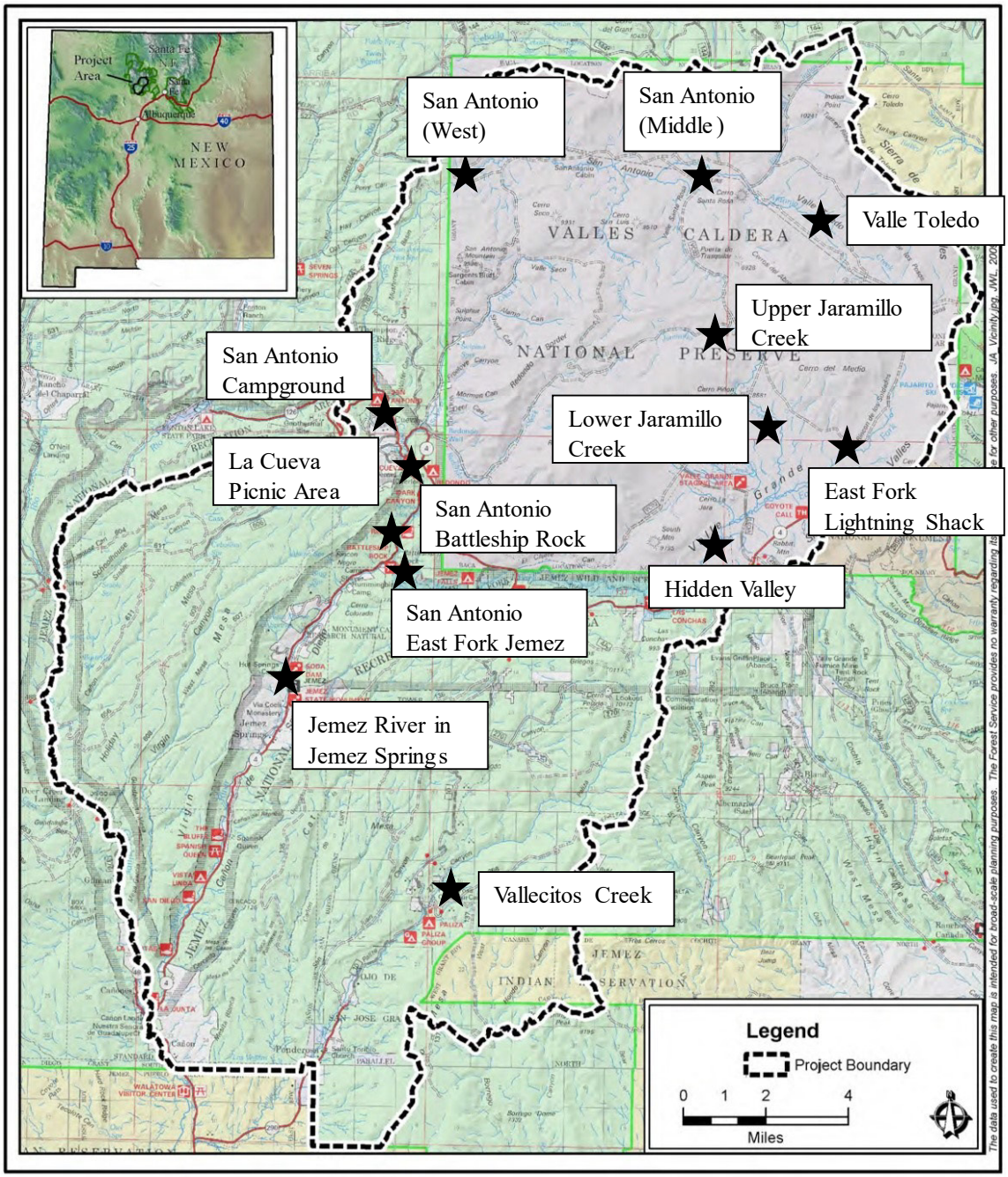



Figure 6.1. Map of 13 fish monitoring sites in the SWJM CFLRP project area.

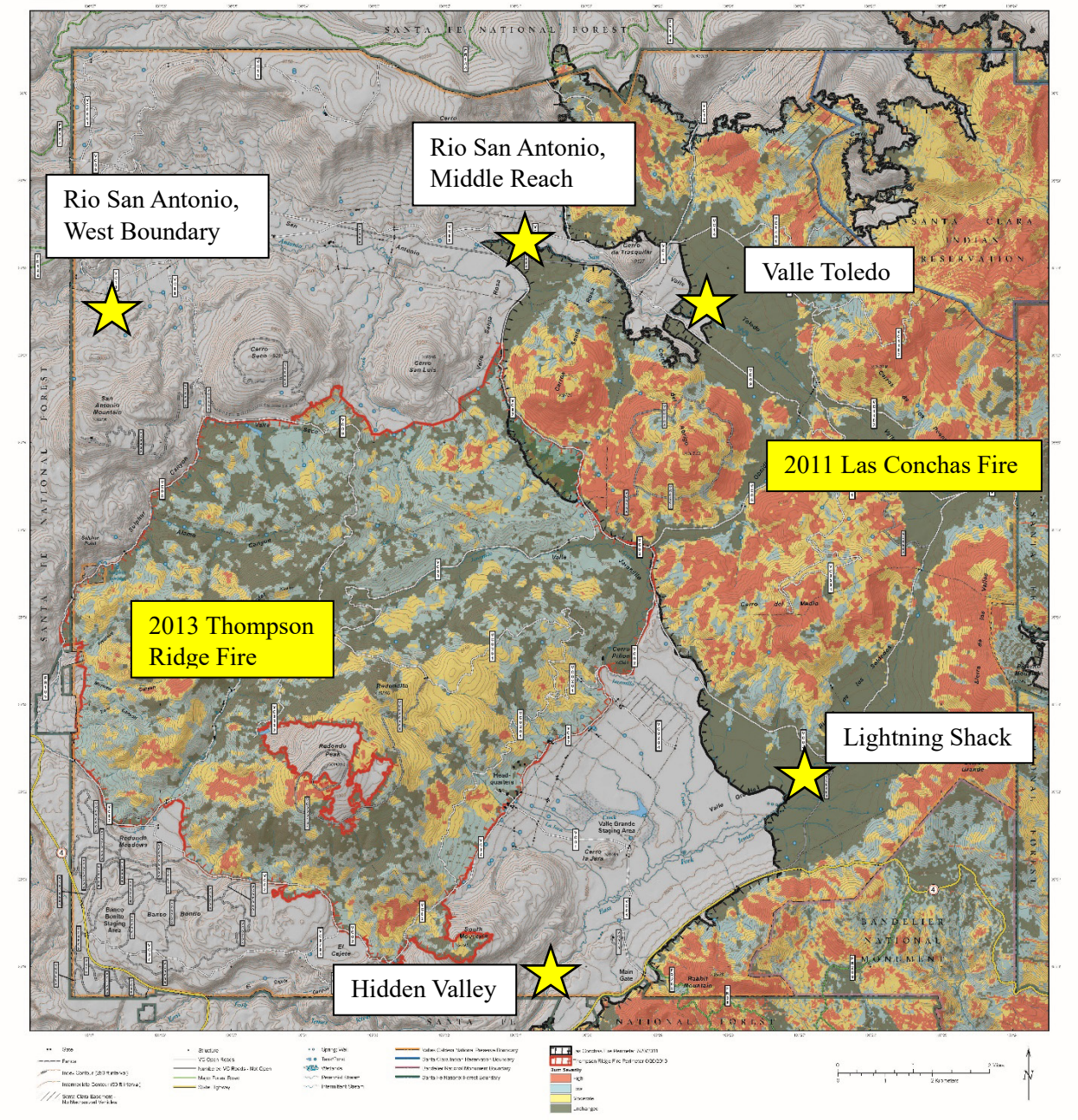



Figure 6.2. Map of burn severities in the 2011 Las Conchas Fire (right) and the 2013 Thompson Ridge Fire (left) on Valles Caldera National Preserve. Yellow stars indicate fish monitoring sites on the Preserve that are discussed in this report. Fish monitoring sites on the Santa Fe National Forest are presented below.




Figure 6.3. Left: Electro-shocking fish in Hidden Valley; note white floats of blocking net in background. Right: Processing station on streambank where staff and volunteers identify, measure and weigh captured fish prior to release.

**Results and Discussion**

The fish monitoring program had begun in the upper Jemez River watershed in 2003 and continued through 2024. In total, 56,623 fish were collected and released during this time period. In 2011, near the beginning of the SWJM CFLRP, the Las Conchas Fire burned much of the upstream watershed in late June and early July. As the monsoons arrived in late July, the stream reaches in the upper Rio San Antonio were subjected to out-of-bank flash floods (Figure 6.4), resulting in lethal changes in water quality and extensive kills of brown trout (Figure 6.5).




Figure 6.4. Left: View of the Rio San Antonio during normal flows; stream is usually 1-2 m wide. Right: Post-fire flash flood on July 29, 2011, showing out-of-bank flooding.




Figure 6.5. Left: Water samples collected in the Rio San Antonio immediately before (left bottle) and during the flash flood (right bottle) on July 29, 2011, following the Las Conchas Fire. Data show water chemistry results for each; note the high concentration of ammonia that is lethal for trout. Right top: Brown trout carcasses on July 30, following the flash flood the day before. Right bottom: Ash deposits in gills of dead brown trout.

As a result of the changes in the stream's water quality, the resident population of brown trout were entirely killed in the upper reaches of the Rio San Antonio (see Figure 6.6, Valle Toledo site; refer to Figures 6.1 and 6.2 for site location map; *Note*: Rainbow trout were not present in the upper Rio San Antonio watershed). Recovery of the brown trout population from downstream populations began the following year (2012) and required ~6 years to recover to a sustainable breeding population (Figure 6.6). At this site in the Valle Toledo, brown trout were nearly the only species present and were the top predators in the ecosystem, but when they were extirpated by the post-fire floods, a small number of non-game native fish colonized the site for a few years in the absence of their main predator (Figure 6.7); but as the brown trout population recovered, these native fish species were again eliminated from the site.

The same pattern was observed several kilometers downstream at the Middle Reach of the Rio San Antonio (Figure 6.8; see map in Figures 6.1 and 6.2 for site locations). At this location, the brown trout population had been fairly steady at ~80 individuals per 100 meters of stream, but during the winter of 2009 – 2010, the population dropped by about 50% to ~40 fish/100 m; it is not clear why this occurred, although increased recreational angling pressure may have contributed to the decline. The 2011 Las Conchas Fire floods also extirpated the brown trout from this site, but the native non-game fish species did not appear to be affected by the change in water quality. We hypothesize that these native species had evolved with landscape fires over many millennia and have become more tolerant to pulses of low water quality. Introduced trout species, the German brown trout from Europe, and the rainbow trout from the Pacific Northwest, had not evolved with high-severity fires, and were apparently more susceptible to post-fire flood water

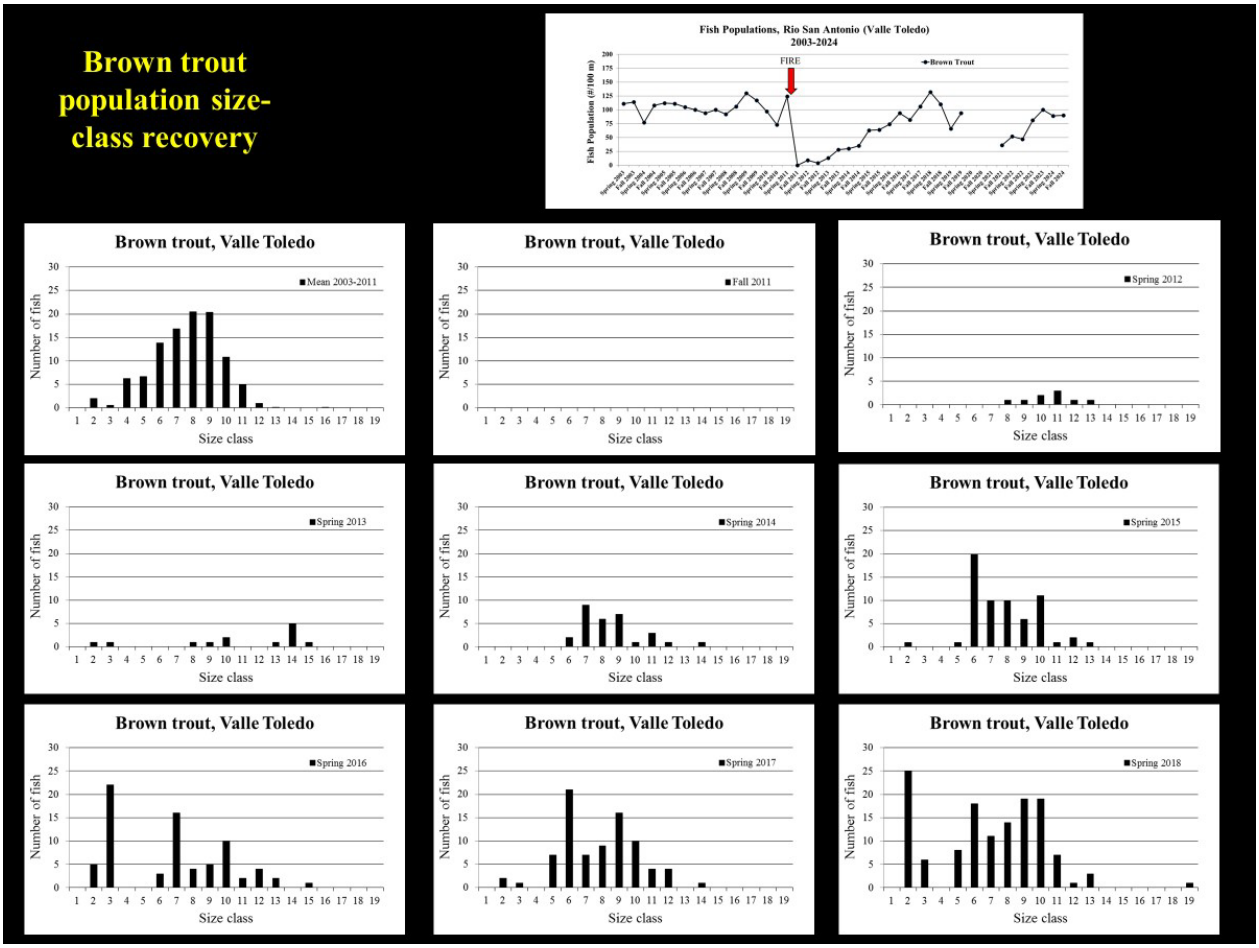



Figure 6.6. Graphs of the Rio San Antonio brown trout population (top) and size class distributions for pre-fire (2003-2011) and post-fire (2011-2024) showing extirpation of this species in 2011 in the Valle Toledo by the Las Conchas Fire floods, and the post-fire recovery of the population. Young age classes were mostly present by 2016. Data from spring 2020 to spring 2021 not collected due to the Covid-19 pandemic. Refer to Figure 6.1 for site location map.

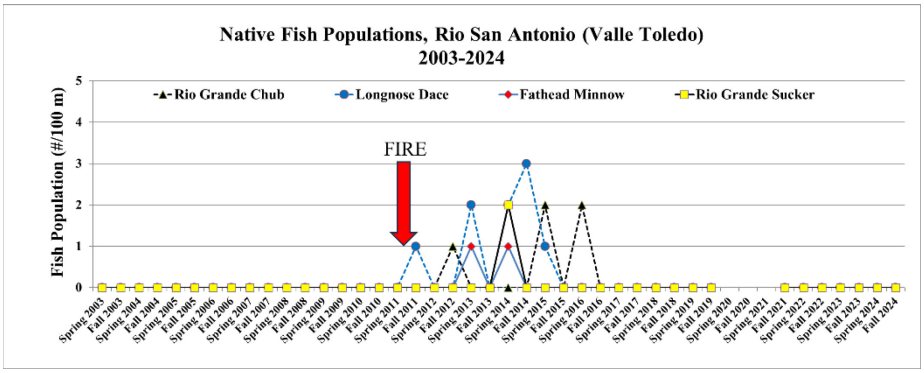



Figure 6.7. Population dynamics of non-game native fish species in Valle Toledo, 2003-2024.

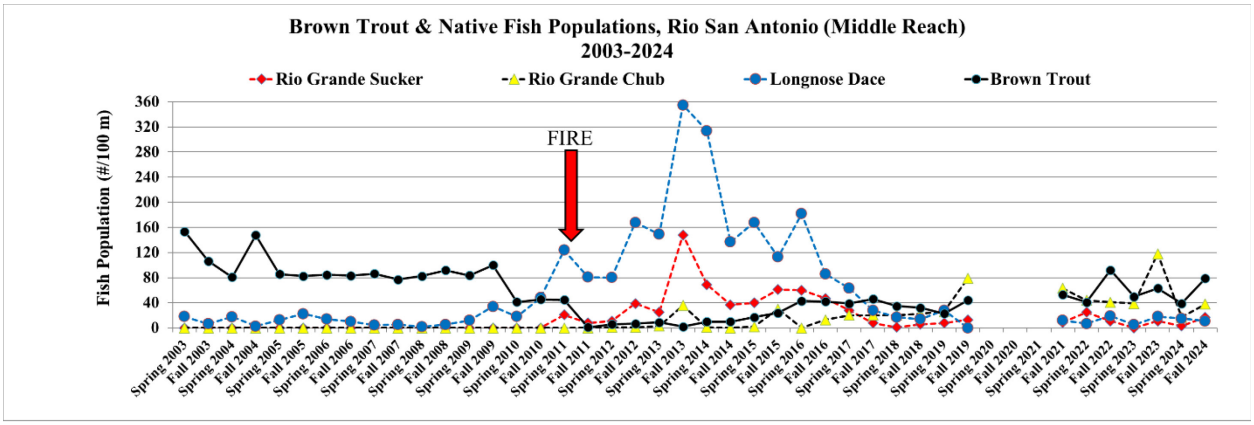



Figure 6.8. Population dynamics of fish species in the Rio San Antonio, Middle Reach, during 2003-2024. Note brown trout population reduction over the winter of 2009-2010 (for unknown reasons) and the population extirpation by the Las Conchas Fire in July, 2011. Non-game fish species populations expanded during this period of "predator release", but eventually returned to pre-fire levels as the brown trout population returned to normal.

quality changes. With the elimination of their main predators, the native fish species (especially the longnose dace) exhibited increased populations for several years, again providing an example of "predator release" during the post-fire period. Note that the Rio Grande sucker and Rio Grande chub, which were reintroduced to the Rio San Antonio in 2008, continue to coexist with the predaceous brown trout.

The fish monitoring site at the western boundary of the Preserve on the Rio San Antonio provided data that indicated the water quality had recovered by the time the floods arrived in that reach. Fish populations here did not show obvious responses to the post-fire floods in July, 2011 (Figure 6.9), although the brown trout population has exhibited a steady decline from 2003 to 2014, with occasional years of increase between 2015 and 2014; again, angling pressure may have contributed to this trend. As with the Middle Reach, the longnose dace showed a dramatic increase in 2010, and Rio Grande chubs have exhibited recent increases in population despite the continued presence of brown trout (Figure 6.9).

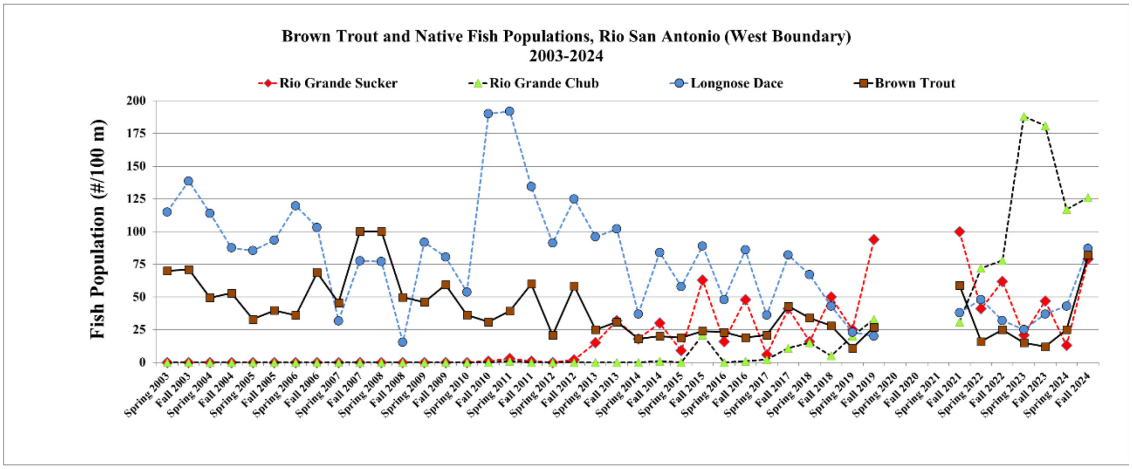



Figure 6.9. Fish population dynamics in the Rio San Antonio, West Boundary reach.

In the watershed of the East Fork Jemez River, the upper fish monitoring site (known as the "Lightning Shack") was located just downstream from the headwater springs of the stream. This location was, again, heavily dominated by large numbers of brown trout, but also with rainbow trout (Figure 6.10). Native fish species were relatively rare, but again showed a "predator release" after the post-fire floods of the Las Conchas Fire (Figure 6.11). For unknown reasons, the trout populations of both species has remained very low since 2014, with episodic population eruptions of Rio Grande chubs and Rio Grande suckers during this time (Figure 6.11).

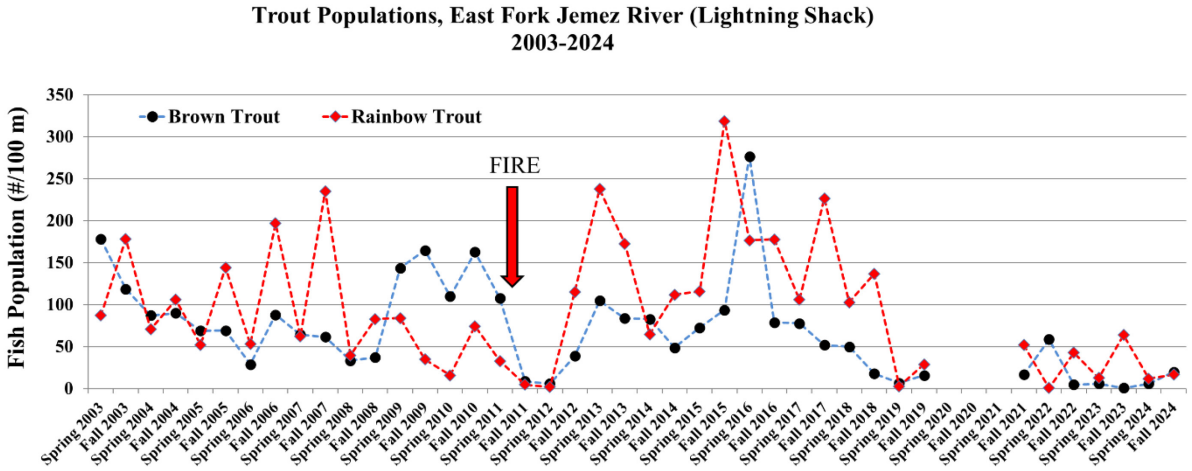



Figure 6.10. Population dynamics of brown and rainbow trout at the headwaters site of the East Fork Jemez River (see map in Fig. 6.1 for site location). Note the extirpation of both species by the July, 2011, post-fire floods from the Las Conchas Fire.

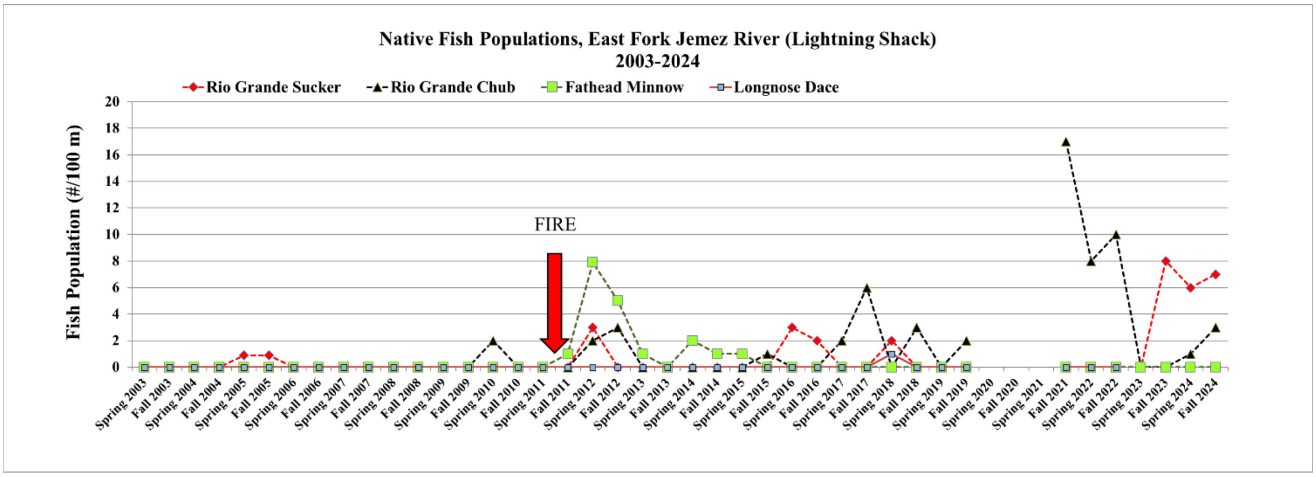



Figure 6.11. Population dynamics of native (non-game) fish species at the headwaters site of the East Fork Jemez River (see map in Fig. 6.1 for site location).

At the downstream site near the southwestern edge of the Valle Grande (known as "Hidden Valley"), brown and rainbow trout were not as dominant, and the native fish species were more common (Figure 6.12); fathead minnows and Rio Grande chubs showed occasional massive increases in populations, exceeding over 600 – 700 individuals/100 m of stream (many of these were young-of-year). This site was >8 km from the burned areas of the Las Conchas Fire, so the post-fire floodwater quality was not sufficient to cause trout mortality; hence, we did not observe an obvious flood effect at this site except for the continued population irruption of fathead minnows which started the year earlier (6.12).

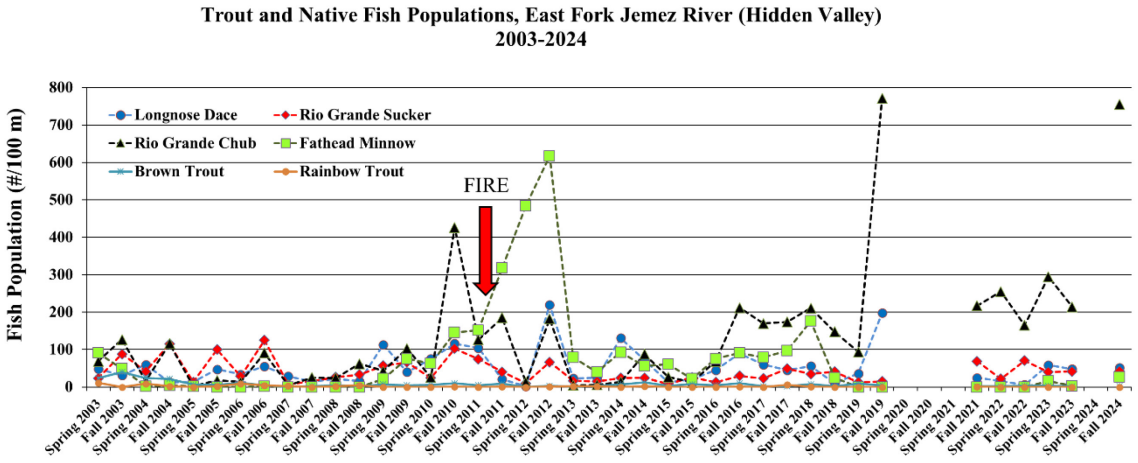



Figure 6.12. Population dynamics of fish species at the Hidden Valley site of the East Fork Jemez River (see map in Fig. 6.1 for site location).

Fish monitoring sites on the Santa Fe National Forest were established in 2013 (see map in Figure 6.1) and sampled each spring and autumn through 2024; protocols were the same as used on the Valles Caldera National Preserve by the same field crews. While these sites were set up two years after the Las Conchas Fire, they were downstream of the 2013 Thompson Ridge Fire, and were subjected to some post-fire floods from tributary streams running off Redondo Peak (Figure 6.2 above).

The first pair of sites (San Antonio Campground and La Cueva Picnic Area) were located on the Rio San Antonio, upstream and downstream of the confluence with Sulphur/Redondo Creeks which sustained post-fire flash floods from the Thompson Ridge fire (see Figure 6.13 for the flood pathways relative to the monitoring sites). The floods during the 2013 summer monsoons eliminated both brown and rainbow trout from the La Cueva Picnic Area monitoring site (Figure 6.14) and appeared to reduce the numbers of longnose dace and Rio Grande suckers (the two native non-game fish on the site). Trout recovered on this site due primarily to release of fish by the New Mexico Department of Game & Fish.

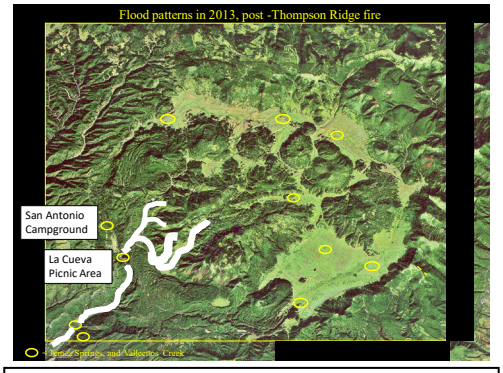



Figure 6.13. Map of fish monitoring sites and post-fire flood pathways.

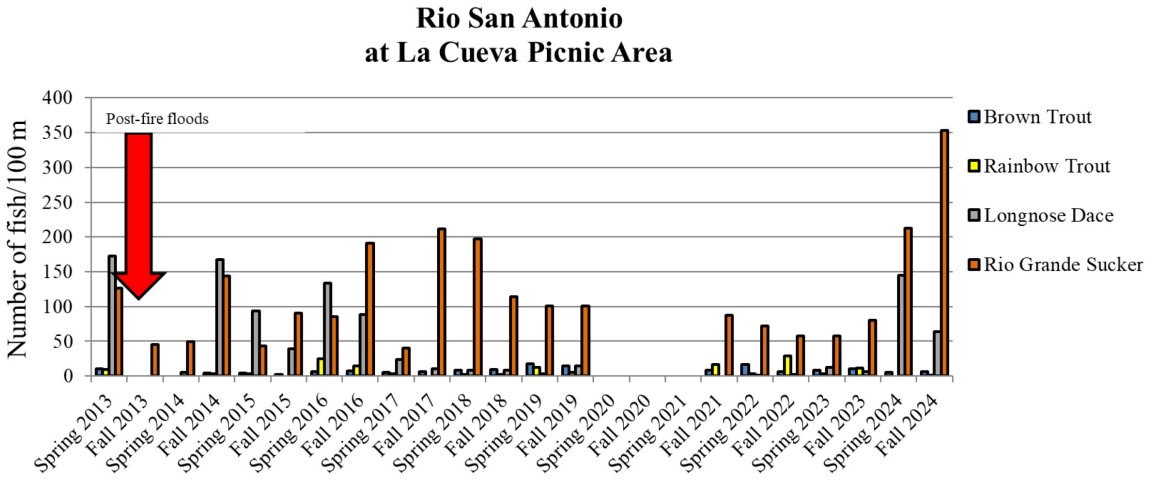



Figure 6.14. Fish population dynamics on the Rio San Antonio at the La Cueva Picnic Area (Santa Fe National Forest) during 2013-2024. Red arrow indicates the Thompson Ridge Fire in the upper watershed of Redondo Creek and Sulphur Creek. Data from spring 2020 to spring 2021 not collected due to the Covid-19 pandemic.

In contrast, the monitoring site at San Antonio Campground, which was upstream of the confluence and did not receive post-fire floods, did not lose any species during the same time period (Figure 6.15), though all species exhibited a general reduction in abundance during the summer of 2013. All species showed increases by spring 2014.

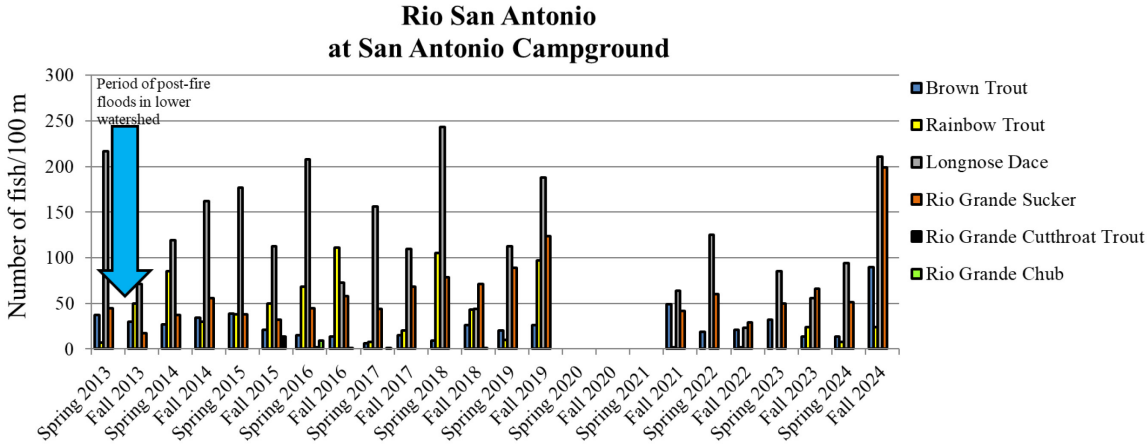



Figure 6.15. Fish population dynamics on the Rio San Antonio at the San Antonio Campground (Santa Fe National Forest) during 2013-2024. Blue arrow indicates the Thompson Ridge Fire for comparison with stream reaches that were impacted by post-fire floods; this site was not subjected to post-fire floods. Data from spring 2020 to spring 2021 not collected due to the Covid-19 pandemic.

Further downstream on the Rio San Antonio, the fish monitoring site at Battleship Rock (Figure 6.16) did not show any responses to the post-fire floods of 2013. This site was sufficiently far downstream that the turbidity and ammonia levels would have declined to sub-lethal levels, allowing all fish species to survive with little if any mortality (Figure 6.17). The paired site on the East Fork Jemez River, which did not sustain any post-fire floods from the Thompson Ridge Fire, also showed no substantial changes in the fish community during this period (Figure 6.18).

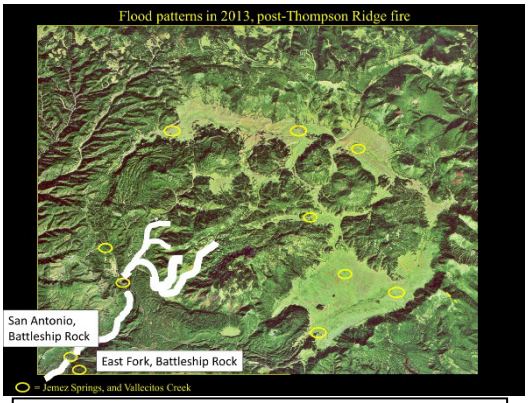



Figure 6.16. Map of fish monitoring sites and post-fire flood pathways.

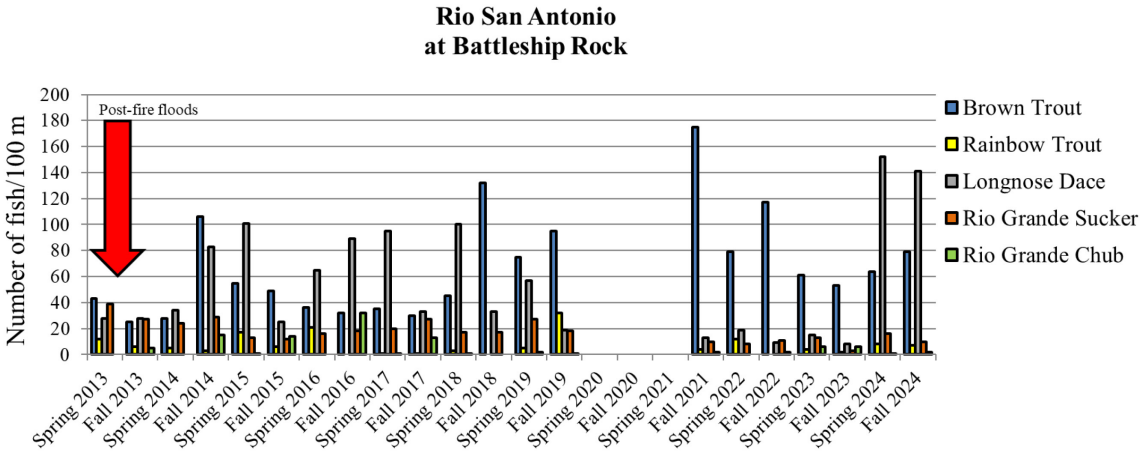



Figure 6.17. Fish population dynamics on the Rio San Antonio at Battleship Rock (Santa Fe National Forest) during 2013-2024. Red arrow indicates the Thompson Ridge Fire in the upper watershed of Redondo Creek and Sulphur Creek. Data from spring 2020 to spring 2021 not collected due to the Covid-19 pandemic.

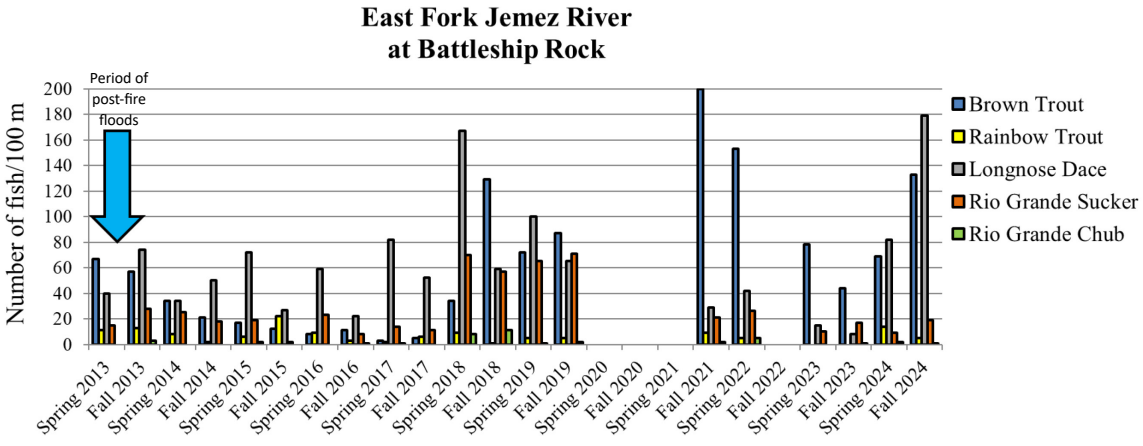



Figure 6.18. Fish population dynamics on the East Fork Jemez River at Battleship Rock (Santa Fe National Forest) during 2013-2024. Blue arrow indicates the period of time of the Thompson Ridge Fire, but this site did not experience any post-fire flooding. Data from spring 2020 to spring 2021 not collected due to the Covid-19 pandemic; data in fall 2022 missed due to weather conditions.

Further downstream on the Jemez River, the fish communities in the Village of Jemez Springs (Figure 6.19), within the NPS Headquarters site of Valles Caldera National Preserve, also displayed no clear response to the Thompson Ridge post-fire flooding. Trout populations were never very high during the period of sampling and were driven mostly by recreational fishing (removal by anglers and replacement by the New Mexico Department of Game & Fish). Native non-game fish species experienced considerable fluctuations in population through time, but were not attributable to fire-related flood events.

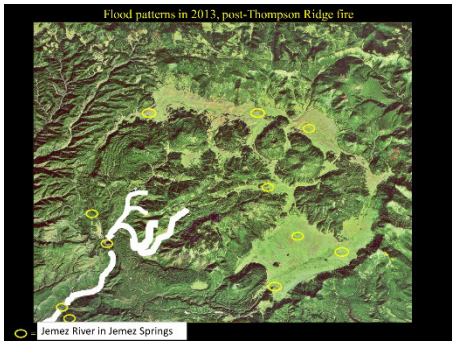



Figure 6.19. Map of fish monitoring sites and post-fire flood pathways.

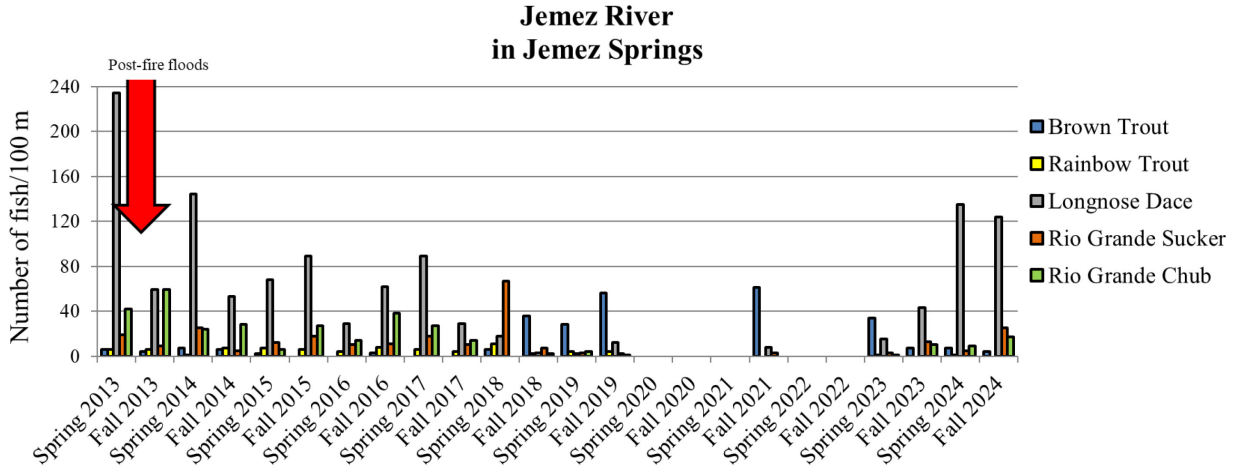



Figure 6.20. Fish population dynamics on the Jemez River in Jemez Springs during 2013-2024. Red arrow indicates the Thompson Ridge Fire in the upper watershed of Redondo Creek and Sulphur Creek. Data from spring 2020 to spring 2021 not collected due to the Covid-19 pandemic; 2022 data missing due to weather conditions.

Finally, fish monitoring on Vallecitos Creek in the Santa Fe National Forest (Figure 6.1) found only brown trout in the watershed, and following the Guacamalla Fire in 2011 that burned 1,558 acres in the upstream watershed, this population of introduced, non-native trout steadily declined to zero (Figure 6.21). This stream is currently under consideration for a potential reintroduction of native fish species, including longnose dace, Rio Grande chub, Rio Grande sucker and possibly Rio Grande cutthroat trout.

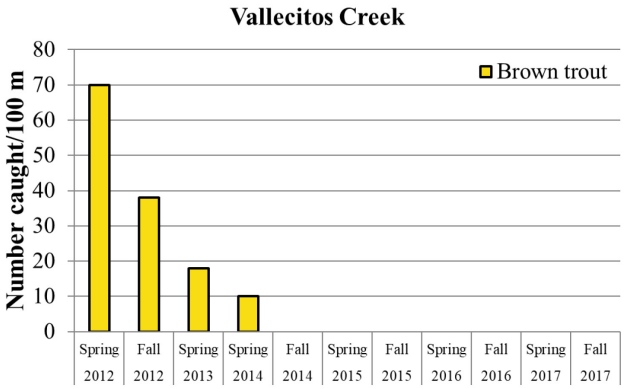



Figure 6.21. Brown trout population in Vallecitos Creek, Santa Fe National Forest. Sampling terminated after 2017, except for visual checks to verify fish absence.

## Conclusions:

1. 13 years post-Las Conchas Fire and 11 years post-Thompson Ridge Fire, the fish monitoring results are meeting expectations of post-fire recovery, in that non-native trout populations have recovered, and native non-game fish species have responded to predator (brown trout) decline and eventual predator recovery.

2. Watershed restoration implementation efforts have had no detectable negative impacts on trout fisheries and native fish populations.

3. Forest restoration activities are predicted to reduce high-severity fires, and this will continue to be the major factor in maintaining healthy fish communities in the streams of the Jemez Mountains.

4. Vallecitos Creek on the Santa Fe National Forest has been de-populated of brown trout, and thus provides an opportunity to reintroduce native fish species in the absence of non-native predatory fish.

# Chapter 7. Wildlife Monitoring – Aquatic Invertebrates

**Monitoring project title: Effects of large-scale wildfire on the spatial and temporal response of benthic aquatic macroinvertebrate communities in Valles Caldera National Preserve.**

**Collaborators:** Lauren Kremer, Colleen A. Caldwell – *U.S. Geological Survey, New Mexico Cooperative Fish and Wildlife Research Unit, Department of Fish, Wildlife and Conservation Ecology, New Mexico State University;* Jerry Jacobi*, New Mexico Highlands University.*

**Summary.** Inventory and monitoring programs are important for multiple land-use practices where land managers must balance the use of natural resources with species conservation and diversity. The Valles Caldera National Preserve (Preserve) represents an unprecedented national experiment in public land management that comprises nearly 36,560 ha of the Valles Caldera volcanic complex in the Jemez Mountains in north central New Mexico. Under private ownership from the early 1820's through 2000, the area was managed for livestock (sheep and cattle) grazing, timber, and geothermal energy exploration. After federal acquisition in 2000, the Preserve was mandated to maintain a multiple land use policy to protect and preserve the sustained yield of renewable natural resources that included grazing, timber, and recreation while maintaining ecological integrity. As such, inventory and monitoring programs were established throughout the Preserve that included wildlife, upland forests, lowland grasslands, and fish and aquatic health of the streams. In conjunction with fish and aquatic health, a macroinvertebrate biomonitoring program began in 2004 to monitor the long-term trends of grazing practices on water quality and the benthic macroinvertebrate community throughout the Preserve's streams. On 26 June 2011, a wildfire (named the Las Conchas Wildfire) started outside of the Preserve's boundary ultimately burning 63,371 ha of Santa Fe National Forest, Bandelier National Monument, Los Alamos Laboratory, and the Preserve until fully contained on 1 August 2011. The Las Conchas wildfire became the largest wildfire in New Mexico's recorded history. The fire burned 12,156 ha or one-third of the Preserve. Immediately after the wildfire in 2011, the Preserve added additional biomonitoring sites to supplement the long-term monitoring of the benthic macroinvertebrate communities throughout the Preserve to describe ecological changes occurring post-fire and recovery trajectory of the aquatic macroinvertebrate communities.

To describe ecological changes related to recovery of the aquatic macroinvertebrate community post-fire, our first objective was to continue biomonitoring of the aquatic macroinvertebrate community. Analysis of the biomonitoring data, however, was not straightforward. From 2004 to 2018, different investigators employed a total of three gear types for biomonitoring of the benthic macroinvertebrate community. Thus, our second objective was to resolve how the differing gear types affected the data and thus interpretation of our ecological metrics. When we compared the use of a commercial Surber sampler with 500 µm mesh (2004-2006), a Hess-like sampler with 500 µm mesh (2005-2014) and a commercial Hess with 250 µm mesh (2015-2018), we determined that the Surber sampler did not capture similar number of taxa relative to the Hess sampler, which resulted in different density and diversity estimates of macroinvertebrates. Moreover, we learned the two mesh sizes (250 µm versus 500 µm) resulted in size selectivity of taxa whereby larger and more common taxa were captured by the 500 µm mesh and the smaller and rarer taxa were captured by the 250 µm mesh. We learned that there

were no comparable diversity metrics among the gear types or mesh sizes. Given this, we characterized environmental variables that were critical to the recovery and health of the aquatic macroinvertebrate community while treating each gear type or mesh size as a unique dataset. As such, our third objective was to identify which environmental variables co-occurred with aquatic macroinvertebrate assemblages as primary drivers of ecological change before and after the wildfire. We used random-forest algorithms to characterize ecological metrics best explained by the environmental variables. We learned that water temperature, dissolved oxygen, and water turbidity were the important predictors of Shannon diversity and feeding group richness. Our fourth objective compared these ecological metrics before and after the Las Conchas wildfire to determine the impact and time to recovery of the aquatic macroinvertebrate assemblages. We learned the aquatic macroinvertebrate communities across un-burned as well as burned streams of the Preserve exhibited large inter-annual variation before and after the wildfire, which precluded our ability to detect changes pertaining to wildfire effects and recovery. These temporal responses appear unrelated to fire effects and may simply reflect the overarching influence of the larger diel or daily range in stream temperature and seasonal pulses of materials that occur in open meadow grassland streams.

In summary, differences in gear types and mesh sizes prevented the standardization of 11 years of biomonitoring data which precluded long-term and seamless comparisons of ecological metrics of the aquatic macroinvertebrate community before and after the wildfire. However, for the gear-types with similar mesh sizes immediate post-fire, we found significant relationships among ecological metrics with environmental variables of water temperature, dissolved oxygen and turbidity. The ecological metrics suggest that time to recovery of the benthic invertebrate community was relatively short-term and that this recovery was supported by improvement of water quality as the surrounding watershed has begun recovery.

## Introduction

Freshwater systems include the most biodiverse environments in the world despite the low abundance of freshwater relative to marine environments (0.0093%, Bone and Marshal 1982; Woodward et al. 2010). These diverse environments include lakes and streams which are home to a wide array of benthic macroinvertebrates, each adapted to a particular niche within the heterogeneity of the benthos (Townsend et al. 1997). However, streams are susceptible to myriad effects that include anthropogenic changes to watersheds, seasonal changes, and natural disturbances such as fire and drought. These influences can result in stochastic effects that affect the aquatic macroinvertebrate community and as such, the community of macroinvertebrates is considered a sensitive indicator of stream health. Altered composition of the macroinvertebrate community can be used to diagnose recovery of aquatic ecosystems through systematic tracking. Biomonitoring can aid in capturing year-to-year variability of macroinvertebrate communities and track biological responses of largescale disturbances, such as wildfire and flooding (Verkaik et al. 2015) as well as localized disturbances to riparian areas such as grazing and logging (Campbell and Doeg 1989). These communities assimilate the effects of the environmental stressors and serve as indicators of change in stream morphology and habitat heterogeneity. Biomonitoring within an ecoregion can allow one to compare and contrast how communities within that ecoregion are differentially affected by disturbance and to monitor ecosystem recovery or succession (Minshall et al. 2001). To make sound management decisions, it is essential to understand the impacts of disturbances on these freshwater environments to evaluate long-term trends in stream ecosystem health.

In the Jemez Mountains of Sandoval County in northcentral New Mexico, the Valles Caldera National Preserve (Preserve) presents a unique opportunity to characterize the effects of disturbance and subsequent recovery of aquatic macroinvertebrates in streams throughout a montane grassland ecosystem. The Preserve contains low gradient meadow stream systems where a historic lake once resided in a volcanic crater. Within the ancient crater or caldera are remnant valleys (Spanish word *valles*) where headwater streams flow from below caldera rim into these low gradient valley streams and contribute to some of the largest montane wet meadows in the Southern Rocky Mountains Ecoregion (Muldavin and Tonne 2003). The climate of the Southern Rocky Mountain ecoregion is considered semi-arid with winter precipitation primarily from snow (Neely 2001). These high-elevation grassland streams are characterized by large daily and seasonal thermal variability, limited canopy cover, and long temporal and spatial intermittency during severe drought.

In 2011, early snowmelt from a reduced snowpack and high spring and summer temperatures promoted conditions for the largest wildfire in New Mexico recorded history. The Las Conchas wildfire began on June 26, 2011, and, in less than one week, burned one-third of the Preserve (12,156 ha; Dahm et al. 2015). The U.S. Forest Service Burned Area Emergency Response categorized areas throughout the Preserve as having 'high', 'moderate', or 'low' burn severity based on pre- and post- satellite imagery comparisons (Parsons et al. 2010; Figure 7.1). While over 50% of the burn area within the Preserve was categorized as high or moderate burn, some of the landscape within the burn perimeter experienced low or no burn, resulting in heterogeneous fire effects. Within a few weeks after the wildfire was extinguished (September 2011), 24.5 cm of precipitation over several days initiated a series of large flooding events throughout the Preserve (WRCC 2011). The precipitation increased overland and in-channel flow with channel incision and erosional avulsions that exacerbated sediment transport and deposition in some of the more severely burned areas (Dahm et al. 2015; Orem and Pelletier 2015; Reale et al. 2015). In the months following the Las Conchas wildfire, perennial streams exhibited altered hydrophyte community structure (Thompson et al. 2019), dissolved oxygen and pH sags (Dahm et al. 2015), and large nutrient pulses (Sherson et al. 2015). Reale et al. (2015) observed prolonged changes to pH, conductivity, and primary productivity three years post-fire. Seven years post-fire, fish populations remained reduced in streams in closer proximity to the more severely burned areas (Caldwell et al. 2018).

The Las Conchas wildfire provided a unique opportunity to characterize ecological response and recovery from wildfire. As a result, a comprehensive biomonitoring program (post-fire) was built upon an existing biomonitoring (pre-fire) of the aquatic macroinvertebrate stream communities throughout the Preserve. While the wildfire occurred in 2011, we were tasked in 2015 to initiate a more comprehensive biomonitoring program to assess the recovery trajectory of the aquatic macroinvertebrate stream communities throughout the Preserve. Thus, our goal was to characterize spatial and temporal changes reflecting recovery in the aquatic macroinvertebrate community using a suite of ecological metrics pre- and post-fire. We were tasked to continue the biomonitoring program beginning in 2015 and complete the program in 2018. However, from 2004 to 2018, different investigators employed a total of three gear types with two mesh sizes for biomonitoring of the aquatic macroinvertebrate community. We established four objectives which follow in three parts.

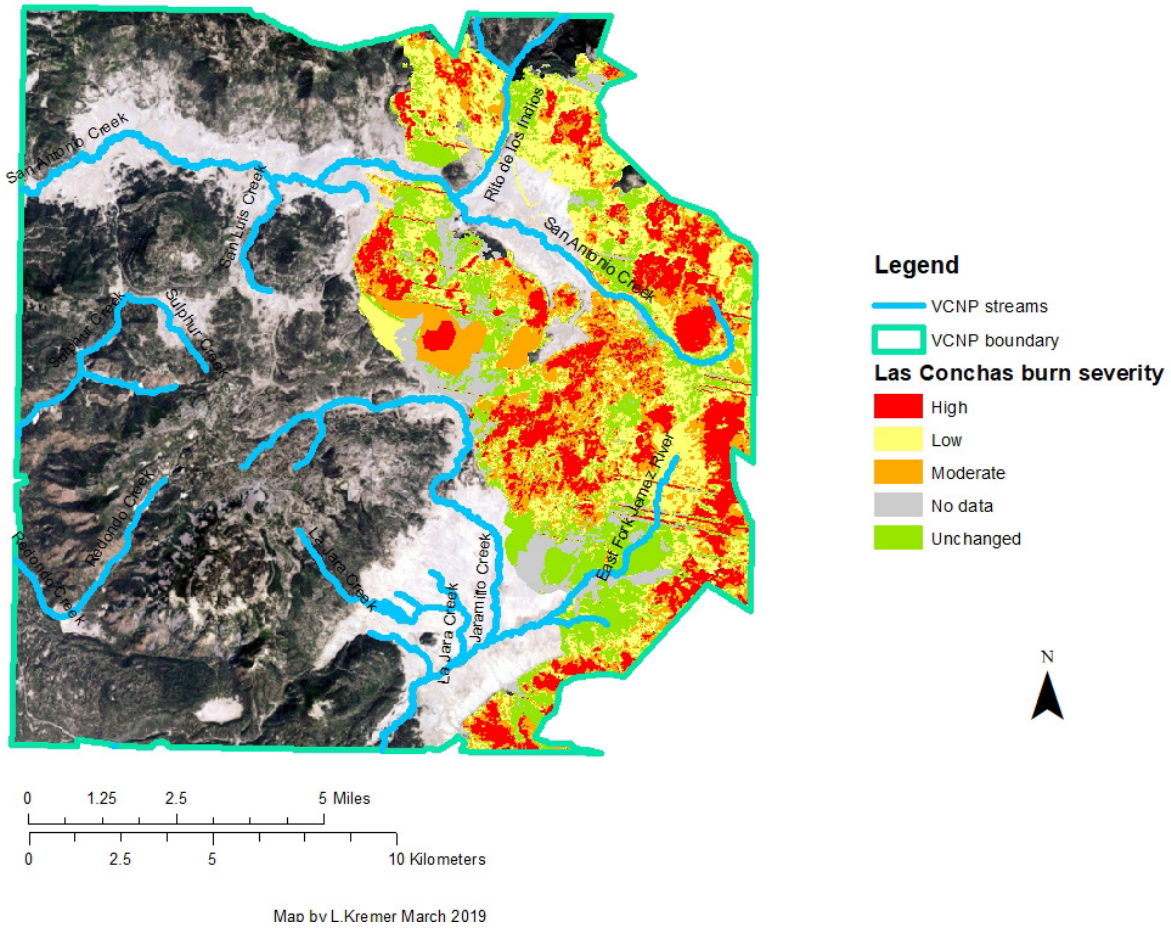



Figure 7.1. The Valles Caldera National Preserve, north-central New Mexico, with burn severity classifications of the 2011 Las Conchas wildfire within the Preserve boundary.

In ***Part 1. Methodology Review and Dataset Selection: Collection Gear-types and Methodologies***, we describe the methodology, analyses, results of the method comparisons, and data standardization among biomonitoring protocols throughout the Preserve to effectively characterize benthic macroinvertebrate assemblage responses to spatial and temporal changes related to wildfire effects. Thus, our first objective tested the feasibility of comparing the three gear types. In other words, if the different gear types selected the benthic macroinvertebrates disproportionately, then we would have to adjust the data prior to analysis. To effectively characterize assemblage responses, our second objective was to compare environmental variables presumed to affect benthic invertebrates and time to recovery. Thus, in ***Part 2. Environmental Drivers of Aquatic Macroinvertebrate Assemblages***, we describe the methodology, analyses, and results of the impact of environmental variables using common composition metrics. In this section, our third objective was to characterize environmental variables that contributed to changes in the benthic macroinvertebrate community across sites and time using general linear models (GLMs). In ***Part 3. Aquatic Macroinvertebrate Assemblage Changes over Time***, we describe the methodology, analyses, and results of assemblage changes throughout the Preserve before and after the Las Conchas wildfire where pre-fire data were available, and post-fire trends over time where comparable pre-fire data were not available. Thus, in ***Part 4. Compare ecological metrics before and after the Las Conchas wildfire*** to determine the impact and time to recovery of the aquatic macroinvertebrate assemblages. This last section establishes a new baseline with recommendations for continuing the monitoring program into the future.

## Methods

*Overview of the Valles Caldera National Preserve*

A volcanic eruption nearly one million years ago resulted in the collapse of several mountains resulting in depressions 300 m below the surrounding terrain. These depressions created the Preserve and its surrounding watershed where streams originate from headwater springs along the Caldera rim and a resurgent dome in the center of the main Caldera. Streams drop in elevation, slope, and substrate size as they enter the valles present at the base of the Caldera. These streams flow through the lower elevation portions of the rim and exit along the western and southern boundaries. Two major drainages occur within the Preserve (Jemez and San Antonio). The East Fork Jemez River and San Antonio Creek represent the main tributaries to the Jemez River in the middle Rio Grande. Under private ownership from the early 1820's through 2000, the area was managed for livestock (sheep and cattle) grazing, timber, and geothermal energy exploration. After federal acquisition in 2000, the Preserve was mandated to maintain a multiple land use policy to protect and preserve the sustained yield of renewable natural resources that included grazing, timber, and recreation while maintaining ecological integrity (Parmenter et al. 2007). As such, inventory and monitoring programs were established throughout the Preserve that included wildlife, upland forests, lowland grasslands, and fish and aquatic health of the streams. In conjunction with fish and aquatic health, a macroinvertebrate biomonitoring program began in 2004 to monitor the long-term trends of grazing practices on water quality and the benthic invertebrate community throughout the Preserve's streams. Monitoring was conducted in streams located within three distinct grassland valles: Valle Toledo, Valle San Antonio, and Valle Grande (Figure 7.2).

*Monitoring Sites- Valle Toledo and Valle San Antonio*

Within the Valle Toledo in the northeast corner of the Preserve, the Rito de los Indios (Indios) originates at 3000 m in elevation as a spring along the north rim of the Valles Caldera (Figure 7.2). It is a first-order perennial stream with a steep gradient and a predominately cobble and boulder bed constrained by a valley on the western side of the Cerro Toledo as it flows south. The riparian vegetation along the headwaters is primarily mixed conifer forest. The gradient reduces to 1.0-1.5% and wetted width of 1.2-5.4 m as the gradient decreases when the Indios reaches the Valle Toledo (USDA 2009). The annual hydrograph is typical for northern New Mexico with peak runoff occurring after snowmelt in spring and low flows occurring in fall-winter. Monsoonal rains in late summer may create episodic flow increases during those events. At the lowest reach or confluence with San Antonio Creek, the discharge varies from 0.1 $m^3/s$ in October to 0.2 $m^3/s$ in May. In the Indios, the dominant substrate varies from coarse gravel in the upper reach to sand in the lowest reach before it joins San Antonio Creek. Several post-fire large flooding events within the Indios valley triggered channel alteration with turbidity, erosion of banks and riparian areas, and increased sediment transport and deposition, all of which are anticipated indirect effects of a severe wildfire (Minshall 2003). The width to depth ratio (bankfull surface width (m)/mean bankfull depth (m), NRCS 2007) was 5.0 immediately post-fire September 2011, but increased to 15.3 (5.55, standard deviation) by 2018. San Antonio Creek emerges from a series of springs and overland flow on the northeastern side of the Preserve in the Valle Toledo (2650 m) (Figure 7.2). Its gradient decreases from 1.5 to 1.0% as it enters the Valle Toledo. Within this valley, the stream increases in size as it flows past an artesian well that was installed in prior ownership. The well contributes sufficient water to increase mean discharge that varies from 0.16 $m^3/s$ in October to 0.48 $m^3/s$ in May. The dominant substrates include cobble and coarse gravel. Sedges and sod-forming grasses dominate the meadow riparian communities.

The San Antonio Creek headwaters, in particular, lie adjacent to the Las Conchas burn scar. Though no major flood events occurred within the catchment, regrowth of the riparian grassland meadows occurred following the Las Conchas wildfire (R. Parmenter, personal observation). The width to depth ratio was 9.7 in September 2011 (i.e., immediately post-fire) and remained stable throughout seven years post-fire (average 9.7 ± 2.95) by 2018. Downstream of the confluence of the Indios with San Antonio Creek, the San Antonio Creek enters the larger open meadow valley system called the Valle San Antonio. In contrast to the Valles Toledo, no post-fire large flooding events occurred in the Valle San Antonio (i.e., channel alteration, erosion of banks and riparian areas).

As the San Antonio Creek traverses the Valle San Antonio, the stream remains a low gradient second-order stream that flows east to west across the northern portion of the Preserve and drains into the Jemez River south of the Preserve boundary. Prior to leaving the Preserve, San Antonio Creek is bordered by a well-developed floodplain containing 2-3 km wide grassland meadows and mixed conifer forests bordering these meadows. Meandering grassland streams of this type, typically contain deep pools with undercut banks that provide shade and fish habitat. However, long term grazing practices have caused streambanks to collapse, which increased stream turbidity and altered substrate composition of smaller sizes and fines (Muldavin and Tonne 2003). These alterations shift the hydrology of the streams, which results in a wider, shallow, homogeneous stream that receives direct amounts of sunlight throughout daylight affecting thermal conditions. Bank erosion transports fine sediments to pools, transforming ideal fish habitat into long runs followed by long riffles. Western reaches of San Antonio Creek,

distant from wildfire effects, contain deep pools and undercut banks though the stream transitions to wider reaches showing signs of sloughing and erosion with increased width-to-depth ratios as the stream flows west as it leaves the Preserve.

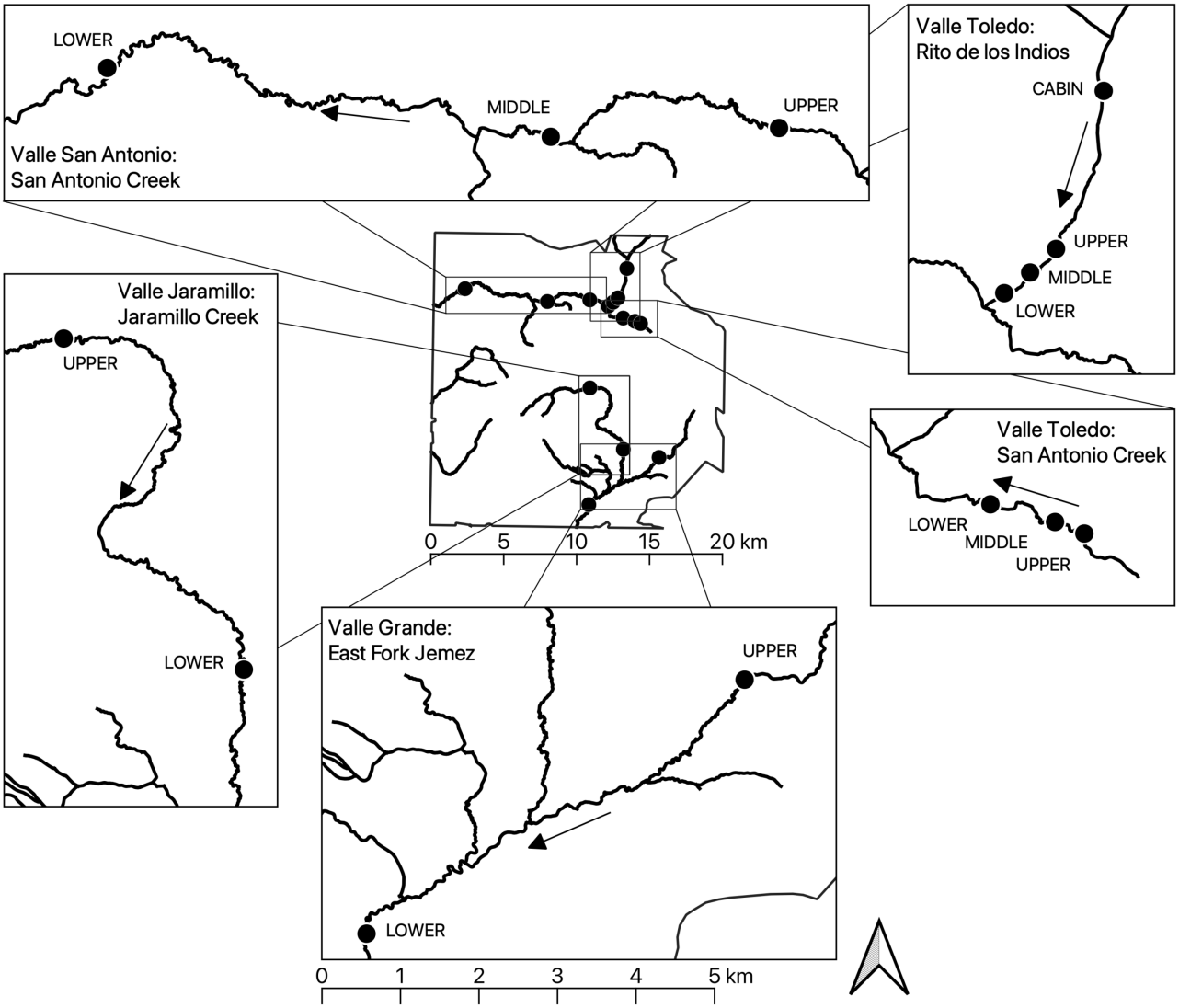



Figure 7.2. Location of aquatic macroinvertebrate community monitoring sites, Valles Caldera National Preserve, New Mexico. Arrows indicate direction of streamflow.

*Monitoring Sites: Valle Jaramillo and Valle Grande*

Within the Jemez River watershed in the Valle Jaramillo, the East Fork of the Jemez River, flows east to southwest where it is joined by Jaramillo Creek. At this point, the East Fork of the Jemez River traverses west to east before joining the Jemez River. Jaramillo Creek, which has headwaters rising from springs on the main resurgent dome in the caldera, is a spring-fed first order tributary to the East Fork Jemez River. The Creek flows through meadows approximately 500 m wide bordered by mixed conifer forests and drains into the Valle Grande. At the upper reaches, Jaramillo Creek is a small, low-gradient stream with a predominately silt and gravel streambed and few to no pools. Mean discharge rates vary from 0.02 $m^3$/s in May to 0.11 $m^3$/s in October near the headwaters (Anderson et al. 2010). As Jaramillo Creek approaches the confluence with the East Fork of the Jemez River, the stream is confined within steep cut banks and predominated by silt in the stream beds 0.5-1.0 m deep (USDA 2009). Jaramillo Creek is perennial with long continuous low gradient runs with few pools or riffles. Approximately 31% of Valle Jaramillo was burned within the lower reaches by the Las Conchas wildfire (Reale 2015).

Within the Jemez River watershed, the East Fork of the Jemez River originates as a spring fed first-order stream on the northeast side of the Valle Grande, an expansive meadow valley containing several ground-fed seeps and streams. The stream begins as a clear, wide, low-gradient with steep banks and evidence of some sloughing from the effects of long-term grazing practices. The benthic substrate in the upper reaches is predominately sand and gravel. Within the Valle Grande, Jaramillo Creek, La Jara Creek, and a series of seeps increases the depth and discharge of the East Fork Jemez as it exits the Preserve's southern boundary. The Las Conchas fire burned 36% of the East Fork Jemez catchment with large areas of severe burn occurring below its confluence with Jaramillo Creek (Reale 2015).

*Part 1. Methodology Review and Dataset Selection: Collection Gear-types and Methodologies*

To evaluate the effects and recovery trajectories of the wildfire on the aquatic macroinvertebrate communities throughout the Preserve, we compared the multiple gear-types and collection methodologies and set forth recommendations for continued biomonitoring of the aquatic macroinvertebrate community. To this end, we compiled six datasets resulting from the biomonitoring of aquatic macroinvertebrate assemblages throughout the Preserve from 2004 to 2018 (Table 7.1). The six datasets spanned four years pre-Las Conchas wildfire (2004-2008), and seven years post-Las Conchas wildfire (2011-2018). Biomonitoring did not take place in 2008, 2009, and 2010. The datasets represent fourteen stream reaches within four sub-catchments or valles (Table 7.1). Sulphur, Redondo, La Jara Creek and upstream reaches of Jaramillo Creek were not affected by the Las Conchas wildfire, but these sites served as reference sites for future pre- and post-assessments.

Despite the different gear-types across years, all samples emerging from the six datasets followed similar collection methodologies and processing. The few exceptions to the standardized protocol are detailed in the Appendices. Briefly stated, collection of the aquatic macroinvertebrates occurred throughout riffle habitats within each site that encompassed the greatest macroinvertebrate diversity. In sites without riffles, areas with the largest substrate (i.e., gravel) or areas of aquatic macrophytes and algae growth were selected. All samples were fixed in 95% ethanol and identified to the lowest taxonomic level with Ephemeroptera, Trichoptera, Plecoptera, Mollusca, Odonata and most Diptera identified to genus (Merritt et al. 2008).

Table 7.1. Biomonitoring timeline of the benthic macroinvertebrate communities on the Valles Caldera National Preserve from 2004 to 2018. Principle investigator (PI), year of collections, seasons (month) of collections, and locations of the sample collections as reflected by the minor drainage (Valle) and the stream system followed by specific sites within the stream. Note that the Valle Toledo does not have exclosure sites to sample. Number of samples are reflected within each site within each stream. Sample method reflects gear-type. Chironomidae (Diptera) identification describes the lowest taxon.

| PI | Year | Seasons | Location | Number of samples/site | Sample method | Chironomidae identification |
|---|---|---|---|---|---|---|
| Anderson et al. (2010) | 2004-2006 | Spring (May) Fall (Sep) | Valle San Antonio- San Antonio Creek: upper, middle, lower elk exclosure sites<br>Valle Jaramillo- Jaramillo Creek: upper, lower elk exclosure sites<br>Valle Grande- East Fork Jemez: lower elk exclosure site | 3<br>3<br>3 | Surber 500 µm mesh (0.10 $m^2$) | family subfamily |
| Jacobi and Associates | 2005-2007 | Summer (July) Fall (Sep) | Valle San Antonio- San Antonio Creek: upper elk exclosure site<br>Valle Toledo- San Antonio Creek, Rito de los Indios | 3<br>1 | Modified Hess 500 µm mesh (0.067 $m^2$) | family subfamily genus |
| Jacobi and Associates | 2011 | Summer (Aug) | Valle San Antonio- San Antonio Creek: upper elk exclosure site<br>Valle Toledo- Rito de los Indios | 3<br>1 | Modified Hess 500 µm mesh (0.067 $m^2$) | family subfamily genus |
| Jacobi and Associates | 2011 | Fall (Sep) | Valle San Antonio- San Antonio Creek: upper elk exclosure site<br>Valle Toledo- San Antonio Creek, Rito de los Indios | 3<br>1 | Modified Hess 500 µm mesh (0.067 $m^2$) | family subfamily genus |
| Jacobi and Associates | 2012-2015 | Spring (May) Fall (Sep) | Valle San Antonio- San Antonio Creek: upper, middle, lower elk exclosure sites<br>Valle Toledo- San Antonio Creek, Rito de los Indios | 3<br>1 | Modified Hess 500 µm mesh (0.067 $m^2$) | family subfamily genus |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | | Valle Jaramillo- Jaramillo Creek: upper, lower elk exclosure sites | 3 | | |
| | | | Valle Grande- East Fork Jemez: upper, lower elk exclosure sites | 3 | | |
| Kremer and Caldwell | 2016-2018 | Spring (May) Fall (Sep) | Valle Toledo- San Antonio: upper, middle and lower sites | 3 | Commercial Hess 250 µm mesh ($0.083\ m^2$) | family subfamily genus |
| | | | Valle Toledo- Rito de los Indios: cabin, upper, middle, lower sites | 3 | | |
| | | | Valle San Antonio- San Antonio: upper, middle, lower elk exclosure sites | 3 | | |
| | | | Valle Jaramillo- Jaramillo Creek: upper, lower elk exclosure sites | 3 | | |
| | | | Valle Grande- East Fork Jemez: upper, lower elk exclosure sites | 3 | | |

Anderson et al. (2010) pre-burn: 2004-2006

From 2004 to 2006, benthic macroinvertebrates were collected by Anderson et al. (2010) in spring (May) and fall (September) from the Valle San Antonio (San Antonio Creek: upper, middle, lower elk exclosures), Valle Grande (Jaramillo Creek: upper and lower elk exclosures) and the East Fork Jemez (lower elk exclosure) (Table 7.1; Figure 7.2). Three replicates were collected from each site and elk exclosure within each stream using a commercial Surber stream bottom sampler (0.10 $m^2$) and a 500 µm mesh. Each replicate was sorted to remove all invertebrates and 100% of each sample identified to lowest possible taxonomic level with Ephemeroptera, Trichoptera, Plecoptera, Mollusca, and Odonata identified to genus. Most Diptera were identified to family and sub-family levels.

Jacobi and Associates pre-burn: 2005-2007

From 2005 to 2007, benthic macroinvertebrates were collected by Jacobi and Associates of Santa Fe, New Mexico in summer (July) and fall (September) from the Valle Toledo (Rito de los Indios and San Antonio Creek) and the upper elk exclosure of the Valle San Antonio (Table 7.1). Replicates were collected from each stream reach using a Hess stream bottom sampler modified from a 5-gallon bucket (0.067 $m^2$) with a 500 µm mesh. Each replicate was sorted and 100% of all invertebrates captured in each sample identified to lowest possible taxonomic level with Ephemeroptera, Trichoptera, Plecoptera, Mollusca, and Odonata identified to genus. Most Diptera were identified to genus, including Chironomidae.

Jacobi and Associates post-burn: 2011-2015

Immediately after the Las Conchas fire was extinguished in August 2011, Jacobi and Associates began collections within the Valle Toledo in fall (September) 2011 to monitor recovery of the benthic macroinvertebrate communities. In 2012, the scope of the sampling program was expanded to resume sample collection in the elk exclosure sites in the Valle San Antonio and the Valle Grande with the addition of one stream reach in the Valle Grande (Table 7.1). This stream reach was selected in the uppermost reaches of the East Fork of the Jemez approximately 200 m from the stream's perennial origination at a spring because it was near the burn perimeter. Additionally, post-fire sampling periods were shifted from summer (July) to spring (May) and fall (September/October) to coincide with fish surveys.

Kremer and Caldwell, NMSU, post-burn: 2016-2018

An analysis using the Bray-Curtis index suggested that dissimilarity among sample replicates within the Valle Toledo streams had increased since the wildfire in 2011 (Caldwell et al. 2016). Therefore in 2016, Kremer and Caldwell increased the number of sample collections from 1 to 3 in the three monitoring reaches of the Rito and the Rio San Antonio in the Valle Toledo. Furthermore, a fourth monitoring reach (with three sample collections) was added in the headwater of the Indios (Cabin; Table 7.1). This monitoring reach was added as a reference because it was outside the Las Conchas wildfire. Beginning May 2016, samples were collected with a commercial Hess sampler (0.083 $m^2$) with a 250 µm mesh. Rather than duplicating Jacobi and Associates' modified Hess (0.067 $m^2$), we chose a standard commercial Hess collector accepted by the biomonitoring industry (Wildco 16-C22 243 µm Nitex nylon net; Dolphin 250 µm mesh). Preserve staff would have access to the same collector in future biomonitoring efforts.

Samples were elutriated (Appendix A), then subsampled using a mock Caton sub-sampler (a gridded screen that divided the sample into a grid of twenty-four even sample squares (Caton 1991; Barbour 1996). This allowed for random selection of individuals lying in gridded squares until at least 250 individuals were identified. The number of sampled squares was transformed into a percentage that was used to estimate the total number of individuals per sample replicate. Specimens were identified to lowest taxonomic level with Ephemeroptera, Trichoptera, Plecoptera, Mollusca, and Odonata identified to genus. Most Diptera were identified to genus, including Chironomidae. Identification of Chironomidae to the genus level was provided by Timberline Aquatics in Fort Collins, CO, USA (Appendix B).

Collection gear-types and comparisons

The resulting six datasets utilized three gear types with different sampling areas and mesh sizes: Surber (0.10 $m^2$) 500 µm mesh, modified Hess (0.067 $m^2$) 500 µm mesh, commercial Hess (0.083 $m^2$) 250 µm mesh. The shift from Jacobi and Associates' modified Hess with its 500 µm mesh and smaller diameter (0.067 $m^2$) to the commercial Hess with the 250 µm mesh and larger diameter (0.083 $m^2$) was intended to obtain more representative sample of rarer species. A sampling protocol will rarely identify a complete census of the population of interest; however, larger sampling areas have a greater probability of sampling more microhabitats or detecting rare species as richness has an asymptotic relationship with both the area sampled and the sample abundances (Colwell and Coddington 1994). In other words, greater sample size will increase the likelihood of revealing rare species. Thus, we chose to use a 250 µm mesh that would potentially increase the chances of obtaining greater abundances of smaller and rarer organisms.

To compare the effectiveness of the three gear types, we first calculated the difference in taxa densities among samplers and sampling periods (Figure 7.3). We then performed a rarefaction analysis to estimate species diversity among each gear type with standardized abundances. All statistical analyses were performed in R (version 3.2.2; R Project for Statistical Computing, Vienna, Austria) with figures produced using ggplot2 (Wickham 2016).

Before performing any tests, the datasets were standardized. Macroinvertebrate density was estimated for each sample by converting total individual counts to individual counts per 0.10 $m^2$. Then macroinvertebrate densities were aggregated by sample period (summer, fall, spring). To standardize diversity estimates, we had to first address the fact that different datasets had identified macroinvertebrates to different taxonomic levels. For the comparison of gear types, the taxonomic identifications were standardized to the lowest common taxonomic level. For example, even though some datasets identified Chironomidae to genus or species levels, Chironomidae among all datasets were classified to the family level because identification was completed only to the family level in some datasets. Then the density estimate from each dataset for each year and season was plotted to visualize the scope of difference in taxa densities among collection samplers and sampling period (Figure 7.3). This Figure demonstrates an overall increase in sample density (approximately 10-fold increase) after May 2015 which we attributed the change in sampling protocols, namely a reduced Hess mesh size from 500 µm to 250 µm. To explore this further, we paired datasets by mesh size (500 µm versus 250 µm) and by collection sampler type (Surber 500 µm versus Hess 500 µm).

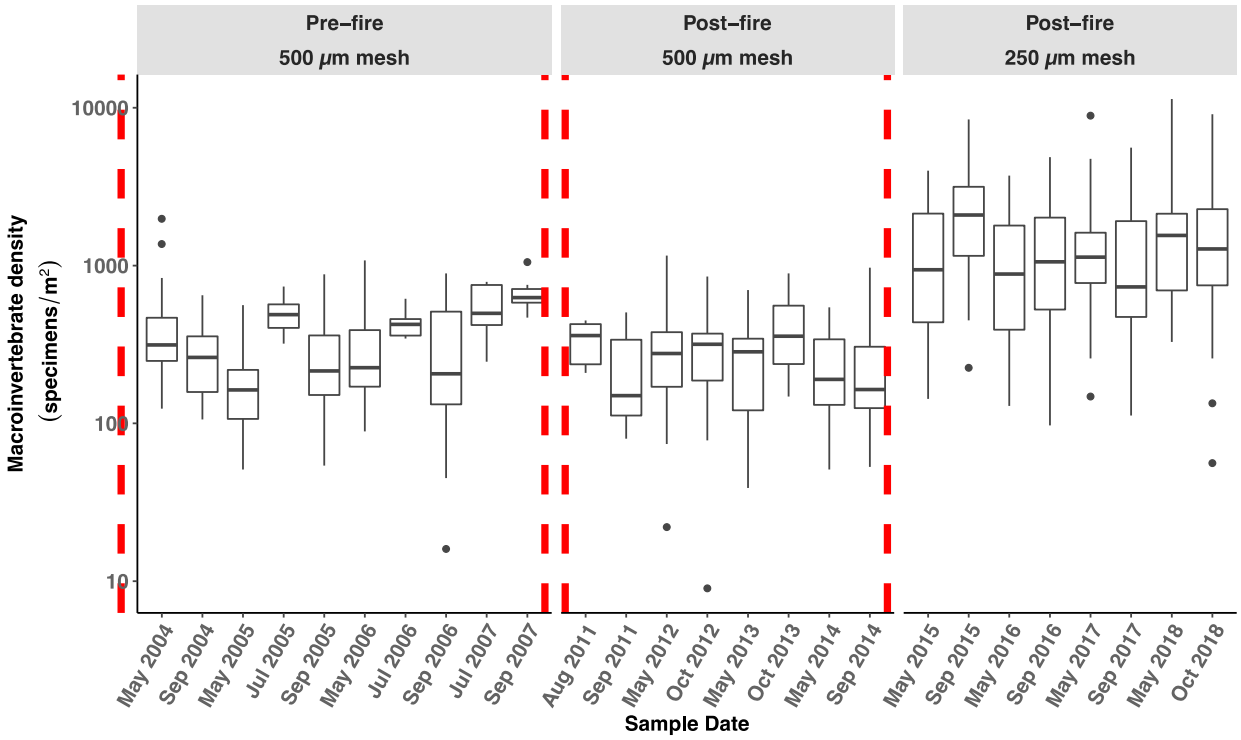



Figure 7.3. Estimated density (macroinvertebrates/$m^2$) on a $\log_{10}$ scale for each Hess replication averaged over all sampled reaches in the Valles Caldera National Preserve. Box plots with whiskers are presented. The horizontal line within each plot represents the median with upper and lower quartiles represented by the whiskers. Outliers are presented as single dots. Increases in sample density after May 2015 may be attributed to reduction in mesh size. Vertical dashed line reflects the timing of the Las Conchas Wildfire.

Hess 500 µm versus Hess 250 µm

We tested for detectable differences between Hess mesh sizes (Hess 500 µm mesh versus 250 µm mesh). Collection methodology and mesh sizes did not overlap across sites; thus, we attempted to ascertain differences in mesh size by comparing samples that had been minimally affected by fire. A comparison of these two methods was generated from a subset of data from the San Antonio Toledo from 2005 to 2007 (500 µm mesh) and from 2016 to 2018 (250 µm mesh). This subset ($n$ = 33) was selected to compare sample periods where fire effects were minimized relative to other reaches in the Preserve as it allows the comparison of pre-fire data to post-fire or 'recovery' data in a stream that was minimally affected by the wildfire. San Antonio Creek within the Valle San Antonio originates within the Valle Toledo below the burn perimeter and though the stream may have been acutely impacted by the fire, in-stream continuous temperature monitoring suggests that the temperature regime from 2016 to 2018 was similar to 2005-2007. Additionally, the riparian in the Valle Toledo grassland meadows visibly recovered within 24 months post-fire (R. Parmenter, personal communication). This subset of data minimizes, but cannot eliminate, the potential effects of time or disturbance on benthic assemblages nor changes and effects of the stream substrate from ash and sediment transport.

We used rarefaction to determine the differences in diversity estimates for the two mesh sizes (Hess 500 µm mesh versus Hess 250 µm mesh). Rarefaction is a technique that performs well with raw count data to estimate species diversity with standardized abundances. Asymptotic diversity estimations are produced from multiple random samples of the dataset and then by plotting average diversity for each sample size. With these estimations, equivalent abundances based on differing sampling effort (Colwell et al. 1994) or sample completeness (Chao and Jost 2012) can be visualized to compare the efficiency of different sampling methods. If discrepancies in total species abundances from the simulated data are due to disparate sampling techniques, the use of rarefaction would aid in the standardization of abundances across samples. This method assumes, however, that differences in abundance among samples are due to differences in sample size, rather than as a response to an environmental parameter (Colwell et al. 2012, Chao and Jost 2012, Chao et al. 2014). Differences in abundance due to environmental parameters will affect diversity estimates and confound comparison across sample methods.

We estimated Hill numbers for species richness, Shannon diversity, and Simpson diversity for the two mesh sizes using the *iNEXT* function in the *iNEXT* package (Chao et al. 2014, Hsieh et al. 2016). Sample-based rarefaction and extrapolation curves were generated to determine how diversity increases with increasing density for each of the methods. All rarefaction curves were plotted with up to 1000 individuals/$m^2$, which adequately reflected the asymptote of each sample type, and 500 bootstrap iterations were used to estimate 95% confidence intervals. In rarified curves, non-overlapping 95% confidence intervals indicate detectable differences at a level <5% (Chao and Jost 2012, Chao et al. 2014).

Surber 500 µm versus Hess 500 µm

We compared assemblage structure between sampling methods with the same mesh sizes (Surber 500 µm mesh versus Hess 500 µm mesh) to explore differences in sample methods without the effect of sampling period. Incidentally, the datasets contained a period from May 2005 to September 2006 ($n$ = 4) where Anderson et al. (2010) and Jacobi and Associates collected samples from the San Antonio upper exclosure using a500 µm Surber and a 500 µm Hess, respectively. We used rarefaction to determine the differences in diversity estimates for the

two gear types (Surber 500 µm mesh versus Hess 500 µm mesh) using the same methods as stated above.

Results of gear type comparisons

Average benthic macroinvertebrate densities declined across all study sites post-fire and then increased in 2015. While this could indicate an increase in macroinvertebrate abundance within the Preserve beginning May 2015, it was most likely due to a reduction in Hess mess size which resulted in an increase in species abundance and appearance of rarer species within our sample collections. However, benthic macroinvertebrate densities declined from September 2011 (immediately after the fire) to September 2014 (post-fire) before the Hess 250 µm mesh was utilized. Thus, the changes in densities pre-fire compared to post-fire (2011-2014) suggests that beginning in 2015, both methodology (Hess mesh sizes) and environmental variables may have confounded our ability to identify benthic macroinvertebrate density changes specifically due to fire.

*Hess 250 µm versus Hess 500 µm*

Rarefaction curves suggests that the smaller 250 µm mesh resulted in greater assemblage diversity in all three ecological indices (species richness, Shannon's diversity, Simpson's diversity) (Figure 7.4, Table 7.2).

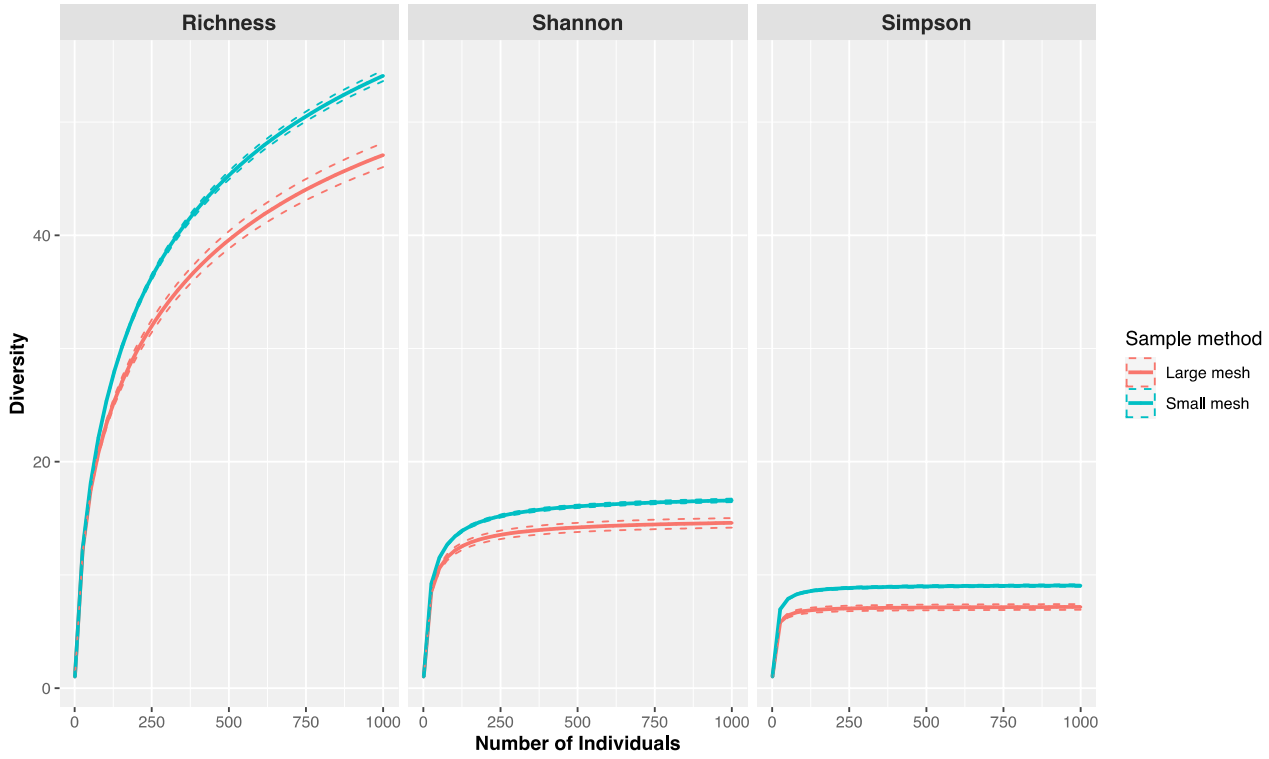



Figure 7.4. Sample-based rarefaction curves reflecting diversity for the ecological metrics of Richness, Shannon and Simpson diversity. Diversity is scaled by sample density (number of individuals/$m^2$) for Hess 500 µm (large mesh) and Hess 250 µm (small mesh) for samples collected in San Antonio Creek of the Valle Toledo from 2005 to 2007 (pre-fire) and from 2016 to 2018 (post-fire). Solid lines represent the mean estimated diversity for each density and each pair of dashed lines represents the corresponding upper and lower 95% confidence intervals. The

smaller mesh demonstrates when Chironomidae is classified to genus (no standardization of taxonomic effort needed). All rarefaction curves were plotted to present 1000 individuals/$m^2$ after 500 bootstrap iterations to estimate 95% confidence intervals. Note that the smaller mesh captures greater richness and diversity.

Table 7.2. A comparison of richness, Simpson and Shannon diversity indices for sample sizes ranging from 103 to 1000 individuals with 95% confidence intervals. There are no overlapping confidence intervals for any of the extrapolated sample sizes. Differences in diversity estimates between the two Hess mesh sizes may be due to differences in environmental parameters between sampling periods or gear type.

| | | Large mesh (500 µm) | | | Small mesh (250 µm) | | |
|---|---|---|---|---|---|---|---|
| Index | Sample Size | Index Value | Lower CI | Upper CI | Index Value | Lower CI | Upper CI |
| Richness | 103 | 23.3 | 23.0 | 23.7 | 25.4 | 25.2 | 25.5 |
| | 205 | 29.9 | 29.4 | 30.4 | 33.9 | 33.6 | 34.0 |
| | 513 | 39.9 | 39.1 | 40.7 | 45.6 | 45.3 | 46.0 |
| | 743 | 43.9 | 43.0 | 44.9 | 50.4 | 50.0 | 50.8 |
| | 1000 | 47.1 | 46.0 | 48.2 | 54.1 | 53.6 | 54.6 |
| Simpson | 103 | 12.2 | 11.9 | 12.5 | 13.4 | 13.3 | 13.5 |
| | 205 | 13.3 | 12.9 | 13.6 | 14.9 | 14.7 | 15.0 |
| | 513 | 14.2 | 13.8 | 14.6 | 16.1 | 15.9 | 16.2 |
| | 743 | 14.4 | 14.0 | 14.9 | 16.4 | 16.2 | 16.5 |
| | 1000 | 14.6 | 14.2 | 15.0 | 16.6 | 16.4 | 16.7 |
| Shannon | 103 | 6.8 | 6.6 | 7.0 | 8.5 | 8.4 | 8.6 |
| | 205 | 7.0 | 6.8 | 7.2 | 8.9 | 8.7 | 8.9 |
| | 513 | 7.1 | 6.9 | 7.4 | 9.0 | 8.9 | 9.1 |
| | 743 | 7.2 | 6.9 | 7.4 | 9.0 | 8.9 | 9.1 |
| | 1000 | 7.2 | 6.9 | 7.4 | 9.1 | 9.0 | 9.2 |

*Surber 500 µm versus Hess 500 µm*

Using Hess subsamples ($n$ = 4), rarefaction curves suggest that the large mesh reflects greater assemblage Shannon and Simpson diversity sizes extrapolated to 1000 individuals (Figure 7.5, Table 7.3). However, richness estimates are similar for both gear types.

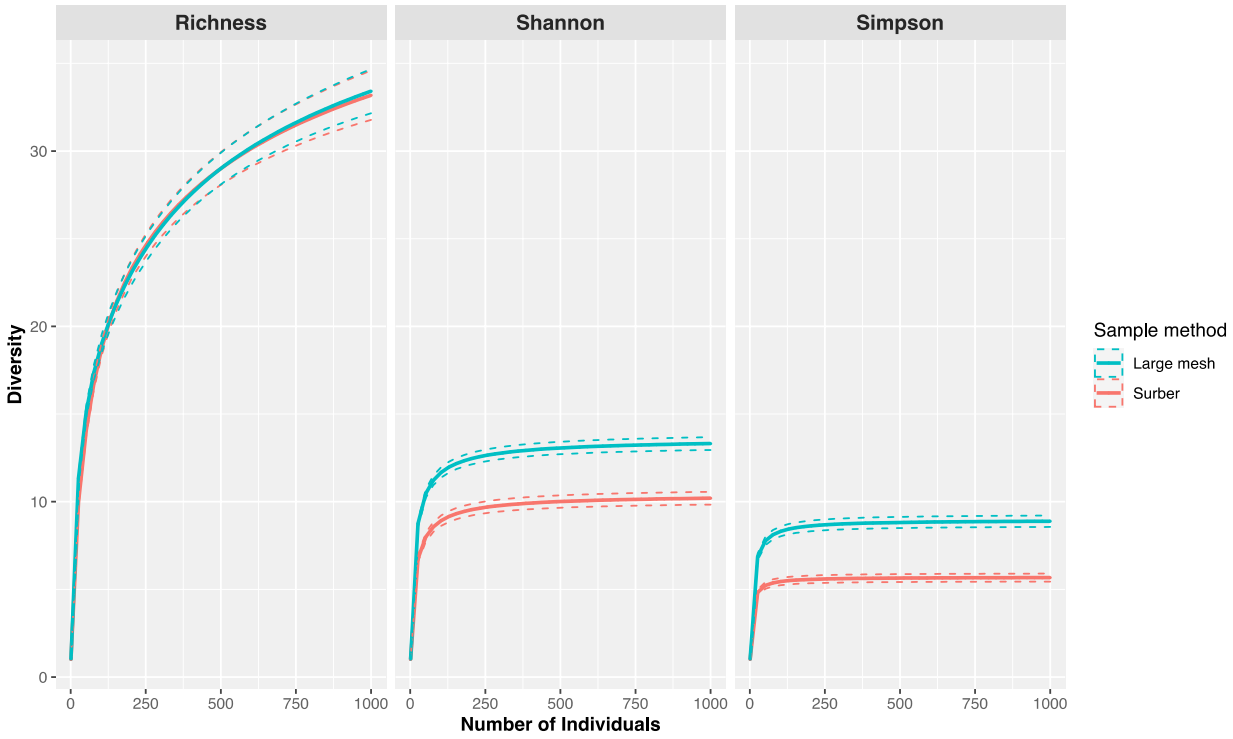



Figure 7.5. Sample-based rarefaction curves reflecting diversity scaled for the ecological metrics of Richness, Shannon and Simpson diversity. Diversity is scaled by sample density (number of individuals/m$^2$) for Surber 500 µm (Surber) and the Hess 500 µm mesh (large mesh) for samples collected from San Antonio Creek Upper Elk Exclosure (2005-2006). Solid lines represent the mean estimated diversity for each density and each pair of dashed lines represents the corresponding upper and lower 95% confidence intervals. The curves suggest that for densities above 250 individuals/m$^2$, the Hess 500 µm mesh sampler collected greater assemblage diversity in two estimated metrics (Shannon and Simpson diversity), than the Surber 500 µm mesh sampler. However, taxa richness is comparable across the two mesh sizes.

Table 7.3. A comparison of richness, Simpson and Shannon diversity indices for sample sizes ranging from 103 to 1000 individuals with 95% confidence intervals. Richness estimates suggest that the 500 µm mesh in both Hess and Surber samplers capture similar numbers of taxa. However, differences in Shannon and Simpson diversity estimates between the gear types suggest that the Hess captures more even proportions of taxa as reflected by higher index values.

| | | Hess 500 µm | | | Surber 500 µm | | |
|---|---|---|---|---|---|---|---|
| Index | Sample Size | Index Value | Lower CI | Upper CI | Index Value | Lower CI | Upper CI |
| Richness | 103 | 19.0 | 18.5 | 19.5 | 18.7 | 18.3 | 19.1 |
| | 205 | 23.1 | 22.4 | 23.8 | 23.3 | 22.8 | 23.9 |
| | 513 | 29.2 | 28.3 | 30.1 | 29.2 | 28.2 | 30.1 |
| | 743 | 31.6 | 30.5 | 32.6 | 31.4 | 30.3 | 32.6 |
| | 1000 | 33.4 | 32.2 | 34.7 | 33.2 | 31.8 | 34.6 |
| Simpson | 103 | 11.7 | 11.4 | 12.0 | 8.9 | 8.6 | 9.2 |
| | 205 | 12.5 | 12.1 | 12.8 | 9.5 | 9.2 | 9.9 |
| | 513 | 13.1 | 12.7 | 13.4 | 10.0 | 9.7 | 10.4 |
| | 743 | 13.2 | 12.9 | 13.6 | 10.1 | 9.8 | 10.5 |
| | 1000 | 13.3 | 13.0 | 13.7 | 10.2 | 9.8 | 10.6 |
| Shannon | 103 | 8.3 | 8.0 | 8.6 | 5.5 | 5.2 | 5.7 |
| | 205 | 8.6 | 8.3 | 8.9 | 5.6 | 5.4 | 5.8 |
| | 513 | 8.8 | 8.5 | 9.1 | 5.6 | 5.4 | 5.9 |
| | 743 | 8.9 | 8.5 | 9.2 | 5.7 | 5.4 | 5.9 |
| | 1000 | 8.9 | 8.6 | 9.2 | 5.7 | 5.4 | 5.9 |

*Part 2. Environmental Drivers of Aquatic Macroinvertebrate Assemblages: Collection of Environmental Parameters*

Multi-parameter sondes (YSI Inc., Model 6920, Yellow Springs, OH, USA) were deployed throughout the study area to obtain continuous in-stream temperature (ºC), dissolved oxygen (mg/L), pH, turbidity (NTU) and conductivity (mS/cm) at 15-minute intervals prior to and after the wildfire. Methods used to collect continuous discharge and water quality data can be found in Dahm et al. (2015) and Reale et al. (2015). Temperature was compiled and validated using Aquarius Workstation 3.3 (Aquatic Informatics, Vancouver, British Columbia, Canada). From a review of the pre-fire (1 July 2005 - 31 October 2007) and post-fire (6 July 2011 - 22 October 2018) data, temperature metrics relevant to biological processes of aquatic ectotherms were selected to include daily maxima, daily minima, daily mean, and daily range (the difference between daily maxima and daily minima) (Koontz 2018). Pre- and post-fire temperature were averaged into daily (i.e., 24 h) means, mean-maxima, mean-minima, and mean-range. Then temperature metrics and stream variables (dissolved oxygen, pH, conductivity, and turbidity) were averaged across time to generate an environmental snapshot for each collection date. For spring samples, daily means for the month of May were averaged ($n$ = 31) into a 'spring' mean for each year that invertebrate data was collected to compare to May samples because continuous in-stream monitoring is typically resumed at the end of April. June and July daily means were averaged across both months to generate one 'summer' mean ($n$ = 61) for each year of invertebrate data and daily August and September means were averaged for 'fall' samples ($n$ = 61) for each year that invertebrates were sampled in the fall. The Western Regional Climate Center provided data for climate stations maintained by the Preserve (WRCC 2020). Daily summaries of wind speed and direction, net radiation, air temperature, humidity and precipitation were selected from four stations, which were in close proximity to the biomonitoring sites.

2. a. Influence of environmental drivers on aquatic macroinvertebrate assemblages: Statistical analyses

Random forest regression models were generated to determine the relative influence of each of the environmental drivers on each response using R statistical software (R Core Team 2015; version 3.2.2). Responses tested were macroinvertebrate diversity (Shannon diversity), density (species richness), and composition (relative abundances of each taxa and of each feeding group). Environmental drivers selected for causal relationships in our models included variables from the multi-parameter sondes and weather stations, and the number of years post-fire. We also designated site as a factor in our analysis to assess site-specific responses for each of the numeric and continuous biotic indices or compositional metrics (response variables). Before developing each model, redundant variables were removed when identified as multicollinear by the *multi.collinear* function of the *rfUtilities* package (Evans and Murphy 2019). The final variables for the model were selected from the remaining variables using the *rf.modelSel* function based on variable importance. Lastly, after correlations of remaining variables were reviewed and, in instances where correlations ($r^2$) between variable pairs were greater than 0.7, the least important variable (as indicated by *rf.modelSel*) was removed.

The random forest algorithm estimates the importance of variables using decision trees. It is a non-parametric technique that does not make assumptions about the residuals of the model. The algorithm selects bootstrapped datasets from the original data and generates decision trees for each of the bootstrapped samples where, for each branch of the regression tree, predictors are randomly sampled. The "out-of-bag" (*oob*), or un-bootstrapped portion of the original dataset is

then used to measure the accuracy of the forest using the proportion of the *oob* samples that were correctly predicted by the forest. Incorrectly classified *oob* is classified as *oob* error or prediction error. In regression models, this is measured by the root mean square error (RMSE), which corresponds to the average difference between the observed known values of the outcome and the predicted value by the model. Variable importance is estimated by the prediction error increase when *oob* data for a given variable is permuted while all other variables are left unchanged (Liaw and Wiener 2002). Datasets were modeled using selected variables by implementing the *randomForest* function of the *randomForest* package (Liaw and Wiener 2002). For each model, the random forest validation statistics were reviewed, including the RMSE and the percentage of variation explained by the model.

2.b. Influence of environmental drivers on aquatic macroinvertebrate assemblages: Results

A total of 45 models were evaluated to assess the relative importance of environmental drivers on macroinvertebrate assemblages. The models explaining the greatest amount of variation and least *oob* error included the feeding guild, richness (5 indices), Shannon diversity, and proportion of Plecoptera. Across all models, the most important environmental variables included dissolved oxygen concentration, turbidity, and minimum daily temperature, though the nature of the correlations between each of these metrics and the response differs among feeding groups (Table 7.4). Our models suggested that Shannon diversity was partly driven by turbidity, dissolved oxygen, and turbidity. While the models using proportion of Plecoptera as a response performed well for both Hess 250 µm and Hess 500 µm mesh samples, the influence of the environmental variables differed between the sampler type (Hess versus Surber). Additionally, site was influential in the models among the Hess 250 µm mesh samples and suggests that other environmental variables were not accounted for throughout the study sites.

2.c. Aquatic macroinvertebrate assemblage changes over time: Data selection and comparisons

In this section, we illustrate trends in the diversity of macroinvertebrate assemblages over time including a comparison of pre- and post-wildfire communities to determine the extent and duration of wildfire effects on diversity. In part 1 of this report, our rarefaction analysis determined that only samples collected by Jacobi and Associates could be compared to examine the effects of the Las Conchas wildfire on macroinvertebrate assemblages. Additionally, pre-fire sampling for this subset of data occurred in mid-summer and fall, while post-fire sampling occurred in spring and fall. Therefore, this section is presented in two parts: 1) comparisons of diversity in streams containing pre- and post-fire data determined to be comparable, and 2) comparisons of diversity among reaches and over time in streams that did not have comparable pre- and post-fire data.

2.c.i. Pre-and post-wildfire comparisons of diversity

This section includes data collected from 2005 to 2014 by Jacobi and Associates in the Rito de los Indios, the San Antonio Creek in the Valle Toledo, and the Upper Elk Exclosure on San Antonio Creek in the Valle San Antonio. To eliminate seasonal effects on pre-and post-fire comparisons, only fall samples were used to characterize the effects of wildfire on benthic assemblages. Generalized linear models were used to compare macroinvertebrate assemblage structure before and after wildfire. Additionally, univariate analysis was used to identify specific taxa abundances that changed from pre-fire abundances. There were no long-term detectable

effects of wildfire on macroinvertebrate assemblages, though interannual variability was observed in streams proximal to the wildfire.

2.c.ii. Pre- and post-wildfire comparisons of diversity: *Rito de los Indios in the Valle Toledo*

Generalized linear models reflected significant differences between pre- and post-fire macroinvertebrate assemblages (deviance = 301.8, $P = 0.001$) collected with the Hess 500 µm mesh. This suggests rapid community responses to varying environmental factors. Of note, diversity of the macroinvertebrate assemblage post-fire within the Rito de los Indios was highly variable among years (Figure 7.6) suggesting that post-fire episodic events affected benthic assemblages. To eliminate the effects of season on assemblage structure, fall samples were used to visually compare pre- and post-fire diversity. Taxa that were detectably different between fall pre- and post- fire assemblages from 2011 through 2014 are compiled in Table 7.5. When assemblages were evaluated according to feeding group using the Hess 500 µm mesh in fall samples, a generalized linear model detected changes in assemblage between pre- and post-fire samples (*deviance* = 22.42, $P = 0.001$). The group with the greatest influence on composition change were shredders (*deviance* = 6.40, $P = 0.001$) and scrapers (*deviance* = 4.02, $P = 0.051$). A stacked bar plot visualizes changes in proportions of feeding groups in fall samples from 2005 to 2014 (Figure 7). While the proportion of feeding groups is variable among all samples, an increase in predators and gatherers corresponded to an increase in diversity one-year post-fire (October 2012).

2.c.iii. Pre-and post-wildfire comparisons: *San Antonio Creek in the Valle Toledo*

General linear models reflected significant differences between fall pre- and post-fire assemblages (deviance = 343.0, $P = 0.001$) collected with the Hess 500 µm mesh. To eliminate the effects of season on assemblage structure, fall samples were used to visually compare pre- and post-fire diversity. A decline in diversity from 2006 to 2007 (Figure 7.8), may be attributed to drought conditions. Samples collected in September 2011 followed severe rain and scouring events throughout the Preserve (*see* Introduction). Taxa were detectably different between fall pre- and post-fire assemblages from 2011 to 2014 (Table 7.5). When assemblages were evaluated according to feeding group using the Hess 500 µm mesh, a generalized linear model detected changes in assemblage between pre- and post-fire samples (*deviance* = 1213.6, $P = 0.001$). Univariate analysis demonstrated that the groups with the greatest influence on composition change were gatherers (*deviance* = 316.6, $P = 0.001$), scrapers (*deviance* = 675.4, $P = 0.001$), and predators (*deviance* = 218.9, $P = 0.001$). A stacked bar plot visualizes changes in proportions of feeding groups from 2005 to 2014 (Figure 7.9), which reflected an increase in filterers and gatherers post-fire that was coupled with a decrease in scrapers.

Table 7.4. Model output and top two environmental variable of each model as indicated by the percent increase in mean standard error (*%IncMSE*) for each variable when it is randomly permuted (not a rank of relative correlation with the response variable). Correlations with the response ($r^2$) demonstrate the direction of the relationship between the driver and the response. NA reflects information not available.

| Variable name | Sampler | Variance Explained | Mean squared of residuals (*RMSE*) | Top predictors in order of *%IncMSE* rank | Predictor correlation with response ($r^2$) |
|---|---|---|---|---|---|
| Shannon Diversity | Both Hess meshes | 48.80 | 10.1 | Mean turbidity<br>Precipitation | -0.13<br>-0.34 |
| Proportion Plecoptera | 500 µm Hess | 44.70 | 0.003 | Min conductivity<br>Average air temperature | 0.36<br>0.68 |
| | 250 µm Hess | 49.07 | 0.014 | Site<br>Min temperature | 0.43<br>-0.46 |
| Scraper Richness | 500 µm Hess | 95.25 | 0.13 | Min conductivity<br>Min turbidity | -0.13<br>-0.27 |
| | 250 µm Hess | 98.89 | 0.005 | Min temperature<br>Min DO | 0.59<br>-0.17 |
| Predator Richness | 500 µm Hess | 95.65 | 0.05 | Maximum DO<br>Min turbidity | 0.43<br>-0.21 |
| | 250 µm Hess | 98.73 | 0.02 | Mean turbidity<br>Min DO | -0.29<br>0.28 |
| Shredder Richness | 500 µm Hess | 98.20 | 0.006 | Max DO<br>Max turbidity | 0.29<br>0.29 |
| | 250 µm Hess | 99.67 | 0.001 | Max turbidity<br>Average air temperature | 0.51<br>0.08 |
| Gatherer Richness | 500 µm Hess | 99.10 | 0.05 | Min turbidity<br>Max conductivity | -0.48<br>0.24 |
| | 250 µm Hess | 99.91 | 0.06 | Mean wind direction<br>Average relative humidity | -0.24<br>0.35 |
| Filterer Richness | 500 µm Hess | NA | NA | ** | |
| | 250 µm Hess | 99.22 | 0.003 | Max turbidity<br>Min air temperature | -0.25<br>0.48 |

**all drivers selected by the *rf.modelSel* are highly correlated

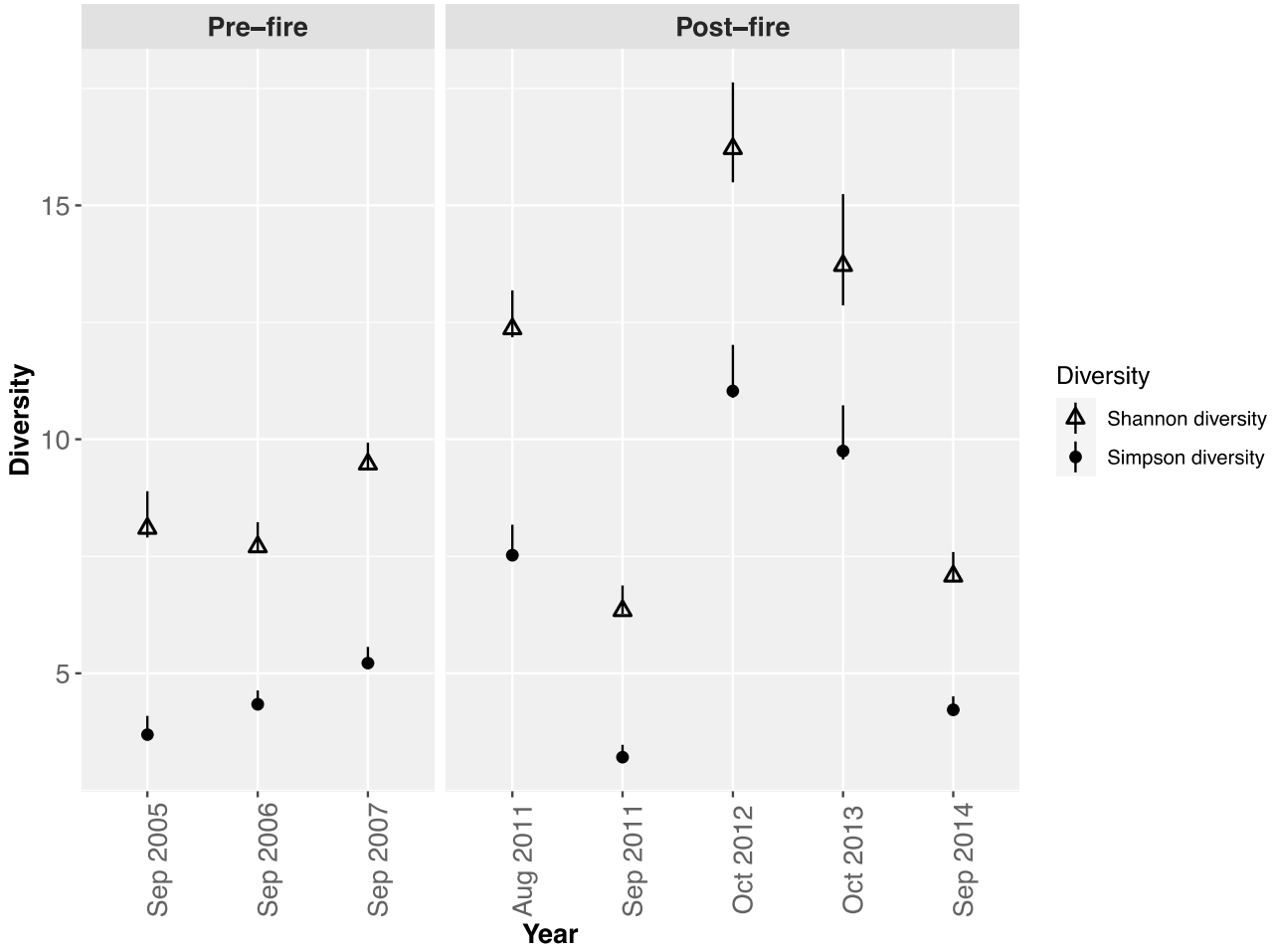



Figure 7.6. Shannon and Simpson diversity estimates for rarified samples (n = 1000) from the Rito de los Indios in the Valle Toledo by sample date. Shannon diversity indicated by open triangles and Simpson diversity indicated by closed circles. Each shape represents the diversity estimate and lines represent the bootstrap 95% confidence interval of each estimate. Annual variation of the diversity estimates suggests that seasonal or annual episodic events influenced the macroinvertebrate assemblage and may be the result of wildfire effects.

Table 7.5. Aquatic benthic macroinvertebrate taxa that reflect detectable differences in abundance between pre- and post-fire sample collections using 500 µm Hess in the Rito de los Indios of the Valle Toledo, Valles Caldera National Preserve.

| Time frame with greatest abundance | Order | Family | Genus | Feeding group | Primary habitat |
|---|---|---|---|---|---|
| Pre-fire | Diptera | Chironomidae | *Pagastia* | collector-gatherer | lotic |
| | Diptera | Tipulidae | *Dicranota* | predator | lotic-erosional and depositional |
| | Ephemeroptera | Ephemerellidae | *Drunella* | scraper | lotic-erosional and depositional |
| | Plecoptera | Nemouridae | *Amphinemeura* | shredder | lotic-erosional |
| | Trichoptera | Brachycentridae | *Micrasema* | shredder | lotic-erosional |
| | Trichoptera | Hydroptilidae | *Hydroptila* | piercer | lotic-erosional and depositional |
| Post-fire | Basommatophora | Planorbidae | *Gyraulus* | scraper | lotic-erosional |
| | Diptera | Simulidae | *Simulium* | collector-filterer | lotic and lentic-erosional |
| | Ephemeroptera | Leptophlebiidae | *Paraleptophlebia* | collector-gatherer | lotic-erosional |
| | Plecoptera | Chloroperlidae | *Suwallia* | predator | lotic-erosional |
| | Haplotaxida | Naididae | | collector-gatherer | lotic-lentic |
| | Hemiptera | Nacouridae | *Ambrysus* | predator | lotic and lentic-erosional |
| | Hygrophila | Physidae | *Physella* | scraper | lotic-erosional |
| | Lepidoptera | Crambidae | *Petrophila* | scraper | lotic-erosional |
| | Rhynchobdellida | Glossiphoniidae | *Helobdella* | scraper | lotic-erosional |
| | Trichoptera | Helicopsychidae | *Helicopsyche* | scraper | lotic and lentic-erosional |

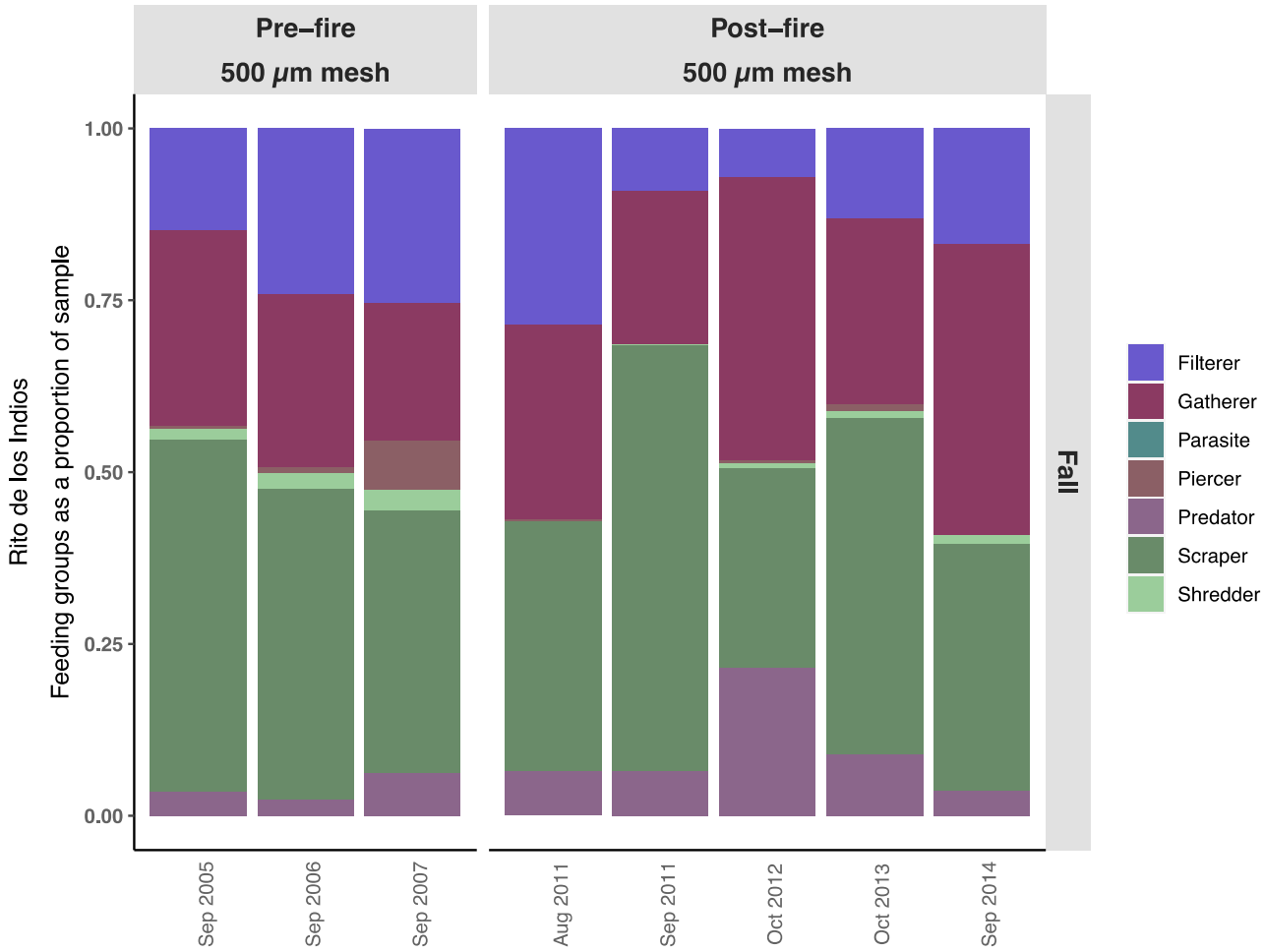



Figure 7.7. Functional feeding group composition in the Rito de los Indios from the Valle Toledo before (2005-2007) and after the Las Conchas fire (2011-2014) reflects variability of feeding groups among all samples. An increase in predators and gatherers and a decrease in scrapers corresponds with an increase in diversity one-year post-fire (October 2012).

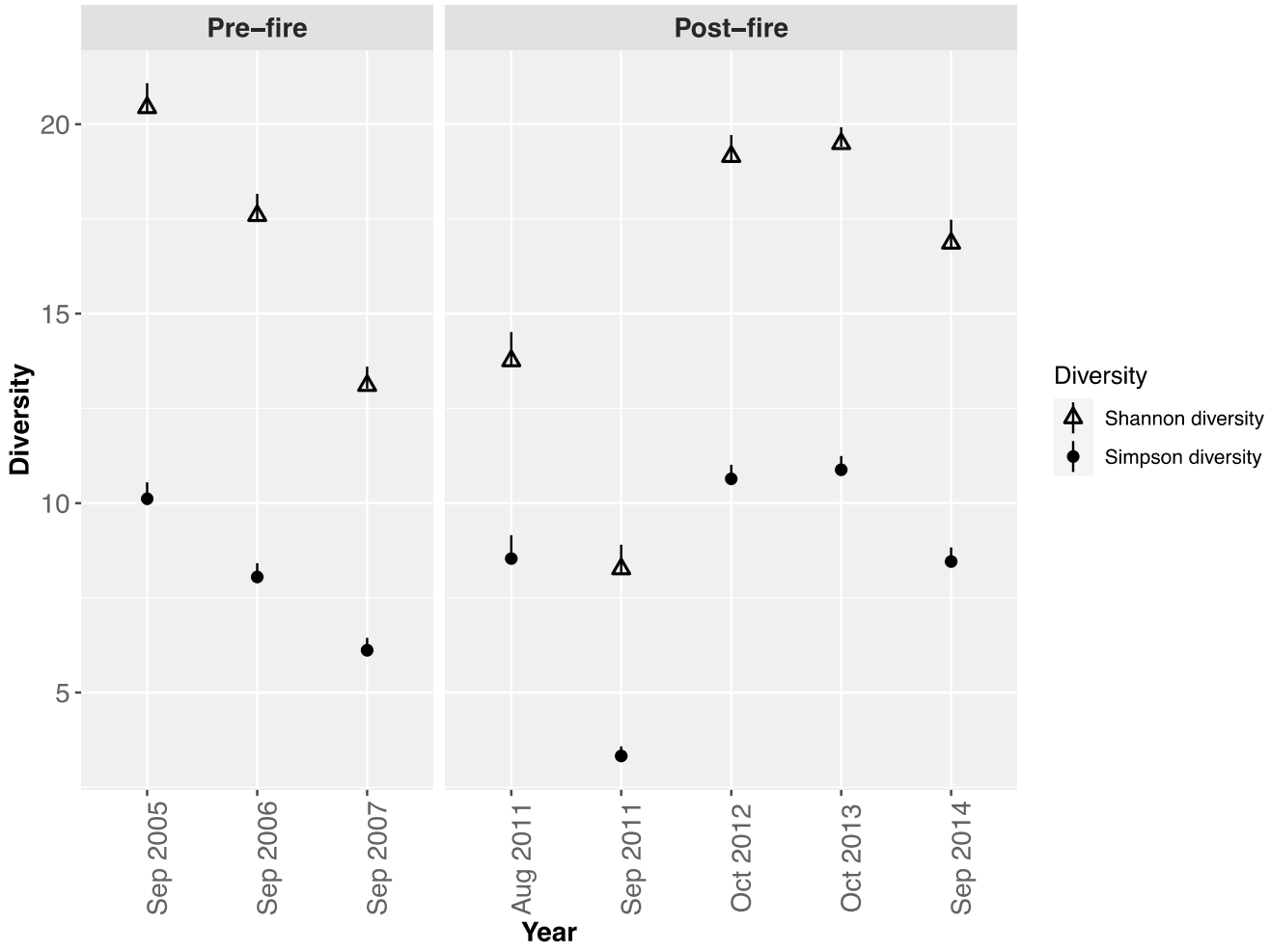



Figure 7.8. Shannon and Simpson diversity estimates for rarified samples (n = 1000) from San Antonio Creek in the Valle Toledo by sample date. Shannon diversity indicated by open triangles and Simpson diversity indicated by closed circles. Each shape represents the diversity estimate and lines represent the bootstrap 95% confidence interval of each estimate. Annual variation of the diversity estimates suggests that seasonal or annual episodic events influenced the macroinvertebrate assemblage and may be the result of fire effects. Diversity estimates vary greatly by year, with reduced diversity evident in September 2011 immediately following post-fire rain events.

Table 7.6. Aquatic benthic macroinvertebrate taxa that reflect detectable differences in abundance between pre- and post-fire sample collections using 500 µm Hess in the San Antonio Creek of the Valle Toledo.

| Time frame with the greatest abundance | Order | Family | Genus | Feeding group | Primary habitat |
|---|---|---|---|---|---|
| Pre-fire | Hemiptera | Nacouridae | *Ambrysus* | predator | lotic and lentic erosional |
| | Diptera | Chironomidae | *Cricotopus* | collector-gatherer | Lotic depositional |
| | Trichoptera | Helicopsychidae | *Helicopsyche* | scraper | lotic and lentic erosional |
| | Diptera | Chironomidae | Orthocladius | collector-gatherer | lentic littoral |
| | Lepidoptera | Crambidae | Petrophila | scraper | lotic erosional |
| Post-fire | Amphipoda | Hyalellidae | Hyalella | collector-gatherer | lotic depositional |
| | Arhynchobdellid | Erpobdellidae | Motobdella | predator | lotic depositional |
| | Trichoptera | Leptoceridae | Oecetis | predator | lotic erosional and depositional |
| | Ephemeroptera | Leptohyphidae | Tricorythodes | collector-gatherer | lotic depositional |
| | Diptera | Simulidae | Simulidae | filterers | lotic erosional |

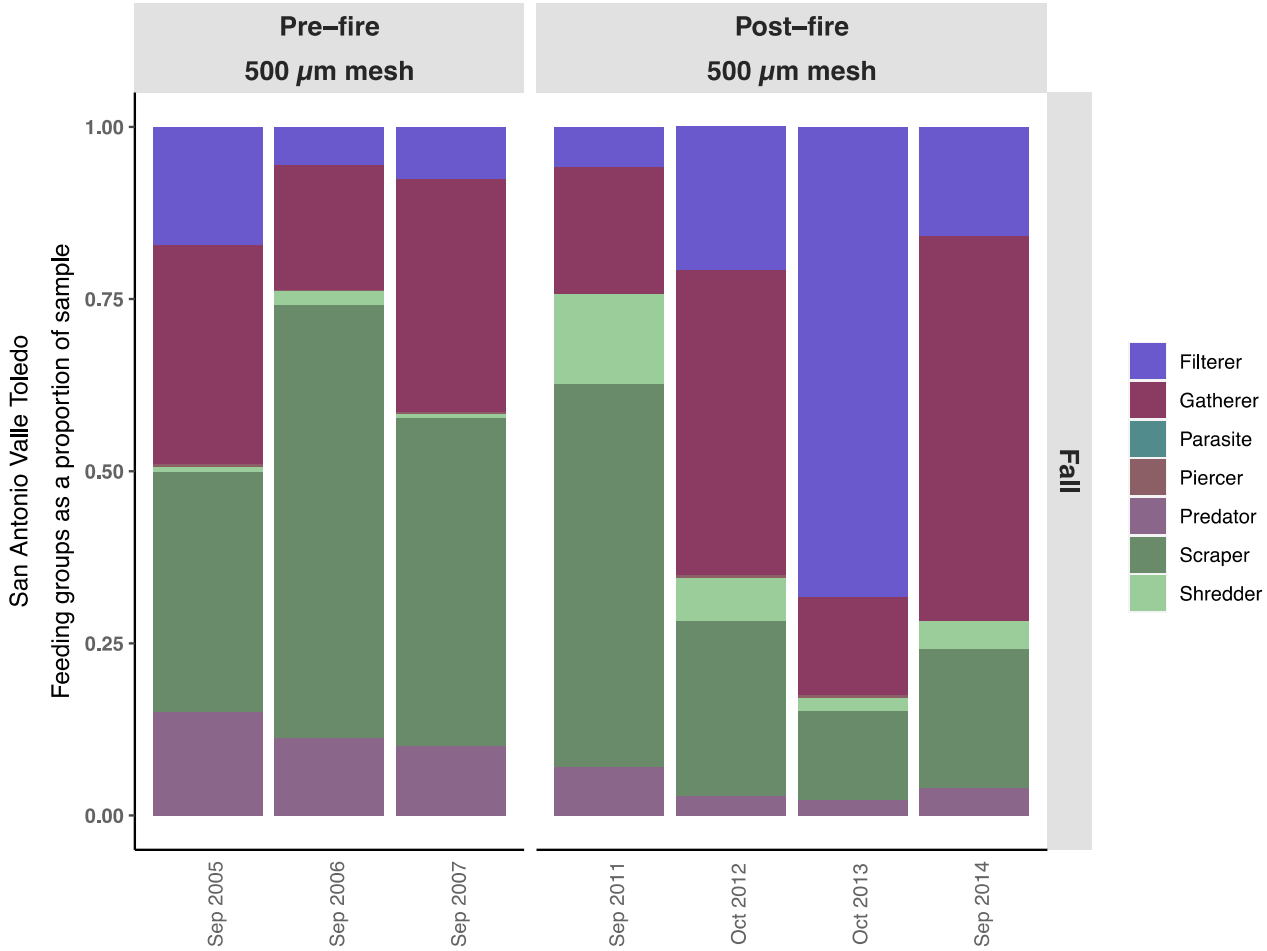



Figure 7.9. Functional feeding group composition in San Antonio Creek of the Valle Toledo pre-fire (2005-2007) and after the Las Conchas fire (2011-2014) reflects an increase in filterers and gatherers post-fire and a decline in scrapers (October 2012). Prior to the wildfire, scrapers were represented by riffle beetle families of Coleopteran Elmidae (*Optioservus* and *Dubiraphia*). These persisted post-fire in high abundances until their decline October 2013, driving the decrease in scrapers. Post-fire filterer increases are largely due to increased abundances of Simulidae (black flies) in 2012 and 2013.

2.c.iv. Pre- and post-wildfire comparisons: *San Antonio Creek in Valle San Antonio (Upper Elk Exclosure)*

The San Antonio Creek within the upper Elk exclosure was sampled pre-fire (2005-2007, summer and fall) and post-fire (2011-2014, spring and fall) with a Hess 500 µm mesh. To eliminate the effects of season on assemblage structure, fall samples were used to visually compare pre- and post-fire diversity (Figure 7.10). Generalized linear models suggested significant differences in the Hess 500 µm mesh samples between pre- and post-fire assemblage (*deviance* = 181.8, *P* = 0.001); however, the Dipteran Simulidae (blackflies) were the only taxa with detectably different densities between pre- and post- fire assemblages (*deviance* = 14.1, *P* = 0.01) as post-fire densities increased (Figure 7.11). Changes in assemblage among fall samples from 2005 to 2014 and between pre- and post-fire samples were evident (*deviance* = 1618.0, *P* = 0.001) in relative densities for filterers (*deviance* = 145.0, *P* = 0.001), gatherers (*deviance* = 347.3, *P* = 0.001), predators (*deviance* = 109.3, *P* = 0.001), and scrapers (*deviance* = 1013.1, *P* = 0.001).

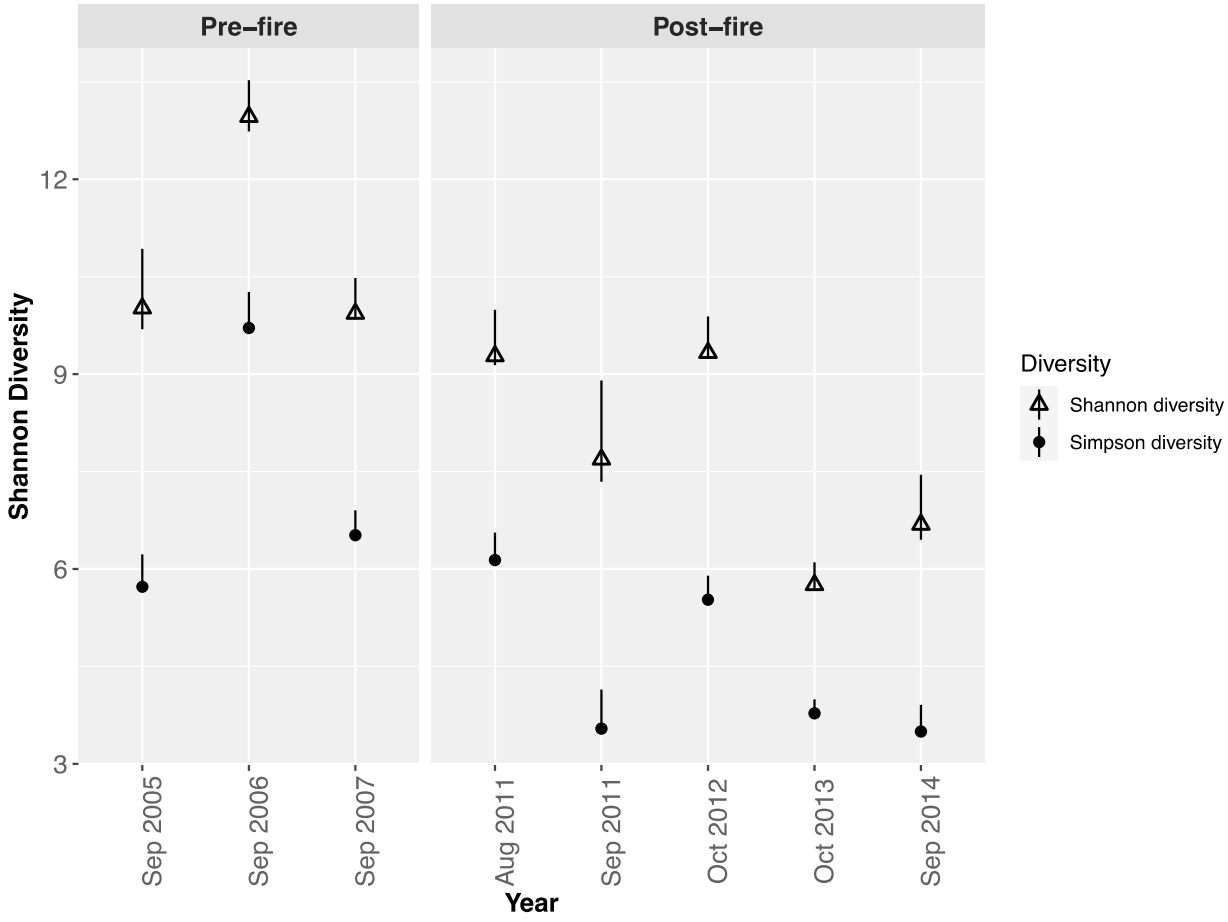



Figure 7.10. Shannon and Simpson diversity estimates for rarified samples (n = 1000) from upper Elk exclosure within San Antonio Creek by sample date. Shannon diversity indicated by open triangles and Simpson diversity indicated by closed circles. Each shape represents the diversity estimate and lines represent the bootstrap 95% confidence interval of each estimate. Annual variation of the diversity estimates suggests that seasonal or annual episodic events influenced the macroinvertebrate assemblage and may be the result of fire effects. Diversity estimates vary greatly by year, with reduced diversity evident in September 2011 immediately following post-fire rain events.

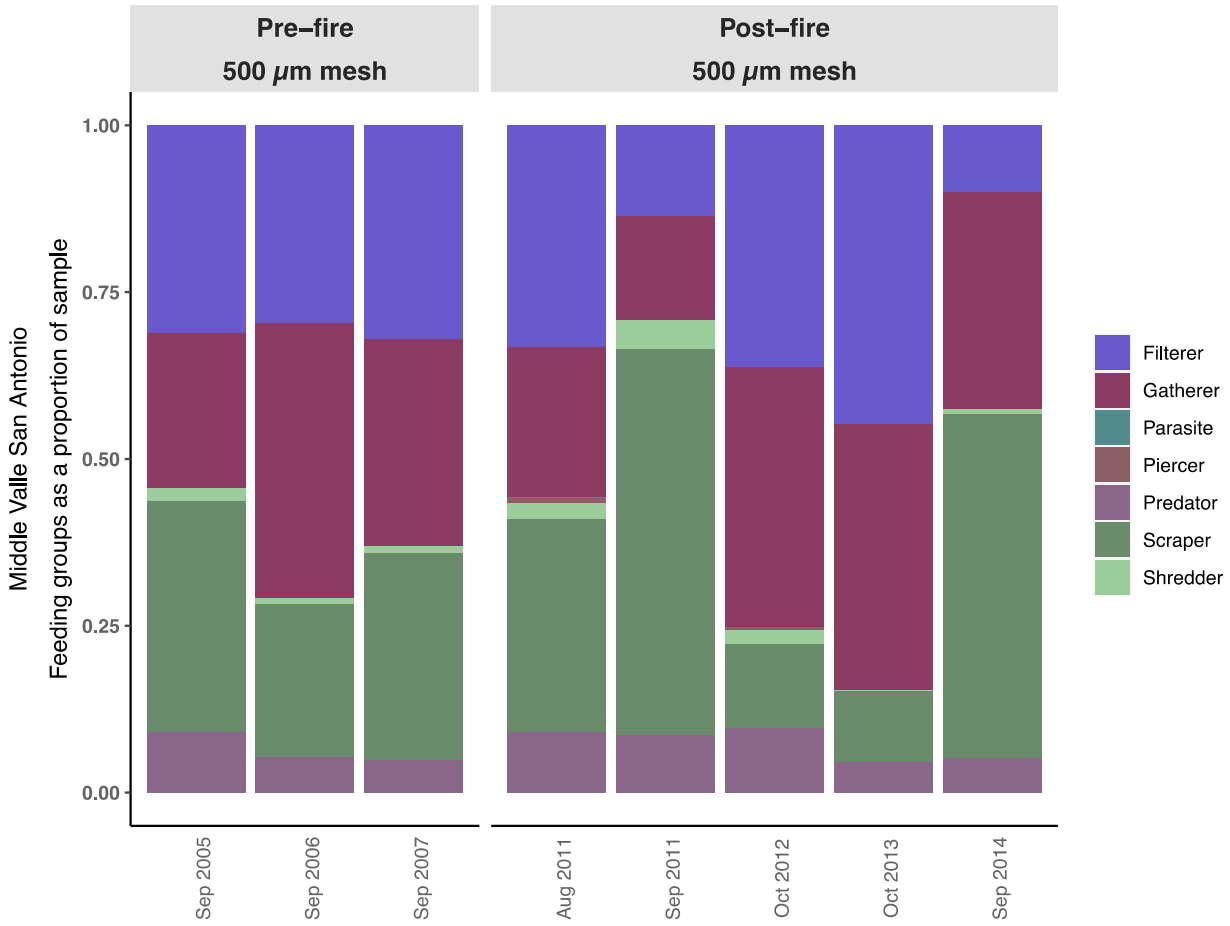



Figure 7.11. Fall functional feeding composition from upper Elk exclosure within in San Antonio Creek of the Valle San Antonio before (2005-2007) and after the Las Conchas wildfire (2011-2014) reflects an increase in functional scrapers and a decline in functional filterers when comparing pre-fire community compositions September 2011 and October 2013 samples. The increases in filterers corresponded with increases in the Dipteran Simulidae (blackflies).

*Part 3. Aquatic Macroinvertebrate Assemblage Changes over Time*

This section evaluates each stream within the biomonitoring program that does not have comparable pre- and post-fire datasets. The purpose of presenting these results is to establish a current baseline of the aquatic macroinvertebrate assemblages for future monitoring throughout the Preserve. These streams include San Antonio Creek in the Valle San Antonio and Jaramillo Creek and the East Fork Jemez, both in the Valle Grande. These streams were characterized by two datasets: one dataset collected with a 500 µm mesh modified Hess sampler from 2012 to 2014 (Jacobi and Associates), and a dataset collected with a 250 µm mesh Hess sampler from 2015 to 2018 (Kremer and Caldwell). Non-metric multidimensional scaling (NMDS) plots using the *metaMDS* function of the *vegan* package (Oksanen et al. 2019) were used to visualize differences in species composition within each stream. It is important to note that differences in diversity estimates and other composition classifiers between the two samplers are attributed to differences in the sensitivity of the gear types rather than actual assemblage or environmental changes.

3.a. Aquatic macroinvertebrate assemblages over time: *San Antonio Creek of the Valle San Antonio*

Before evaluating aquatic macroinvertebrate assemblages over time, we compared the similarity of assemblages among reaches to determine if any could be grouped together. The upper, middle, and lower reaches of San Antonio Creek within the Valle San Antonio exhibited different assemblages (Figure 7.12). The upper most reach within the eastern elk exclosure of the San Antonio (upper) and the middle reach within the central elk exclosure (middle) contained greater Ephemeroptera, Plecoptera and Trichoptera richness than the lower reach in the lower and most western elk exclosure (lower). The fall composition of the middle reach is similar to the spring composition in the lower reach, with greater Chironomidae richness than the upper reach. Given these differences, the reaches of the Valle San Antonio are hereafter considered separately and referred to as “upper”, “middle”, and “lower” reaches. Since pre- and post-fire trends of the upper San Antonio are discussed above, post-fire trends of the only the middle and lower San Antonio Creek are covered in this section.

Estimated diversity trends from 2012 to 2018 in the middle (Figure 7.13) and the lower Elk exclosure (Figure 7.14) of San Antonio Creek are reflected by high interannual variability for both spring and fall samples. Apparent increases in the estimated diversity indices after 2014 are attributed to the change in Hess and mesh size. Stacked bar plots visualized changes in proportions of feeding groups over time with the Hess 250 µm mesh for each of the reaches (middle, lower) in San Antonio Creek of the Valle San Antonio (Figures 7.15 and 7.16, respectively). Increases in the proportion of predators and shredders with a decrease in the proportion of filterers began in 2015. Predaceous Chironomidae (Diptera) and early instar Plecoptera (stoneflies) were the greatest contributors to the increase in predators. Specimens were small (1 mm, total length); thus, apparent increases in abundance may be due to a decrease in sampler mesh size beginning in 2015.

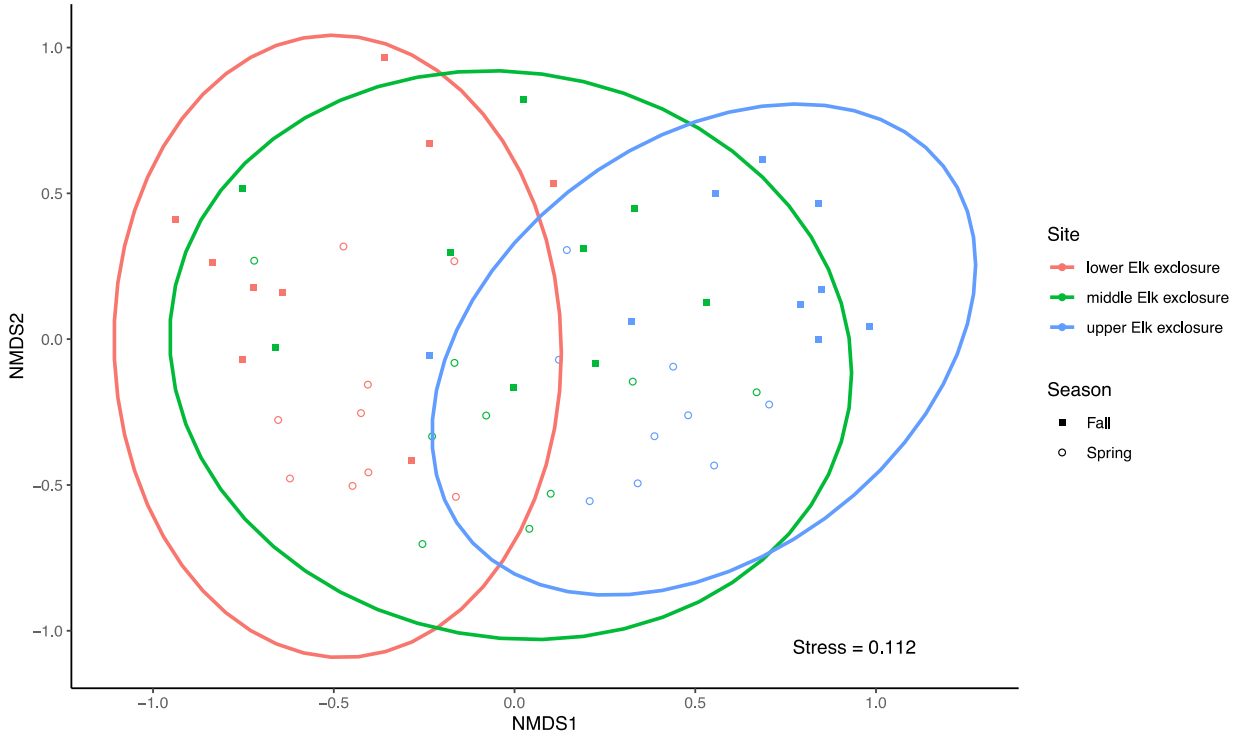



Figure 7.12. Non-metric multidimensional scaling ordination plot of 250 µ*m* Hess samples (2015-2018; Kremer and Caldwell data) reflecting changes in benthic macroinvertebrate composition with season and location (upper, middle, and lower Elk exclosures) throughout the San Antonio Creek continuum (from upper to lower San Antonio). Stress (0.112) is an indication of the fit of the ordination, where values less than 0.20 indicate a good fit. This plot highlights composition dissimilarity across upper, middle, and lower reaches of San Antonio Creek in the Valle San Antonio.

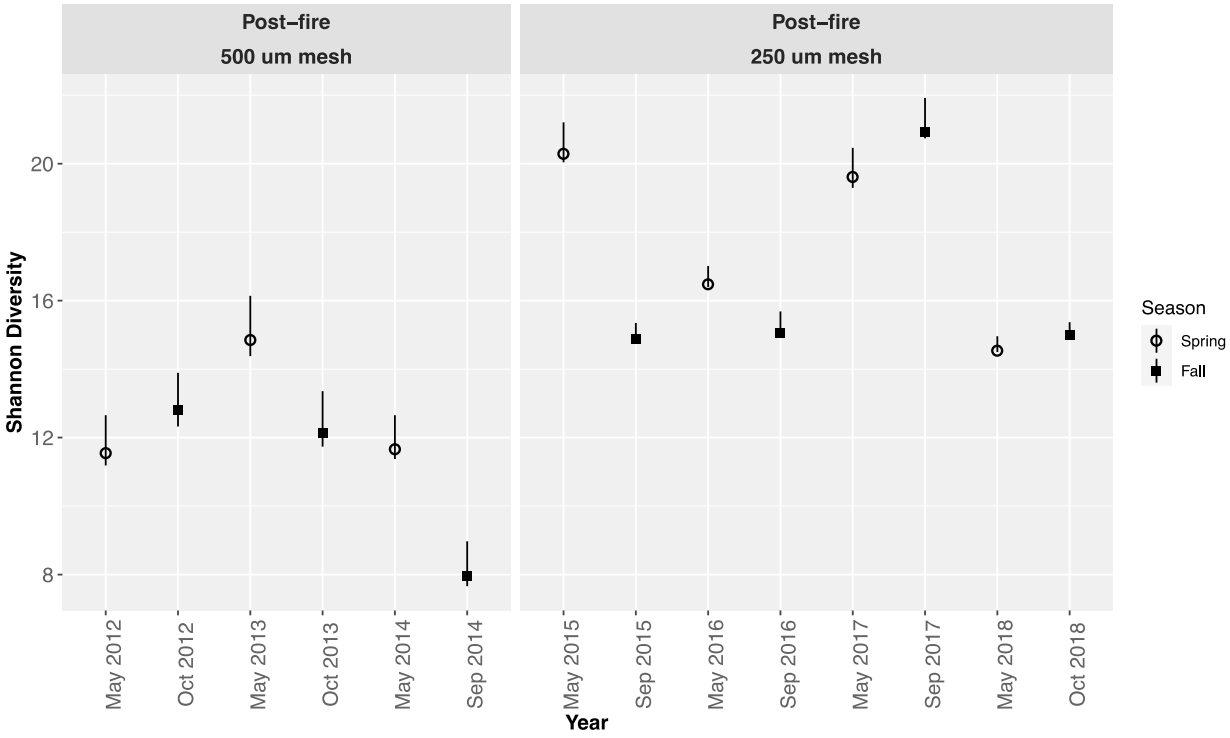



Figure 7.13. Spring and fall sample collections in the middle Elk exclosure within San Antonio Creek of the Valle San Antonio from 2012 to 2018. Shannon diversity estimates for rarified samples (n = 1000) by sample date with spring samples indicated by open triangles and fall samples indicated by closed circles. Each shape represents the diversity estimate and lines represent the bootstrap 95% confidence interval of each estimate. Diversity estimates vary due to seasonal and interannual environmental variation.

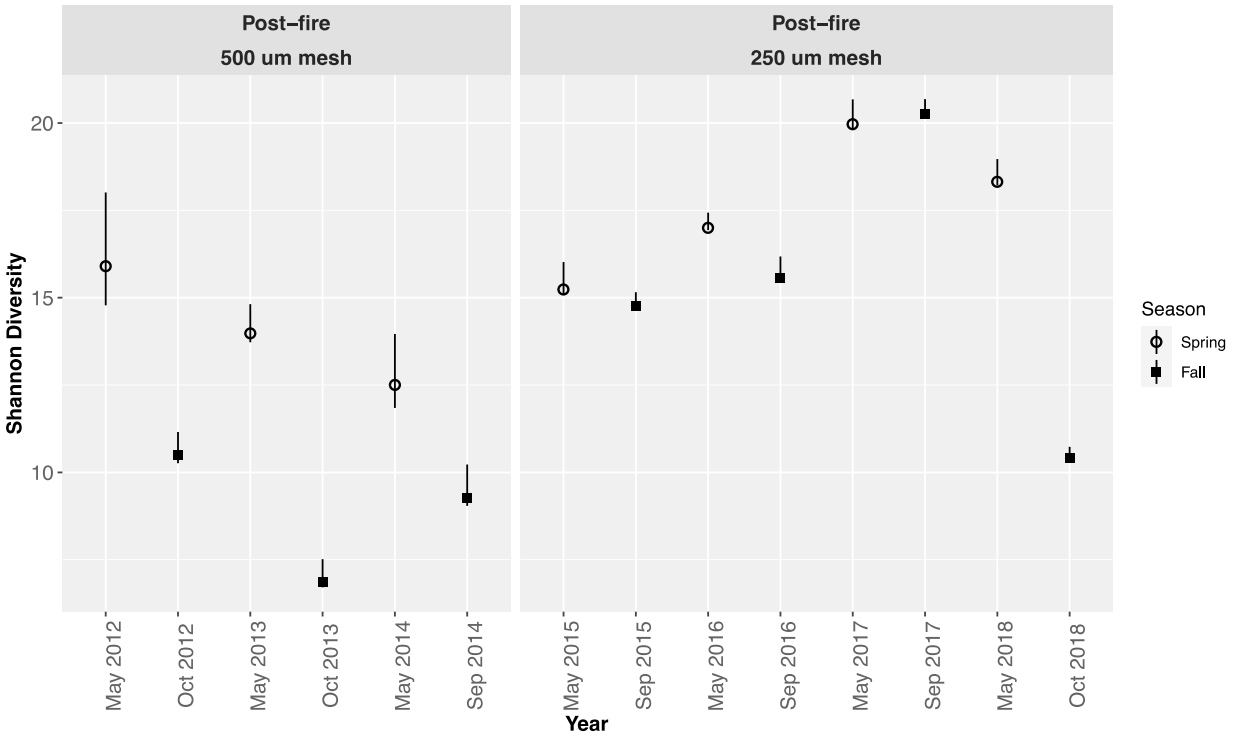



Figure 7.14. Shannon diversity estimates for rarified samples (n = 1000) within the lower Elk exclosure in San Antonio Creek of the Valle San Antonio by sample date with spring samples indicated by open triangles and autumn and summer samples indicated by closed circles. Each shape represents the diversity estimate and lines represent the bootstrap 95% confidence interval of each estimate. Variation in diversity is influenced by environmental drivers such as precipitation and turbidity.

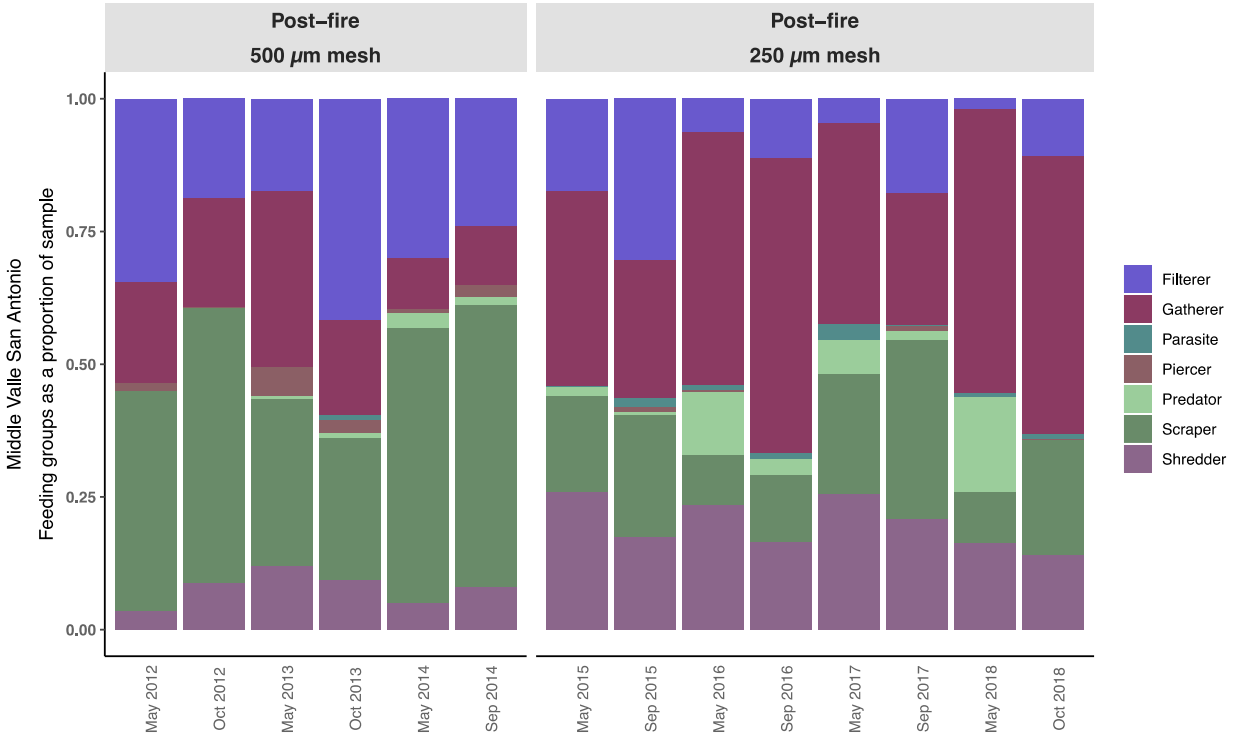



Figure 7.15. Spring and fall functional feeding group composition in middle San Antonio Creek of the Valle San Antonio after the Las Conchas wildfire (2012-2018). Proportion of the feeding group reflects a decrease in functional scrapers and an increase in functional gatherers between two mesh sizes. Post-fire increases in gatherers were represented by Chironomidae and Baetidae (Ephemeroptera) when using smaller mesh size.

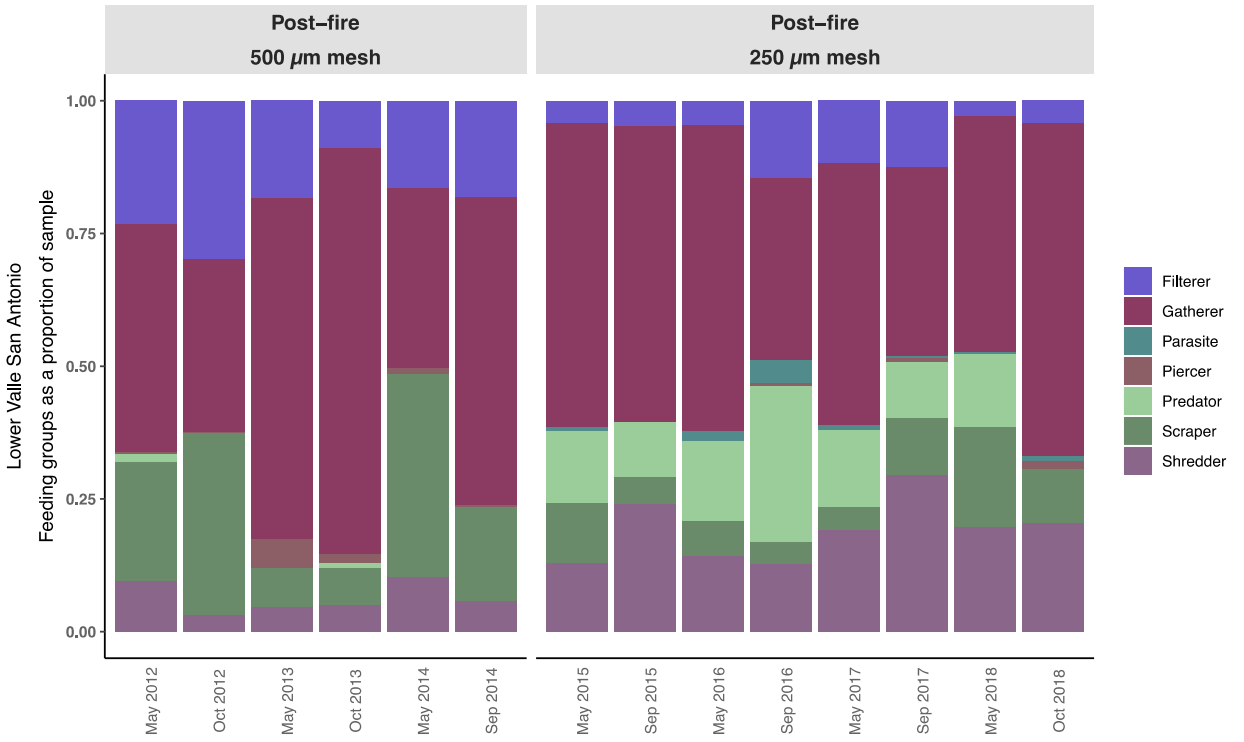



Figure 7.16. Spring and fall functional feeding group composition in lower San Antonio Creek of the Valle San Antonio after the Las Conchas wildfire (2012-2018). Proportion of the feeding group reflects an increase in functional shredders and predators with a decline in functional filterers between two mesh sizes. Filterers were represented largely by Hydropsychidae (caddisflies) and Simulium (blackflies). Chironomidae in the Cricotopus and Orthocladius genus were the greatest contributors to the increase in shredders in the small Hess mesh. Predators were represented by Chironomidae when using smaller mesh size.

3.b. Aquatic macroinvertebrate assemblages over time: *Jaramillo Creek in the Valle Grande*

Monitoring reaches within Jaramillo Creek in the Valle Grande exhibited assemblage changes with changes in stream location ($F_{1,31}$ = 11.46, $P$ = 0.001) (Figure 7.17). The upper reach within the western elk exclosure of Jaramillo Creek contained greater Ephemeroptera and Trichoptera richness than that of the lower reach in the eastern elk exclosure. In contrast to the upper reach, the lower reach contained greater Chironomidae and Elmidae richness. Given these differences, the upper and lower reaches of Jaramillo Creek (Figures 7.18 and 7.19, respectively) were analyzed separately and revealed diversity estimates that were highly variable from 2012 to 2018 in both reaches. Composition of the functional feeding groups in both the upper (Figure 7.20) and lower Jaramillo (Figure 7.21) after the Las Conchas wildfire (2012-2018) reflected a greater proportion of gatherers. These proportions were largely due to an increase in the abundance of Chironomidae within the samples. Whether this difference was a response to the sampler mesh size or to environmental variables that changed with time is confounded and unclear.

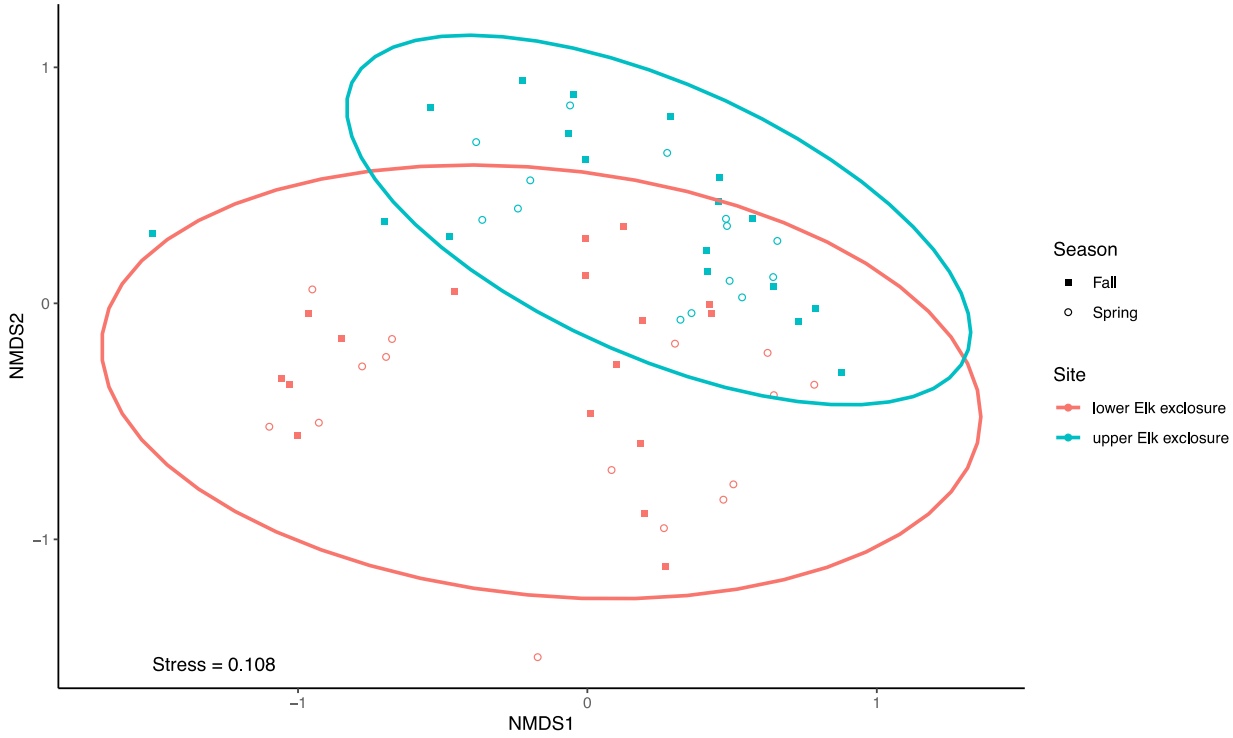



Figure 7.17. Non-metric multidimensional scaling ordination plot displaying fall and spring sample collections using the Hess 250 µm mesh of Jaramillo Creek (2015-2018) that reflect changes in community composition from upper to lower stream reaches. Stress (0.108) is an indication of the fit of the ordination, where values less than 0.20 indicate a good fit. Symbol shape indicates pre-fire or post-fire status, and ellipses are 95% confidence ellipses for each group centroid. Composition within upper (western) reaches of Jaramillo Creek were dissimilar to those in the lower (eastern) reach.

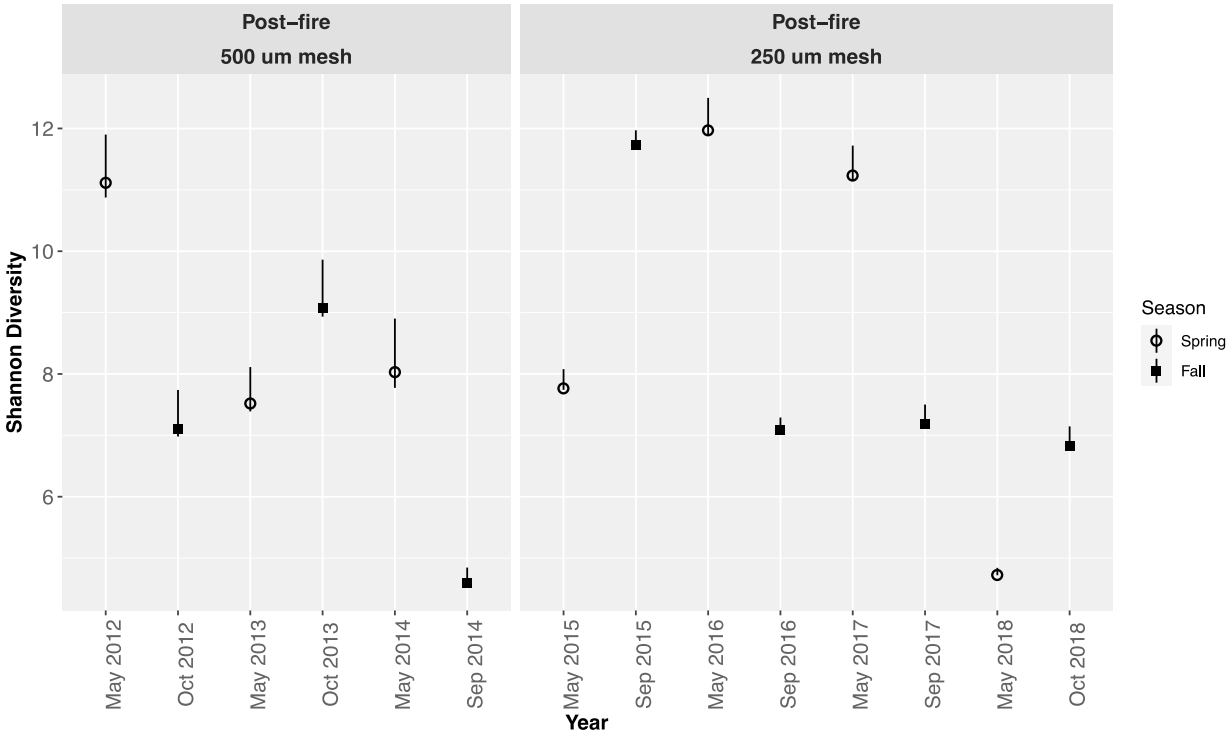



Figure 7.18. Shannon diversity estimates compared between the two mesh sizes for rarified samples (n = 1000) within the upper elk exclosure in Jaramillo Creek by sample date with spring samples indicated by open triangles and autumn and summer samples indicated by closed circles. Each shape represents the diversity estimate and lines represent the bootstrap 95% confidence interval of each estimate. Variation in diversity is likely influenced by environmental drivers such as precipitation and turbidity.

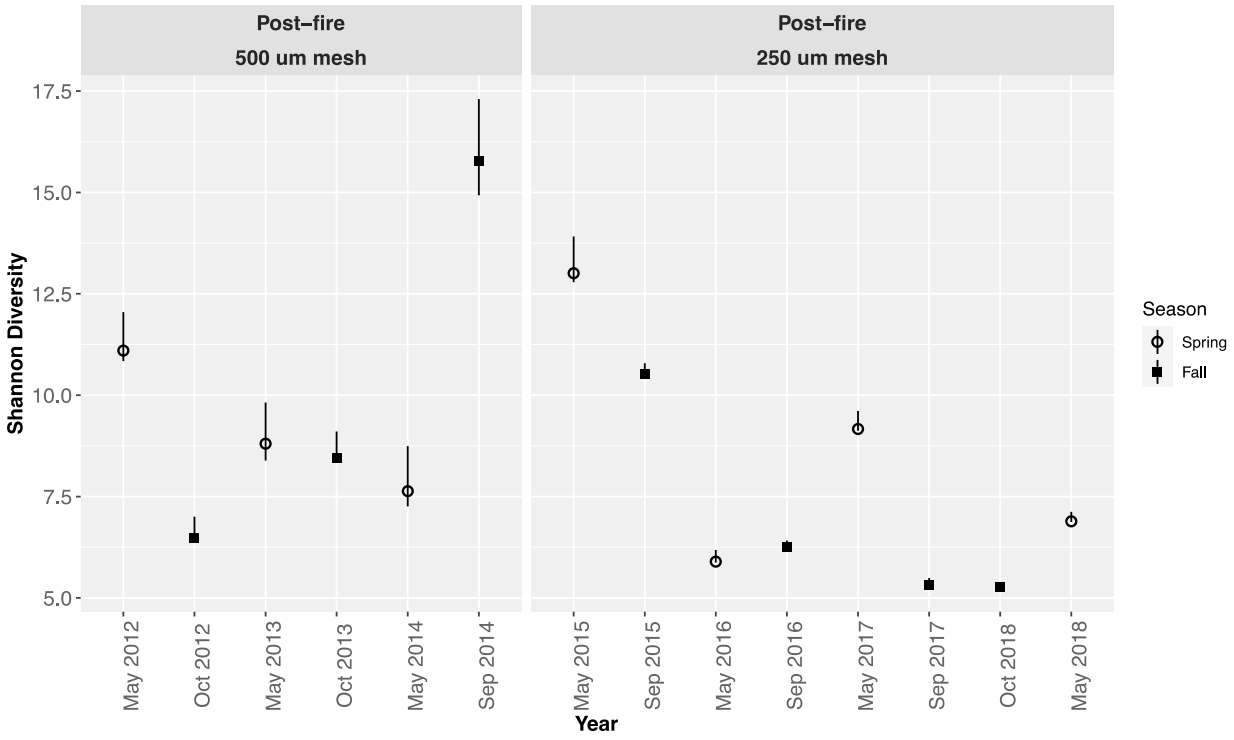



Figure 7.19. Shannon diversity estimates compared between the two mesh sizes for rarified samples (n = 1000) within the lower elk exclosure in the Jaramillo Creek by sample date with spring samples indicated by open triangles and autumn and summer samples indicated by closed circles. Each shape represents the diversity estimate and lines represent the bootstrap 95% confidence interval of each estimate. Variation in diversity is likely driven by environmental drivers such as precipitation and turbidity.

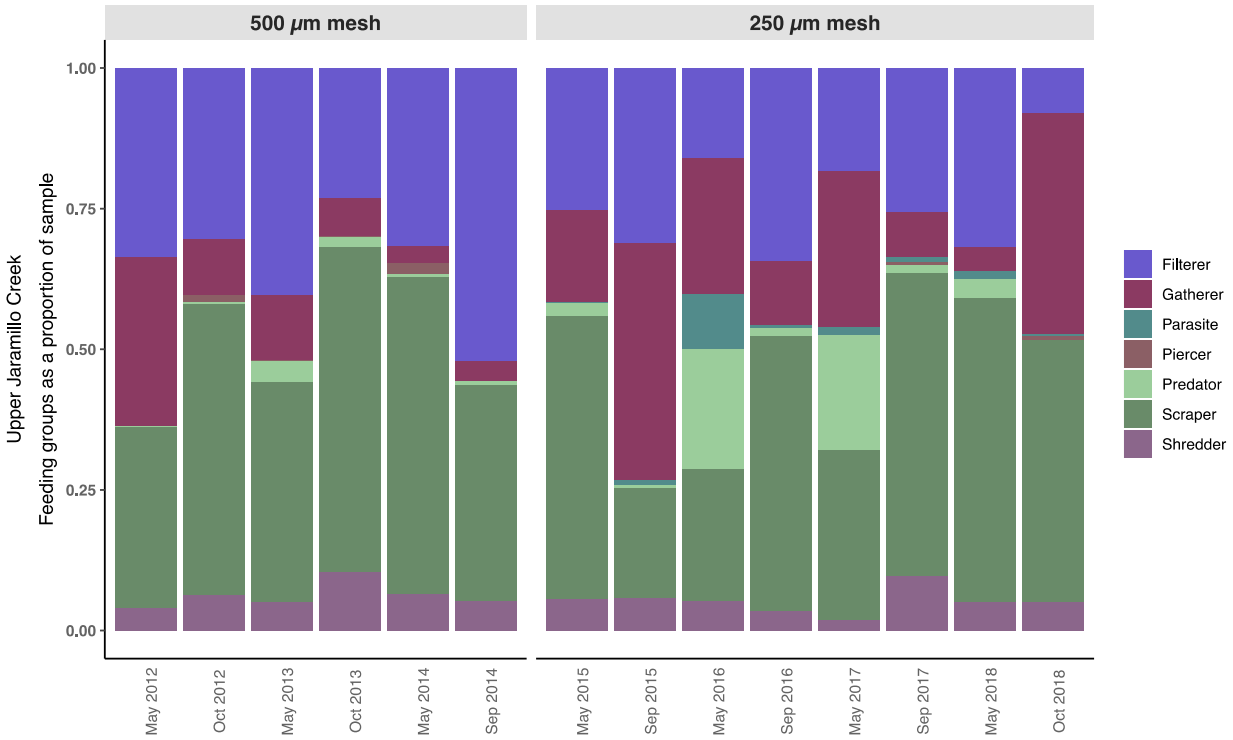



Figure 7.20. Functional feeding group composition compared between the two mesh sizes in the upper Jaramillo after the Las Conchas wildfire (2012-2018) reflected a greater proportion of functional gatherers and shredders in assemblages sampled with a smaller mesh.

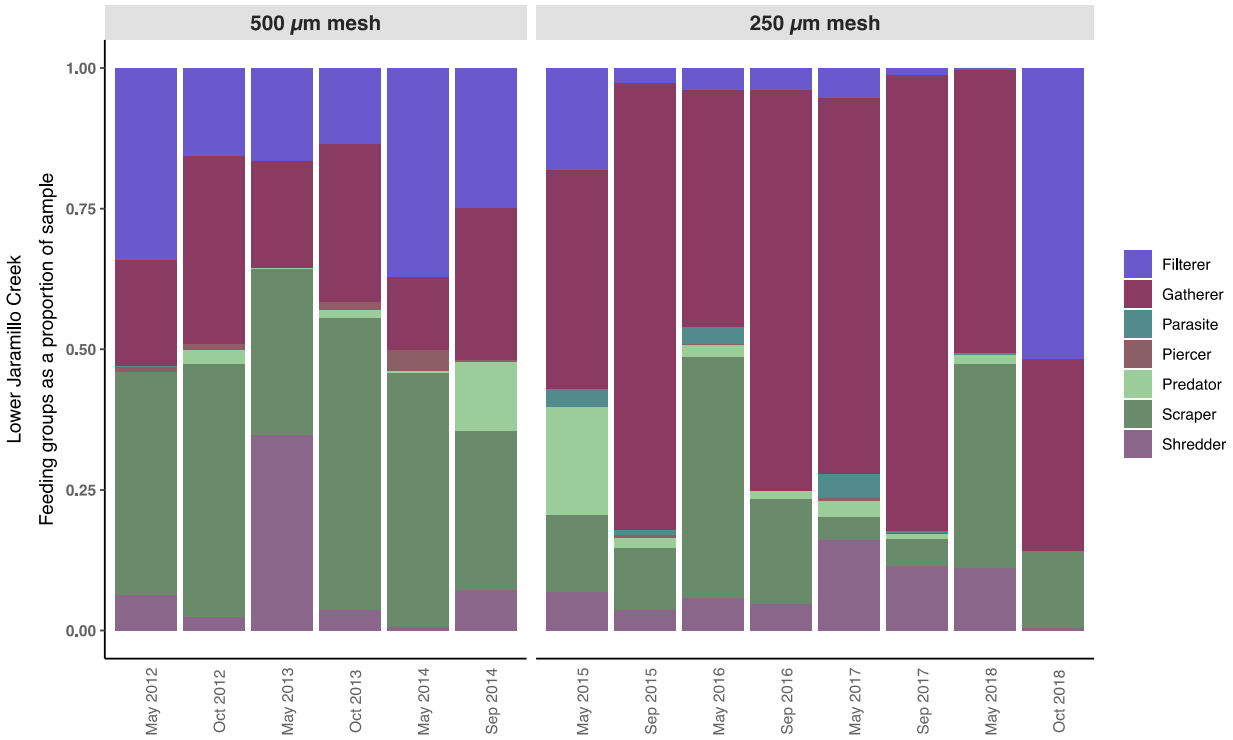



Figure 7.21. Functional feeding group composition compared between the two mesh sizes in lower Jaramillo Creek after the Las Conchas wildfire (2012-2018) reflected an increase in functional gatherers and a decline in functional scrapers over time.

3.c. Aquatic macroinvertebrate assemblages over time: *East Fork Jemez in the Valle Grande*

Upper and lower reaches within the East Fork Jemez were characterized by similar assemblages of macroinvertebrates (Figure 7.22). Therefore, diversity estimates were generated by combining both reaches from 2012 to 2018. Diversity estimates were highly variable (Figure 7.23) and the composition of functional feeding groups reflected an increase in gatherers and shredders with a decline in piercers (Figure 7.24). High proportions of gatherers within the Jemez were attributed to variation in densities of *Tricorythodes sp.* (Ephemeroptera Leptohyphidae), Naididae (Haplotaxida), and various Chironomidae (Dipetera) families. Similarly, the increased density of shredders was largely due to *Cricotopus* or *Orthocladius sp*. (Chironomidae). However, when assemblages were collected with the small 250 µm mesh Hess, each of these taxa, including *Tricorythodes sp.* were heavily represented by early instars with less than 2 mm in length. Specimen lengths were not recorded prior to 2016; however, higher densities of small organisms were likely an artifact of the smaller sampler mesh size, rather than a response to an environmental driver. Scrapers of both Hess sampler types were largely represented by Elmidae (Coleoptera) and *Physella sp.* (Hygrophila).

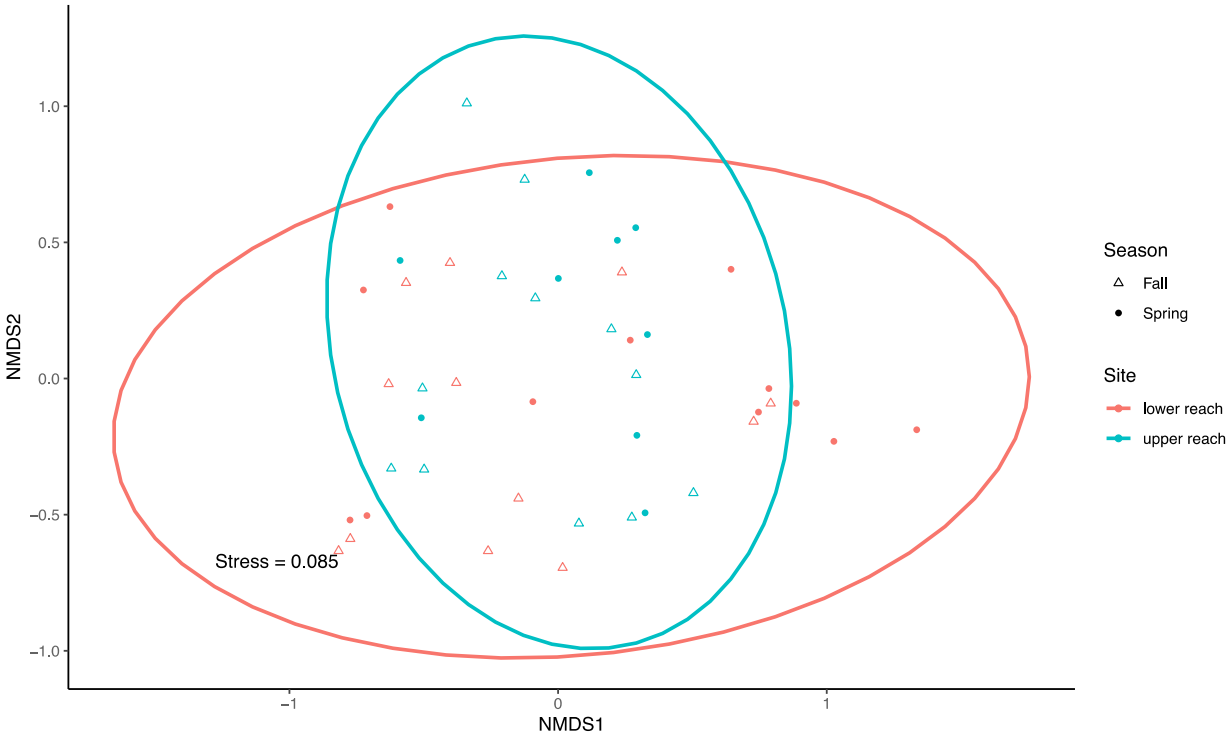



Figure 7.22. Non-metric multidimensional scaling ordination plot visualizing 250 µm Hess samples (2015 - 2018) reflected compositional similarity between the lower and upper sample reaches of the East Fork Jemez in the Valle Grande. Stress (0.085) is an indication of the fit of the ordination, where values less than 0.20 indicate a good fit. Symbol shape indicates season, and ellipses are 95% confidence ellipses for each group centroid.

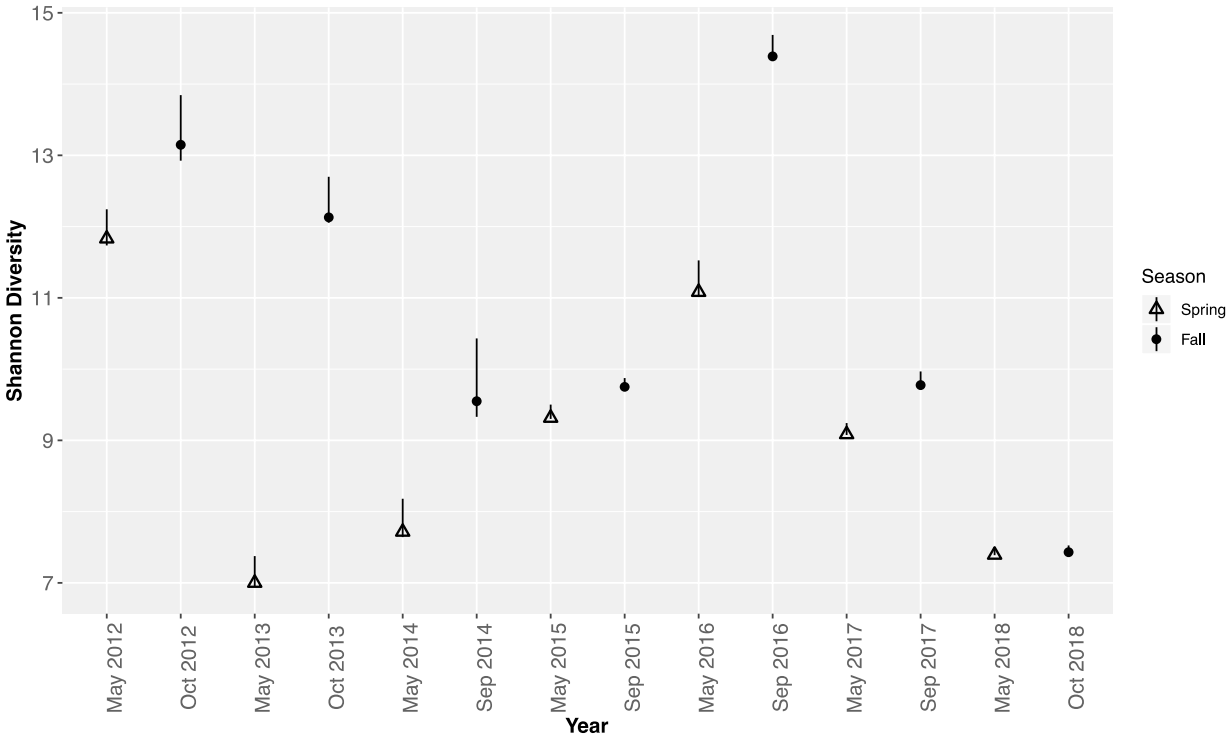



Figure 7.23. Shannon diversity estimates for rarified samples (n = 1000) within the East Fork Jemez by sample date. Spring samples indicated by open triangles and autumn and summer samples indicated by closed circles. Each shape represents the diversity estimate and lines represent the bootstrap 95% confidence interval of each estimate. Variation in diversity is likely driven by environmental drivers such as precipitation and turbidity.

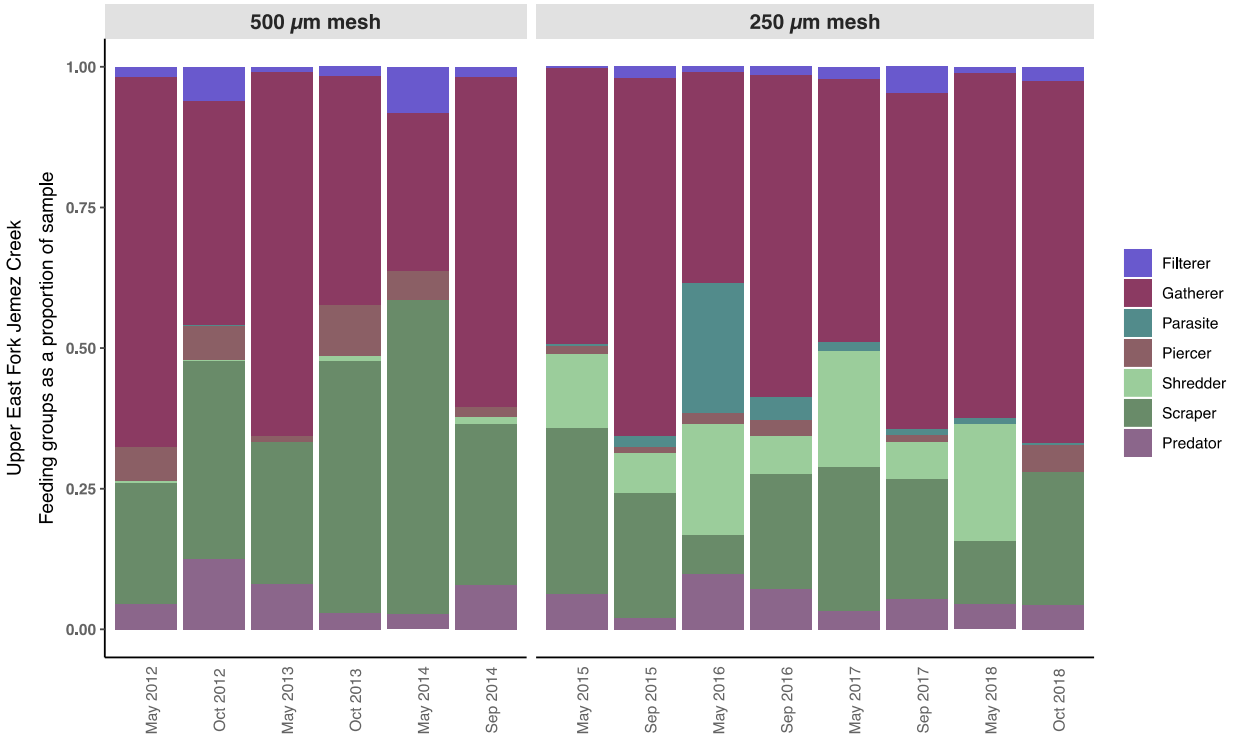



Figure 7.24. Functional feeding group composition compared between the two mesh sizes in the East Fork Jemez from 2012 to 2018 reflected an increase in the relative proportions of gatherers and shredders and a decline in piercers through time.

## Discussion

*Methodology Review and Dataset Selection*

The methodologies employed in the Valles Caldera macroinvertebrate biomonitoring program and long-term datasets resulting from various investigators before and after the Las Conchas wildfire had to be standardized to effectively characterize benthic macroinvertebrate responses to assess temporal and environmental changes. Differences in taxa density among sampling gear types required review of diversity estimations using rarefaction analysis. When we compared three sampling methods applied as part of the Preserve's long term benthic macroinvertebrate biomonitoring program, we determined that the Surber sampler did not capture sufficient number of taxa, which resulted in different density estimates from samples collected by the Hess that used the same mesh size (500 um). In addition, two mesh sizes (250 µm versus 500 µm) of the Hess sampler confounded interpretation of the data because it resulted in differences in the abundance of common taxa (i.e., larger, more common taxa were selected for in the 500 µm mesh in contrast to more rarer taxa captured with the 250 µm mesh). For either mesh size, we determined that a minimum of 750 individuals/0.10 $m^2$ would result in comparable diversity of taxa according to the Shannon Index. An assumption of rarefaction analysis, however, is that differences in macroinvertebrate abundance among samples are due to differences in sample size (e.g., the number of individuals per sample), rather than in response to an environmental parameter (Colwell et al. 2012; Chao and Jost 2012; Chao et al. 2014). Differences in abundance due to environmental parameters will affect diversity estimates and confound comparison across sample methods.

*Recommendations*

Consistency (i.e., standardization) in spatial and temporal sample collection methods will be key to the future success of the Preserve's aquatic biomonitoring program. While we were able to make some inferences with the current data that came from three different gear types across seasons, we cannot truly identify environmental variation due to wildfire or climate as casual to changes in assemblages.

The reliability of benthic organism assemblages as bioindicators of environmental degradation has led to the development of many rapid assessment programs, which are patterned after the U.S. Environmental Protection Agency's (EPA) Rapid Bioassessment Protocol (e.g., Barbour and Gerritsen 1996). These protocols include efficient and previously statistically verified sampling and data analyses. The Preserve might consider adopting a benthic macroinvertebrate biomonitoring protocol that closely follows the EPA National Rivers and Streams Assessment Methods and the protocol be peered reviewed. This would ensure that industry standards of collection and analysis in both the field and the laboratory be repeatable, which would provide the Preserve a robust and manageable dataset. For example, the authors of this report (Kremer and Caldwell) applied these industry standards from May 2015 to October 2018 for benthic macroinvertebrate collection and processing (see Appendices).

We recommend the Preserve develops specific objectives designed around the collection of benthic macroinvertebrates that not only allows for the analysis of temporal changes of benthic assemblages but should not require an expensive and exhaustive taxonomic inventory. If a complete inventory of macroinvertebrates is desired, it is advised that the inventory is conducted separately and less frequently than collection and analysis of biomonitoring samples. This will save time and expense. If a comprehensive inventory is required, we suggest that at least three sampling methods be employed simultaneously: 1) kick sampling captures taxa that

prefer to cling to overhanging vegetation or shorelines; 2) nocturnal light traps capture adult stages as nymphs and larvae are often inadequately developed for species resolution; and 3) small-mesh Hess samples are used to collect multiple replicates (see Vieira et al. 2009). A 250 µm mesh Hess is recommended for these comprehensive inventory collections, as these samples reflect greater richness of sampled assemblages compared to the 500 µm mesh Hess (i.e., rare species are captured in the smaller mesh). This combination of methodologies can be performed across seasons within a year to account for the incongruous life-cycles of benthic taxa and that identification be conducted by a taxonomist with the ability to identify Chironomidae to the lowest taxonomic levels. There are several macroinvertebrate laboratories in the Southwest that offer Chironomidae identification to species level (e.g., Timberline Aquatics, Fort Collins, CO and the National Aquatic Monitoring Center, Utah State University, Logan UT); however, sample processing and shipping to these facilities will incur additional time and expense.

Biomonitoring protocols do not need to be labor intensive, though monitoring should occur frequently to effectively characterize the extent to which environmental variables influence macroinvertebrate assemblages. The intersection of objectives of the biomonitoring, and the ability of macroinvertebrates to recolonize streams should be considered so that the level of sensitivity needed in the response signal is achieved. Complete inventories with suitable frequency for observing temporal and spatial responses to environmentally induced changes are not feasible when considering processing time and expense. However, biomonitoring samples will require some standards of accuracy and completeness to adequately reflect community composition and function. The 500 µm mesh Hess demonstrated acceptable estimates of diversity for community analysis that were similar to the 250 µm mesh Hess. While the smaller mesh size may capture additional rare and small taxa relative to the larger mesh, it is likely that the extra time sorting high densities of small (less than 1 mm) specimens will not be cost effective. Surber samplers are not recommended for streams in the Preserve. These are best suited to riffle sampling, yet many of the Preserve's streams require sampling of runs, which may result in greater loss of specimens that "escape" or drift around the Surber funnel once suspended in the water column. Hess samplers are designed to confine the suspended benthos by isolating the sampled water column, unless water is deeper than height of Hess. When deciding on the level of taxonomic effort for biomonitoring samples (mesh 250 µm versus 500 µm), it will be important to consider the cost of Chironomidae identification to genus or species levels, due to time and expertise required to dissect and slide mount many of these taxa.

*Environmental Drivers of Macroinvertebrate Assemblages*

Analyses linking environmental conditions to specific taxa groups were insightful in understanding what conditions influence and alter macroinvertebrate communities. Additionally, the models identified which variables exhibited multi-collinearity versus which influenced responsive community metrics. Turbidity, conductivity, dissolved oxygen, and temperature were the prominent drivers in the strongest models. Large-scale and repeated disturbances such as wildfires affect thermal regimes and water turbidity in aquatic systems (Minshall et al. 1997; Earl et al. 2003; Dunham et al. 2007) and these affect the local distribution of aquatic organisms (Mihuc et al. 1996; Isaak et al. 2010). Water temperature contributes largely to aquatic ectotherm growth, development, and fecundity, and thereby affects geographic distribution and diversity of the macroinvertebrates (Vannote and Sweeney 1980; Ward and Stanford 1982). Changes in turbidity and conductivity result in shifts in the basal resources and habitat quality available to macroinvertebrate communities (Wood and Armitage 1997). However, most of the biotic

indices tested were not explained by the majority of the data from sondes or the weather stations. Furthermore, stream location appeared in models as an explanatory variable, indicating that other unforeseen variables differed among streams, but were not included in the models.

*Recommendations*

To characterize the effects of wildfire on macroinvertebrates, we required a better understanding of how wildfire and catchment characteristics influenced the drivers of macroinvertebrate assemblages. The extent of any disturbance on a stream largely depends on the characteristics of the watershed (e.g., catchment slopes, riparian vegetation), the severity, intensity and heterogeneity of the wildfire, and the features of each stream (e.g., discharge, gradient, and geomorphology) (Gresswell 1999; Bixby et al. 2015). Though it is not within the scope of this report to estimate the impacts of disturbance on all variables known to influence benthic communities, characteristics of each stream such as size (Minshall 2001), temperature (Dunham 2007), burn extent (Minshall 2001), post-fire resources (Mihuc 1995), benthic substrates (Oliver 2012) and water chemistry (Earl and Blinn 2003) contribute to the differential responses of macroinvertebrate communities.

The Preserve's macroinvertebrate biomonitoring program can facilitate future studies that evaluate many potential relationships between catchment, disturbance or environmental characteristics and stream function. If desired, to identify additional environmental drivers of biotic indices, we recommend that the program include routine collection of other environmental variables that stream biota may respond to. This was exemplified by differences between pre-fire diversity and assemblage relative to post-fire in the Valle Toledo. Pre-fire samples were collected in low precipitation years, possibly resulting in reduced stream discharge and increased periphyton growth. The concomitant change in functional feeding groups, such as the increase in scrapers, may be indicative of available resources and have little to do with the indirect effects of wildfire. Examples of environmental variables that could be added to the program with little additional field time include pebble counts, discharge, and identification of microhabitat or run-riffle-pool sequences to characterize benthic habitat heterogeneity and identify changes over time.

To identify relationships between disturbance or catchment characteristics and stream biota, remote sensing tools that quantify topographic characteristics of streams or measure changes in stream width and sinuosity over time may be informative. Additionally, remote sensing and field studies could compare vegetation coverage and composition among streams pre- and post-fire to inform relationships between riparian composition and aquatic community composition. The Preserve's headwater streams contribute to some of the largest montane wet meadows in the Southern Rocky Mountains Ecoregion (Muldavin and Tonne 2003) and to the water supply of the Albuquerque metropolitan area. Thus, such studies would aid in management and restoration efforts in a way that positively impact New Mexico ecosystems and municipalities.

*Macroinvertebrate Assemblage Changes over Time*

The intricate effects of disturbance on aquatic communities can be complicated by continued or episodic perturbations that are indirect effects of disturbance. Continuous in-stream water quality monitoring within the Preserve suggests that small streams in close proximity to burn perimeters, such as the Indios may be subjected to such continued effects with highly variable turbidity (i.e., long term episodic perturbation) as was seen in 2013 and 2015.

Anecdotally, we noted that from 2015 to 2018, the riffles from which our samples were collected in the Indios changed dramatically in a 6-month period. We observed that sedimentation changed the width, depth, and sinuosity of the thalweg within the sampling reaches. In the lower reaches of the Indios, it was not uncommon to find that the three riffles for sample collections were not the same as the previous sample period. Observed fluctuations in diversity and composition may be in response to these repeated erosional disturbances. Reduced post-fire abundances in the Valle Toledo and Valle San Antonio are consistent with previous studies of the effects of wildfire on stream ecosystems in the western United States (Minshall et al. 2001, 2003; Vieira et al. 2004; Arkle et al. 2010). These studies were within catchments where high post-fire stream flows resulted in channel alterations and reduced post-fire habitat stability, and a severe decline of macroinvertebrate densities or biomass occurred within 1–3 years post-fire and persisted for several years.

Functional feeding groups are determined by a taxa's principal mode of feeding and not by nutritional sources. Functional feeding group richness reflects the number of taxa within a functional feeding group and can be indicative of the relative diversity of nutritional resources available to feeding groups. Some of the most notable assemblage changes in our study were observed through changes in the relative abundance and diversity of functional feeding groups. A theoretical understanding of feeding group responses (Minshall et al. 1989; MacKay 1992) led us to expect an increase in the proportion and diversity of collector gatherers and collector filterers following the fire. These feed groups best reflect the availability of primary resources and do not necessarily have selective diets. Some taxa in these groups are opportunistic and may serve as colonizers following disturbance events (MacKay 1992). Predators may be considered similarly, but with respect to secondary resources. The availability of these resources may fluctuate in response to both acute and chronic disturbance, as erosion, turbidity, changes in temperature and dissolved oxygen may affect primary and secondary resources. As time allowed for the recovery of burned riparian meadows or upstream forests within the watershed, we anticipated an increase of shredders with an increase in allochthonous leaf and grass detritus. Due to changes in sampling methods in 2015, we cannot be certain of the drivers of the observed increase in shredders in the Indios from 2015 to 2018, however, increased foliage in the upper reaches of the watershed is expected to induce a similar response. Changes in scraper abundance may be indicative of changes autotrophic periphyton. This may be exemplified by the decrease in scrapers from 2012 to 2014 in the San Antonio of the Valle Toledo. Though some meadow patches were burned throughout the valles, local observations revealed the grasslands returned to pre-fire conditions within two years (R. Parmenter, personal communication). Nutrients released by the intense wildfire may have allowed for increased grass density and height, limiting light needed for autotrophic growth within the Valle Toledo stream. Stream coverage by riparian grasses may also be expected to change seasonally, as grasses shade stream edges, or the entire stream width.

*Recommendations*

In addition to issues caused by non-standardized monitoring methods, future studies that utilize data from the Preserve's biomonitoring program will need careful consideration of the geographic aspect of the study design if they aim to compare macroinvertebrate assemblages within and among reaches. Each stream reviewed here differed in size, substrate, gradient, and riparian catchment characteristics so that each supported a slightly different macroinvertebrate assemblage. For these reasons, sufficient replicates throughout the same stream should be

included to reflect the stream continuum. The River Continuum Concept (Vannote et al.1980) hypothesizes that structure and function of stream assemblages are adapted to their immediate environment. Therefore, taxa will change in a replacement series as physical variables change along the stream gradient. Benthic taxa are adapted to exploit resources in their optimal environment, resulting in an assembly line of taxa each serving a function to advance the decomposition of organic inputs as the detritus moves downstream. This principle was evidenced in the Preserve by streams with disparate communities when comparing "upstream" and "downstream" assemblages. A key consideration for a biomonitoring program will be to have long term, site-specific trends as indicators of the stream environment. Conversely, if program objectives are to compare among sites, then the Preserve may consider joining with another land use agency to have a concurrent monitoring program among comparable streams.

*Conclusions*

Historically, relatively frequent mixed-severity wildfires have been an important driver in Southern Rockies montane forests (Brown et al. 2014). However, a history of wildfire suppression in conjunction with a changing climate and reduced snowpack has increased the severity and frequency of stand-replacing wildfires (Westerling et al. 2006). Vegetation within forested water catchments regulates water quality through retention and release of nutrients. However, vegetation mortality due to wildfire instantaneously reduces plant demand of nutrients (Rhoades et al., 2019) and in combination with hydrophobic soils (Reale et al., 2015), results in high post-fire erosional sediment and nutrient inputs to headwater streams. Post-fire runoff with increased sediment loading may alter channel morphology and temperature regime at baseflow conditions (Gresswell 1999; Dunham et al. 2007). The unpredictability and heterogeneous effects of wildfire on stream temperature have been challenging as researchers attempt to associate changes in stream characteristics with the recovery of aquatic biota (Issac 2010; Beakes et al. 2014; Cooper et al. 2015).

Characterization of the impact of environmental disturbance was not within the scope of the biomonitoring program; however, the data clearly showed that the majority of measured environmental variables contributed to differential responses of macroinvertebrate community composition. The effects that wildfires have on aquatic systems are both direct with short-term or acute effects and indirect with lasting long-term or chronic effects. While direct effects may briefly affect aquatic organisms, indirect effects such as flooding, sediment deposition, and changes in stream morphology result in wholesale changes in community structure and function (Earl and Blinn 2003; Malison and Baxter 2010; Vieira et al. 2004; Minshall et al. 2001). Benthic macroinvertebrates are often monitored after wildfire as sensitive indicators of stream health. Variations in macroinvertebrate assemblages imply changes to a system including thermal regime changes, nutrient inputs, or streambed modifications (Downing 1984; Vannote and Sweeney 1980). The indirect effects of wildfire can temporarily increase macroinvertebrate growth and density, which has been attributed to increases in nutrient inputs from overland flow (Rugenski and Minshall 2014; Silins et al. 2014) and dominance by r-strategists with several generations per season (Malison and Baxter 2010). Macroinvertebrate density and richness will be reduced due to flood pulses and sediment movement (Vieira et al. 2004; Oliver 2012).

Throughout the Preserve, interannual variation in diversity was high. Streams were all differentially affected by the wildfire, despite several sites distant from burn perimeters. These appeared to recover quickly once the riparian grassland was re-established the following year. It

is difficult to attribute variability in diversity to post-fire effects. However, we hypothesize that a slight increase in shredders and scrapers in the Indios of the Valle Toledo from 2011 to 2014 was indicative of increased coarse material and periphyton in the stream as a result of post-fire effects. Similarly, an increase in filterers in San Antonio Creek of the Valle Toledo may be in response to an increase in fine particulates within the water column as a result of post-fire effects. These hypotheses assume that resources were suitable to some groups but not others pre- and post-fire and requires additional research that tie post-fire effects to ecological recovery in southwestern montane grassland streams.

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## APPENDIX 7.A
## Valles Caldera National Preserve
## Aquatic Benthic Macroinvertebrate Sample Site Locations
## 2011-2018

| Site ID | Stream | Easting | Northing | Latitude | Longitude |
|---|---|---|---|---|---|
| **SALE-1** | San Antonio Lower Exclosure (Elk) | 355938 | 3982236 | N 35°58'26.1" | W 106°35'52.2" |
| **SALE-2** | San Antonio Lower Exclosure (Elk) | | | | |
| **SALE-3** | San Antonio Lower Exclosure (Elk) | | | | |
| **SAME-1** | San Antonio Middle Exclosure (Elk) | 361584 | 3981359 | N 35°58'00.6" | W 106°32'06.3" |
| **SAME-2** | San Antonio Middle Exclosure (Elk) | | | | |
| **SAME-3** | San Antonio Middle Exclosure (Elk) | | | | |
| **SAUE-1** | San Antonio Upper Exclosure (Elk) | 364499 | 3981471 | N 35°58'05.7" | W 106°30'10.0" |
| **SAUE-2** | San Antonio Upper Exclosure (Elk) | | | | |
| **SAUE-3** | San Antonio Upper Exclosure (Elk) | | | | |
| **JCLE-1** | Jaramillo Creek Lower Exclosure (Elk) | 366800 | 3971222 | N 35°52'34.3" | W 106°28'32.9" |
| **JCLE-2** | Jaramillo Creek Lower Exclosure (Elk) | | | | |
| **JCLE-3** | Jaramillo Creek Lower Exclosure (Elk) | | | | |
| **JCUE-1** | Jaramillo Creek Upper Exclosure (Elk) | 364214 | 3975515 | N 35°54'48.8" | W 106°30'17.5" |
| **JCUE-2** | Jaramillo Creek Upper Exclosure (Elk) | | | | |
| **JCUE-3** | Jaramillo Creek Upper Exclosure (Elk) | | | | |
| **EFJLS-1** | East Fork Jemez Lightning Shack | 369262 | 3970667 | N 35°52'17.5" | W 106°26'53.5" |
| **EFJLS-2** | East Fork Jemez Lightning Shack | | | | |
| **EFJLS-3** | East Fork Jemez Lightning Shack | | | | |
| **EFJLE-1** | East Fork Jemez Lower Exclosure (Elk) | 364444 | 3967433 | N 35°50'30.2" | W 106°30'03.6" |
| **EFJLE-2** | East Fork Jemez Lower Exclosure (Elk) | | | | |
| **EFJLE-3** | East Fork Jemez Lower Exclosure (Elk) | | | | |
| **RDLI-1-1** | Rito de los Indios (Toledo) | 366016 | 3981217 | N 35°57'52.6" | W 106°29'19.2" |
| **RDLI-1-2** | Rito de los Indios (Toledo) | | | | |
| **RDLI-1-3** | Rito de los Indios (Toledo) | | | | |
| **RDLI-2-1** | Rito de los Indios (Toledo) | 366169 | 3981294 | N 35°58'01.3" | W 106°29'05.4" |
| **RDLI-2-2** | Rito de los Indios (Toledo) | | | | |
| **RDLI-2-3** | Rito de los Indios (Toledo) | | | | |
| **RDLI-3-1** | Rito de los Indios (Toledo) | 366384 | 3981538 | N 35°58'11.2" | W 106°28'52.9" |
| **RDLI-3-2** | Rito de los Indios (Toledo) | | | | |
| **RDLI-3-3** | Rito de los Indios (Toledo) | | | | |
| **RDLI-4-1** | Rito de los Indios (Toledo) | 367061 | 3984239 | N 36°0'27.2" | W 106°27'25.6" |
| **RDLI-4-2** | Rito de los Indios (Toledo) | | | | |
| **RDLI-4-3** | Rito de los Indios (Toledo) | | | | |
| **SAVT-1-1** | San Antonio (Toledo) | 366785 | 3980225 | N 35°57'26.4" | W 106°28'38.0" |
| **SAVT-1-2** | San Antonio (Toledo) | | | | |
| **SAVT-1-3** | San Antonio (Toledo) | | | | |
| **SAVT-2-1** | San Antonio (Toledo) | 367616 | 3979999 | N 35°57'19.5" | W 106°28'04.7" |
| **SAVT-2-2** | San Antonio (Toledo) | | | | |
| **SAVT-2-3** | San Antonio (Toledo) | | | | |
| **SAVT-3-1** | San Antonio (Toledo) | 367982 | 3979852 | N 35°57'14.9" | W 106°27'50.0" |
| **SAVT-3-2** | San Antonio (Toledo) | | | | |
| **SAVT-3-3** | San Antonio (Toledo) | | | | |

# Chapter 8. Wildlife Monitoring – Mammals

## Monitoring project title: Monitoring large- and small-mammal dynamics with wildfire and landscape restoration.

**Collaborators:** James W. Cain – *U.S. Geological Survey, New Mexico Cooperative Fish and Wildlife Research Unit, Department of Fish, Wildlife and Conservation Ecology, New Mexico State University;* Mark A. Peyton, Matt Keeling, Leah White, Gary Roemer, Fitsum Abadi – *Department of Fish, Wildlife and Conservation Ecology, New Mexico State University;* Sharon Valverde, Warren Conway – *Department of Natural Resources Management, Texas Tech University;* Robert R. Parmenter, Tyler Obermeit – *National Park Service, Valles Caldera National Preserve.*

**Summary:** One of the primary objectives of the SWJM CFLRP was to "improve wildlife habitat" for resident species of large and small mammals. The landscape scale forest thinning and prescribed burning were anticipated to alter vast areas of forest stand structure and increasing understory herbaceous vegetation. These actions would have changed habitat characteristics and food/shelter resources; as such, the collaborating organizations decided to undertake detailed monitoring of large mammals throughout the CFLRP project area and in the surrounding countryside. In addition, following the Las Conchas Fire in 2011, we conducted a 3-year monitoring program to assess the response of small mammals to wildfire in both ponderosa pine forests and mixed conifer forests.

The responses of large mammals to forest restoration treatments and wildfires were variable and depended on the type of disturbance, post-disturbance recovery period, and general diet and habitat use characteristics of each species. Elk, with their grass dominated diets, responded most strongly to wildfires, with selection for high severity burns, as well as prescribed fire and thinning treatments shortly after treatment. Mule deer selected recent prescribed fire and wildfire-burned areas in spring, but avoided more recently thinned areas. Mule deer strongly selected for thinned areas > 5 years old, after recovery of woody browse. Black bears selected areas with more structural complexity, providing both foraging resources and security cover. Bears generally avoided thinned areas, but did select for more recent prescribed fire. Bears selected for areas that were more recently burned with high severity wildfires, as well as areas experiencing high severity fires >10 years prior. Mountain lion habitat use was strongly related to the distribution and vulnerability of mule deer and elk, their primary prey in the study area. Day beds and den sites were typically found in rugged terrain that offered high levels of concealment. Overall, initial results suggest complex responses to restoration treatments and wildfire, driven mostly by species ecology related to security cover and food resources. Further analyses will account for independent and interactive effects of restoration treatments and wildfires in combination with other habitat characteristics.

For the small mammal project, we sampled small mammals for three years (2012-2014) on replicated burned and unburned (control) stands in both forest types. We recorded a total of 15 small mammal species (11 rodents, one rabbit, and three shrews). Small mammals exhibited lower species richness on burned mixed conifer sites than control sites, but species richness was only slightly lower on burned ponderosa pine sites compared to control sites. Deer mouse abundances were comparable between burned and control sites, but two species of chipmunks, woodrats, red squirrels, three species of shrews, and mountain cottontail rabbits were rare or absent in burned forests. Golden-mantled ground squirrels were more abundant on burned mixed conifer sites than control sites, and voles (*Microtus*) colonized the burned sites within three years of the fire (montane voles in 2013, and long-tailed voles in 2014). Rodent demographic data indicated mixed sex and age classes were present for the more abundant species, including reproductively active adult females; the exception was the long-tailed vole in burned ponderosa pine forests, which was represented only by male adults and subadults. Vole colonization occurred after burned sites attained 80 – 90% herbaceous cover. Given the lack of conifer regeneration, we anticipate that the small mammal community will retain its composition of meadow-grassland species for the foreseeable future.

## Section 8.1. Large mammals – elk, mule deer, black bears and cougars.

### Background

Our original goal was to monitor the responses of mule deer (*Odocoileus hemionus*), elk (*Cervus canadensis*), mountain lions (*Puma concolor*) and black bears (*Ursus americanus*) to forest restoration treatments associated with the Southwest Jemez Mountains CFLRP and Valles Caldera's Landscape Restoration and Management Project (LRMP). However, given the large extent and likely influence of the Las Conchas and other wildfires on large mammal habitat use, we incorporated wildfires as unplanned treatments. We monitored the short- and long-term changes in vegetation, particularly forage conditions, and the responses in movements and habitat selection of mule deer, elk, mountain lions and black bears. Our specific research objectives were to: 1) assess changes in abundance of key forage species in response to forest restoration treatments and wildfires including herbaceous forage for elk, browse for mule deer, and mast producing species for black bear and mule deer; 2) estimate forage quality in treated and untreated areas; and 3) determine habitat selection and space use patterns of large mammals in relation to forest restoration treatments and wildfire including the time between restoration activities and use by large mammals. The results presented below are preliminary and mostly reflect data collected through 2018. Data collected during the project extension from 2021 to 2023 are currently being analyzed and we expect that the final report for large mammal monitoring will be completed during summer of 2025.

### Study animal capture and handling

We captured and fitted GPS/VHF collars on 156 adult female elk, 53 adult female mule deer, 29 female and 49 male adult black bear, and 23 female and 21 male subadult or adult mountain lions to identify movements and assess habitat selection patterns in relation to wildfires and forest restoration treatments (Fig. 8.1.1). Additionally, through collaborations with New Mexico Department of Game and Fish and Santa Ana Pueblo we were provided with GPS collar data from 21 mountain lions from the greater Jemez Mountains.

### Habitat Selection

***Elk*** – We used selection ratios to assess selection or avoidance of wildfires and restoration treatments. Selection ratios above 1 indicate selection for a treatment/disturbance class whereas selection ratios less than 1 indicate avoidance. At the landscape scale, selection ratios indicated that elk selected for areas treated with prescribed burns during the first 6 months following treatment but avoided older prescribed burns. At the within home range scale, elk selected areas treated with prescribed fire during the first 1-2, and 4 years following prescribed fire, then avoided prescribed burn areas ≥ 5 after treatment at the within home range scale (Table 8.1.1). Selection ratios indicated that elk avoided thinned areas at the landscape scale and the strength of avoidance increased with treatment age (Table 8.1.2). Conversely, at the within home range scale, elk selected for thinned areas 1-2 years after treatment but then avoided thinned areas > 2 years post-treatment and thereafter (Table 8.1.2).

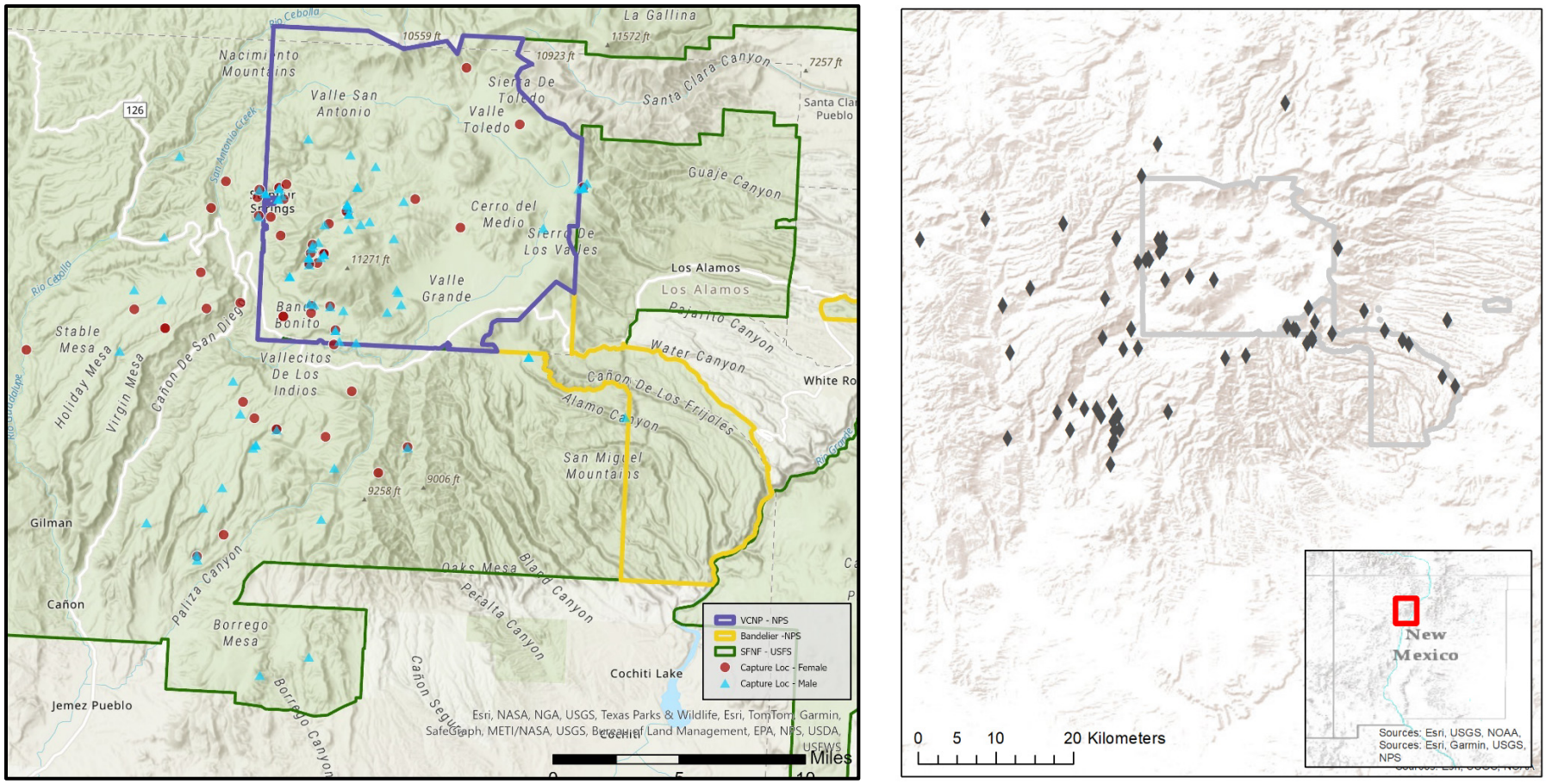



Figure 8.1.1. Capture locations for black bears (left) and mountain lions (right).

Table 8.1.1. Elk selection ratios at the landscape and within home range scales for prescribed burns by time since treatment, 2012-2018.

| Landscape Scale | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Untreated | Burned RX | | | | | | | | |
| Age (years) | | 0.5 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 |
| Selection | 0.63 | 1.90 | 0.89 | 0.59 | 0.01 | 0.20 | 0.11 | 0.04 | 0.00 | 0.00 |
| SE | 0.002 | 0.06 | 0.04 | 0.03 | 0.005 | 0.033 | 0.02 | 0.01 | | |
| Average | | 0.42 | | | | | | | | |
| | | | | | | | | | | |
| Home Range Scale | | | | | | | | | | |
| Age (years) | | 0.5 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 |
| Selection | 0.91 | 0.97 | 1.16 | 1.14 | 0.12 | 1.09 | 0.59 | 0.09 | 0.00 | 0.00 |
| SE | 0.003 | 0.030 | 0.055 | 0.059 | 0.044 | 0.187 | 0.135 | 0.031 | | |
| Average | | 0.57 | | | | | | | | |

Table 8.1.2. Elk selection ratios at the landscape and within home range scales for thinned areas by time since treatment, 2012-2018.

| Landscape Scale | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Untreated | Thinned | | | | | | | | |
| Age (years) | | 0.5 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 |
| Selection | 0.63 | 0.16 | 0.86 | 0.53 | 0.05 | 0.12 | 0.03 | 0.04 | 0.00 | 0.00 |
| SE | 0.002 | 0.01 | 0.04 | 0.03 | 0.011 | 0.020 | 0.01 | 0.02 | | |
| Average | | 0.20 | | | | | | | | |
| | | | | | | | | | | |
| Home Range Scale | | | | | | | | | | |
| Age (years) | | 0.5 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 |
| Selection | 0.91 | 0.58 | 2.17 | 1.52 | 0.15 | 0.50 | 0.21 | 0.13 | 0.00 | 0.00 |
| SE | 0.003 | 0.047 | 0.123 | 0.111 | 0.034 | 0.086 | 0.105 | 0.089 | | |
| Average | | 0.58 | | | | | | | | |

For wildfire burned areas, selection ratios at the landscape scale indicated initial avoidance of low and moderate severity burns at 6 months post fire, but strong selection of high severity burns (Table 8.1.3). After 1-year post fire, elk strongly selected for wildfire burned areas of all burn severities and selection strength increased with fire age until about 6 years post-fire (Table 8.1.3). Similarly, at the within home range scale, elk avoided low and moderate severity burns within 6 months of the fire, but strongly selected for young high severity fires (Table 8.1.4). At 1-2 years post fire, elk continued to select for moderate and high severity burns, but selection of all burn severities began to diminish for all severities after 3 years at the within-home range scale (Table 8.1.4).

Table 8.1.3. Elk selection ratios at landscape scale for selection of wildfire burned areas by time since fire and burn severity, 2012-2018.

| Age (years) | Severity | Selection Ratio | SE | Average across severities | Average across ages and severities |
|---|---|---|---|---|---|
| 0.5 | Low | 0.80 | 0.022 | | |
| | Med | 0.79 | 0.041 | | |
| | High | 2.01 | 0.098 | 1.20 | |
| 1 | Low | 1.44 | 0.027 | | |
| | Med | 1.86 | 0.050 | | |
| | High | 2.32 | 0.058 | 1.87 | |
| 2 | Low | 1.97 | 0.026 | | |
| | Med | 1.85 | 0.037 | | |
| | High | 2.05 | 0.037 | 1.96 | |
| 3 | Low | 1.03 | 0.017 | | |
| | Med | 0.98 | 0.022 | | |
| | High | 1.13 | 0.021 | 1.05 | |
| 4 | Low | 2.31 | 0.024 | | |
| | Med | 2.00 | 0.030 | | |
| | High | 2.16 | 0.030 | 2.16 | |
| 5 | Low | 1.72 | 0.021 | | |
| | Med | 1.77 | 0.030 | | |
| | High | 2.02 | 0.030 | 1.84 | |
| 6 | Low | 3.04 | 0.047 | | |
| | Med | 2.03 | 0.042 | | |
| | High | 2.02 | 0.034 | 2.36 | |
| 7 | Low | 1.73 | 0.037 | | |
| | Med | 1.30 | 0.035 | | |
| | High | 1.14 | 0.027 | 1.39 | 1.73 |

Table 8.1.4. Elk selection ratios at the with-home range scale for selection of wildfire burned areas by time since fire and burn severity, 2012-2018.

| Age (years) | Severity | Selection Ratio | SE | Average across severities | Average across ages and severities |
|---|---|---|---|---|---|
| 0.5 | Low | 0.65 | 0.018 | | |
| | Med | 0.59 | 0.031 | | |
| | High | 1.65 | 0.079 | 0.96 | |
| 1 | Low | 0.83 | 0.015 | | |
| | Med | 1.32 | 0.035 | | |
| | High | 3.19 | 0.082 | 1.78 | |
| 2 | Low | 1.12 | 0.014 | | |
| | Med | 1.29 | 0.025 | | |
| | High | 1.76 | 0.031 | 1.39 | |
| 3 | Low | 0.89 | 0.014 | | |
| | Med | 0.98 | 0.022 | | |
| | High | 1.12 | 0.021 | 1.00 | |
| 4 | Low | 1.29 | 0.013 | | |
| | Med | 1.13 | 0.016 | | |
| | High | 1.11 | 0.014 | 1.18 | |
| 5 | Low | 1.01 | 0.012 | | |
| | Med | 1.00 | 0.017 | | |
| | High | 1.11 | 0.016 | 1.04 | |
| 6 | Low | 1.24 | 0.018 | | |
| | Med | 0.92 | 0.018 | | |
| | High | 0.94 | 0.015 | 1.03 | |
| 7 | Low | 0.86 | 0.018 | | |
| | Med | 0.77 | 0.200 | | |
| | High | 0.64 | 0.015 | 0.76 | 1.14 |

***Mule deer*** – At the landscape scale, we observed differences in mule deer habitat selection of vegetation types and treatments in association with seasonal elevation movements. Winter habitat selection by mule deer were unrelated to prescribed burns and thinning treatments because the treatments were located at higher elevations than mule deer winter ranges. During spring, mule deer selected wildfire-burned areas, prescribed burns <2 years old, and thinned areas, with stronger selection for thinned areas ≥5 years old (Fig. 8.1.2). In summer, collared mule deer avoided wildfire-burned areas, selected prescribed-burned areas, avoided thinned areas <5 years old, and strongly selected thinned areas ≥5 years old compared to unburned or untreated areas. In fall, mule deer avoided wildfire-burned areas and selected for prescribed burns <2 years old, avoided thinned areas <5 years old, but strongly selected for thinned areas ≥5 years old (Fig. 8.1.2)

Habitat selection by female mule deer was similar at the within home range scale. Winter home ranges of collared mule deer largely did not overlap restoration treatments or wildfire-burned areas. During spring, mule deer selected for wildfire-burned areas and prescribed burns compared with unburned areas, and thinned versus unthinned areas (Fig. 8.1.3). In summer, mule deer more strongly selected for wildfire and prescribed burn areas than unburned areas, they avoided thinned areas <5 years old, and selected for thinned areas ≥5 years old. During the fall, mule deer avoided wildfire-burned areas, selected areas treated with prescribed fire, particularly burns <2 years old, and selected older thinned areas (≥ 5 years old) over unburned and unthinned areas, respectively (Fig. 8.1.3).

At both the landscape and home-range scale, grasslands were avoided during most seasons, pinyon-juniper woodlands were selected in winter, and oak vegetation and mixed-conifer forests were selected during summer. Our data suggests that mule deer may benefit from recent prescribed burns and older forest thinning, but the duration of post-treatment vegetation recovery influences the strength and direction of selection.

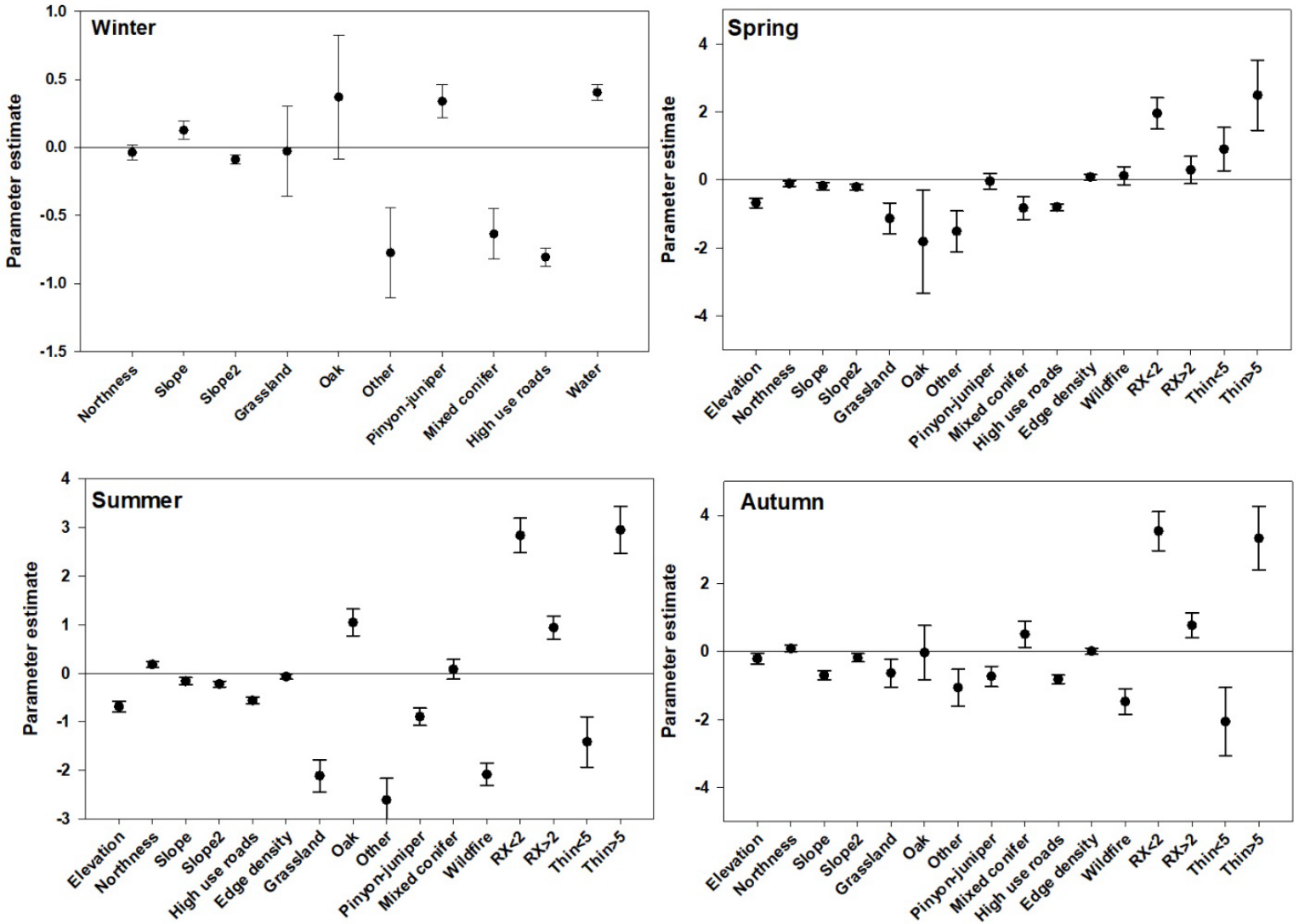



Figure 8.1.2. Parameter estimates from the top models for seasonal landscape-scale (2nd order) habitat selection of female mule deer. Error bars represent the 95% confidence intervals. Covariates: Roads = distance to high use roads, Water = distance to nearest perennial water source, Wildfire = wildfire burned areas, RX<2 = prescribed burn less than 2 years old, RX≥2 = prescribed burn greater than 2 years old, Thin<5 = thinning less than 5 years old, Thin≥5 = thinning greater than 5 years old.

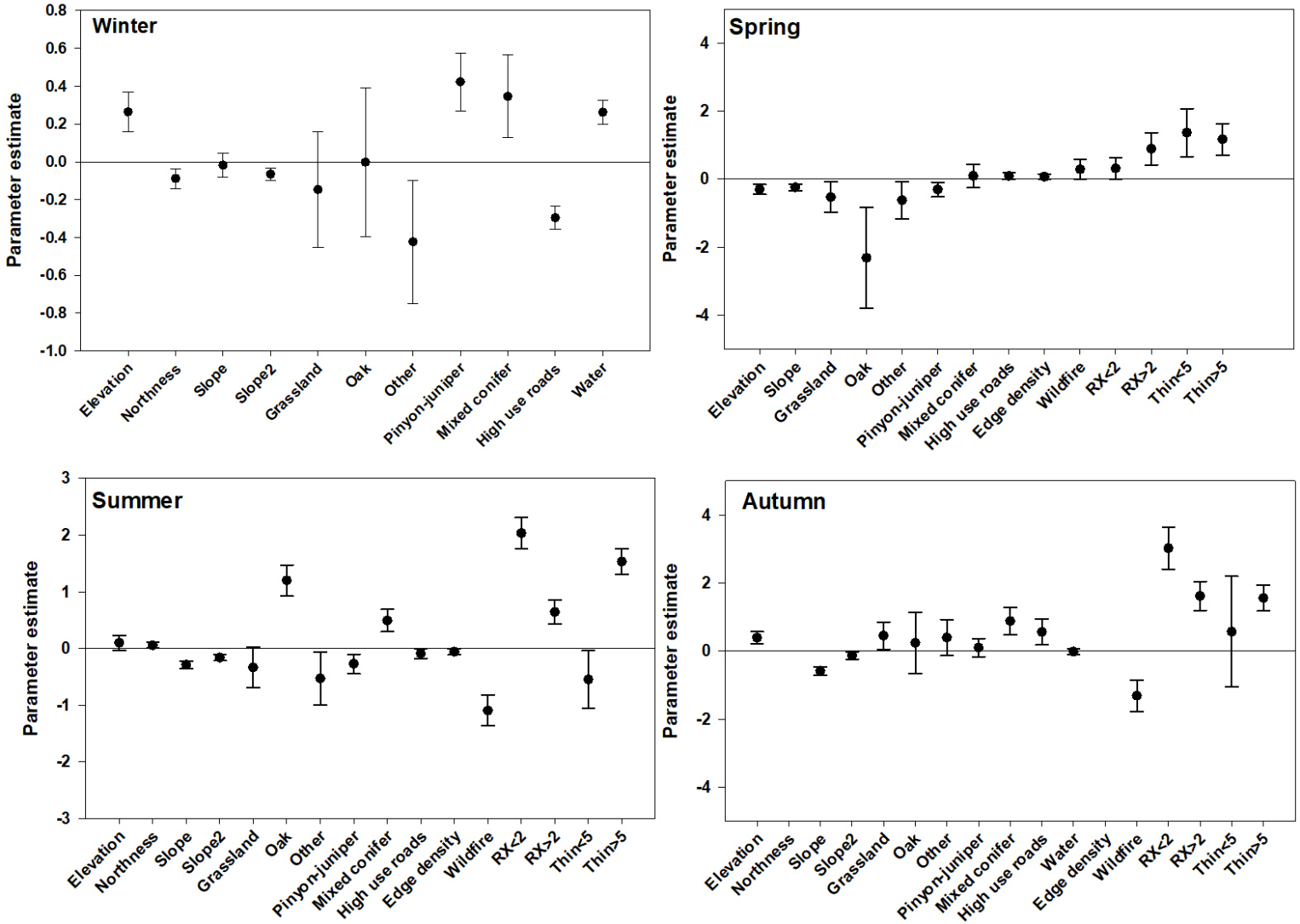



Figure 8.1.3. Parameter estimates from the top models for seasonal within-home range scale (3rd order) habitat selection of female mule deer. Error bars represent the 95% confidence intervals. Covariates: Roads = distance to high use roads, Water = distance to nearest perennial water source, Wildfire = wildfire burned areas, RX<2 = prescribed burn less than 2 years old, RX≥2 = prescribed burn greater than 2 years old, Thin<5 = thinning less than 5 years old, Thin≥5 = thinning greater than 5 years old.

***Black bears*** – We investigated habitat selection of American black bears at multiple spatial scales, including landscape scale habitat selection, within home range habitat selection and selection of day beds and den sites relative to wildfires and restoration treatments.

**Bed and den site selection**.

We investigated bed and den site selection of black bears using GPS location data and a use/available study design to assess the influence of habitat characteristics, including wildfires, prescribed burns, and thinning treatments. Black bears used all disturbed sites to varying degrees, although 48% of bed sites were located in undisturbed habitat while only 11% and 2% of bed sites were located in thinned and prescribed burn sites, respectively. Thirty-nine percent of bed sites were located in previous wildfire-burned locations but 67% of these were in areas with low burn severity. The most supported models suggested that black bears were more likely to select bed sites with a combination of low horizontal visibility and high stand basal area (Table 8.1.5). Thirty-eight percent of den sites were located in disturbed habitat, 8 den sites were in areas burned by wildfires. The highest-ranking model for den selection indicated that black bears were more likely to select den sites with low horizontal visibility (Table 8.1.6). Accounting for the timing, size, and proximity of future restoration would aid in mitigating potential short-term negative effects on black bear bed and den site selection.

Table 8.1.5. Parameter estimates, standard errors, and *P*-values for the five highest ranking models for black bear bed site selection.

| Model No. | Covariate | Estimate | *SE* | *P* |
|---|---|---|---|---|
| 12 | Visibility | -0.007 | 0.002 | 0.002 |
| | Basal Area | 0.013 | 0.005 | 0.004 |
| 13 | Visibility | -0.007 | 0.002 | 0.003 |
| | Canopy | 0.002 | 0.003 | 0.532 |
| | Basal Area | 0.012 | 0.005 | 0.014 |
| 14 | Visibility | -0.008 | 0.002 | 0.002 |
| | Hard Mast | 0.047 | 0.023 | 0.041 |
| 8 | Visibility | -0.008 | 0.002 | 0.002 |
| 4 | Basal Area | 0.014 | 0.005 | 0.003 |

[a]Visibility = percent horizontal visibility; Basal Area = stand basal area $m^2$/ha; Canopy = percent canopy cover; Hard Mast = average percent cover of hard mast species.

Table 8.1.6. Parameter estimates, standard errors, and *P*-values for the five highest ranking models for black bear den site selection.

| Model No. | Covariate | Estimate | *SE* | *P* |
|---|---|---|---|---|
| 8 | Visibility | -0.010 | 0.004 | 0.006 |
| 13 | Visibility | -0.010 | 0.004 | 0.008 |
| | Canopy Cover | 0.003 | 0.010 | 0.733 |
| | Basal Area | 0.003 | 0.025 | 0.903 |
| 39 | Visibility | -0.010 | 0.005 | 0.054 |
| | Northness | -0.007 | 0.332 | 0.982 |
| | Visibility × Sex | 0.001 | 0.008 | 0.934 |
| | Northness × Sex | 0.136 | 0.454 | 0.764 |
| 34 | Visibility | -0.010 | 0.005 | 0.054 |
| | Basal Area | 0.003 | 0.023 | 0.896 |
| | Visibility × Sex | -0.001 | 0.008 | 0.982 |
| | Basal Area × Sex | -0.003 | 0.043 | 0.943 |
| 23 | Visibility | -0.010 | 0.005 | 0.006 |
| | Road | 0.00009 | 0.001 | 0.929 |
| | Visibility × Road | -0.00001 | 0.0001 | 0.999 |
| | Sex × Road | 0.0001 | 0.001 | 0.932 |

[a]Visibility = percent horizontal visibility; Canopy = percent canopy cover; Basal Area = stand basal area $m^2$/ha; Northness = northly aspect value; Visibility × Sex = interaction effect between visibility and sex of bear; Northness × Sex = interaction effect between northness value and sex of bear; Basal Area × Sex = interaction effect between stand basal area $m^2$/ha and sex of bear; Road = distance to nearest road (m); Visbility × Road = interaction effect between visibility and the distance to nearest road (m); Sex × Road = interaction effect between the sex of bear and the distance to nearest road (m).

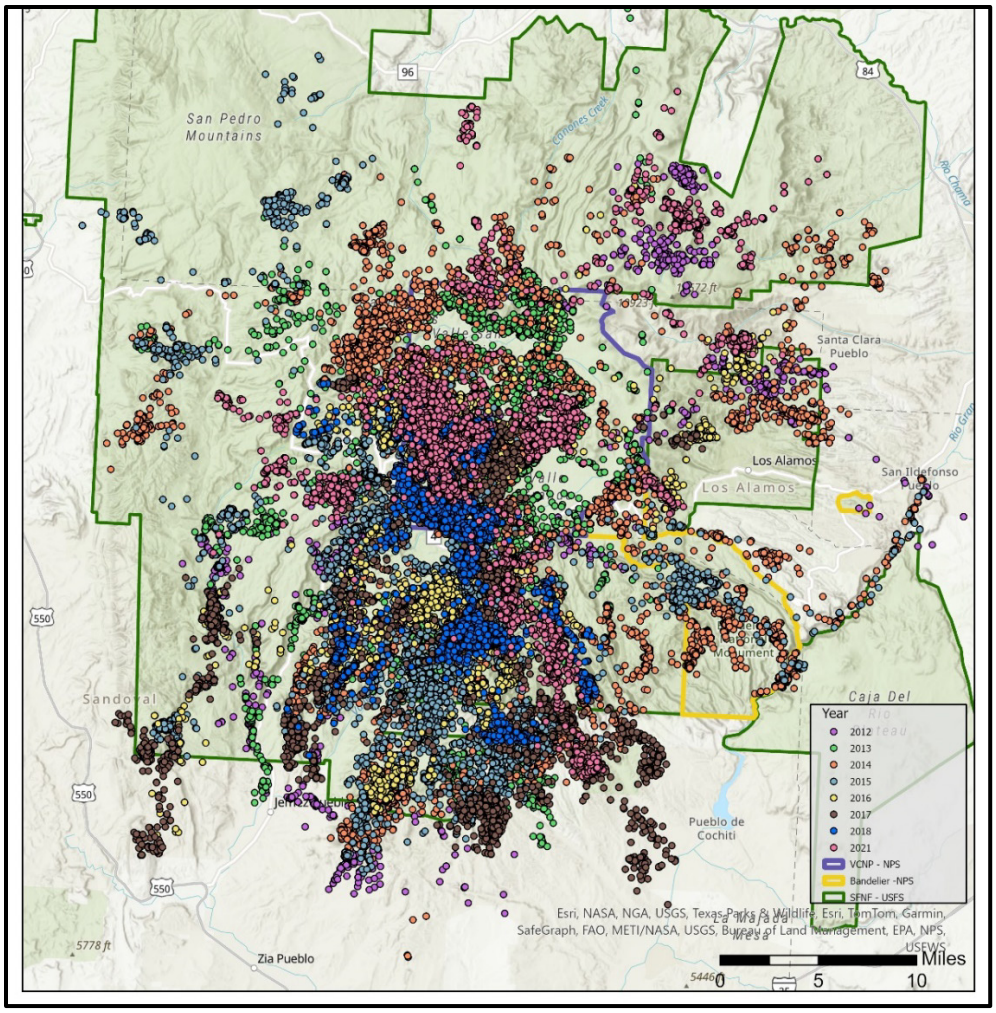



Figure 8.1.4. GPS collar data collected on black bears in southwest Jemez Mountains, 2012 to 2022.

**Landscape and within home range habitat selection**

We assessed the response of black bears to wildfires, prescribed burns, and forest thinning restoration treatments using GPS location data (Fig. 8.1.4) and habitat selection modelling. Our approach included the effects of time since wildfires, fire severity, and forest restoration treatments on bear habitat selection during spring (den emergence-pre/early green-up), summer (mating, dispersal), and fall (hyperphagia).

During spring, black bears selected areas that had recently experienced low-moderate severity fire and avoided areas where prescribed burns and thinning treatments occurred (Fig. 8.1.5). During all seasons, bears avoided areas where prescribed burns occurred ≥ 4 years ago, but during summer and fall bears selected areas that had more recently experienced prescribed burns (Fig. 8.1.6). Black bears also generally avoided all areas where thinning treatments occurred across all seasons. During summer, black bears selected areas that had most recently been burned by wildfires and prescribed burns, but avoided these areas of recent disturbances in the fall. During fall, bears avoided areas that were recently burned by wildfires (Fig. 8.1.6).

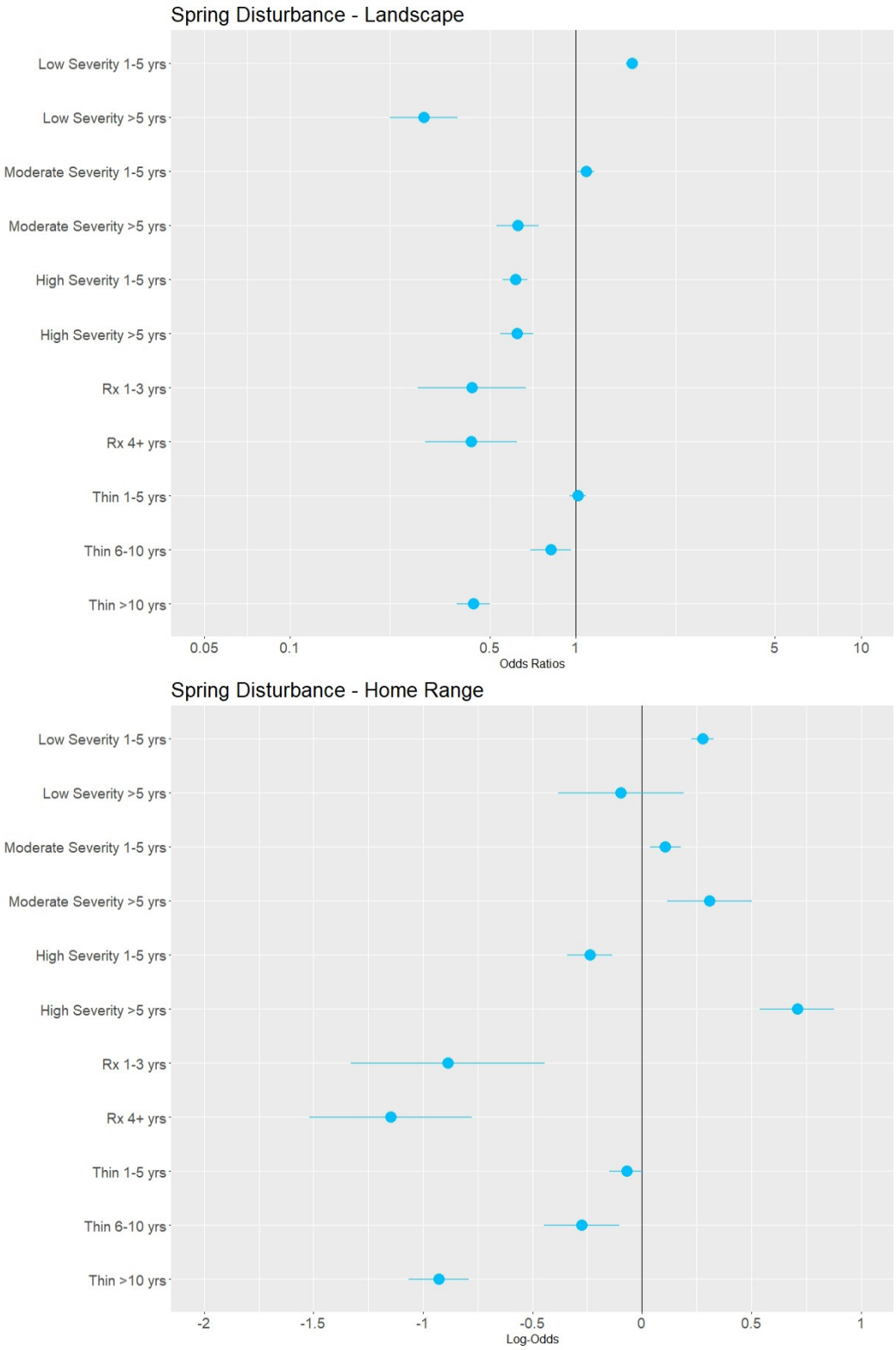



Figure 8.1.5. Odds ratios for black bear habitat selection in relation to wildfire and restoration treatments during spring at the landscape (upper panel) and within home range (lower panel) scales.

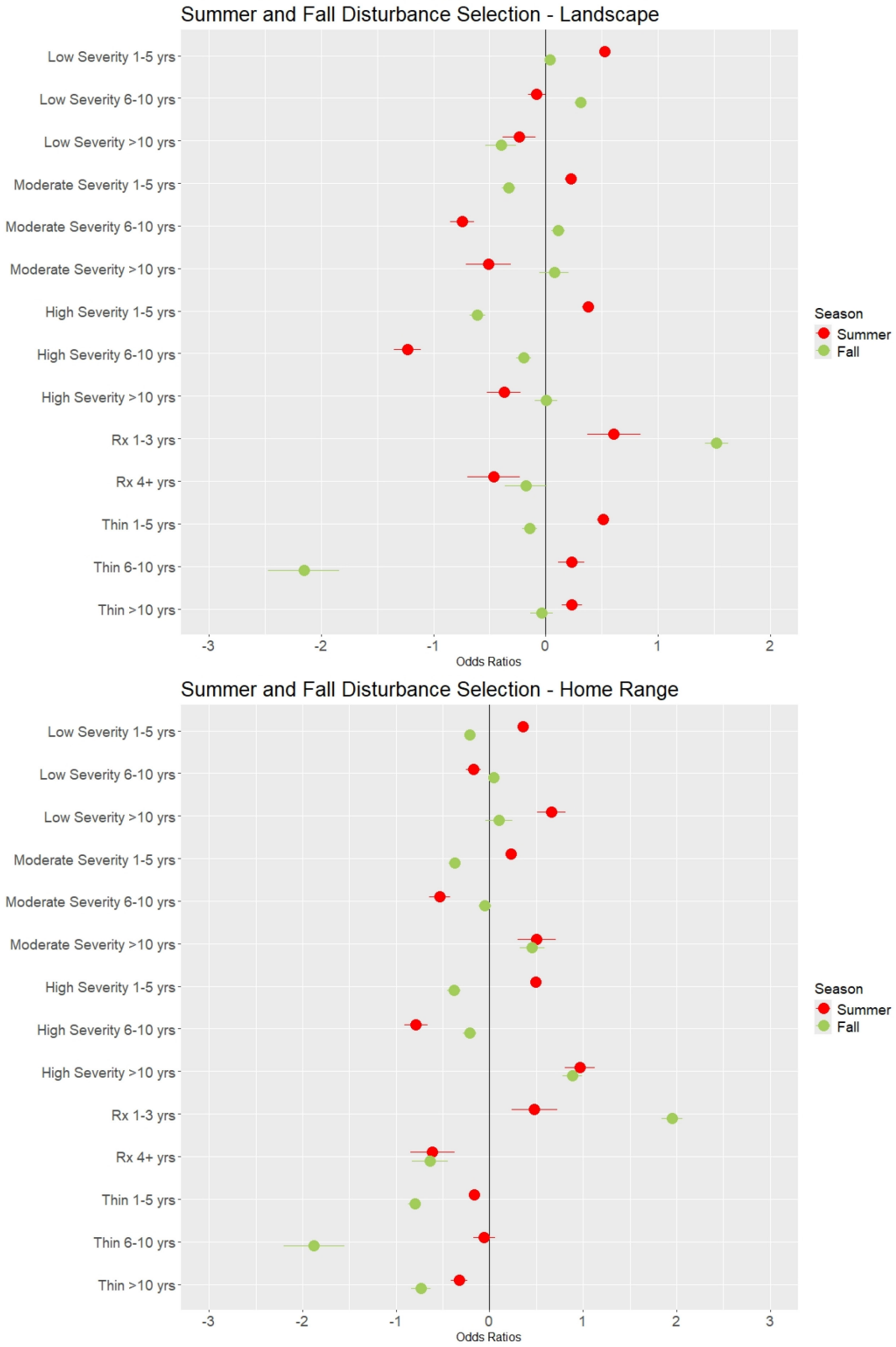



Figure 8.1.6. Odds ratios for black bear habitat selection in relation to wildfire and restoration treatments during summer and fall at the landscape (upper panel) and within home range (lower panel) scales.

These results indicate black bears increased their use and selection of areas that experience fire-based disturbances (prescribed burns, wildfires), generally avoiding forest thinning treatments. The responses to these disturbances varied by the time since disturbance, wildfire severity, season, and magnitude of effect. We hypothesize that in the attempt to reduce large severity fires and extreme fire behavior, a liberal reduction in ladder fuels such as small trees, shrubs, and low-hanging branches may be creating unfavorable habitat conditions for black bears, at least in the short-term. Bed and den site models suggest that black bears were more likely to select bed sites with a combination of low horizontal visibility and high stand basal area. Black bears may benefit from a heterogeneous landscape composed of a mosaic of wildfire, prescribed burns, and potentially more conservative forest thinning treatments. This mosaic creates diverse plant communities across the landscape that bears can benefit from by using different patches for forage, cover, and other needs across all seasons as resource availability and nutritional demands change.



Capture and collaring of black bear in Jemez Mountains.

***Mountain lions*** – We tracked individual mountain lions for 43 – 1695 days (Fig. 8.1.7). We found that humans (e.g. hunting, depredation, and vehicles) were the leading cause of death for both subadult and adult mountain lions followed by intraspecific killing. Litter sizes ranged from 1 – 3 kittens, with dispersal documented at 13 – 24 months of age. Home range sizes were highly variable but followed a general trend seen elsewhere with smaller female ranges and larger male ranges (Fig. 8.1.8).

To investigate mountain lion habitat use, we categorized their activities into four key behavioral states: day resting locations, feeding locations, traveling locations, and den sites/nursery areas. This approach recognizes that individual mountain lions may utilize their environment differently depending on their specific needs at any given time. By examining these distinct activities, we aimed to gain a comprehensive understanding of how mountain lions interact with their habitat, both in treated areas (e.g., vegetation managed for ecological purposes) and untreated areas.

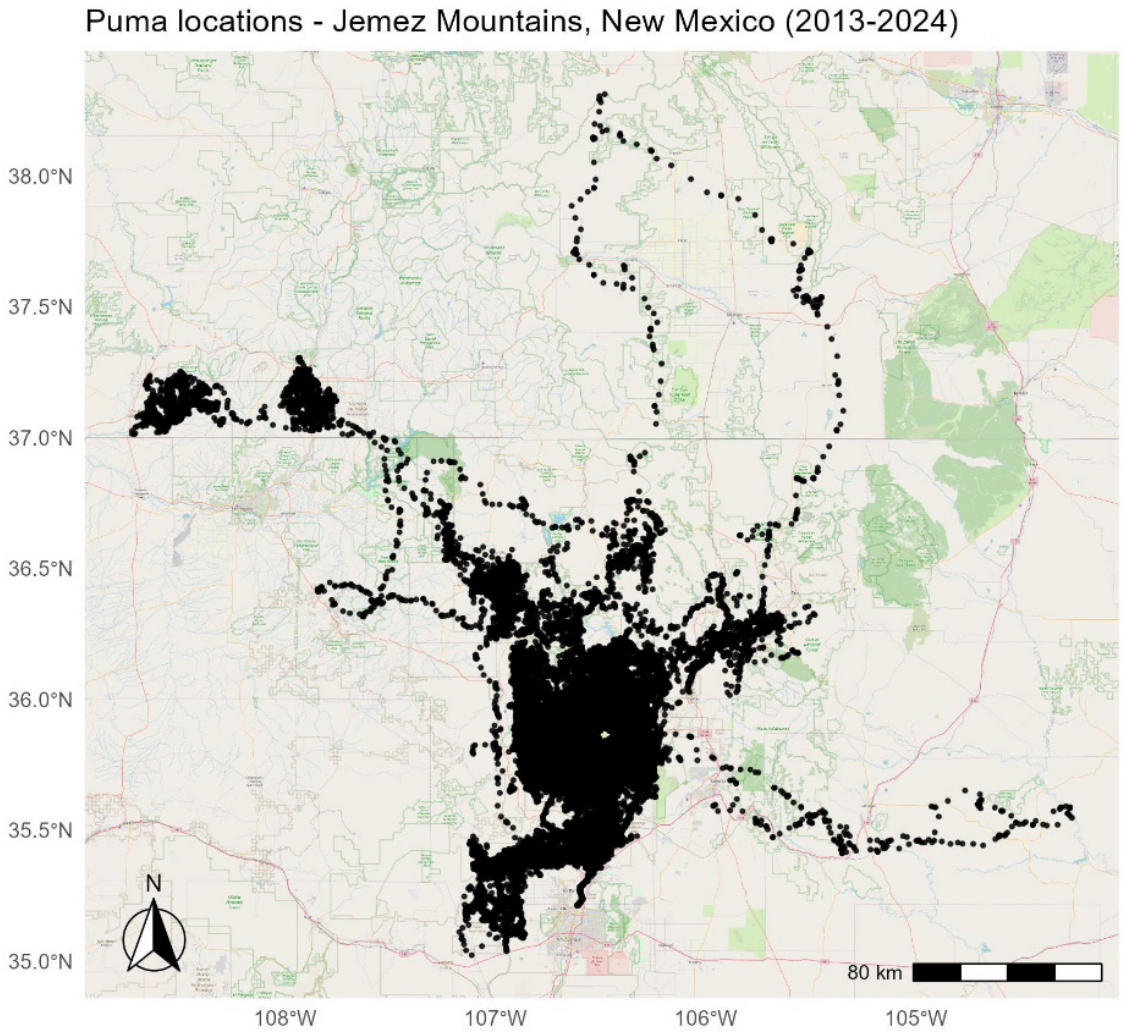



Figure 8.1.7. GPS locations collected on collared mountain lions captured in the Jemez Mountains, New Mexico between 2013 and 2024.

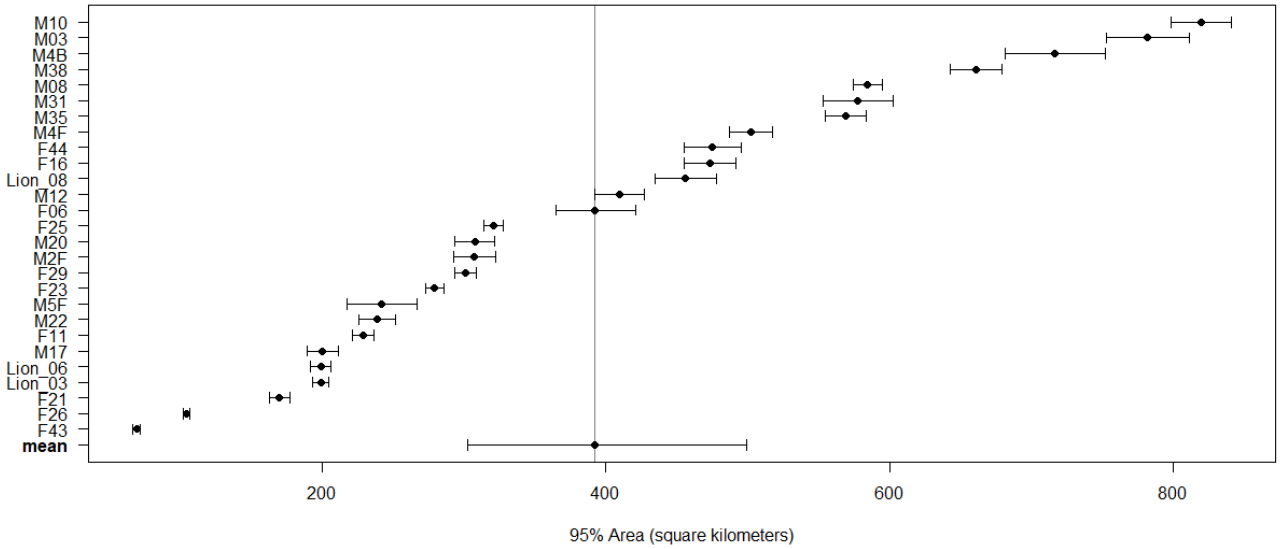



Figure 8.1.8. Home range estimates (KDE href) for resident adult pumas in the Jemez Mountains, New Mexico between 2013 and 2024.

Day resting locations, or day beds, were typically found in rugged terrain that offered high levels of concealment and overhead cover, providing both a secure and shaded environment. These areas were typically used between 6 – 12 hours, either as a mid-day stopping point during movement activities, or as a resting location between feeding bouts. Feeding activities revealed that mule deer and elk were the primary prey species, with kills occurring in both treated and untreated vegetation. Notably, mountain lions often dragged their kills a short distance to areas with dense cover, where they cached the prey likely to protect the kill from scavengers and reduce the risk of detection by competitors, thereby extending the time mountains lions have to consume their kill.

Den sites were predominantly located in steep, rugged terrain with a high proportion of rock and thick impenetrable vegetation. The den site structures were most frequently located under large boulders, or within erosion crevices along cliff walls in canyons. These locations provide natural concealment and protection from predators and environmental threats. We found that female mountain lions frequently moved their kittens between multiple dens during the early developmental stages, potentially to minimize risks from competitors and allowing the mother to remain mobile while accessing critical resources such as prey.

All behavioral categories (movement, day resting, feeding, and denning) were observed in both thinned and unthinned forest vegetation, and in burned and unburned vegetation. Continued analysis of mountain lion movement data will identify if mountain lions display selection or avoidance of treatments.




Collared mountain lion captured on game camera used to monitor scavengers at kill sites.

**Ungulate Migrations**

Elk captured as part of this study area reside primarily in and around the Valles Caldera National Preserve (VCNP) and along the mesas to the north and west of VCNP. These elk are part of a partially migratory population, with residents remaining in the VCNP year-round, and migrant individuals traveling to the surrounding lower elevation slopes depending on the year and snowpack levels. Migrants exhibit two different seasonal movement patterns. The first movement pattern occurs during midwinter (January–February) when numerous elk move to the lower elevation slopes of the VCNP, primarily southeast towards Bandelier National Monument (BNM; Fig. 8.1.9). The second movement is more typical of a spring and fall migratory movement; elk migrate north to La Grulla Plateau for the summer and return to the VCNP for the winter (Fig. 8.1.9). Elk moving southeast through BNM traverse the Las Conchas, Cerro Grande, and Cerro Pelado burn scars. A few individuals migrating to the southwest go through thinned areas on Banco Bonito and prescribed burns along San Juan Mesa. Migrations averaged ~9 miles with a maximum migration of 17.5 miles. Detailed description is available in Kauffman et al. 2024.

Mule deer captured and monitored as part of this study are also partially migratory. These mule deer winter in the pinyon-juniper woodlands in the foothills south and east of Jemez Springs, near Ponderosa New Mexico (Fig. 8.1.10). Migratory mule deer migrated an average of 16.2 miles, to Blue Bird Mesa on the western edge of the Jemez Mountains, or to the Valles Caldera National Preserve. The central migration route follows the top Virgin Mesa, with deer utilizing numerous stopovers along Virgin Mesa while traversing large expanses of prescribed burns and thinned areas along the mesa. Mule deer that migrate to the Valles Caldera, traverse the extensive thinned areas in Banco Bonito; there is also a large mule deer stopover in the thinned areas in the Banco Bonito area (Fig. 8.1.10). The summer range is dominated by ponderosa pine and mixed-conifer forests, along with open grasslands in the Valles Caldera. The herd is only partially migratory, with some residents found on the winter range and on Lake Fork Mesa, west of the VCNP. Detailed descriptions of mule deer migration are in Kauffman et al. 2022.

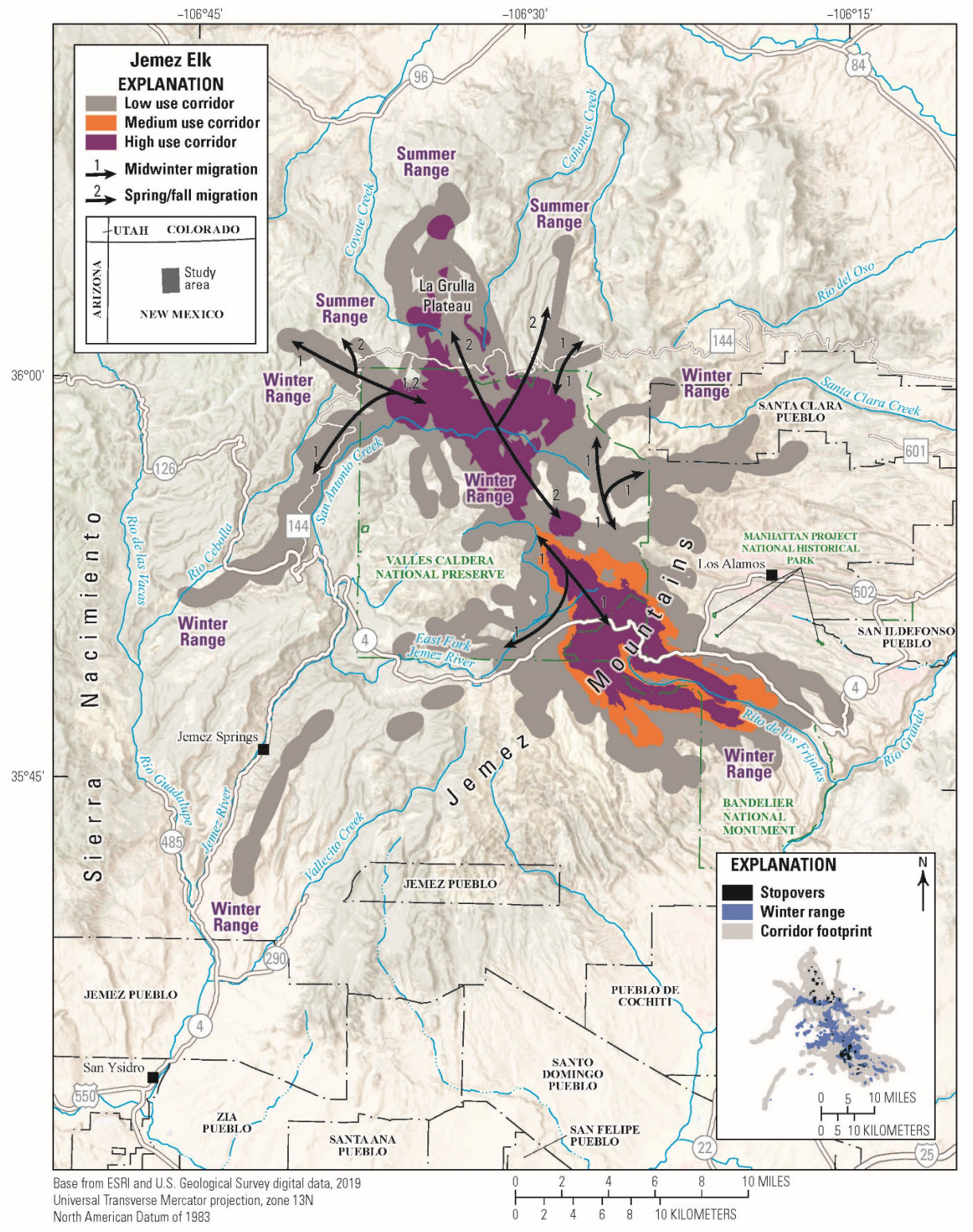



Figure 8.1.9. Migration routes, seasonal ranges, and stopover locations of elk in the southwest Jemez Mountains (Kauffman et al. 2024).

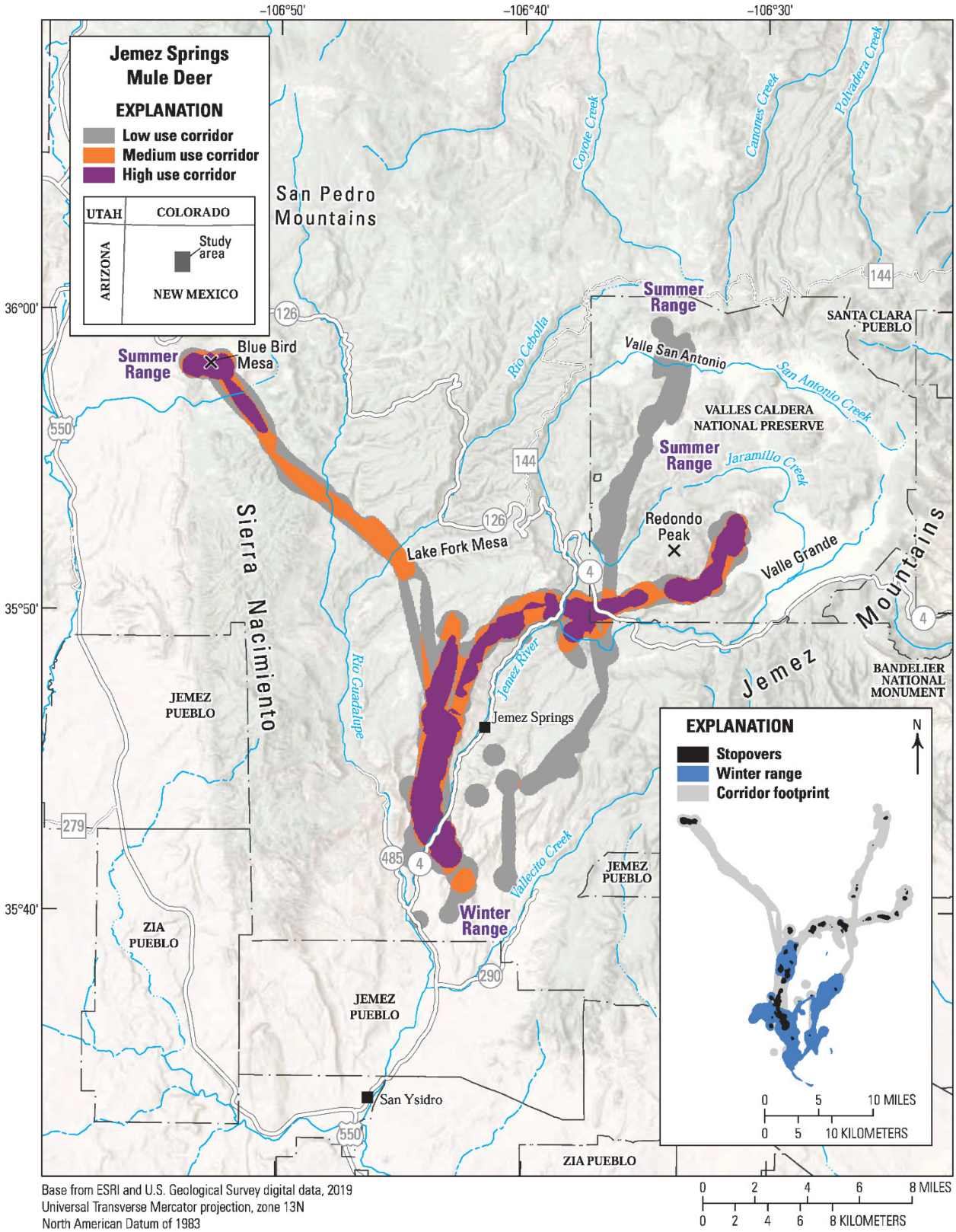



Figure 8.1.10. Migration routes, seasonal ranges, and stopover locations of mule deer in the southwest Jemez Mountains (Kauffman et al. 2022).

**Vegetation monitoring**

***Sampling and collection*** – We sampled forage and vegetation characteristics annually from 2013 to 2018 and 2021 to 2023. In total over 2,039 plots (Fig. 8.1.11) were sampled between 2013 and 2023 across 6 stand types and 5 treatment types, including 34 exclosures and their controls. From these plots, we collected > 10,000 forage samples for nutritional analyses. In addition, we collected over 1,000 herbaceous and 2,300 shrub samples to be used in biomass-predicting regressions. We collected over 470 seed bank samples in 2017 and 2018, and seed bank trials on those samples were completed in 2019. We have germinated and tracked over 6,400 seedlings from those trials.

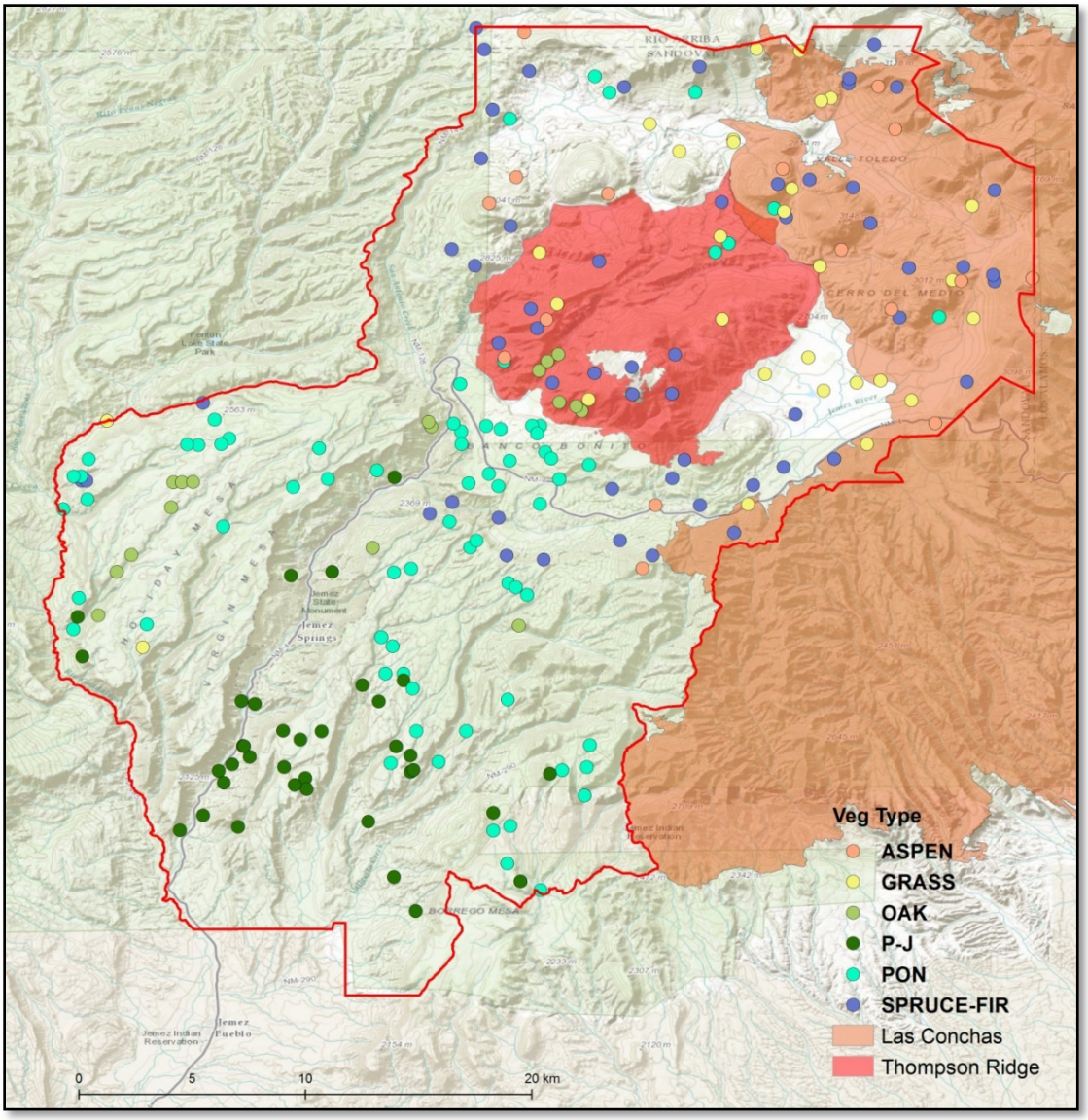



Figure 8.1.11. Location of long-term plots where samples for forage biomass and nutritional quality were collected. Plots are color-coded by stand type. Footprint of Las Conchas and Thompson Ridge fires are also shown.

***Forage quality*** – We collected 8,400 forage samples between 20013 and 2018 for nutritional content analysis (i.e., crude protein, lignin, ash, NDF, ADF, and other analyses). A further 1,600 samples were sent to the WSU Wildlife Habitat and Nutrition Lab (WHNL) for condensed tannin analyses. Lastly, 1,800 samples were analyzed for gross energy via bomb calorimetry. An additional 1,300 forage samples were collected from 2021 to 2023; samples are currently at the lab for nutritional content analysis.

Preliminary results regarding the nutritional quality of herbaceous and woody forage across stand types and treatments suggest crude protein levels may vary by both stand type and treatment type, lignin levels vary significantly ($P < 0.005$) as expected between herbaceous and woody species but not significantly across treatments or stand types, and mineral levels (via ash) vary significantly ($P < 0.005$) between herbaceous and woody species, and possibly by treatment in grassland plots.

Digestible energy content on a per hectare basis (kcal/ha) peaked with intermediate ages following wildfires for ponderosa and mixed conifer stands (Fig. 8.1.12), but total digestible energy of forage per hectare showed an increasing trend with time since prescribed fire (Fig. 8.1.13) and with thinning treatments (Fig. 8.1.13).

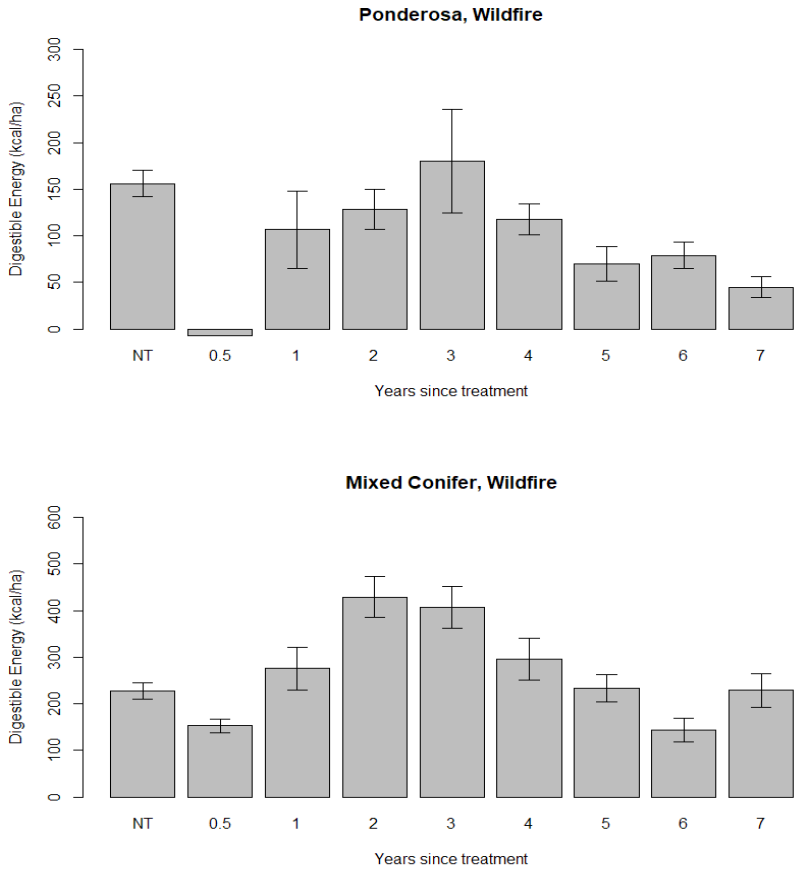



Figure 8.1.12. Forage digestible energy content on a biomass basis (kcal/ha) in relation to time since wildfire for ponderosa and mixed conifer stands.

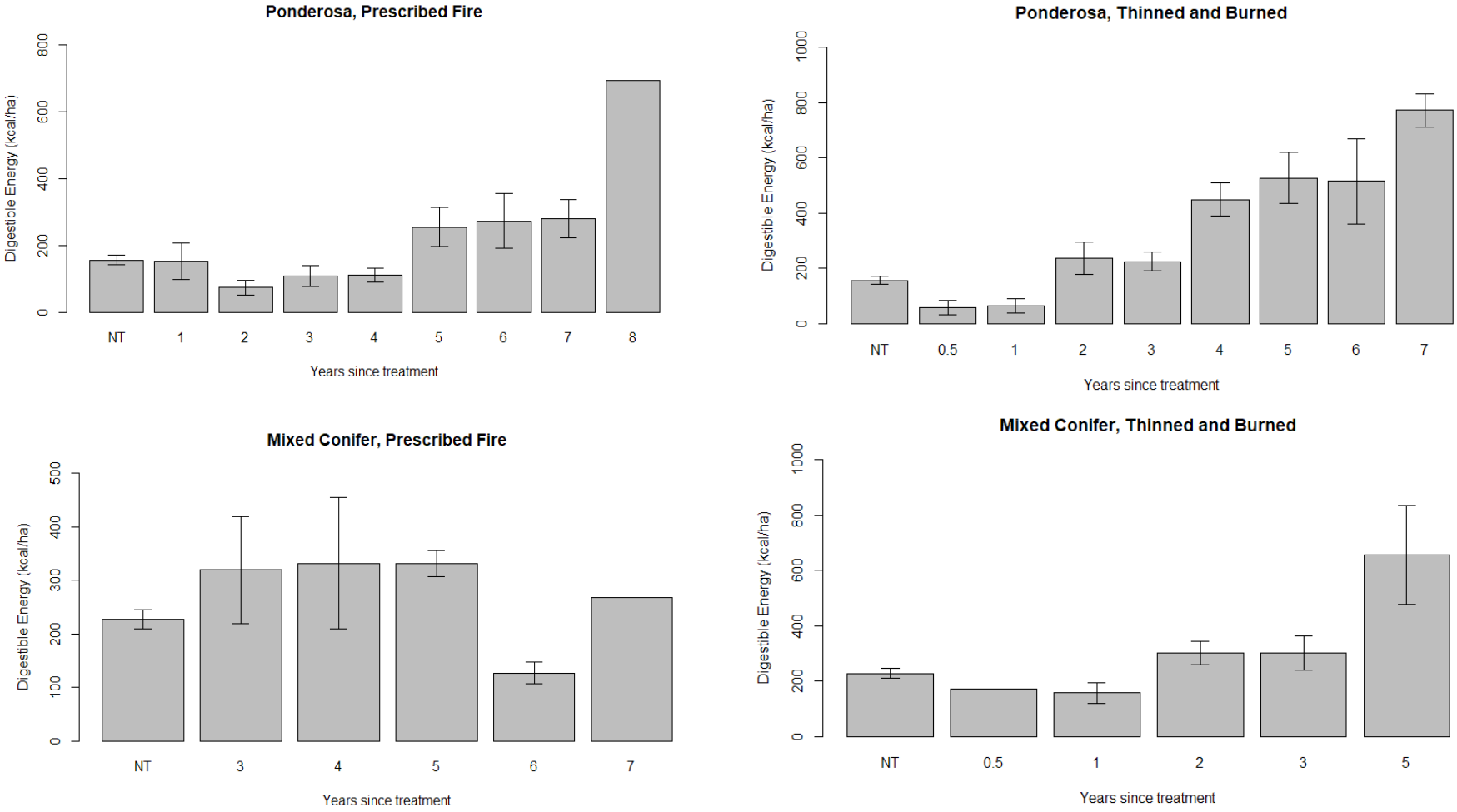



Figure 8.1.13. Forage digestible energy content on a biomass basis (kcal/ha) in relation to time since prescribed fire and thinning treatments for ponderosa and mixed conifer stands.

***Forage biomass*** – Preliminary analyses of changes in herbaceous and woody forage biomass over time (2013-2018) in untreated plots in the 6 dominant stand types (aspen, grassland, oak woodland, pinon-juniper, ponderosa, and spruce-fir) show no significant changes in measured biomass within stands over time, except aspen stands, though individual stands varied greatly in their forage availability. Woody biomass in untreated aspen stands decreased significantly over time ($P < 0.05$). Untreated grassland and spruce-fir stands consistently produced the greatest available herbaceous forage consistently, while oak shrublands consistently produced the greatest available woody forage. There was high variability in herbaceous or woody biomass across the 5 treatments (not treated, thinned, prescribed fire, wildlife, and both thinned and burned) due to high variability across stand types. However, when our results were broken out by stand type and time since treatment, some stand types differed between treated and untreated.

Biomass of herbaceous forage in ponderosa stands peaked 3 years after thinning, then declined (Fig. 8.1.14), whereas biomass of herbaceous forage was more variable for thinned

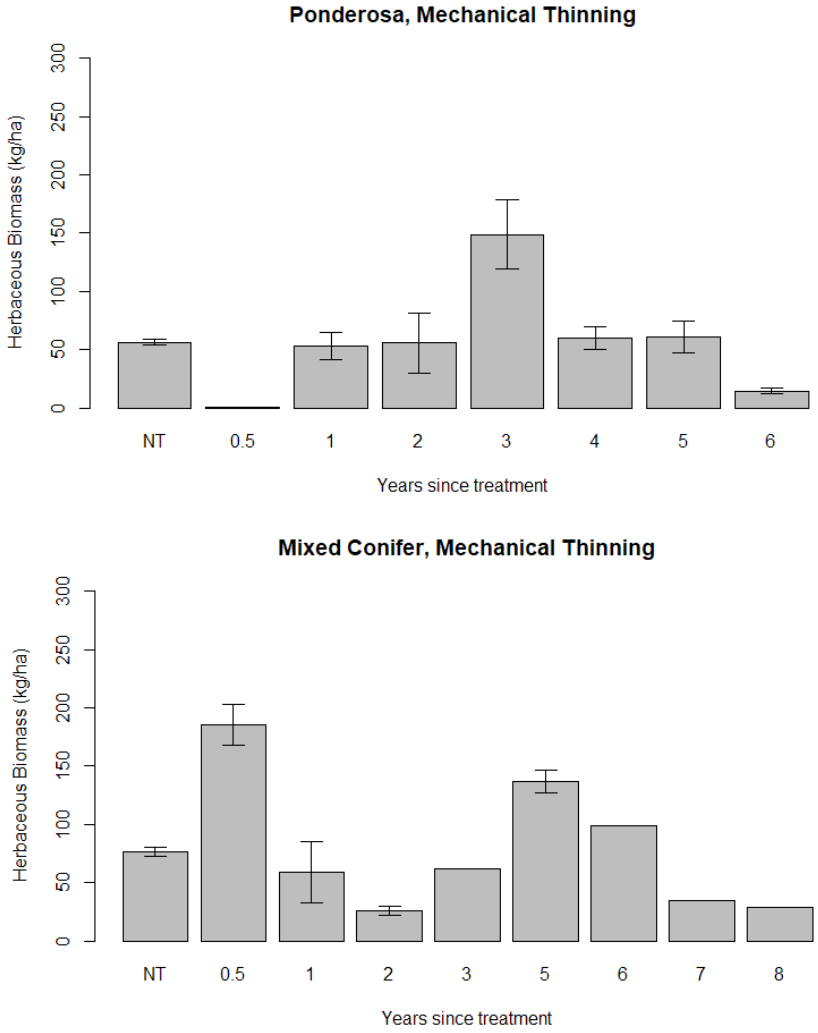



Figure 8.1.14. Biomass of herbaceous forage(kg/ha) in relation to time since thinning treatments for ponderosa and mixed conifer stands.

mixed conifer stands (Fig. 8.1.14). Biomass of herbaceous forage also peaked 3-4 years following both wildfires and prescribed fires in ponderosa and mixed conifer stands (Fig. 8.1.15).

Biomass of woody browse (kg/ha) peaked at intermediate ages of wildfires and prescribed burns for mixed conifer stands, with peaks at 2-3 years post wildfire and 4-5 years post prescribed fire (Fig. 8.1.16). Woody browse biomass in ponderosa stands was higher in untreated stands than those burned by wildfire, at least until 7 years post-fire (Fig. 8.1.16), whereas the biomass of woody browse in ponderosa stands treated with prescribed fires generally increased with highest biomass in stands > 5 years following prescribed fires (Fig. 8.1.16). Biomass of woody browse following thinning treatments initially decreased in both ponderosa and mixed conifer stands (Fig. 8.1.17), but increased above that in untreated stands after 6 years for ponderosa stands and 5 years in mixed conifer stands (Fig. 8.1.17).

All results presented above should be considered preliminary in nature and subject to change upon further analyses and addition of data on forage nutritional content from outstanding lab samples and the inclusion of additional plot samples collected between 2021 and 2023.

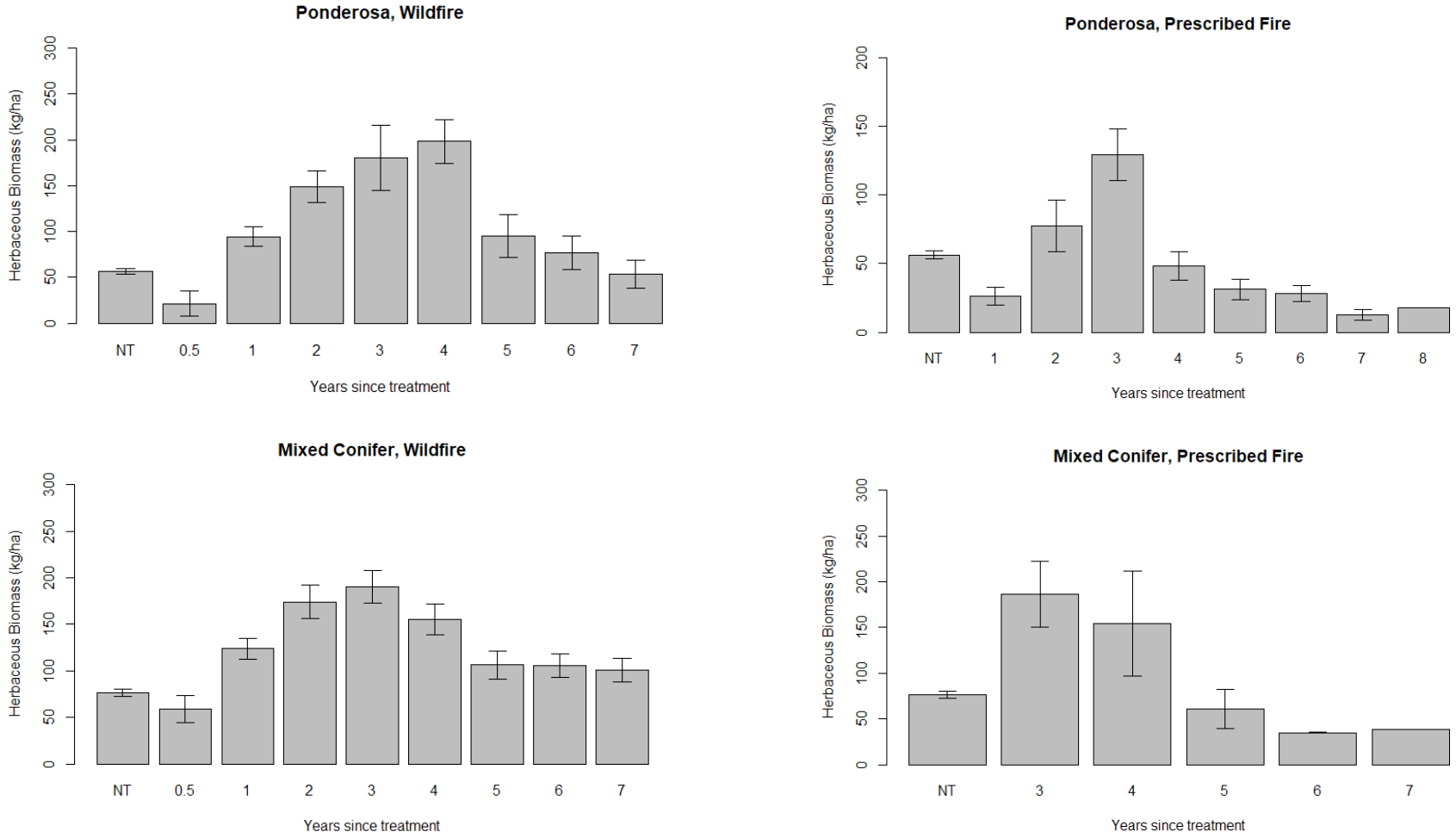



Figure 8.1.15. Biomass of herbaceous forage (kg/ha) in relation to time since wildfire and prescribed fires for ponderosa and mixed conifer stands.

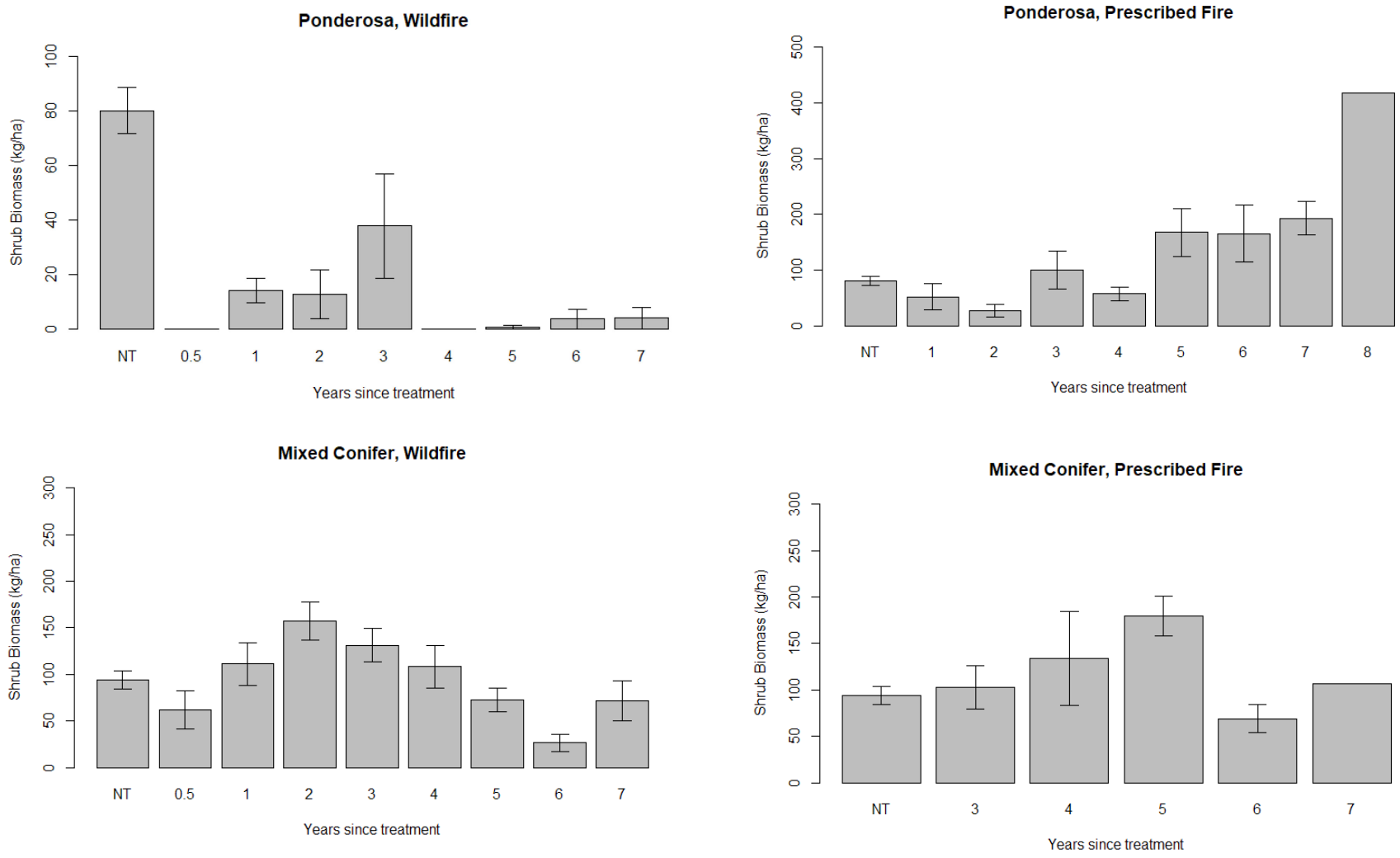



Figure 8.1.16. Biomass of woody browse (kg/ha) in relation to time since wildfire and prescribed fires for ponderosa and mixed conifer stands.

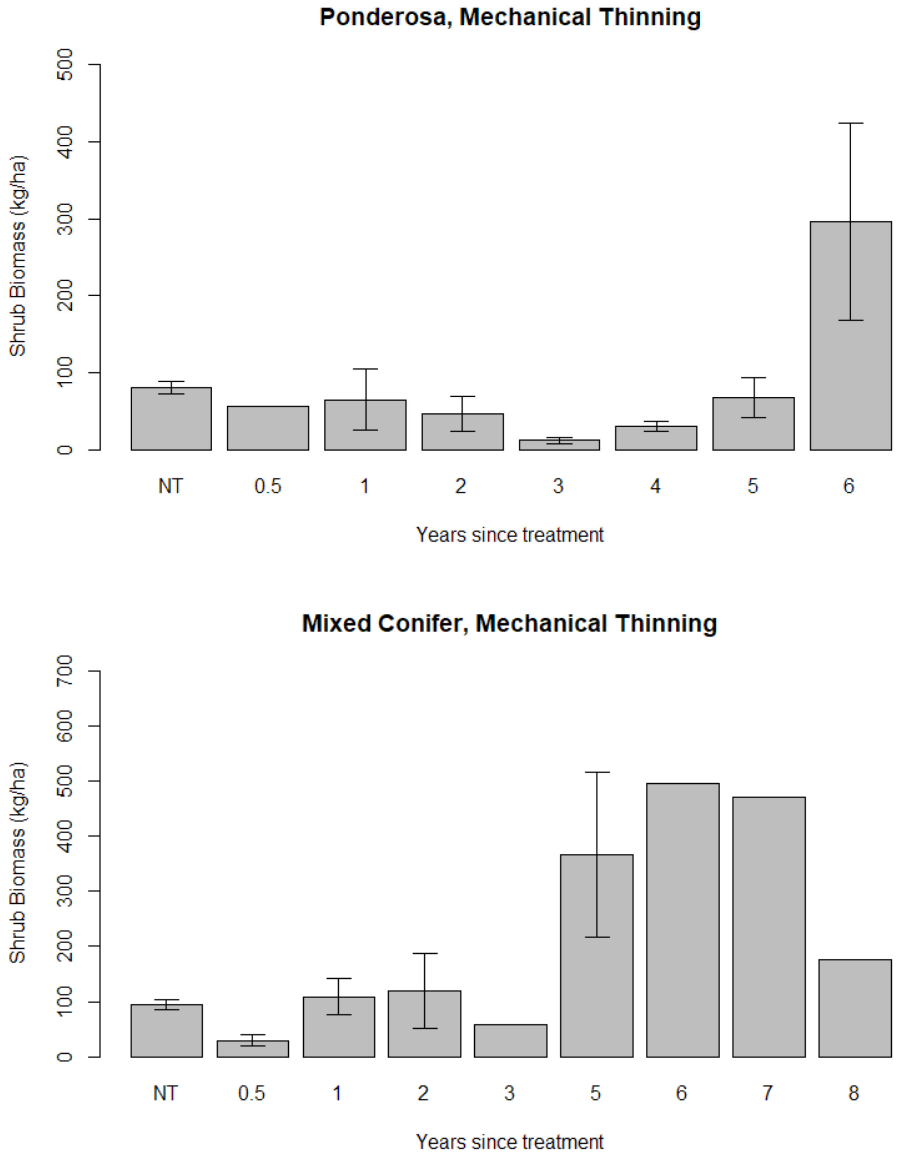



Figure 8.1.17. Biomass of woody browse (kg/ha) in relation to time since thinning treatments for ponderosa and mixed conifer stands.

**Community distribution and spatial interaction of large mammals**

We deployed 149 remote cameras to investigate the impacts of restoration activities on the distribution, habitat use, and co-occurrence of large mammals (i.e., mule deer (*Odocoileus hemionus*), elk (*Cervus canadensis*), black bears (*Ursus americanus*), mountain lions (*Puma concolor*), and coyotes (*Canis latrans*) (Fig. 8.1.18). We placed cameras proportionally over the range of restoration treatments and wildfires in the region and processed images to identify animals. We estimated relative habitat use following the approach described by Royle and Nichols (2003) to evaluate whether the disturbance history influences species' probability of use, specifically accounting for type of disturbance, time since treatment, and landscape metrics of disturbance type and severity.

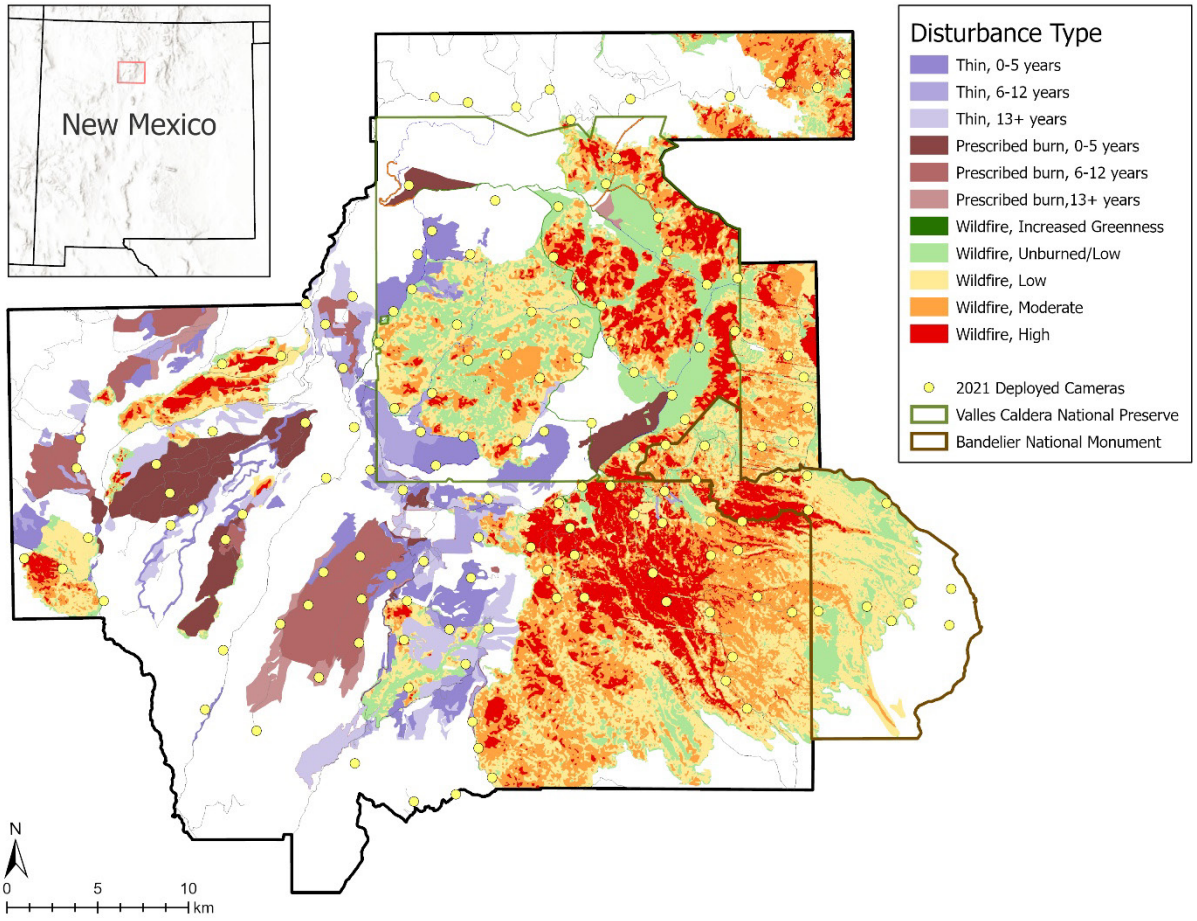



Figure 8.1.18. Map of wildlife camera deployments across the Santa Fe National Forest, Valles Caldera National Preserve and Bandelier National Monument.

In 2021, we collected images of ≥27 mammal species over 19,029 trap-nights (see examples in Fig. 8.1.19). Elk was the most commonly detected species, with 50,945 independent events at 121 cameras; ringtail (*Bassariscus astutus*), porcupine (*Erethizon dorsatum*), and long-tailed weasel (*Mustela frenata*) were the least commonly detected, with one event each. There were 11,089 mule deer detections at 137 cameras, 3,858 coyote events at 111 cameras, 1,315 black bear events at 88 cameras, and 193 mountain lion events at 40 cameras.

Detection of all five species was positively associated with increased camera effort (i.e., number of days the camera was active per 7-day occasion) and negatively associated with off-trail camera placement. Relative habitat use of black bears was positively associated with wildfire heterogeneity within 1500 m, coverage of prescribed fire within 1500 m, and overstory density (Table 8.1.7). Coyote relative habitat use was positively associated with fire heterogeneity at 1500 m. Relative habitat use by coyotes declined with more high severity fire within 300 m, more thinned area within 1500 m, and more treated areas, whether thin or prescribed fire, within 900 m. Elk relative habitat use was positively associated with more high severity fire within 300 m. Relative habitat use by elk declined with more prescribed fire coverage within 1500 m and with more undisturbed area within 100 m (Table 8.1.7). Mule deer relative habitat use increased with more thinned area within 300 m. Mountain lion relative habitat use did not appear to be associated with any of the examined disturbance variables.

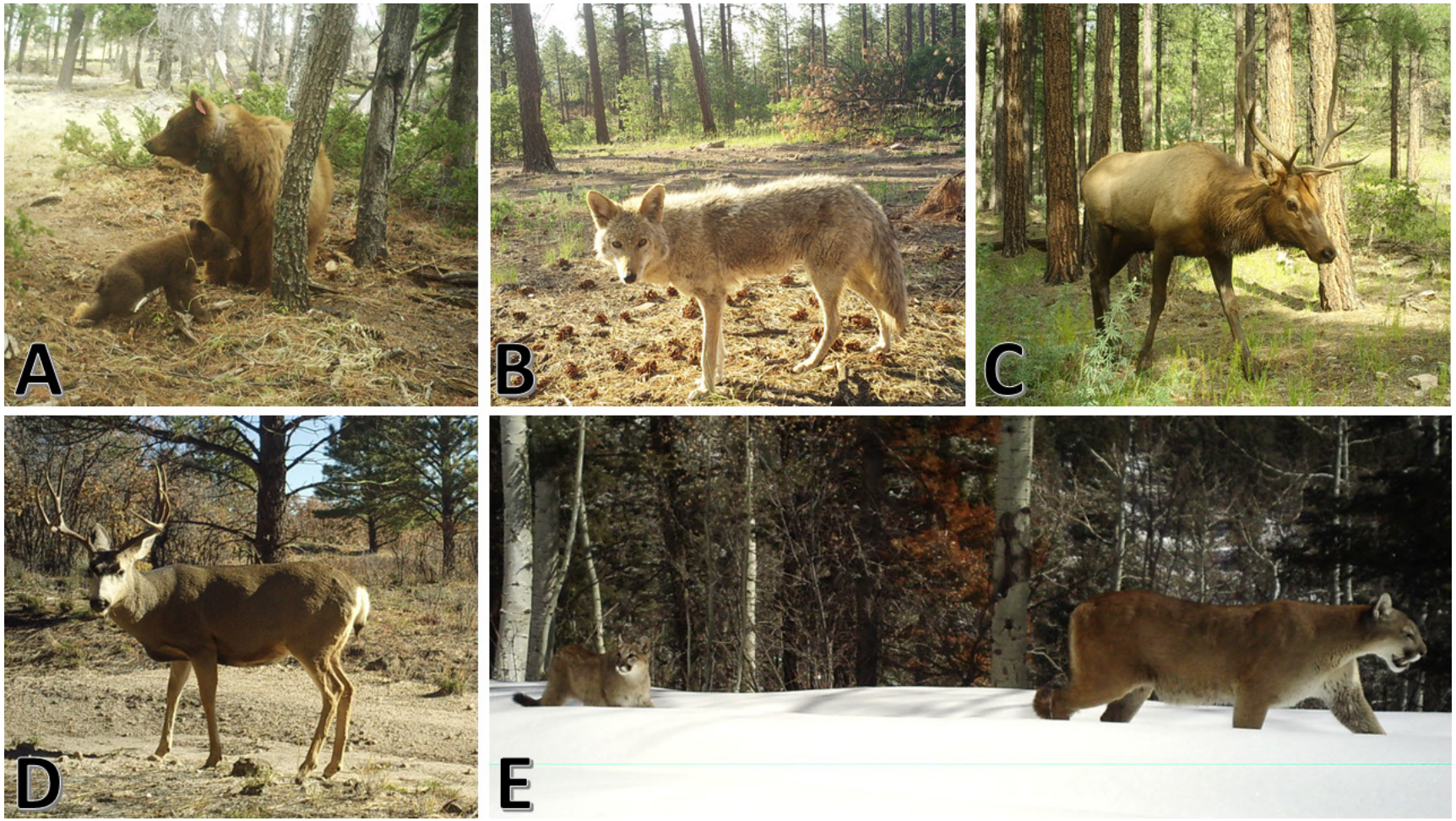



Figure 8.1.19. Representative images of five study species; (A) black bear (*Ursus americanus*), (B) coyote (*Canis latrans*), (C) elk (*Cervus canadensis*), (D) mule deer (*Odocoileus hemionus*), and (E) mountain lion (*Puma concolor*) in the southwest Jemez Mountains, New Mexico, US, May – October 2021.

Table 8.1.7. Model-averaged beta coefficients and 95% confidence intervals (95% CI) from top-ranked models of Royle-Nichols model of relative habitat use in relation to vegetation characteristics and disturbance history for large mammals in the Southwest Jemez Mountains, 2021.

| | **Species** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Black bear | | Coyote | | Elk | | Mule deer | | Mountain lion | |
| | Estimate | 95% CI | Estimate | 95% CI | Estimate | 95% CI | Estimate | 95% CI | Estimate | 95% CI |
| Intercept | –0.33 | −0.01–0.67 | 0.77* | 0.52–1.04 | 1.06* | 0.82–1.3 | 1.15 | 0.93–1.37 | −0.02 | −0.6–0.56 |
| Severity SD | | | | | | | | | | |
| 300 m | | | | | | | | | −0.25 | −0.6–0.09 |
| 1500 m | 0.37* | 0.14–0.6 | 0.17* | 0.01–0.34 | | | | | | |
| % Low | | | | | | | | | | |
| 100 m | | | | | | | 0.09 | −0.04–0.21 | | |
| 300 m | −0.19 | −0.42−0.04 | | | | | | | | |
| % Moderate | | | | | | | | | | |
| 1500 m | | | | | | | | | −0.54 | −1.64–0.56 |
| % High | | | | | | | | | | |
| 300 m | | | −0.24 | −0.6–0.13 | 0.14* | 0.09–0.19 | | | | |
| 900 m | | | | | | | | | 0.2 | −0.07–0.48 |
| % Prescribed fire | | | | | | | | | | |
| 100 m | | | | | | | −0.1 | −0.23–0.03 | | |
| 300 m | | | | | | | | | −0.33 | −0.76–0.11 |
| 1500 m | 0.28* | 0.08–0.47 | | | −0.21* | −0.38–−0.04 | | | | |
| % Thin | | | | | | | | | | |
| 300 m | | | | | | | 0.09* | 0.02–0.16 | | |
| 1500 m | | | −0.18* | −0.35−0.01 | | | | | | |
| % Thin or prescribed fire | | | | | | | | | | |
| 100 m | 0.15 | −0.04--0.33 | | | | | | | | |
| 300 m | | | | | | | | | −0.34 | −0.77–0.1 |
| 900 m | | | −0.19* | −0.36−0.01 | | | | | | |
| % No disturbance | | | | | | | | | | |
| 100 m | | | | | −0.19* | −0.36–−0.01 | | | | |
| 300 m | | | | | | | −0.12 | −0.25–0.02 | | |
| 1500 m | −0.24 | −0.51–0.03 | | | | | | | | |
| Overstory density | 0.33* | 0.13–0.54 | 0.01 | −0.14–0.16 | 0 | −0.15–0.15 | 0.07 | −0.05–0.19 | | |

| | | | | |
|---|---|---|---|---|
| Overstory density SD | | | −0.2 | −0.5–0.11 |
| Distance to no disturbance | −0.15 | −0.32–0.03 | | |

*Denotes variable significance at α = 0.05.

**PRODUCTS**

*Theses and Dissertations from former graduate students involved with the project include:*

Kindschuh, S. R. 2015. Efficacy of GPS cluster analysis in predicting black bear carnivory sites. MS Thesis. M.S., New Mexico State University.

Roberts, C. 2015. Seasonal and diel elk habitat selection in the Jemez Mountains of New Mexico. MS Thesis. M.S., Texas Tech University.

Humagain, K. 2016. Ecological variation among vegetation cover types and forest stand dynamics in the South-West Jemez Mountains, New Mexico. PhD. Dissertation. Doctor of Philosophy, Texas Tech University.

Roerick, T. M. 2017. Habitat restoration effects and habitat selection of female mule deer in the Jemez Mountains, New Mexico. MS Thesis. M.S., New Mexico State University.

Bard. S. M. 2018. Microhabitat Selection and Pathogen Prevalence of American Black Bears (*Ursus americanus*) in the Jemez Mountains, New Mexico, USA. MS Thesis. M.S., New Mexico State University.

Anaya, A. J. 2019. Mountain Lion (*Puma concolor*) Kill rates and prey composition in north-central New Mexico. MS Thesis. M.S., New Mexico Highlands University.

*Theses and Dissertations in Progress:*

Matthew Keeling, M.S. student. Thesis: Habitat selection of American black bears in north-central New Mexico. Department of Fish, Wildlife, and Conservation Ecology, New Mexico State University, Las Cruces.

Mark Peyton, Ph.D. student. Dissertation. Influence of landscape-scale habitat disturbances on mountain lion habitat selection, foraging ecology, and intra-specific competition. Department of Biology, New Mexico State University, Las Cruces.

Leah White, M.S. student. Thesis: Investigation of interspecific interactions of mule deer, elk, mountain lions, black bears and coyotes using multi-species occupancy modeling. Department of Fish, Wildlife, and Conservation Ecology, New Mexico State University, Las Cruces

*Publications from former graduate students involved with the project include:*

Roberts, C.P., J.W. Cain III, and R.D. Cox. 2016. Application of activity sensors for estimating behavioral patterns. Wildlife Society Bulletin 40:764–771.

Kindschuh, S.R., J.W. Cain III, D. Daniel and M.A. Peyton. 2016. Efficacy of GPS cluster analysis for predicting carnivory sites of a wide-ranging omnivore: the American black bear. Ecosphere 7: art e01513.

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## Section 8.2. Small mammals – rodents, rabbits and shrews.

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The objective of our small mammal monitoring was to characterize the small mammal community responses to high-severity, stand-replacement fire in ponderosa pine and mixed conifer forests of the SWJM CFLRP project area during the first three years of post-fire succession following the 2011 Las Conchas Fire. Specifically, we examined differences between burned and unburned (control) stands in each forest type for species richness and abundances of individual species, along with population demographic variables (sex ratios, age structure, reproductive activities); in addition, we determined the timeline in which species preferring open meadow habitat successfully colonized post-burn stands, and what vegetation changes were associated with small mammal succession. We then related the observed small mammal responses to known species-specific natural history traits to predict future successional changes in the post-fire environments.

We selected a total of 16 sites using the US Forest Service fire severity map (Figure 8.2.1) and purposefully targeted only areas of high-severity fire for burned study sites. We then used GIS data and site reconnaissance to match forest stand characteristics (tree density, species composition, basal area) to select the burned vs. unburned (control) sites (Figure 8.2.2). We established four burned and four control ponderosa pine forest sites and four burned and four control mixed conifer sites for small mammal and vegetation sampling. Permanent vegetation monitoring transects were installed immediately after the fire in summer 2011. Small mammal sampling plots were set up on these sites in spring 2012. See Peyton et al. (2025) for site details and methods. Sites were sampled every spring and autumn during 2012 and 2014.

During the entire study, we trapped for a total of 25,080 trap-nights. We captured or observed 15 small mammal species, comprising a total of 687 small mammals representing 11 species of rodents, 1 rabbit species and 3 species of shrews (Table 8.2.1). A total of 10 species were collected or observed in ponderosa pine control sites, with eight species recorded in burned sites. Mixed conifer control sites yielded 12 species, whereas burned mixed conifer sites supported only six species. Analysis of Variance (ANOVA) indicated strong differences in mean species richness per site per sample period between burned and control mixed conifer forest stands, with consistently lower species richness in burned sites ($F = 8.78$; $df = 1,47$; $P = 0.006$), but weaker differences between burned and control sites for ponderosa pine stands ($F = 2.98$; $df = 1,47$; $P = 0.094$; Figure 8.2.3).

Deer mice (*Peromyscus maniculatus*) were the most abundant small mammal captured in both ponderosa pine and mixed conifer forest types but showed no substantial differences among population abundances between burned and control sites in either forest type (Figure 8.2.4, Table 8.2.1). The only other *Peromyscus* species observed during the study was the brush mouse (*P. boylii*), and only two individuals were trapped in a ponderosa pine control site (Table 8.2.1).

Least chipmunks (*Neotamias minimus*) and Colorado chipmunks (*N. quadrivitattus*) both showed lower abundances in burned ponderosa pine sites, and least chipmunks exhibited lower abundance in burned mixed conifer sites (Figure 8.2.4, Table 8.2.1). Golden-mantled ground squirrel (*Callospermophilus lateralis*) was the only species to immediately show higher population abundance in burned mixed conifer sites than control sites, but showed no substantial difference in burned vs control ponderosa pine sites where it was considerably less common (Figure 8.2.4).

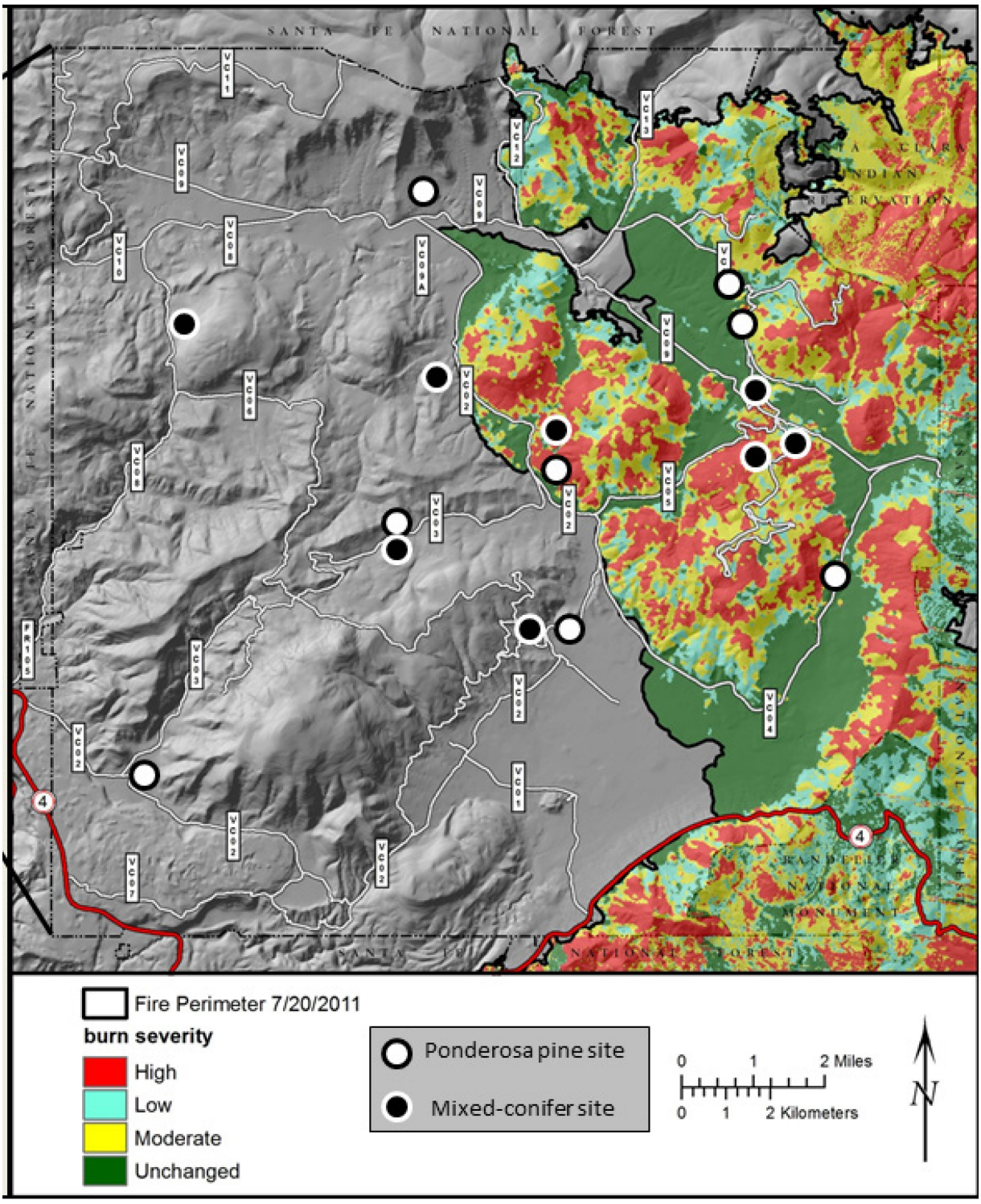



Figure 8.2.1 Map of Valles Caldera National Preserve showing locations of small mammal monitoring sites and burn severity for the 2011 Las Conchas Fire.

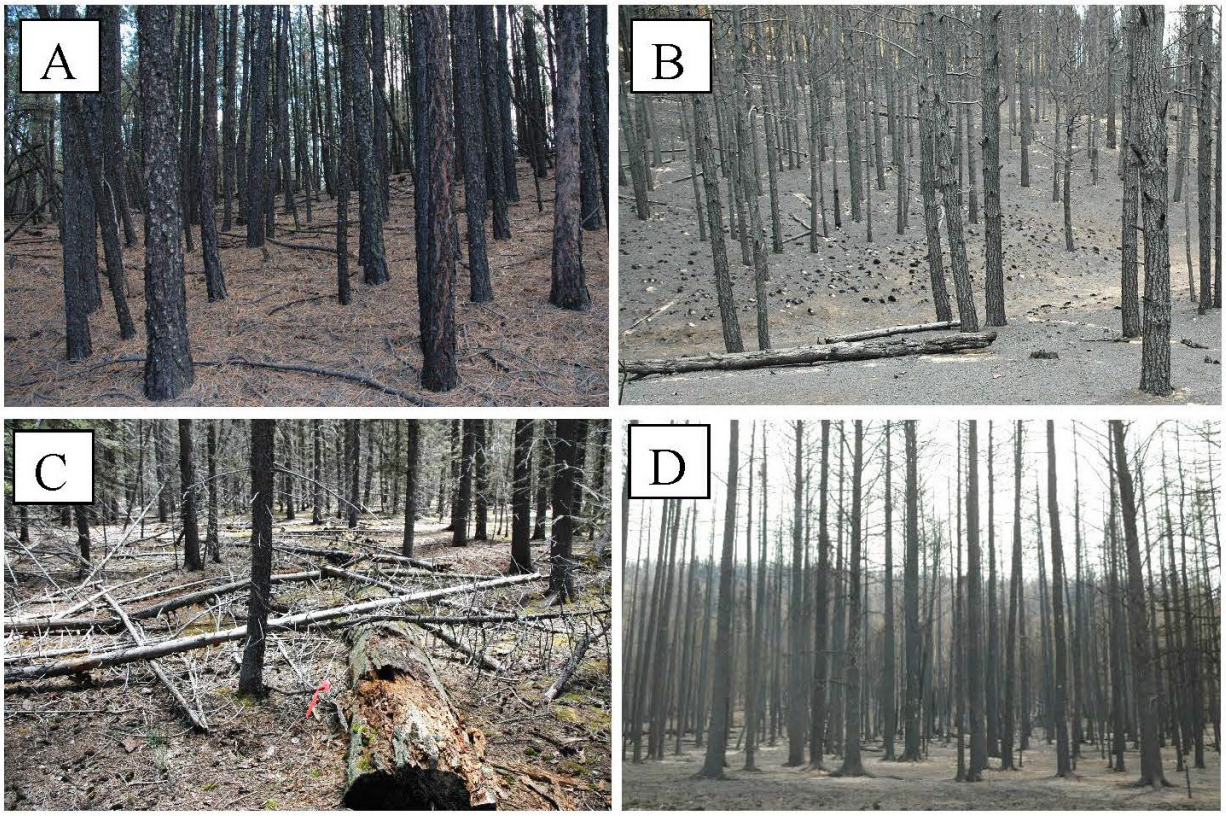



Figure 8.2.2. Small mammal study sites in Valles Caldera National Preserve: (A) unburned (control) ponderosa pine forest; (B) burned ponderosa pine forest; (C) unburned (control) mixed conifer forest; (D) burned mixed conifer forest.

Though the arboreal red squirrel (*Tamiasciurus hudsonicus*) was a common resident in mixed conifer control sites, it was not detected in any burned sites of either forest type (Figure 8.2.5, Table 8.2.1). Abert's squirrel (*Sciurus aberti*), a ponderosa pine specialist, was only observed on one occasion each in a control and a burned ponderosa pine site. Both the Mexican wood rat (*Neotoma mexicana*) and the bushy-tailed wood rat (*N. cinerea*), while captured in low numbers, were nearly always found in control sites (as were their nest structures); only a single Mexican wood rat was trapped in a burned ponderosa pine site in fall 2014, and no nearby nest structure was found. The mountain cottontail rabbit (*Sylvilagus nuttallii*) was occasionally observed in control sites and appeared in burned mixed conifer sites in spring 2012 but was not detected in any burned sites in later years (Figure 8.2.5). The three species of shrews (*Sorex monticolus*, *S. nanus*, and *S. preblei*) were collected mostly in mixed conifer sites, and collectively were eight times more numerous in samples from control sites ($n$ = 16) than in burned sites ($n$ = 2; Table 8.2.1, Figure 8.2.5).

Montane voles (*Microtus montanus*) appeared in burned ponderosa pine sites in fall 2013, and in greater numbers in fall 2014; they also were trapped in mixed conifer sites during those same time periods in both burned and control sites with greater numbers in burned sites (Figure 8.2.5). Long-tailed voles (*M. longicaudus*) finally appeared in burned ponderosa pine sites in fall 2014, four growing seasons following the fire (Figure 8.2.5, Table 8.2.1).

Table 8.2.1. Number of individual small mammals and ANOVA results* comparing abundance in burned vs. control sites in mixed conifer and ponderosa pine forests in northern New Mexico, 2012-2014.

| | Mixed conifer | | | Ponderosa pine | | |
|---|---|---|---|---|---|---|
| Species | Burn | Control | *P* | Burn | Control | *P* |
| Deer mouse (*Peromyscus maniculatus*) | 114 | 117 | 0.896 | 73 | 56 | 0.310 |
| Brush mouse (*Peromyscus boylii*) | 0 | 0 | - | 0 | 2 | - |
| Least chipmunk (*Neotamias minimus*) | 4 | 42 | **0.000** | 0 | 18 | **0.012** |
| Colorado chipmunk (*Neotamias quadrivitattus*) | 0 | 6 | - | 0 | 12 | **0.025** |
| Golden-mantled ground squirrel (*Callospermophilus lateralis*) | 92 | 27 | **0.016** | 5 | 8 | 0.490 |
| Mexican woodrat (*Neotoma mexicana*) | 0 | 6 | - | 1 | 5 | - |
| Bushy-tailed woodrat (*Neotoma cinerea*) | 0 | 3 | - | 0 | 0 | - |
| Red Squirrel (*Tamiasciurus hudsonicus*) | 0 | 28 | **0.000** | 0 | 3 | - |
| Abert's squirrel (*Sciurus aberti*) | 0 | 0 | - | 1 | 1 | - |
| Montane Vole (*Microtus montanus*) | 9 | 2 | 0.104 | 9 | 0 | - |
| Long-tailed Vole (*Microtus longicaudus*) | 0 | 0 | - | 10 | 0 | **0.007** |
| Mountain cottontail (*Sylvilagus nuttallii*) | 3 | 7 | 0.296 | 0 | 2 | - |
| Montane shrew (*Sorex monticolus*) | 0 | 6 | - | 0 | 0 | - |
| Dwarf shrew (*Sorex nanus*) | 1 | 4 | - | 1 | 2 | - |
| Preble's shrew (*Sorex preblei*) | 0 | 6 | - | 1 | 0 | - |

*ANOVA *P*-values for statistical significance of number of individuals under different fire treatments; analyses limited to species with ≥ 10 individuals in samples within a forest type. Values in boldface: $P \leq 0.05$.

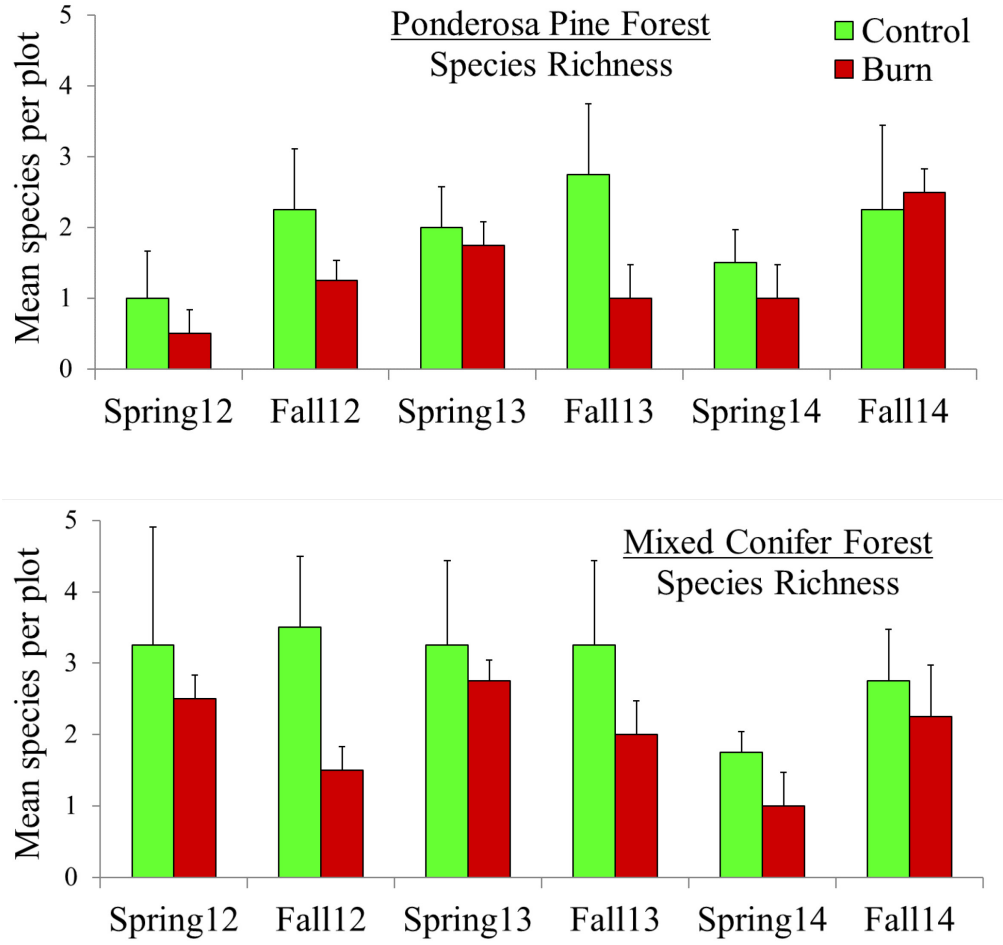



Figure 8.2.3. Mean species richness of small mammals per site in control and burned ponderosa pine forests (top) and mixed conifer forests (bottom), 2012-2014, on Valles Caldera National Preserve following the 2011 Las Conchas Fire. Error bars indicate standard errors.

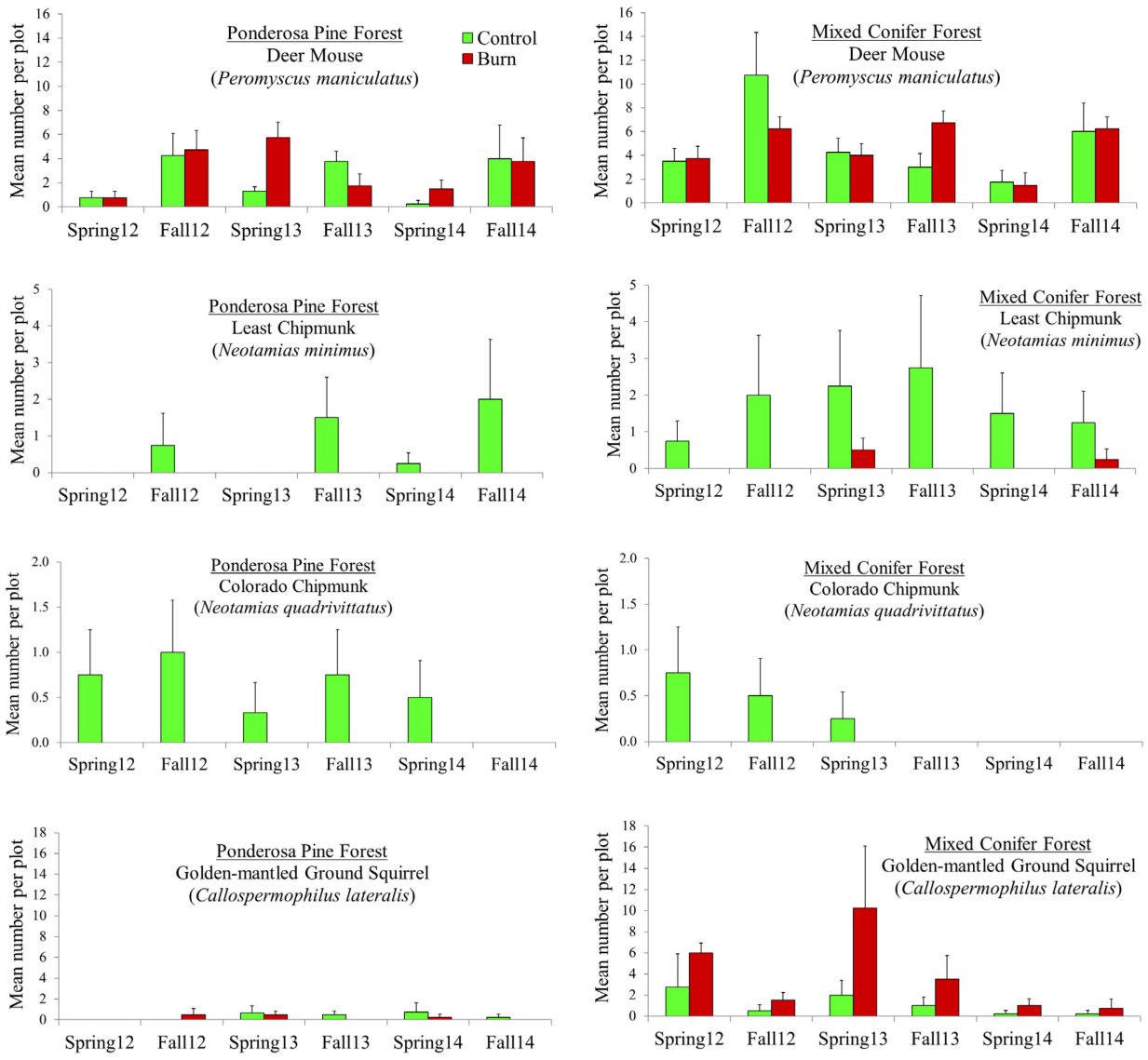



Figure 8.2.4. Mean numbers of deer mice, chipmunks and ground squirrels per plot in control and burned ponderosa pine sites (left) and mixed conifer sites (right), 2012-2014, on Valles Caldera National Preserve following the 2011 Las Conchas Fire. Error bars indicate standard errors.

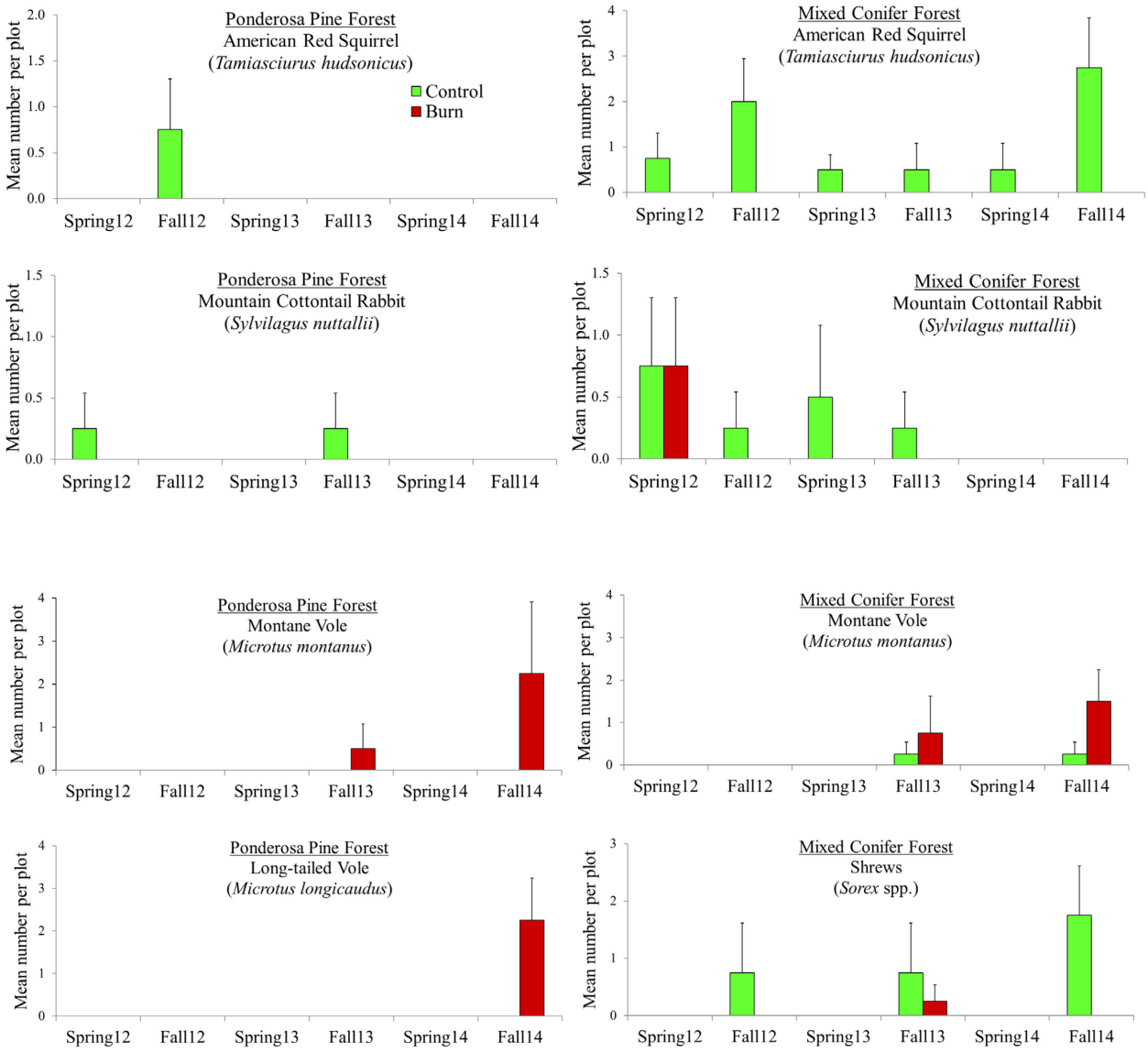



Figure 8.2.5. Mean numbers of tree squirrels, cottontail rabbits, voles and shrews per plot in control and burned ponderosa pine sites (left) and mixed conifer sites (right), 2012-2014, on Valles Caldera National Preserve following the 2011 Las Conchas Fire. Error bars indicate standard errors. Not shown: Long-tailed voles were not found in mixed conifer sites, only two Abert's squirrels were observed, and only four shrews were collected in ponderosa pine sites.

Examination of age, sex, and reproduction attributes of live-trapped rodents showed that populations in burned sites comprised primarily adults from both sexes (Table 8.2.2). The deer mouse population in control ponderosa pine sites exhibited a male:female (M:F) sex ratio of 49:51, with a third of the population represented by subadults (≤15 g body mass). In burned ponderosa pine sites, deer mouse populations had a male prevalence, with a M:F sex ratio of 65:35, while also consisting of one-third subadults. Both unburned and burned ponderosa pine populations included reproductively active females, with 14% and 42% of adult female deer mice being pregnant or lactating in control and burned sites, respectively (Table 8.2.2). In mixed conifer forest, deer mouse M:F sex ratios were 47:53 in control sites and 55:45 in burned sites; subadults made up 37% and 18% in control and burned sites, respectively (Table 8.2.2). Reproductively active females constituted 27% and 23% of the adult female population in control and burned mixed conifer sites, respectively (Table 8.2.2).

Golden-mantled ground squirrels occurred most commonly in mixed conifer forest, and exhibited M:F sex ratios of 59:41 in control sites and 44:56 in burned sites; subadults (≤120 g body mass) made up 33% of the population in control sites, and 52% of the population in burned sites (Table 8.2.2). None of the adult females was observed to be pregnant or lactating during the trapping periods.

Most of the other rodent species in burned forests were recorded in relatively small numbers, but these species usually exhibited a mix of adult males and females, with few instances of subadults and juveniles and little evidence of reproductive activity (Table 8.2.2). Voles proved to be exceptions to this pattern. In burned ponderosa pine sites, long-tailed voles were all male ($n$ = 10) and 70% subadults (≤42 g body mass). Montane voles inhabiting burned sites in both forest types exhibited female-biased M:F sex ratios of 44:56 and 33:67 (ponderosa pine and mixed conifer, respectively), with reproductively-active adult females at 20% and 33%, respectively (Table 8.2.2).

Some small mammal presence/absence and abundances were associated with vegetation characteristics. Burned sites, which completely lost all live tree canopy cover, exhibited lower abundances (or zero) of least chipmunks, Colorado chipmunks, woodrats, red squirrels and shrews compared to control sites (Table 8.2.1). As herbaceous plant cover on burned sites recovered and eventually exceeded that of control sites, montane voles and long-tailed voles colonized the sites. Long-tailed voles were found in burned ponderosa pine sites when herbaceous cover achieved 93 ± 3% (SE), and montane voles arrived when cover was 91 ± 6%. In burned mixed conifer sites, montane voles colonized when herbaceous cover attained 82 ± 11%. Regression analyses yielded positive relationships with herbaceous cover ($HC$) for montane voles in ponderosa pine sites ($Abundance = 0.02HC - 0.78$; $n = 24$; $r^2 = 0.27$; $P = 0.009$) and in mixed conifer sites ($Abundance = 0.02HC - 0.29$; $n = 24$; $r^2 = 0.35$; $P = 0.002$). Similarly, long-tailed voles showed a positive association with herbaceous cover in ponderosa pine sites ($Abundance = 0.03HC - 0.85$; $n = 24$; $r^2 = 0.41$; $P = 0.032$). Regression analyses for other species (with sufficient sample sizes) produced no substantial relationships with herbaceous plant cover in either forest type (deer mice, $r^2 = 0.04 - 0.10$, $P = 0.14 - 0.33$; golden mantled ground squirrels, $r^2 = 0.00$, $P = 0.86 - 0.91$).

Table 8.2.2. Demographic results for rodent species live-trapped in burned and control sites on Valles Caldera National Preserve. Values are numbers of rodents recorded over 2012-2014.

| Forest type, species | Fire treatment | Adult males | Sub-adult males | Adult females | Sub-adult females | Pregnant or lactating |
|---|---|---|---|---|---|---|
| Ponderosa pine forest | | | | | | |
| Species | | | | | | |
| Deer mouse | Control | 17 | 11 | 21 | 8 | 3 |
| | Burn | 31 | 24 | 24 | 6 | 10 |
| Brush mouse | Control | 1 | - | 1 | - | - |
| | Burn | - | - | - | - | - |
| Golden-mantled ground squirrel | Control | 4 | - | 4 | - | - |
| | Burn | 4 | - | 1 | - | - |
| Least chipmunk | Control | 11 | - | 7 | - | - |
| | Burn | - | - | - | - | - |
| Colorado chipmunk | Control | 8 | 1 | 3 | - | - |
| | Burn | - | - | - | - | - |
| Montane vole | Control | - | - | - | - | - |
| | Burn | 4 | - | 5 | - | 1 |
| Long-tailed vole | Control | - | - | - | - | - |
| | Burn | 3 | 7 | - | - | - |
| Mexican wood rat | Control | 3 | - | 2 | - | - |
| | Burn | 1 | - | - | - | - |
| Mixed conifer forest | | | | | | |
| Species | | | | | | |
| Deer mouse | Control | 29 | 25 | 44 | 18 | 12 |
| | Burn | 53 | 11 | 43 | 10 | 10 |
| Golden-mantled ground squirrel | Control | 9 | 7 | 9 | 2 | - |
| | Burn | 17 | 21 | 25 | 24 | - |
| Least chipmunk | Control | 23 | - | 19 | - | - |
| | Burn | 3 | - | 1 | - | - |
| Colorado chipmunk | Control | 1 | - | 5 | - | - |
| | Burn | - | - | - | - | - |
| Montane vole | Control | - | - | 2 | - | 2 |
| | Burn | 3 | - | 6 | - | 2 |
| Mexican wood rat | Control | 2 | - | 3 | - | - |
| | Burn | - | - | - | - | - |
| Bushy-tailed wood rat | Control | 2 | - | 1 | - | - |
| | Burn | - | - | - | - | - |

The monitoring project demonstrated that vegetation succession after the fire rapidly created a nearly complete coverage of herbaceous plants, converting the original forest stands to montane meadow habitats; no aspen or conifer seedling regeneration was observed on our study sites (although some other areas of the Las Conchas Fire did support aspen regeneration). The high-severity stand-replacement fires resulted in a significant decline of forest small mammal species in mixed conifer stands (from 12 to 6 species), and a smaller (non-significant) decline in ponderosa pine stands (from 10 to 8).

The most common rodent in both unburned forest types, the deer mouse, also dominated the burned sites. This generalist omnivore species occupied all of the study sites, with a mix of both sexes and adult/subadult age classes; adult female deer mice were observed both pregnant and lactating, supporting the conclusion that this species maintained permanent, self-supporting populations in both burned and unburned environments. Species that are considered forest specialists were absent in the burned forest stands. Red squirrels were clearly eliminated in burned forests, detected only in unburned stands. Bushy-tailed wood rats and Mexican wood rats also were undetected in the burned stands in mixed conifer forest, possibly because there were no small-diameter woody twigs remaining that woodrats typically require for nest construction. After the spring of 2012, no mountain cottontail rabbits were observed in burned areas, potentially due to removal of all woody shrub and tree cover. Both species of chipmunks, the least chipmunk and Colorado chipmunk, also were undetected in burned ponderosa pine forests, and were rarely observed in burned mixed conifer forests. In addition, the distribution of the three species of shrews, while present in small numbers in burned forests, appeared to favor unburned mixed conifer forest stands. The golden-mantled ground squirrel was the only squirrel species to maintain populations in the burned forest stands; while less common in the ponderosa pine forest, this species occurred in higher abundance in burned mixed conifer stands than unburned.

Finally, colonization of the burned forest sites by montane voles was accomplished in 2013, two years after the fire, and comprised adults of both sexes and reproductively active females. The long-tailed vole appeared on burned ponderosa pine sites in fall 2014, three years after the fire; however, of the 10 individuals recorded, all were males with both adults and subadults. These may have been dispersing individuals and would not represent an established breeding population; we speculate that such a permanent population would establish within the next year or two as grass and forb cover continues to develop.

Thus, at the landscape level, we expect meadow-dwelling small mammals to continue to increase or maintain high abundances for decades to come in these post-burn meadows, while forest specialists will be constrained to suitable unburned or low-severity burn habitat with existing live forest canopy.

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# Chapter 9. Wildlife Monitoring – Birds

**Collaborators:** Trevor Fetz, Gail Garber, Mike Hill, Mike Fugagli, Brian Dykstra, John Stanek and Amanda Schluter – *Hawks Aloft, Inc., Albuquerque, NM.*

**Summary**
As part of the Southwest Jemez Collaborative Forest Landscape Restoration Project, Hawks Aloft, Inc. was subcontracted to conduct avian point count surveys from 2012-2024 to evaluate avian use within eight habitat types located in the Valles Caldera National Preserve and the Santa Fe National Forest. Cumulatively, over 13 years of surveys, avian density was highest in mixed conifer forest (2.51 birds/ha) and burned mixed conifer forest (2.28 birds/ha). Avian richness was highest in riparian habitat (115 species) and ponderosa pine (*Pinus ponderosa*) forest (89 species). We documented a total of 141 bird species. In a comparison of unthinned and thinned ponderosa pine forest plots, cumulative (2012-2024) avian density was significantly higher at treated points (2.27 birds/ha) than untreated (1.68 birds/ha); however, the difference was largely driven by data collected after the thinned points were subjected to prescribed fire. Density was significantly higher at thinned and burned points (2.41 birds/ha) than points that were only thinned (1.97 birds/ha). Avian richness was higher at thinned points (59 species) than unthinned points (58 species). We documented 57 avian species of conservation concern, including two species listed as threatened by the State of New Mexico: Peregrine Falcon (*Falco peregrinus*) and Gray Vireo (*Vireo vicinior*). The response of relatively common species of concern was variable among unburned and burned habitats of like type and between untreated and treated ponderosa pine forest. Among 15 relatively common species of concern, nine were more abundant in unburned forest and six were more abundant in burned forest, while eight were more abundant in unthinned forest and seven were more abundant in thinned forest. Our results suggest that the use of a combination of treatments, including prescribed fire, thinning, and areas of no treatment, conducted in patchy arrangements over time and area could improve overall avian diversity and benefit the greatest number of species.

## Introduction

Beginning in 2012, Hawks Aloft Inc. was subcontracted to conduct avian point count surveys for the Southwest Jemez Collaborative Forest Landscape Restoration Project (CFLRP). This project is a long-term monitoring effort focused on forest and watershed restoration activities in the southwest Jemez Mountains of north-central New Mexico. The overall goal of this project is to improve the resilience of ecosystems to recover from wildfires and other natural disturbance events in order to sustain healthy forests and watersheds (Santa Fe National Forest and Valles Caldera National Preserve 2010). Specific objectives include 1) reducing the risk of wildfire and restoring natural fire regimes; 2) increasing forest diversity and old growth characteristics; 3) improving fish and wildlife habitat; 4) improving water quality and watershed function; 5) mitigating climate change impacts; 6) protecting cultural resources; and 7) utilizing woody by-products to create local economic development opportunities.

Although mitigating climate change is listed as one of the key objectives of this project, it also is inherent in the overall goal of the project and many of the other project objectives. Climate change models project that the southwestern U.S. will become steadily hotter and drier over the remainder of the 21st century (e.g. Karl et al. 2009, Garfin et al. 2013, Kunkel et al. 2013, Thorne et al. 2018, McKinnon et al. 2021), resulting in more frequent and severe drought, increases in insect outbreaks and associated tree mortality (Raffa et al. 2008, Bentz et al. 2010, McDowell et al. 2016, Overpeck and Udall 2020), shifts in vegetation distributions (Allen and Breshears 1998, Gonzales et al. 2010, Nolan et al. 2021) and increased risk of catastrophic fire (Westerling et al. 2006, Abatzoglou and Williams 2016, Harvey 2016, Harvey et al. 2016, Mueller et al. 2020). The reality of increased fire risk was illustrated by the recent occurrence of several major fires within the project area. In 2011, the Las Conchas fire burned over 156,000 acres on the Valles Caldera National Preserve and adjacent Santa Fe National Forest land, making it the largest fire in New Mexico history up to that point (National Park Service 2012). In 2013, the Thompson Ridge fire burned nearly 24,000 acres within the Valles Caldera National Preserve. The Pino fire burned approximately 4,300 acres of the Santa Fe National Forest land in 2014. Additionally, several smaller fires have occurred in or near the study area since 2014.

The effects of climate change, such as those listed above, will have a substantial impact on birds (van Riper et al. 2014, Pearce-Higgins and Green 2014, Stephens et al. 2016, Bateman et al. 2020). Thus, monitoring avian response to project activities designed to mitigate the negative impacts of climate change at both the community and individual species levels will be important factors in assessing project success. In addition, birds comprise the most diverse vertebrate taxa within the study area, and can serve as ecological indicators of overall environmental health (Temple and Wiens 1989, Canterbury et al. 2000, Gregory and van Strien 2010). In this report, we summarize avian data collected from 2012-2024, including an inventory of avian species observed, identification of species of conservation concern, and analyze avian use by habitat type, between burned and unburned points and between treated and untreated points.

**Study Area**

The study area encompasses approximately 210,000 acres in the Southwest Jemez Mountains of north-central New Mexico, comprising the entire upper Jemez River watershed. The area includes 110,000 acres of the Santa Fe National Forest (USFS) and nearly 89,000 acres of the Valles Caldera National Preserve (VCNP), managed by the National Park Service. The Valles Caldera is a 13-mile circular depression that was formed about 1.23 million years ago after a supervolcano erupted in the Jemez Mountains (Goff et al. 2011). This eruption was preceded by an almost identical eruption some 400,000 years earlier that formed the equally large Toledo Caldera. The Toledo Caldera was mostly obliterated when the Valles Caldera formed in the same place. Resurgent doming of the central caldera floor occurred between 1.25 and 1.22 million years ago, and was followed by additional dome eruptions around the caldera margins up until about 40,000 years ago. These eruptions are interlayered with lake deposits, indicating the caldera has been at least partially filled by lakes since its formation. The depressions in Valles Caldera currently contain several grassland valleys that sustain a variety of wildlife unique to the area.

Avian point counts were established in five of the habitat types found within the overall study area: mixed conifer forest, ponderosa pine (*Pinus ponderosa*) forest, mountain valley grassland, riparian, and mountain meadow. Three additional habitat types were created at established points by the Las Conchas, Thompson Ridge, Pino, Hidden Valley, and Cerro Pelado fires: mixed conifer burn, ponderosa pine burn, and mountain valley grassland burn. Mixed conifer forest plots were dominated by various mixtures of conifer, including Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), blue spruce (*Picea pungens*), southwestern white pine (*P. strobiformis*), limber pine (*Pinus flexilis*), and ponderosa pine. Mountain valley grassland plots were primarily devoid of woody vegetation and dominated by different mixtures of grasses, including Arizona fescue (*Festuca arizonica*), pine dropseed (*Blepharoneuron tricholepis*), Thurber fescue (*F. thurberi*), and Parry's oatgrass (*Danthonia parryi*). Vegetation in riparian habitat was highly variable among survey routes and elevation. Dominant woody species at lower elevations included cottonwood (*Populus fremontii*), coyote willow (*Salix exigua*) and juniper (*Juniperus* spp.). Higher elevation riparian sites incorporated components of thinleaf alder (*Alnus incana*), willow, Douglas-fir and ponderosa pine. The mountain meadow plot was devoid of trees, with tufted hairgrass (*Deschampsia cespitosa*) and sedges (*Carex* spp.) comprising the primary vegetation.

Between 2012 and 2021, we initiated surveys at 17 different avian point count routes, including 10 on USFS land, six in VCNP, and one with points in both management areas (Fig. 9.1). Among the 11 routes with point counts on USFS land (N = 112 survey points), five were classified as ponderosa pine forest, four were classified as riparian, and two were classified as mixed conifer forest. Among the seven routes with points located in VCNP (N = 91 survey points), one route was classified as mixed conifer forest, one route was classified as ponderosa pine forest, one route was classified as riparian, and four were comprised of a mixture of habitats, including mixed conifer, mountain valley grassland, ponderosa pine, and mountain meadow. Many of the points in mixed conifer, ponderosa pine, and mountain valley grassland habitats burned during the Las Conchas and/or Thompson Ridge fires. Details of each route are provided below.

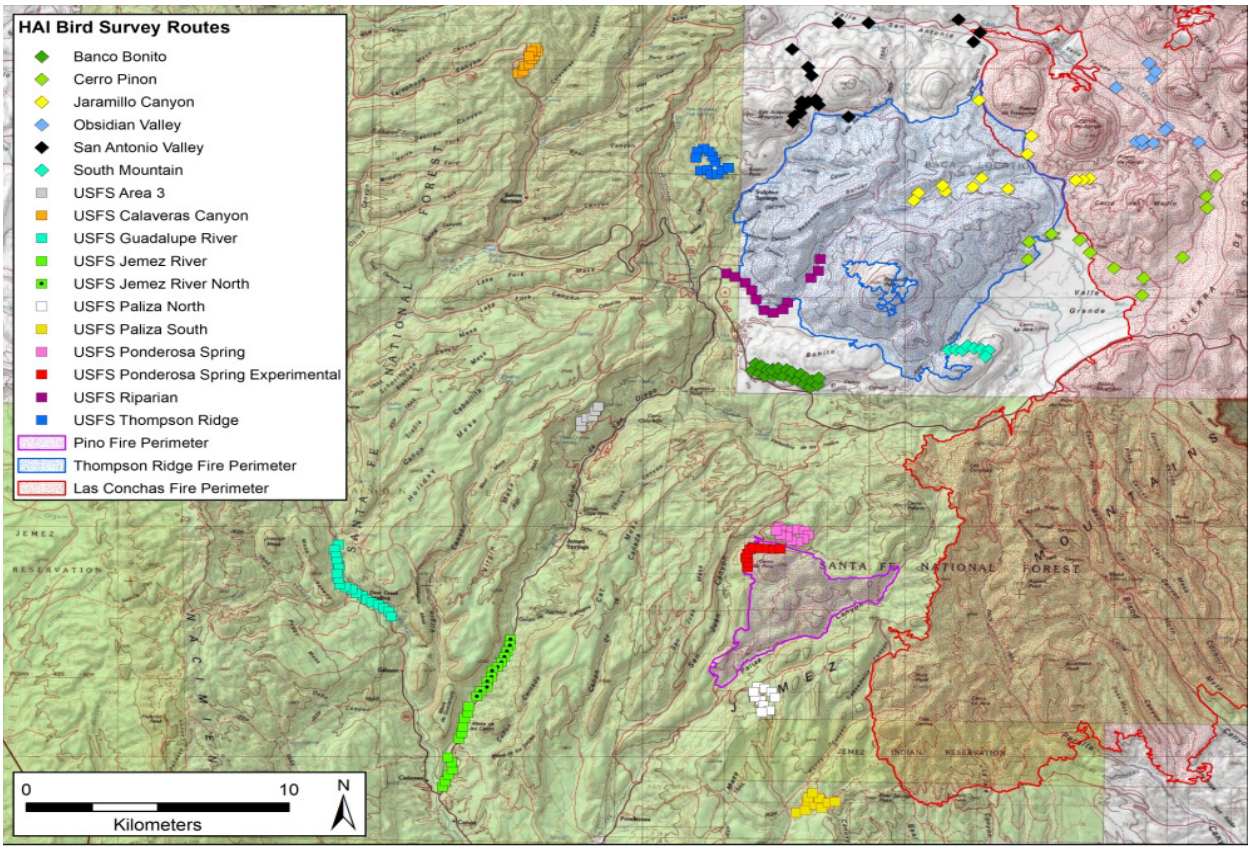


Fig. 9.1. Map of point count routes in the Valles Caldera National Preserve and Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico. Colored squares represent specific point count locations.

**Banco Bonito** - Located at the southern end of Banco Bonito in the southwestern portion of VCNP (Figs. 9.1 and 9.2). All 22 points on this route (BB01-BB22) were classified as ponderosa pine forest. Points BB02, BB04, and BB06 were thinned prior to the 2012 field season, BB08, BB10, and BB12 were thinned during 2014, and BB14 and BB16 were thinned prior to the 2015 field season. Points BB17-BB22 were established in 2017, with BB18, BB20, and BB22 having been treated prior to the initiation of point count surveys. All 11 points subjected to thinning also were subjected to prescribed fire prior to the 2018 field season. With the addition of points BB17-BB22, it required two survey days (or two simultaneous observers) to complete all the points on this route. Banco Bonito was surveyed during all years from 2012-2024.

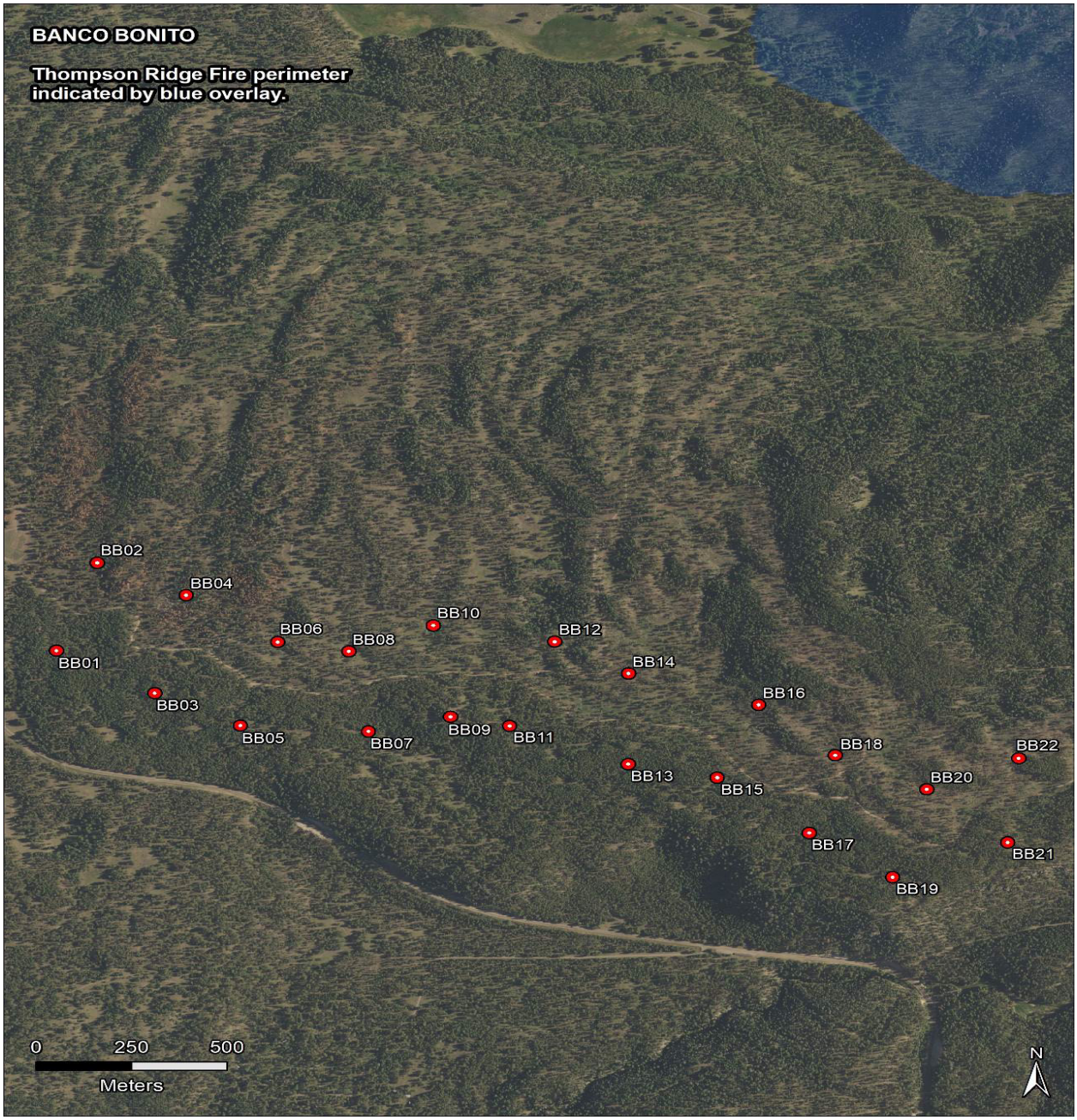



Fig. 9.2. Point locations for the Banco Bonito survey route, Valles Caldera National Preserve, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**Cerro Pinon**- Located in the southeastern portion of VCNP, the 12 points on this route follow Jaramillo Creek and the east fork of the Jemez River (Figs. 9.1 and 9.3). One point on this route (MC10) was classified as mixed conifer, but burned during the 2013 Thompson Ridge fire. Seven points (MV01a, MV01b, MV02a, MV02b, MV03, MV04, MV05) were classified as mountain valley grassland. All the mountain valley grassland points except for MV01a burned during the 2011 Las Conchas fire. Four points (PP04, PP05, PP09, PP10) were classified as ponderosa pine forest. PP04 and PP05 burned during the Las Conchas fire. PP09 burned during the Thompson Ridge fire. Portions of this route were surveyed during all years from 2012-2024.

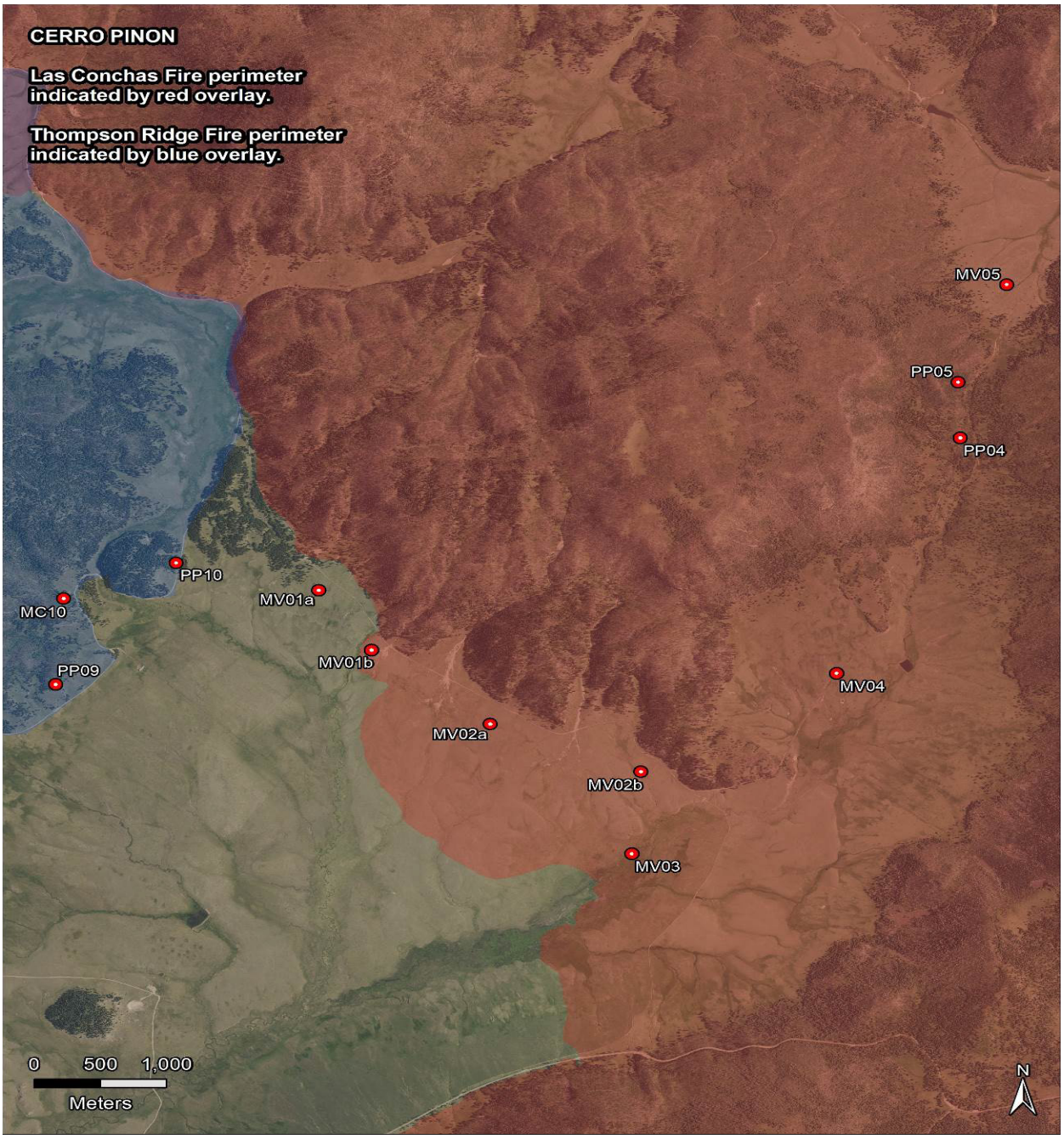



Fig. 9.3. Point locations for the Cerro Pinon survey route, Valles Caldera National Preserve, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**Jaramillo Canyon**- Located in the center of VCNP, the 11 points on this route are situated near Jaramillo Creek and bordered by the southeastern flanks of Cerros del Abrigo, the southern flanks of Puerta de Trasquilar and the northern flanks of Redondo Peak (Figs. 9.1 and 9.4). Eight points (MC03, MC03b, MC03c, MC05, MC07, MC08, MC09, MC11) were classified as mixed conifer forest, one point (MV14) was classified as mountain valley grassland, and four points (PP03, PP07, PP08, PP11a) were classified as ponderosa pine forest. MC03, MC03b, MC03c, MC05, and PP03 burned during the 2011 Las Conchas fire. MC07-MC09, MC11, MV14, PP07, PP08, and PP11a burned during the 2013 Thompson Ridge fire. Portions of this route were surveyed during all years from 2012-2024.

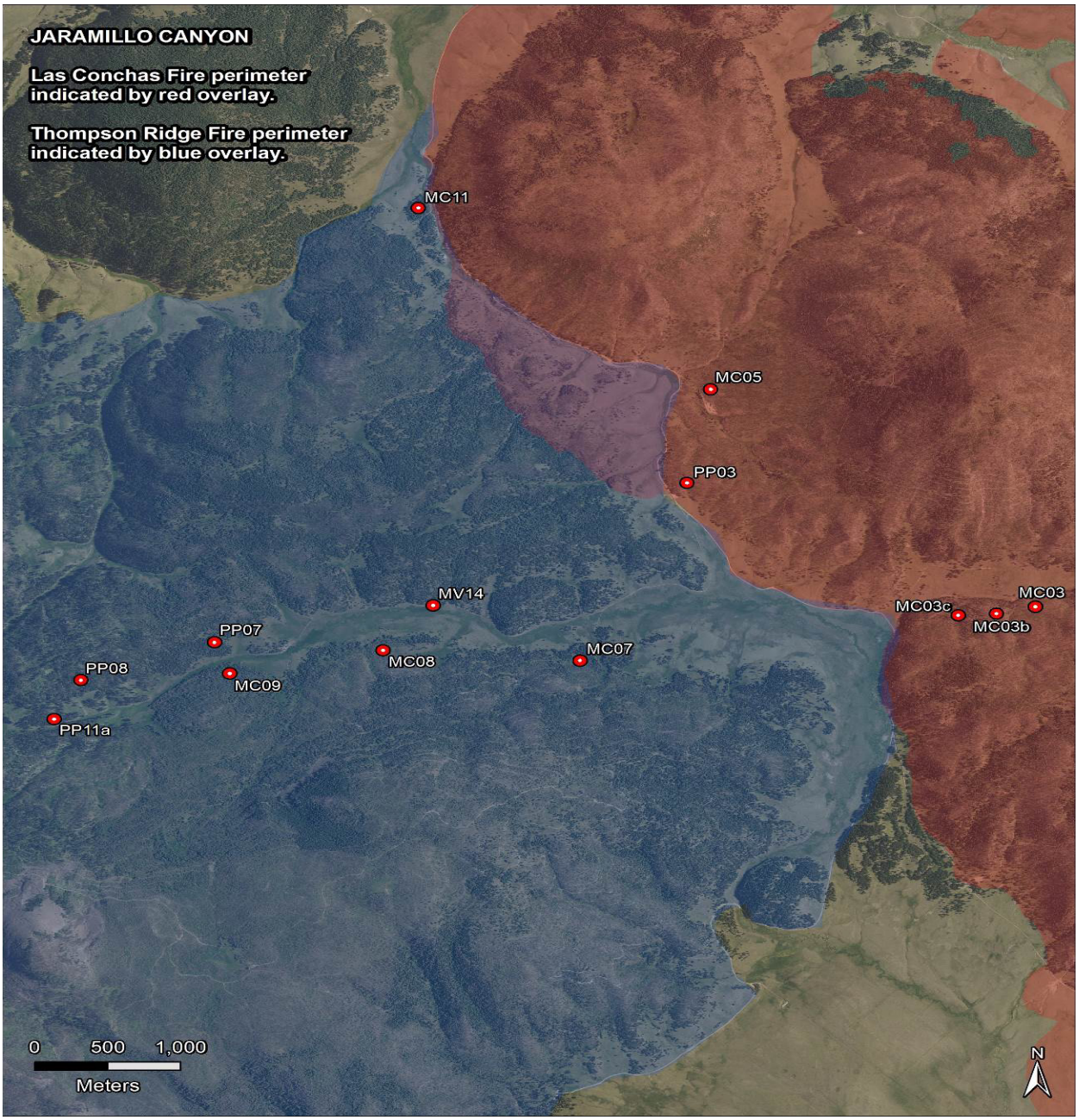


Fig. 9.4. Point locations for the Jaramillo Canyon survey route, Valles Caldera National Preserve, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**Obsidian Valley**- Located in the northeast portion of VCNP, this route in the Valle Toledo, was bounded by the southwestern flanks of the Cerros de los Posos (Figs. 9.1 and 9.5). All 11 points on this route burned during the 2011 Las Conchas fire. Five points (MC01, MC02, MC04, MCNew4, MC06) were classified as mixed conifer forest, three points (MV06, MV07, MVNew3) were classified as mountain valley grassland, and three points (PP01, PP02, PP06) as ponderosa pine forest. Portions of the Obsidian Valley route were surveyed in 2012-2015 and 2021-2024.

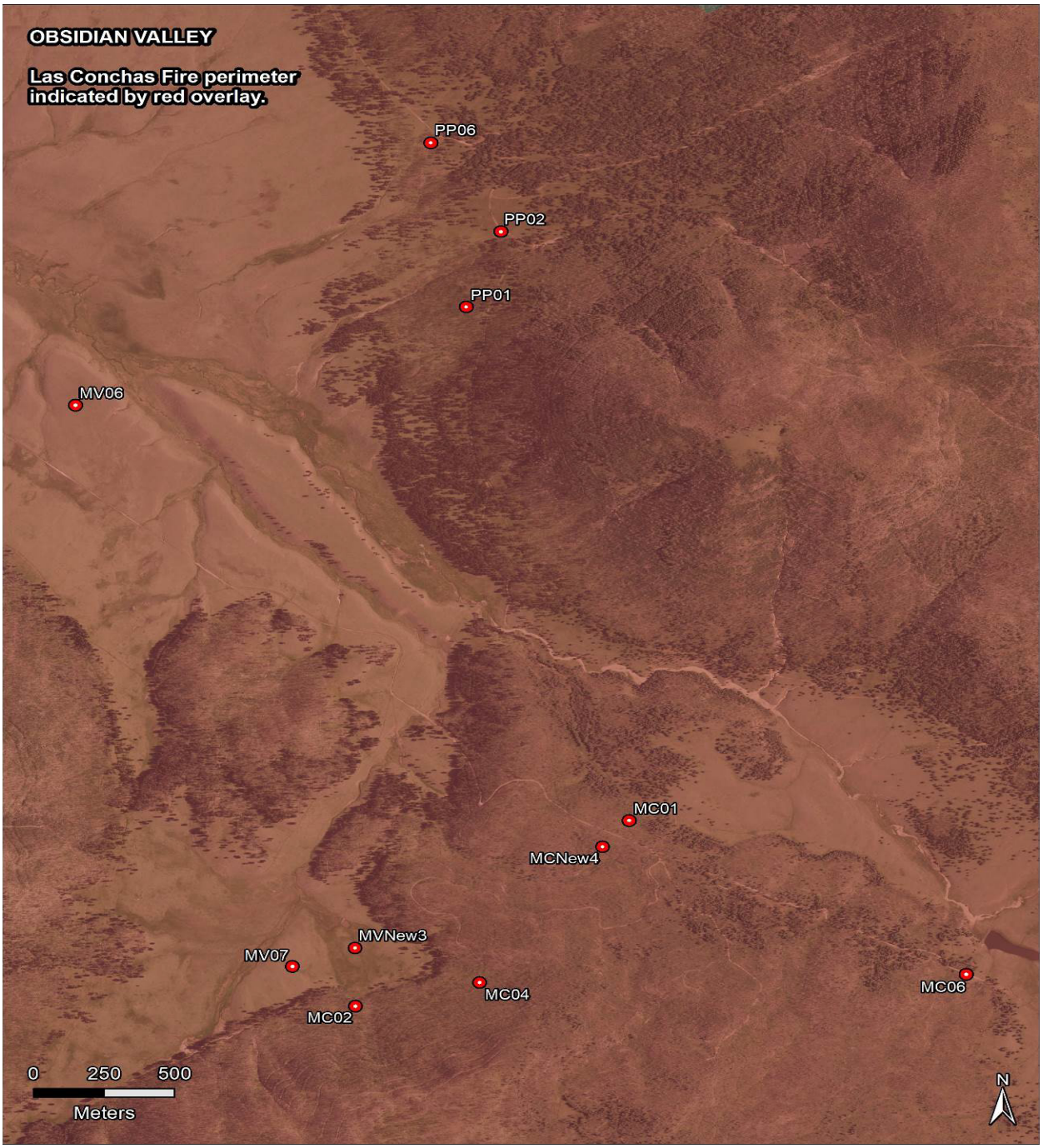



Fig. 9.5. Point locations for the Obsidian Valley survey route, Valles Caldera National Preserve, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**San Antonio Valley**- Located in the northwestern portion of VCNP, the 17 points on this route were situated within the Valle San Antonio and on San Antonio Mountain (Figs. 9.1 and 9.6). Ten points (MCNew1, MCNew2, MC12, SAV01-SAV07) were classified as mixed conifer forest, one point (MM07) was classified as mountain meadow grassland, five points (MV08a, MV09, MV10a, MV11, MV12) were classified as mountain valley grassland, and one point (PP12) was classified as ponderosa pine forest. MV08a burned during the 2011 Las Conchas fire and MV12 burned during the 2013 Thompson Ridge fire. Points MCNew1 and SAV01-SAV07 were thinned prior to the 2021 survey season. Portions of the San Antonio Valley route were

surveyed during all years from 2012-2024.

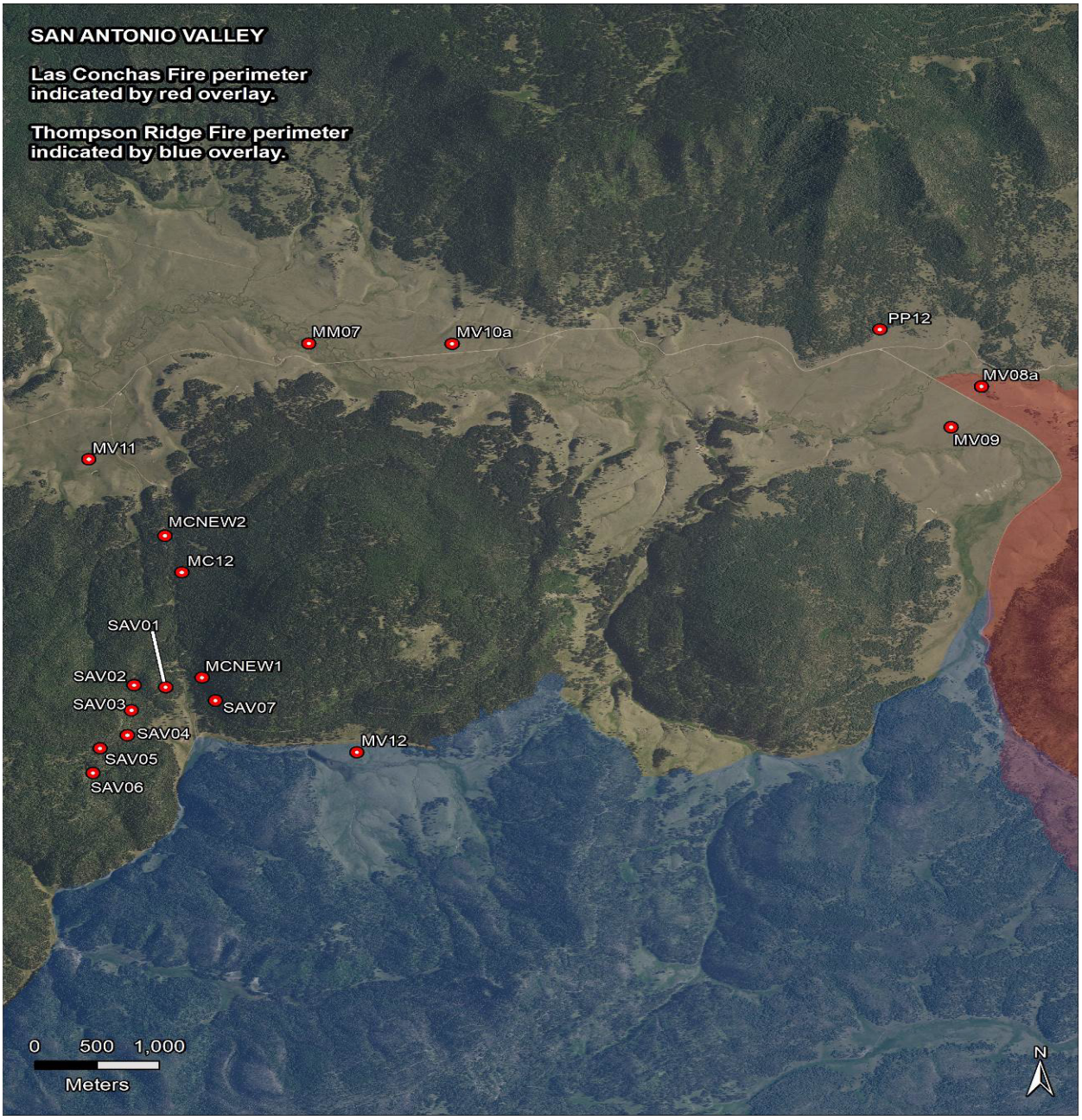



Fig. 9.6. Point locations for the San Antonio Valley survey route, Valles Caldera National Preserve, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**South Mountain-** Located in the southern portion of VCNP, the eight points on this route were on South Mountain to the west of Valle Grande (Figs. 9.1 and 9.7). All eight points on this route (SM01-SM08) were classified as mixed conifer forest, but six points were at least somewhat impacted by the 2018 Hidden Valley fire that was managed for resource benefit, making it similar to a prescribed burn. This route was established in 2017 to increase the sample size of mixed conifer forest plots, and surveyed in 2017-2024.

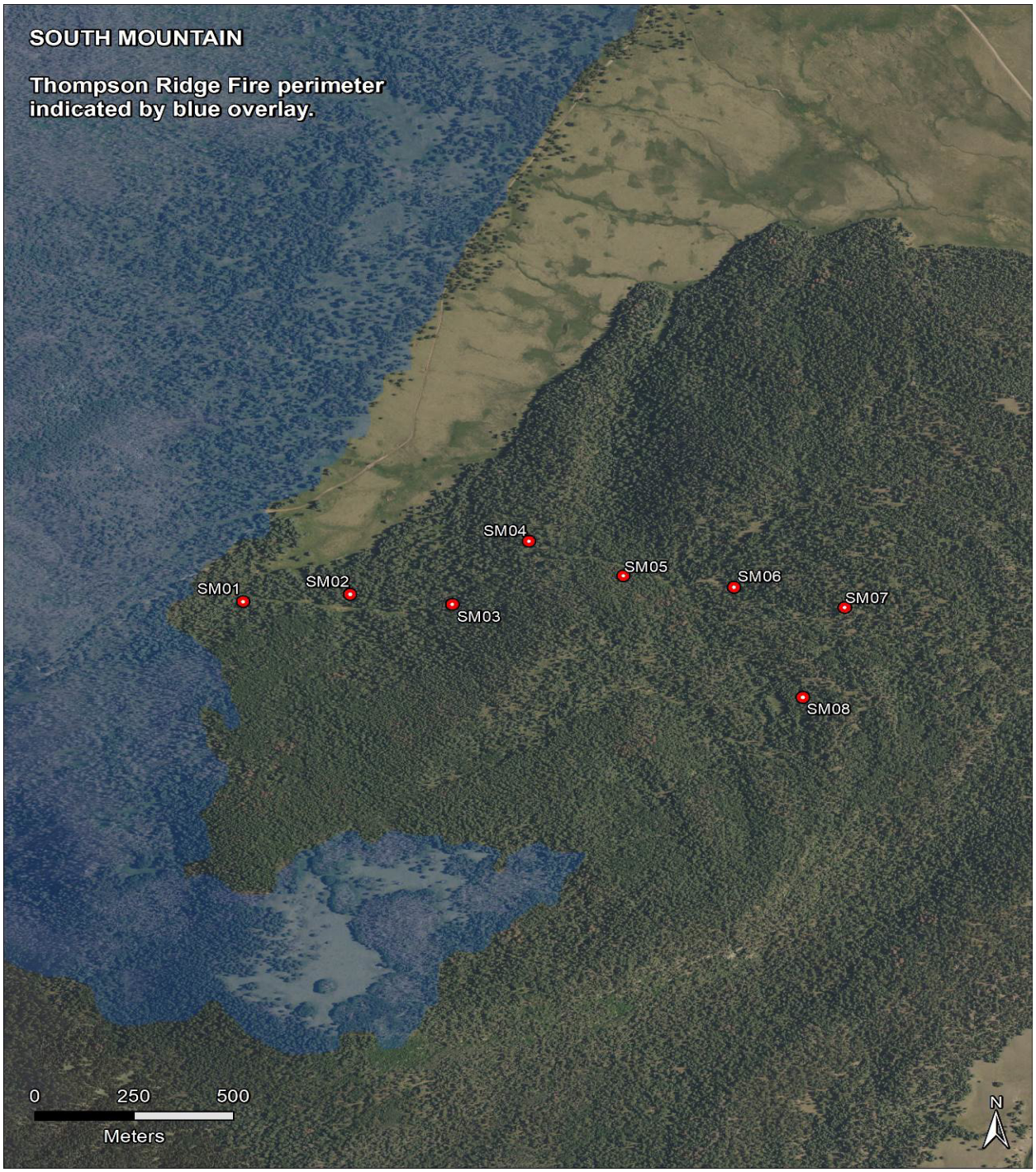



Fig. 9.7. Point locations for the South Mountain survey route, Valles Caldera National Preserve, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**USFS Area 3-** Located on a large bank on the side of Virgin Mesa above residential Area 3 north of Jemez Springs and to the west of NM 4 (Figs. 9.1 and 9.8). The six points on this route (A01-A06) were classified as ponderosa pine forest. The points on this route were thinned following the 2019 field season. This route was surveyed in 2019-2024.

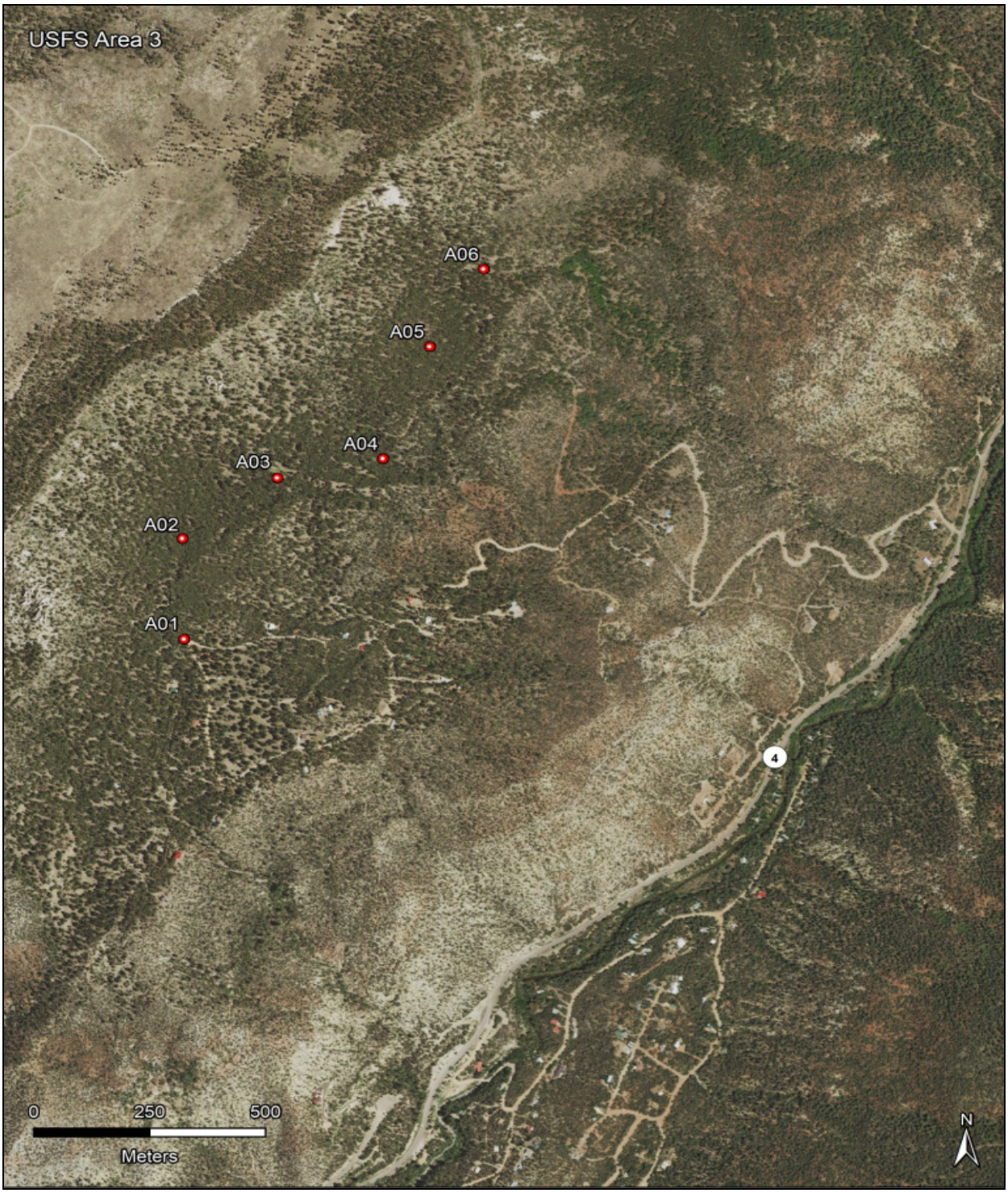


Fig. 9.8. Point locations for the USFS Area 3 survey route, Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**USFS Calaveras Canyon**- Located on USFS land northwest of Calaveras Canyon and west of VC (Figs. 9.1 and 9.9). All 10 points on this route (FMCC01-FMCC10) were classified as mixed conifer forest. This route was surveyed in 2012 and 2013.

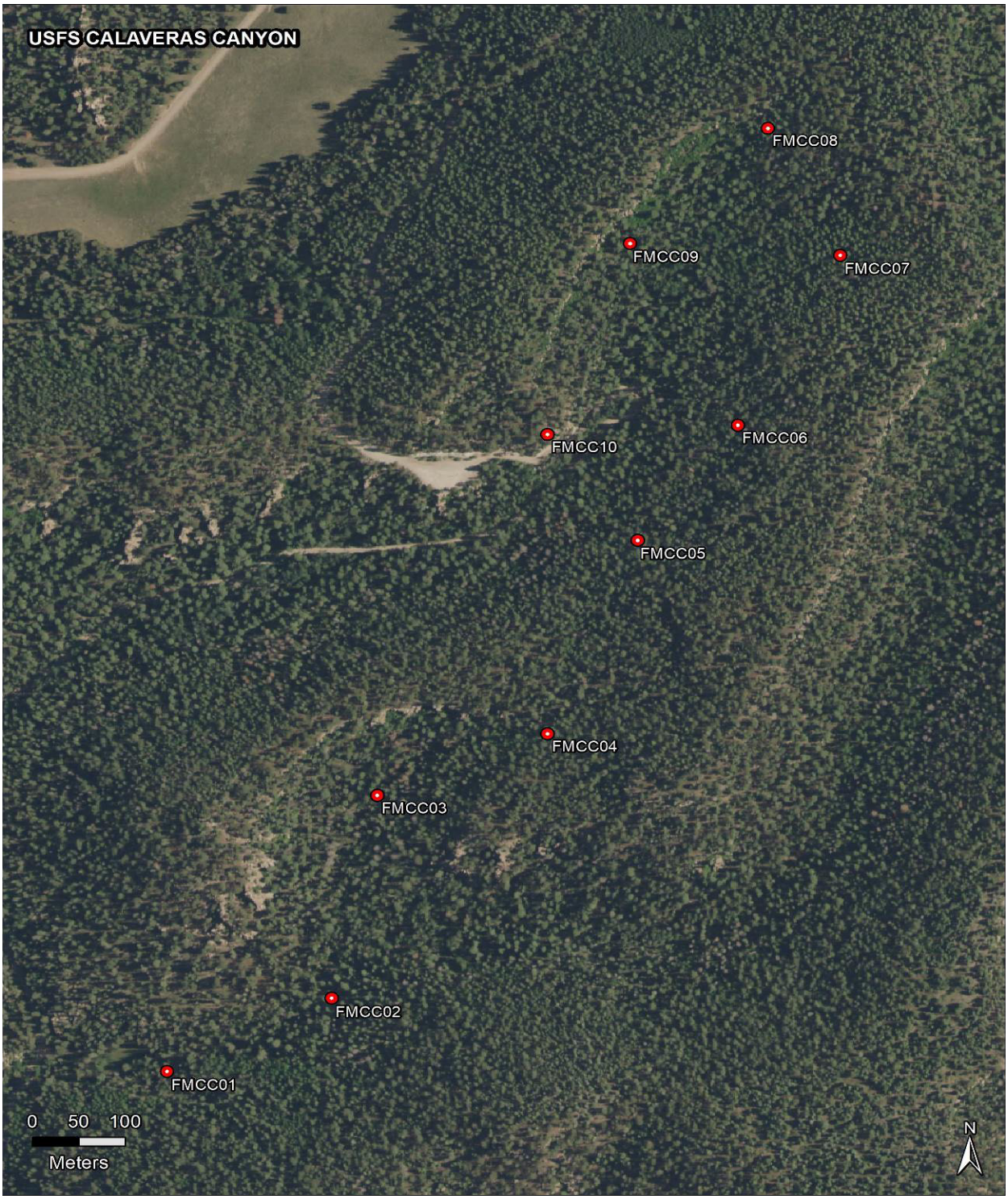



Fig. 9.9. Point locations for the USFS Calaveras Canyon survey route, Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**USFS Guadalupe River**- Located on USFS land along the west side of the Guadalupe River in the southwestern portion of the study area (Figs. 9.1 and 9.10). All 18 points on this route (GR01-GR18) were classified as riparian habitat. This route was established in 2013 and points GR01-GR12 were surveyed in 2013 and 2014. Prior to the 2016 field season, points GR02-GR07 were replaced because they were centered on a road cut above the riparian area. In 2016, points

GR01 and GR08-GR18 were surveyed.

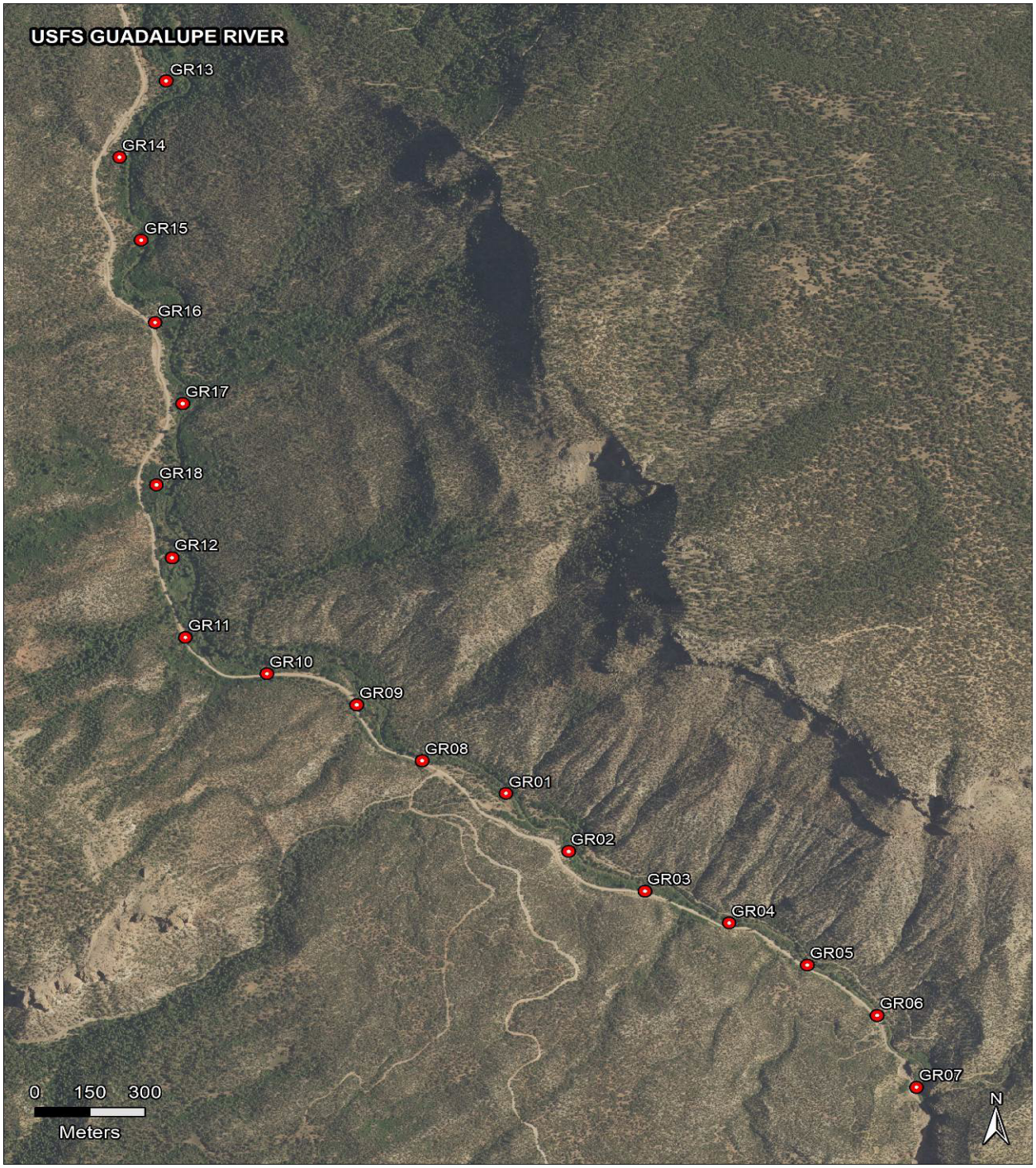



Fig. 9.10. Point locations for the USFS Guadalupe River survey route, Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico. Points GR13-GR18 replaced points GR02-GR07 in 2016.

**USFS Jemez River**- Located on USFS land along the Jemez River in the southern portion of the study area (Figs. 9.1 and 9.11). All 12 points on this route (JR01-JR12) were classified as riparian habitat. The points on this route were established in 2016 and surveyed in 2016-2024.

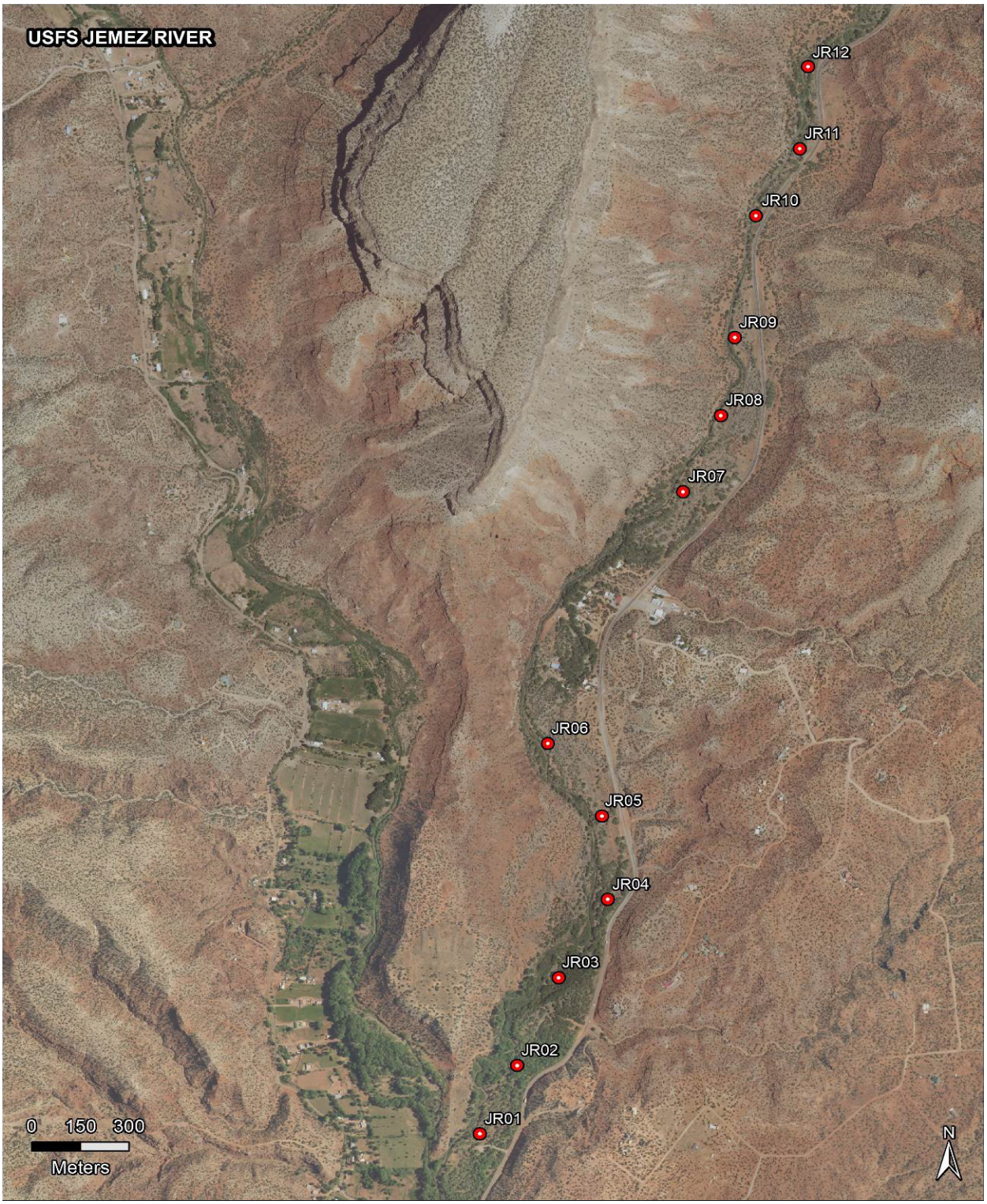



Fig. 9.11. Point locations for the USFS Jemez River survey route, Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**USFS Jemez River North**- Located on USFS land along the Jemez River in the southern portion of the study area and north of the previously-established USFS Jemez River points (Figs. 9.1 and 9.12). All 12 points on this route (JRN01-JRN12) were classified as riparian habitat. The points on this route were established in 2017 and surveyed in 2017-2024.




Fig. 9.12. Point locations for the USFS Jemez River North survey route, Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**USFS Paliza North**- Located on USFS land east of Paliza Canyon in the southern portion of the study area (Figs. 9.1 and 9.13). All 10 points on this route (PZN01-PZN10) were classified as ponderosa pine forest. Three points on this route (PZN03, PZN04, and PZN09) burned in the 2019 Conejos fire. The points on this route were established in 2014 and surveyed in 2014-2016

and 2020-2022, and 2024.

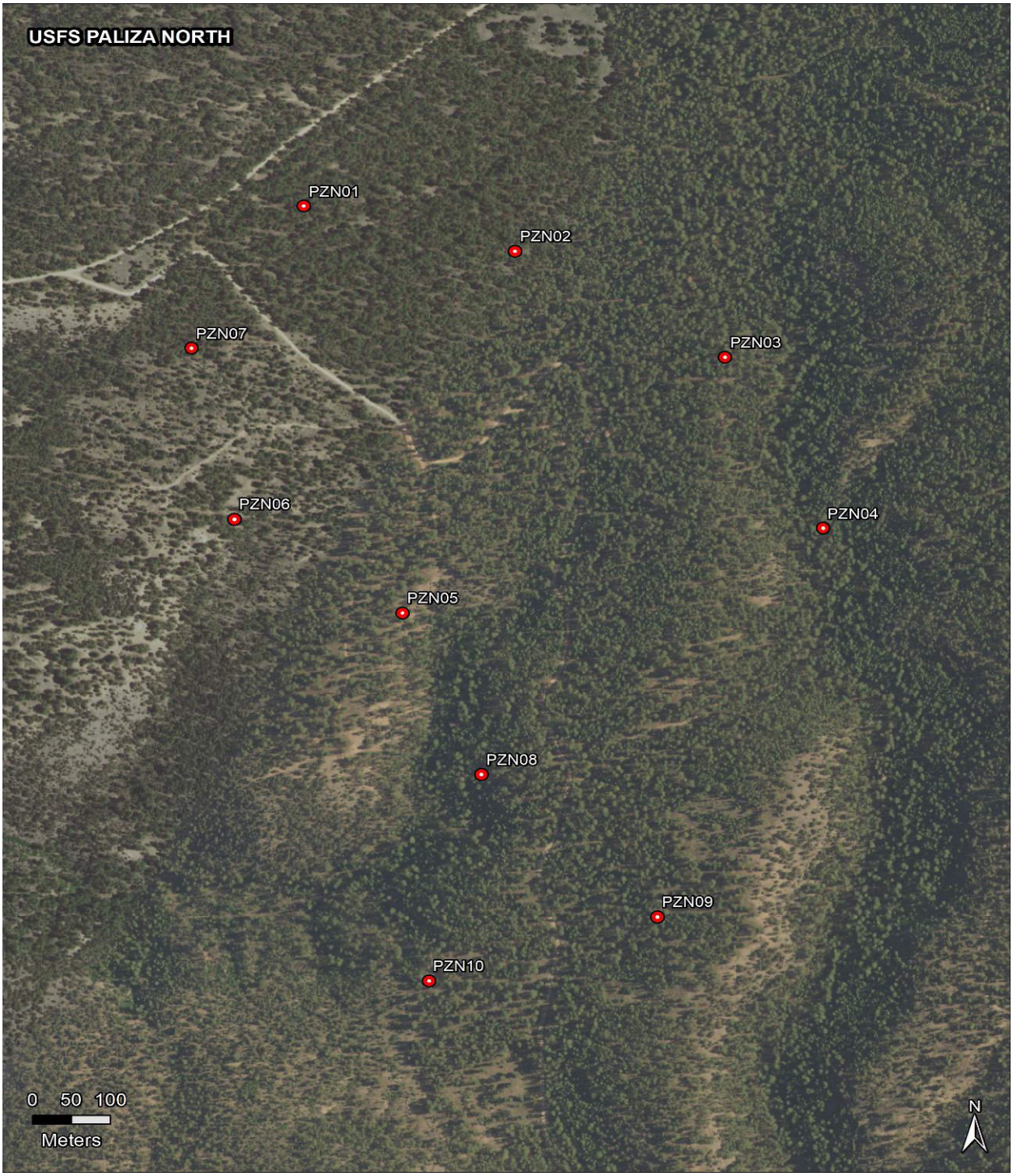



Fig. 9.13. Point locations for the USFS Paliza North survey route, Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**USFS Paliza South**- Located on USFS land south of Jemez Pueblo land near Hondo Canyon in the southern portion of the study area (Figs. 9.1 and 9.14). All 10 points on this route (PZS01-PZS10) were classified as ponderosa pine forest; however, these points fell into a transition zone between pinyon-juniper woodland and pure ponderosa pine forest. Thus, although ponderosa

pine was the dominant tree species in all 10 points, components of pinyon pine and juniper were present at these points. Two points on this route (PZS09 and PZS10) were thinned prior to the 2020 field season. The points on this route were established in 2014 and surveyed in 2014-2016 and 2020-2022, and 2024.

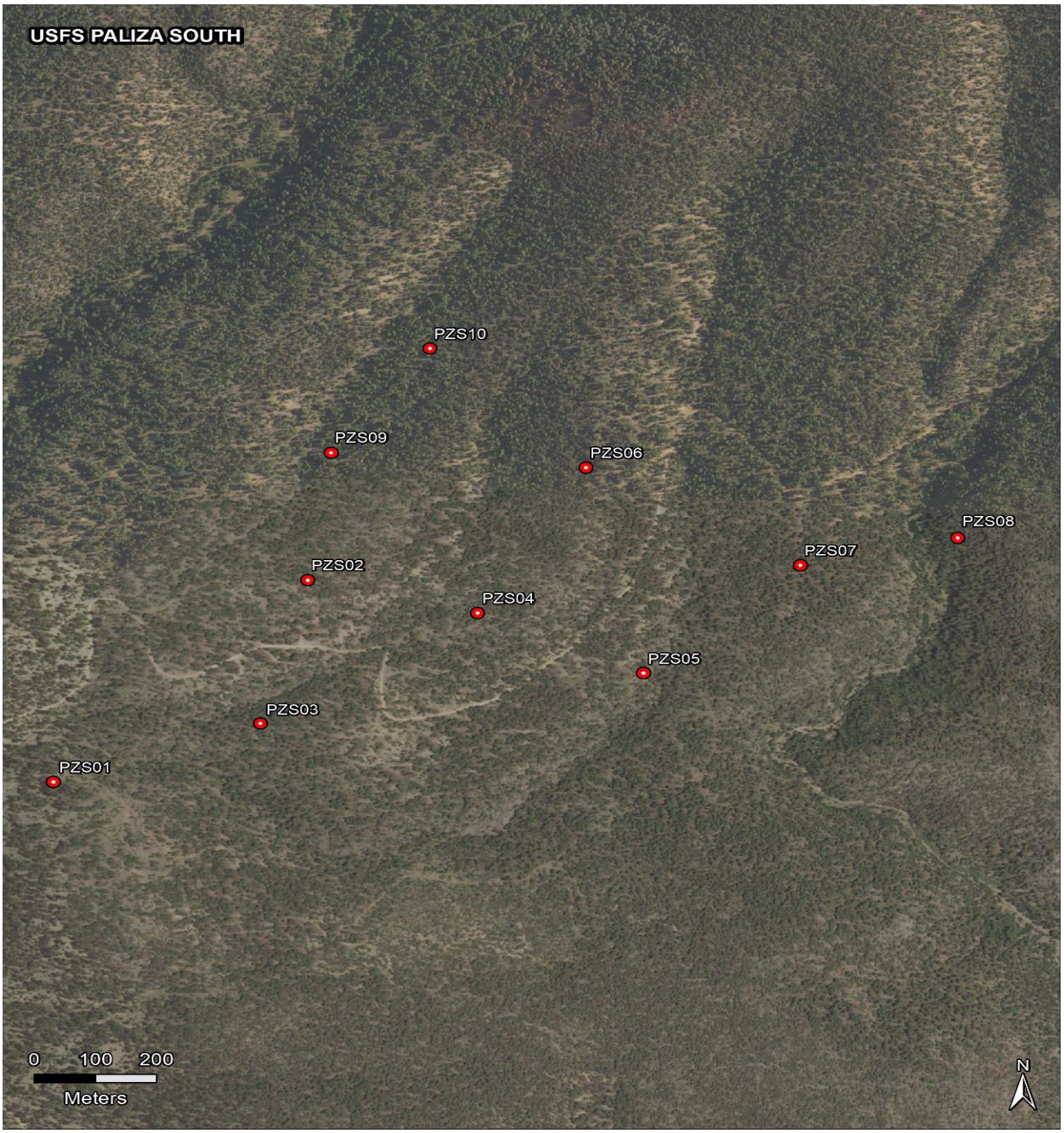



Fig. 9.14. Point locations for the USFS Paliza South survey route, Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**USFS Ponderosa Spring**- Located on USFS land on the northern flanks of Cerro del Pino to the south of VCNP (Figs. 9.1 and 9.15). All 12 points on this route (PS01-PS12) were classified as ponderosa pine forest. All 12 points burned to varying extents in the 2022 Cerro Pelado Fire. This route was surveyed during all years from 2012-2019 and in 2023.

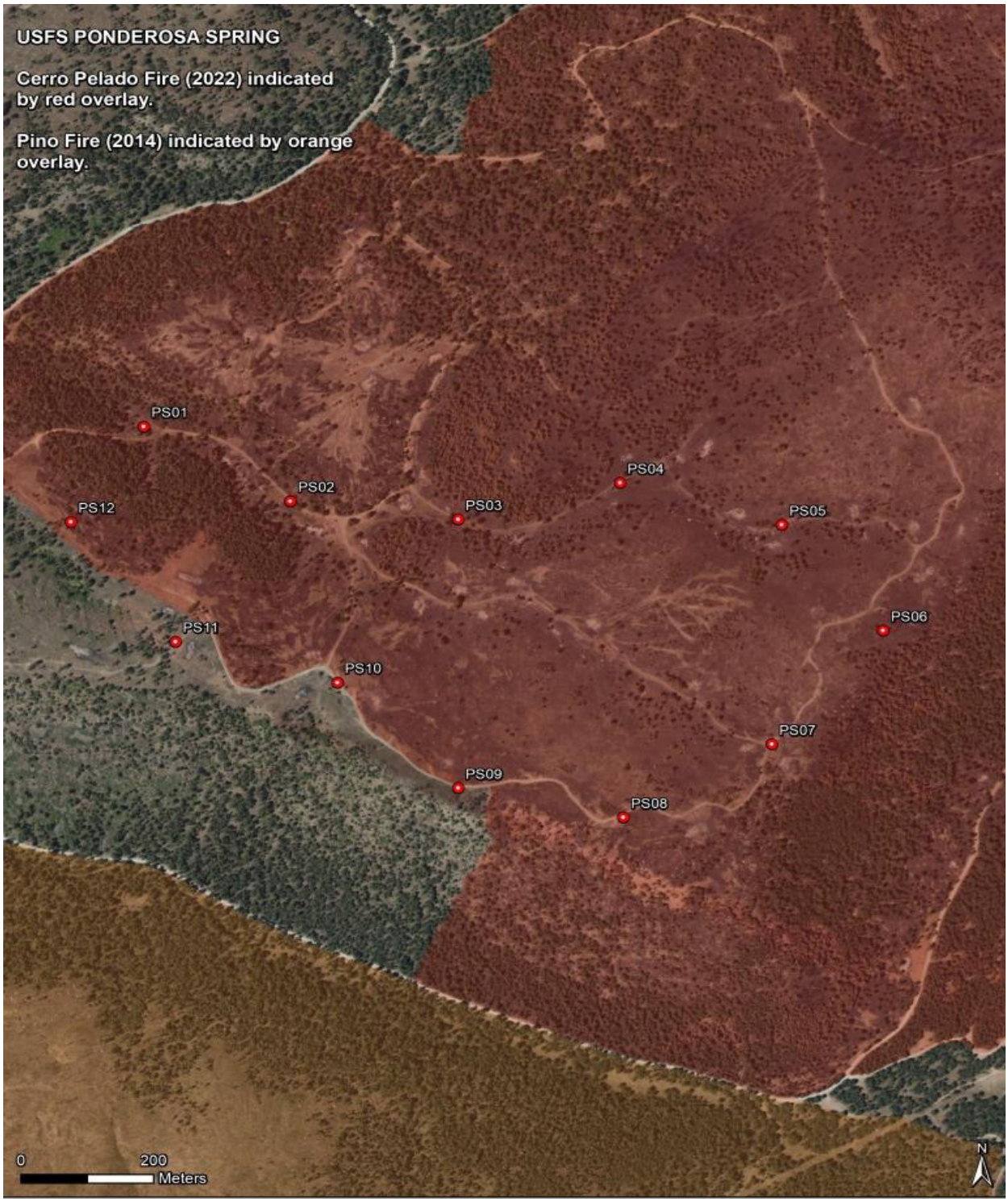



Fig. 9.15. Point locations for the USFS Ponderosa Spring survey route, Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**USFS Ponderosa Spring Experimental**- Located on USFS land on the northern and western flanks of Cerro del Pino and northeast of San Juan Canyon (Figs. 9.1 and 9.16). All 10 points on this route (PSE01-PSE10) were classified as ponderosa pine forest. But, all 10 points burned after the 2014 survey season in the Pino Fire. This route was surveyed during all years from

2012-2019 and in 2023.

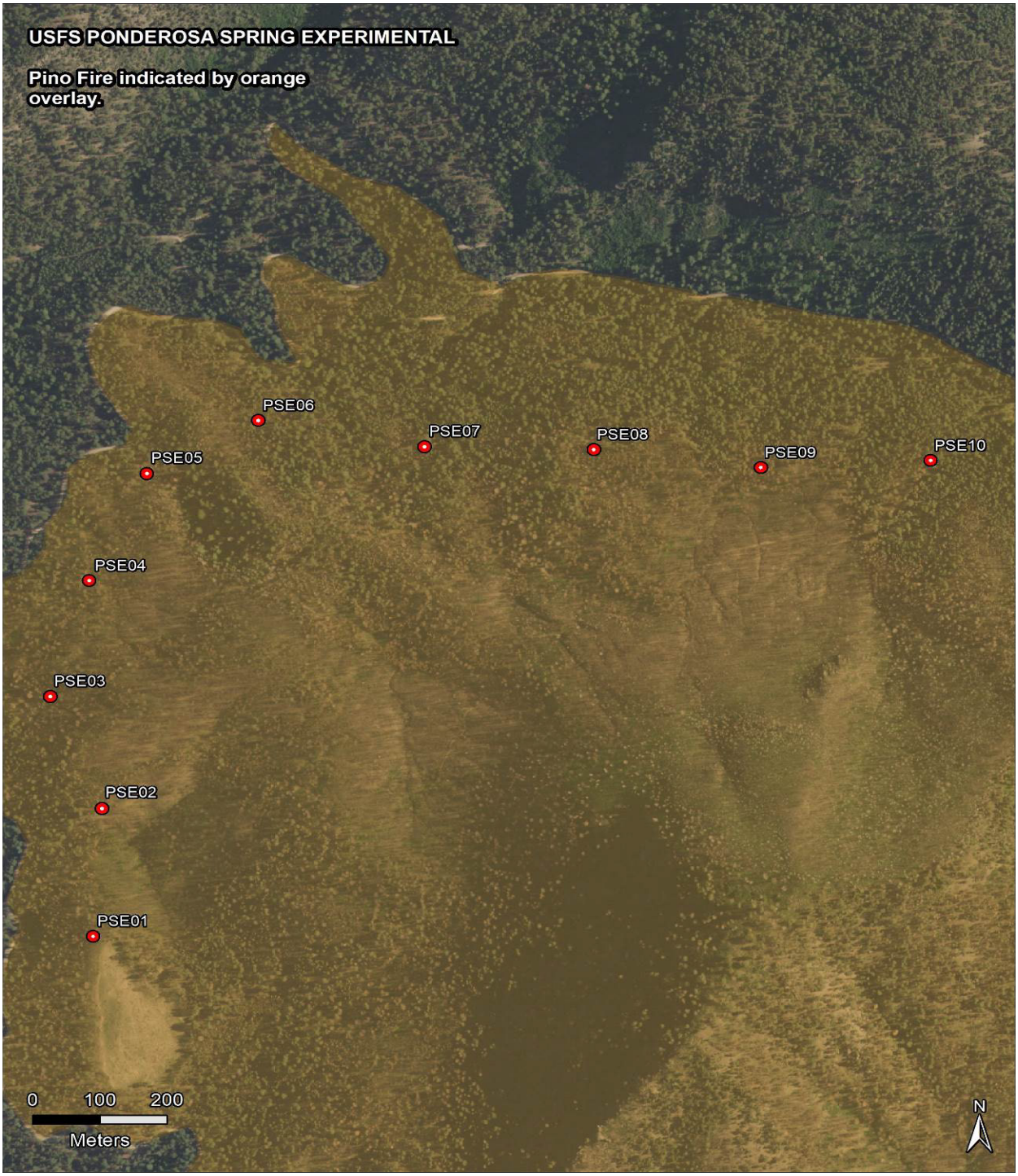



Fig. 9.16. Point locations for the USFS Ponderosa Spring Experimental survey route, Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**USFS Riparian**- Located on the edge of Redondo Meadow on both USFS land and southwestern portions of VCNP land (Figs. 9.1 and 9.17). Two points on this route were on USFS land (RIP01, RIP02) and 10 points (RIP03-RIP12) were on VCNP. All 12 points were classified as riparian habitat. This route was surveyed from 2012-2014.

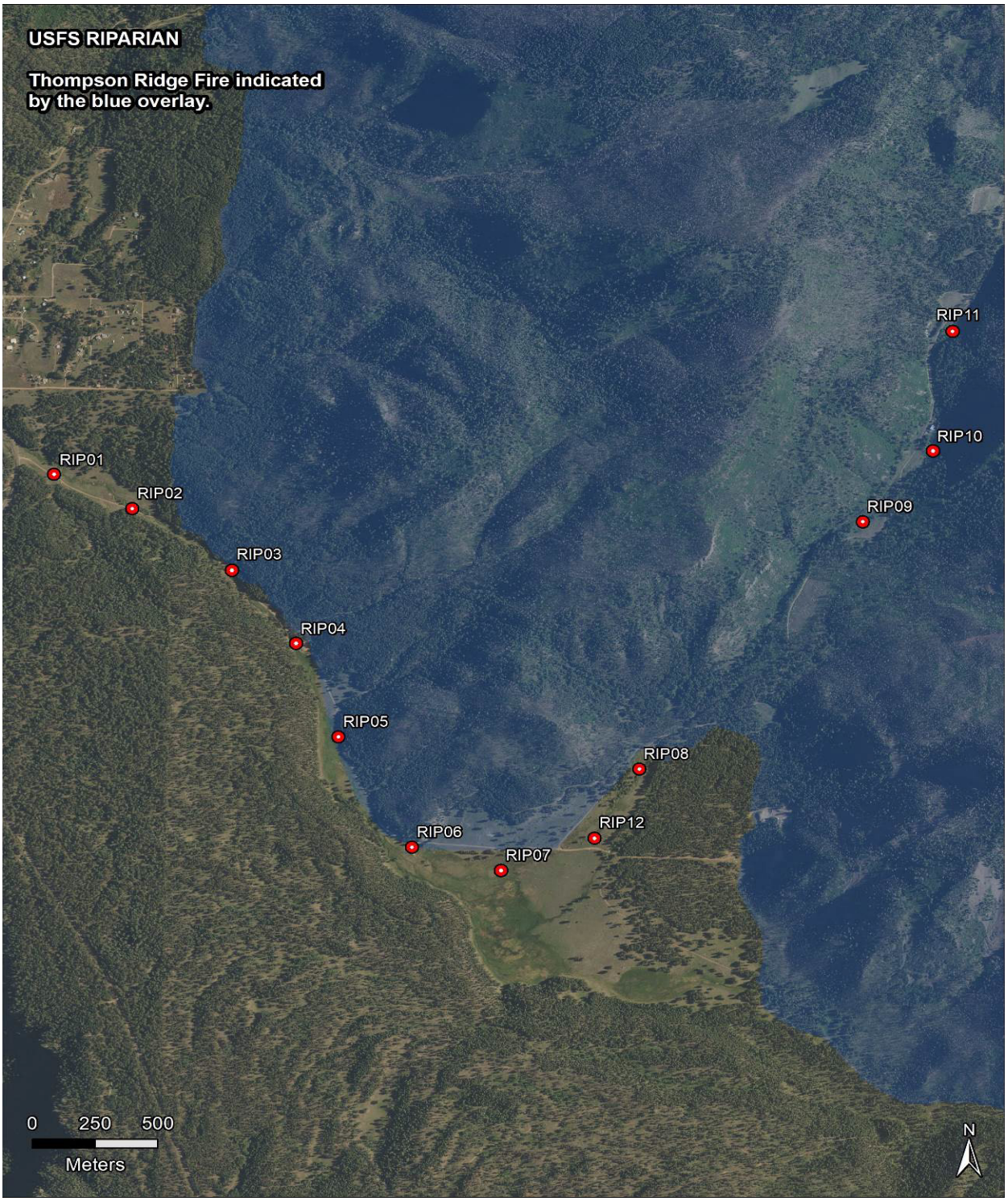



Fig. 9.17. Point locations for the USFS Riparian survey route, Santa Fe National Forest and Valles Caldera National Preserve, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

**USFS Thompson Ridge**- Located on USFS land adjacent to the Thompson Ridge residential area immediately west of VCNP (Figs. 9.1 and 9.18). All 12 points on this route (TR01-TR12) were classified as mixed conifer. This route was surveyed during 2012, 2013, and 2015-2024.

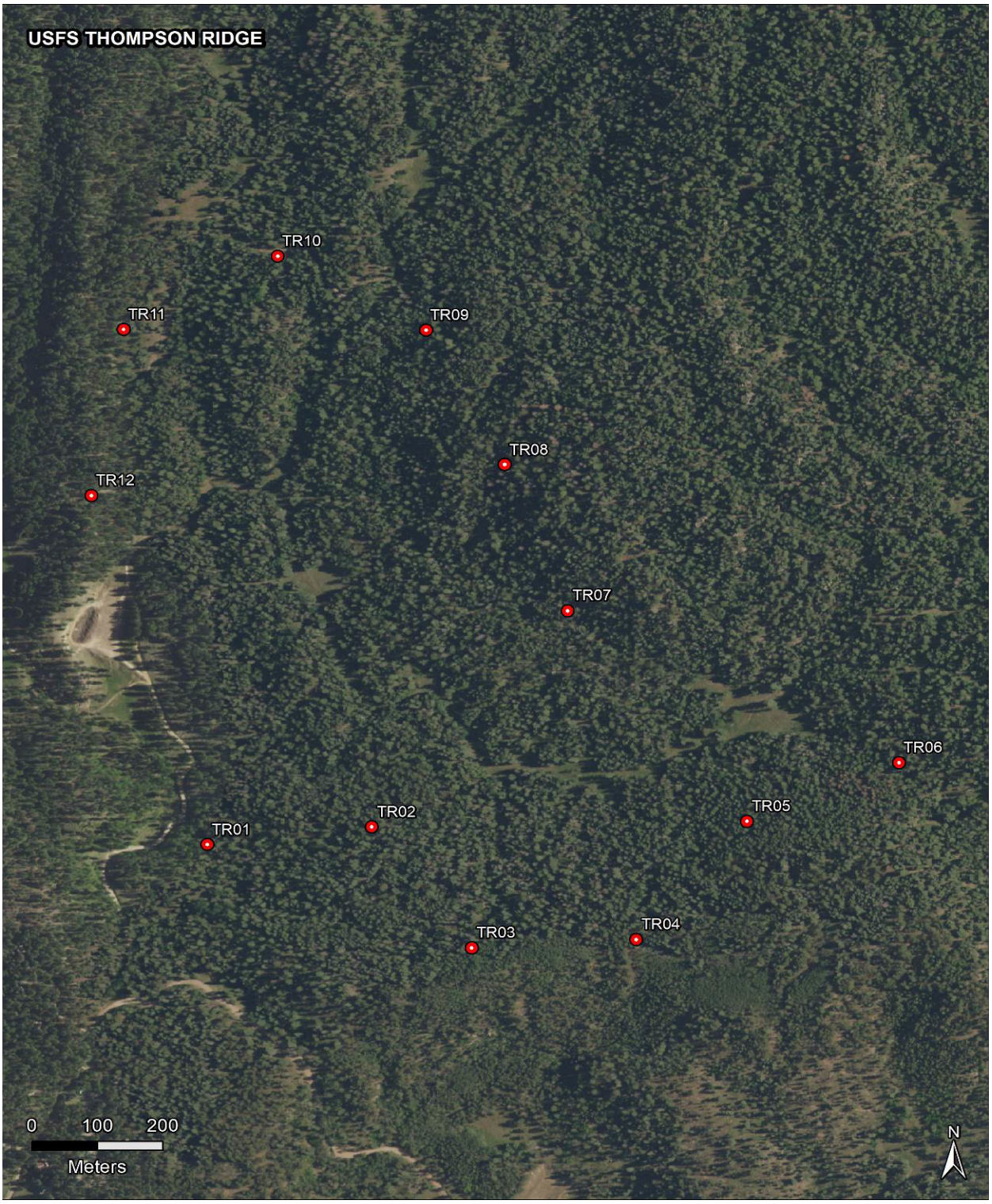


Fig. 9.18. Point locations for the USFS Thompson Ridge survey route, Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

In June 2016, three North American Breeding Bird Survey (BBS) routes were established (Fig. 9.19). The Route 10 BBS route follows FS-10 and NM Highway 4 through USFS land and the southeastern corner of the VCNP. The Redondo Border route follows roads in the VCNP running west of Redondo Peak and north into the Valle San Antonio. The Valle Grande route follows

roads from the southern border of the VCNP north through the Valle Grande to the Valle San Antonio and east through the Valle Toledo.

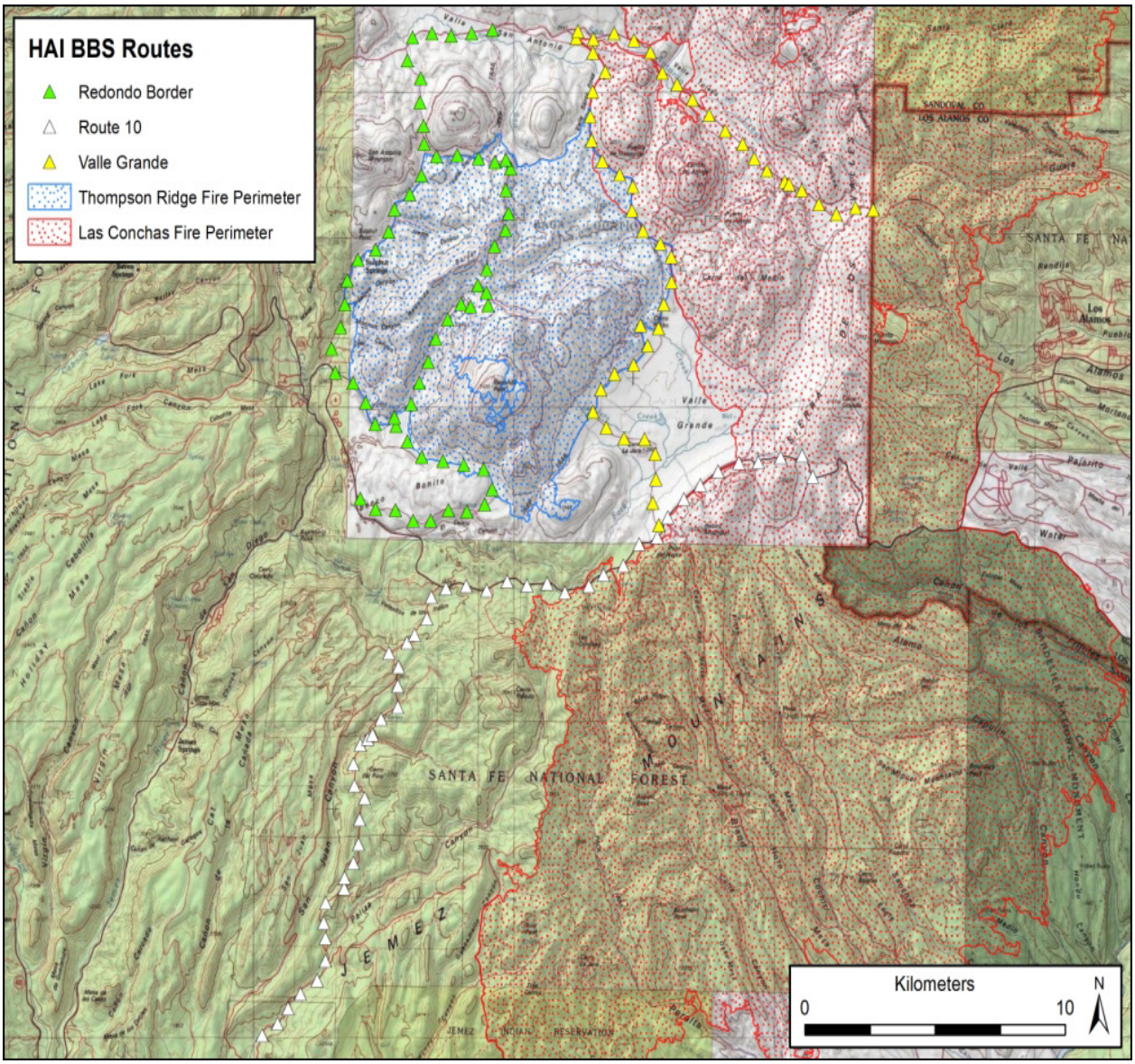



Fig. 9.19. Map of North American Breeding Bird Survey (BBS) routes in the Valles Caldera National Preserve and Santa Fe National Forest, Southwest Jemez Mountains CFLRP, Sandoval County, New Mexico.

## Methods

From 2012-2024, Hawks Aloft personnel established 213 point count locations based on input and assistance from VCNP and USFS personnel. Points were centered at least 250 m from the nearest neighboring point to assure survey independence. Universal Transverse Mercator Coordinates (UTM, NAD27) were recorded for each point to assist with relocation.

We conducted standard, 10-minute point count surveys (see Bibby et al. 2000) three times per year during the breeding season (i.e., 15 May to 30 July) at each of the points selected for survey in a given year. In some cases, fewer than three visits were completed in a given year at certain points due to inaccessibility and/or logistical issues (e.g., fire closures, washed out roads, etc.). Survey route selection for each year occurred prior to the beginning of the field season and was based on input from VCNP and USFS personnel. Nineteen observers (Gail Garber in 2012 and 2017-2024, Erin Greenlee in 2012, Michael Hilchey in 2012, Kieran Sullivan in 2012-2013, Raymond VanBuskirk in 2012-2013 and 2018, Mike Fugagli in 2013-2016, Trevor Fetz in 2014 and 2017-2024, Jennifer Goyette in 2014, Carol Fugagli in 2015-2016, Amanda Schluter in 2016-2018, Lisa Morgan in 2017, Katrina Hucks in 2018, Adam Johnson in 2018, Greg Finkelberg in 2019, John Stanek in 2019, Brian Dykstra in 2020-2022, Joe Acord in 2022 and 2023, Jessalyn Ayars in 2024, and Karl Buermeyer in 2024), each experienced with avian identification by sight and sound, conducted the surveys. At each survey point, the observer recorded all birds seen or heard during a ten-minute period. We recorded the distance of birds from the survey point using Leupold RX-600i laser rangefinders whenever possible; in many instances, distance to the bird's location was estimated. We noted flyovers separately without measuring distance. We began surveys in the morning within 30 minutes after sunrise and attempted to finish within four hours after sunrise.

BBS routes were established by National Park Service personnel (Stephen Fettig) prior to the onset of this study, in accordance with U.S. Geological Survey (1998) protocol. Roadside survey stops for each of the three routes were located approximately 800 meters apart. UTM coordinates were recorded for each stop to assist with relocation. The Redondo Border and Route 10 survey routes were comprised of 50 points each. The Valle Grande route originally was comprised of 49 points because the road ended before a 50th point could be established; however, only 41 points were accessible in 2017, 2019, and 2020. All 49 points were surveyed in 2021-2024. Each route was surveyed once during the month of June. Surveys started within 30 minutes of sunrise and were completed within four hours after sunrise. Surveyors drove each route, stopping at all survey points. At each stop, observers exited the vehicle and conducted a 3-minute count, recording all birds seen and heard within a 400 m radius. Traffic and excessive noise were noted at each point.

All standard, 10-minute point count data presented are based on observations within 125 m of a point in order to minimize the likelihood of double-counting individuals at adjacent points. Although we include species detected only as flyovers in a cumulative species list, we define species richness for all analyses as the total number of species detected within 125 m from a point, excluding species observed only as flyovers. We used post-hoc Tukey-Kramer HSD tests to determine statistical significance. We set statistical significance for all comparisons at $\alpha < 0.05$. All statistical analyses were conducted using JMP 5.0 statistical software (SAS institute 2002). Because BBS data were collected based on a separate protocol, we excluded those from

all data summaries and analyses and provide a separate species list for detections collected during 2016, 2017, and 2019-2024 BBS surveys. No BBS surveys were conducted in 2018 due to fire closures in the Valles Caldera National Preserve and the Santa Fe National Forest.

**Results**

Cumulatively from 2012-2024, we documented 141 bird species and 43,976 total bird detections (Appendix 9.1). Western Tanager (*Piranga ludoviciana*, 6.78% of all detections), Yellow-rumped Warbler (*Setophaga coronata*, 5.68%), and Pygmy Nuthatch (*Sitta pygmaea*, 5.41%) were the most common species detected. No federally listed bird species were detected during surveys, but we documented two New Mexico State Threatened Species: Peregrine Falcon (*Falco peregrinus*; N = 9) and Gray Vireo (*Vireo vicinior*; N = 8) (Bison-M 2024). We also documented 10 species of conservation concern as listed by the U.S. Fish and Wildlife Service (2021): Grace's Warbler (*Setophaga graciae*; N = 1,489), Broad-tailed Hummingbird (*Selasphorus platycercus*, N = 550), Evening Grosbeak (*Coccothraustes vespertinus*, N = 242), Clark's Nutcracker (*Nucifraga columbiana*, N = 220), Virginia's Warbler (*Leiothlypis virginiae*, N = 196), Cassin's Finch (*Haemorhous cassinii*, N = 52), Olive-sided Flycatcher (*Contopus cooperi*, N = 43), Lewis's Woodpecker (*Melanerpes lewis*, N = 19), Pinyon Jay (*Gymnorhinus cyanocephalus*, N = 7), and Flammulated Owl (*Psiloscops flammeolus*, N = 2). In addition, we detected a total of 57 species of conservation concern (see Appendix 9.2) as identified by relevant entities in New Mexico. These entities include New Mexico Avian Conservation Partners (2024), Natural Heritage New Mexico (2024), the U.S. Forest Service, and the New Mexico Department of Game and Fish (Bison-M 2024).

Among habitat types, mean cumulative avian density was highest in mixed conifer (2.51 birds/ha) and burned mixed conifer (2.28 birds/ha), and lowest in burned mountain valley grassland (1.16 birds/ha) and mountain valley grassland (1.22 birds/ha; Table 9.1). Avian density was significantly higher in mixed conifer forest than all other habitat types (Tukey-Kramer test). Further, avian density was significantly higher in all forest types (unburned and burned) and riparian habitat than mountain valley grassland and burned mountain valley grassland. But, avian density in unburned ponderosa pine was significantly lower than all other forest types. Among years, avian density was significantly higher in 2023 and 2024 than all other years except for 2018, and 2020-2022; and density in 2020 and 2022 was significantly higher than all years from 2013-2017 and 2019 (Tukey-Kramer test; Table 9.2). Avian density was significantly lower in 2014-2017 than all other years except 2019.

The most abundant species by habitat were: mixed conifer forest - Western Tanager (8.89% of all detections), Yellow-rumped Warbler (8.74%), and Mountain Chickadee (*Poecile gambeli*, 7.63%); mixed conifer burn - House Wren (*Troglodytes aedon*, 8.05%), Violet-green Swallow (*Tachycineta thalassina*, 7.90%), and Ruby-crowned Kinglet (*Corthylio calendula*, 7.28%); mountain meadow - Western Meadowlark (*Sturnella neglecta*, 28.30%), Savannah Sparrow (*Passerculus sandwichensis*, 25.47%), and Vesper Sparrow (*Pooecetes gramineus*, 16.04%); mountain valley grassland - Vesper Sparrow (28.65%), Western Meadowlark (24.67%), and Chihuahuan Meadowlark (*Sturnella lilianae*, 16.18%); mountain valley grassland burn - Vesper Sparrow (24.84%), Western Meadowlark (12.11%) and Chihuahuan Meadowlark (10.74%); ponderosa pine forest - Pygmy Nuthatch (12.09%), Western Tanager (8.64%), and Grace's

Warbler (7.16%); ponderosa pine burn - Western Tanager (7.88%), Western Wood-Pewee (*Contopus sordidulus*, 6.28%), and Pygmy Nuthatch (6.01%); riparian - Yellow-breasted Chat (*Icteria virens*; 12.28%), Black-chinned Hummingbird (*Archilochus alexandri*, 5.73%), and Spotted Towhee (*Pipilo maculatus*, 5.67%).

Table 9.1. Cumulative avian density (mean number of birds/ha) and statistical significance by habitat type, Jemez CFLRP, 2012-2024. Habitat types not connected by a common letter are significantly different (Tukey-Kramer test).

| Habitat | # Surveys | Statistical Significance | | | | | # Birds/ha |
|---|---|---|---|---|---|---|---|
| Mixed Conifer | 880 | A | | | | | 2.51 |
| Mixed Conifer Burn | 427 | | B | | | | 2.28 |
| Ponderosa Pine Burn | 694 | | | C | | | 2.07 |
| Riparian | 790 | | | C | | | 2.05 |
| Ponderosa Pine | 1443 | | | | D | | 1.71 |
| Mountain Meadow | 14 | | B | C | D | E | 1.54 |
| Mountain Valley Grassland | 63 | | | | | E | 1.22 |
| Mountain Valley Grassland Burn | 141 | | | | | E | 1.16 |

Table 9.2. Cumulative avian density (mean number of birds/ha) and statistical significance by year, Jemez CFLRP, 2012-2024. Years not connected by a common letter are significantly different (Tukey-Kramer test).

| Year | # Routes | # Surveys | Statistical Significance | | | | | | # Birds/ha |
|---|---|---|---|---|---|---|---|---|---|
| 2023 | 12 | 366 | A | | | | | | 2.41 |
| 2024 | 12 | 360 | A | | | | | | 2.41 |
| 2020 | 11 | 351 | A | B | | | | | 2.32 |
| 2022 | 12 | 360 | A | B | | | | | 2.30 |
| 2018 | 10 | 294 | A | B | C | | | | 2.27 |
| 2021 | 12 | 360 | A | B | C | | | | 2.25 |
| 2012 | 10 | 334 | | B | C | | | | 2.09 |
| 2013 | 11 | 273 | | | C | D | | | 2.03 |
| 2019 | 11 | 357 | | | | D | E | | 1.81 |
| 2014 | 11 | 354 | | | | | E | F | 1.65 |
| 2015 | 10 | 328 | | | | | E | F | 1.61 |
| 2017 | 10 | 339 | | | | | | F | 1.54 |
| 2016 | 11 | 376 | | | | | | F | 1.53 |

Cumulative avian richness was highest in riparian habitat (115 species) and ponderosa pine forest (89 species; Table 9.3). Avian richness was lowest in mountain meadow (14 species) and mountain valley grassland (33 species). By year, avian richness was highest in 2024 (102 species) and lowest in 2015 (70 species; Table 9.4).

Table 9.3. Cumulative avian richness (total number of species detected within 125 m of points) by habitat type, Jemez CFLRP, 2012-2024.

| Habitat | # Surveys | # Species |
|---|---|---|
| Riparian | 790 | 115 |
| Ponderosa Pine | 1443 | 89 |
| Ponderosa Pine Burn | 694 | 86 |
| Mixed Conifer | 880 | 81 |
| Mixed Conifer Burn | 427 | 79 |
| Mountain Valley Grassland Burn | 141 | 48 |
| Mountain Valley Grassland | 63 | 33 |
| Mountain Meadow | 14 | 14 |

Table 9.4. Avian richness (total number of species detected within 125 m of points) by year, Jemez CFLRP, 2012-2024.

| Year | # Surveys | # Species |
|---|---|---|
| 2024 | 360 | 102 |
| 2022 | 360 | 100 |
| 2018 | 294 | 97 |
| 2023 | 366 | 96 |
| 2020 | 351 | 94 |
| 2021 | 360 | 92 |
| 2019 | 357 | 86 |
| 2014 | 354 | 84 |
| 2013 | 273 | 78 |
| 2017 | 339 | 78 |
| 2016 | 376 | 74 |
| 2012 | 334 | 72 |
| 2015 | 328 | 70 |

Banco Bonito (classified as ponderosa pine forest) was the survey route where the longest term and most extensive prescribed understory thinning treatments occurred. Cumulative avian density from 2012-2024 was significantly higher at thinned points (2.27 birds/ha) than at unthinned points (1.68 birds/ha; Tukey-Kramer test). But further breakdown of data from Banco Bonito based on both thinning treatments and prescribed fire revealed that avian density was highest at points subjected to thinning and prescribed fire (2.41 birds/ha; N = 231 visits), followed by points where only thinning had occurred (1.97 birds/ha; N = 109 visits), and unburned control points (1.68 birds/visit; N = 411 visits). Cumulative avian density at points that were thinned and burned was significantly higher than density at points where only thinning had occurred and density at control points was significantly lower than points where only thinning

had occurred (Tukey-Kramer test).

Among points where data were collected before and after thinning (N = 5), cumulative avian density through 2024 was highest after both thinning and prescribed fire at four of the five points and highest after thinning but before fire at one point (Table 9.5). Avian density after thinning but before prescribed fire was higher than before thinning at three points and lower than before thinning at the other two points. At two points (BB12 and BB14), avian density was significantly higher after both thinning and prescribed fire than before thinning (Tukey-Kramer test). At one point (BB16), avian density was significantly higher after both thinning and prescribed fire than after only thinning.

Table 9.5. Comparison of mean avian density (mean number of birds/ha) before and after thinning and prescribed burning at five points in ponderosa pine forest, Banco Bonito survey route, Valles Caldera National Preserve, Jemez CFLRP, 2012-2024.

| **Point & Treatment Type** | **# Surveys** | **# Birds/ha** |
|---|---|---|
| BB08 PP Thinned | 10 | 2.22 |
| BB08 PP Thinned & Burned | 21 | 2.20 |
| BB08 PP Unthinned | 7 | 1.89 |
| BB10 PP Thinned & Burned | 21 | 2.81 |
| BB10 PP Unthinned | 7 | 2.62 |
| BB10 PP Thinned | 10 | 2.47 |
| BB12 PP Thinned & Burned* | 21 | 2.58 |
| BB12 PP Thinned | 11 | 1.96 |
| BB12 PP Unthinned | 5 | 1.35 |
| BB14 PP Thinned & Burned* | 21 | 2.55 |
| BB14 PP Thinned | 9 | 2.26 |
| BB14 PP Unthinned | 8 | 1.48 |
| BB16 PP Thinned & Burned** | 21 | 2.13 |
| BB16 PP Unthinned | 8 | 1.76 |
| BB16 PP Thinned | 9 | 1.40 |

*Avian density was significantly higher after thinning and burning than before thinning.
**Avian density was significantly higher after thinning and burning than after only thinning.

Cumulative avian richness (2012-2024) at Banco Bonito was higher at thinned points (59 species) than unthinned points (56 species; Table 9.6). The most common species at unthinned points were Pygmy Nuthatch (15.10% of detections), Western Tanager (9.57%), and Yellow-rumped Warbler (8.83%). The most common species at thinned points were Western Wood-Pewee (9.57% of detections), Pygmy Nuthatch (9.33%), and Western Tanager (8.59%).

Table 9.6. Comparison of species richness (total number of species detected w/in 125 m) at 22 unthinned and thinned points in ponderosa pine forest by year, Banco Bonito survey route, Valles Caldera National Preserve, Jemez CFLRP, 2012-2024.

| Year | Type | # Points | # Visits | # Species |
|---|---|---|---|---|
| 2012 | Unthinned | 13 | 39 | 31 |
| 2012 | Thinned | 3 | 9 | 18 |
| 2013* | Unthinned | 13 | 26 | 32 |
| 2013* | Thinned | 3 | 6 | 16 |
| 2014 | Unthinned | 12** | 34 | 31 |
| 2014 | Thinned | 6** | 13*** | 27 |
| 2015 | Unthinned | 8 | 24 | 26 |
| 2015 | Thinned | 8 | 24 | 28 |
| 2016 | Unthinned | 8 | 24 | 24 |
| 2016 | Thinned | 8 | 24 | 27 |
| 2017 | Unthinned | 11 | 33 | 26 |
| 2017 | Thinned | 11 | 33 | 29 |
| 2018 | Unthinned | 11 | 33 | 32 |
| 2018 | Thinned | 11 | 33 | 35 |
| 2019 | Unthinned | 11 | 33 | 24 |
| 2019 | Thinned | 11 | 33 | 30 |
| 2020 | Unthinned | 11 | 33 | 34 |
| 2020 | Thinned | 11 | 33 | 34 |
| 2021 | Unthinned | 11 | 33 | 31 |
| 2021 | Thinned | 11 | 33 | 35 |
| 2022 | Unthinned | 11 | 33 | 34 |
| 2022 | Thinned | 11 | 33 | 34 |
| 2023 | Unthinned | 11 | 33 | 40 |
| 2023 | Thinned | 11 | 33 | 34 |
| 2024 | Unthinned | 11 | 33 | 38 |
| 2024 | Thinned | 11 | 33 | 34 |
| Totals | Unthinned | 11 | 411 | 56 |
| | Thinned | 11 | 340 | 59 |
| | All | 22 | 738 | 66 |

*Only two surveys occurred at each point in 2013 due to accessibility limitations.
**Two points changed from Control to Treated for the 3rd survey in 2014.
***No first period survey occurred at BB12 in 2014 due to thinning activity.

Prescribed thinning of ponderosa pine forest also occurred at three USFS survey routes. The six USFS Area 3 points and two points on the USFS Paliza South route were thinned prior to the 2020 survey season. Eight points on the USFS Paliza North route were thinned prior to the 2024 survey season. Because surveys were initiated at USFS Area 3 in 2019, the sample size for pre-thinning data was very small (n = 18 total surveys). Avian density was lowest in 2019 (0.68 birds/ha), increased in 2020 (1.00 birds/ha) and 2021 (1.12 birds/ha), and was highest in 2022 and 2023 (1.65 birds/ha both years), but decreased in 2024 (0.84 birds/ha). Avian density at the USFS Area 3 points was significantly higher in 2022 and 2023 than all other years (Tukey-Kramer test). There were no significant differences in avian density between the only pre-thinning year surveys conducted in 2019 and three of the post-thinning years (2020, 2021 and 2024). Avian richness was lowest in 2019 (17 species) and slightly increased in 2020 (18 species) and 2021 (20 species), before a more substantial increase in 2022 (27 species). Avian richness decreased in 2023 (22 species) and 2024 (19 species).

With only two thinned points on the USFS Paliza South route, the sample size for that area also is small (n = 16 total pre-thinning surveys, n = 24 post-thinning surveys). Pre-thinning surveys at USFS Paliza South were conducted in 2014-2016, but not all scheduled surveys were completed during those years due to fire restrictions. Post-thinning surveys at USFS Paliza South were conducted in 2020-2022 and 2024. Cumulative avian density at the two USFS Paliza South points was slightly higher during post-thinning years (1.88 birds/ha) than pre-thinning (1.82 birds/ha), but the difference was not statistically significant (Tukey-Kramer test). Avian richness was higher post-thinning (35 species) than pre-thinning (30 species).

Because 2024 was the only year the USFS Paliza North route was surveyed after thinning occurred, the post-thinning sample size is small (n = 24 surveys), but the pre-thinning sample size is robust (n = 144 surveys). Pre-thinning surveys were conducted at USFS Paliza North in 2014-2016 and 2020-2022. Cumulative avian density at the eight USFS Paliza North points was higher pre-thinning (1.74 birds/ha) than post-thinning (1.45 birds/ha), but the difference was not statistically significant (Tukey-Kramer test). Avian richness was higher pre-thinning (54 species) than post-thinning (23 species).

Prescribed thinning also occurred at eight of the 10 mixed conifer points on the San Antonio Valley route. Among the eight points where data were collected before and after thinning, cumulative avian density through 2024 was significantly higher after thinning (3.10 birds/ha) than before thinning (2.24 birds/ha; Tukey-Kramer test). Data collection at seven of the eight thinned points did not begin until 2017, and thinning occurred before the 2020 survey season (2 points) and the 2021 survey season (6 points). For thorough evaluation of mixed conifer on this route and to increase sample size, we included the two unthinned points in data analyses. Pre-thinned mixed conifer density increased slightly with the inclusion of the two unthinned points (2.32 birds/ha), but remained significantly lower than the post-thinning density (Tukey-Kramer test). Among points where data were collected before and after thinning, cumulative avian density through 2024 was higher after thinning at all eight points and significantly higher at six points (Tukey-Kramer test; Table 9.7).

Table 9.7. Comparison of mean avian density (# birds/ha) before and after thinning at eight points in mixed conifer forest, San Antonio Valley survey route, Valles Caldera National Preserve, Jemez CFLRP, 2012-2024. Density differences were statistically significant at points marked with an asterisk.

| Point & Treatment Type | # Surveys | # Birds/ha |
|---|---|---|
| MCNew1 Thinned | 15 | 2.42 |
| MCNew1 None | 22 | 1.94 |
| SA01 Thinned* | 12 | 3.09 |
| SA01 None | 11 | 2.11 |
| SA02 Thinned* | 12 | 3.19 |
| SA02 None | 11 | 2.35 |
| SA03 Thinned* | 12 | 3.40 |
| SA03 None | 11 | 2.56 |
| SA04 Thinned* | 12 | 3.30 |
| SA04 None | 11 | 2.09 |
| SA05 Thinned* | 12 | 3.53 |
| SA05 None | 11 | 2.54 |
| SA06 Thinned | 12 | 3.70 |
| SA06 None | 11 | 2.85 |
| SA07 Thinned* | 15 | 2.49 |
| SA07 None | 8 | 1.61 |

Cumulative avian richness (2012-2024) at the mixed conifer San Antonio Valley points was higher at unthinned points (53 species) than thinned points (49 species; Table 9.8). Avian richness among species of conservation concern was slightly higher after thinning (23 species) than before thinning (21 species). The most common species at unthinned points were Western Tanager (9.10% of detections), Mountain Chickadee (8.53%), and Yellow-rumped Warbler (8.01%). The most common species at thinned points were Western Tanager (10.83% of detections), Yellow-rumped Warbler (8.25%), and Ruby-crowned Kinglet (5.22%).

A direct comparison of burned and unburned mixed conifer forest was made possible by the 2018 Hidden Valley fire that impacted the South Mountain survey route. For increased detail, points with less than 10% of their survey area within the burn perimeter (n = 3) were classified as unburned mixed conifer, points entirely inside the burn perimeter were classified as burned mixed conifer (n = 3), and points with 60-70% of their survey area outside the burn perimeter were classified as “mixed” (n = 2). Cumulative avian density was significantly higher at mixed points (3.49 birds/ha) and burn points (3.28 birds/ha) than unburned points (2.70 birds/ha; Tukey-Kramer test). Post-fire avian density was higher than pre-fire density at all three burn points and at both “mixed” points (Table 9.9).

Table 9.8. Comparison of species richness (total number of species detected w/in 125 m) at 10 unthinned and thinned points in mixed conifer forest by year, San Antonio Valley survey route, Valles Caldera National Preserve, Jemez CFLRP, 2012-2024.

| Year | Type | # Points | # Visits | # Species |
|---|---|---|---|---|
| 2012 | Unthinned | 3 | 9 | 27 |
| 2013* | Unthinned | 3 | 6 | 22 |
| 2014 | Unthinned | 3 | 9 | 18 |
| 2015 | Unthinned | 3 | 9 | 21 |
| 2016 | Unthinned | 3 | 9 | 20 |
| 2017 | Unthinned | 10 | 30 | 34 |
| 2018* | Unthinned | 10 | 20 | 32 |
| 2019 | Unthinned | 10 | 30 | 36 |
| 2020 | Unthinned | 8 | 24 | 33 |
| 2020 | Thinned | 2 | 6 | 23 |
| 2021 | Unthinned | 2 | 6 | 21 |
| 2021 | Thinned | 8 | 24 | 35 |
| 2022 | Unthinned | 2 | 6 | 21 |
| 2022 | Thinned | 8 | 24 | 36 |
| 2023 | Unthinned | 2 | 6 | 20 |
| 2023 | Thinned | 8 | 24 | 40 |
| 2024 | Unthinned | 2 | 6 | 19 |
| 2024 | Thinned | 8 | 24 | 39 |
| Totals | Unthinned | 10 | 170 | 53 |
| | Thinned | 8 | 102 | 49 |
| | All | 10 | 272 | 63 |

*Only 2 surveys were conducted at each point in 2013 and 2018 due to fire closures

During the eight years of data collection on the three North American Breeding Bird Survey (BBS) routes established in 2016, a total of 111 species were detected. Cumulatively, the most common species were Violet-green Swallow (7.74% of all detections), Western Tanager (7.29%), and American Robin (*Turdus migratorius*; 5.58%). On the Redondo Border route, the most common species were Violet-green Swallow (10.18% of all detections), Western Tanager (9.77%), and Western Wood-Pewee (6.03%). On Route 10, the most common species were Western Tanager (7.89% of all detections), Violet-green Swallow (6.75%), and Pygmy Nuthatch (6.62%). On the Valle Grande route, the most common species were Vesper Sparrow (9.32%), Common Raven (9.19% of all detections), and Chihuahuan Meadowlark (8.06%). The BBS routes were not surveyed in 2018 due to closures of the study area related to fire danger throughout the survey window.

Table 9.9. Comparison of mean avian density (# birds/ha) before and after fire at five points in mixed conifer forest, South Mountain survey route, Valles Caldera National Preserve, Jemez CFLRP, 2017-2024. At MIX points, less than 40% of the survey areas were within the fire perimeter.

| **Point & Habitat Type** | **# Surveys** | **# Birds/ha** |
|---|---|---|
| SM01 MC MIX | 18 | 3.99 |
| SM01 MC | 5 | 2.36 |
| SM03 MC Burn | 18 | 3.26 |
| SM03 MC | 5 | 1.79 |
| SM04 MC Burn | 18 | 3.39 |
| SM04 MC | 5 | 2.36 |
| SM05 MC Burn | 18 | 3.18 |
| SM05 MC | 5 | 1.83 |
| SM06 MC MIX | 18 | 3.00 |
| SM06 MC | 5 | 2.20 |

Table 9.10. Comparison of species richness (total number of species detected w/in 125 m) at 8 unburned and burned points in mixed conifer forest by year, South Mountain survey route, Valles Caldera National Preserve, Jemez CFLRP, 2017-2024. At MIX points, less than 40% of the survey areas were within the fire perimeter.

| Year | Habitat | # Points | # Visits | # Species |
|---|---|---|---|---|
| 2017 | MC | 8 | 24 | 28 |
| 2018 | MC | 8 | 16 | 35 |
| 2019 | MC | 3 | 9 | 23 |
| 2019 | MC MIX | 2 | 6 | 25 |
| 2019 | MC Burn | 3 | 9 | 25 |
| 2020 | MC | 3 | 9 | 31 |
| 2020 | MC MIX | 2 | 6 | 24 |
| 2020 | MC Burn | 3 | 9 | 26 |
| 2021 | MC | 3 | 9 | 30 |
| 2021 | MC MIX | 2 | 6 | 25 |
| 2021 | MC Burn | 3 | 9 | 29 |
| 2022 | MC | 3 | 9 | 26 |
| 2022 | MC MIX | 2 | 6 | 28 |
| 2022 | MC Burn | 3 | 9 | 25 |
| 2023 | MC | 3 | 9 | 23 |
| 2023 | MC MIX | 2 | 6 | 25 |
| 2023 | MC Burn | 3 | 9 | 36 |
| 2024 | MC | 3 | 9 | 31 |
| 2024 | MC MIX | 2 | 6 | 25 |
| 2024 | MC Burn | 3 | 9 | 34 |
| Totals | MC | 8 | 78 | 52 |
| | MC MIX | 2 | 36 | 43 |
| | MC Burn | 4 | 54 | 45 |
| | All | 8 | 184 | 59 |

*Only 2 surveys were conducted at each point in 2018 due to a fire closure

## Discussion

Among the more important outcomes of the Southwest Jemez CFLRP is an increased knowledge of avian trends at both the community and individual species levels and a better understanding of how fire and vegetation thinning affects birds. This knowledge is particularly important given the large number of species of conservation concern that occur within the study area. A number of these species are common within the habitats we surveyed, but have experienced steep declines in their regional population trends (e.g., Broad-tailed Hummingbird [*Selasphorus platycercus*], Chipping Sparrow [*Spizella passerina*], Mountain Chickadee, Townsend's Solitaire [*Myadestes townsendi*], Violet-green Swallow, Western Bluebird [*Sialia mexicana*], Vesper Sparrow) or are faced with substantial deterioration of habitat or other negative impacts related to climate change (e.g., Brown Creeper [*Certhia americana*]). Some species are facing both of these issues (e.g., Grace's Warbler, Mountain Bluebird [*Sialia currucoides*], Pygmy Nuthatch, Steller's Jay [*Cyanocitta stelleri*], Virginia's Warbler) (U.S. Geological Survey 2024, New Mexico Avian Conservation Partners 2024).

Cumulatively, avian density was significantly higher in unburned mixed conifer forest than burned mixed conifer forest. Mixed conifer forest supported the highest avian density during each of the 13 years of this study. Among 42 species of conservation concern, 23 were documented at higher densities in mixed conifer forest and 19 were at higher densities in burned mixed conifer. Most notably, Grace's Warbler density was significantly higher in unburned mixed conifer than burned mixed conifer, where it was rare (Tukey-Kramer test). The status of Grace's Warbler is particularly important because of its high priority as a species of conservation concern due to its range, limited to forests with a substantial pine component in the southwestern U.S. and the prevalence of this species within the study area (Stacier and Guzy 2020, U.S. Fish and Wildlife Service 2021, New Mexico Avian Conservation Partners 2024, Bison-M 2024). Among relatively common species of conservation concern, densities of Brown Creeper, Evening Grosbeak, Golden-crowned Kinglet (*Regulus satrapa*), Mountain Chickadee, Pygmy Nuthatch, Red-naped Sapsucker (*Sphyrapicus nuchalis*), Steller's Jay, Townsend's Solitaire, and Williamson's Sapsucker (*Sphyrapicus thyroideus*) also were significantly higher in unburned mixed conifer than burned mixed conifer. In contrast, American Three-toed Woodpecker (*Picoides dorsalis*), Chipping Sparrow, Green-tailed Towhee (*Pipilo chlorurus*), House Wren, Mountain Bluebird, Violet-green Swallow, and Western Bluebird densities were significantly higher in burned mixed conifer than unburned mixed conifer.

In contrast to mixed conifer forest, cumulative avian density in burned ponderosa pine forest was significantly higher than unburned ponderosa pine forest. But among 42 species of conservation concern, 23 were present in higher densities in unburned ponderosa pine forest and 19 were at higher densities in burned ponderosa pine forest. Cumulatively, Grace's Warbler density was moderately higher in unburned ponderosa pine forest than burned ponderosa pine forest, and significantly higher in unburned ponderosa pine forest than all other habitat types (Tukey-Kramer test). Mountain Chickadee, Pygmy Nuthatch, Steller's Jay, Gray Flycatcher (*Empidonax wrightii*), Black-throated Gray Warbler (*Setophaga nigrescens*), Townsend's Solitaire, and Virginia's Warbler were other species of concern that we documented at significantly higher densities in unburned ponderosa pine forest than burned ponderosa pine. Additional, relatively common species of concern documented at substantially higher (but not significantly different)

densities in unburned ponderosa pine than burned ponderosa pine included Brown Creeper and Golden-crowned Kinglet.

But species preferring more open habitat and many woodpeckers were more abundant in burned ponderosa pine. For example, we documented American Three-toed Woodpecker during all 13 years of surveys. To our knowledge, American Three-toed Woodpecker had not been documented breeding in the Valles Caldera National Preserve prior to our 2012 observations. This woodpecker is particularly attracted to disturbed and/or recently burned conifer forests with insect infested snags (Tremblay et al. 2020). Other woodpecker species exhibit a similar, but lower attraction to burned forest. American Three-toed Woodpecker density was significantly higher in burned ponderosa pine forest than unburned (Tukey-Kramer test). Other species of concern documented at significantly higher densities in burned ponderosa pine than unburned ponderosa pine included Broad-tailed Hummingbird, Chipping Sparrow, Green-tailed Towhee, House Wren, Mountain Bluebird, Olive-sided Flycatcher, Violet-green Swallow, and Western Bluebird. The stark contrast in the impact of fire on these species of conservation concern in both mixed conifer and ponderosa pine forest illustrates the complexities of the forest ecosystem and the challenges presented in attempting to manage the forests appropriately.

Although avian density was significantly higher in unburned mixed conifer forest than burned mixed conifer forest, burned ponderosa pine points supported a significantly higher avian density than unburned ponderosa pine points. The influx of bird species preferring more open habitat into burned areas somewhat offset the decrease in numbers of birds less tolerant of fire in mixed conifer plots and resulted in increased avian use in burned ponderosa pine plots. But it is important to note that we generally lacked the ability to reliably quantify different ways in which fire impacted burned points. Any point count plot that showed extensive evidence of fire was classified as burned. The intensity and amount of burn, however, varied widely among plots. Fire impact ranged from 100% consumption of woody vegetation at some plots to other plots where fire primarily stayed out of the canopy and most mature trees survived. In general, tree survival was higher in burned ponderosa pine plots than burned mixed conifer plots, primarily due to the resistance of ponderosa pine to fire. Avian use was substantially reduced at points where all vegetation was consumed compared to points where fire was largely excluded from the canopy. The gradients of fire impact at different points between these two extremes ran a wide spectrum and generally left detailed quantification beyond our scope of work.

We did, however, quantify some gradient of fire impact at the eight South Mountain points following the 2018 Hidden Valley fire. As explained above, points were classified as unburned mixed conifer, burned mixed conifer or, “mixed.” The vegetation and structure characteristics of “mixed” points were more similar to unburned points than burned points, with minimal evidence of fire impact. Fire impact at burned points was generally moderate with numerous surviving, mature trees. This likely explains why avian density was higher at the South Mountain burn points than other mixed conifer burn points in the study area where fire impact was much more extensive.

Among 28 species of conservation concern documented at South Mountain, 12 were most abundant in burn points, nine most abundant in unburned points, and seven most abundant in “mixed” points. Six species of conservation concern were documented at significantly higher

densities in a specific mixed conifer gradient. Mountain Chickadee and Golden-crowned Kinglet density was significantly higher at unburned points than burned points, but not significantly higher than "mixed" points. Chipping Sparrow was significantly more abundant at both "mixed" and unburned points than burned points. In contrast, American Three-toed Woodpecker density was significantly higher at burned points than "mixed" and unburned points, and Brown Creeper density was significantly higher in burned points than unburned points. House Wren density was significantly higher in burned points than "mixed" points and significantly lower at unburned points than the other two mixed conifer types.

Among the 25 most common species at South Mountain, 10 were most common at "mixed" points, 10 were most common at burned points, four were most common at unburned points, and one, Red-breasted Nuthatch (*Sitta canadensis*), was equally common at both unburned and burned points. In addition to the six species of concern mentioned above, seven other species were present at significantly higher densities in specific mixed conifer burn gradients. Western Tanager, Western Flycatcher (*Empidonax difficilis*), and Warbling Vireo (*Vireo gilvus*) densities were significantly higher in burned and "mixed" points than unburned points. Western Wood-Pewee was significantly higher in burned points than both "mixed" and unburned points, while Hermit Thrush (*Catharus guttatus*) was significantly higher in both "mixed" and unburned points than burned points. Black-headed Grosbeak (*Pheucticus melanocephalus*) density was significantly higher in burned points than unburned points, but not significantly higher than "mixed" points. Ruby-crowned Kinglet density was significantly higher in "mixed" points than burn points, but not significantly higher than unburned points. The wide variation in avian species abundance relative to mixed conifer burn gradient at South Mountain illustrates the complexity of avian habitat preferences and the importance of relatively subtle habitat variation to the avian community.

Cumulative data from mountain valley grassland plots indicated little difference in overall avian density between burned and unburned plots, but there were notable differences among regularly occurring individual species. The two species of conservation concern most dependent on grassland habitat, Savannah Sparrow and Vesper Sparrow, exhibited differing trends relative to preference of unburned vs. burned grassland. Vesper Sparrow was documented at a significantly higher cumulative density in unburned mountain valley grassland than burned mountain valley grassland. Savannah Sparrow density was significantly higher in burned grassland than unburned grassland. The response of Savannah Sparrow illustrates how quickly species may return to burned grassland habitat. In 2012, the first year after the Las Conchas fire, no Savannah Sparrows were documented in burned grassland. In contrast, during all years from 2013-2016, Savannah Sparrow density was higher in burned grassland than unburned grassland. Avian populations in burned grassland tend to stabilize within three or four years after a fire, likely in response to relatively rapid changes in food availability and other resources (Roberts et al. 2012). This is the scenario that played out on the grassland points. Avian density was higher in unburned grassland during 2012-2014, but was higher in burned grassland in 2015 and 2016. We did not survey the grassland points in the study area after 2016.

In a comparison of thinned and unthinned forest, data from the Banco Bonito route indicate that understory thinning in ponderosa pine forest may have a net benefit to the avian community. This is consistent with the findings in similar studies (e.g., Gaines et al. 2007, Kalies and

Rosenstock 2013). In 2020, we were able to begin comparing the impact of thinning at a total of eight ponderosa pine points on USFS survey routes (all six points at USFS Area 3 and two points at USFS Paliza South). In 2024, we surveyed eight thinned ponderosa pine points at USFS Paliza North. The sample sizes were very small for the USFS routes and the results were variable. Data from USFS Area 3 and USFS Paliza South data followed a similar trend to what was documented at Banco Bonito, with higher avian density post-thinning than pre-thinning. In contrast, avian density was higher pre-thinning than post thinning at USFS Paliza North.

Several important factors need to be considered relative to the benefits of thinning. First, the impact of prescribed fire at Banco Bonito was significant. In a meta-analysis of 22 studies in southwestern ponderosa pine and dry mixed conifer forests, Kalies et al. (2010) concluded that thinning and prescribed burning produced an overall benefit for passerine birds. Our data appear to support that conclusion, as the difference in cumulative avian density between thinned and unthinned points at Banco Bonito was largely due to data collected during the seven years after the use of prescribed fire at the thinned points. Cumulatively, avian density at thinned points was significantly higher after burning than before burning. The positive impact on overall bird numbers following prescribed fire at Banco Bonito mirrors the trend we documented of ponderosa pine forest burned during unintentional wildfires supporting a significantly higher avian density than unburned ponderosa pine forest.

Second, data from thinned ponderosa pine points on USFS land varied relative to the trend observed at Banco Bonito and the sample sizes were relatively small. Pre-thinning data are particularly limited at USFS Area 3, as 2019 was the only year surveys were conducted before thinning occurred (n = 18 surveys). Additionally, avian density across the entire study area as a whole was significantly lower during 2019 than all subsequent years. Thus, bird numbers at Area 3 in 2019 likely were suppressed relative to 2020-2024, at least in part due to the presence of moderate to exceptional drought conditions for the 15 months leading up to the 2019 breeding season (National Drought Mitigation Center 2024). Data from the two thinned ponderosa pine points at USFS Paliza South are both limited and largely inconclusive. Only 16 surveys were conducted pre-thinning and only 24 surveys were conducted post-thinning. Cumulative avian density from USFS Paliza South (1.88 birds/ha post-thinning vs 1.82 birds/ha pre-thinning) was very similar. In contrast, the pre-thinning data from USFS Paliza North was much more robust (n = 144 surveys), but post-thinning data was limited (n = 24). And, cumulative avian density was higher pre-thinning (1.74 birds/ha) than post-thinning (1.45 birds/ha). Results from small scale data sets are not reliable predictors of the longer-term response of birds to forest management treatments (Cahall et al. 2013), and long-term studies are necessary, in terms of both pre- and post-treatment data collection, to more completely understand the impacts of forest treatments on birds (Kalies et al. 2010). Although the pre- and post-thinning data set for Banco Bonito is relatively strong, the corresponding data sets for USFS Area 3 and USFS Paliza South are relatively weak.

Consideration also needs to be given to the impact of secondary plant species and forest stand composition on the avian community in ponderosa pine stands. For example, the presence of quaking aspen (*Populus tremuloides*) at Banco Bonito definitely impacted bird numbers. Aspen was present at all thinned points, but was virtually absent at unthinned points. Aspen is known to benefit birds both in terms of foraging and nesting opportunities. Invertebrate abundance in

aspen has been shown to be greater than invertebrate abundance in conifers (Schimpf and MacMahon 1985, Rumble and Anderson 1996), thus providing increased foraging opportunities for birds relative to ponderosa pine monocultures. Aspen also provides an important nesting substrate, especially for cavity nesting birds (Dobkin et al. 1995). We documented numerous active nests in aspen cavities in thinned points at Banco Bonito. Three species of conservation concern, House Wren, Western Bluebird, and Violet-green Swallow, were regularly documented using aspen cavities for nesting. Thus, the presence of aspen at thinned points and its absence at unthinned points was certainly a factor in these three species being significantly more abundant in thinned stands and a factor in the higher overall avian use of thinned stands.

Finally, the impact of treatment on specific bird species needs to be considered. Understanding the impact on species of conservation concern is particularly important. Cumulatively, among the 15 most common species of conservation concern at Banco Bonito, eight were present at higher densities at unthinned points (Pygmy Nuthatch, Mountain Chickadee, Townsend's Solitaire, Steller's Jay, Brown Creeper, Clark's Nutcracker, Golden-crowned Kinglet, and Evening Grosbeak) and seven were present at higher densities at thinned points (Grace's Warbler, Violet-green Swallow, Western Bluebird, Chipping Sparrow, Broad-tailed Hummingbird, House Wren, and Williamson's Sapsucker). Gaines et al. (2007) found similar results in terms of the impact differing among bird species following thinning in ponderosa pine forest.

But further breakdown of the Banco Bonito data revealed some interesting caveats. For example, although Mountain Chickadee was cumulatively more abundant at unthinned points, it was more abundant at thinned points before burning than unthinned points and was significantly more abundant at thinned points before burning than after burning. In contrast, both Broad-tailed Hummingbird and House Wren were significantly more abundant at thinned and burned points than both unthinned points and thinned points before burning; and both species were more abundant at unthinned points than thinned points before burning. Thus, from an avian perspective, the value of ponderosa pine thinning and the incorporation of controlled burning may depend on the species of interest. It is likely that a combination of treatments, including no treatment, conducted in patchy arrangements over time and area will result in higher overall avian diversity and benefit a greater number of species than any single treatment. A variety of studies have suggested this type of approach to forest management (e.g., Noss et al. 2006, Gaines et al. 2007, Kalies et al. 2010, Cahall et al. 2013, Akresh et al. 2023).

Analysis of pre- and post-thinning data at mixed conifer points on the San Antonio Valley route showed similar trends to those we documented at Banco Bonito. Avian density was higher at all eight treatment points after thinning, and the increase was significant at six of the eight points. But there are caveats with the comparison of pre- and post- thinning at San Antonio that make it a less reliable comparison than Banco Bonito. Only one of the eight treatment points was surveyed during 2012-2016, and that point consistently supported lower avian use both pre- and post-thinning. It supported the second-lowest avian density among the eight treatment points pre-thinning and the lowest avian density among the eight treatment points post-thinning. Because it was the only point surveyed during the first five years of the study, it accounts for 23% of all pre-thinning surveys. Additionally, although all eight treatment points were surveyed for three years before thinning, two of those years (2017 and 2019) supported the second and fifth lowest avian densities across the Jemez study area as a whole. Furthermore, the seven years of the study

with the lowest cumulative avian densities across the study area occurred before thinning began at San Antonio, while five of the six years with the highest cumulative avian densities across the study area occurred after thinning began. Thus, while the general trends in the San Antonio data are likely reliable, broader environmental factors probably impacted those trends and made them appear more extreme than would have been the case if both thinned and unthinned points were surveyed during all years, as was the case at Banco Bonito. Broader environmental factors impacting bird numbers were largely mitigated in the Banco Bonito data set, because both thinned and unthinned points were surveyed during all years.

Trend comparisons among species of conservation concern between Banco Bonito and San Antonio showed both similarities and differences. Similar to Banco Bonito, Mountain Chickadee, Brown Creeper and Golden-crowned Kinglet densities at San Antonio were higher at unthinned points, while House Wren, Violet-green Swallow, Chipping Sparrow, and Williamson's Sapsucker were at higher densities at thinned points. In contrast to Banco Bonito, Broad-tailed Hummingbird and Grace's Warbler densities at San Antonio were higher at unthinned points, while Steller's Jay, Pygmy Nuthatch, Evening Grosbeak, Clark's Nutcracker, and Townsend's Solitaire densities were higher at thinned points. The apparent differences in individual species' preferences for thinned or unthinned plots between Banco Bonito and San Antonio could be due to a variety of factors, including elevational differences between the two sites, differences in conifer species composition between ponderosa pine and mixed conifer forest, differences in ground cover, differences in graminoid and forb species composition, and differences in stand structure and relative stand densities between the two sites. Another possibility, because all pre-thin surveys at San Antonio occurred between 2012-2020 and all post-thin surveys occurred between 2020-2024, is that substantial changes in species density occurred over the last 13 years.

We documented widespread changes in individual bird species numbers over the course of this study, varying from statistically significant overall increases to significant overall decreases. Among 19 relatively common species of concern, we documented 10 species that showed annual fluctuations but generally stayed near the same density levels throughout the study: Broad-tailed Hummingbird, Brown Creeper, Clark's Nutcracker, Evening Grosbeak, Golden-crowned Kinglet, Grace's Warbler, Mountain Bluebird, Pygmy Nuthatch, Steller's Jay and Townsend's Solitaire. Three species, Chipping Sparrow, Western Bluebird, and Williamson's Sapsucker showed substantial increases, and four species, Green-tailed Towhee, Blue-gray Gnatcatcher (*Polioptila caerulea*), House Wren, and Yellow-breasted Chat showed statistically significant increases (Tukey-Kramer tests). In contrast, two species, Mountain Chickadee and Virginia's Warbler, showed significant decreases.

Among 15 of the most common species in the study area (excluding species of concern), most showed some change in density during the study. Black-headed Grosbeak density fluctuated, but generally remained consistent. Yellow-rumped Warbler had a significant decrease in density during the middle years of this study, but rebounded to numbers moderately lower than the beginning of the study over the last few years. Black-chinned Hummingbird, Plumbeous Vireo, Western Flycatcher, Western Tanager, and Warbling Vireo densities increased over the course of the study, while Dark-eyed Junco (*Junco hyemalis*), Hairy Woodpecker (*Dryobates villosus*), Hermit Thrush, Northern Flicker (*Colaptes auratus*), and Pine Siskin (*Spinus pinus*) decreased.

Hammond's Flycatcher (*Empidonax hammondii*) and Yellow Warbler (*Setophaga petechia*) had significant density increases, while American Robin had a significant decrease in density.

Cumulative avian density among years was significantly higher in 2023 and 2024 than all other years except for 2018 and 2020-2022, and density in 2014-2017 was significantly lower than all other years except for 2019. There likely were multiple contributing factors to the variation in density among years. But it does appear that the general trends are accurate. One observer conducted a majority of the surveys during all years from 2013-2016, and his detection rates showed steady decreases each year. It also appears that the significant increase in detection rates from 2017 to 2018 was real. All three surveyors who surveyed in both 2017 and 2018 had increased detection rates in 2018, suggesting bird numbers improved that year.

The cumulative impact of long-term drought also likely was a factor in fluctuating bird numbers. Drought can reduce abundance and species richness in avian communities (Albright et al. 2010a, 2010b, Cady et al. 2019, Roberts et al. 2021), with the negative impacts becoming increasingly pronounced during long-term drought and recovery taking a number of years (George et al. 1992, Bock and Bock 1999, Bennett et al. 2014). The four-year period from 2010-2014 was the 10th driest on record for Sandoval County (27.5 cm below normal; NOAA National Centers for Environmental Information 2024). This helps explain the steady decrease in overall avian density during the first five years (2012-2016) of this study. In contrast, the 7-month period prior to the 2015 breeding season (November 2014-May 2015) was the 12th wettest on record for Sandoval County (6.91 cm above average). Nevertheless, overall avian density in 2015 was the lowest of the first four project years. The short-term alleviation of drought conditions following a long period of drought is not sufficient to allow for the avian community to experience any immediate recovery. The precipitation level in the 7-month period prior to the 2016 breeding season was slightly drier than average (1.3 cm below normal), and overall avian density in 2016 was the lowest documented during this study. This provides evidence that avian recovery from long-term drought is a long-term process that requires more than a few months of wet conditions and a return to normal precipitation levels.

Further evidence of the long-term drought recovery process was provided in 2017, when avian density was the second lowest documented during this study, despite the fact that the entire study area was free of drought conditions from July of 2015 through the end of 2017 (National Drought Mitigation Center 2024). In contrast, the avian community clearly benefitted in 2018 from the improved conditions present during the previous two years, as avian density in 2018 was higher than all previous years of the study, despite the entire study area suffering through severe to exceptional drought conditions from January of 2018 through the end of the breeding season. But drought conditions only partially explain the variation in avian density among years, as the four years with the highest documented avian densities during this study all occurred between 2020 and 2024, and the study area was in a nearly perpetual state of drought from October 2019 through February 2023. Drought conditions subsided from early March through early July in 2023, but again covered the entire survey area from mid-July 2023 through July 2024.

The timing of surveys also impacted avian density among years. Cumulative avian density was significantly higher for surveys conducted during July (2.11 birds/ha) than surveys conducted in

June (2.00 birds/ha) and May (1.94 birds/ha) (Tukey-Kramer test). This likely was a key factor in the high density documented in 2018. The study area was closed throughout June of 2018 due to extreme fire danger, forcing the second and third rounds of surveys to be conducted in July and pushing the survey period later than other years. The late season surveys likely resulted in increased detections of females who were no longer sitting on nests and newly-fledged young, which likely contributed to the higher densities documented in 2018.

Another factor likely impacting avian density among years was variation in the points surveyed among years. For example, mixed conifer forest consistently supported the highest avian density throughout the study. But, a lower percentage of the surveys in 2014-2016 were conducted in mixed conifer (range = 3-14%) than in 2012, 2013, and 2017-2024 (range = 22-28%). Correspondingly, cumulative avian density was lower in 2014-2016 than all other years except 2017, while 2012, 2013, and 2018-2024 were the nine years with the highest cumulative avian density. Similarly, the percentage of surveys conducted in ponderosa pine, the forest type consistently supporting the lowest avian densities, was higher in 2014-2016 (range = 40-48%) than during other years (range = 27-39%). Additionally, mountain meadow, mountain valley grassland, and burned mountain valley grassland, the three habitat types consistently supporting the lowest avian densities, were only surveyed in 2012-2016. The addition of those open habitats drove down the cumulative avian densities in 2012-2016 relative to 2017-2024.

Finally, observer bias likely accounts for some differences within the dataset. Nineteen different observers have conducted surveys during the course of this study. Although all were well qualified, some bias due to variation in ability and protocol interpretation is almost inevitable. The observer with the second-highest cumulative detection rates was only involved with the project during 2012, 2013, and 2018; it is probably not a coincidence that we documented higher avian densities during those three years than all other years prior to 2020. Similarly, the observer with the third-highest detection rate conducted a majority of the surveys in 2020-2024, years when we documented four of the five highest annual detection rates. It is likely that a combination of all the above factors impacted avian density documentation among years.

Data collected during the 13 years of this study have revealed a number of interesting trends at both the avian community and individual species levels. Despite relatively persistent drought conditions, all four years with the highest annual avian densities during this study have occurred since 2020. The general absence of large, catastrophic fires in the survey area since 2013 has likely played a role in the increased bird numbers documented in recent years. The 2011 Las Conchas fire and 2013 Thompson Ridge fire directly impacted substantial portions of our survey area and probably played a key role in lower bird numbers during earlier years of this study. But a decade of successional recovery without widespread catastrophic disturbance has probably helped bird numbers recover from lows documented during several of the early post-fire years. Other than the 2022 Cerro Pelado fire, which directly impacted only one of our survey routes, large, catastrophic wildfires have been absent from the survey area since 2013.
Habitat structure and food resources change rapidly following a fire and those changes continue as burned areas undergo succession. Correspondingly, avian population dynamics continue to change as the number of post-fire years increases (Raphael et al. 1987, Smucker et al. 2005). The initial response of many bird species may be mediated either by short-term increases or decreases in food availability. For example, the availability of seeds from cones, early-successional forbs

and grasses, and easy access to invertebrates beneath the bark of fire-killed standing trees may increase post-fire, providing increased food availability for species that exploit such resources (Hutto 1995, Brawn et. al. 2001). In contrast, decreases in food availability, such as reductions in arthropod abundance (e.g., grasshoppers or moths) in burned grasslands or foraging opportunities within burned forest may negatively affect other species (Hobson and Schieck 1999).

After widespread, catastrophic wildfire, the number of species negatively affected likely exceeds the number that benefit, especially in areas where essentially all vegetation has burned. But at a smaller scale with less intense burn damage, such as that provided by prescribed burns, species that benefit from fire can be enhanced without a substantial negative impact on species that don't. Additionally, these smaller burns can help reduce the risk of catastrophic fire. The prescribed burning at Banco Bonito and the management of the Hidden Valley fire at South Mountain are examples of smaller scale burns that provided overall benefit to the avian community. Species of concern such as House Wren, Western Bluebird, and Violet-green Swallow appeared to benefit without a substantial negative impact on species more adverse to fire, such as Grace's Warbler, Golden-crowned Kinglet, and Mountain Chickadee. Thus, we think that the use of prescribed fire in carefully considered situations can be a valuable resource for land managers in helping to enhance habitat characteristics for certain avian species of concern.

In addition to food availability, architectural differences in habitat, such as reduced or increased litter, changes in standing dead trees, nest site availability, vegetation regeneration, and predator abundance are among the key components regulating avian abundance and richness (MacArthur and MacArthur 1961, Raphael et al. 1987, Hobson and Schieck 1999). These factors impact different bird species in different ways, and characteristics that are beneficial to some species are detrimental to others. The careful use of thinning projects can increase the diversity of habitat structure and other factors important to birds, including nest site availability. But their use must be limited and accompanied by consideration for species that prefer denser forest stands.

The contrasts in avian response to both fire and habitat thinning during this project illustrate the complexity of the habitat types monitored during this study and the challenges of appropriate management. The use of thinning and prescribed fire can benefit avian communities. But, avian response to thinning and fire varies widely between species. Thus, careful consideration needs to be given to the needs of all species, especially species of concern. A combination of treatments, including prescribed fire, thinning, and areas of no treatment, conducted in patchy arrangements over time and area are likely to provide the greatest benefit to overall avian diversity and benefit the greatest number of species.

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Appendix 9.1. Alphabetical list and total numbers of 141 bird species detected within 125 m of point count locations, Santa Fe National Forest (USFS) and Valles Caldera National Preserve (VCNP), Jemez CFLRP, 2012-2024. Species with an asterisk (*) were detected only as flyovers. Species of conservation concern are listed in bold print.

| Species | USFS | VCNP | Total Detections |
|---|---|---|---|
| American Crow | 35 | 56 | 91 |
| **American Goshawk** | 3 | 5 | 8 |
| American Kestrel | 5 | 63 | 68 |
| **American Pipit*** | 0 | 0 | 0 |
| American Robin | 588 | 1074 | 1662 |
| **American Three-toed Woodpecker** | 16 | 74 | 90 |
| Ash-throated Flycatcher | 439 | 7 | 446 |
| **Band-tailed Pigeon** | 12 | 27 | 39 |
| Barn Swallow | 1 | 0 | 1 |
| Bewick's Wren | 16 | 8 | 24 |
| Black Phoebe | 23 | 0 | 23 |
| Black-billed Magpie | 0 | 10 | 10 |
| **Black-capped Chickadee** | 18 | 0 | 18 |
| Black-chinned Hummingbird | 452 | 8 | 460 |
| Black-headed Grosbeak | 274 | 195 | 469 |
| **Black-throated Gray Warbler** | 35 | 5 | 40 |
| Blue Grosbeak | 293 | 1 | 294 |
| Blue-gray Gnatcatcher | 343 | 1 | 344 |
| Brewer's Blackbird | 1 | 77 | 78 |
| **Brewer's Sparrow** | 2 | 6 | 8 |
| **Broad-tailed Hummingbird** | 285 | 265 | 550 |
| **Brown Creeper** | 133 | 83 | 216 |
| Brown-headed Cowbird | 59 | 10 | 69 |
| Bullock's Oriole | 14 | 0 | 14 |
| **Bushtit** | 239 | 0 | 239 |
| Calliope Hummingbird | 2 | 0 | 2 |
| **Canada Jay** | 0 | 8 | 8 |
| Canyon Wren | 7 | 1 | 8 |
| **Cassin's Finch** | 23 | 29 | 52 |
| Cassin's Kingbird | 39 | 2 | 41 |
| Cedar Waxwing | 12 | 0 | 12 |
| Chihuahuan Meadowlark | 0 | 180 | 180 |
| **Chipping Sparrow** | 619 | 1190 | 1809 |
| **Clark's Nutcracker** | 91 | 129 | 220 |
| Cliff Swallow | 1 | 0 | 1 |
| **Common Nighthawk** | 10 | 8 | 18 |

Appendix 9.1 continued.

| Species | USFS | VC | Total Detections |
|---|---|---|---|
| Common Raven | 114 | 192 | 306 |
| Common Yellowthroat | 34 | 0 | 34 |
| Cooper's Hawk | 12 | 3 | 15 |
| Curve-billed Thrasher | 1 | 0 | 1 |
| Dark-eyed Junco | 582 | 798 | 1380 |
| Downy Woodpecker | 46 | 27 | 73 |
| Dusky Flycatcher | 161 | 75 | 236 |
| **Dusky Grouse** | 1 | 11 | 12 |
| **Eastern Bluebird** | 3 | 0 | 3 |
| *Empidonax* spp. | 7 | 11 | 18 |
| Eurasian Collared-Dove | 4 | 0 | 4 |
| **Evening Grosbeak** | 102 | 140 | 242 |
| **Flammulated Owl** | 0 | 2 | 2 |
| Gambel's Quail | 26 | 0 | 26 |
| **Golden-crowned Kinglet** | 46 | 77 | 123 |
| **Grace's Warbler** | 808 | 681 | 1489 |
| **Gray Catbird** | 1 | 0 | 1 |
| **Gray Flycatcher** | 159 | 0 | 159 |
| **Gray Vireo** | 8 | 0 | 8 |
| Great Blue Heron | 5 | 0 | 5 |
| Great Horned Owl | 2 | 13 | 15 |
| Greater Roadrunner | 1 | 0 | 1 |
| **Green-tailed Towhee** | 25 | 92 | 117 |
| Hairy Woodpecker | 415 | 506 | 921 |
| Hammond's Flycatcher | 338 | 360 | 698 |
| Hepatic Tanager | 14 | 4 | 18 |
| Hermit Thrush | 542 | 240 | 782 |
| Hooded Warbler | 2 | 0 | 2 |
| Horned Lark | 1 | 78 | 79 |
| House Finch | 290 | 0 | 290 |
| House Sparrow | 4 | 0 | 4 |
| **House Wren** | 215 | 745 | 960 |
| Indigo Bunting | 0 | 1 | 1 |
| **Juniper Titmouse** | 25 | 0 | 25 |
| Killdeer* | 0 | 0 | 0 |
| Ladder-backed Woodpecker | 21 | 0 | 21 |
| Lark Sparrow | 5 | 9 | 14 |
| **Lazuli Bunting** | 5 | 3 | 8 |
| Lesser Goldfinch | 423 | 36 | 459 |

Appendix 9.1 continued.

| Species | USFS | VC | Total Detections |
|---|---|---|---|
| **Lewis's Woodpecker** | 0 | 19 | 19 |
| **Lincoln's Sparrow** | 6 | 12 | 18 |
| **Lucy's Warbler** | 5 | 0 | 5 |
| **MacGillivray's Warbler** | 81 | 25 | 106 |
| Mallard | 11 | 14 | 25 |
| **Marsh Wren** | 0 | 2 | 2 |
| **Mountain Bluebird** | 10 | 229 | 239 |
| **Mountain Chickadee** | 879 | 904 | 1783 |
| Mourning Dove | 221 | 96 | 317 |
| Nashville Warbler | 3 | 9 | 12 |
| Northern Flicker | 248 | 643 | 891 |
| Northern Mockingbird | 11 | 9 | 20 |
| **Northern Pygmy-Owl** | 1 | 7 | 8 |
| Northern Rough-winged Swallow | 13 | 5 | 18 |
| **Olive-sided Flycatcher** | 12 | 31 | 43 |
| Orange-crowned Warbler | 141 | 185 | 326 |
| **Peregrine Falcon** | 9 | 0 | 9 |
| Pine Siskin | 273 | 561 | 834 |
| **Pinyon Jay** | 7 | 0 | 7 |
| Plumbeous Vireo | 579 | 409 | 988 |
| **Purple Martin** | 0 | 3 | 3 |
| **Pygmy Nuthatch** | 1257 | 1120 | 2377 |
| Red Crossbill | 87 | 119 | 206 |
| Red-breasted Nuthatch | 78 | 222 | 300 |
| **Red-naped Sapsucker** | 17 | 39 | 56 |
| Red-tailed Hawk | 2 | 16 | 18 |
| Red-winged Blackbird | 2 | 2 | 4 |
| **Rock Wren** | 2 | 8 | 10 |
| Ruby-crowned Kinglet | 356 | 867 | 1223 |
| Rufous Hummingbird | 4 | 23 | 27 |
| **Savannah Sparrow** | 0 | 71 | 71 |
| Say's Phoebe | 6 | 0 | 6 |
| **Scaled Quail** | 2 | 0 | 2 |
| Sharp-shinned Hawk | 2 | 3 | 5 |
| Spotted Sandpiper | 1 | 0 | 1 |
| Spotted Towhee | 627 | 58 | 685 |
| **Steller's Jay** | 494 | 528 | 1022 |
| Summer Tanager | 118 | 0 | 118 |
| Swainson's Hawk | 0 | 2 | 2 |
| **Townsend's Solitaire** | 277 | 286 | 563 |

Appendix 9.1 continued.

| Species | USFS | VC | Total Detections |
|---|---|---|---|
| Townsend's Warbler | 3 | 10 | 13 |
| **Tree Swallow** | 0 | 3 | 3 |
| Turkey Vulture | 14 | 5 | 19 |
| **Vesper Sparrow** | 7 | 373 | 380 |
| **Violet-green Swallow** | 369 | 1051 | 1420 |
| **Virginia's Warbler** | 148 | 48 | 196 |
| Warbling Vireo | 918 | 594 | 1512 |
| **Western Bluebird** | 223 | 586 | 809 |
| Western Flycatcher | 409 | 472 | 881 |
| Western Kingbird | 8 | 0 | 8 |
| Western Meadowlark | 0 | 260 | 260 |
| Western Tanager | 1447 | 1534 | 2981 |
| Western Wood-Pewee | 741 | 784 | 1525 |
| White-breasted Nuthatch | 548 | 501 | 1049 |
| White-crowned Sparrow | 0 | 6 | 6 |
| White-eyed Vireo | 1 | 0 | 1 |
| **White-throated Swift** | 1 | 0 | 1 |
| White-winged Dove | 72 | 1 | 73 |
| Wild Turkey | 3 | 21 | 24 |
| **Williamson's Sapsucker** | 79 | 188 | 267 |
| Willow Flycatcher | 1 | 0 | 1 |
| **Wilson's Warbler** | 39 | 4 | 43 |
| **Woodhouse's Scrub-Jay** | 44 | 3 | 47 |
| Yellow Warbler | 440 | 3 | 443 |
| Yellow-bellied Sapsucker | 1 | 0 | 1 |
| **Yellow-breasted Chat** | 975 | 0 | 975 |
| Yellow-rumped Warbler | 1146 | 1353 | 2499 |
| unknown spp. | 5 | 1 | 6 |
| **Totals** | **22022** | **21954** | **43976** |

Appendix 9.2. Alphabetical list of 57 avian species of conservation concern documented during point count surveys, 2012-2024, Jemez CFLRP, Santa Fe National Forest and Valles Caldera National Preserve. Conservation concern status is based on ratings by the U.S. Fish and Wildlife Service (FWS), U.S. Forest Service (USFS), New Mexico Avian Conservation Partners (NMACP), New Mexico Department of Game and Fish (NMGF), the State of New Mexico (NM), and Natural Heritage New Mexico (NHNM). For FWS, BCC = birds of conservation concern. For USFS, SCC = Region 3 Species of Conservation Concern and SS = Region 3 Sensitive Species. For NMACP, species of concern are assigned to a species conservation list (SC1 or SC2) or a biodiversity conservation list (BC1 or BC2). SC lists contain species for which New Mexico has a high stewardship responsibility, while BC lists contain species for which New Mexico does not have a high stewardship responsibility, but that are important for the state's biodiversity. For both lists, level 1 species are of higher concern than level 2 species. For NM, T = State Threatened Species. For NMGF, SGCN = Species of Greatest Conservation Need. For NHNM, S1 = critically imperiled, S2 = imperiled, and S3 = vulnerable.

| Species | Conservation Status |
|---|---|
| American Goshawk | USFS - SCC; NMACP - BC1; NHNM - S2 |
| American Pipit | NMACP - BC1; NHNM - S3 |
| American Three-toed Woodpecker | NHNM - S3 |
| Band-tailed Pigeon | NMACP - SC2 |
| Black-capped Chickadee | NHNM - S3 |
| Black-throated Gray Warbler | NMACP - SC2; NMGF - SGCN; NHNM - S3 |
| Brewer's Sparrow | NMACP - SC2; NHNM - S3 |
| Broad-tailed Hummingbird | FWS - BCC; NMACP - SC2 |
| Brown Creeper | NMACP - BC1 |
| Bushtit | NMACP - SC2 |
| Canada Jay | NMACP - BC1 |
| Cassin's Finch | FWS - BCC; NMACP - BC2; NMGF - SGCN; NHNM - S3 |
| Chipping Sparrow | NMACP - BC2 |
| Clark's Nutcracker | FWS - BCC; NMACP - SC2; NMGF - SGCN |
| Common Nighthawk | NMGF - SGCN |
| Dusky Grouse | NMACP - BC2; NHNM - S3 |
| Eastern Bluebird | NHNM - S1 |
| Evening Grosbeak | FWS - BCC; NMACP - SC2; NMGF - SGCN |
| Flammulated Owl | FWS - BCC; USFS - SS; NMACP - SC1; NMGF - SGCN; NHNM - S3 |
| Golden-crowned Kinglet | NMACP - BC1 |
| Grace's Warbler | FWS - BCC; USFS - SCC; NMACP - SC1; NMGF- SGCN; NHNM - S3 |
| Gray Catbird | USFS - SS; NHNM - S3 |
| Gray Flycatcher | NHNM - S3 |
| Gray Vireo | NM - T; USFS - SCC; NMACP - SC2; NMGF - SGCN |
| Green-tailed Towhee | NMACP - SC2; NHNM - S3 |
| House Wren | NMACP - BC1; NHNM - S3 |

Appendix 9.2 continued.

| Species | Conservation Status |
|---|---|
| Juniper Titmouse | NMACP - SC1; NMGF - SGCN; USFS - SS |
| Lazuli Bunting | NMACP - BC2 |
| Lewis's Woodpecker | FWS - BCC; USFS - SCC; NMACP - SC1; NMGF- SGCN; NHNM - S3 |
| Lincoln's Sparrow | USFS - SCC; NHNM - S2 |
| Lucy's Warbler | NMACP - SC2; NMGF - SGCN; NHNM - S3 |
| MacGillivray's Warbler | NMACP - BC2 |
| Marsh Wren | NHNM - S1 |
| Mountain Bluebird | NMACP - SC2; NMGF - SGCN |
| Mountain Chickadee | NMACP - SC2; NHNM - S3 |
| Northern Pygmy-Owl | NMACP - BC1 |
| Olive-sided Flycatcher | FWS - BCC; NMACP - BC1; NMGF - SGCN; NHNM - S3 |
| Peregrine Falcon | NM - T; USFS - SCC; NMGF - SGCN; NHNM - S2 |
| Pinyon Jay | FWS - BCC; USFS - SCC; NMACP - SC1; NMGF - SGCN; NHNM - S2 |
| Purple Martin | NHNM - S3 |
| Pygmy Nuthatch | NMACP - SC2; NMGF - SGCN; NHNM - S3 |
| Red-naped Sapsucker | NHNM - S3 |
| Rock Wren | NMACP - SC2 |
| Savannah Sparrow | NHNM - S2 |
| Scaled Quail | NMACP - SC1, NHNM - S3 |
| Steller's Jay | NMACP - SC2 |
| Townsend's Solitaire | NMACP - SC2 |
| Tree Swallow | NHNM - S3 |
| Vesper Sparrow | NMACP - SC2; NMGF - SGCN |
| Violet-green Swallow | NMACP - SC2; NHNM - S3 |
| Virginia's Warbler | FWS - BCC; NMACP - SC1; NMGF - SGCN; NHNM - S3 |
| Western Bluebird | NMACP - SC2; NMGF - SGCN |
| White-throated Swift | NHNM - S3 |
| Williamson's Sapsucker | NMACP - SC2; NMGF - SGCN |
| Wilson's Warbler | USFS - SCC; NMACP - BC1; NHNM - S1 |
| Woodhouse's Scrub-Jay | NMACP - SC1; NHNM - S3 |
| Yellow-breasted Chat | NHNM - S3 |

# Chapter 10. Wildlife Monitoring – Terrestrial Invertebrates

## Monitoring project title: Effects of restoration and fire on pest and beneficial insects and other arthropods.

**Collaborators:** Robert R. Parmenter, Mark Ward, Richard Buss – *National Park Service, Valles Caldera National Preserve;* Sandra Brantley, David Lightfoot – *University of New Mexico;* Michael Gates, John Brown, Michael Pogue – *USDA Systematic Laboratory and Smithsonian Institution*; Eileen Brantley, Arden Jones – *University of the South*; Jonathan Knudsen, Robin Verble – *Texas Tech University*; Andrew Graves, Tom Coleman – *US Forest Service*.

**Summary:** Ecologists have long understood that "invertebrates rule the world" due to their incredible diversity, abundance and world-wide distribution, along with the importance of their ecosystem roles in plant herbivory, decomposition, nutrient cycling, predator-prey interactions, parasitism, and disease. As part of the SWJM CFLRP monitoring, we undertook monitoring of common groups of invertebrate taxa that typically include "pest and beneficial" insects, along with other important taxa such as spiders. We focused on three phenomena: (1) effects of the high-severity wildfire on common species of beetles, grasshoppers, ants, spiders and moths; (2) forest thinning and prescribed fire impacts on ground-dwelling arthropods; and (3) post-fire impact of the Douglas fir beetle on old-growth trees in the History Grove on Valles Caldera. Results showed that wildfire and restoration actions shifted arthropod species composition from forest obligates to grassland meadow species, although considerable overlap among communities continued to persist. Douglas fir beetle populations expanded following wildfires, and resulted in widespread mortalities of old-growth Douglas firs; young Douglas firs were not significantly affected. Additional monitoring of soil invertebrates is reported in Chapter 11.

## Section 10.1. High-severity wildfire effects on forest invertebrates.

The 2011 Las Conchas Fire, ignited by a tree falling on a powerline on private property, burned a total of 63,371 ha of forests and grasslands, including 12,156 ha within Valles Caldera National Preserve (Figure 10.1.1); at the time, the Las Conchas fire was the largest wildfire ever recorded in the state of New Mexico. On the Preserve, the fire created a landscape mosaic of burned forest stands, with moderate- to high-severity stand-replacement fires occurring in 3,546 ha of mixed conifer forest and 643 ha of ponderosa pine forests. Following the fire in 2011, herbaceous vegetation (grasses and forbs) became established, and over the next 5 years (2011-2015), increased rapidly to eventually attain 100% plant cover; this effectively converted the previous forested ecosystem, which sustained stand-replacement fire, into a montane meadow/grassland ecosystem (Figure 10.1.2).

Immediately following the fire, we established replicated monitoring plots in burned and un-burned mixed conifer forests, ponderosa pine forests and montane grasslands (Figure 10.1.3), and sampled arthropods using a variety of trapping methods: pitfall traps for ground-dwelling species, malaise traps for flying insects, and UV light traps for nocturnal moth species. While analytical work continues on these samples and data sets, some taxa have been analyzed and are presented below. Potential responses of arthropods to fire impacts are summarized in Table 10.1.1.

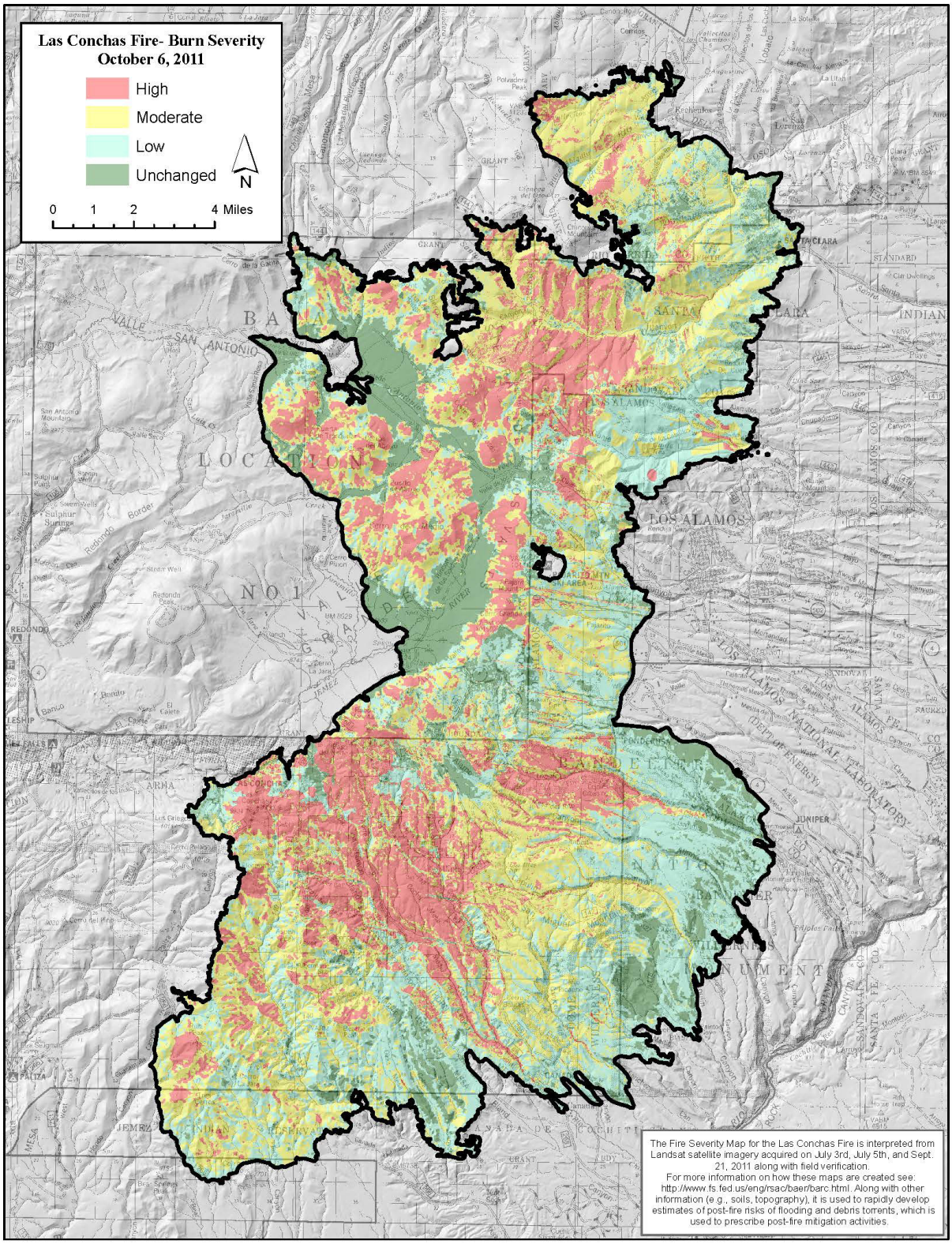



Figure 10.1.1. Fire severity map of the Las Conchas Fire, summer 2011.

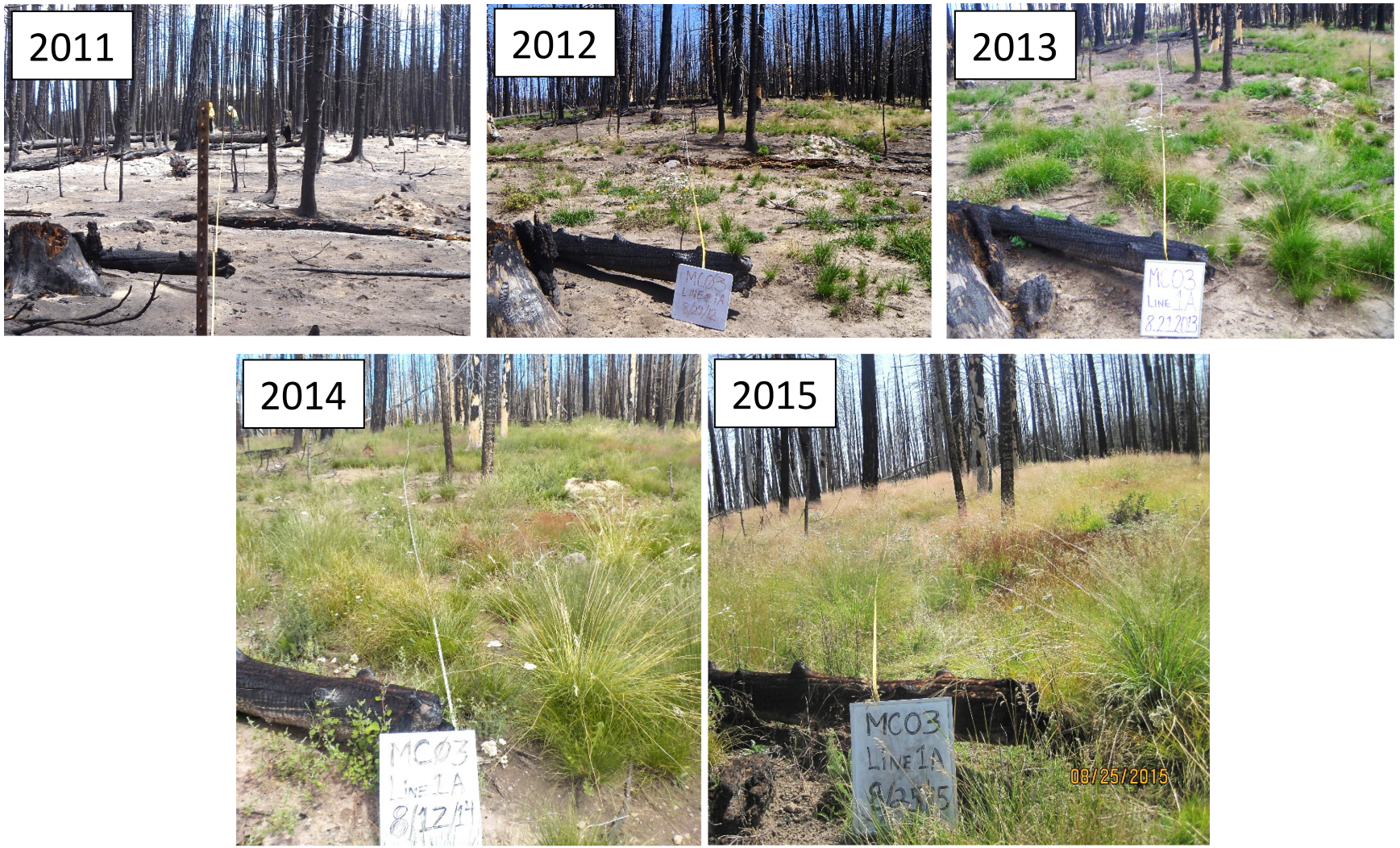



Figure 10.1.2. Repeat photographs of vegetation sampling transect in mixed conifer forest following the Las Conchas Fire, 2011-2015. Herbaceous vegetation achieved 100% cover in 2015.

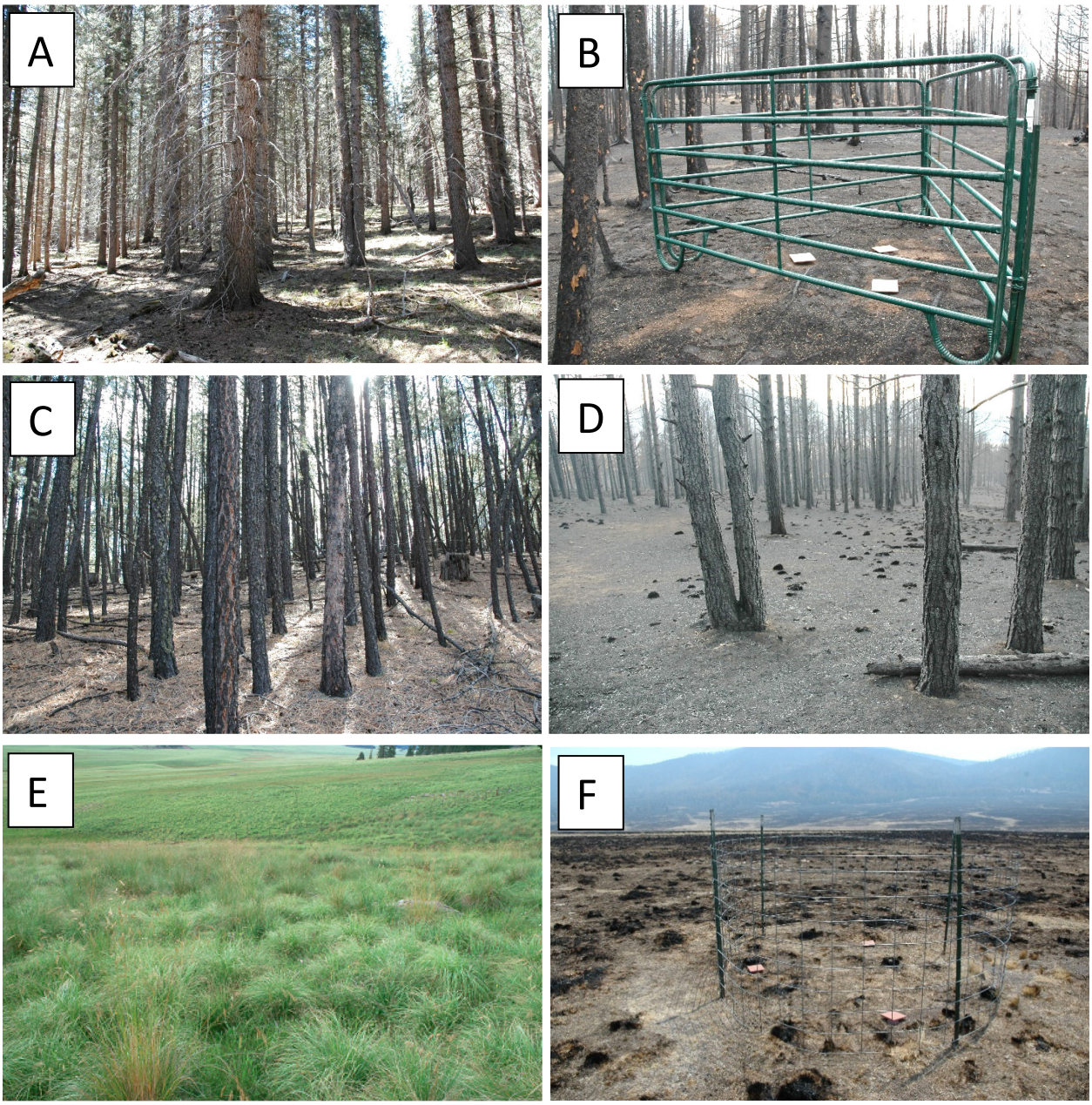



Figure 10.1.3. Photos of arthropod monitoring sites following the Las Conchas Fire in 2011-2015: (A) Control mixed conifer forest; (B) burned mixed conifer forest; (C) control ponderosa pine forest; (D) burned ponderosa pine forest; (E) control grassland; (F) burned grassland. Photos B and F show pitfall traps inside elk exclosure fences.

Table 10.1.1. Matrix of positive and negative impacts of forest fires on arthropods during immediate and short-term post-fire time periods.

| | Negative Response | Positive Response |
|---|---|---|
| **Immediate (1-3 months)** | Mortality (or rapid outmigration) | Rapid in migration (or increase in activity) |
| **Delayed (1-3+ years)** | Changes that reduce long term success | Successful occupation of new habitats |

*Grasshoppers and crickets (Orthoptera)*. From samples collected during 2011-2015, we identified 18 species of grasshoppers, 2 species of camel crickets, and one katydid (the Mormon cricket) in ponderosa pine forests (burned and unburned), and 20 species in mixed conifer forests. Crickets dominated the unburned forest sites, and thus could be considered "indicator species" of undisturbed forest arthropod assemblages (Table 10.1.2). In contrast, grasshoppers began to colonize the newly-burned forest sites, and increased in numbers and species as herbaceous vegetation (grasses and forbs) became established on the burned sites (Figure 10.1.2). By 2015, plant cover attained 100% on the burned forest sites, and grasshoppers were abundant.

Table 10.1.2. List of grasshoppers, crickets and katydids (Orthoptera) considered to be "indicator species" for burned (B) and unburned (U) ponderosa pine forests following the Las Conchas Fire during the monitoring program in 2011-2015.

**Results of Indicator Species Analysis by Year for Select Orthoptera, Ponderosa Pine Sites, 2011-2105. Indicator species: p<.05**

| Family | Species | 2011 | 2012 | 2013 | 2014 | 2015 |
|---|---|---|---|---|---|---|
| Katydids (Tettigoniidae) | *Anabrus simplex* | | B | | B | B |
| Grasshoppers (Acrididae) | *Aeropodellus clavatus* | | | B | B | |
| | *Melanoplus femurnigrum* | | | B | B | |
| | *Trimerotropis suffusa* | | B | B | | |
| | *Melanoplus bruneri* | | | | B | |
| | *Camnula pellucida* | | | B | | |
| Camel Crickets (Rhaphidophoridae) | *Styracosceles neomexicanus* | U | U | | U | U |







*Beetles (Coleoptera).* During 2011-2015, we collected a total of 124 species of beetles in ponderosa pine forests and 130 beetle species in mixed conifer forests. Nearly all of the resident beetles would have been killed by the wildfire, but most beetles are good fliers and recolonized

the burned forest sites fairly quickly. Some species were more abundant in the burned forests, and could therefore be classified as "indicator species" for post-fire insect communities; others were predominantly found in unburned forests, and were therefore indicators of undisturbed forest floor communities. Table 10.1.3 below lists 6 species that were found more often in burned ponderosa pine forests than in unburned, along with three species indicative of undisturbed forests. Overall, the species composition in the post-fire forests was considerably different from unburned forests, and displayed signs of moving towards a community dominated by grassland species (similar to the results for grasshoppers discussed above).

Table 10.1.3. List of beetles (Coleoptera) considered to be "indicator species" for burned and unburned ponderosa pine forests following the Las Conchas Fire in 2011.

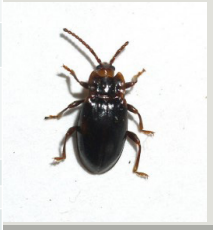


**Results of Indicator Species Analysis by Year for Beetles, Ponderosa Pine Sites,2011-2015. Indicator species: p<.05**

| Family | Species | 2011 | 2012 | 2013 | 2014 | 2015 |
|---|---|---|---|---|---|---|
| Carabidae | *Apristus sp.* | | B | B | B | B |
| Ground beetles (Carabidae) | *Agonum placidum*<br>*Amara quenseli*<br>*Bembidion mutatum*<br>*Sericoda bembidioides* | | | B | B | B |
| Ground beetles (Carabidae) | *Agonum cupreum*<br>*Agonum cupripenne*<br>*Sericoda quadripunctata*<br>*Amara ellipsis*<br>*Pterostichus adstrictus*<br>*Harpalus fraternus*<br>*Dicheiotrichus cognatus* | | | | B | B |
| Ground beetles (Carabidae) | *Amara littoralis*<br>*Amara lunicollis*<br>*Bembidion sp. 5* | | | | | B |
| Carabidae<br>Weevils | *Rhadine sp.*<br>*Pseudorimus orbicollis* | | B | B | B | |
| Carabidae<br>Tenebrionidae<br>Anthicidae<br>Trachypachidae | *Amara obesa*<br>*Blapstinus sp.*<br>*Anthicus hastatus*<br>*Trachypachus inermis* | | | B | B | |
| Endomychidae | *Aphorista morosa* | U | | U | U | U |
| Elateridae | *Pseudanostirus pudicus* | | U | U | U | U |
| Tenebrionidae | *Eleodes nigrinus* | U | U | U | U | U |







*Moths (Lepidoptera)*.

We assessed forest-dwelling macrolepidopteran moth communities in mixed conifer and ponderosa pine forests during 2012, the first year after the 2011 Las Conchas fire (see Brantley et al. 2023). We deployed blacklight traps in replicated burned and unburned forest stands during June, July, and August in 2012. In total, we collected 9,478 individuals representing 211 species. Noctuidae (124 species) and Geometridae (53) were the best-represented families, followed by Erebidae (21), Sphingidae (5), Notodontidae (3), Lasiocampidae (2), Saturniidae (2), and Drepanidae (1). Ponderosa pine forests supported 161 moth species, and mixed conifer forests supported 169 species. Moth communities (species composition and abundances) in each forest type (mixed conifer vs. ponderosa pine) were statistically distinguishable, but shared 56.4% (119) of observed species. Overall, compared to unburned forests, post-fire moth communities in both forest types had significantly lower numbers of individuals, species richness and diversity, and lower evenness in ponderosa pine forests. As expected, categorizing moth taxa by larval host

plant taxa revealed that reductions of moth populations following fire were associated with the elimination or reduction of available larval host plants (particularly conifers, oaks, and junipers); see Figure 10.1.4. Future moth community succession will likely parallel the overall transformation from a forested landscape to a montane meadow/grassland ecosystem, with continued reduction in tree-feeding species and increasing dominance by forb/grass-feeding species.

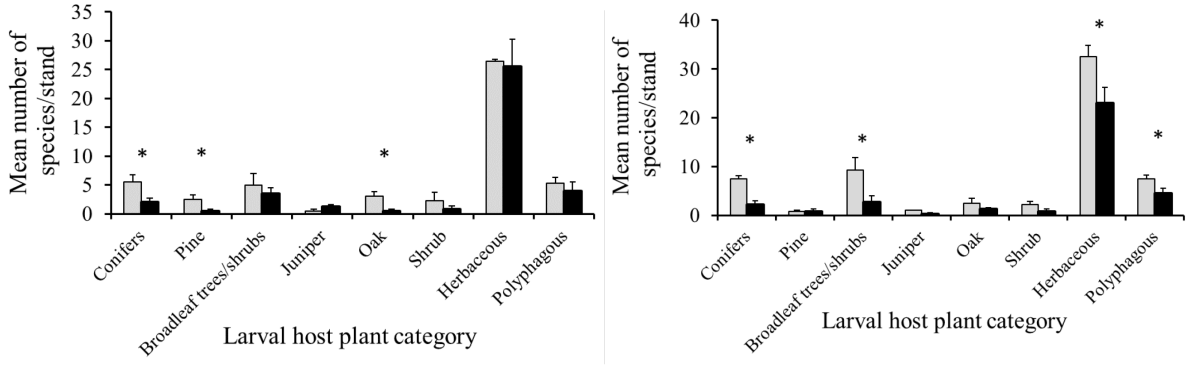



Figure 10.1.4. Differences in moth numbers of species among host plant categories in burned (solid black histograms) and unburned (grey histograms) stands of (left) ponderosa pine forests and (right) mixed conifer forests on Valles Caldera National Preserve. Friedman's ANOVA: * = $P < 0.05$. *Source:* Brantley et al. 2023.

*Ants (Hymenoptera)*. Ants are indicators of ecosystem health and function that are sensitive to disturbance and fill a variety of roles in their ecosystems, including altering soil chemistry, dispersing seeds, and serving as a key food resource for many species, including the federally endangered Jemez Mountain salamander (*Plethodon neomexicanus*). We examined the post-fire effects of the 2011 Las Conchas Wildfire on ant communities in Valles Caldera National Preserve. We collected ants via pitfall traps in replicated burned and unburned sites across three habitats: ponderosa pine forests, mixed-conifer forests, and montane grassland. We analyzed trends in species richness, abundance, recruitment, loss, turnover and composition over five sequential years of post-fire succession (2011-2015). We collected 66,203 ants from 2011-2015, comprising 60 species representing 16 genera (Knudsen et al., in review). Ant community structure was influenced by burn presence, season of sampling, and macrohabitat. We also found strong seasonal trends and decreases over time since fire in ant species richness and ant abundance. However, habitat and seasonal effects were a stronger predictor of ant species richness than presence of fire or post-fire successional patterns.

*Spiders (Arachnida).* Ground-dwelling arachnids were collected in pitfall traps in burned and unburned habitats from 2011–2015. Spider taxa used in analyses of the monitoring effort were: 64 species in 29 genera and 10 families for ponderosa pine forest species; 69 species in 34 genera in 10 families for mixed conifer forests; and 58 species in 37 genera in 10 families for montane grasslands. The common co-dominant spider families in temperate areas, wolf spiders (Lycosidae) and ground spiders (Gnaphosidae), along with harvestmen and the erythraeid mites, made up about 60% of the abundance. Permutational multivariate analysis of variance

PERMANOVA), mostly at the genus level with some higher taxon levels, showed significant fire, year, and interaction effects (see S. Brantley 2020). Abundance was at or near unburned levels by 2014, but species composition changed in burned areas. Spiders in the genera *Pardosa* and *Haplodrassus* were dominant genera across habitats, and provided species that were indicators of fire severity (Table 10.1.4). Linyphiidae spiders were also strong indicators of unburned sites. Harvestmen were among the dominant species in the forest habitats, and erythraeid mites were abundant in the burned ponderosa pine forest and the grassland. Years were not significantly autocorrelated, unsurprising given the interannual variation in precipitation in this generally arid region. Although fire is a common feature of these habitats, future fires may be outside of historical patterns, preventing spider communities from re-establishing fully.

Table 10.1.4. Lists of indicator spiders common in burned (B) and unburned (U) forest sites following the Las Conchas Fire on Valles Caldera National Preserve.

Selected arachnid indicator species p=0.001



| Wolf spiders Genus *Pardosa* | 2011 | 2012 | 2013 | 2014 | 2015 |
|---|---|---|---|---|---|
| *P. coloradensis* | | | B | B | B |
| *P. distincta* | | | B | B | B |
| *P. xerophila* | | | B | B | B |
| *P. concinna* | | B | B | B | B |
| *P. montgomeryi* | U | | B | B | B |
| *P. uncata* | U | U | U | U | U |
| *P. yavapa* | U | U | U | U | U |



| Ground spiders | 2011 | 2012 | 2013 | 2014 | 2015 |
|---|---|---|---|---|---|
| *Micaria riggsi* | | B | B | B | B |
| *Micaria rossica* | | B | B | B | B |
| *Haplodrassus eunis* | | U | U | U | U |
| *Drassodes neglectus* | | U | U | U | U |
| *Gnaphosa muscorum* | U | U | U | U | U |
| *Zelotes fratris* | U | U | U | U | U |

Many specialists by habitat or year

Lag time 1-2 yrs.

## Section 10.2. Forest thinning and prescribed fire impacts on forest ground-dwelling invertebrates.

Much of the SWJM CFLRP activities involved forest thinning followed by prescribed fire to burn off the remaining slash and expose mineral soils for herbaceous plant establishment. On one of these thinning/burning forest units, we assessed microclimate, vegetation and arthropod communities through the entire process. We conducted this monitoring on the Cerro Seco Unit 5 on Valles Caldera National Preserve. The monitoring included three periods of measurements: (1) A two-year pre-treatment monitoring period (Oct. 2015 to Dec. 2017); (2) a forest thinning treatment in Dec. 2017, followed by two years of post-thinning measurements (Dec. 2017-Oct. 2019); and (3) a planned managed broadcast burn during Oct. 17-18, 2019, followed by three years of post-fire measurements.

The site was mechanically thinned by a contractor in December 2017, using a feller-buncher and skidder; log decks were removed via trucks during winter-spring in 2018 (Figure 10.2.1, top). The thinning prescription was to reduce canopy closure from 80-85% to 45% over the total unit area; the thinning also was to create open gaps of meadow to function as fire breaks and increase 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.

NPS fire crews conducted a broadcast slash burn on 17-18 Oct. 2019 (Figure 10.2.1, bottom), which further reduced the tree density via burning of some saplings; wind-throw contributed to additional losses. The fire also removed much of the slash and forest litter and exposed mineral soil; mean litter depth at the treatment log sites was reduced to 0.9 cm (94% reduction).

Herbaceous vegetation and arthropods were monitored from 2017-2022. Herbaceous vegetation showed considerable increases following thinning and burning. Ground-dwelling arthropods also demonstrated responses similar to those from the Las Conchas Fire described above in Section 10.1. While analyses are still underway for many of the arthropod taxa, preliminary results are available for the Orthoptera (grasshoppers and crickets).

As the herbaceous vegetation increased on the site post-treatment, so too did the grasshopper assemblage (Table 10.2.1). Grasshoppers were very rare in undisturbed forest prior to the treatment, while camel crickets (*Ceuthophilus utahensis*) were very common. Once the canopy was opened up and grasses and forbs established, grasshoppers quickly colonized the area; by 2022 there were 15 species of grasshoppers and 1 katydid (the Mormon cricket, *Anabrus simplex*) on the site, and were represented by both adult males and females, along with nymphs – indicating successful reproduction and established breeding populations. The presence of these grasshoppers indicates that a healthy understory of herbaceous vegetation was establishing, leading us to conclude that the restoration objectives were being achieved: improve wildlife habitat while reducing fuel loads to mitigate high-severity fire risk.



Fig. 3. Top: Photograph of study site during thinning operations, Dec. 2017. Bottom: Photograph of planned, managed slash burn on study site during Oct. 17-18, 2019.

Table 10.2.1. List of Orthoptera recorded on treated and control plots on the Cerro Seco Unit 5 stand unit between 2017 (pre-treatment) and 2022 (post thinning and burning). Note that untreated (control) forest sites were dominated almost exclusively by camel crickets (*Ceuthophilus utahensis*), with grasshoppers (family Acrididae) being very rare in undisturbed forest. Grasshoppers increased greatly following treatment, in both species and abundances.

| | | **2017 (Pre-treatment)** | | **2022 (Post-treatment)** | |
|---|---|---|---|---|---|
| **Family** | **Species** | **Control** | **Thin/Rx Burn** | **Control** | **Thin/Rx Burn** |
| Rhaphidophoridae | *Ceuthophilus utahensis* | 90.67 | 135.63 | 39.75 | 48.00 |
| Rhaphidophoridae | *Styracosceles neomexicanus* | 0.10 | | | 0.12 |
| Acrididae | *Aeropedellus clavatus* | | | | 8.12 |
| Acrididae | *Arphia conspersa* | | 0.03 | | 3.60 |
| Acrididae | *Camnula pellucida* | | | | 18.24 |
| Acrididae | *Chorthippus curtipennis* | | | | 0.16 |
| Acrididae | *Cratypedes neglectus* | | | 0.08 | 0.08 |
| Acrididae | *Eritettix simplex* | | | | 0.04 |
| Acrididae | *Melanoplus bivitattus* | | | | 0.12 |
| Acrididae | *Melanoplus bruneri* | | | | 0.04 |
| Acrididae | *Melanoplus dodgei* | | | | 0.04 |
| Acrididae | *Melanoplus femurnigrium* | 0.30 | 0.03 | | 0.20 |
| Acrididae | *Melanoplus nufioi* | | | | 0.04 |
| Acrididae | *Trimerotropis modesta* | | | | 0.08 |
| Acrididae | *Trimerotropis pallidipennis* | | | | 0.04 |
| Acrididae | *Trimerotropis suffusa* | | | | 0.08 |
| Acrididae | *Xanthippes corallipennis* | | | | 0.08 |
| Tettigoniidae | *Anabrus simplex* | | | | 0.36 |
| Acrididae | Nymphs of various acridid grasshoppers | | 0.03 | | 28.04 |
| | | | | | |
| | **Total grasshoppers and crickets** | **91.07** | **135.72** | **39.83** | **107.48** |
| | **Total number of species present** | **3** | **4** | **2** | **18** |

## Section 10.3. Post-fire dynamics of bark beetles.

During the fall of 2016, USDA Forest Service entomologists assessed the condition of an old growth stand on Valles Caldera National Preserve, known as the History Grove; the stand of old Douglas fir and ponderosa pine trees was located on the western edge of the Valle Grande. In late May and June 2013, the Thompson Ridge fire started and spread throughout nearby and adjacent forests and ultimately burned ponderosa pine and Douglas fir trees near the western edge of the Grove (Figure 10.3.1). The larger trees in this stand were left uncut during various logging times around the Preserve and represent some of the older trees in the area (Figure 10.3.2). Although most of the trees likely survived the direct effects of the fire, many of the trees subsequently became infested with bark beetles that were spreading to unaffected trees throughout the stand.



Figure 10.3.1. A satellite image of the History Grove showing burned trees and shrubs on the western edge of the Grove caused by the Thompson Ridge fire. The red oval indicates the area with the highest number of trees with bark beetle damage.



Figure 10.3.2. The History Grove with large, old ponderosa pine and Douglas fir trees.

The area inspected was approximately 43 acres in size and included mostly ponderosa pine trees with a smaller component of Douglas fir and scattered white fir. The overall stand basal area ranged from ~70 to 110 $ft^2$ per acre. We inspected all trees for evidence of attack by bark beetles based on fading or yellowing of the needles in the crown and looking for frass (on

Douglas fir trees, see Figure 10.3.3A) or pitch tubes (on ponderosa pine trees, see Figure 10.3.3B) on the lower portion of the tree bole.

The Douglas fir beetle (*Dendroctonus pseudotsugae*) is a native bark beetle that occurs throughout the range of Douglas fir. It has one generation per year. The beetles generally spend the winter under the bark as adults and when temperatures begin to warm in the spring beetles fly and attack trees. The female beetle initiates the attack and produces a pheromone that is attractive to males which then join the female under the bark and both produce attractive pheromones that attract additional beetles to the tree. Under the bark, beetles mate, and females lay eggs in vertical galleries. Larvae hatch and feed horizontally away from the egg galleries (Fig. 10.3.3D). The effect of the larval feeding and their mutualistic staining fungi cuts off the downward nutrient movement in the phloem and clogs upward, water moving vessels which causes the death of the tree. Following pupation, the new adult beetles spend the winter underneath the bark and repeat the cycle the following year.

Of the more than 500 trees surveyed (the majority being ponderosa pine), 80 trees were attacked by bark beetles. Of the attacked trees, 75 were Douglas fir and 5 were ponderosa pine. We were able to determine that 54 of the Douglas fir trees had been attacked during 2016, whereas 21 had been attacked and killed in 2015 or earlier. All five of the ponderosa pines had been attacked in 2016 by either western pine beetle during the summer or by roundheaded pine beetle in late summer. The GPS coordinates of currently infested tress were noted during the survey. We anticipated that all of the attacked trees would produce successful brood and subsequent adult beetles that flew in the following year and attack additional trees. The potential spread of beetles attacking ponderosa pine was considerably less than that of the Douglas fir beetle, primarily because of the low population levels of beetles in those trees and pine bark beetles do not commonly attack unaffected trees outside of burned areas, unlike Douglas fir beetle.

In 2017-2019, we installed 20 Lindgren funnel traps with pheromone lures in the History Grove to collect Douglas fir beetles. Traps were emptied every 1-2 weeks from April to September, and the number of Douglas fir beetles were counted. Other species of beetles were also identified and counted. During the peak of the beetle infestation (2017), we collected 277,848 Douglas fir beetles (more than 1,000 beetles per trap per day!), and these numbers declined precipitously in 2018 (26,434 individuals) and 2019 (14,467 individuals); see Fig. 10.3.4. In addition to the Douglas fir beetles, Dr. Richard Buss (NPS volunteer) identified 368 other beetle species, illustrating the enormous diversity of beetles in the CFLRP project area. Of these other beetles, 67 species are known to feed on or burrow into various tree species in the Jemez Mountains, and therefore are of potential economic importance to the timber industry in the region.

Following this outbreak of Douglas fir beetles, it became obvious that the vast majority of old Douglas fir trees in the Preserve succumbed to bark beetle attacks; standing dead firs could be seen across the landscape. Younger trees appeared healthy and were thriving, so the next generation of Douglas firs were replacing the old trees lost to the beetles. The dead trees were functioning as excellent snag habitat for wildlife, and as these trees decay and fall, their logs will provide preferred habitat for the endangered endemic Jemez Mountains Salamander. Thus, the Douglas fir beetle outbreak following the Thompson Ridge Fire provided an outstanding example of how “invertebrates rule the world.”

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Figure 10.3.3. Red boring dust on the bark surface of a Douglas fir tree (A) compared to white and reddish pitch tubes on the bark surface of a ponderosa pine tree (B) as indicators of bark beetle attack. The History Grove with characteristic "fading" Douglas fir trees in the background (C) and a close-up of the underside of the bark of a Douglas fir bark with adult and larval Douglas fir beetle galleries (D).

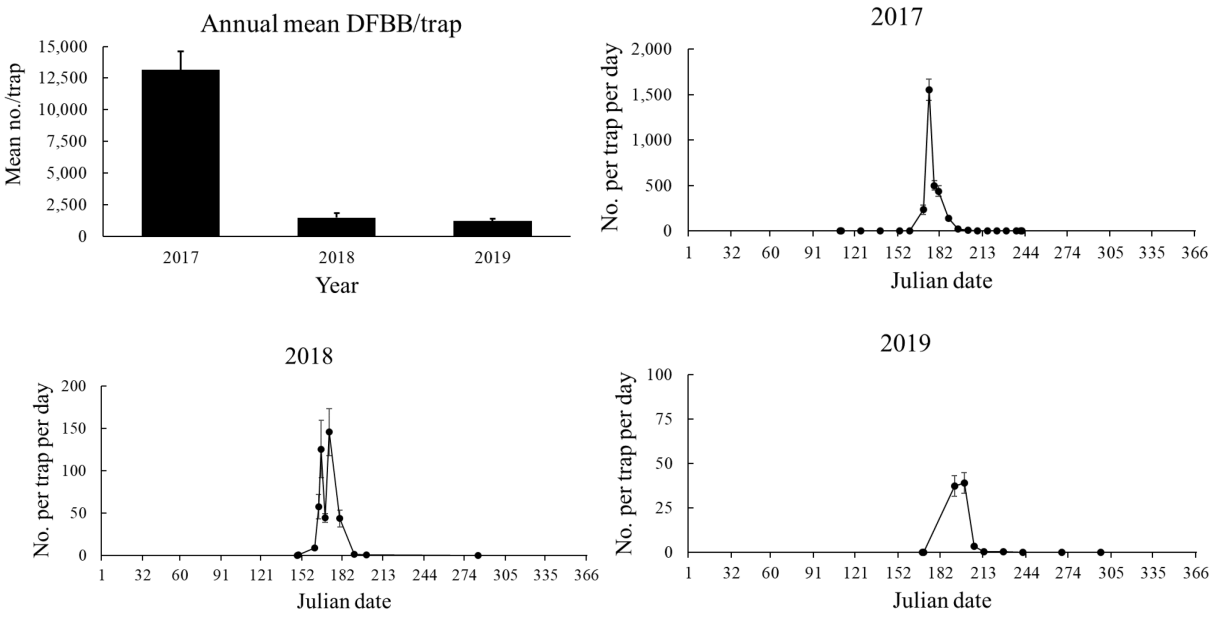



Figure 10.3.4. Numbers of Douglas fir bark beetles (DFBB) collected in Lindgren funnel traps in the History Grove, Valles Caldera National Preserve, NM, during 2017-2019. Error bars represent standard errors; note scale changes in y-axes.

Table 10.3.1. List of forest beetles of potential economic importance and species abundances collected with Lindgren funnel traps in the History Grove of Valles Caldera National Preserve, NM, during 2017-2019.

| Family | Subfamily/Tribe | Species | Total |
|---|---|---|---|
| Curculionidae | Ceutorhynchinae | *Ceutorhynchus neglectus?* | 4 |
| Curculionidae | Ceutorhynchinae | *Ceutorhynchus* sp. | 2 |
| Curculionidae | Ceutorhynchinae | *Glocianus punctiger* C. R. Sahlberg | 1 |
| Curculionidae | Conoderinae | *Cylindrocopturus* sp. | 2 |
| Curculionidae | Conoderinae | *Lechriops californicus* (LeConte) | 1 |
| Curculionidae | Cossoninae | *Cossonus* sp. | 9 |
| Curculionidae | Cossoninae | *Rhyncolus* sp. | 1 |
| Curculionidae | Curculioninae | *Mecinus linnavuori (Korotyaev)* | 1 |
| Curculionidae | Curculioninae | *Phyllotrox* sp. | 1 |
| Curculionidae | Entiminae | *Orimodema protracta* Horn | 1 |
| Curculionidae | Entiminae | *Thricolepis inornata* Horn | 1 |
| Curculionidae | Hylesinini | *Dendroctonus frontalis* Zimmermann | 1 |
| Curculionidae | Hylesinini | *Dendroctonus ponderosae* Hopkins | 3 |
| Curculionidae | Hylesinini | *Dendroctonus pseudotsugae* Hopkins | 277,848 |
| Curculionidae | Hylesinini | *Dendroctonus valens* LeConte | 1 |
| Curculionidae | Hylesinini | *Hylastes asperatus* Wood | 11 |

| | | | |
|---|---|---|---|
| Curculionidae | Hylesinini | *Hylastes gracilis* LeConte | 10 |
| Curculionidae | Hylesinini | *Hylastes macer* LeConte | 1 |
| Curculionidae | Hylesinini | *Hylastes ruber* Swaine | 5 |
| Curculionidae | Hylesinini | *Hylurgops subcostulatus subcostulatus* (Mannerheim) | 1 |
| Curculionidae | Hylesinini | *Pseudohylesinus nebulosus nebulosus* (LeConte) | 1 |
| Curculionidae | Mesoptiliinae | *Magdalis sp.* | 2 |
| Curculionidae | Scolytinae | *Dendroctonus pseudotsugae* Hopkins | 3 |
| Curculionidae | Scolytini | *Chramesus asperatus* Schaeffer | 2 |
| Curculionidae | Scolytini | *Conophthorus edulis* Hopkins | 1 |
| Curculionidae | Scolytini | *Conophthorus ponderosae* Hopkins | 4 |
| Curculionidae | Scolytini | *Gnathotrichus denticulatus* Blackman | 3 |
| Curculionidae | Scolytini | *Gnathotrichus sulcatus* (LeConte) | 3 |
| Curculionidae | Scolytini | *Ips borealis lanieri* Wood | 2 |
| Curculionidae | Scolytini | *Ips grandicollis* (Eichhoff) | 1 |
| Curculionidae | Scolytini | *Ips latidens* (LeConte) | 3 |
| Curculionidae | Scolytini | *Ips pini* (Say) | 6 |
| Curculionidae | Scolytini | *Pityogenes fossifrons* (LeConte) | 1 |
| Curculionidae | Scolytini | *Pityophthorus* sp. 1 | 5 |
| Curculionidae | Scolytini | *Pityophthorus* sp. 2 | 5 |
| Curculionidae | Scolytini | *Scolytus reflexus* Blackman | 1 |
| Curculionidae | Scolytini | *Scolytus unispinosus* LeConte | 1 |
| Curculionidae | Scolytini | *Xyleborus intrusus* Blandford | 2 |
| Curculionidae | Scolytini | *Xyleborus* sp. | 4 |
| Bostrichidae | Dinoderinae | *Stephanopachys cribratus* (LeConte) | 4 |
| Bostrichidae | Lyctinae | *Lyctus carbonarius* Waltl | 7 |
| Buprestidae | Buprestinae | *Anthaxia* sp. | 7 |
| Buprestidae | Buprestinae | *Buprestis maculiventris* Say | 5 |
| Buprestidae | Buprestinae | *Chrysobothris ludificata?* | 13 |
| Buprestidae | Buprestinae | *Phaenops (drummondi?)* | 2 |
| Buprestidae | Buprestinae | *Phaenops gentilis* (LeConte) | 1 |
| Buprestidae | Buprestinae | *Phaenops* sp. | 1 |
| Buprestidae | Chrysochroinae | *Chalcophora angulicollis* (LeConte) | 1 |
| Cerambycidae | Cerambycinae | *Neoclytus muricatulus* Kirby | 2 |
| Cerambycidae | Cerambycinae | *Phymatodes dimidiatus* (Kirby) | 24 |
| Cerambycidae | Cerambycinae | *Phymatodes varius* (Fabricius) | 1 |
| Cerambycidae | Cerambycinae | *Xylotrechus sp.* | 50 |
| Cerambycidae | Lamiinae | *Acanthocinus obliquus* (LeConte) | 3 |
| Cerambycidae | Lamiinae | *Monochamus clamator* LeConte | 10 |
| Cerambycidae | Lepturinae | *Acmaeops proteus* Kirby | 36 |
| Cerambycidae | Lepturinae | *Gnathacmaeops pratensis* (Laicharting) | 4 |
| Cerambycidae | Lepturinae | *Rhagium inquisitor* (Linnaeus) | 4 |
| Cerambycidae | Lepturinae | *Stictoleptura canadensis* (Olivier) | 1 |

| | | | |
|---|---|---|---|
| Cerambycidae | Lepturinae | *Xestoleptura cockerelli* (Fall) | 6 |
| Cerambycidae | Prioninae | *Prionus californicus* Motschulsky | 1 |
| Cerambycidae | Prioninae | *Prionus emarginatus* Say | 3 |
| Cerambycidae | Prioninae | *Tragosoma harrisii* LeConte | 31 |
| Cerambycidae | Spondylinae | *Arhopalus rusticus montanus* (LeConte) | 3 |
| Cerambycidae | Spondylinae | *Asemum striatum* (Linnaeus) | 12 |
| Cerambycidae | Spondylinae | *Tetropium parallelum* Casey | 1 |
| Cerambycidae | Spondylinae | *Tetropium velutinum* LeConte | 1 |
| Cerambycidae | Cerambycinae | *Neoclytus muricatulus* Kirby | 2 |

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Brantley, S. L. 2020. Responses of ground-dwelling spider (Arachnida: Araneae) communities to wildfire in three habitats in northern New Mexico, USA, with notes on mites and harvestmen (Arachnida: Acari, Opiliones). *Diversity* 2020, 12, 396; https://doi.org/10.3390/d12100396

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# Chapter 11. Soils and Soil Biota Monitoring

**Monitoring project title: Monitoring soils and soil biota during landscape restoration.**

**Collaborators:** Anita Antoninka, Kara Gibson – *Northern Arizona University*; Nancy Johnson – *School of Earth and Sustainability and Department of Biological Sciences, Northern Arizona University;* Henry Grover – *United States Department of Agriculture Forest Service*
Robert R. Parmenter – *National Park Service, Valles Caldera National Preserve;* Mark Losleben – *University of Arizona;* Rachel Loehman – *USFS Rocky Mountain Research Station.*

**Summary:** 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.

Soil biotic communities are responsible for a disproportionate number of ecosystem functions, including carbon and nutrient cycling, soil fertility, soil stability and water retention. Understanding how these communities change based on land management activities is a first step in understanding how ecosystem function might change. Forest treatments (thinning and use of logging machinery) can strongly impact soil abiotic parameters (soil compaction, bulk density, hydrology, organic matter, carbon, and nutrients) and important groups of soil organisms (nematodes, microarthropods, bacteria, and fungi). We addressed the effects of forest thinning and prescribed fire on soil biotic communities and abiotic parameters. Responses of soil invertebrates to forest management vary across taxa, treatment type, and disturbance intensity. Although our findings suggest that many soil invertebrates are relatively resilient or resistant to forest treatments, certain faunal groups may be negatively impacted by some management activities, especially at higher levels of disturbance.

## Section 11.1. Forest thinning and prescribed fire effects on soil temperature and moisture.

**(*Sections reprinted/edited from Parmenter and Losleben, 2023 [with permission]*)**

*Introduction*

The objective of this soil monitoring program 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 (Parmenter and Losleben 2023). 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.

*Methods*

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 11.1.1); 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.

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.

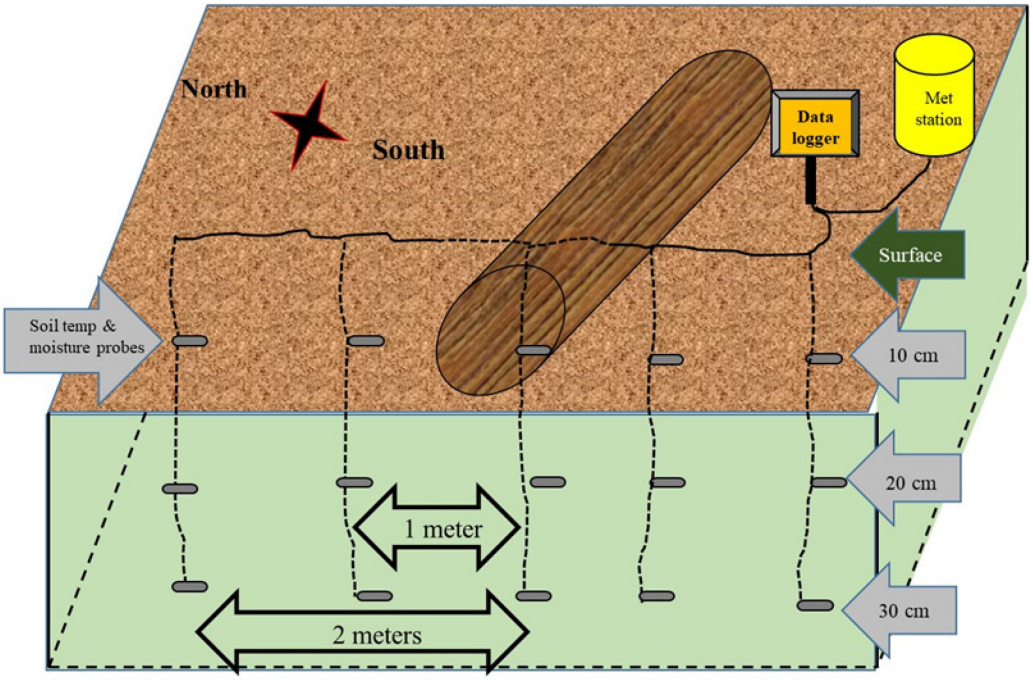



Figure 11.1.1. Schematic layout of the soil sensors relative to a log (not to scale).

We used a 2 × 3 × 4 × 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.

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

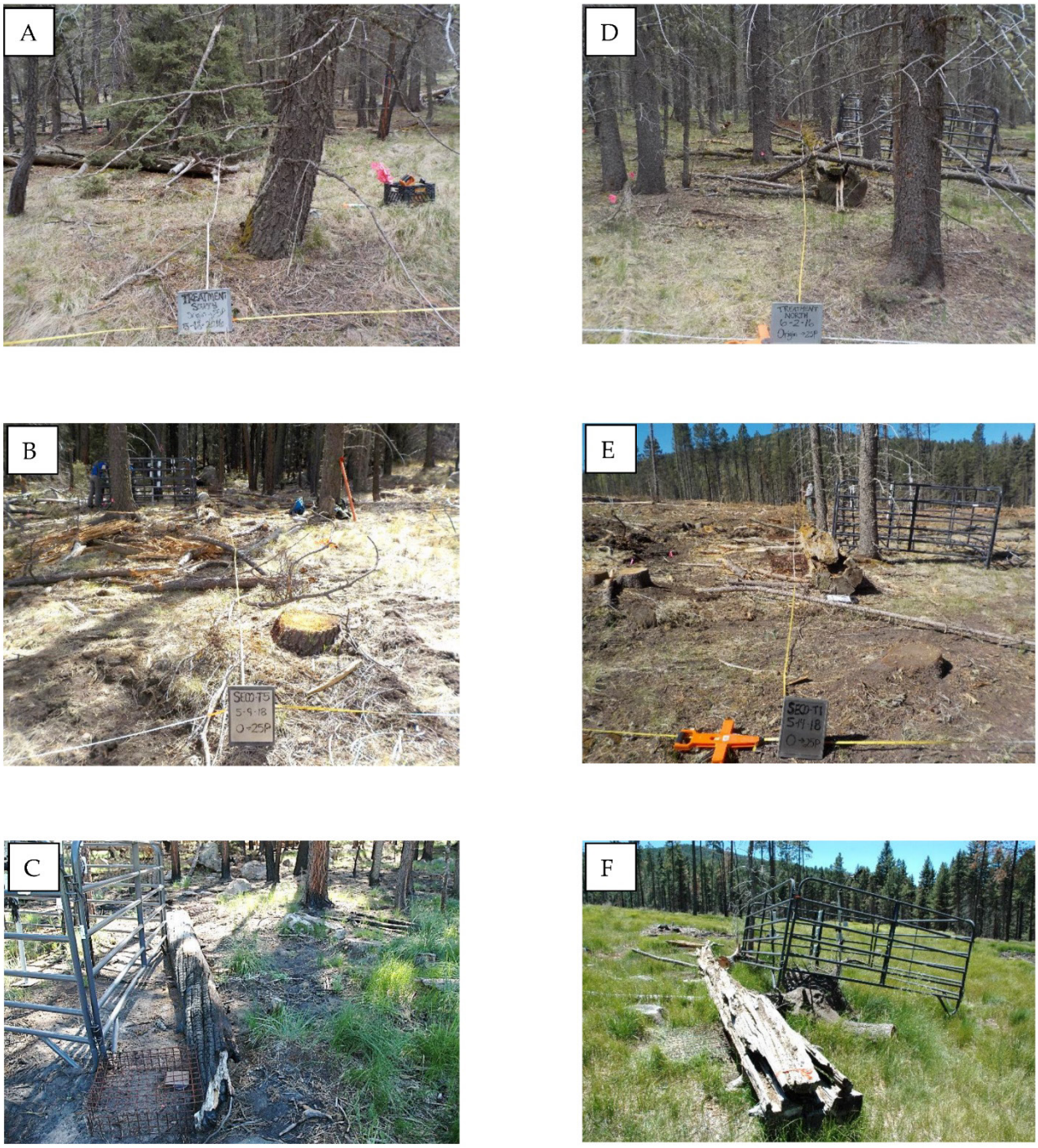



Figure 11.1.3. Photographs of logs on the treatment sites: South Site (left column) and North Site (right column). (A,D): Pre-thinning, undisturbed condition in 2016. (B,E): Post-thinning condition in 2018. (C,F): Post-fire condition in 2022.

*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. 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. The treatment effects (thinning and burning) resulted in small but significant changes in the air temperature (Table 11.1.1). The temperatures across all sites ranged from −16.8 °C to 22.7 °C.

Table 11.1.1. 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. 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 11.1.1).

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 11.1.1); 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. In contrast, the wind speeds on the Treatment sites showed much larger increases following the thinning and burning treatments (Figure 11.1.4 A,B).

Pre-treatment solar radiation (daily PAR) averaged 51 µmol/m²s across the study sites and increased by 15 µmol/m²s after the pre-treatment period on the Control sites (Table 11.1.1). On the Treatment sites, PAR exhibited large increases (mean of 269 µmol/m²s) following canopy reduction from the thinning treatment (Figure 11.1.4 C).

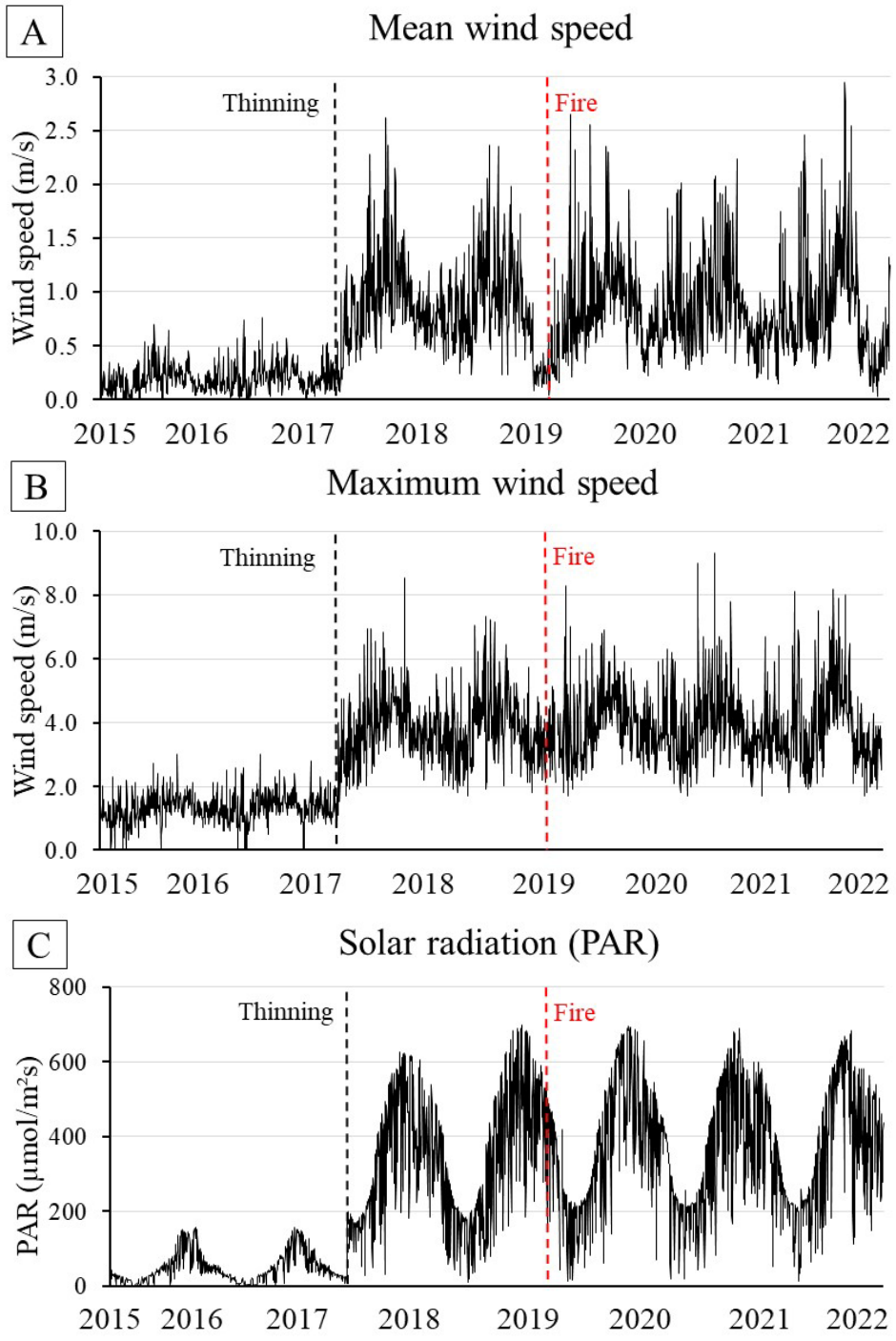



Figure 11.1.4. Example of the thinning and burning treatment effects from the North Treatment site on (A) mean wind speed, (B) maximum wind speed, and (C) Photosynthetically Active Radiation (PAR) during 2015–2022.

*Soil Temperature 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. 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 11.1.5, Table 11.1.2); 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 11.1.5; Table 11.1.2). 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 11.1.6; Table 11.1.2).

In addition, the diel patterns of soil temperatures over time scales of several days differed following the thinning and burning treatments (Figure 11.1.7). 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 11.1.7 A,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 11.1.7 C,D).

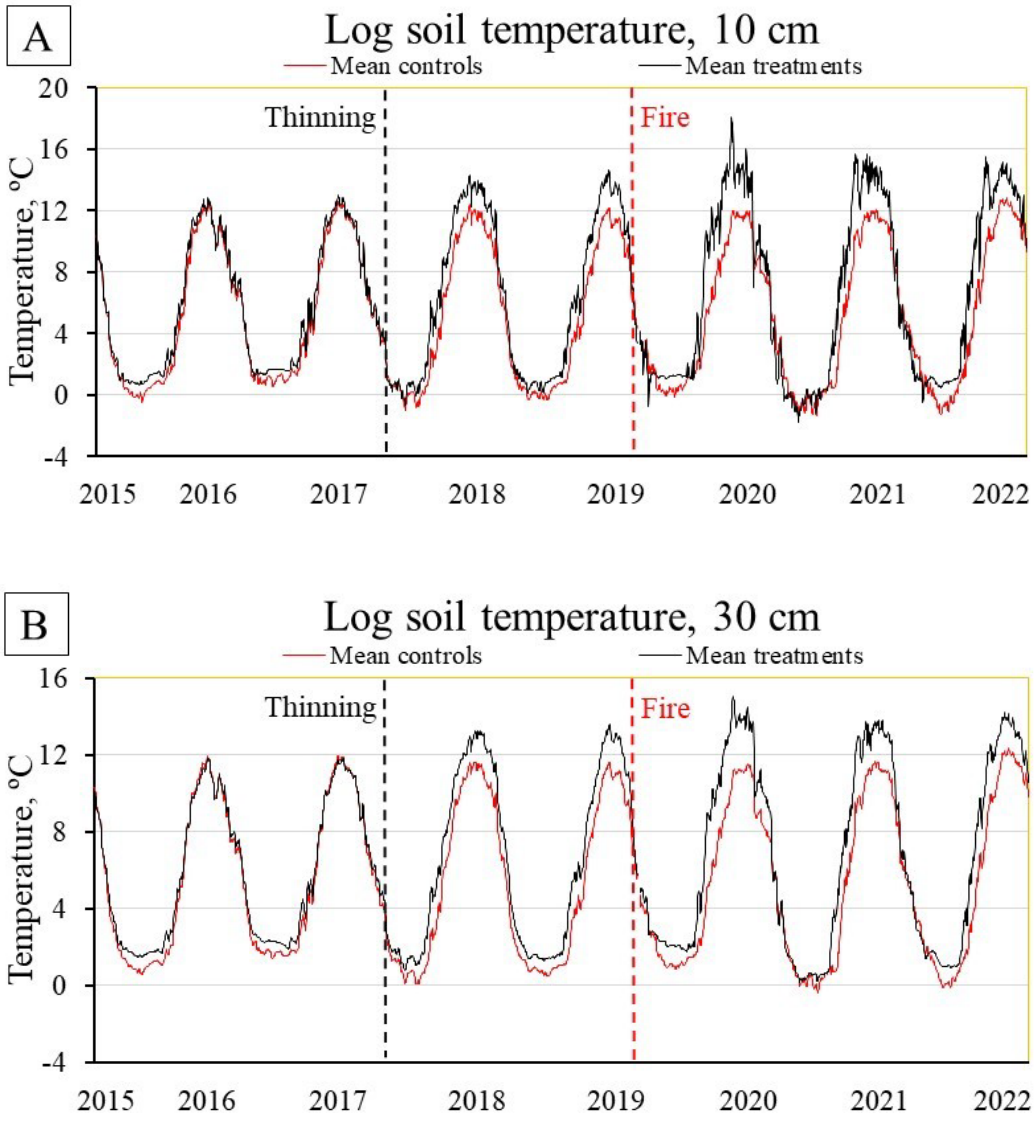



Figure 11.1.5. Mean soil temperature dynamics at 10 cm (A) and 30 cm (B) depths between Control and Treatment sites beneath the logs during 2015–2022 on the Cerro Seco study site.

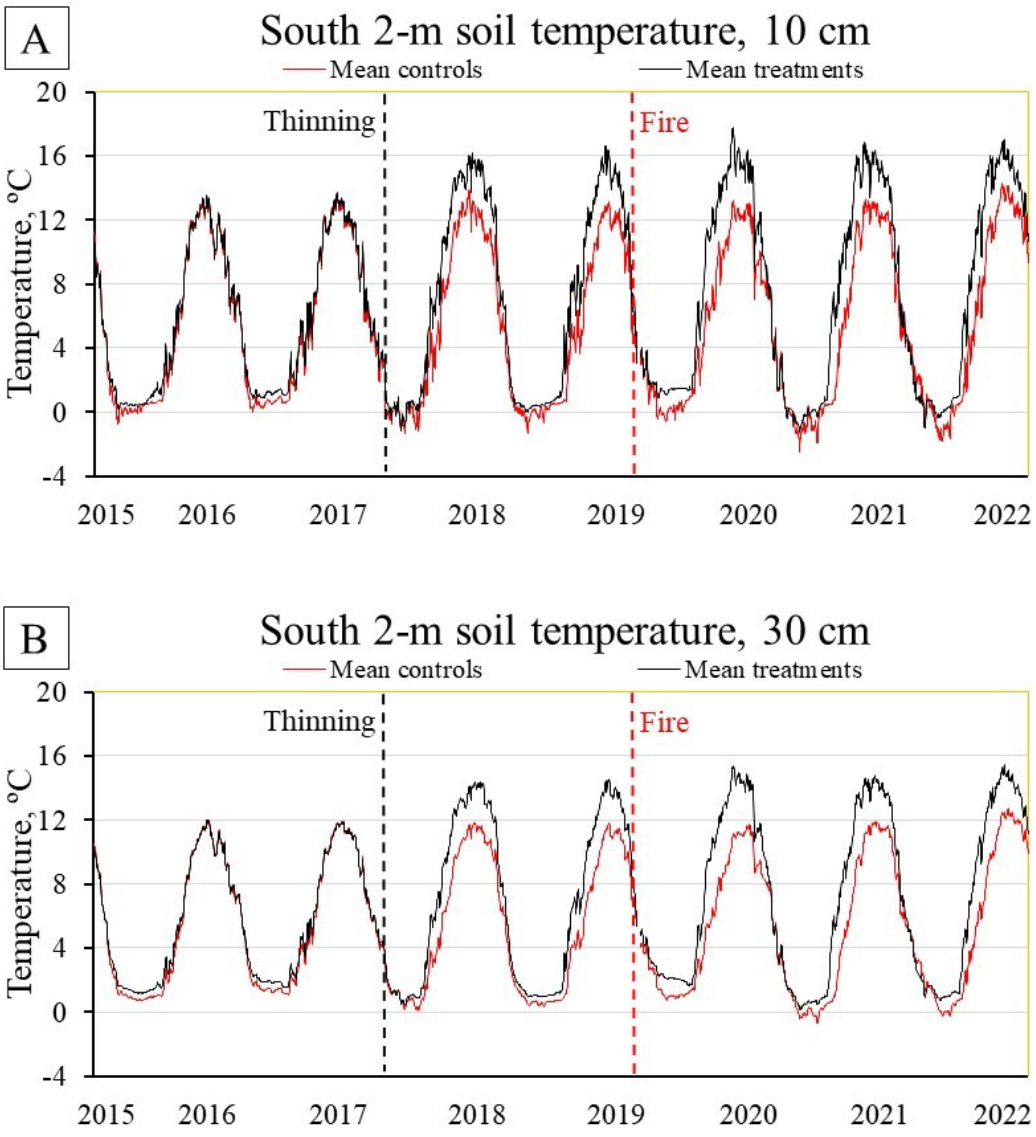



Figure 11.1.6. Example graphics for mean soil temperature dynamics at 10 cm (A) and 30 cm (B) depths between Control and Treatment sites at sites 2 m south of the logs during 2015–2022 on the Cerro Seco study site.

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. 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 (Water Years 2018, 2019) | | | | Post-Fire (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 |

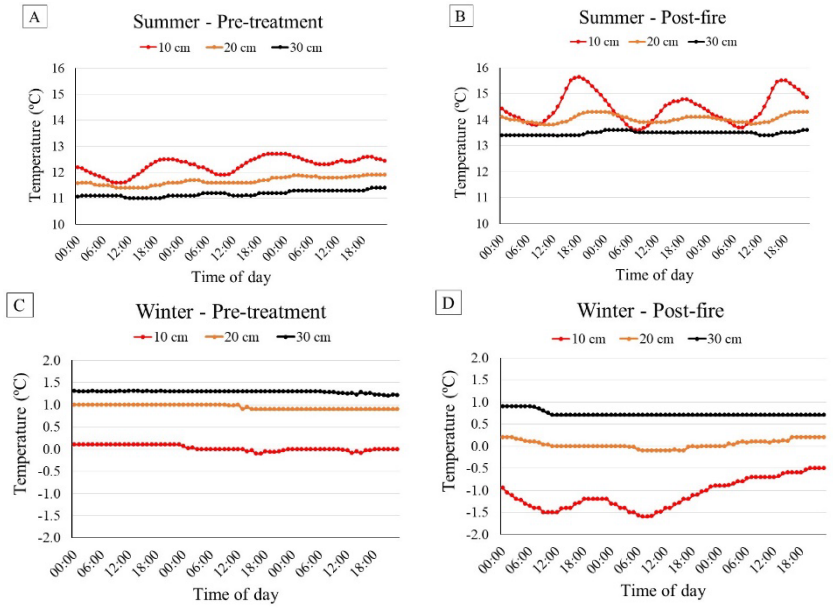



Figure 11.1.7. Examples of diel patterns of the soil temperature beneath the logs at different depths at the Cerro Seco study site. Data from the North Treatment Site. (A): 22–24 July 2016 (pre-treatment); (B): 14–16 July 2020 (post-fire); (C): 26–28 January 2016 (pre-treatment); (D): 12–14 February 2019 (post-fire).

*Soil Moisture Dynamics*

Soil moisture (VWC) also exhibited significant treatment, period, season, location, and most interaction effects in the Repeated-Measures ANOVA results (Table 11.1.3); 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 $m^3/m^3$. The soils beneath the logs at 10 cm depth were significantly drier than the soils at both 20 and 30 cm depths (Table 11.1.2). The soil moisture peaked in spring, concomitant with snowmelt, and exhibited smaller spikes during summer and fall associated with monsoon rain events. 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. 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 11.1.8, Table 11.1.3). Deeper soils (20 and 30 cm) beneath the logs following burning showed larger VWC increases in the range of 0.05–0.08 $m^3/m^3$.

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 11.1.3). 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 11.1.3). The 2 m South Site displayed no significant changes in VWC (Figure 11.1.9), despite showing the largest increase in temperature (Figure 11.1.6).

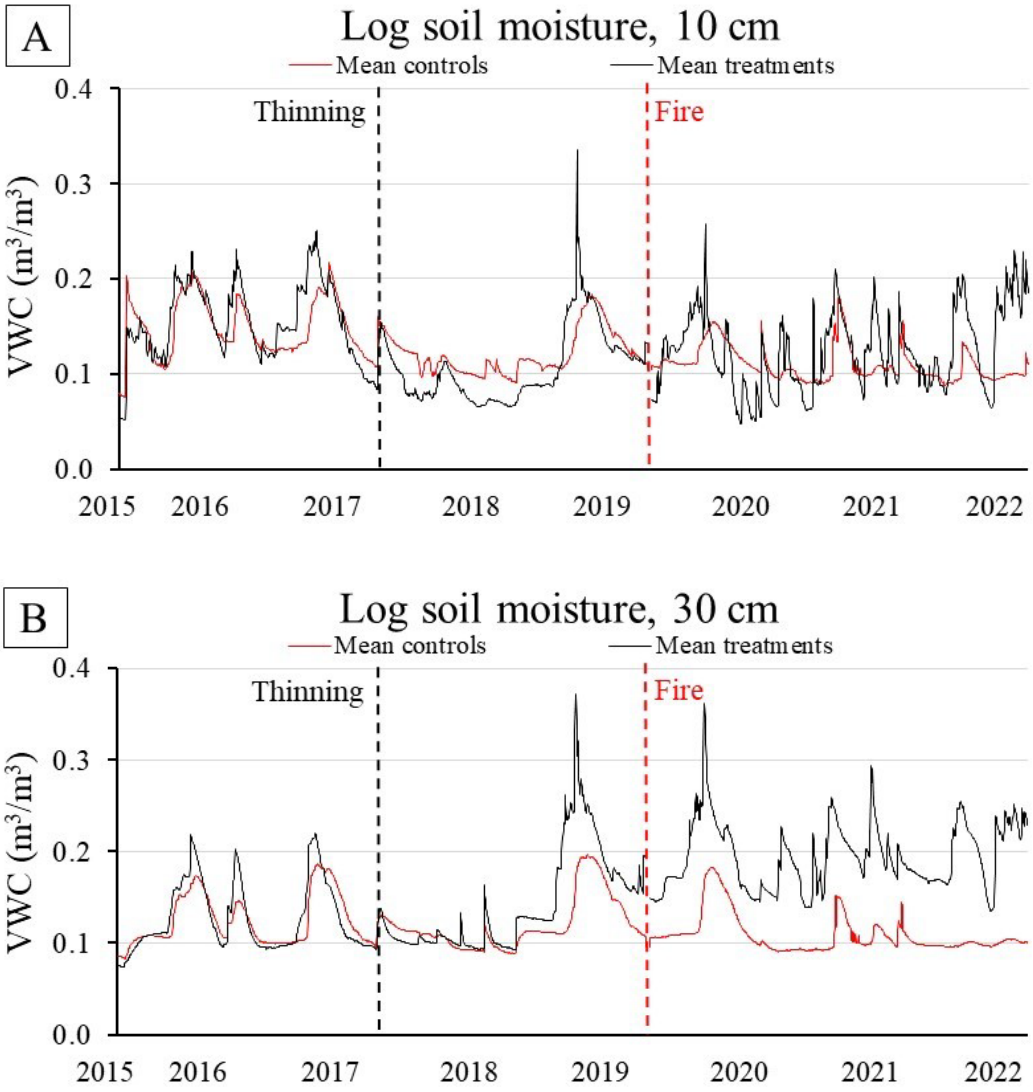



Figure 11.1.8. Mean soil moisture dynamics at 10 cm (A) and 30 cm (B) depths between Control and Treatment sites beneath the logs during 2015–2022 in the Cerro Seco study site.

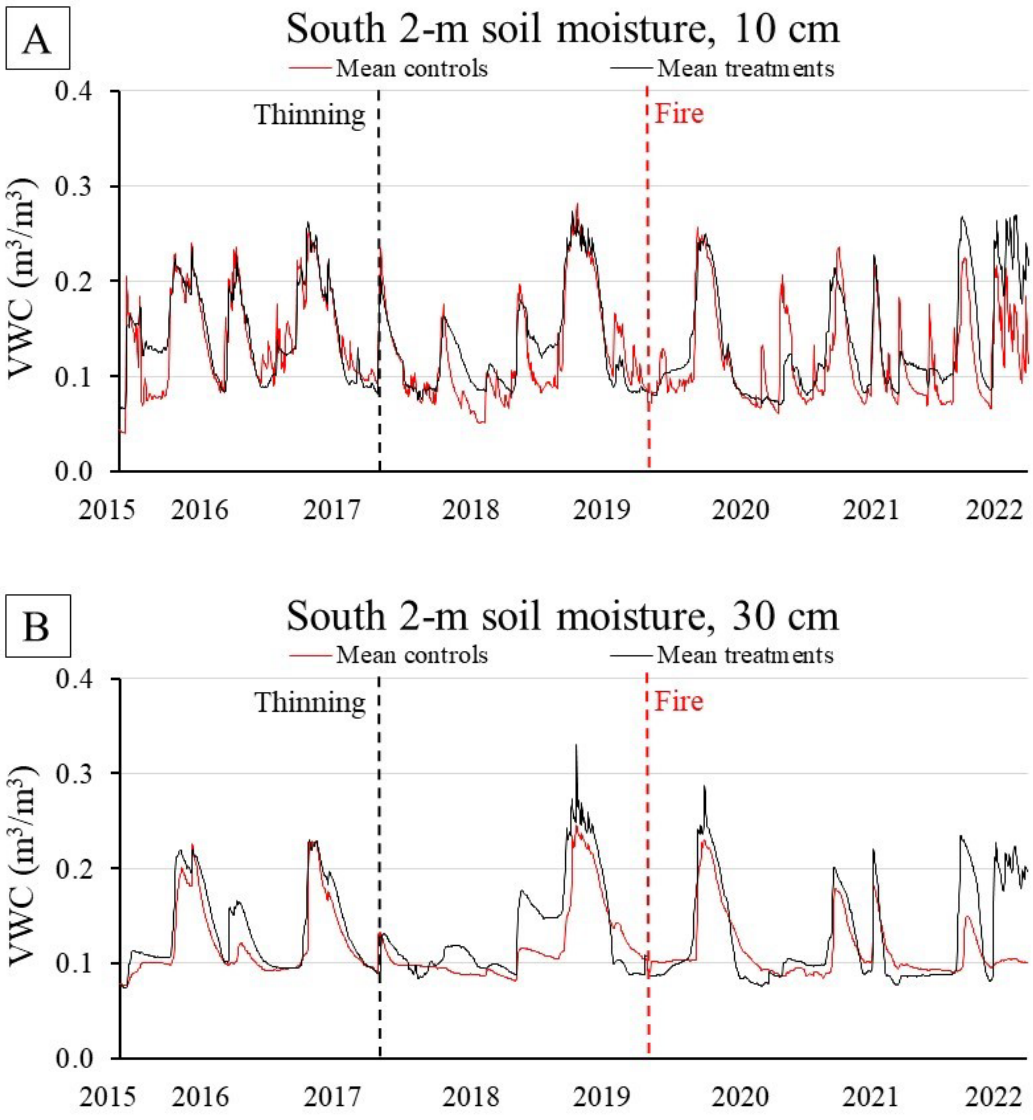



Figure 11.1.9. Mean soil moisture dynamics at 10 cm (A) and 30 cm (B) depths between Control and Treatment sites 2 m south of the logs during 2015–2022 in the Cerro Seco study site.

Table 11.1.3. 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. 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 (Water Years 2018, 2019) | | | | Post-Fire (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 |

*Discussion*

On this monitoring site, the thinning operations and associated post-treatment wind-throw removed nearly 90% of the trees (from a mean of 1,429 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 11.1.2). The treatments opened the site to much greater levels of PAR and wind speeds, but only minimal increases in the average air temperatures occurred.

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 11.1.2). 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 11.1.7 A,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 11.1.7 C,D). This was especially evident in the winter of 2017–2018, which had virtually no snowpack to insulate the ground from radiative heat losses. 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 11.1.2).

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 11.1.3). 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. 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. Open gaps in the forest canopy allow 25% more snow depth on the ground, versus closed canopy stands. 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 11.1.3).

The results of our monitoring 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 Jemez Mountains 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.

## Literature Cited

Parmenter, R. R., and M. V. Losleben. 2023. 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. *Forests,* 14, 1117. https://doi.org/10.3390/f14061117

## Section 11.2. Soil Biota

**Monitoring Project Title: Effects of thinning and prescribed fire on abundance of soil microarthropods and nematodes in a *Pinus ponderosa* forest.**

In the first soil monitoring study, we quantified nematodes, mites, and collembolans in soil and litter habitats within three adjacent ponderosa pine forest management units at Banco North in VALL: an untreated control unit (hereafter referred to as the "control" unit), a thinned-only unit (comprising thinning and masticating wood; hereafter referred to as the "thinned" unit), and a thinned/burned unit (comprising thinning and masticating followed by a broadcast burn, hereafter referred to as the "burned" unit). We linked patterns in animal abundance to resource and habitat characteristics, hypothesizing that resources and available habitat for many taxa would increase with thinning and decrease with burning.

Soil abiotic characteristics and densities of hyphae formed by arbuscular mycorrhizal fungi (symbionts of grasses and many other plants, but not pines) and dark septate fungi are summarized in Table 11.2.1. Two years after thinning, densities of collembolans and nematodes in the thinned unit were higher than in the control unit, but one year post-fire, their densities in the burned unit were similar to those of the untreated control unit (Fig. 11.2.1). Mite abundance, however, was not elevated in the thinned unit and was lower in the burned unit (Gibson et al., 2022) (Fig. 11.2.1). Although faunal communities were highly heterogeneous, a significant proportion of the variance in faunal abundances was explained by easily and inexpensively measured habitat and resource characteristics: bulk density, soil organic matter (SOM), pH, grass cover, and litter cover and depth (Table 11.2.2). These findings demonstrate the abiotic and biotic factors that structure faunal habitats so that forest managers have a more complete understanding of the impacts of forest restoration treatments.

Table 11.2.1. Litter depth, soil abiotic properties, and lengths of arbuscular mycorrhizal (AM) and septate hyphae (m $g^{-1}$). Groups with different letters are significantly different at $P<0.1$ according to univariate ANOVA or Kruskal-Wallis (KW), after correction for multiple comparisons using Tukey's test of honestly significant differences for ANOVA or Benjamini Hochberg adjustment for KW.

| | Test | *P* | *Control* | | | *Thinned* | | | *Burned* | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | *n* | Mean | SE | *n* | Mean | SE | *n* | Mean | SE |
| Litter depth (cm) | KW | 0.21 | 7 | 3.92 | 1.25 a | 8 | 3.03 | 1.01 a | 6 | 1.44 | 0.64 a |
| Bulk density ($g/cm^3$) | ANOVA | 0.014 | 8 | 0.92 | 0.05 ab | 8 | 0.77 | 0.04 b | 7 | 1.02 | 0.07 a |
| pH | ANOVA | 0.084 | 8 | 4.85 | 0.16 ab | 8 | 4.56 | 0.22 b | 8 | 5.25 | 0.23 a |
| % SOM | KW | 0.047 | 8 | 6.00 | 0.77 ab | 8 | 8.66 | 1.45 a | 8 | 5.46 | 1.26 b |
| % C | KW | 0.37 | 8 | 2.82 | 0.54 a | 8 | 3.34 | 0.81 a | 8 | 2.34 | 0.71 a |
| $PO_4$ µg/g | KW | 0.26 | 7 | 3.44 | 0.69 a | 8 | 3.73 | 0.75 a | 8 | 6.43 | 1.57 a |
| $NH_4$ µg/g | KW | 0.093 | 7 | 5.90 | 0.39 a | 8 | 7.92 | 1.14 a | 8 | 18.48 | 9.48 a |
| $NO_3$ µg/g | KW | 0.37 | 7 | 0.21 | 0.02 a | 8 | 0.33 | 0.10 a | 8 | 0.59 | 0.21 a |
| % N | KW | 0.37 | 8 | 0.12 | 0.02 a | 8 | 0.13 | 0.02 a | 8 | 0.11 | 0.04 a |
| % Clay | ANOVA | 0.004 | 8 | 9.28 | 0.94 a | 8 | 5.10 | 0.40 b | 8 | 6.43 | 0.90 b |
| % Silt | ANOVA | 0.008 | 8 | 43.71 | 4.18 a | 8 | 25.78 | 2.24 b | 8 | 32.06 | 4.31 b |
| % Very fine sand | ANOVA | 0.006 | 8 | 19.60 | 1.39 a | 8 | 12.50 | 0.89 b | 8 | 16.08 | 1.71 ab |
| % Fine sand | ANOVA | 0.070 | 8 | 14.32 | 2.02 b | 8 | 20.17 | 1.56 a | 8 | 19.33 | 1.86 ab |
| % Medium sand | ANOVA | 0.008 | 8 | 7.67 | 2.28 b | 8 | 18.91 | 1.61 a | 8 | 13.92 | 2.78 ab |
| % Coarse sand | ANOVA | 0.004 | 8 | 5.00 | 2.00 b | 8 | 16.41 | 1.88 a | 8 | 11.29 | 2.43 ab |
| Fine AM hyphae (m/g) | ANOVA | 0.90 | 8 | 7.49 | 0.95 a | 8 | 8.01 | 1.20 a | 8 | 7.22 | 1.51 a |
| Coarse AM hyphae (m/g) | ANOVA | 0.041 | 8 | 1.81 | 0.33 b | 8 | 3.19 | 0.40 a | 8 | 2.31 | 0.34 ab |
| Septate hyphae (m/g) | ANOVA | 0.63 | 8 | 3.41 | 0.84 a | 8 | 4.12 | 0.72 a | 8 | 4.59 | 1.03 a |

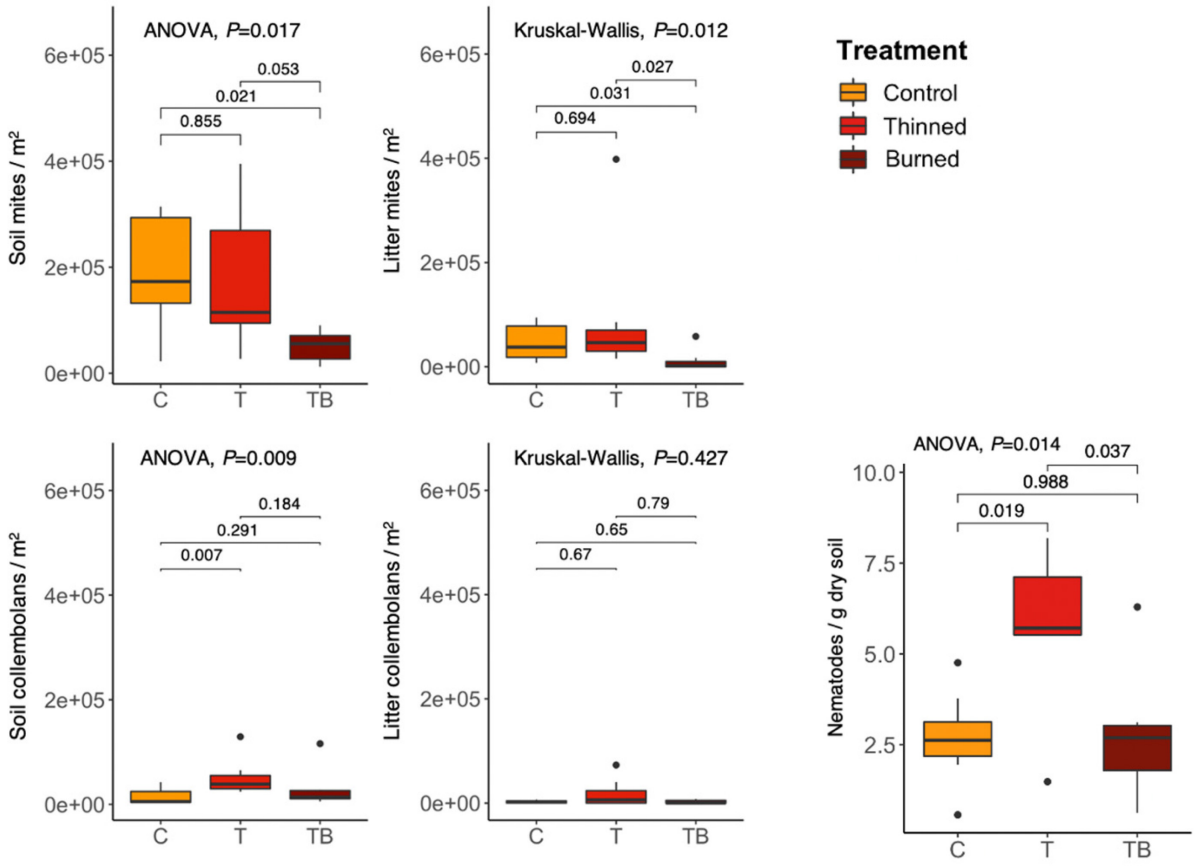



Fig. 11.2.1. Densities of mites and collembolans in soil and litter habitats and nematodes in soil within untreated control, thinned, and burned management units. The center line within each box represents the median for the treatment, and the lower and upper bounds of the box represent the first and third quartiles, respectively. The whiskers show any data points extending up to 1.5 times the interquartile range from the bounds of the box. Any dots are extreme values more than 1.5 times the interquartile range from the upper or lower box boundaries. Horizontal bars above the boxes show *P*-values for pairwise comparisons, adjusted for multiple comparisons by Tukey's test of honestly significant differences for ANOVA or Benjamini Hochberg adjustment for Kruskal-Wallis.

**Table 11.2.2.** Summaries of models predicting abundances of faunal groups. Optimal models were selected using an automated stepwise procedure based on Akaike information criterion (AIC) values.

| | Parameter Estimate | Std. Error | *t* value | Parameter *P*-value | Adjusted $R^2$ | Overall model *P*-value |
|---|---|---|---|---|---|---|
| ln (Soil mites / $m^2$) | | | | | | |
| Intercept | 8.52 | 0.91 | 9.404 | <0.001 | | |
| % Litter cover | 0.01 | 0.01 | 1.823 | 0.085 | | |
| Fine AM hyphae (m/g) | 0.15 | 0.07 | 2.222 | 0.039 | | |
| ln (% SOM) | 0.72 | 0.39 | 1.854 | 0.080 | | |
| | | | | | 0.32 | 0.017 |
| ln (Litter mites / $m^2$) | | | | | | |
| Intercept | 8.89 | 2.81 | 3.164 | <0.01 | | |
| % Litter cover | 0.02 | 0.01 | 1.804 | 0.092 | | |
| Bulk density | -5.96 | 1.29 | -4.618 | <0.001 | | |
| % Clay | -0.23 | 0.09 | -2.695 | 0.017 | | |
| ln (% Bare ground) | -0.68 | 0.21 | -3.275 | <0.001 | | |
| ln ($NO_3$ µg/g) | -0.94 | 0.31 | -3.000 | <0.001 | | |
| % Wood cover | -0.03 | 0.01 | -1.739 | 0.104 | | |
| ln (% Forb cover) | 0.46 | 0.31 | 1.510 | 0.153 | | |
| | | | | | 0.82 | <0.001 |
| ln (Total mites / $m^2$) | | | | | | |
| Intercept | 13.64 | 0.89 | 15.364 | <0.001 | | |
| % Litter cover | 0.02 | 0.01 | 4.012 | <0.001 | | |
| Bulk density ($g/cm^3$) | -2.42 | 0.91 | -2.645 | 0.016 | | |
| % Clay | -0.12 | 0.06 | -1.891 | 0.074 | | |
| | | | | | 0.50 | <0.01 |
| ln (Soil collembolans / $m^2$) | | | | | | |
| Intercept | 12.38 | 1.67 | 7.472 | <0.001 | | |
| % Clay | -0.21 | 0.05 | -3.936 | <0.01 | | |
| ln (% Grass cover) | 0.43 | 0.12 | 3.684 | <0.01 | | |
| % Wood cover | 0.03 | 0.01 | 2.997 | <0.01 | | |
| ln ($NO_3$ µg/g) | 0.27 | 0.19 | 1.449 | 0.165 | | |
| | | | | | 0.67 | <0.001 |
| ln (Litter collembolans / $m^2$) | | | | | | |
| Intercept | 14.90 | 5.71 | 2.611 | 0.019 | | |
| Bulk density ($g/cm^3$) | -8.81 | 2.58 | -3.419 | <0.01 | | |
| ln (% Forb cover) | -1.97 | 0.61 | -3.227 | <0.01 | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| % Clay | -0.45 | 0.17 | -2.694 | 0.016 | | |
| Fine AM hyphae (m/g) | -0.43 | 0.16 | -2.672 | 0.017 | | |
| ln (% Bare ground) | -0.73 | 0.29 | -2.527 | 0.023 | | |
| ln ($NO_3$ µg/g) | -0.87 | 0.60 | -1.452 | 0.167 | | |
| | | | | | 0.62 | <0.01 |
| ln (Total collembolans/ $m^2$) | | | | | | |
| Intercept | 11.62 | 0.88 | 13.236 | <0.001 | | |
| % Litter cover | -0.02 | 0.01 | -2.295 | 0.034 | | |
| % Clay | -0.23 | 0.06 | -3.674 | <0.01 | | |
| ln (% Bare ground) | -0.39 | 0.14 | -2.820 | 0.011 | | |
| ln (% SOM) | 0.80 | 0.32 | 2.518 | 0.022 | | |
| | | | | | 0.62 | <0.001 |
| Nematodes / g dry soil | | | | | | |
| Intercept | -1.31 | 1.82 | -0.716 | 0.482 | | |
| ln (% SOM) | 2.78 | 1.00 | 2.783 | 0.012 | | |
| | | | | | 0.26 | 0.012 |

**Impacts of logging machinery on soil physical properties and nematodes**

The objective of this soils monitoring study was to determine the thresholds for impacts of logging machinery on soil nematodes and physical properties. Mechanized tree thinning causes considerable soil disturbance, but little information is available regarding thresholds for impacts on the dominant multicellular animals in soils: nematodes. These trophically diverse microfauna perform important ecological functions (for example, nitrogen mineralization, dispersal of microbes, and regulation of lower trophic levels) and are widely consumed by predatory and omnivorous mesofauna. Because nematodes are too small to physically modify soil pore structure, their movement is restricted to pores of sufficient size to accommodate them. Compaction from heavy logging machinery may thus reduce nematode abundances by decreasing available pore space. Nematodes also exhibit a wide range of life history characteristics, and some taxa are slow to recover from disturbance due to relatively long generation times and low reproductive output. We examined responses of nematode assemblages and soil physical characteristics to increasing number of passes (one, three, or nine) by a tracked harvester (a feller buncher) during thinning of a xeric mixed conifer forest at unit Seco 5 in VALL. Within and between the harvester tracks (**Fig. 11.2.2**), we measured soil surface penetration resistance and shear strength, quantified bulk density at four depth increments to a maximum depth of 27 cm, and characterized nematode assemblages in the upper 10 cm. We hypothesized that nematode responses to harvester traffic would vary according to their life history characteristics (i.e., whether they were *r*- or *K*-strategists *sensu lato,* as designated by the colonizer-persister (cp) classification system (Bongers, 1990)) and feeding habits, and that compaction would disproportionately affect large bodied taxa.

Eight months after treatment, we found that nematode communities were less impacted than soil physical properties by harvester passes (Gibson et al., 2023). Soil compaction was evident with a single pass and extended deep into the soil profile to at least 23-27 cm (**Fig. 11.2.3** and **Table 11.2.3**). Conversely, total nematode abundance was unaffected by any level of feller buncher disturbance, and sensitive *K*-selected nematode groups (cp 3-5 nematodes including Dorylaimida) were reduced only with nine passes (**Figure 11.2.4** and **Table 11.2.4).** We did not detect an impact of harvester disturbance on other nematode groups including herbivores, nematodes in the family Tylenchidae (which includes herbivorous and fungivorous taxa), *r*-strategist bacterivores, and *r*-strategist fungivores (**Table 11.2.4**). Results of this study suggest that nematode communities may be relatively resistant, and potentially resilient, to disturbance from heavy harvesters. If so, nutrient cycling, microorganism dispersal, and pest regulation services provided by nematode communities are unlikely to be impacted by low levels of logging machinery traffic. Nevertheless, our results emphasize the importance of minimizing areas subjected to harvester disturbance because of impacts to soil physical properties.



Fig. 11.2.2. One of three experimental feller buncher disturbance transects. Flags represent sampling points within and between the tracks (each cluster of three flags represents one sampling location). Each of the three transect blocks included three sections treated with 1, 3, and 9 passes, respectively. Samples and measurements were taken at three points per track and intertrack transect section (N=54 sampling locations).

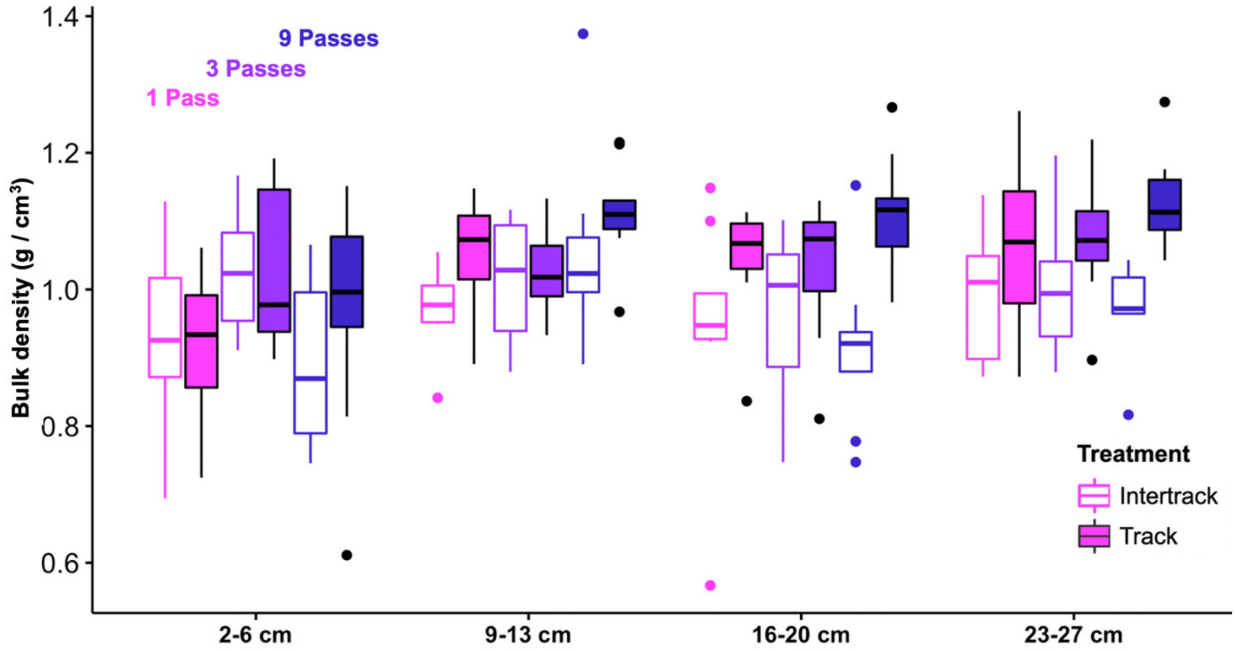



Fig. 11.2.3. Soil bulk densities (g/cm$^3$) from four depths beneath track and intertrack sampling locations that received 1, 3, or 9 passes (pink, blue and purple, respectively) from a feller buncher. Open boxplots show data from between the feller buncher tracks, and filled boxplots show data from the tracks. The horizontal line within each box shows the median for that treatment, and the lower and upper bounds of the box indicate the 25th and 75th percentiles, respectively. Whiskers extend to 1.5 x the interquartile range from the lower and upper box bounds, and outlier observations are plotted as individual dots.

Table 11.2.3. Non-parametric ANOVA-type analysis of bulk density; ATS = ANOVA-type test statistic; df = numerator degrees of freedom; passes = number of passes applied to the transect segment (1, 3, or 9).

| **Bulk density** | | | |
|---|---|---|---|
| | df | ATS | *p* |
| Passes | 1.99 | 1.21 | 0.298 |
| Treatment (track vs. intertrack) | 1.00 | 15.99 | < 0.001 |
| Depth | 2.52 | 6.70 | < 0.001 |
| Passes * Treatment | 1.93 | 2.12 | 0.122 |
| Depth * Treat | 2.62 | 3.06 | 0.033 |
| Passes* Depth | 4.69 | 1.91 | 0.094 |
| Passes * Treatment * Depth | 4.45 | 0.45 | 0.795 |

Table 11.2.4. Non-parametric ANOVA-type analysis of nematode abundance; ATS = ANOVA-type test statistic; df = numerator degrees of freedom; passes = number of passes applied to the transect segment (1, 3, or 9).

| **Total nematodes** | df | ATS | *p* |
|---|---|---|---|
| Passes | 1.97 | 0.66 | 0.513 |
| Treatment (track vs. intertrack) | 1.00 | 2.04 | 0.154 |
| Passes * Treatment | 1.99 | 1.93 | 0.145 |
| **cp2 bacterivores** | | | |
| Passes | 2.00 | 0.09 | 0.911 |
| Treatment (track vs. intertrack) | 1.00 | 1.84 | 0.175 |
| Passes * Treatment | 1.95 | 3.88 | 0.022 |
| **cp2 fungivores** | | | |
| Passes | 1.97 | 1.77 | 0.172 |
| Treatment (track vs. intertrack) | 1.00 | 10.16 | 0.001 |
| Passes * Treatment | 1.63 | 0.85 | 0.407 |
| **Total cp3, cp4, and cp5 nematodes** | | | |
| Passes | 1.93 | 1.49 | 0.227 |
| Treatment (track vs. intertrack) | 1.00 | 3.40 | 0.065 |
| Passes * Treatment | 1.92 | 3.77 | 0.025 |
| **cp3 and cp4 bacterivores** | | | |
| Passes | 1.77 | 0.26 | 0.747 |
| Treatment (track vs. intertrack) | 1.00 | 5.49 | 0.019 |
| Passes * Treatment | 1.84 | 4.92 | 0.009 |
| **Dorylaimida** | | | |
| Passes | 2.00 | 1.99 | 0.137 |
| Treatment (track vs. intertrack) | 1.00 | 3.21 | 0.073 |
| Passes * Treatment | 1.84 | 3.93 | 0.023 |
| **Herbivores** | | | |
| Passes | 1.80 | 2.65 | 0.077 |
| Treatment (track vs. intertrack) | 1.00 | 0.60 | 0.439 |
| Passes * Treatment | 1.63 | 1.01 | 0.353 |
| **Tylenchidae** | | | |
| Passes | 1.98 | 0.25 | 0.779 |
| Treatment (track vs. intertrack) | 1.00 | 2.17 | 0.141 |
| Passes * Treatment | 1.69 | 0.45 | 0.606 |

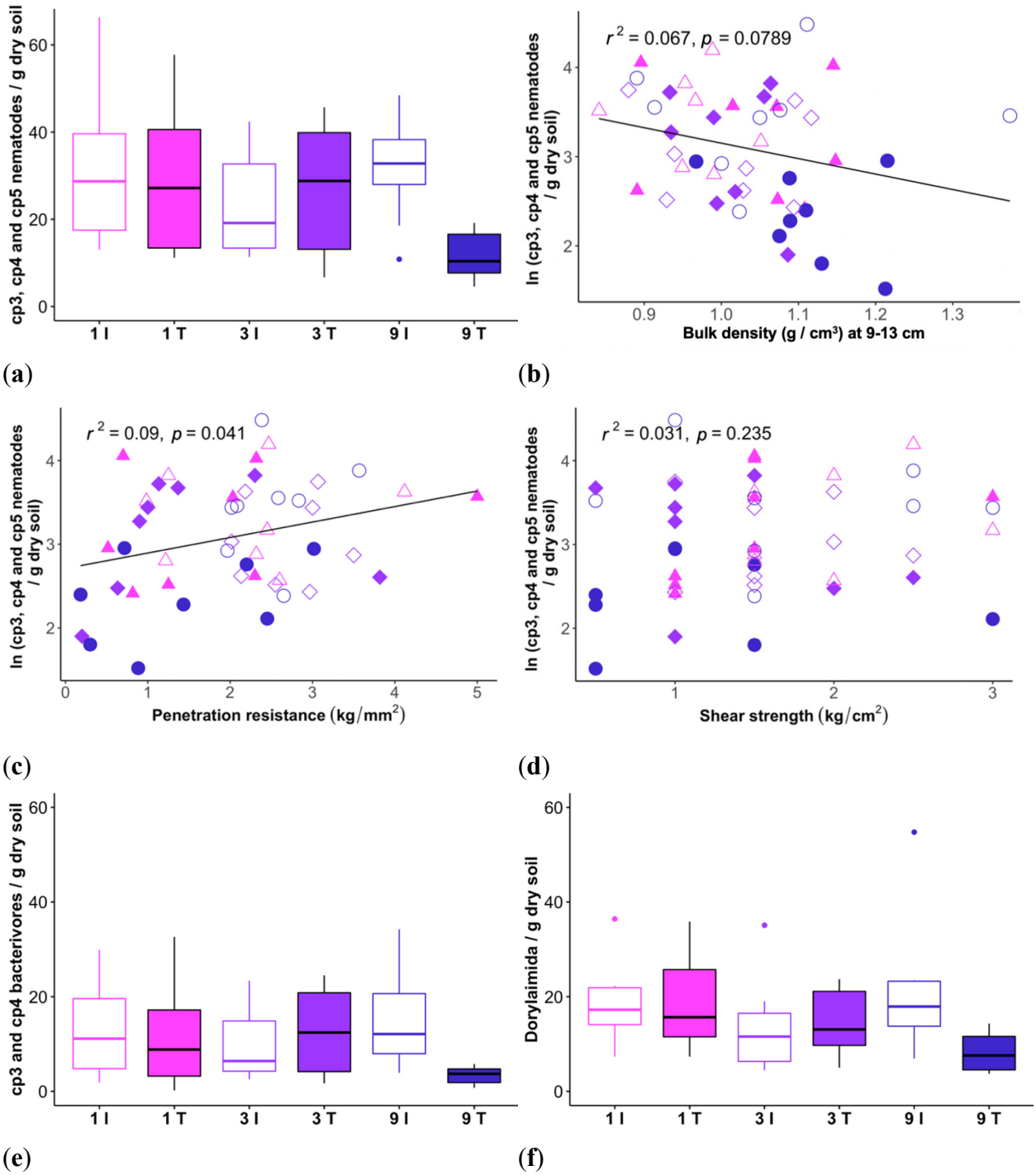



Fig. 11.2.4. Total abundances of cp3, cp4, and cp5 nematodes in feller buncher tracks (T) that received one, three, or nine passes, and matched intertrack (I) sampling locations (a); and correlations between total cp3, cp4, and cp5 nematodes and (b) bulk density at 9-12 cm, (c) surface penetration resistance, and (d) soil shear stress. (e) Harvester disturbance treatment responses of total cp3 and cp4 bacterivores. (f) Responses of Dorylaimida to harvester traffic. The horizontal line within each box shows the median for that treatment, and the lower and upper bounds of the box indicate the 25$^{th}$ and 75$^{th}$ percentiles, respectively. Whiskers extend to 1.5 x the interquartile range from the lower and upper box bounds. Outlier observations are plotted as dots.

**Modulation of the Gadgil effect by soil mesofauna in thinned/burned and untreated ponderosa pine forests**

Soil mesofauna likely influence the distribution and associated functions of fungal communities, but few manipulative studies have parsed the direct and indirect contributions of mesofauna to fungal-mediated ecosystem processes in the field. We used mesocosms of novel design to untangle relationships among microarthropods, fungal communities, and decomposition of labile (cellulose) and recalcitrant (wood) substrates in contrasting ecological contexts: thinned/burned and untreated ponderosa pine forest management units (Fig. 11.2.5 and Fig. 11.2.6) at Banco North in VALL. Our study took place over two growing seasons and concluded five years after the prescribed fire in the thinned/burned management unit. Our mesocosms were engineered to manipulate microarthropod communities via mesh treatments while minimizing—and enabling measurement of—mesh treatment side effects. Because fungivorous microarthropods may preferentially graze saprotrophic hyphae over ectomycorrhizal hyphae, we hypothesized that they could influence decomposition indirectly by modulating the Gadgil effect (where ectomycorrhizal fungi decelerate decomposition by outcompeting saprotrophic fungi for nitrogen). This experiment also provided an opportunity to test the resource-ratio theory-based prediction of (Smith and Wan, 2019) that the Gadgil effect should occur in recalcitrant—but not labile—substrates. We anticipated, however, that decomposition of a labile substrate would be directly increased by comminution (or fragmenting of substrates) by microarthropods. At the conclusion of the study, we extracted microarthropods that had colonized the mesocosms and used multi-group structural equation modeling (SEM) to quantify direct and indirect effects of mesofauna on decomposition.

Our mesocosms successfully manipulated microarthropods > 150 μm for fifteen months (Fig. 11.2.7) but were only partially effective at manipulating microarthropods < 150 μm (Table 11.2.5). Mesocosms in the untreated unit were colonized by more microarthropods, and had more morphospecies > 300 μm, than those in the thinned/burned unit (Fig. 11.2.7). SEM indicated that medium and large microarthropods (> 150 μm) increased the ratio of ectomycorrhizal (EcM) fungi to litter and wood saprotrophic (LWS) fungi, but only in the thinned/burned management unit. This EcM:LWS ratio was the best predictor of wood decomposition in the thinned/burned forest, where higher ratios were correlated with reduced decomposition, consistent with the Gadgil effect (Fig. 11.2.8). However, we found no evidence of the Gadgil effect in the untreated forest, despite higher EcM:LWS ratios and higher densities of microarthropods there. Decomposition of the labile substrate was unaffected by the EcM:LWS ratio, as resource-ratio theory would predict. We also observed no effect of microarthropods on decomposition of the labile substrate, a result consistent with other reports from moisture-limited systems. Our findings indicate that the Gadgil effect may play out differently in contrasting ecological theaters, with decomposition rates determined by the stage (abiotic context and substrate recalcitrance) as well as the cast (communities of microarthropods and fungi; Fig. 11.2.9). Our results also suggest that forest restoration treatments might increase the functional importance of microarthropods, despite reducing their abundance.

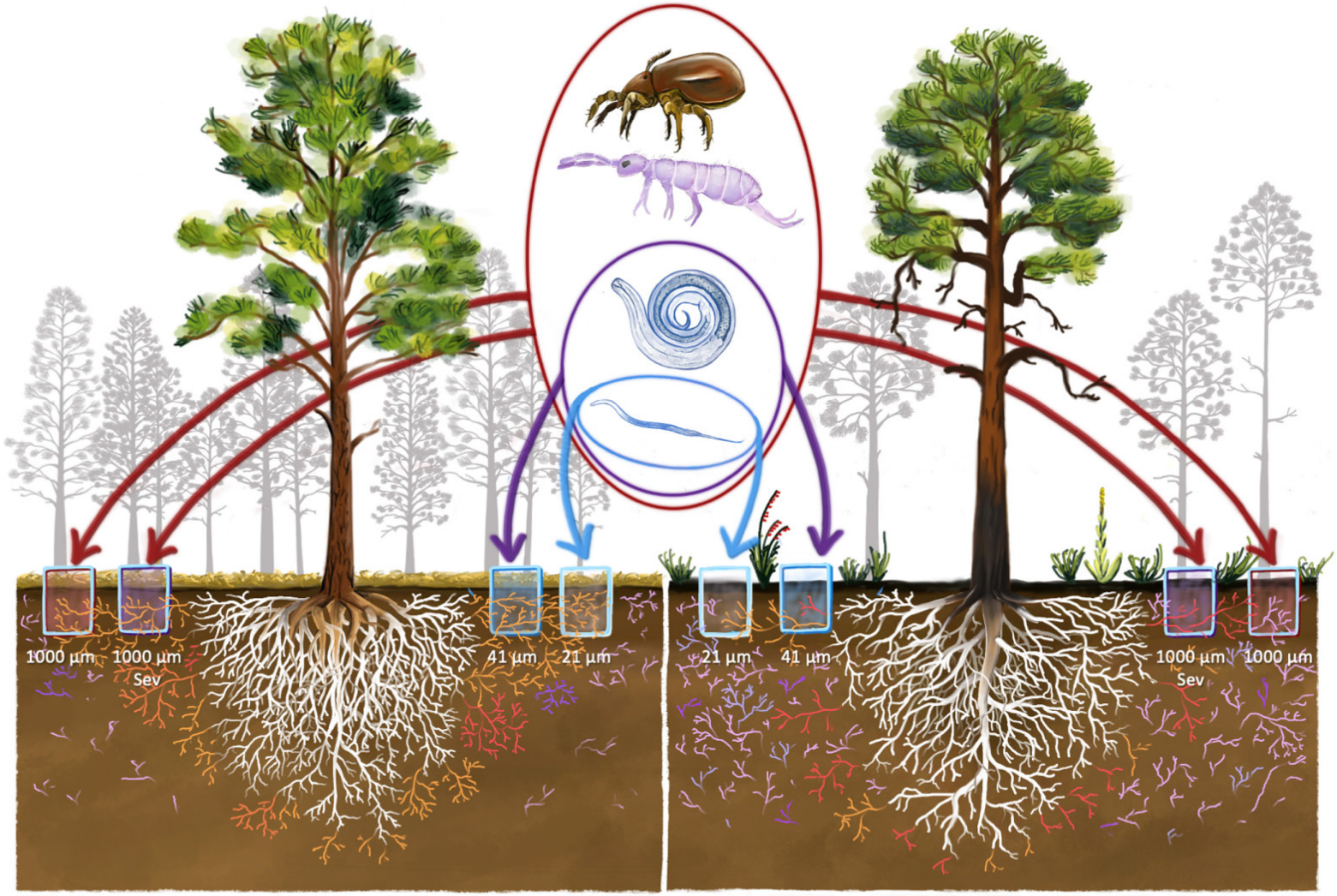



Fig. 11.2.5. Overview of our field mesocosm treatments. At ponderosa pine driplines in untreated (left) and thinned/burned (right) forest management units, we installed mesocosms designed to allow colonization by microfauna only (21 µm mesh and 41 µm mesh) or by microfauna and mesofauna (1000 µm mesh). Because roots could access the 1000 µm mesh mesocosms, we also included a fourth treatment of 1000 µm mesh treatment with root severing (1000 µm Sev).



Fig. 11.2.6. Experimental and sacrificial mesocosms at the dripline of a tree in the thinned/burned management unit. Other trees in the study, flagged with orange tape and marked with stars, are visible in the background.

Table 11.2.5. Results of two-way ANOVA. Effects of mesh treatment, forest management unit (untreated or thinned/burned (TB)), and mesh x forest management unit interactions on soil mesofauna abundances (individuals per core sample) and soil abiotic properties. Where *F*-tests justified the inclusion of an interaction term, the management unit x mesh results are listed below the main effects results. Abundances of mites and collembolans, percent soil organic matter, total available nitrogen, and ammonium content were log transformed prior to analysis.

| | *Intercept* | | *Management treatment: TB* | | *Mesh treatment: 21 µm* | | *Mesh treatment: 41 µm* | | *Mesh treatment: 1000 µm Sev* | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Estimate (Std. err) | *t p(t)* | Estimate (Std. err) | *t p(t)* | Estimate (Std. err) | *t p(t)* | Estimate (Std. err) | *t p(t)* | Estimate (Std. err) | *t p(t)* |
| **Mites >150 µm** | 4.772 (0.259) | 18.424 $p<0.001$ | **-1.846 (0.232)** | **-7.968 $p<0.001$** | **-1.879 (0.331)** | **-5.675 $p<0.001$** | **-1.319 (0.328)** | **-4.026 $p<0.001$** | -0.206 (0.328) | -0.630 $p=0.530$ |
| Adj. $R^2$: 0.528 $F=26.96$ (4, 89); $P<0.001$ | | | | | | | | | | |
| **Collembolans >150 µm** | 3.248 (0.287) | 11.320 $p<0.001$ | **-1.787 (0.257)** | **-6.961 $p<0.001$** | **-1.034 (0.367)** | **-2.818 $p=0.006$** | -0.314 (0.363) | -0.865 $p=0.389$ | 0.251 (0.363) | 0.690 $p=0.492$ |
| Adj. $R^2$: 0.384 $F=15.47$ (4, 89); $P<0.001$ | | | | | | | | | | |
| **Mites <150 µm** | 5.343 (0.327) | 16.335 $p<0.001$ | 0.410 (0.473) | 0.867 $p=0.388$ | 0.041 (0.463) | 0.090 $p=0.929$ | 0.092 (0.463) | 0.200 $p=0.842$ | -0.190 (0.463) | -0.410 $p=0.682$ |
| *Interaction (TB x mesh):* | | | | | **-1.952 (0.669)** | **-2.918 $p=0.004$** | *-1.165 (0.662)* | *-1.760 $p=0.082$* | -0.566 (0.662) | -0.855 $p=0.395$ |
| Adj. $R^2$: 0.133 $F=3.042$ (7, 86); $P=0.007$ | | | | | | | | | | |
| **Collembolans <150 µm** | 2.387 (0.339) | 7.044 $p<0.001$ | **-1.122 (0.490)** | **-2.289 $p=0.025$** | **-2.061 (0.479)** | **-4.301 $p<0.001$** | -0.009 (0.479) | -0.019 $p=0.985$ | 0.112 (0.479) | 0.233 $p=0.816$ |
| *Interaction (TB x mesh):* | | | | | 0.942 (0.693) | 1.359 $p=0.178$ | -0.248 (0.685) | -0.362 $p=0.718$ | 0.190 (0.685) | 0.277 $p=0.783$ |
| Adj. $R^2$: 0.323 $F=7.339$ (7, 86); $P<0.001$ | | | | | | | | | | |
| **Soil organic matter (%)** | 1.74 (0.08) | 22.64 $p<0.001$ | **-0.16 (0.07)** | **-2.27 $p=0.026$** | 0.01 (0.10) | 0.09 $p=0.932$ | -0.03 (0.10) | -0.26 $p=0.795$ | 0.00 (0.10) | -0.04 $p=0.967$ |
| Adj. $R^2$: 0.014 $F=1.326$ (4, 88); $P=0.267$ | | | | | | | | | | |
| **Mean soil moisture Z score** | -0.59 (0.18) | -3.32 $p=0.001$ | 0.04 (0.26) | 0.14 $p=0.886$ | **1.58 (0.25)** | **6.28 $p<0.001$** | **1.08 (0.25)** | **4.30 $p<0.001$** | **0.66 (0.25)** | **2.61 $p=0.011$** |
| *Interaction (TB x mesh):* | | | | | **-0.90 (0.36)** | **-2.47 $p=0.016$** | -0.21 (0.36) | -0.58 $p=0.561$ | -0.30 (0.36) | -0.83 $p=0.406$ |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Adj. $R^2$: 0.370 $F$=8.804 (7, 86); $P$<0.001 | | | | | | | | | | |
| **Total available N (mg N $g^{-1}$ dry soil)** Adj. $R^2$: 0.001 $F$=1.02 (4, 89); $P$=0.402 | 1.78 (0.19) | 9.46 $p$<0.001 | -0.16 (0.17) | -0.95 $p$=0.344 | -0.04 (0.24) | -0.15 $p$=0.881 | 0.15 (0.24) | 0.65 $p$=0.517 | 0.35 (0.24) | 1.46 $p$=0.149 |
| **Ammonium (mg N $g^{-1}$ dry soil)** Adj. $R^2$: 0.000 $F$=1.011 (4, 89); $P$=0.406 | 1.73 (0.19) | 8.96 $p$<0.001 | -0.22 (0.17) | -1.25 $p$=0.214 | 0.00 (0.25) | 0.02 $p$=0.985 | 0.18 (0.24) | 0.76 $p$=0.452 | 0.33 (0.24) | 1.35 $p$=0.179 |

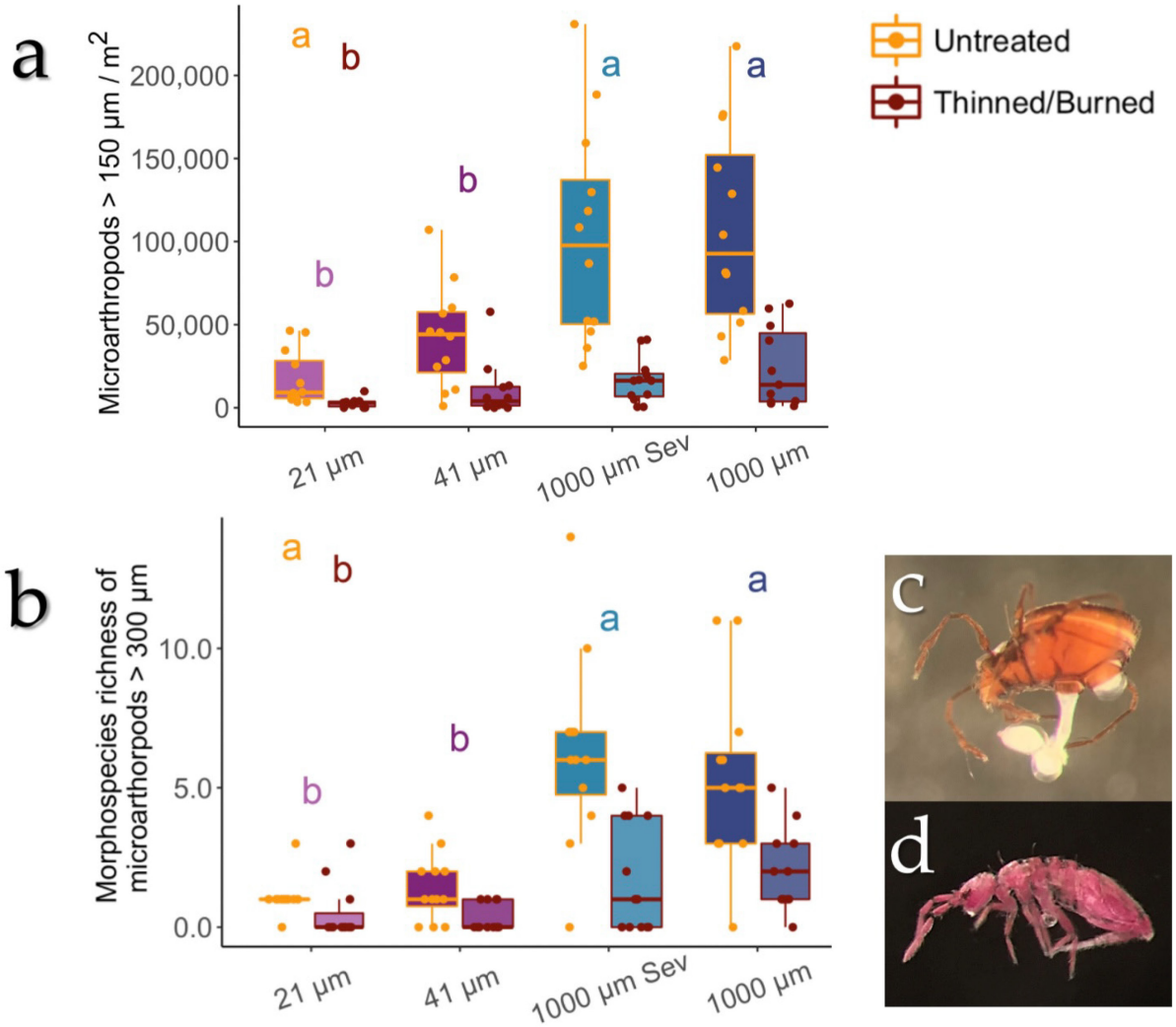



Fig. 11.2.7. Abundances of microarthropods > 150 µm (a) and morphospecies richness of microarthropods > 300 µm (b). Contrasting letters indicate significant group differences according to two-way ANOVA at $p<0.05$ after adjustment for multiple pairwise comparisons using Tukey's method (gold and red letters indicate forest management treatment differences, and purple and blue letters indicate mesh treatment differences). The horizontal line within each box indicates the median, the lower and upper bounds of the box correspond to the 25$^{th}$ and 75$^{th}$ percentiles, and whiskers show the minimum and maximum values that do not exceed 1.5 x the interquartile range from lower and upper box bounds. (c) *Odontodamaeus* sp. (Oribatida: Gymnodamaeidae) caught in the act of oviposition, and (d) an Entomobryidae sp. (Entomobryomorpha), two of the morphospecies encountered.

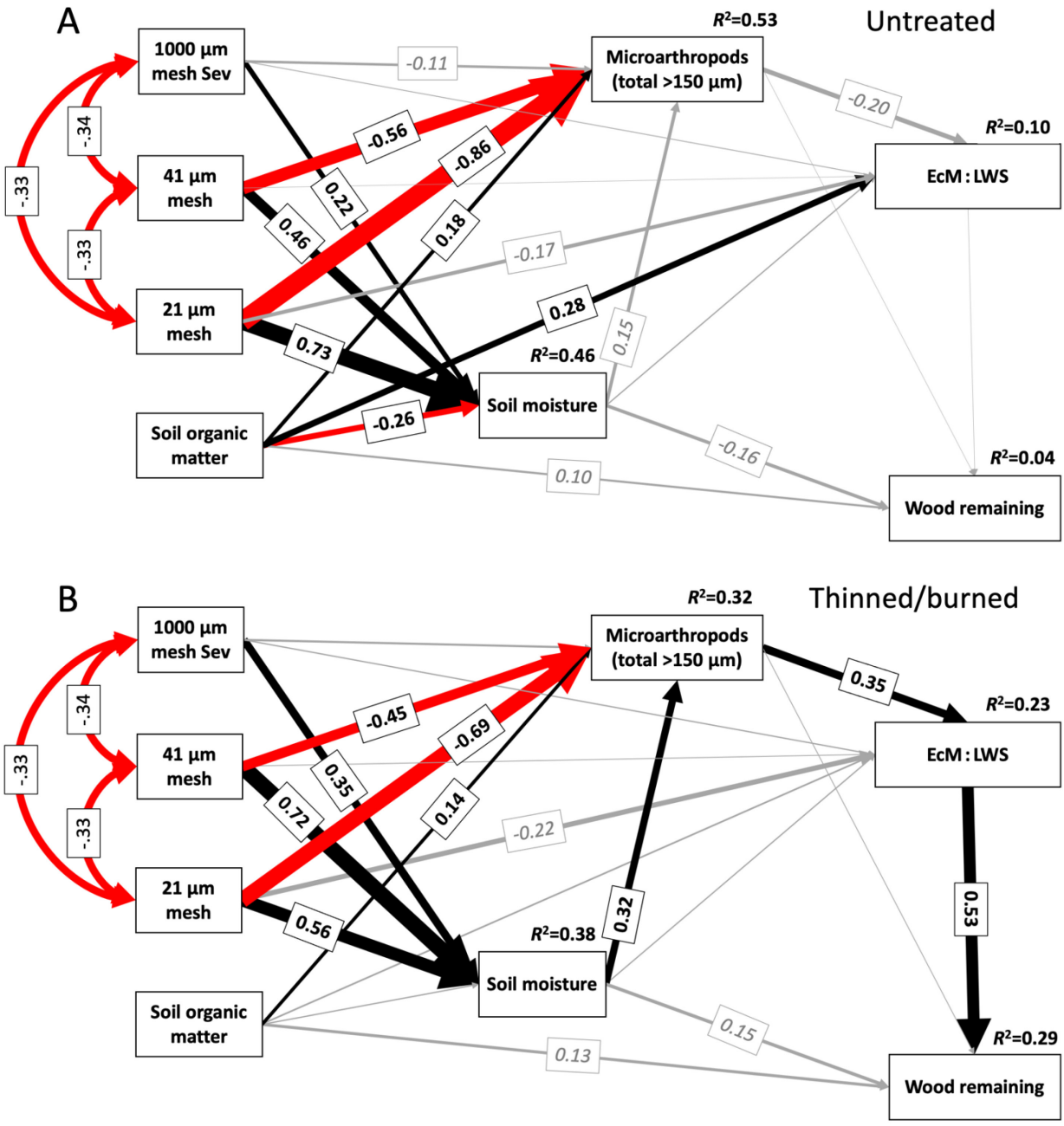



Fig. 11.2.8. Multigroup structural equation model of wood decomposition. (A) Untreated management unit. (B) Thinned/burned management unit. Soil moisture=mean Z scores of soil moisture measurements taken at five timepoints. Significant standardized path coefficients (λ) with $p < 0.05$ are shown in bolded black text; all other path coefficients are shown in gray italics where $\lambda \geq 0.1$ or are omitted if $\lambda < 0.1$. Black arrows indicate significant positive paths, red arrows represent significant negative paths. Arrow widths are scaled according to the standardized path coefficients. EcM:LWS = ratio of ectomycorrhizal reads to litter and wood saprotroph reads. Percent soil organic matter, microarthropod abundances, and EcM:LWS were log transformed prior to analysis.

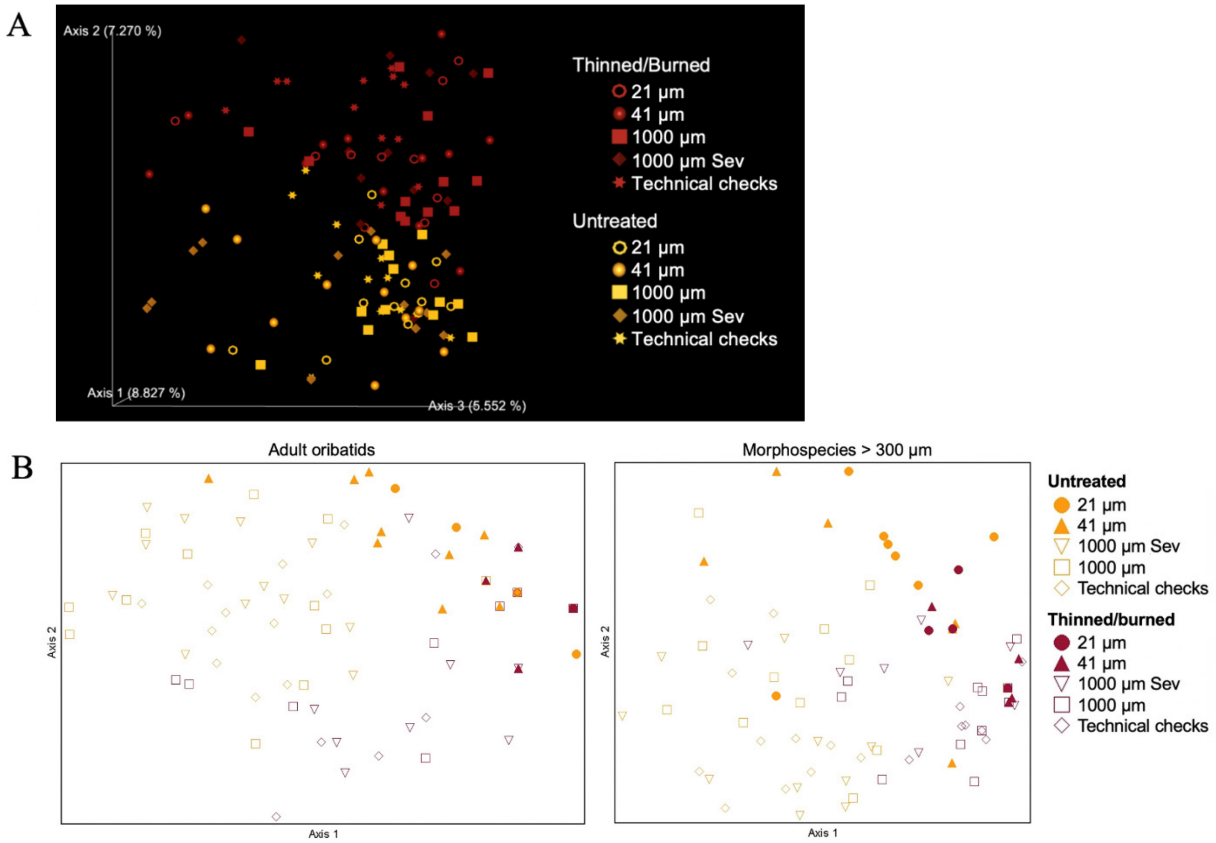



Fig. 11.2.9. Ordinations of fungal and faunal communities at the conclusion of the field mesocosm study. (A) Variation along axes two and three of a three-dimensional principal coordinate analysis (PCoA) ordination of fungal communities based on weighted Bray-Curtis distance, visualized with Emperor QIIME2View. Colors correspond to ponderosa restoration treatment, and shapes designate mesh treatment: 21 µm, 41 µm, 1 mm with root severing (1 mm Sev), 1 mm without root severing (1 mm), and technical checks without mesh or pipe. (B) Nonmetric multidimensional scaling (NMDS) ordinations of (left) adult oribatid mites > 150 µm (final stress = 9.37; two-dimensional ordination) and (right) abundances of all microarthropod morphospecies > 300 µm (final stress=12.21; axes 1 and 2 shown of three-dimensional ordination). Note that mesh treatments are equally replicated (N=12), but many of the fine mesh mesocosms with few or no large fauna or oribatids are plotted atop one another in the NMDS.

## Exploring the potential of fire mosses as a Burned Area Emergency Response treatment

In an effort to determine the potential of fire mosses as a Burned Area Emergency Response (BAER) restoration treatment, a field experiment on the Cajete Fire was set up in November of 2017 by Henry Grover.

Land managers use BAER hillslope treatments to mitigate devastating post-fire impacts and protect values at risk. However, the effectiveness of BAER seeding hillslope treatments have been debated in the literature and by managers. Perhaps a focus on early successional, functionally different species could help increase seeding success. With wildfires increasing in extent and severity through the Western US, land managers need new tools to facilitate post wildfire soil stabilization as well as ecosystem recovery.

One unexplored BAER seeding approach is using the fire increasing, early successional mosses *Ceratodon purpureus*, *Funaria hygrometrica*, and *Bryum argentum* as hillslope treatments. These so-called fire mosses have functional differences which could complement existing seed mixes and mulching efforts: (1) they are desiccation tolerant, (2) they can be propagated and dispersed vegetatively, (3) they can aggregate disturbed soils, reducing erosion and increasing infiltration, and (4) fire mosses have global distributions and are native plants virtually everywhere. Thus, they may be a valuable restoration material throughout the world.

With this study we explored 3 major research questions:

1. What is fire mosses' potential to stabilize and restore severely burned soils in Valles Caldera National Preserve (VC)? We hypothesized that inoculation of fire moss gametophytes would increase colonization of moss, vascular plant, and soil microbial colonization of severely burned soils and could rapidly improve post-fire ecosystem stability and recovery when compared with uninoculated controls.
2. Can we improve fire moss colonization by applying it in a pelletized form (i.e., by mixing moss with diatomaceous earth and organic matter and rolling it into pellets before applying it to the soil)? We hypothesized that mixing moss into pellets would improve colonization by increasing the connection between the soil and moss and reducing the sensitivity of moss to climatic fluctuations.
3. How will ex-situ growth affect fire moss function in the field? To understand this impact, we inoculated plots with either field collected moss or greenhouse grown moss in every block. We hypothesized that ex-situ growth would not have a large impact on fire moss function but could affect soil microbial communities dramatically.

### Methods

*Study Area and Plot Selection*: The study was set up on the northern edge of the Cajete Fire on the Santa Fe National Forest just north of Highway 4 and South of Valles Caldera. We selected a roughly 200 m x 200 m patch of high burn severity where 100% overstory tree mortality had occurred (Figure 11.3.3.1, Lat 35.818208 Lon -106.554691). This wildfire allowed us to test the effectiveness of inoculating moss in a post-fire setting similar to a real BAER treatment. Sixty 1 m x 1 m plots were marked with rebar in two opposing corners. Inoculation occurred on 11/4/2017 roughly 4 months after the wildfire. There was some initial colonization of moss prior to inoculation during the monsoon season; these patches, and locations downslope of them were avoided during plot setup.

*Treatments*: Using a completely randomized design we inoculated levels of: greenhouse grown moss inoculation, field collected moss inoculation, two concentrations of greenhouse grown moss rolled into pellets, and no inoculation (Fig. 11.2.10). Variability of fire moss colonization from a previous study was used to conduct a power analysis, allowing us to determine adequate replication for this experiment. Treatments were replicated 12 times for a total of 60 experimental units, and a study wide ground impact of 60 $m^2$. Greenhouse grown moss and field collected moss were dried, ground through a sieve, and inoculated at an initial cover of 10%. The greenhouse grown pellet treatments were also inoculated at a cover of 10% as well as a cover of 50%.

*Measurements*: Sampling of plots occurred every Spring and Fall for two years post-inoculation. Cover estimates of plant functional type, moss to species, bare ground, rock, and litter were made using standardized ocular estimates. Moss colonies to species were counted in every plot, allowing us to measure inoculation survivorship at a higher resolution. Infiltration (3 per plot) and soil aggregate stability (4 per plot) measurements were taken at randomly generated positions on every plot in fall 2019.

**Results**

Pelletization at 50% cover was the most successful treatment with *B. argenteum* reaching a maximum mean cover of 10.5% in June 2019 ($F = 2.73$, $p = .046$, Fig. 11.2.11.A). The sieve and pellet 10% treatments had similar cover values throughout the experiment and performed only slightly better than the controls (Fig. 11.2.11.A). Natural recruitment on control plots of *B. argenteum* was high, reaching a maximum cover of 6.8% in June 2019. Moss cover at the first monitoring visit, six months after inoculation, was undetectable on any plot; however, pellets were still visible. This was most likely due to the exceptional drought that occurred over the winter of 2017/2018 (Fig. 11.2.11.C). As treatment cover did not strongly diverge from controls in this experiment, we did not see effects of moss cover on infiltration ($F = .18$, $p = .67$) or soil aggregate stability ($F = .46$, $p = .50$). *C. purpureus* was detected on one pelletized and one sieve plot at the final monitoring visit. No *F. hygrometrica* was found at any time.

The number of colonies for all treatments increased through time (Fig. 11.2.11.B, $F = 3.96$, $p = .01$). In spring 2018 there had been so little precipitation that pellets were intact, and diatomaceous earth was still visible making it difficult to assess colony counts. Strong monsoon precipitation in 2018 lead to a large increase in colony counts by the fall 2018 monitoring. The colony count number plateaued from fall 2018 to spring 2019; however, there was a relatively large increase in cover during that time period (Fig. 11.2.11.A).



Figure 11.2.10. Fire moss plots post inoculation. White squares are plots inoculated with pellets.

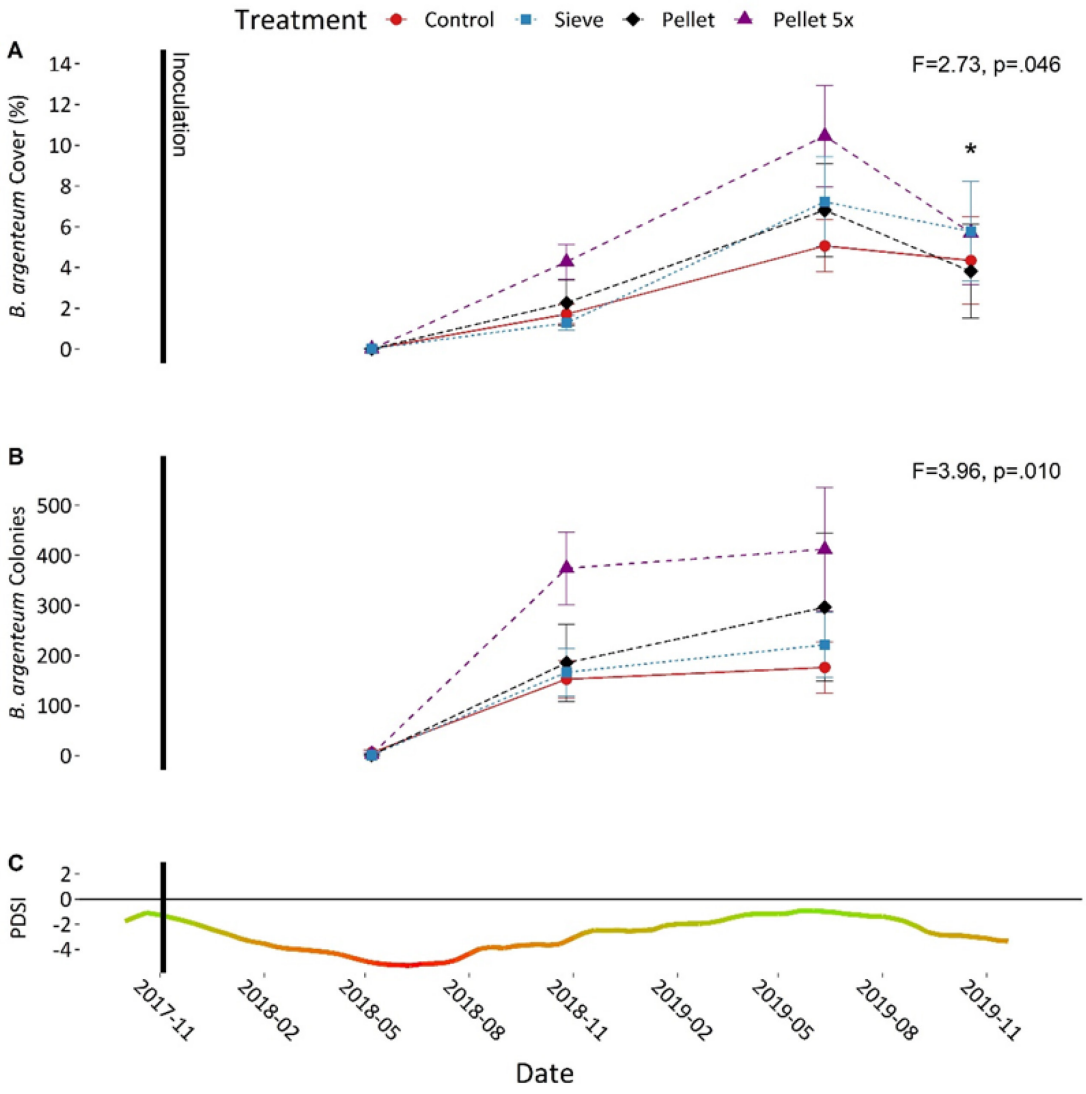



Figure 11.2.10. *Bryum argenteum* (A) cover and (B) colony count at monitoring visits. Vertical line indicates date of inoculation. Error bars represent ±SE. *Cover methodology changed, and no colony count data was taken in final monitoring visit due to extensive vascular plant colonization. (C) Palmer drought severity index (PDSI) throughout the experiment, negative values indicate drought while positive indicate moisture.

**Summary of other non-CFLRP financial support**

*Undergraduate Funded grants:*

Justine Baca, How do diatoms affect soil microarthropod community assembly. NASA (2021, $1,200).

Mildred Diaz. The Effects of Forest Management Practices and Feedbacks between Plants and Soil Organisms. HURA and NASA (2018, $4,000).

Channing Laturno, Effects of thinning and burning on microarthropods. NAS. (2017, $2,500).

*General grants:*

Antoninka and Gibson. Assessing the contribution of soil faunal complexity to ecosystem services after restoration with thinning and fire. JFS-GRIN to PhD student, K. Gibson 2018-2020 ($25,000).

Antoninka and Gibson. Wildfire and land management impacts on habitat and survival of Jemez Mountains salamanders: Quantifying immediate and long-term ecosystem and soil responses to inform endangered species stewardship and forest restoration. NRPP. 2022-2025. ($11,643)

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Gibson, K.S., Johnson, N.C., Laturno, C., Parmenter, R.R., Antoninka, A., 2022. Abundance of mites, but not of collembolans or nematodes, is reduced by restoration of a Pinus ponderosa forest with thinning, mastication, and prescribed fire. *Trees, Forests and People* 100190.

Gibson, K.S., Neher, D.A., Johnson, N.C., Parmenter, R.R., Antoninka, A.J., 2023. Heavy Logging Machinery Impacts Soil Physical Properties More than Nematode Communities. *Forests* 14, 1205.

Smith, G.R., Wan, J., 2019. Resource-ratio theory predicts mycorrhizal control of litter decomposition. *New Phytol.* 223, 1595–1606.

# Chapter 12. Monitoring for Threatened and Endangered Species

**Collaborators:** Robert R. Parmenter, Mark Peyton, Steve Zanoni, Tyler Obermeit – *National Park Service, Valles Caldera National Preserve;* Andre Silva, Leslie Welch – *US Forest Service, Santa Fe National Forest;* Trevor Fetz, Gail Garber – *Hawks Aloft Inc.;* Carol Chambers – *Northern Arizona University.* Nancy Karraker – *University of Rhode Island;* Rachel Loehman – *US Geological Survey.*

**Summary:** The SWJM CFLRP project area supported three species listed by the US Fish and Wildlife Service as Threatened or Endangered under the Endangered Species Act. These included the Jemez Mountains Salamander (*Plethodon neomexicanus*) [Endangered], the Mexican Spotted Owl (*Strix occidentalis lucida*) [Threatened], and the New Mexican Meadow Jumping Mouse (*Zapus hudsonius luteus*) [Endangered]. Two other species have recently been added to the potential list of protected species within the SWJM CFLRP project area, but neither has as yet been observed to have permanent populations on our sites. These include the Endangered subspecies of Mexican wolf (*Canis lupus baileyi*) and the Candidate species monarch butterfly (*Danaus plexippus*). Monitoring surveys have thus far indicated that restoration activities have had little direct effect on any protected species, and that habitat improvements should promote continued occupancy in existing habitats through reduction in stand-replacement fire and provide additional suitable habitat for population expansions.

### Jemez Mountains Salamander (*Plethodon neomexicanus*)

This species of terrestrial salamander inhabits coniferous forests in the SWJM CFLRP project area, with a preference for mixed conifer stands in mesic environments. The Jemez Mountains salamander spends most of the year underground, emerging during the summer monsoons for surface activities (typically at night when humidities are higher). The major threat to this species is stand-replacement wildfires that remove forest canopy and desiccate the soil surface, reducing the habitability of the burned area. As such, the SWJM CFLRP forest restoration actions will benefit the species in the long term by reducing the frequency and areal extent of stand-replacing fires, allowing the continued existence of mixed conifer forests in the Jemez Mountains. However, the collateral disturbance of thinning the forest using machinery or hand crews, including piling of logs for eventual burning, coupled with post-thinning broadcast fires, runs the risk of injuring or killing individual salamanders in the treated areas. To reduce these risks, restoration activities (thinning and burning) were restricted within salamander habitat to late fall (post-November 1) through the winter/spring when the salamanders were underground.



Figure 12.1. Jemez Mountains salamander. Photo: Mark Watson.

Surveys for this salamander involve searching beneath logs and rocks for resident salamanders. Jemez Mountains salamanders are notoriously difficult to find and require extensive survey effort to assess occupancy in any given area. Surveys for the salamanders were conducted in the SWJM CFLRP project area during 2018-2024, and over these 7 years, only 152 salamanders were observed (2018 = 7; 2019 = 26; 2020 = 30; 2021 = 40; 2022 = 32; 2023 = 8; 2024 = 7). These sample sizes were too small to establish population trends, but occupancy analyses are currently in progress. Details of survey efforts are as follows:

*2018 and 2019:*

In 2018 and 2019, we established 972 artificial cover objects (of three types - artificial logs, artificial rocks, artificial rock piles) distributed evenly among six sites (3 on the Preserve and 3 on the Santa Fe National Forest (SFNF)). We sampled those cover objects every other week during the months of July and August. Across the two years, we found only two salamanders beneath two of the cover objects. We ceased sampling the artificial cover object arrays after the 2019 season because of apparent lack of effectiveness of the artificial cover objects.

In alternate weeks, we: (1) conducted mark-recapture surveys in 50 x 50 m plots (three per site) at one site on the Preserve and two sites on SFNF, and (2) conducted two-person-hour time-constrained surveys at sites known to be historically occupied by Jemez Mountains salamanders.

*2019-2023:*

We conducted mark-recapture surveys in 50 x 50 m plots (three per site) at one site on the Preserve and two sites on SFNF every other week. Having conducted mark-recapture surveys on the same plots for four years, we ceased mark-recapture surveys in these plots after 2023 over concerns of potential habitat degradation due to: (1) alteration of microclimate and microhabitat associated with repeated turning of cover objects, and (2) erosion on steep slopes because of repeated visits to the same plots. We also conducted two-person-hour time-constrained surveys at 15 sites known to be historically occupied by Jemez Mountains salamanders along a burn severity gradient. Those sites included five unburned sites, five sites that burned at low severity, and five sites that burned at moderate/high severity. Each of the 15 sites were surveyed five times during the season.

*2024:*

Every other week, we conducted two-person-hour, time-constrained surveys at the same 15 sites along a burn severity gradient surveyed in 2023. Those sites included five unburned sites, five sites that burned at low severity, and five sites that burned at moderate/high severity. Each of the 15 sites were surveyed five times during the season.

In all years, we swabbed most salamanders captured for *Batrachochytrium dendrobatidis* (amphibian chytrid fungus). Samples are currently being batch-analyzed at a lab in North Carolina; we are awaiting results.

**Mexican Spotted Owl (*Strix occidentalis lucida*)**



Figure 12.2. Female Mexican Spotted Owl in the SWJM CFLRP project area. Photos: Trevor Fetz.

The SFNF conducted Protected Activity Center (PAC) monitoring for the Mexican Spotted Owl (MSO) as required by the Biological Opinion from FWS. There were seven MSO PACs originally within the SWJM CFLRP boundary and many of them had PAC monitoring conducted for over 5 years. A new PAC was discovered in 2021 when a Forest user contacted the SFNF. A second new PAC was recently discovered in 2023 by another Forest user. Both of these new PACs were unaffected by activities within the SWJM CFLRP but can now inform project implementation through time.

The SWJM CFLRP surveys greatly expanded our understanding of the complex life history of the MSO within the Jemez Mountains. It became clear through monitoring that the MSO relies heavily on canyon systems with crevices, rock walls, alcoves and scrapes in addition to mature, old growth mixed conifer. The monitoring set up the Santa Fe National Forest (SFNF) to manage wildfire, plan restoration projects and evaluate whether our activities were resulting in PAC abandonment. The SFNF has documented continued occupancy and reproduction in most of the PACs during the life of SWJLRP and now has a firm understanding of the types of habitats which can support the MSO and how they may be impacted by specific activities.

*Fogon Protected Activity Center Meadow Treatment* – In 2018, the Forest treated meadow habitat within the Fogon PAC. This PAC is located off FR 376 in a high use recreation corridor but the PAC is south of the busy road. The PAC contains a large meadow in the center. The meadow has been experiencing conifer encroachment effects and the meadow had been overgrown with small diameter (< 9" dbh) trees. Ladder fuels had fully developed and were threatening the overall condition of the PAC. The treatment included felling meadow encroaching conifers at various size classes and all trees over 18" were retained. Cut trees were left on site as downed wood and one side of the bole had branches cut so that the bole would achieve ground contact. Approximately 5 acres were completed. Prior to and after the treatment, the Fogon PAC has continued to be occupied by the MSO and they generally produce one to two fledglings each year. The SWJM CFLRP was able to show a case of treatments within PACS being acceptable to occupancy and reproduction for the MSO.

*Stable Canyon Prescribed Fire* – An MSO PAC prescribed fire was conducted in 2021 under the SWJLRP. Prior to ignition, the Forest collected common stand exam data as consistent with the biological opinion, and the R3 MSO Habitat Management Strategy. The District and Forest had conducted MSO PAC monitoring since 2017 and knew the location of nest sites (Figure 12.3) documented MSO reproduction, and biologists conducted hand ignition and habitat monitoring during the implementation of the prescribed fire. MSO PAC monitoring the spring after the fire found MSOs still occupying the PAC and attempting to breed. A year after the fire, a known nest site was evaluated for burning results in November 2022. The biologists found that the nest site was completely intact due to the nest site being canyon habitat and mostly comprised of rock and south aspect-fire adapted vegetation (12.4). The north aspect slope of this PAC contains prominent mixed conifer habitat which did not burn at desired condition despite multiple aerial

ignition attempts for the duration of prescribed fire (12.5). Downed wood was hardly consumed on the north aspect slope and high volumes of snags and downed wood remain.

*San Antonio Mountain thinning operation* – On Valles Caldera National Preserve, the southern unit (Unit 1) of San Antonio Mountain had been thinned in a cut-and-pile hand crew operation during the fall-winter of 2020-2021. In May, 2021, an owl surveyor observed a pair of sub-adult MSOs in mixed conifer forest on the boundary of the thinned unit (see Figure 12.2 for a photo of the subadult female). No nest was identified, and while the pair remained in the area for several weeks, they eventually disappeared and did not return. Additional surveys across the Preserve during 2020-2024 failed to identify any nesting pairs, but recorded several transient MSOs that were only detected on 1 or 2 nights.



Figure 12.3. Stable Canyon MSO PAC Nest Site pre - Rx fire. June 3, 2020.



Figure 12.4. Stable Canyon PAC nesting site - post prescribed burn conditions (showing low severity from October 2021 Rx fire



Figure 12.5. Stable Canyon Rx Fire 2021 - North aspect mixed conifer showing minimal fire effects to canopy despite connected ladder fuels.

**New Mexico Meadow Jumping Mouse (*Zapus hudsonius luteus*)**

The New Mexico Meadow Jumping Mouse occurs in several watersheds within the SWJM CFLRP project area. Surveys have identified individuals in two watersheds on the Preserve (Jaramillo Creek and Redondo Creek) and in 9 locations along 3 streams on the Santa Fe National Forest (Rio Cebolla, Rio San Antonio, Rio Guadalupe). Populations are apparently patchy along these streams, as many surveys failed to detect jumping mice.

Habitat improvements for this species are being undertaken as part of the stream/wetland restoration activities (see Chapter 5). Activities include planting willows along streambanks and installing various structures to raise water tables and increase riparian wetlands. Among the structures installed in stream reaches that have known jumping mouse populations, Beaver Dam Analogs (BDAs) have been shown to not only successfully restore streambanks with small ponds behind the BDAs, but also have no apparent negative effect on the jumping mouse. Jumping mice have been recorded in these areas before and after installation of the BDAs; Figure 12.7 shows a jumping mouse foraging at the base of one of the posts used to anchor the BDA to the bank.



Figure 12.6. New Mexico Meadow Jumping Mouse. Photo: NPS



Figure 12.7. New Mexico Meadow Jumping Mouse (in yellow circle) foraging at the base of a BDA post. Photo: USFS.

**Mexican wolf (*Canis lupus baileyi*)**



Figure 12.8. Photo of "Asha", the female Mexican wolf that dispersed into the Jemez Mountains. Photo: Bryan Ramsay.

The Mexican wolf has been reintroduced into southern Arizona and New Mexico as an experimental population, and a boundary for the wolf population has been established using Interstate 40 as the northern limit of the Mexican Wolf Experimental Population Area (MWEPA). This northern boundary is ~50 miles south of the SWJM CFLRP project area. As such, there are no permanent breeding populations of Mexican wolf in the CFLRP project area. However, there has been at least one transient Mexican wolf that dispersed out of the established MWEPA and entered the Jemez Mountains. A female wolf, known as "Asha" in the popular press (Figure 12.8), ventured into the CFLRP area in November 2023. This non-resident wolf resided on the Valles Caldera National Preserve and the northwest portion of the CFLRP on the Jemez and Coyote Ranger Districts of the Forest for several weeks. As consistent with the revised recovery plan for the Mexican wolf, wolves that leave the MWEPA are recaptured and relocated within the MWEPA. The female wolf was captured by FWS and the New Mexico Game & Fish Department on the Forest Service's Coyote Ranger District and relocated to the FWS Sevilleta National Wildlife Refuge wolf breeding facility in December 2023. It is unknown how or if the SWJM CFLRP restoration activities influenced wolf activity on the Forest and Preserve. No Mexican wolves are known to reside on the Forest or Preserve at this time, but it is likely that these instances of dispersing individuals will increase through time.

**Monarch butterfly (*Danaus plexippus*)**



Figure 12.9. The monarch butterfly. Photo: FWS

The monarch butterfly (Figure 12.9) is listed as a Candidate Species at this time (2024), due to lack of FWS personnel resources to adequately evaluate its status. There have been extensive surveys of butterflies on Valles Caldera National Preserve since 2003, but no monarch butterflies have ever been detected. This is likely due to the high elevations in the Jemez Mountains that are outside of the monarch's typical elevational range. However, with future climate warming (see Chapter 3), we anticipate that this species will eventually colonize the SWJM CFLRP project area. The forest restoration work is increasing the herbaceous vegetation

layer in thinned and burned forest stands, and assuming native milkweed plants (used for reproduction by monarchs) will be part of the plant community developing on restored forest sites, then these restored habitats will benefit the monarch population as it establishes occupancy. A pollinator monitoring program has been established in collaboration with the University of New Mexico's Museum of Southwestern Biology (Arthropod Division), and this program should be able to detect colonization in the future by monarch butterflies. In addition, the Santa Fe National Forest is pursuing participating in the River for Monarchs program and some sites restored under the CFLRP will be used for planting sites for milkweed.

# Chapter 13. Cultural Resources

**Collaborators:** Rebecca Baisden – *US Forest Service.* Ana Steffen, Nick Jarman, Monte White, David McCray – *National Park Service.* Rachel Loehman – *US Geological Survey.*

**Summary.** The SWJM CFLRP included considerable activities by Cultural Resource specialists in both the Forest Service and the National Park Service. These activities included pre-restoration survey work to meet compliance requirements of the National Historic Preservation Act (NHPA), as well as direct fuels assessments and reductions projects on cultural resource sites throughout the project area. These activities were funded by USFS and NPS as part of the cost-match requirements, as well as outside grants to evaluate restoration techniques in reducing fire risk on cultural resource sites. In total, pre-thinning and pre-burning cultural resource surveys were conducted on 26,000 acres, and fuels assessments and/or treatments were completed on 3,100 cultural resource sites (mostly ancient field houses and pueblo ruins). In addition, with funding from the Joint Fire Science Program and the USFS Rocky Mountain Research Station, archaeologists and fire specialists from the NPS and USFS evaluated drought as a key variable causing forest fires that result in a wide range of effects to archeological sites, historic structures, cultural landscapes, and traditional cultural places. They assessed impacts to archeological resources seen in the Jemez Mountains over the past forty years. This interdisciplinary effort provided the SWJM CFLRP with insights on archeological fire effects and the fuels and fire environments that cause them, leading to the implementation of cultural resource site fuels management during the 15 years of the project period.

## 13.1. Santa Fe National Forest Cultural Resources Activities, 2010-2024

The Jemez Ranger District of the Santa Fe National Forest implemented the Archaeological Site Thinning Project from 2013 to 2022 (Figure 13.1.1). Approximately 90 percent of sites on the Jemez Ranger District are constructed of volcanic tuff rock, highly porous and susceptible to fire. The rock tends to crack, spall, and weaken under high intensity fire and prolonged exposure. During the project approximately 1,700 sites were manually thinned by Quicksilver Contracting (Figure 13.1.2) and Jemez Ranger District Fire personnel. Fuel loading assessments were conducted on 3,000 cultural resources. Sites treated include pre-contact small structures, pueblos, cavates, rock art, and artifact scatters and historic log structures, rock art, and artifact scatters.

Archaeological site thinning was accomplished by hand with chainsaws and all fuels were carried out of the site boundary and piled. Jemez Ranger District archaeologists were responsible for monitoring all thinning and determined what the thinning prescription would be at each archaeological site.

Throughout the life of the Archaeological Site Thinning Project the approach to thinning changed because of continued collaboration between archaeologists, fuel specialists and fire personnel. In 2013, the first year of implementation, a somewhat rigid design criteria were utilized resulting in more standing trees being cut and more fuel on the ground (Figure 13.1.3). Unfortunately, this resulted in some piles being around 6ft high and 6 ft wide and when burned caused torching of surrounding trees during prescribed fire and soil sterilization.

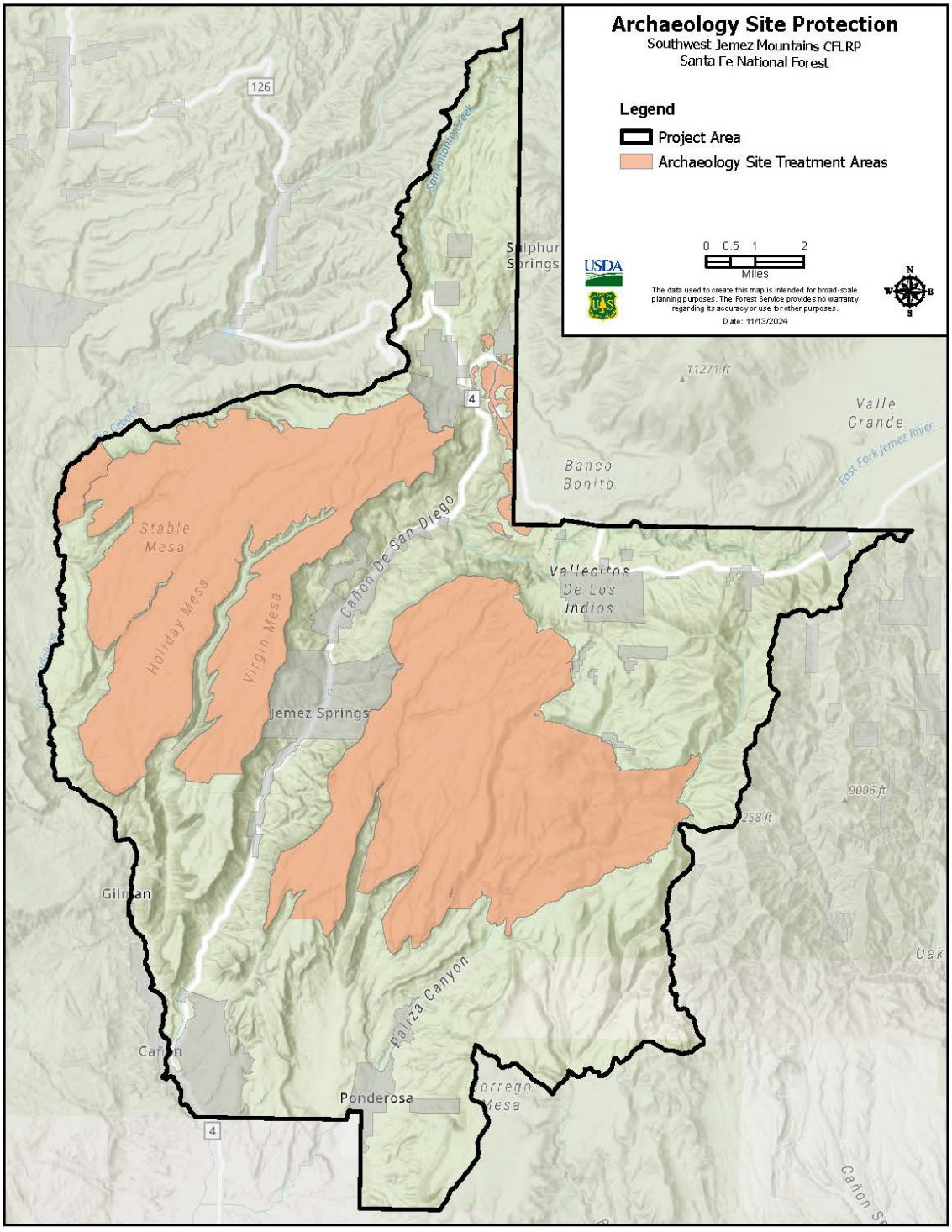



Figure 13.1.1. Map of archaeological site thinning areas on the Jemez Ranger District of the Santa Fe National Forest during the SWJM CFLRP project period.



Figure 13.1.2. Quicksilver Contracting crew works to remove wood fuels from a Puebloan cultural resource site to prevent future fire damage to the structure.
Jemez Ranger District, Santa Fe National Forest.



Figure 13.1.3. Pre-contact structural site after fuels reduction treatment in 2013. When future fires move across this site, the lack of heavy fuels will prevent rock spalling and cracking, while removal of nearby trees will prevent sub-terranean root burning that could compromise rock wall foundations and cause collapse.

In 2016 the thinning prescription was altered to mimic historic forest conditions and current forestry prescriptions, what some call "groupy clumpy". This is defined as stands with large openings in the canopy adjacent to clusters or groups of healthy ponderosa pine (Figure 13.1.4). This approach to thinning cultural resources allowed for a more natural landscape and smaller piles of cut fuels measuring 4ft high by 4ft wide. Cultural resources with this level of thinning still had minimal fire effects during prescribed burning, piles were fully consumed, and soil was not sterilized in the process.



Figure 13.1.4. View of pre-contact structural site post-treatment in 2016.

The Archaeological Site Thinning Project was a proactive approach to cultural resource management in the face of changing climate and uncharacteristic wildfire. Not only did this project foster increased collaboration between fire, fuels, and archaeology, it also provided current and accurate condition assessments and geospatial information for thousands of cultural resources on the Jemez Ranger District which will allow for better stewardship and site protection in the future.

## 13.2 Valles Caldera National Preserve Cultural Resource Activities

In support of the many SWJM CFLRP activities on the Preserve, the NPS Cultural Resources (CR) team (including Preserve CR staff and the University of New Mexico's Office of Contract Archaeology) have conducted a number of field surveys to identify cultural resource sites prior to restoration actions. In general, project activities such as thinning and burning avoided these identified site locations, and specially-trained teams went in later to the CR sites and implement treatments to preserve and protect the on-site resources. Cultural Resource surveys during 2003-2024 are shown in Figure 13.2.1, and the vast majority of these survey areas (~26,000 acres of the 36,888 total survey acres) were done for NHPA compliance purposes prior to forest thinning and prescribed burning operations. The summary table shows CR site preservation work accomplished during 2010-2024 (Table 13.2.1).

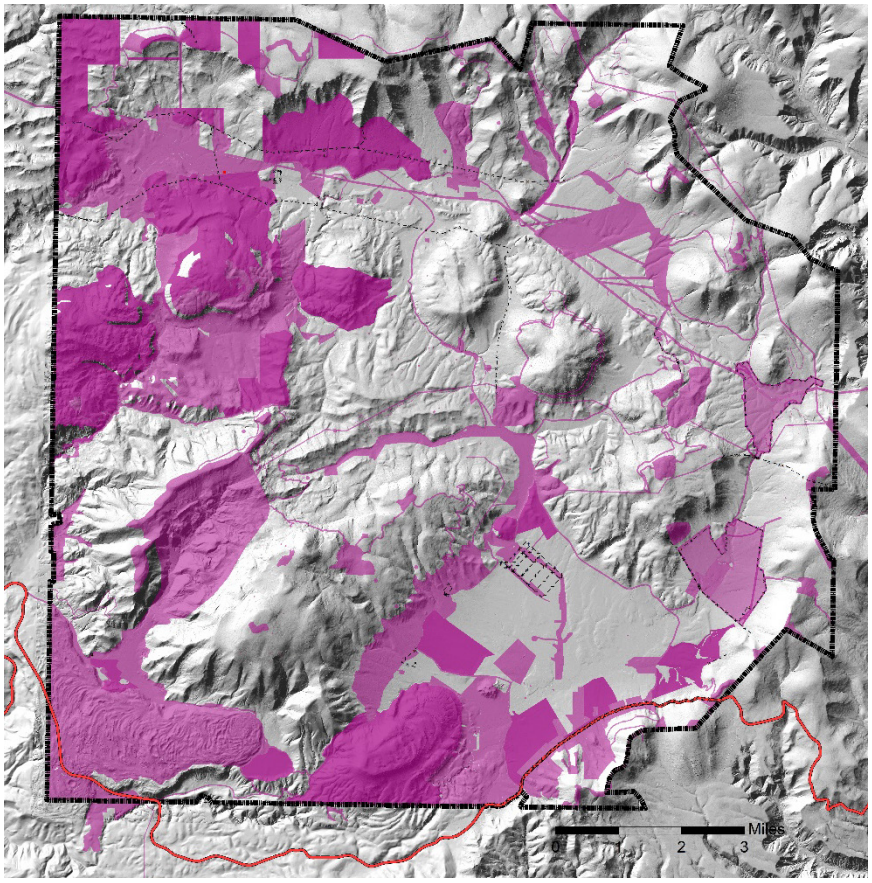



Fig. 13.2.1. Map of Cultural Resource surveys on Valles Caldera National Preserve.

Table 1.4. List of Cultural Resource Activities on Valles Caldera National Preserve

| VCNP CR Activity | TOTAL 2010-2024 |
|---|---|
| Cultural resources inventory (acres surveyed) | 26,089 |
| Sites documented during inventory (# of sites) | 405 |
| Site assessments prior to forest & watershed restoration treatments (# of sites) | 258 |
| Protective treatment at CR sites (# of sites) | 222 |
| Post-fire monitoring for effects (# of sites) | 92 |
| Fire/Rx monitoring locations (# installations) | 13 locations |
| Erosion monitoring locations | 7 locations |

Of the many Cultural Resource projects on Valles Caldera National Preserve, the major cultural resources site restoration projects were located on Banco Bonito in the southwestern corner of the Preserve. This was the area generally below 8,500 ft elevation, which is the highest elevation that can support agricultural activities. This western region of Banco Bonito supported more than a hundred field house sites. NPS and the Pueblo of Jemez's forestry crew worked together during 2020-2024 to reduce the fuel loads on these sites, so that future fires would not damage the rock walls of the field house ruins. In total, the archaeologists assessed 104 total sites, of which 102 were field houses and 2 were artifact scatters. Of these, 93 field house sites required treatments, along with one of the artifact scatter sites. In addition, fuels were removed from 3 cairns and a hunting blind.

## 13.3 Fire Impacts on Cultural Resources
**(*Sections reprinted/edited from Steffen et al. 2024*)**

Like many areas across the Southwest, the Jemez Mountains in north-central New Mexico can be characterized as a rich cultural landscape representing all time periods of human occupation in North America. In addition to abundant archeological resources, numerous traditional cultural places (areas with high cultural significance or important resources) are also a part of Indigenous peoples' living traditions and cultural practices. Many "natural resources" are a part of the "cultural resources" realm, as well. Climate and fire induced changes to the biological elements of landscapes, the traditional use of fire by Indigenous peoples to promote valued conditions and resources, and the important role of Tribal Consultation to assess and prioritize treatment areas and mitigation are critical elements of cultural resources management within fire-prone and fire-adapted landscapes.

In the context of warming climate trends (see Chapter 3) and the cumulative human alteration of forests, fuels, and fire regimes, several "big fires" (>10,000 acres) over the last four decades in the eastern and central Jemez Mountains have swept across large swaths of forest and burned with greater severity across larger extents than fires that occurred in the past (see selective list below). Archeological resources in these burns were likely exposed to higher temperatures (energy from combustion) and longer fire residence time than during fires before this era. During this time, dramatic and unexpected effects to archeological resources began to be reported, and each of these fires provided extraordinary opportunities for analyzing fire impacts (see this webpage, Wildfire and Archeology in the Jemez Mountains, for more details). To address these issues, a SWJM CFLRP Cultural Resources team led by Dr. Ana Steffen and Dr. Rachel Loehman acquired funding for a project entitled "ArcBurn" and funded by the Joint Fire Sciences Program.

*Overview of Direct and Indirect Effects to the Material Record at Archeological Sites*

Fire effects to archeological resources (sites and artifacts) can be **direct** (caused by combustion and heat exposure during a fire) or **indirect** (the result of secondary processes that occur after a fire). **Direct effects** to architectural sites such as pueblos and small residential structures include fracturing and discoloration of masonry (architectural stone) and destabilized walls (Figure 13.3.1). Direct fire effects to artifacts include fracturing, discoloration, or changes in decorative elements of ancient pottery (ceramic sherds), such that the style, time period, and in some cases function of these artifacts becomes unidentifiable, and cracking or fracturing of stone artifacts (lithics), in a way that can obscure the material source or the artifact's function. For obsidian, a volcanic glass commonly used for tool-making in the Jemez Mountains, shattering or bubbling of the glass can destroy the artifact shape (Figures 13.3.2, 13.3.3, 13.3.4). In addition, fire fracture produces non- cultural objects that mimic obsidian artifacts and confound analyses at quarry sites, because they are easily mistaken by archeologists who have not been specifically trained to recognize this fire effect. Fire can also damage obsidian artifacts by interfering with obsidian hydration dating potential. This chronometric technique is based on measurements of microscopic depths of hydration bands in obsidian that slowly form over centuries or millennia, as water is adsorbed into the silica structure of the geological glass: exposure to the heat of a fire can alter or reset this obsidian hydration "clock."



Figure 13.3.1. Ancestral Puebloan dry-laid masonry site burned during the Las Conchas fire in the Jemez Mountains of New Mexico, 2011. Fire effects include loss of protective vegetation along with sooting, spalling, and cracking of architectural stone (left) and destabilization of walls (right). Credit: NPS.



Figure 13.3.2. Obsidian direct fire effects at the Capulin Quarry site in the Jemez Mountains, NM, photographed six months after the 2011 Las Conchas fire. This site was used for obsidian procurement and reduction. Observed fire effects include burned ground surface with obsidian artifacts and geological nodules (left) and vesiculated nodules (right, top of photograph), partially vesiculated flake artifacts and fracture fragments, and fire fractured nodules (right, center), Credit: NPS / A. Steffen.



Figure 13.3.3. This frothy mass was a chunk of obsidian (volcanic glass) from a prehistoric quarry site before heat exposure during the Las Conchas fire in the Jemez Mountains of New Mexico, 2011 caused it to melt and bubble, destroying all discernible artifact characteristics. This "vesiculated" obsidian is a direct archeological fire effect first recognized by Fred Trembour, following New Mexico's La Mesa fire in 1977. Photo Credit: NPS / A. Steffen.

**Indirect effects** can occur days, months, or years after a fire, and can be localized or occur at a landscape scale. Many architectural sites exhibit significant pedestaling and displacement as post-fire erosion rates are high in and around features after vegetation is consumed. Architecture and other site features can also be disturbed or displaced by soil movement resulting from fire-killed tree fall. Other important post-fire effects include erosion and flooding common after high-severity wildfires, which may result in devastating damage to the archeological record occurring when the soil matrix that composes subsurface archeological sites washes away. Post-fire sheet washing can remove artifacts from their context within cultural deposits, disassociate artifacts from other materials deposited prehistorically (such as charcoals, pollen, seeds and plant fragments), and displace and transport artifacts, causing them to be redeposited into jumbled contexts. Post-fire erosion also can include gullying and trenching that cut into or eradicate deeply buried and previously stable archeological deposits (Figure 13.3.4).




Figure 13.3.4. This comparative photo time series, taken from the same location, shows a deep trench forming on a slope of Cerro Del Medio in Valles Caldera National Preserve after the 2011 Las Conchas fire in New Mexico. This indirect fire effect began with overland debris flows, followed by narrow rilling, and then developed into a deep broad trench that ripped through archeological site deposits that likely had been undisturbed for millennia. Credit: NPS / A. Steffen.

Many different archeological fire effects are possible, but not all effects are permanent or alter artifacts or sites in a way that compromises their interpretation or conveyance of information about the past (e.g., how peoples lived, when they lived, and/or how they behaved or interacted with landscapes and resources). While a wide variety of considerations can contribute to the meaning and value of artifacts, ArcBurn targeted characteristics that render them significant for archeological preservation as mandated by federal regulations. A novel contribution of the ArcBurn project is the explicit consideration of what constitutes meaningful ("substantial") fire alteration in this context. For example, sooting is a common but usually temporary fire effect that may not obscure an artifact's history; in contrast, fire fracture or breakage can render an artifact unidentifiable, undatable, or no longer attributable to a cultural group. Other examples of substantial alteration include sherd color change sufficient to interfere with identification, fracturing of obsidian nodules leading to misidentification of resulting fragments, and masonry sufficiently cracked to undermine the stability of a wall. The project's assessment of relevance or non-relevance for individual fire effects provides a stronger basis for prioritizing preservation measures and treatment actions.

*Findings and Conclusions*

Fire effects occurred in most fire environments but not all effects were assessed as substantial, and artifact types responded uniquely to simulated fire environments.

For **ceramics**, the widest range of substantial alterations were produced in surface fire experiments, as the result of direct flame contact; these included color change, residue deposits, cracking, and melting. Ground fire experiments produced only color change and residue deposits, and crown fire experiments resulted in only color change (e.g., darkening on the sherd surfaces), some instances of which were assessed to be substantial.

For **chert**, the most common fire effects observed were residue deposits and color changes (surface and ground fire environments) and fracturing (crown fire experiments).

For **masonry**, residue deposits and color changes were the most prevalent fire effect, occurring in most crown fire experiments and all surface and ground fire experiments. Cracking was mainly associated with high temperature and long duration crown fire (900 degrees C for 90 seconds). This is at the high end of crown fire "dose" in real-world fire environments but is an exposure of a type that can also occur outside of crown fires; for example, heavy downed fuels ('jackpots') will produce a similar fire 'dose' and can result in similar substantial effects. These results accord with field observations at Bandelier National Monument, where fire-caused cracking in masonry can compromise the stability and persistence of standing walls and architectural features.

For **obsidian**, observed substantial effects included fire fracture and loss of obsidian hydration dating potential. Fire fracture of obsidian nodules (i.e., the artifact cores and non-artifact geological materials common at obsidian quarries) occurred both under radiant heating doses typical of high-temperature crown fires (or under jackpots, as described above for masonry) and in open flame environments (where obsidian was exposed to contact with direct flames). Fire fracture was more common in the larger nodule sizes (Figures 13.3.2, 13.3.3), while obsidian flake artifacts did not experience fire fracture in high percentages – probably because their thin profile and lack of chunky mass resulted in lesser thermal stress. Fire-caused loss of obsidian hydration dating potential was experimentally tested for ground fire (smoldering) environments; loss of dating potential occurred in 82-100% of the flake artifacts tested. This

profound degree of information loss may be due to the long residence time associated with the smoldering fire environment, highlighting potential risks of high fuel loading (whether jackpots, well-dispersed heavy fuels, or deep beds of light fuels) on archeological sites.

Improving and expanding preservation strategies at archeological sites in fire prone landscapes presents numerous methodological and operational challenges. The ArcBurn project identified several major arenas of research – ideally conducted in practitioner-researcher partnerships – to address these challenges. These include continued characterization of the range of fire effects present across a diversity of artifacts and archaeological site types, identification of the fuels and fire environments that cause these effects, and development of operational tools (e.g., prediction models, vulnerability assessment tools, treatment guidelines) to aid managers in development of fuels treatments and characterization of risk. Archaeologists' perspectives and assessments are a necessary and critical input to design of fuel treatments for archaeological preservation; for example, in adjusting fuel treatments and fire management options based on the specific tolerances or sensitivities of the most common or most information-rich artifacts. Because diverse artifacts are often clustered together within sites, treatment decisions may need to be made on a site-by-site basis to address the most sensitive type(s) of artifact in archeological assemblages, but at landscape scales, forest treatment preservation decisions can emphasize the most frequent kinds of sites or artifacts present across large areas or apply more refined or finer-scale strategies that target protection of specific archeological resources that are more rare, more sensitive, or with higher information potential or cultural value. Design and implementation of fuel treatments with benefits across both cultural and natural resources domains is a critical step in mitigating threats and decreasing the vulnerability of all resources to impacts from climate changes and uncharacteristic fire.

## Literature Cited

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