A COMPREHENSIVE STUDY ON FOREST MANAGEMENT AND WILDFIRE TRENDS IN DRY WESTERN CONIFEROUS REGIONS OF THE UNITED STATES

by

Richard Herman Schonenberg

A thesis submitted in partial fulfillment
of the requirements for the degree

of

Master of Science

in

Land Rehabilitation

MONTANA STATE UNIVERSITY
Bozeman, Montana

May 2024

©COPYRIGHT

by

Richard Herman Schonenberg

2024

All Rights Reserved

DEDICATION

To the steadfast guardians of the forested mountains, the Ponderosa pine and the Douglas fir, whose towering presence and resilient tenacity have stood as welcoming beacons of strength amidst the complexities of our ever-evolving societies.

ACKNOWLEDGEMENTS

I am overwhelmingly fortunate and entirely grateful to those who have generously supported me both academically and spiritually throughout the journey of completing my master's thesis. First and foremost, I want to sincerely thank my Graduate committee—Tony Hartshorn, William Kleindl, and Clayton Marlow—for their valuable guidance, thoughtful feedback, and consistent support. Their expertise and encouragement have significantly contributed to the development of this thesis. I am indebted to the members of my lab group, Morgan Suddreth, Alexandria Lin, Matt Deyoe, Tyler Boyd, Mustabshira Ikram, and Eliza Gillian, for their camaraderie and collaboration, which have enriched my research experience immeasurably. Special thanks are due to the vibrant bluegrass music communities of Bozeman and Livingston for providing moments of recess and lighting joy amidst the rigors of academic pursuit. Your music and inclusivity have been a source of respite and rejuvenation throughout this journey. Lastly, I sincerely thank my family and friends for their unwavering support, encouragement, and perspective throughout this endeavor. Your belief in me has been a constant source of strength and motivation, and I am profoundly grateful for your presence in my life.

TABLE OF CONTENTS

TABLE OF CONTENTS CONTINUED

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

In response to the growing concern over the escalating severity of wildfires in dry coniferous forests across the Western United States, this study aims to evaluate the effectiveness of Managed Fire Operations (MFO) in mitigating wildfire severity. By leveraging satellite-derived fire severity data, specifically the Difference Normalized Burn Ratio (dNBR), I conducted a comprehensive analysis comparing fire severity between forest lands with and without MFO, subsequently affected by wildfires. Employing a paired study design, I analyzed wildfire events from 1985 to 2021 within dry coniferous forests, limited to south-facing slopes with moderate terrain gradients, using fire perimeter data from the Monitoring Trends in Burn Severity (MTBS) program. Geospatial analysis identified areas where wildfire perimeters intersected with MFO-designated zones, enabling the delineation of new burn perimeters for each wildfire. The results revealed that regions subjected to MFO before wildfires experienced a statistically significant decrease in fire severity compared to areas without MFO (Wilcoxon signed-rank test, p-value < .01). Ecoregion stratification revealed notable variations, with the Northwestern Forested Mountains showing an estimated median fire severity difference nearly three times greater than the Temperate Sierras. Further stratification by time since MFO implementation displayed consistent, modest reductions in fire severity across two intervals (0-15 years and 15-40 years), with minimal variations between the temporal categories. These results highlight the effectiveness of MFO in mitigating wildfire severity and emphasize the importance of regional context and temporal factors in evaluating MFO efficacy in Western U.S. dry coniferous forests over the past four decades.

INTRODUCTION

Dry western coniferous forests (DWCF) in the U.S. encompass a variety of forest types characterized by their specific climate conditions and dominant tree species. These forests, typically found in regions with lower precipitation and warmer temperatures, are primarily characterized by fire-resistant species such as ponderosa pine (*Pinus ponderosa*), sugar pine (*Pinus lambertiana*), western larch (*Larix occidentalis*), and Douglas-fir (*Pseudotsuga menziesii*) (Barbour, 2007). These forest ecosystems play a crucial role in the region's ecological and hydrological processes, biodiversity, and carbon sequestration, yet they are susceptible to high-severity wildfire events due to historical fire exclusion policies, climate change impacts, and land management practices (Hagmann et al., 2021; Parks et al., 2023).

Before European-American expansion in the western United States, dry conifer forests experienced fires every 5–35 years, primarily low-to-moderate severity (Fulé et al., 2003; Margolis & Balmat, 2009). The arrival of European Americans precipitated substantial changes in the landscape, including the cessation of indigenous burning, the implementation of high-grade logging practices coupled with passive forest management, and active fire suppression, which collectively facilitated the accumulation of live and dead fuels and fundamentally altered the structure of DWCF (Hagmann et al., 2021; Hessburg et al., 2019). The combination of these actions, alongside a warmer and drier climate, has further contributed to substantial shifts in the fire regime of this forested ecosystem, with high-severity fire events occurring more frequently and posing a threat to the persistence of these forests (Parks et al., 2023).

Over the past five decades, Managed Fire Operations (MFO), which involve the deliberate and controlled use of fire under specific environmental conditions, has gained prominence as a land management tool to reduce the potential impacts of high-severity wildfires in DWCF (Fernandes, 2015; Prichard et al., 2021). The rationale behind this approach is that prescribed or managed fires reduce the available fuel load, decreasing the severity of subsequent wildfires. However, a regional-scale assessment of the reduction in wildfire severity resulting from these operations has not yet been conducted.

In the same time frame, satellite technology has advanced landscape assessment by providing consistent, high-resolution data on Earth's surface characteristics. Satellites equipped with multi-spectral sensors offer invaluable insights into the spatial extent and intensity of burned areas, enabling a more precise determination of fire severity. Integrating satellite-derived data with indices such as the Normalized Burn Ratio (NBR) and the Difference Normalized Burn Ratio (dNBR) yields valuable quantitative measures for evaluating the ecological impact of wildfires on regional scales. When merged with existing landscape ecology and fire science research, these datasets aid in developing effective land management and conservation strategies for land managers and policymakers.

To address the knowledge gap regarding the effectiveness of Managed Fire Operations (MFO) in reducing subsequent wildfire severity, this study aims to (1) develop a regional-scale satellite-derived dataset characterizing MFO-treated lands impacted by wildfires, (2) conduct a statistical analysis incorporating time and ecoregion variations to assess the impact of MFO on wildfires in the context of temporal and biophysical gradients, and (3) contextualize the results with field-based calibration assessments to evaluate the ecological impact. Given the escalating trend of high-severity wildfire events (Westerling et al., 2006), forest resilience is essential for forest managers navigating uncertain climate conditions. The findings of this study provide quantitative insights to shape and inform effective forest management strategies for greater forest resiliency.

LITERATURE REVIEW

The Natural Fire Environment

Wildfires have existed on Earth since terrestrial biomass aligned with natural ignition sources like lightning and volcanic eruptions (Bowman et al., 2009). Evidence of fire's ancient presence dates back hundreds of millions of years to the Paleozoic era, with extensive fossil charcoal deposits spanning geological epochs (Komarek, 1973). Moreover, wildfire has been recognized as a regular phenomenon since the Mesozoic era, aligning with gymnosperms' dominance and angiosperms' emergence (Scott, 2000).

Natural plant communities reflect species assembled in transitory states, each exhibiting distinct lag times in response to historical climatic fluxes and disturbances, migrating along latitudinal and elevational gradients (Lockwood et al., 1997). Despite differing migration rates and limited coevolution with other species, each extensively coevolves with specific processes and disturbances over time (Pickett & White, 1985). Thus, throughout most, if not all, of their respective evolutionary developments, naturally occurring fires have been associated with most species of angiosperms and gymnosperms (He et al., 2016).

Fires in natural systems exhibit diverse behaviors and characteristics shaped by climate, weather patterns, landscape configuration, biomass quantity, and vegetation structure (Moritz et al., 2011). The fire regime concept serves as an integrated framework, categorizing the impacts of these multifaceted spatial and temporal patterns of fire behavior at the ecosystem or landscape level (Schoennagel et al., 2004; Taylor & Skinner, 2003). Understanding historical and contemporary fire regimes and their modifying factors is essential for comprehending and projecting interactions between fire, natural ecosystems, forest management practices, and the impacts of a changing climate.

Fire Regimes

Quantifying Wildfire

The temporal and spatial patterns of fire behavior and effects within a specific vegetation type or ecosystem across multiple fire cycles (ranging from decades to centuries) define the fire regime for a given ecosystem (Moritz et al., 2011). At a minimum, fire regimes can be differentiated by their frequency of occurrence (expressed as incidence, fire interval, or fire rotation) and evaluation of their impact on the ecosystem (e.g., mortality of canopy or understory vegetation) (Sommers et al., 2011). More detailed fire regime classifications consider additional features such as fire location characteristics (e.g., surface fire, crown fire, ground fire), fire size (patch size), intensity, seasonality, topographic position, variability within ecosystem types, and fire severity (degree of ecosystem impact) (Sommers et al., 2011).

Fire severity, a fire-induced change to physical ecosystem components such as vegetation and soil, is a critical fire regime component that influences landscape heterogeneity, soil erosion, nutrient cycling, wildlife habitat, and post-fire successional trajectories. The effects of high-severity fires can immediately compromise ecosystem integrity, threaten natural and economic resources, and limit the ability to suppress wildfires (Higuera et al., 2023). For this reason, fire severity studies have increased in research and applications over the past decade, coinciding with the development of satellite-based remote-sensing technologies (Cova et al., 2023; Stevens et al., 2017).

Furthermore, understanding the historical context of fire in ecosystems, which typically involves data on fire frequency and severity in a stand or landscape, is essential for characterizing comprehensive fire regimes. Historical fire regimes provide a temporal framework for fire in an area, informing changes in regional biodiversity trajectories and potential human-induced influences on fire regimes (Fulé et al., 2003; Swetnam et al., 1999). Two disciplines primarily contribute to understanding historical fire regimes: pollen and fossilized charcoal analyses inform periods dating back hundreds of thousands of years, while dendrochronology focuses on more recent centuries.

Historical Fire Regimes

Approaching the present, understanding of fire history improves, with evidence indicating substantial interactions between vegetation, climate, and fire that persist to the present (Agee, 1993). Pollen analysis studies, often conducted on samples from peatlands and dated using radiocarbon dating, enable the inference of shifts in species composition and the derivation of a fire activity index from the charcoal found in the same layers, facilitating the reconstruction of regional vegetation patterns throughout the Holocene (Whitlock & Larsen, 2001). During periods of climatic change, many sites experience the colonization of new species facilitated by burning, accelerating the expansion of shade-intolerant vegetation while reducing the spread of shade-tolerant late-successional species (Brubaker et al., 2009; Heyerdahl et al., 2002).

Fire history in the current millennium has become more definitive due to the long lifespans of many forest tree species, which can reliably span 500 to 1000 years (Waring & Franklin, 1979). Analysis of forest age structure and fire scars from these trees directly records fire activity (Morrison & Swanson, 1990). Additionally, more recent fire records are informed by survey documents, fire atlases (Andrews & Cowlin, 1940), and paintings, but these are often incomplete and may be limited by land ownership or political constraints (Agee, 1993). Thus, recent efforts have intensified to map fire regime characteristics using satellite remote sensing, with initiatives like Canada's extensive fire database covering fires larger than 200 hectares from 1959-1999 and representing about 97% of the burned area (CWFIS, 2024) while in the United States, the Monitoring Trends in Burn Severity (MTBS) Project utilizes satellite data to map all large fires since 1984 (Eidenshink et al., 2007).

Contemporary Fire Regime Quantification

Since 2006, the Monitoring Trends in Burn Severity (MTBS) project has been gathering and analyzing data from 1984 to 2020 to discern national forest health and wildfire trends, encompassing nearly 50 years of U.S. wildfire frequency, severity, and size information (Eidenshink et al., 2007). Established to analyze the effects of forest management policies on fire occurrence and severity, this collaborative project maps indices of fire severity, including the Normalized Burn Ratio (NBR) and Difference Normalized Burn Ratio (dNBR) within fire perimeters using moderate-resolution satellite data acquired before and after fires. These indices, derived from multi-temporal change detection through remote sensing, quantify burn severity by describing the impact of fire intensity (Figure 2.1) on ecosystem functioning and the degree of alteration, providing an informative basis for evaluation (Key & Benson, 2006).

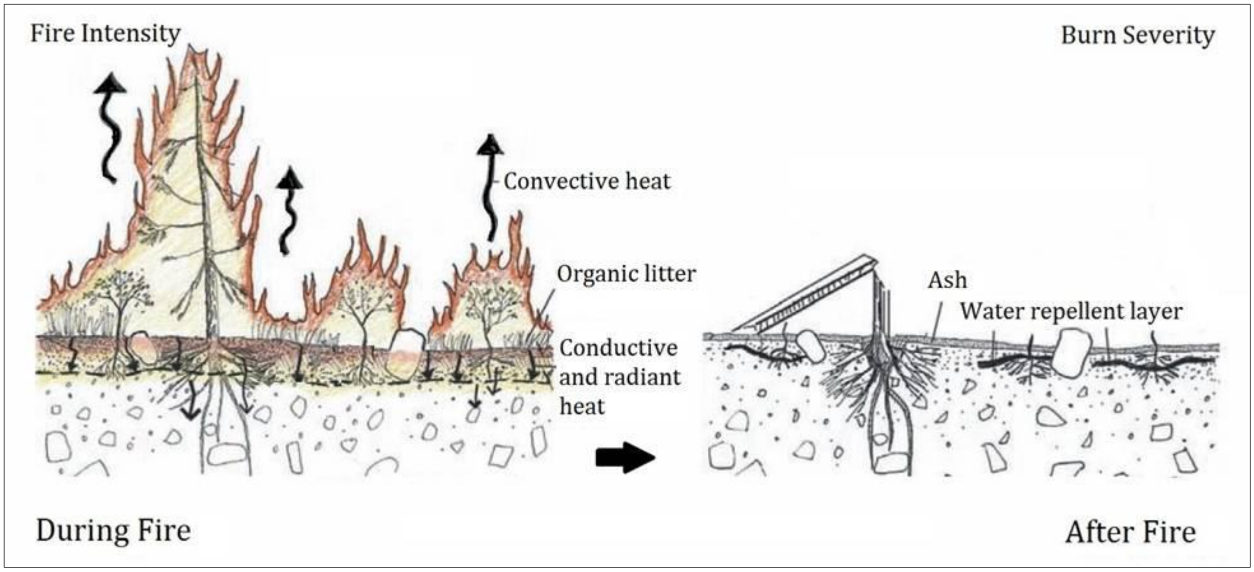


Figure 2.1. Fire Intensity and Burn Severity. Illustration describing the relationship between fire intensity (measured in heat output) and burn severity (evaluated by the ecological impact on vegetation and soil) (US Forest Service)

The Normalized Burn Ratio employs both near-infrared (NIR) and shortwave infrared (SWIR) wavelengths to detect burnt areas in large fire zones. Healthy vegetation typically reflects high NIR and low SWIR, contrasting with recently burnt areas where NIR reflection is low and SWIR reflection is high (Figure 2.2a.) (Key & Benson, 2006).

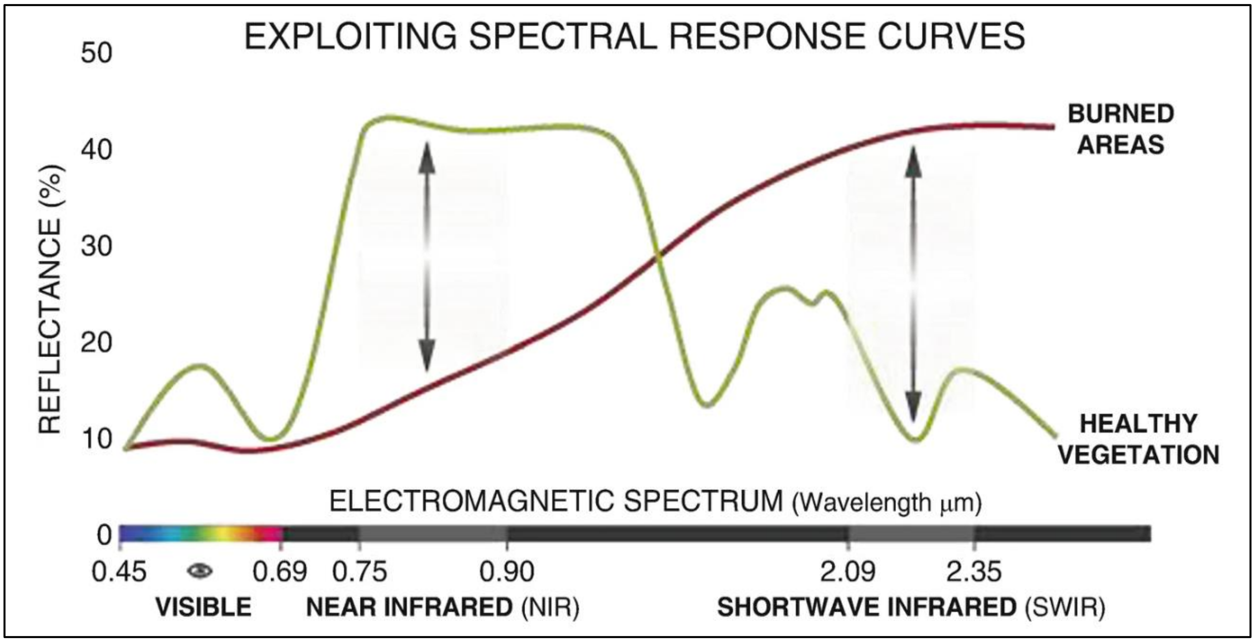
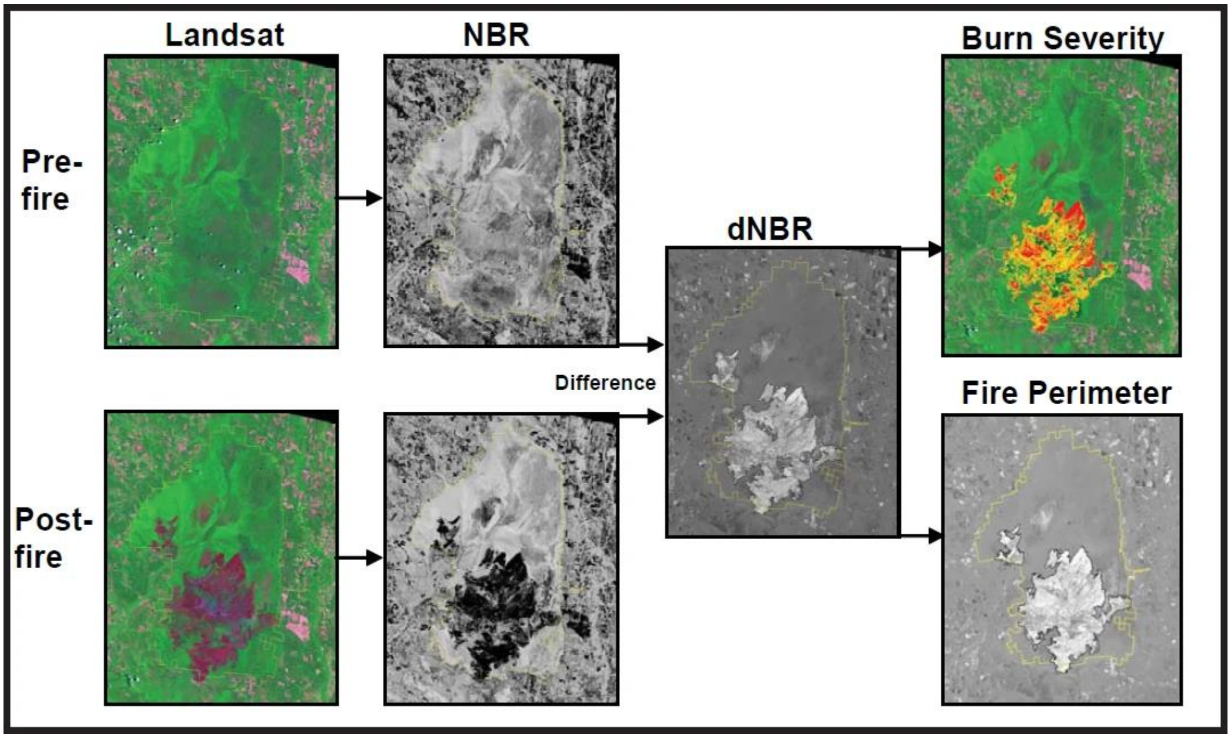


Figure 2.2a. Spectral Reflectance Curve. Spectral reflectance curve comparing healthy vegetation and burned areas. Healthy vegetation shows high near-infrared (NIR) and low short-wave (SWIR) infrared reflectance, while burned areas exhibit low NIR and high SWIR reflectance, highlighting distinct spectral differences (US Forest Service)

The contrast between healthy vegetation and burnt areas is most pronounced in the NIR and SWIR regions of the spectrum, where the Normalized Burn Ratio (NBR) calculates the ratio between NIR and SWIR bands, with a high NBR value signifying healthy vegetation (Figure 2.2a.). A low NBR value indicates bare ground or recently burnt areas, and non-burnt areas typically register values close to zero. The difference between pre-fire and post-fire NBR values (Figure 2.2b) is used to compute the Difference Normalized Burn Ratio (dNBR), which indicates burn severity, typically ranging from -200 to 1300. Low and negative dNBR values indicate unburned or increased vegetation in each area, while higher dNBR values signify more severe fire damage.



$$\textbf{Equation 1)} \qquad NBR = \frac{NIR - SWIR}{NIR + SWIR}$$

$$\textbf{Equation 2)} \qquad dNBR = ((PrefireNBR - PostfireNBR) * 1000) - dNBR_{offset}$$

Figure 2.2b. Process Sequence and Equations for Calculating the Normalized Burn Ratio and Difference Normalized Burn Ratio. (Equation 1) The Normalized Burn Ratio (NBR) is calculated from Landsat satellite imagery by subtracting the shortwave infrared (SWIR) reflectance from the near-infrared (NIR) reflectance and dividing by their sum, with higher positive values indicating healthier vegetation and negative values often associated with burned or bare ground. The Difference Normalized Burn Ratio (Equation 2) is calculated by subtracting the pre-fire NBR value from the post-fire NBR value, with an offset applied to account for differences due to phenology or precipitation between the pre-and post-fire images, providing a quantitative measure of burn severity. Satellite imagery shows the processing sequence for Landsat images to map burn severity and fire printer in the Okefenokee National Wildlife Refuge (yellow line is the refuge border) sourced from (Eidenshink et al., 2007).

To facilitate comparisons of burn severity across multiple fires and landscapes, an offset value is applied to the dNBR equation (Figure 2.2b). This offset value, derived from an unburned or reference area within the same scene as the burned area, serves as the baseline dNBR value and is subtracted from the computed dNBR to normalize the burn severity values (Kolden et al., 2015). Despite the strong evidence of dNBR accurately characterizing on-the-ground fire severity (Parks et al., 2014), field assessment is recommended for accurate ecological interpretation (Kolden et al., 2015).

Calibration of dNBR to Field-based Assessments

Satellite-derived spectral indices like NBR and dNBR have limited practicality as standalone metrics because their values are not empirically field-validated (Eidenshink et al., 2007). While NBR and dNBR datasets are linked to various observed fire effects metrics, such as vegetation cover and soil burn severity, their accuracy varies considerably across ecosystems (Eidenshink et al., 2007; Lentile et al., 2007). Thus, these indices' variable sensitivity and accuracy in representing fire effects require ecosystem-specific or location-specific evaluation (French et al., 2008; Lentile et al., 2007).

While the MTBS project offers an "offset value" specific to each dNBR file that normalizes interannual variability phenology, relying solely on this does not comprehensively refine the accuracy of fire severity assessment (Kolden et al., 2015). Consequently, calibration methods involve aligning dNBR values with field-based metrics such as vegetation mortality, soil burn depth, and scorch height to improve precision and reliability in measuring fire severity (Cansler & McKenzie, 2012; Holden et al., 2009; Miller et al., 2009). For these assessments to be valuable in calibration, they require standardized data collection, such as the Composite Burn Index (CBI), a measure of post-fire ecosystem impact supported by various land management agencies (Key & Benson, 2006).

Composite Burn Index The Composite Burn Index (CBI) is a comprehensive assessment of ground-level burn severity effects, employing 30-meter fixed-radius plot diameters to evaluate various strata, including litter, duff, fuel, soil, and vegetation layers ranging from herbs to large trees (Key & Benson, 2006). Each stratum undergoes evaluation based on 4-5 rating factors, quantifying burn severity on a scale from 0.0 (indicating no burning) to 3.0 (representing high severity). The ratings are then averaged to derive a single value per stratum, and an overall CBI value is calculated by averaging across all strata. This value can then be compared with satellite-derived burn severity data to develop regression equations for further analysis (Figure 2.3).

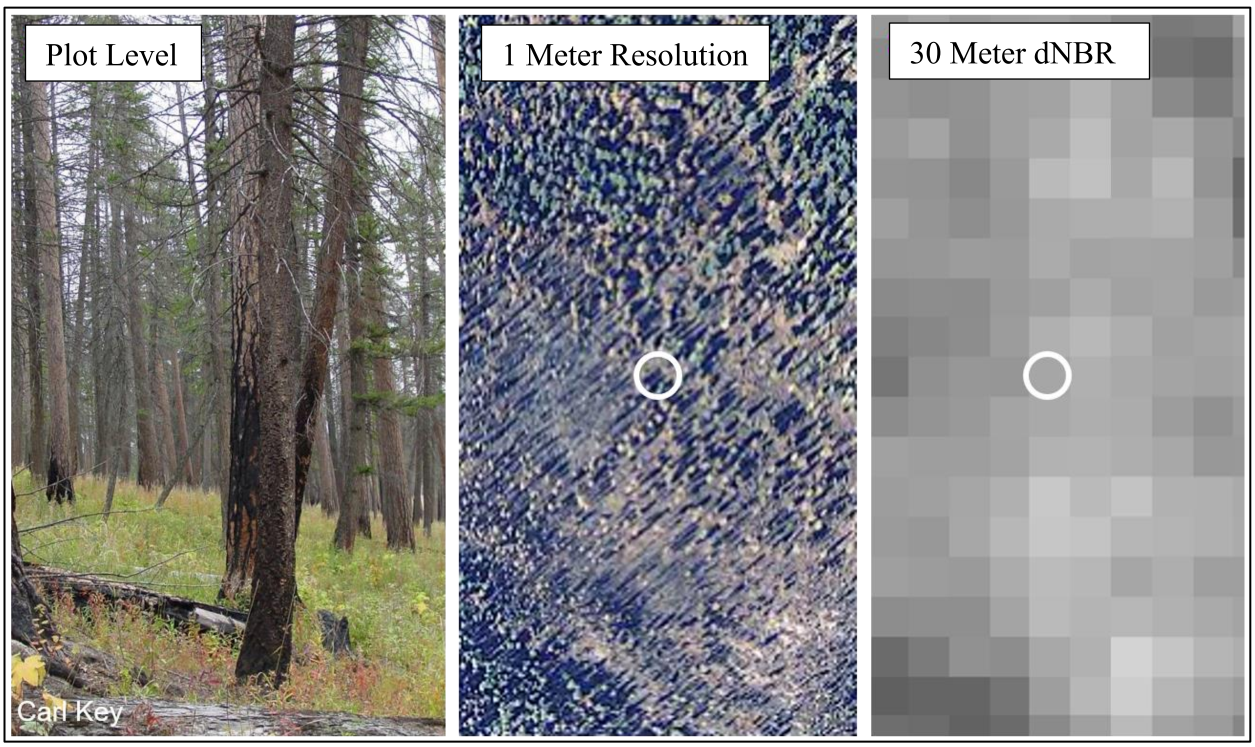


Figure 2.3. Severity Mapping using Composite Burn Index and Difference Normalized Burn Ratio. Example of burned areas where composite burn index analysis was performed at the plot level, 1-meter resolution imagery, and 30-meter dNBR imagery (Key & Benson, 2006).

Dry Western Coniferous Forests

Among forested ecosystems, DWCF's fire regimes are extensively studied, providing increased confidence in understanding historical fire regimes compared to others. This confidence in dry western conifer systems stems from dominant conifer species in these forests, which possess longevity, relative fire resistance, and the ability to exhibit fire-scarring (Franklin & Dyrness, 1973), making them valuable indicators of fire history, particularly within the western United States (US) context. Additionally, dry conifer forests are acknowledged as being substantially altered from reference conditions in the western US, primarily due to 20th-century logging practices and ongoing fire suppression efforts (Parks et al., 2023), which has prompted increased research on these fire-adapted systems.

General Description

Dry coniferous forests in the US are characterized by fire-resistant trees such as ponderosa pine, sugar pine, western larch, and Douglas-fir, which flourish in mesic environments with occasional drought. Marked by limited understory growth, wide crown-spacing, and nutrient-deficient soils (Barbour, 2007), the distribution of these forests in the United States spans a vast range, covering various elevations from the lower regions of the northern Rocky Mountains to the higher altitudes of Arizona and New Mexico (Figure 2.4). A comprehensive fire regime for this forest ecoregion is derived from a combination of palaeoecological studies, dendrochronology investigations, and recent remotely sensed analyses, all of which indicate a notable division in fire regimes between pre-European-American westward expansion and post-European westward expansion.

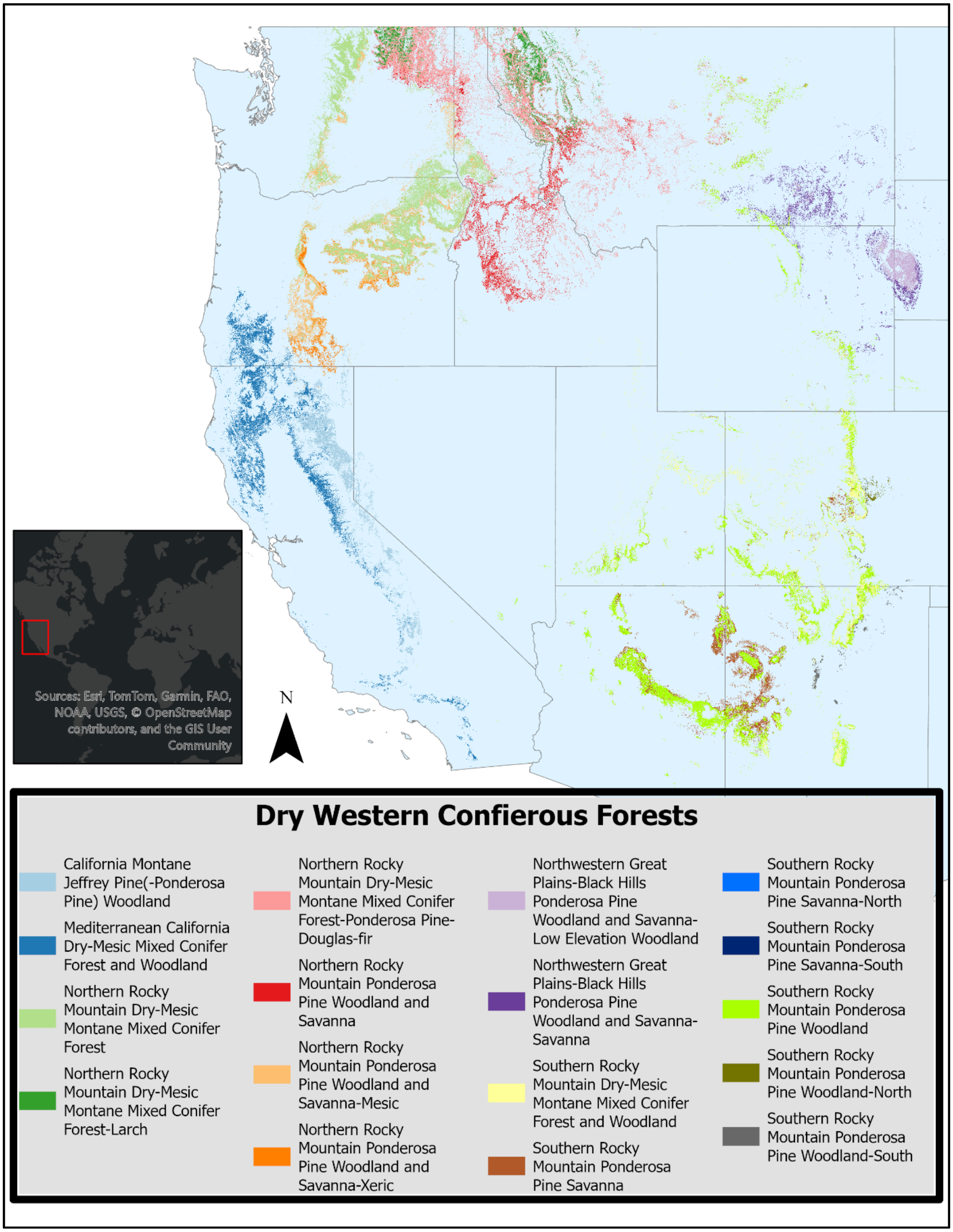


Figure 2.4. Map of Biophysical Settings that Define Dry Coniferous Forests. Dry Coniferous forests are defined by seventeen distinct biophysical settings spanning various elevations and latitudes (LANDFIRE 2020).

Fire Regime Classification of Dry Western Forests

Historic Fire Regime Palaeoecological studies inform the fire regimes of dry US coniferous forests over the past millennia by linking large-scale climatic events to fire activity. High-severity fires and erosional events were prevalent during the Medieval Climatic Anomaly (950-1250 CE), a period marked by multi-decadal drought, while evidence suggests that smaller, less severe, and more frequent fires occurred during the Little Ice Age (1300-1850 CE) (Pierce et al., 2004). Furthermore, charcoal studies in northern Idaho indicated high-severity fires occurring every 175 years from 208 to 980 CE, with low-intensity fires predominating until European expansion in the area in the early- to mid-1800s and beyond (Smith, 1983). While these coarse temporal estimates inform historical regimes, more refined studies focus on the recent 200 years, delineating stark differences between fire patterns before and after European American inhabitation into these forested regions.

Before European American expansion into the western United States, dry coniferous forest ecosystems experienced frequent low and mixed-severity fires (Arno & Allison-Bunnell, 2002). Fires with low and mixed-severity led to a forest type characterized by low tree density, clustered tree distribution, sparse and patchy fuel beds, relatively uniform canopy layering, and a fire-tolerant composition of trees and associated species (Hessburg et al., 2005). This impact extends to both stand-level and regional-level forest dynamics, influencing individual stands' structural and ecological characteristics and the broader composition and functioning of forested regions.

Low and mixed-severity fires in forest stands favor fire-resistant structures by eliminating lower crown classes. These fires also cycle nutrients from foliage and branches into the soil, stimulate the growth of a sparse and varied shrub and herb cover, diminish the risk of running crown fires by consistently thinning stands, removing fuel ladders, and decreasing competition among surviving trees, shrubs, and herbs (Uhl & Kauffman, 1990). Frequent surface fires benefit the largest trees with thick bark, resulting in varied ages within stands due to the ongoing regeneration of trees through fires (Harrod et al., 2009).

At the landscape scale, natural patterns of dry forest structure and composition, maintaining a semi-predictable mosaic, supported low- or mixed-severity fires, while spatially isolated conditions conducive to high-severity fires were infrequent (Hessburg et al., 2005). Consequently, severe fire behavior and effects were uncharacteristic of landscapes dominated by dry forests (Hessburg & Agee, 2003). Dry forest landscapes seldom experienced widespread synchronization in vegetation and fuel conditions, but when they did, it was due to climate-driven, high-severity fire events (Agee, 1998; Swetnam & Lynch, 1993; Whitlock & Knox, 2002).

<u>Contemporary Fire Regime</u> Following the arrival of European American colonization in the region (~1875), key change agents, categorized by their primary ecological effects, worked to exclude fires, directly advance secondary succession, suppress fires, or achieve a mix of these impacts. Various human activities, including domestic livestock and wild ungulate grazing, road and rail construction, grassland conversion to agriculture, urbanization, and rural development, collectively contributed to the direct or indirect exclusion of fires.

Since the 1870s, domestic livestock grazing has notably diminished fast-burning fuels, reducing the potential for rapid surface fire spread across the landscape (Belsky et al., 1999; USDA, 1994). The construction of roads and railways further fragmented expansive forest landscapes into smaller, isolated pieces, with road and rail beds acting as effective barriers against low-flame-length surface fires. Furthermore, changes in land allocation, such as lands allotted to railroad companies being sold or harvested, contributed to historical landscape fragmentation, reducing fire spread (Hessburg & Agee, 2003).

Similarly, the conversion of grasslands into agricultural areas prevented the occurrence and spread of fires. Historic fires, initially ignited by natural lightning or intentional aboriginal burning, became diminished in spread and occurrence with urbanization, rural development, and displacement of Native American communities (Whitlock & Knox, 2002). This led to the nuanced fragmentation of fire-prone dry forests and the replacement of highly flammable vegetation types with inflammable bare ground, concrete, or asphalt, as well as less flammable grazed, irrigated, or cultivated areas (Halofsky et al., 2020).

Repeated selection cutting, mainly targeting ponderosa pine, Douglas-fir, and western larch, directly advanced secondary succession, contributing forest structure more prone to high severity fire occurrence (Hessburg et al., 2019). Selective logging, which removed large-diameter, fire-resistant ponderosa pine and Douglas-fir, resulted in the formation of small canopy gaps subsequently filled by small-diameter Douglas-fir, grand fir, and white fir (Hessburg et al., 2000). Furthermore, the exclusion and suppression of fires further facilitated the indirect advancement of secondary succession, as they hindered disturbances occurring at a frequency and spatial scale conducive to promoting the dominance of early seral cover species, leading to higher fuel loads and more continuous fuel beds (Hessburg et al., 2005).

The 20th-century shift in management culture, characterized by widespread logging that removed the most fire-resistant trees and successful fire suppression initiatives (Mallek et al., 2013), resulted in increased tree density and a greater prevalence of shade-tolerant, fire-sensitive tree species (Naficy et al., 2010). The accompanying alterations in forest structure, coupled with rising temperatures and an increased prevalence of drought, have prompted a shift in the fire regime towards more frequent and intense fires (Hessburg et al., 2019; Parks et al., 2023; Westerling et al., 2006). Furthermore, these altered forests are unlikely to absorb the shock of naturally occurring fires in the summer months, as fuel consumption and high-intensity wildfires will likely result in the death of remaining old-growth trees (Agee, 1993)

Navigating the Anthropocene: Adaptive Management Strategies in Dry Western Conifer Forests

Despite not being recognized as an official subdivision of geologic time (ICS, 2024), the Anthropocene marks a critical period in Earth's history characterized by unprecedented human influence on natural ecosystems (Steffen et al., 2011). In DWCF, this has profound implications for fire regimes, increasing the frequency and intensity of wildfires (Schoennagel et al., 2004). Human activities, including fire suppression, land use change, and climate change, have disrupted historical fire regimes, altering vegetation composition and forest structure (Hagmann et al., 2021), thereby exacerbating fire risk and challenging ecosystem resilience, with potential long-term consequences for biodiversity and ecosystem function (Williams et al., 2023). Moreover, the interplay between anthropogenic factors and natural fire processes complicates efforts to manage and mitigate wildfire impacts in these ecosystems (Stephens et al., 2018).

Managing wildfires in DWCF during the Anthropocene epoch has emerged as a critical challenge (Higuera et al., 2023; McWethy et al., 2019). Traditional fire suppression strategies have inadvertently contributed to altered fire regimes and increased the risk of catastrophic wildfires (Parks et al., 2023). Adaptive fire management has emerged as a promising approach, incorporating ecosystems' dynamic nature, structure, and function with socioeconomic considerations (McWethy et al., 2019).

Adaptive management involves a flexible and iterative process, continuously refining strategies based on new information and changing conditions (Sample et al., 2022). This approach recognizes the intricate presence of fire occurrence in DWCF, encompassing factors including inter- and intra-annual climate variability, vegetation succession dynamics, and human activities. Additionally, by incorporating diverse perspectives and engaging stakeholders, adaptive management has the potential to enhance ecosystems' resilience to wildfires (Sample et al., 2022).

In the context of forest management, the careful and deliberate implementation of managed fire operations (MFO) has the potential to restore ecological processes while also mitigating severe wildfire risks. MFO, when meticulously planned and executed under specific conditions, can emulate natural fire regimes, fostering forest health and reducing fuel loads (Fernandes, 2015; Hunter & Robles, 2020). Given the lengthening of fire seasons attributed to climate change, it is plausible that prescribed fire seasons may also expand in certain regions in response to these environmental shifts (Abatzoglou et al., 2021; Swain et al., 2023). MFO encompasses two primary approaches: Wildland fire use operations (WFU), which oversee and regulate natural fires ignited by lightning, and prescribed fires (RX), involving intentional and controlled ignitions under specified conditions. Both MFO strategies deviate from the traditional wildfire suppression approach of recent centuries by acknowledging fire's integral role in natural processes, yet their scale and effectiveness in achieving goals to reduce wildfire severity have not been assessed. Thus, this research aims to evaluate the effectiveness of MFOs in reducing subsequent wildfire severity.

ASSESSING THE IMPACT OF 40 YEARS OF MANAGED FIRE OPERATIONS IN DRY WESTERN CONIFEROUS FORESTS

Study Objective

The primary objective of this study is to quantify the extent and effectiveness of MFO in attenuating wildfire severities within dry coniferous forests across the western US over the past four decades. Utilizing satellite-derived fire severity data, specifically the dNBR, a comparative analysis of fire severity is conducted between regions with and without managed fire operations, subsequently impacted by wildfires, across two prominent ecoregions in the United States (Figure 3.1). This investigation required the creation of an unprecedented dataset, employing a paired study design of managed fire and wildfire interactions that spans the expanse of dry western forests in the US.

Dry Conifer Forest Identification

This research focuses on the ecosystem of conifer savannas, woodlands, and forests typically found in the lower elevational range of arid forested regions in the Western United States, referred to as 'dry western conifer forests'. To identify and delineate DWCF, the satellite-derived LANDFIRE Biophysical Settings (BPS) geospatial dataset (Rollins, 2009) was employed, with a resolution of 30 meters. Consistent with the classification criteria outlined by Parks et al. (2023), 17 BPS classifications were identified that meet the definition of dry coniferous forests. The classifications, primarily characterized by tree species like ponderosa pine, Jeffrey pine, sugar pine, Douglas-fir, and western larch (Table A.1), were integrated to produce a unified raster layer representing the estimated extent of DWCF (Figure 3.1)

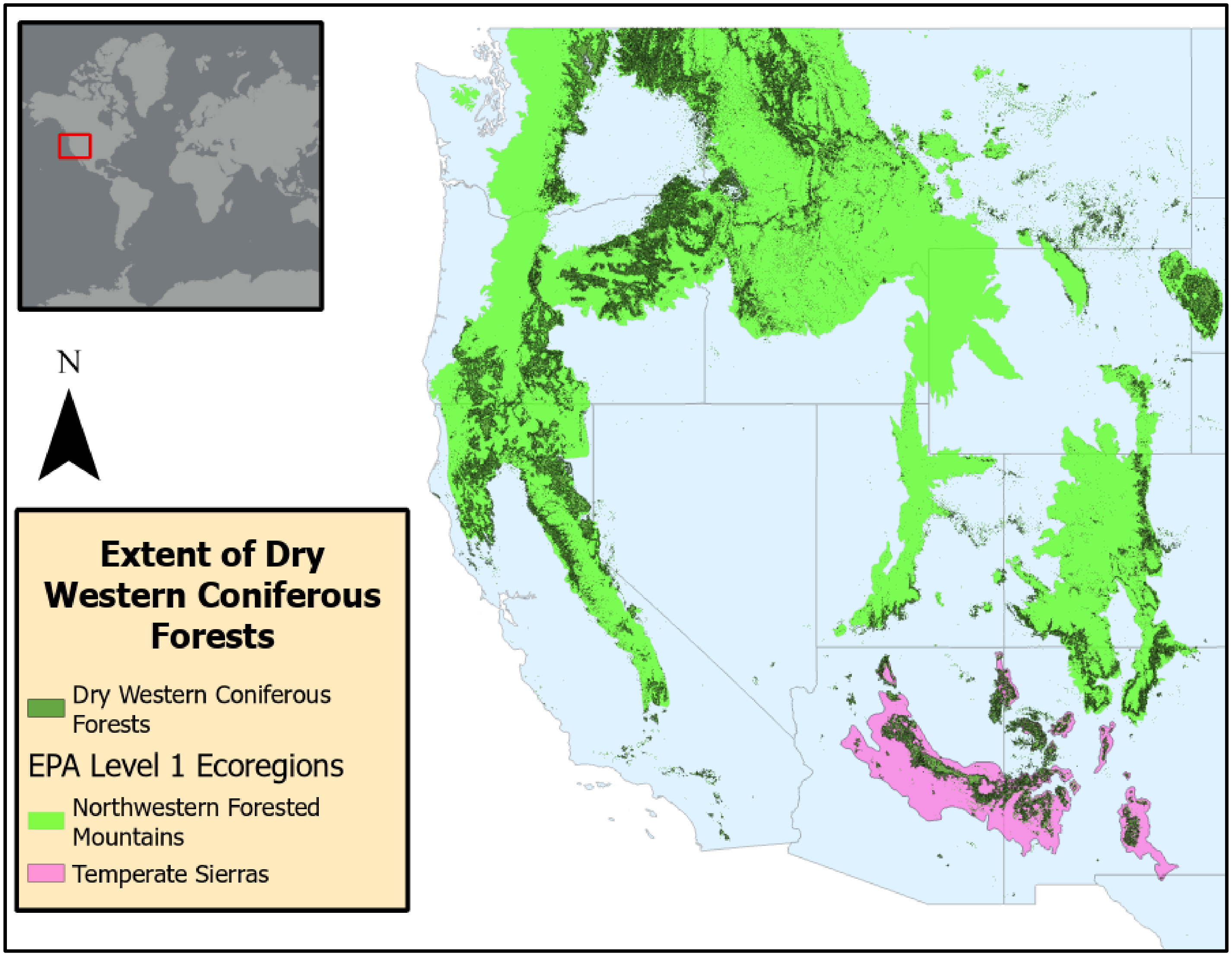


Figure 3.1. Extent of Dry Western Coniferous Forest and Two Ecoregions. This map depicts the extent of dry western coniferous forest and two ecoregions, where wildfire severities are compared between areas identified as managed fire operations and untreated areas.

Recognizing that the classification of dry conifer forests by LANDFIRE BPS may not always reflect the current forested state due to land-use alterations or past disturbances such as stand-replacing fires or land conversion, the analysis was refined by exclusively focusing on pixels classified as forest, woodland, or savanna, as outlined Dillon et al. (2020). These designated areas were identified by synthesizing landscape-level vegetation datasets, including LANDFIRE's Existing Vegetation Cover, Environmental Site Potential, and Landsat Time Series Stacks-Vegetation Change Tracker (Dillon et al., 2020; Huang et al., 2010). This approach ensured an accurate regional-scale characterization of the dry conifer ecosystem and has been used in similar regional-scale wildfire severity studies (Dillon et al., 2020; Parks et al., 2023).

Topographical Stratification

Given the substantial influence of topography on fire behavior, particularly the documented increased rate of spread observed on steeper slopes (Rothermel, 1972) and its correlation with biophysical gradients such as solar radiation and topographic moisture, which affect fuel characteristics and availability (Holden et al., 2009), the study focused on south-facing slopes with gradients less than 24 degrees. This criterion was guided by the United States Department of Agriculture slope classification, which defines slopes with gradients less than 24 degrees as moderately steep, aligning with conditions most relevant to the managed fire operations in this study (USDA, 2022). Leveraging the publicly available LANDFIRE 2020 slope degree layer and the LANDFIRE 2020 slope aspect layer (both with a resolution of 30m) covering the contiguous United States, I utilized the raster calculator geoprocessing tool in ArcGIS Pro software to create new raster datasets, delineating areas with slopes equal to or less than 24 degrees and aspects ranging from 35 degrees to 225 degrees, indicative of the southern direction on the compass rose. Subsequently, these criteria were combined iteratively using the extract by mask tool to integrate slopes less than 24 degrees, southern aspects, and dry western coniferous tree cover, resulting in a comprehensive dataset for statistical analysis (Figure 3.2.)

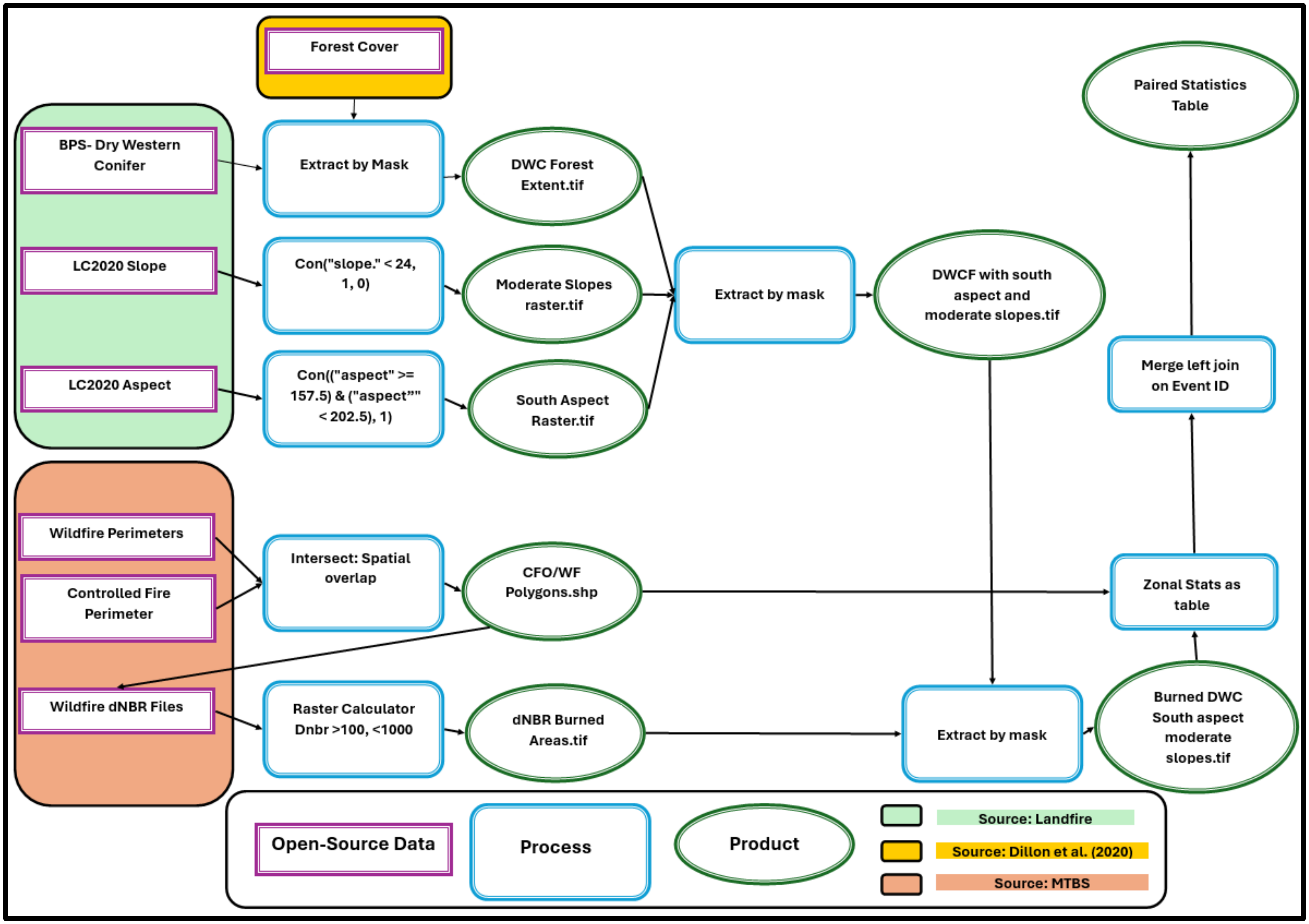


Figure 3.2. GIS Analysis Flowchart. Flowchart depicting data sources, processes, and products used and created in this GIS analysis

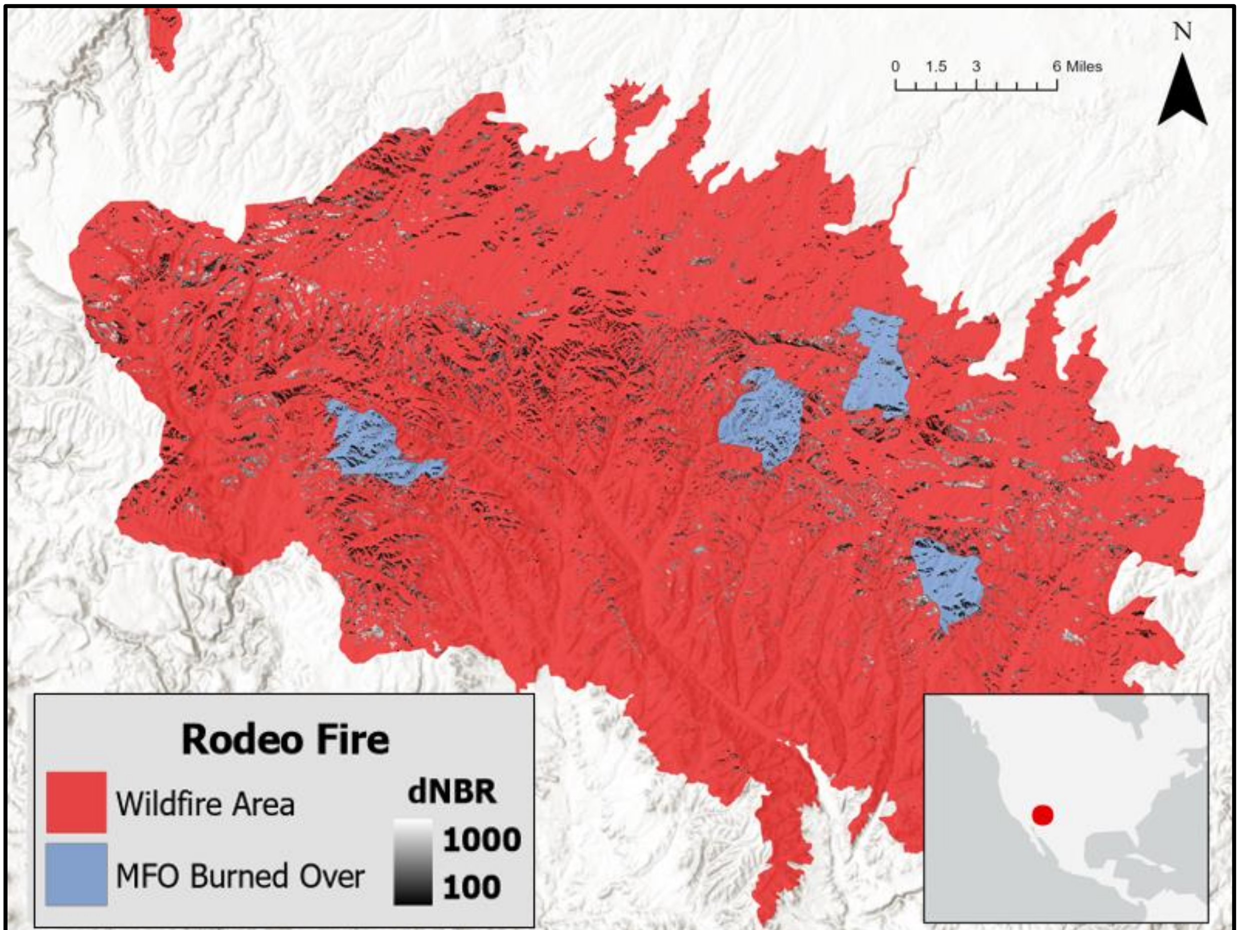
Ecoregion Stratification

To accommodate the climatic variations within the expanse of dry western forests spanning various elevations and latitudes, I employed the broad EPA Ecoregion level 1 designation to mitigate differences in aridity and evapotranspiration gradients across the western US. The dataset, acquired from the EPA website (USEPA, 2016), was integrated into our analysis using the spatial join tool in ArcGIS Pro, which categorized fire perimeters into distinct ecoregions based on the majority of overlapping areas. This approach enabled a standardized assessment of fire regimes in varied ecological zones, particularly the Northwest Mountains and Temperate Sierras, where the overlap between MFO operations and wildfires was most notable (n=86).

Wildfire Data

Wildfire and Managed Fire Area Delineation

To pinpoint areas where wildfires affected MFO-designated lands in the western United States, I utilized the ArcGIS Pro intersect geoprocessing tool on two polygon layers sourced from MTBS, representing all MFO and wildfire extents from 1984 to 2020. Using R software and the raster and SF packages, new burn perimeters were generated for fires with overlapping areas, distinguishing between zones impacted by fire operations and those unaffected (R Core Team 2023). This delineation enabled analysis of areas affected by the same wildfire, with and without MFO (Figure 3.3).



Incident Name	Incident type	Acreage	Ignition Date	Zonal Median dNBR	IQR Zonal Median dNBR
CHUCK BOX	Prescribed Fire	4181	10/21/1998	264	215-299
HOPCOT	Prescribed Fire	4580	3/25/1999	364	286-468
DEER SPRIN	Prescribed Fire	5037	11/1/1999	243	201-323
HOP-COT	Prescribed Fire	3742	10/11/2001	246	215-291
RODEO	Wildfire	461774	6/18/2002	403	295-522

Figure 3.3. GIS Analysis Snapshot. Map and table illustrating a single wildfire event (highlighted in red) burning over areas designated as Managed Fire Operations (MFO, highlighted in blue). The 30-meter dNBR raster cells depict wildfire severity, limited to DWCF, south-facing slopes, and a moderate slope gradient. The table lists wildfire severity data from zonal statistics corresponding to MFO from left to right.

Wildfire Severity Data

The dNBR dataset, obtained from the Monitoring Trends in Burn Severity (MTBS) database (Eidenshink et al., 2007), offers a comprehensive assessment of post-wildfire burn severity for each observed wildfire at a resolution of 30 meters. Using pre- and post-fire satellite imagery, MTBS computes the normalized difference in reflectance values and assigns a unique dNBR offset specific to the affected area (Kolden et al., 2015), with higher dNBR values indicative of more severe burning. Raw dNBR values can range from -2000 to 2000, which includes unburned areas, clouds, and satellite errors. To focus solely on burned areas, a filter was applied to retain values exceeding 100 and ≤1000. This aligns with established studies that define dNBR ranges of -200 to 100 as unburned (Delcourt et al., 2021; Lutz et al., 2011) and limits values to ≤1000 to minimize land cover misidentification.

Using R software (R Core Team 2023), this filtered dNBR raster was merged with the south-facing aspect and moderate slope rasters of DWCF, producing a formatted raster optimized for analysis of burned areas within dry forests on moderate or gentler slopes with south-facing aspects (Figure 3.2). Although dNBR datasets are commonly used to evaluate fire effects like vegetation cover and soil organic matter consumption, their accuracy can vary significantly across ecosystems (Eidenshink et al., 2007; Lentile et al., 2007). Therefore, an ecosystem-specific evaluation is essential due to these indices' variable sensitivity and accuracy in representing fire effects (French et al., 2008; Kolden et al., 2015).

Ecological Context

To impart ecological context to the MTBS dNBR datasets, I conducted a calibration process using field-derived Composite Burn Index (CBI) values. The CBI, a field-based assessment method, systematically captures fire effects across diverse strata within burned areas. This assessment protocol visually estimates and scores fire effects across five vertical strata, encompassing surface fuels and soils to large trees. After assessing each stratum, the total CBI value is computed by summing the scores from all variables within each stratum and dividing by the number of variables measured (Key & Benson, 2006). The resulting CBI value offers a comprehensive evaluation of fire severity within the burned area, graded on a scale from zero to three, where higher values correspond to greater severity, zero indicating no effect, and three denoting maximum severity (Key & Benson, 2006). These CBI values can then be compared with satellite-derived burn severity data to model regression equations and more accurately characterize fire effects.

CBI assessments conducted by land management agencies, such as the United States Forest Service and National Park Service, have been aggregated into publicly accessible shapefiles containing tabular data, including location, date, data logger designation, CBI values, and corresponding geo-specific dNBR values. These shapefiles are openly accessible via direct download from the United States Geological Survey (USGS 2024). While this dataset provides a consistent methodology for evaluating severity, it encompasses various climatic and ecological regions across the continental United States (CONUS). Consequently, plots of interest overlapping with the DWCF dataset were delineated using the clip tool in ArcGIS Pro, resulting in 1931 CBI plots characteristic of the study extent.

Establishing thresholds for wildfire severity requires careful consideration of the distribution of CBI values and their associated severity categories. Following widely used and established protocols (Miller et al., 2009; Miller & Thode, 2007), thresholds were determined at the midpoint between specified values on the CBI spectrum ranging from 0 to 3. For example, when considering the transition from "moderate" to "high" severity categories, the midpoints proposed by Miller and Thode (2007) indicate a transition between CBI values of 1.5 and 2.0 for "moderate" severity and between 2.5 and 3.0 for "high" severity. This guidance established thresholds of 1.25 for moderate severity and 2.25 for high-severity fire classification (Table 3.1).

The Composite Burn Index (CBI) demonstrated a moderate predictive capacity for characterizing satellite-derived fire severity (dNBR) within the DWCF dataset ($r^2 = 0.62$). Utilizing this non-linear regression model (Figure 3.4), thresholds of wildfire severity specific to DWCF were established. Based on the regression equation (Figure 3.4), fire severity was subsequently categorized as follows: dNBR <148 corresponded to Low Severity, a range of 149-357 denoted Moderate Severity, and values exceeding 357 indicated High Severity (Table 3.1).

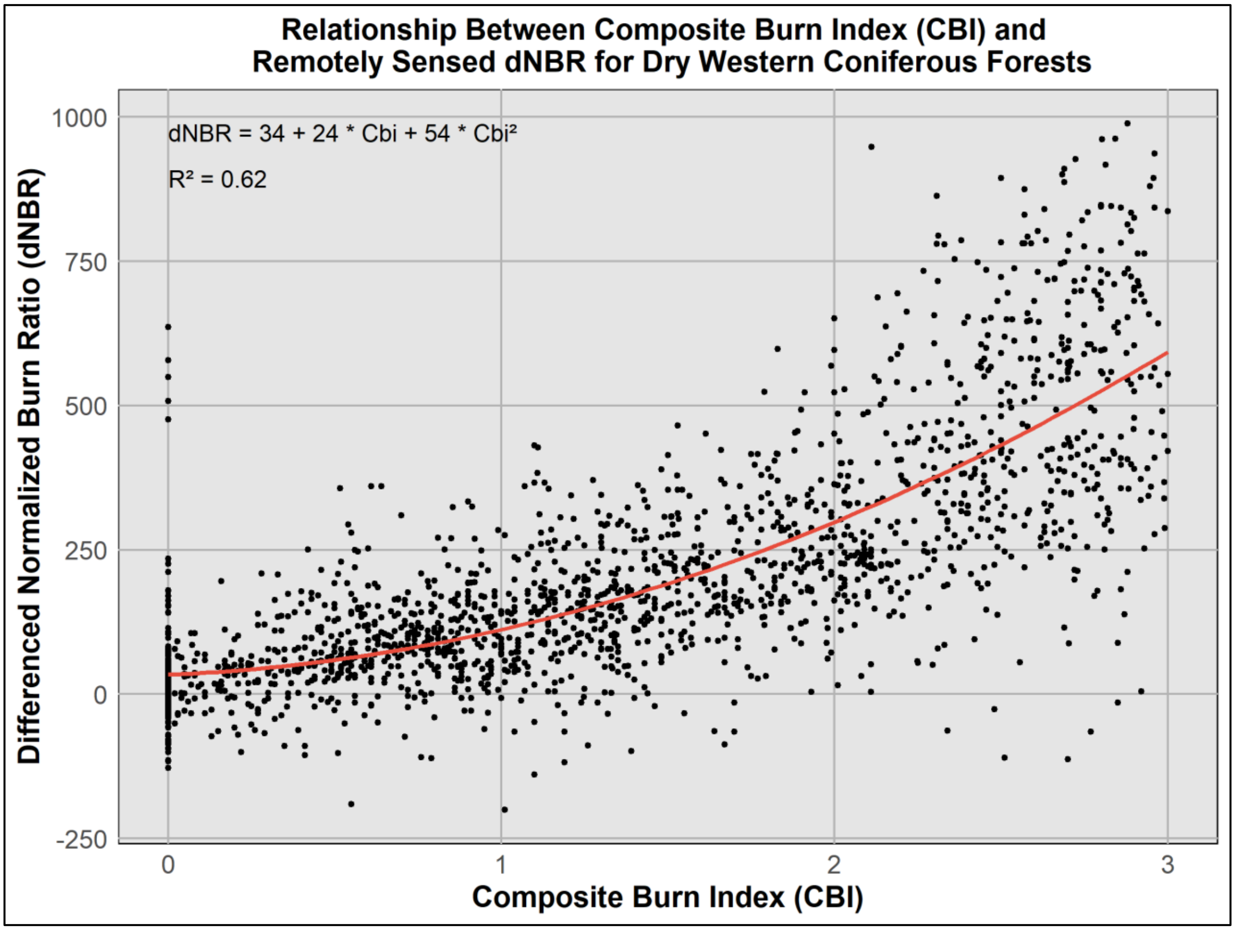


Figure 3.4. Non-linear Regression between CBI and dNBR values. This graph illustrates the non-linear relationship between field-based CBI assessments in DWCF and corresponding dNBR values from the same locations.

DNBR Range	Predicted CBI	CBI Severity Classification
0-148	0-1.24	Low/Unburned
>148-357	1.25-2.24	Moderate
>357	2.25-3	High

Table 3.1. Thresholds of Fire Severity. Thresholds of fire severity used in this analysis based on nonlinear regression (Figure 3.4) and Miller and Thode's (2007) CBI Severity Classification.

Statistical Analysis and Study Design

Using the raster and statistics packages in R software, zonal statistics were performed on raster cells for burned DWCF areas with south-facing, moderately sloped terrain, distinguishing between areas affected by managed fire operations and those that were not. To address the potential limitations of using average dNBR values across extensive fire areas, which could introduce data skewness, median dNBR values were employed as they offer a more robust measure of fire severity across large areas. Additionally, the dataset was refined to include only areas greater than or equal to one acre, ensuring an accurate representation of wildfire severity by area.

The data, characterized by zonal median dNBR values for fire severity, exhibited a non-normal distribution. Thus, the Wilcoxon signed-rank test was employed on the paired dataset to evaluate its statistical significance. This statistical approach facilitated the evaluation of differences in zonal median dNBR between the two areas, allowing computation of corresponding p-values, confidence intervals, and pseudo-medians, providing insights into the observed disparities.

The dataset for statistical analysis consisted of 86 paired data points with dNBR values characterizing fire severity, each representing a MFO impacted by a subsequent wildfire. Prior to statistical analysis, data distributions indicated zonal median dNBR values centered around 190 for areas treated with MFO and 300 for untreated areas (Figure 4.1). Given that many large fires affected multiple MFO operations, the zonal median dNBR (Difference Normalized Burn Ratio) of a single wildfire could be associated with multiple MFO operations. Of these data points, 46 were located in the Northwestern Forested Mountains ecoregion, and 40 were in the Temperate

Sierra ecoregion. Furthermore, differences in ignition dates between MFO and subsequent wildfire impacts facilitated the calculation of time intervals (0-15 years and 15-40 years) and the evaluation of these temporal variables.

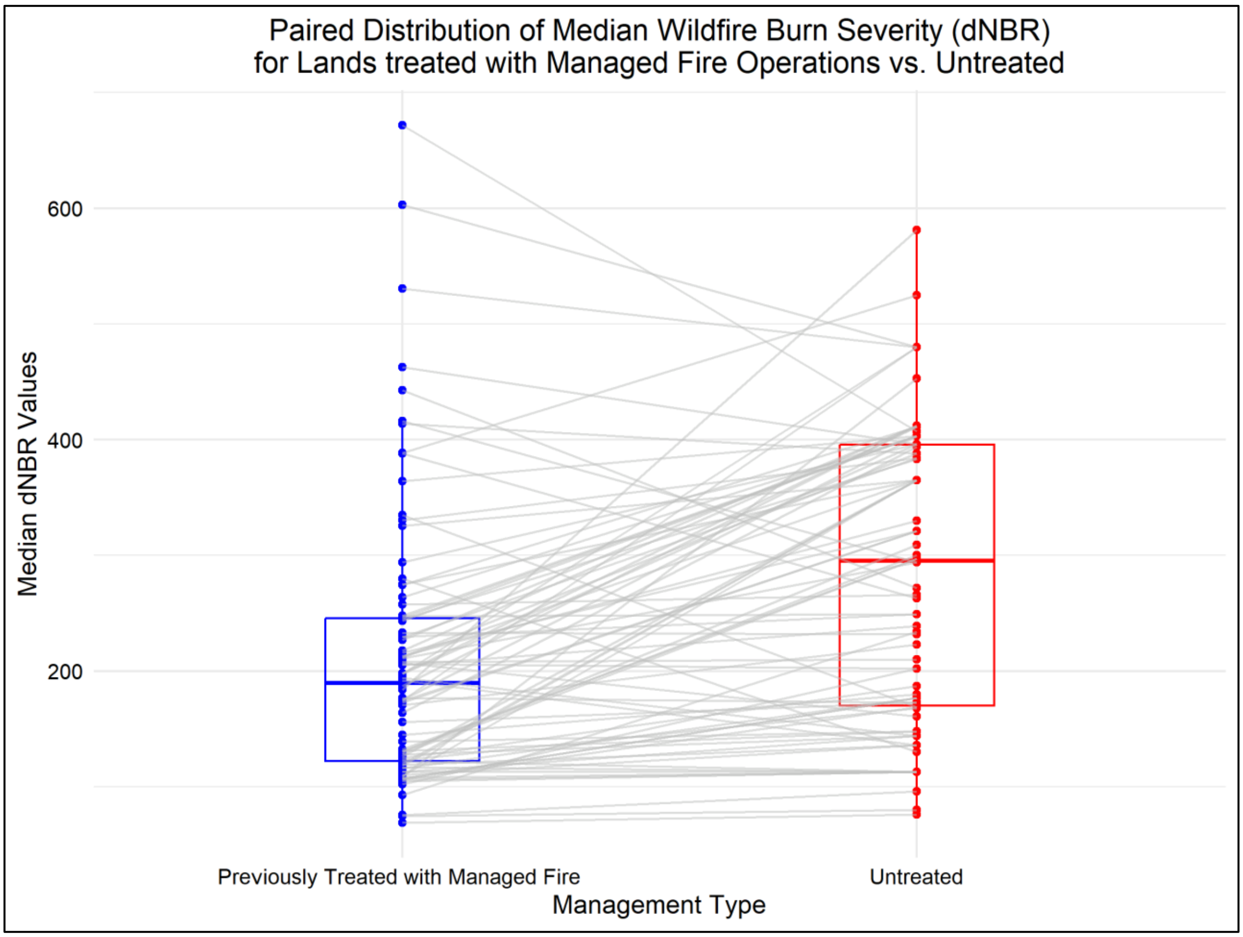


Figure 4.1. Paired Distribution of Median Wildfire Burn Severity. Paired distribution of zonal median wildfire burn severity (dNBR) for lands treated with Managed Fire Operations (MFO, blue) versus untreated lands (red). Each dot represents median dNBR values derived from zonal statistics, and grey lines connect MFO-designated lands to untreated lands impacted by the same wildfire event

RESULTS

Statistical Conclusions

In dry conifer forests of the western United States, regions that underwent managed fire operations prior to experiencing subsequent wildfires demonstrated a highly significant reduction in wildfire severity compared to areas without managed fire operations. (pseudo-median: -76 dNBR, 90% CI: -118 -45 dNBR, p-value < .01, Wilcoxon signed-rank test). Upon stratifying the entire dataset by the time elapsed since MFO into two intervals, 0-15 years and 15-40 years, consistent findings emerged, indicating a statistically significant reduction in median fire severity between areas subjected to MFO and those that were not, regardless of the time since MFO (p-value < .05, Wilcoxon signed-rank test) (Figure 5.1). Furthermore, only a modest discrepancy in estimated pseudo medians was observed between the 0-15-year interval (-85 dNBR) and the 15-40-year interval (-56 dNBR), with the 90 percent confidence interval widening in the 0–15-year category (-104 to -59) compared to the 15-40-year classification (-113 to -7), likely due to the reduced number of samples in the latter category (Figure 4.1).

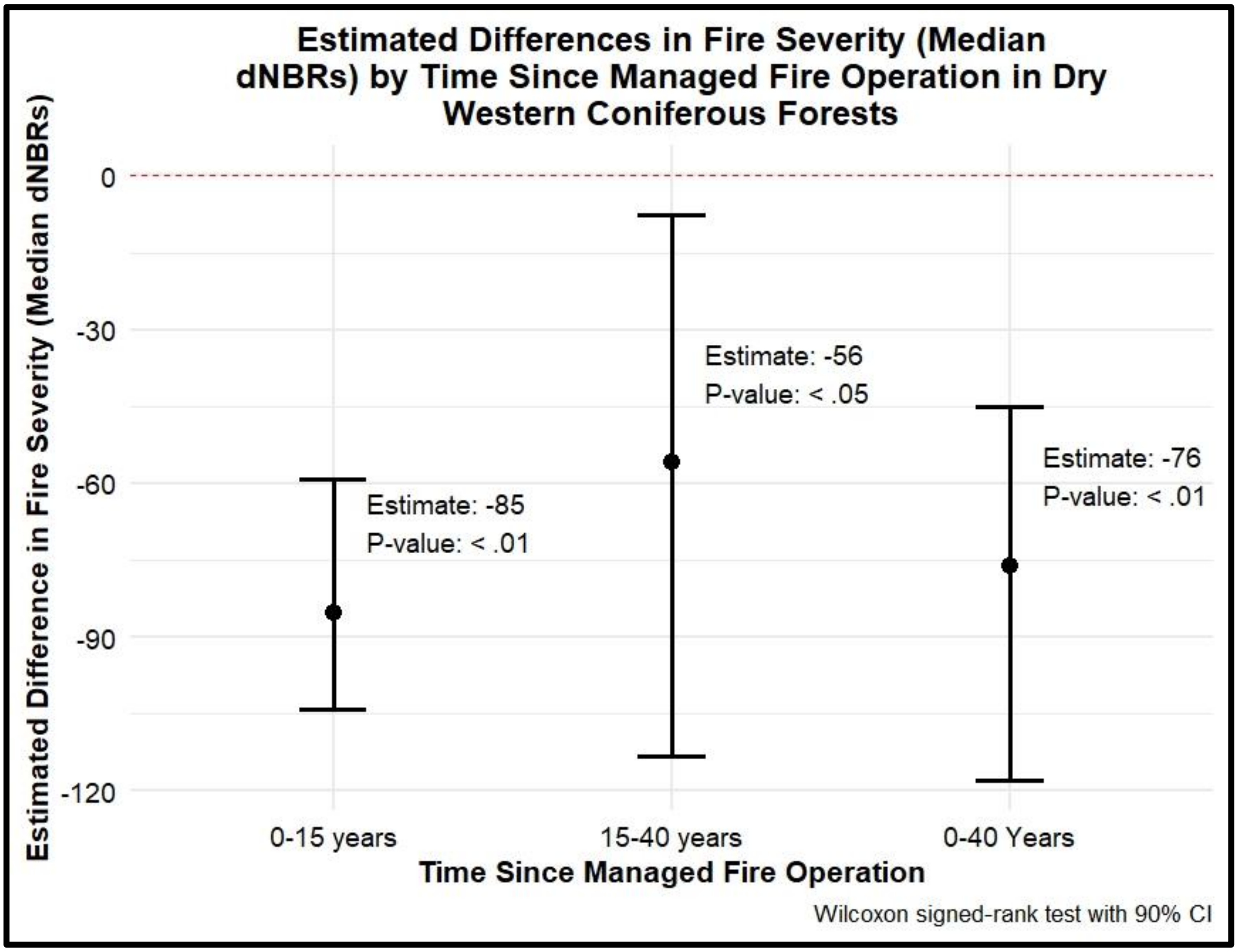


Figure 4.2. Estimated Differences in Fire Severities by Time since Managed Fire Operation. Estimated Differences in Wildfire Severity impacting MFO areas across temporal classifications (0-15, 15-40, 0-40) in dry western coniferous forests. Error bars represent 90% confidence intervals. Annotations denote estimates and p-values from the Wilcoxon rank sum test. All categories indicate a statistical reduction in impacting wildfire severity compared to untreated lands, with the most considerable estimated difference in the 0-15 category.

Upon stratification of the dataset (n=86) by ecoregions, the estimated median difference in fire severity was found to be more than two times larger in the Northwestern Forested Mountains (n=46, pseudo-median: -104 dNBR, 90% CI: -142 -62 dNBR, p-value < .01, Wilcoxon signed-rank test) compared to the Temperate Sierras (n=40, pseudo-median: -38, 90% CI: -78 to -20 dNBR, p-value < .05, Wilcoxon signed-rank test). Of all stratifications performed in this study, The Norwest Forested Mountains Ecoregion showed the most robust and most pronounced estimated difference in fire severity between MFO and untreated lands (Figure 4.2). These findings highlight significant variations in fire severity between MFO lands in the two Ecoregions studied; however, the context of the severity distributions of impacting wildfires should be considered.

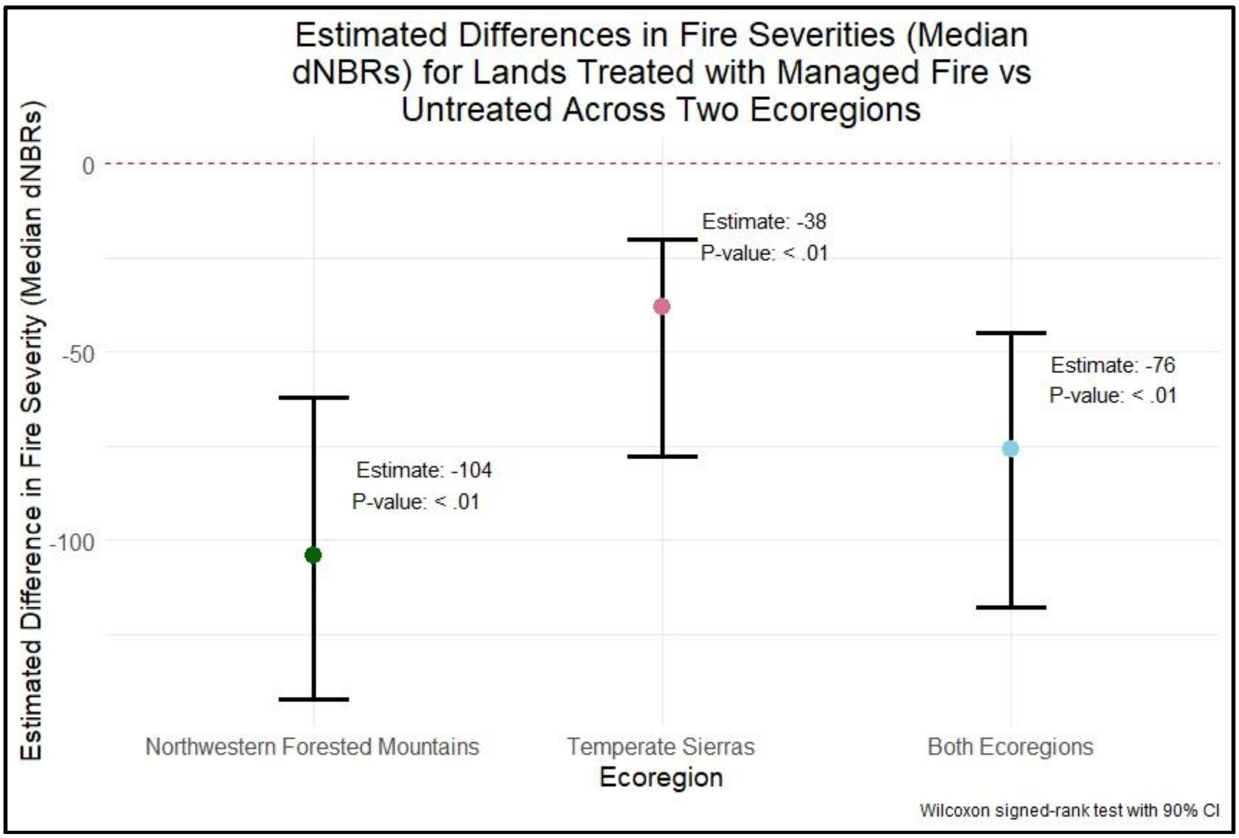


Figure 4.3. Estimated Differences in Fire Severities Across Ecoregions. Error bars represent 90% confidence intervals. Point colors indicate ecoregions: Northwestern Forested Mountains (dark green), Temperate Sierras (pale red), and Both Ecoregions (pale blue). Annotations display Estimated differences and p-values from the Wilcoxon signed-rank test. This analysis suggests a more pronounced impact of managed fire treatments in the Northwestern Forested Mountains compared to the Temperate Sierras.

While the estimated reductions in dNBR from Managed Fire Operations (MFO) were more pronounced in the Northwestern Forested Ecoregion than in the Temperate Sierra region, it is crucial to acknowledge the distinct distributions in wildfire severity that influenced the MFO-designated zones (Figure 4.3). This understanding is key to interpreting our findings: in the Northwestern Forested Mountains, wildfires that impacted MFO-designated zones had median dNBR values of 365, whereas, in the Temperate Sierra region, the median dNBR was substantially lower at 175. This suggests that reductions in fire severity may be more readily measurable in more severe wildfire occurrences than in lower-severity events, a context that should be considered in any wildfire management strategy.

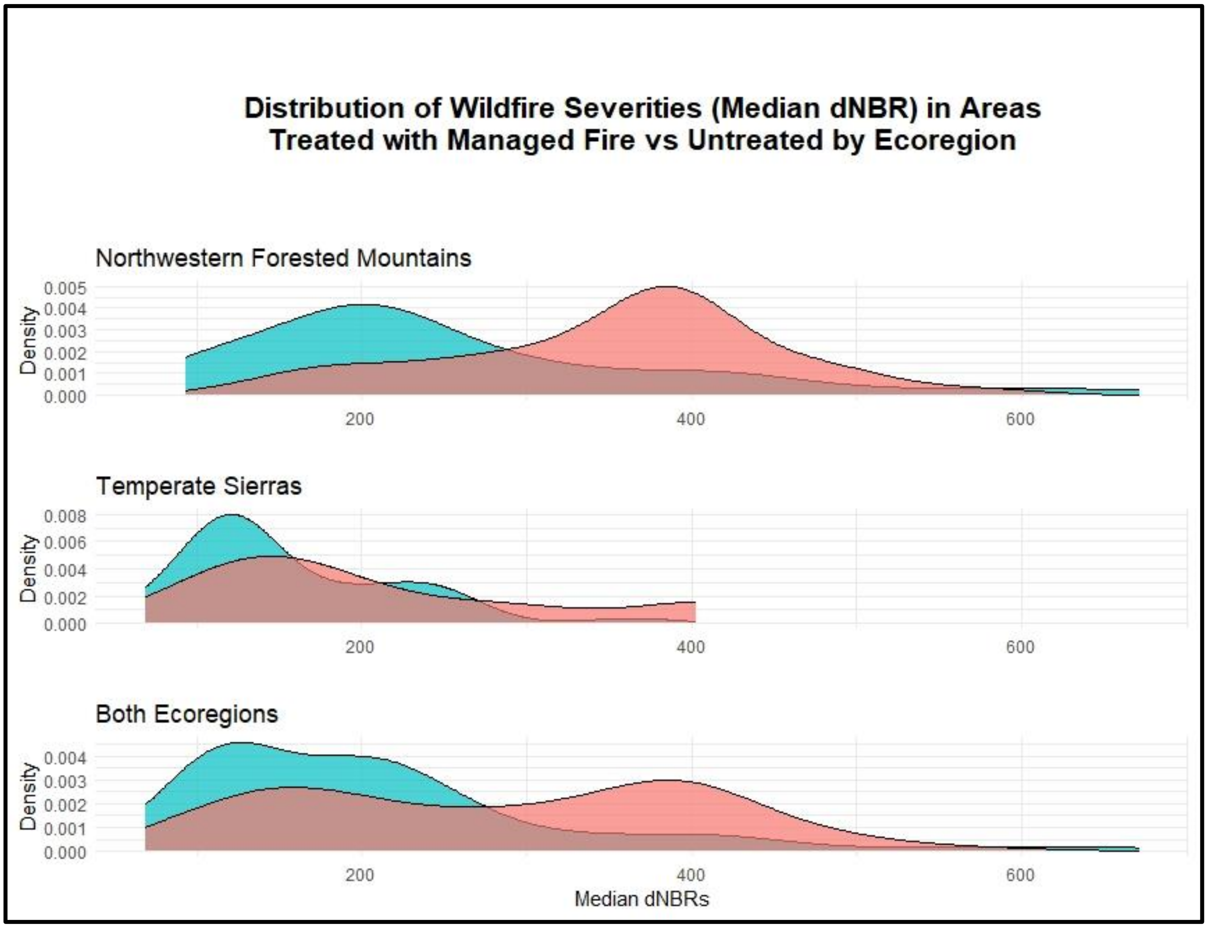


Figure 4.4. Density Distribution of Wildfire Severities Across EPA Ecoregions. This plot illustrates the density distribution of median dNBR values across different EPA ecoregions, where dashed lines indicate the median dNBR values for each ecoregion. For this dataset, wildfires that impacted lands designated as MFO were typically of higher severity in the Northwestern forested Mountains region than in the Temperate Sierras.

Ecological Context

To contextualize the findings within an ecological framework, the thresholds established through regression analyses of field-based severity CBI and dNBR values in DWCF were utilized (figure 3.4). This wildfire severity analysis spanning the CONUS dataset (n=86), comparing lands treated with MFO to untreated lands using the Wilcoxon rank test, revealed a significant dNBR estimated difference (pseudo-median) of -76. However, the ecological significance of this difference is relatively modest when juxtaposed against established thresholds of fire severity. For instance, the estimated median dNBR reduction of 76 represents approximately one-third of the predefined difference (214) that delineates the boundary between low and moderate levels of fire severity (Table 3.1). Consequently, the findings in this study suggest a moderate ecological impact of MFO treatments on reducing wildfire severity in DWCFs, with estimated differences less robust when contextualized within established severity thresholds.

The most pronounced ecological impact of MFO on reducing wildfire severity was evident upon stratifying the dataset by EPA Ecoregions of Northwestern Forested Mountains and Temperate Sierras. Within the Northwestern Forested Mountains subgroup (n=46), the median estimated dNBR difference stood at -104 (90% CI: -142 to -62), suggesting a potential ecological impact approximately midway between the predefined thresholds for low and moderate fire severity (214), as established by regression analysis of dNBR and CBI. In contrast, the Temperate Sierras subgroup manifested the least pronounced impact within this regression framework, as indicated by an estimated median dNBR difference of -38 (90% CI: -78 to -20), which is quite modest when contextualized within fire severity classification thresholds presented in (Table 3.1).

In the analysis of ecological impact across two post-fire time categories (0-15 years and 15-40 years), minimal variations were discerned. Despite the statistical significance of the estimated dNBR differences, the overlapping confidence intervals and analogous estimated differences within the 0-15 years interval (-85, 90% CI -104 to -59) and the 15-40 years interval (-56, 90% CI -113 to -7) are relatively modest when juxtaposed against the regression-derived thresholds. For instance, the dNBR values delineating moderate severity span from 148 to 362, rendering a difference of 36 dNBR between the two-time intervals notably slight. Consequently, the time intervals employed in this investigation (0-15 years and 15-40 years post-fire) appear to exert a limited influence on the quantified ecological outcomes in areas subjected to MFO treatments compared to untreated counterparts.

DISCUSSION

In DWCFs of the CONUS, wildfires from 1985-2021 in areas previously treated with managed fire showed a significant, albeit moderate, reduction in severity compared to untreated lands, with an estimated dNBR difference of -76 (p-value < .01, 90% CI: -118 to -45), representing one-third of the value needed to shift from Miller and Thode's (2007) moderate to low severity classification. In recent decades, the number of areas burned by wildland fires and wildfire severity has increased in the western US, raising concerns about shifting wildfire regimes (Abatzoglou et al., 2021; Parks et al., 2023). One of the most frequently advocated and implemented land management practices aimed at mitigating the adverse ecological effects resulting from increased severity is the implementation of managed fire operations; however, limited research quantifies the effectiveness of this practice (McKinney et al., 2022). While recognizing the uncertainty in comparing satellite-derived median wildfire severities to field-based severity metrics, these findings indicate the measured effectiveness of MFO in reducing wildfire severity impacts over the past 40 years.

In the DWCF from 1985-2021, wildfires impacting managed fire-treated areas, stratified into two intervals (0-15 years and 15-40 years post-treatment), demonstrated moderate reductions in wildfire severity proxies and marginal differences between the temporal categories. Observational studies suggest that the time since managed fire operations positively correlates with wildfire severities due to likely increases in fuel structure recovery over time (Fernandes, 2015). However, quantifying this relationship is limited in the literature, often attributed to challenges such as the scarcity of treatments older than 30 years and limited fuel succession in semiarid regions (Prichard & Kennedy, 2014). The minimal difference in wildfire severities between the 0-15 years and 15-40 years post-treatment categories in this study supports the notion that a more recent MFO in an area, compared to an older MFO, may not necessarily reduce the probability of severe wildfire events (McKinney et al., 2022). Consequently, areas previously considered effective in buffering severe wildfire events may be less efficacious than previously believed (McKinney et al., 2022).

Additionally, differing geographical settings revealed significant variations in the ecological impact of MFO. The Northwestern Forested Mountains ecoregion exhibited a larger median difference in fire severity compared to the Temperate Sierras, indicating the influence of regional environmental factors on fire behavior and response to management interventions, as documented in previous studies (Parks et al., 2014). Additionally, this study emphasized pronounced differences in wildfire severity affecting MFOs across regions. This highlights the importance of considering typical or probable wildfire severity distributions, along side other regional variations such as climate, weather, and topography, when formulating and prioritizing MFO strategies to mitigate the adverse ecological effects of high-severity wildfire events.

Application to Fire Management in DWCF

The findings of this study carry substantial implications for fire and ecological restoration management strategies in DWCF. Firstly, while MFOs have effectively reduced wildfire severity, the observed reductions remain modest in the context of ecological impact. This suggests that relying solely on MFOs may not be sufficient to mitigate the negative ecological impacts of the escalating severity of wildfires in DWCF. Consequently, a multifaceted approach integrating MFOs with other land management practices, such as mechanical fuel reduction and forest restoration, may offer a more effective strategy for reducing widespread high-severity wildfires (McWethy et al., 2019; Prichard et al., 2021; Prichard & Kennedy, 2014).

Secondly, the minimal difference in wildfire severities between the 0-15 years and 15-40 years post-MFO suggests that the efficacy of MFOs may not necessarily diminish over time. This, along with recent studies (McKinney et al., 2022), challenges the conventional assumption that a more recent MFO in an area, compared to an older MFO, would inherently reduce the probability of severe wildfire impact in DWCF ecosystems.

Thirdly, fuel loading and its management are critical factors in determining fire severity; however, our study underscores the complex interplay between fuel and weather in influencing wildfire behavior and severity (Bowman et al., 2009). By evaluating fire severity MFO designated lands, which are representative of reduced fuel loads, I found only marginal differences in fire severity, except for the Northwestern Forested Mountains ecoregion. These findings are consistent with existing literature, suggesting that climatic and weather conditions may exert a more pronounced influence on the overall ecosystem impact of wildfires than underlying fuel loads (Abatzoglou et al., 2021).

Lastly, the substantial regional variability in the ecological impact of MFOs underscores the need to tailor fire management strategies to specific regional environmental conditions. The Northwest Forested Mountains region exhibited the most substantial reduction in wildfire severity impacting MFO, suggesting that MFO may be more effective in this region. This could be due to several factors, including past disturbance history, fuel loading levels, specific fire regime characteristics, or how extreme droughts may affect areas less adapted to these climate anomalies. This implies that a one-size-fits-all approach to fire management may not be practical and that management strategies should be regionally customized to maximize their effectiveness in mitigating the negative ecological impacts of high-severity wildfire events (Syphard et al., 2017; Thompson et al., 2020).

In conclusion, this study provides evidence that managed fire operations show promise in reducing wildfire severity in DWCF. However, their effectiveness is moderate within established severity thresholds and subject to regional fire regime variability. Thus, a comprehensive and adaptive fire management strategy that incorporates various land management practices and is tailored to regional environmental conditions is essential for effectively mitigating the adverse ecological effects of increasing wildfire severities in DWCF.

Limitations and Considerations

It is necessary to acknowledge the limitations of this study, which are influenced by constraints related to data availability and quality, as well as the complexity of natural fire regimes. Firstly, satellite-derived datasets inherently contain estimates, introducing inherent uncertainties. The study utilized five geospatial datasets—DWCF, aspect, slope, fire perimeters, and fire severities—each introducing uncertainties not explicitly addressed within the study design. Future research should develop and incorporate methods such as uncertainty raster layers to account for and enhance data reliability.

Secondly, wildfire behavior is influenced by fuel availability, topography, and fire weather. (Bowman et al., 2009). While this study assessed reduced fuel availability due to MFO within a relatively constant topography of moderate gradient south-facing slopes, fire weather exhibits temporal variability due to wind events, temperature and humidity fluctuations, and atmospheric changes. Representing fire severity as an aggregate of the total burned wildfire incident area limits the analysis of specific fire season climates, fire weather events, and climate change impacts on fire severity departures. This study did not analyze fire season climate or climate change, though determining the relative influence of fuel reduction from MFO versus climate change would be valuable.

Thirdly, the variability in MFO goals and outcomes across sites poses challenges. Data on specific goals, outcomes, and management history for each MFO operation are lacking, necessitating the grouping of all operations despite nuanced differences. Incorporating current fuel load data and MFO treatment history within and across MFOs would reveal valuable relationships between fire severity and these management actions.

Lastly, the classification of fire severity varies across studies. Miller and Thode's partitions of low, moderate, and high severity are comparatively interpretable but may lack nuance, while Sikkink and Keane (2012) developed nine classification groups based on soil heating, intensity, fire duration, and fuel consumption. Additionally, the CBI protocol offers a consistent methodology for rapidly assessing relative severity and allows for quick sampling in many locations; variability can arise due to the protocol's reliance on ocular estimates. For example, Korhonen et al. (2006) have shown discrepancies between ocular and instrumented measurements of forest vegetation, potentially amplified by data collection variations across organizations and personnel.

CONCLUSIONS

Despite significant efforts in fire prevention, dry conifer forests remain susceptible to high-severity fires due to fuel accumulation and extreme weather conditions, jeopardizing their long-term survival (Hagmann et al., 2022; Higuera & Abatzoglou, 2021). MFO, by reducing available fuels, can offer substantial benefits to these ecosystems, including a diminished threat of high-severity wildfires and enhanced ecosystem resilience. However, quantifying their effectiveness remains a challenge.

MFOs and their outcomes are shaped by a multifaceted interplay of factors, encompassing the characteristics of the burning environment (e.g., fuels, fire weather, and topography), ignition properties, the specific attributes of the impacted ecosystems, and the enduring impacts of past disturbances, climatic conditions, soil properties, and land management practices (Bonner et al., 2021; O'Brien et al., 2018). Given the intricacy of these factors and the temporal and resource constraints that managers often encounter, evaluating, interpreting, and applying findings from scientific studies can pose significant challenges. Nonetheless, an expanding body of quantitative evidence underscores the importance of MFO as a crucial strategy in reducing fuel loads and facilitating the restoration of natural fire regimes in DWCF, fostering biodiversity, enhancing wildlife habitat, and bolstering overall ecosystem health. (Cannon et al., 2018; Huffman et al., 2017; Jain et al., 2007; Kennedy & Johnson, 2014; Latif et al., 2020; Lydersen et al., 2017; Stevens-Rumann & Morgan, 2016; Tubbesing et al., 2019)

With the increasing research on MFO effectiveness in mitigating wildfire severity, questions about the appropriate scale of implementation arise. Recent studies suggest that the current pace and extent of treatment efforts may not be sufficient to restore resilience in dry conifer forests (Haugo et al., 2015; North et al., 2012, 2015; Prichard et al., 2021). Large-scale regional restoration initiatives are imperative to bolster forest resilience against landscape-scale recurrent wildfires. Projects like the Collaborative Forest Landscape Restoration Program (CFLRP) in the Pacific Northwest prioritize prescribed fires to improve forest health and reduce high-severity wildfire risks across extensive forested areas (Schultz et al., 2012). Similarly, the United States Forest Service has launched a management framework (Wildfire Crisis Implementation Plan) to identify and target high-risk forested watersheds, facilitating an expansion of prescribed and cultural fire to cover 20 million acres by 2032 (USFS, 2022). Continued large-scale implementation and initiatives targeting specific locations and ecosystems are essential to address restoration scale and overcome cross-cultural stakeholder challenges (Prichard et al., 2021; Sample et al., 2022).

Furthermore, it is crucial to recognize that the effectiveness and impact of MFOs can vary significantly across different biophysical settings and regions. Factors such as climate conditions, soil types, vegetation composition, and topographical features can influence the outcomes of MFOs and the subsequent ecosystem responses. Therefore, tailored approaches accounting for these diverse conditions are essential for optimizing the benefits of MFOs and ensuring their successful implementation across various landscapes and ecosystems (Dillon et al., 2011; Parks et al., 2014; Parisien, et al., 2014).

These findings underscore the importance of incorporating MFO into wildfire management strategies to reduce fire severity and highlight nuanced perspectives. This aligns with previous research emphasizing the significance of prescribed burning in modifying wildfire fire behavior (Cannon et al., 2018; Tubbesing et al., 2019). However, this analysis suggests time since MFO may not conclusively contribute to diminished fire severity, and the overall ecosystem impact of wildfires is driven by the interplay of climate, weather, topographic, and historic management conditions alongside underlying fuel loads. This highlights the need for further research to refine management practices and assess long-term impacts (McKinney et al., 2022). Additionally, our study contributes to the broader understanding of fire regimes in natural ecosystems, emphasizing the complex interactions between human activities, natural fire processes, and ecosystem resilience. Within the Anthropocene epoch, characterized by unparalleled human influence on natural ecosystems, our extensive evaluation of MFO's effectiveness enhances understanding and supports evidence-based fire management strategies.

REFERENCES CITED

Abatzoglou, J. T., Battisti, D. S., Williams, A. P., Hansen, W. D., Harvey, B. J., & Kolden, C. A. (2021). Projected increases in western US forest fire despite growing fuel constraints. *Communications Earth & Environment*, *2*(1), 1–8. https://doi.org/10.1038/s43247-021-00299-0

Agee, J. K. (1993). *Fire Ecology of Pacific Northwest Forests*. Island Press.

Agee, J.K. (1998). The Landscape Ecology of Western Forest Fire Regimes. *Northwest Science*, *72*, 24–34.

Andrews, H. J., & Cowlin, R. W. (1940). *Forest resources of the Douglas-fir region /.* https://doi.org/10.5962/bhl.title.65691

Arno, S., & Allison-Bunnell, S. (2002). Flames in Our Forest: Disaster Or Renewal? *Bibliovault OAI Repository, the University of Chicago Press*.

Barbour, M. G. (Ed.). (2007). *North American terrestrial vegetation* (2. ed., repr). Cambridge Univ. Press.

Belsky, A. J., Matzke, A., & Uselman, S. (1999). Survey of livestock influences on stream and riparian ecosystems in the western United States. *Journal of Soil and Water Conservation*, *54*(1), 419–431.

Bonner, S. R., Hoffman, C. M., Kane, J. M., Varner, J. M., Hiers, J. K., O'Brien, J. J., Rickard, H. D., Tinkham, W. T., Linn, R. R., Skowronski, N., Parsons, R. A., & Sieg, C. H. (2021). Invigorating Prescribed Fire Science Through Improved Reporting Practices. *Frontiers in Forests and Global Change*, *4*. https://doi.org/10.3389/ffgc.2021.750699

Bowman, D. M. J. S., Balch, J. K., Artaxo, P., Bond, W. J., Carlson, J. M., Cochrane, M. A., D'Antonio, C. M., Defries, R. S., Doyle, J. C., Harrison, S. P., Johnston, F. H., Keeley, J. E., Krawchuk, M. A., Kull, C. A., Marston, J. B., Moritz, M. A., Prentice, I. C., Roos, C. I., Scott, A. C., … Pyne, S. J. (2009). Fire in the Earth system. *Science (New York, N.Y.)*, *324*(5926), 481–484. https://doi.org/10.1126/science.1163886

Brubaker, L. B., Higuera, P. E., Rupp, T. S., Olson, M. A., Anderson, P. M., & Hu, F. S. (2009). Linking sediment-charcoal records and ecological modeling to understand causes of fire-regime change in boreal forests. *Ecology*, *90*(7), 1788–1801. https://doi.org/10.1890/08-0797.1

Canada, N. R. (n.d.). *Canadian Wildland Fire Information System*. Retrieved March 26, 2024, from https://cwfis.cfs.nrcan.gc.ca/home

Cannon, J. B., Barrett, K. J., Gannon, B. M., Addington, R. N., Battaglia, M. A., Fornwalt, P. J., Aplet, G. H., Cheng, A. S., Underhill, J. L., Briggs, J. S., & Brown, P. M. (2018). Collaborative restoration effects on forest structure in ponderosa pine-dominated forests of Colorado. *Forest Ecology and Management*, *424*, 191–204. https://doi.org/10.1016/j.foreco.2018.04.026

Cansler, C. A., & McKenzie, D. (2012). How Robust Are Burn Severity Indices When Applied in a New Region? Evaluation of Alternate Field-Based and Remote-Sensing Methods. *Remote Sensing*, *4*(2), 456–483. https://doi.org/10.3390/rs4020456

Cansler, C. A., & McKenzie, D. (2014). Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern Cascade Range, USA. Ecological Applications, 24(5), 1037-1056.

Cova, G., Kane, V. R., Prichard, S., North, M., & Cansler, C. A. (2023). The outsized role of California's largest wildfires in changing forest burn patterns and coarsening ecosystem scale. *Forest Ecology and Management*, *528*, 120620. https://doi.org/10.1016/j.foreco.2022.120620

Crutzen, P. J., & Stoermer, E. F. (2000). The "Anthropocene". Global Change Newsletter, 41,

17-18.

Dillon, G. K., Holden, Z. A., Morgan, P., Crimmins, M. A., Heyerdahl, E. K., & Luce, C. H. (2011). Both topography and climate affected forest and woodland burn severity in two regions of the western US, 1984 to 2006. *Ecosphere*, *2*(12), art130. https://doi.org/10.1890/ES11-00271.1

Dillon, G. K., Panunto, M. H., Davis, B., Morgan, P., Birch, D. S., & Jolly, W. M. (2020). Development of a Severe Fire Potential map for the contiguous United States. *Gen. Tech. Rep. RMRS-GTR-415. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 107 p.*, *415*. https://www.fs.usda.gov/research/treesearch/60733

Eidenshink, J., Schwind, B., Brewer, K., Zhu, Z.-L., Quayle, B., & Howard, S. (2007). A Project for Monitoring Trends in Burn Severity. *Fire Ecology*, *3*(1), Article 1. https://doi.org/10.4996/fireecology.0301003

Fernandes, P. M. (2015). Empirical Support for the Use of Prescribed Burning as a Fuel Treatment. *Current Forestry Reports*, *1*(2), 118–127. https://doi.org/10.1007/s40725-015-0010-z

Franklin, J. F., & Dyrness, C. T. (1973). Natural vegetation of Oregon and Washington. *Gen. Tech. Rep. PNW-GTR-008. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 427 p*, *008*. https://www.fs.usda.gov/research/treesearch/26203

French, N. H. F., Kasischke, E. S., Hall, R. J., Murphy, K. A., Verbyla, D. L., Hoy, E. E., & Allen, J. L. (2008). Using Landsat data to assess fire and burn severity in the North

American boreal forest region: An overview and summary of results. *International Journal of Wildland Fire*, *17*(4), 443–462. https://doi.org/10.1071/WF08007

Fulé, P. Z., Crouse, J. E., Heinlein, T. A., Moore, M. M., Covington, W. W., & Verkamp, G. (2003). Mixed-severity fire regime in a high-elevation forest of Grand Canyon, Arizona, USA. *Landscape Ecology*, *18*(5), 465–486. https://doi.org/10.1023/A:1026012118011

Hagmann, R. K., Hessburg, P. F., Prichard, S. J., Povak, N. A., Brown, P. M., Fulé, P. Z., Keane, R. E., Knapp, E. E., Lydersen, J. M., Metlen, K. L., Reilly, M. J., Sánchez Meador, A. J., Stephens, S. L., Stevens, J. T., Taylor, A. H., Yocom, L. L., Battaglia, M. A., Churchill, D. J., Daniels, L. D., … Waltz, A. E. M. (2021). Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests. *Ecological Applications*, *31*(8), e02431. https://doi.org/10.1002/eap.2431

Hagmann, R. K., Hessburg, P. F., Salter, R. B., Merschel, A. G., & Reilly, M. J. (2022). Contemporary wildfires further degrade resistance and resilience of fire-excluded forests. *Forest Ecology and Management*, *506*, 119975. https://doi.org/10.1016/j.foreco.2021.119975

Halofsky, J. E., Peterson, D. L., & Harvey, B. J. (2020). Changing wildfire, changing forests: The effects of climate change on fire regimes and vegetation in the Pacific Northwest, USA. *Fire Ecology*, *16*(1), 4. https://doi.org/10.1186/s42408-019-0062-8

Harrod, R., Fonda, R., & McGrath, M. (2009). Vegetation Response to Thinning and Burning in a Ponderosa Pine Forest, Washington. *Northwest Science*, *82*, 141–150. https://doi.org/10.3955/0029-344X-82.2.141

Haugo, R., Zanger, C., DeMeo, T., Ringo, C., Shlisky, A., Blankenship, K., Simpson, M., Mellen-McLean, K., Kertis, J., & Stern, M. (2015). A new approach to evaluate forest structure restoration needs across Oregon and Washington, USA. *Forest Ecology and Management*, *335*, 37–50. https://doi.org/10.1016/j.foreco.2014.09.014

He, T., Belcher, C. M., Lamont, B. B., & Lim, S. L. (2016). A 350-million-year legacy of fire adaptation among conifers. *Journal of Ecology*, *104*(2), 352–363. https://doi.org/10.1111/1365-2745.12513

Hessburg, P. F., & Agee, J. K. (2003). An environmental narrative of Inland Northwest United States forests, 1800–2000. *Forest Ecology and Management*, *178*(1), 23–59. https://doi.org/10.1016/S0378-1127(03)00052-5

Hessburg, P. F., Agee, J. K., & Franklin, J. F. (2005). Dry forests and wildland fires of the inland Northwest USA: Contrasting the landscape ecology of the pre-settlement and modern eras. *Forest Ecology and Management*, *211*(1), 117–139. https://doi.org/10.1016/j.foreco.2005.02.016

Hessburg, P. F., Miller, C. L., Parks, S. A., Povak, N. A., Taylor, A. H., Higuera, P. E., Prichard, S. J., North, M. P., Collins, B. M., Hurteau, M. D., Larson, A. J., Allen, C. D., Stephens, S. L., Rivera-Huerta, H., Stevens-Rumann, C. S., Daniels, L. D., Gedalof, Z., Gray, R. W., Kane, V. R., … Salter, R. B. (2019a). Climate, Environment, and Disturbance History Govern Resilience of Western North American Forests. *Frontiers in Ecology and Evolution*, *7*. https://www.frontiersin.org/articles/10.3389/fevo.2019.00239

Hessburg, P. F., Miller, C. L., Parks, S. A., Povak, N. A., Taylor, A. H., Higuera, P. E., Prichard, S. J., North, M. P., Collins, B. M., Hurteau, M. D., Larson, A. J., Allen, C. D., Stephens, S. L., Rivera-Huerta, H., Stevens-Rumann, C. S., Daniels, L. D., Gedalof, Z., Gray, R. W., Kane, V. R., … Salter, R. B. (2019b). Climate, Environment, and Disturbance History Govern Resilience of Western North American Forests. *Frontiers in Ecology and Evolution*, *7*. https://doi.org/10.3389/fevo.2019.00239

Hessburg, P. F., Smith, B. G., Salter, R. B., Ottmar, R. D., & Alvarado, E. (2000). Recent changes (1930s–1990s) in spatial patterns of interior northwest forests, USA. *Forest Ecology and Management*, *136*(1), 53–83. https://doi.org/10.1016/S0378-1127(99)00263-7

Heyerdahl, E. K., Brubaker, L. B., & Agee, J. K. (2002). Annual and decadal climate forcing of historical fire regimes in the interior Pacific Northwest, USA. *The Holocene*, *12*(5), 597–604. https://doi.org/10.1191/0959683602hl570rp

Higuera, P. E., & Abatzoglou, J. T. (2021). Record-setting climate enabled the extraordinary 2020 fire season in the western United States. *Global Change Biology*, *27*(1), 1–2. https://doi.org/10.1111/gcb.15388

Higuera, P. E., Cook, M. C., Balch, J. K., Stavros, E. N., Mahood, A. L., & St. Denis, L. A. (2023). Shifting social-ecological fire regimes explain increasing structure loss from Western wildfires. *PNAS Nexus*, *2*(3), pgad005. https://doi.org/10.1093/pnasnexus/pgad005

Holden, Z. A., Morgan, P., & Evans, J. S. (2009). A predictive model of burn severity based on 20-year satellite-inferred burn severity data in a large southwestern US wilderness area. *Forest Ecology and Management*, *258*(11), 2399–2406. https://doi.org/10.1016/j.foreco.2009.08.017

Huang, C., Goward, S. N., Masek, J. G., Thomas, N., Zhu, Z., & Vogelmann, J. E. (2010). An automated approach for reconstructing recent forest disturbance history using dense Landsat time series stacks. *Remote Sensing of Environment*, *114*(1), 183–198. https://doi.org/10.1016/j.rse.2009.08.017

Huffman, D. W., Sánchez Meador, A. J., Stoddard, M. T., Crouse, J. E., & Roccaforte, J. P. (2017). Efficacy of resource objective wildfires for restoration of ponderosa pine (*Pinus ponderosa*) forests in northern Arizona. *Forest Ecology and Management*, *389*, 395–403. https://doi.org/10.1016/j.foreco.2016.12.036

Hunter, M. E., & Robles, M. D. (2020). Tamm review: The effects of prescribed fire on wildfire regimes and impacts: A framework for comparison. *Forest Ecology and Management*, *475*, 118435. https://doi.org/10.1016/j.foreco.2020.118435

International Commission on Stratigraphy. (n.d.). Retrieved March 30, 2024, from https://stratigraphy.org/news/152

Jain, T., Juillerat, M., Sandquist, J., Ford, M., Sauer, B., Mitchell, R., Mcavoy, S., Hanley, J., & David, J. (2007). *Vegetation and soil effects from prescribed, wild, and combined fire events along a ponderosa pine and grassland mosaic*.

Kennedy, M. C., & Johnson, M. C. (2014). Fuel treatment prescriptions alter spatial patterns of fire severity around the wildland-urban interface during the Wallow Fire, Arizona, USA. *Forest Ecology and Management*, *318*, 122–132. https://doi.org/10.1016/j.foreco.2014.01.014

Key, C., & Benson, N. (2006). Landscape Assessment: Ground measure of severity, the Composite Burn Index; and Remote sensing of severity, the Normalized Burn Ratio. In *FIREMON: Fire Effects Monitoring and Inventory System* (p. LA 1-51).

Key, C. H., & Benson, N. C. (2006). *Landscape Assessment (LA)*.

Kolden, C. A., Smith, A. M. S., & Abatzoglou, J. T. (2015). Limitations and utilisation of Monitoring Trends in Burn Severity products for assessing wildfire severity in the USA. *International Journal of Wildland Fire*, *24*(7), 1023. https://doi.org/10.1071/WF15082

Komarek, E. V. (1973). Ancient fires. *Proceedings of the Twelfth Tall Timbers Fire Ecology Conference. Tall Timbers, Tallahassee, Florida*, 219–240. https://talltimbers.org/wp-content/uploads/2018/09/219-Komarek1972_op.pdf

Korhonen, L., Korhonen, K. T., Rautiainen, M., & Stenberg, P. (2006). *Estimation of forest canopy cover: A comparison of field measurement techniques*. https://jukuri.luke.fi/bitstream/handle/10024/532615/Lkorhonen.pdf?sequence=1

Latif, Q. S., Truex, R. L., Sparks, R. A., & Pavlacky, D. C. (2020). Dry conifer forest restoration benefits Colorado Front Range avian communities. *Ecological Applications*, *30*(6), e02142. https://doi.org/10.1002/eap.2142

Lentile, L. B., Morgan, P., Hudak, A. T., Bobbitt, M. J., Lewis, S. A., Smith, A. M. S., & Robichaud, P. R. (2007). Post-Fire Burn Severity and Vegetation Response Following Eight Large Wildfires Across the Western United States. *Fire Ecology*, *3*(1), Article 1. https://doi.org/10.4996/fireecology.0301091

Lockwood, J. L., Powell, R. D., Nott, M. P., & Pimm, S. L. (1997). Assembling Ecological Communities in Time and Space. *Oikos*, *80*(3), 549–553. https://doi.org/10.2307/3546628

Lydersen, J. M., Collins, B. M., Brooks, M. L., Matchett, J. R., Shive, K. L., Povak, N. A., Kane, V. R., & Smith, D. F. (2017). Evidence of fuels management and fire weather influencing fire severity in an extreme fire event. *Ecological Applications. 27(7): 2013-2030*, *27*, 2013–2030. https://doi.org/10.1002/eap.1586

Mallek, C., Safford, H., Viers, J., & Miller, J. (2013). Modern departures in fire severity and area vary by forest type, Sierra Nevada and southern Cascades, California, USA. *Ecosphere*, *4*(12), art153. https://doi.org/10.1890/ES13-00217.1

Margolis, E., & Balmat, J. (2009). Fire history and fire-climate relationships along a fire regime gradient in the Santa Fe Municipal Watershed, NM, USA. *Forest Ecology and Management*, *258*, 2416–2430. https://doi.org/10.1016/j.foreco.2009.08.019

Marris, E. (2011). Rethinking the Anthropocene. Nature, 471(7336), 32-34.

McKinney, S. T., Abrahamson, I., Jain, T., & Anderson, N. (2022). A systematic review of empirical evidence for landscape-level fuel treatment effectiveness. *Fire Ecology*, *18*(1), 21. https://doi.org/10.1186/s42408-022-00146-3

McWethy, D. B., Schoennagel, T., Higuera, P. E., Krawchuk, M., Harvey, B. J., Metcalf, E. C., Schultz, C., Miller, C., Metcalf, A. L., Buma, B., Virapongse, A., Kulig, J. C., Stedman, R. C., Ratajczak, Z., Nelson, C. R., & Kolden, C. (2019). Rethinking resilience to wildfire. *Nature Sustainability. Doi: 10.1038/S41893-019-0353-8.* https://doi.org/10.1038/s41893-019-0353-8

Miller, J. D., Safford, H. D., Crimmins, M., & Thode, A. E. (2009). Quantitative Evidence for Increasing Forest Fire Severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. *Ecosystems*, *12*(1), 16–32. https://doi.org/10.1007/s10021-008-9201-9

Miller, J. D., & Thode, A. E. (2007). Quantifying burn severity in a heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). *Remote Sensing of Environment*, *109*(1), 66–80. https://doi.org/10.1016/j.rse.2006.12.006

Moritz, M. A., Hessburg, P. F., & Povak, N. A. (2011). Native Fire Regimes and Landscape Resilience. In D. McKenzie, C. Miller, & D. A. Falk (Eds.), *The Landscape Ecology of Fire* (pp. 51–86). Springer Netherlands. https://doi.org/10.1007/978-94-007-0301-8_3

Morrison, P. H., & Swanson, F. J. (1990). *Fire history and pattern in a Cascade Range landscape.* PNW-GTR-254. https://doi.org/10.2737/PNW-GTR-254

Naficy, C., Sala, A., Keeling, E. G., Graham, J., & DeLuca, T. H. (2010). Interactive effects of historical logging and fire exclusion on ponderosa pine forest structure in the northern Rockies. *Ecological Applications*, *20*(7), 1851–1864. https://doi.org/10.1890/09-0217.1

North, M. P., Collins, B. M., & Stephens, S. L. (2012). Using fire to increase the scale, benefits and future maintenance of fuels treatments. *Journal of Forestry. 110(7): 492-401*, *110*, 392–401. https://doi.org/10.5849/jof.12-021

North, M. P., Stephens, S. L., Collins, B. M., Agee, J. K., Aplet, G., Franklin, J. F., & Fulé, P. Z. (2015). Reform forest fire management. *Science*, *349*(6254), 1280–1281. https://doi.org/10.1126/science.aab2356

O'Brien, J. J., Hiers, J. K., Varner, J. M., Hoffman, C. M., Dickinson, M. B., Michaletz, S. T., Loudermilk, E. L., & Butler, B. W. (2018). Advances in Mechanistic Approaches to Quantifying Biophysical Fire Effects. *Current Forestry Reports*, *4*(4), 161–177. https://doi.org/10.1007/s40725-018-0082-7

Parks, S. A., Dillon, G. K., & Miller, C. (2014). A New Metric for Quantifying Burn Severity: The Relativized Burn Ratio. *Remote Sensing*, *6*(3), Article 3. https://doi.org/10.3390/rs6031827

Parks, S. A., Holsinger, L. M., Blankenship, K., Dillon, G. K., Goeking, S. A., & Swaty, R. (2023). Contemporary wildfires are more severe compared to the historical reference period in western US dry conifer forests. *Forest Ecology and Management*, *544*, 121232. https://doi.org/10.1016/j.foreco.2023.121232

Parks, S. A., Parisien, M.-A., Miller, C., & Dobrowski, S. Z. (2014). Fire Activity and Severity in the Western US Vary along Proxy Gradients Representing Fuel Amount and Fuel Moisture. *PLOS ONE*, *9*(6), e99699. https://doi.org/10.1371/journal.pone.0099699

Pickett, S. T., & White, P. S. (1985). *The ecology of natural disturbance and patch dynamics*. Academic Press.

Pierce, J. L., Meyer, G. A., & Jull, A. J. T. (2004). Fire-induced erosion and millennial-scale climate change in northern ponderosa pine forests. *Nature*, *432*(7013), 87–90. https://doi.org/10.1038/nature03058

Prichard, S. J., Hessburg, P. F., Hagmann, R. K., Povak, N. A., Dobrowski, S. Z., Hurteau, M. D., Kane, V. R., Keane, R. E., Kobziar, L. N., Kolden, C. A., North, M., Parks, S. A., Safford, H. D., Stevens, J. T., Yocom, L. L., Churchill, D. J., Gray, R. W., Huffman, D. W., Lake, F. K., & Khatri-Chhetri, P. (2021). Adapting western North American forests to climate change and wildfires: 10 common questions. *Ecological Applications*, *31*(8), e02433. https://doi.org/10.1002/eap.2433

Prichard, S. J., & Kennedy, M. C. (2014). Fuel treatments and landform modify landscape patterns of burn severity in an extreme fire event. *Ecological Applications*, *24*(3), 571–590. https://doi.org/10.1890/13-0343.1

Rothermel, R. C. (1972). A mathematical model for predicting fire spread in wildland fuels. *Res. Pap. INT-115. Ogden, UT: U.S. Department of Agriculture, Intermountain Forest and Range Experiment Station. 40 p.*, *115*. https://www.fs.usda.gov/research/treesearch/32533

Sample, M., Thode, A. E., Peterson, C., Gallagher, M. R., Flatley, W., Friggens, M., Evans, A., Loehman, R., Hedwall, S., Brandt, L., Janowiak, M., & Swanston, C. (2022). Adaptation Strategies and Approaches for Managing Fire in a Changing Climate. *Climate*, *10*(4), 58. https://doi.org/10.3390/cli10040058

Schoennagel, T., Veblen, T. T., & Romme, W. H. (2004). The Interaction of Fire, Fuels, and Climate across Rocky Mountain Forests. *BioScience*, *54*(7), 661–676. https://doi.org/10.1641/0006-3568(2004)054[0661:TIOFFA]2.0.CO;2

Schoennagel, T., Balch, J. K., Brenkert-Smith, H., Dennison, P. E., Harvey, B. J., Krawchuk, M. A., ... & Moritz, M. A. (2017). Adapt to more wildfires in western North American forests as climate changes. Proceedings of the National Academy of Sciences, 114(18), 4582-4590.

Schultz, C. A., Jedd, T., & Beam, R. D. (2012). The Collaborative Forest Landscape Restoration Program: A History and Overview of the First Projects. *Journal of Forestry*, *110*(7), 381–391. https://doi.org/10.5849/jof.11-082

Scott, A. C. (2000). The Pre-Quaternary history of fire. *Palaeogeography, Palaeoclimatology, Palaeoecology*, *164*(1–4), 281–329. https://doi.org/10.1016/S0031-0182(00)00192-9

Smith, C. S. & Craig S. Smith, Unknown Location. (1983). *4300 Year History of Vegetation, Climate, and Fire From Blue Lake, Nez Perce County, Idaho* (Utah Division of State History, Salt Lake City, Utah).

Soil Survey Manual | Natural Resources Conservation Service. (2020). Retrieved March 28, 2024, from https://www.nrcs.usda.gov/resources/guides-and-instructions/soil-survey-manual

Sommers, W., Coloff, S., & Conard, S. (2011). Synthesis of Knowledge: Fire History and Climate Change. *Joint Fire Science Program Synthesis Reports*. https://digitalcommons.unl.edu/jfspsynthesis/19

Steffen, W., Grinevald, J., Crutzen, P., & McNeill, J. (2011). The Anthropocene: Conceptual and historical perspectives. *Philosophical Transactions: Mathematical, Physical and Engineering Sciences*, *369*(1938), 842–867.

Stephens, S. L., Collins, B. M., Fettig, C. J., Finney, M. A., Hoffman, C. M., Knapp, E. E., North, M. P., Safford, H., & Wayman, R. B. (2018). Drought, tree mortality, and wildfire in forests adapted to frequent fire. *BioScience. 68(2): 77-88*, *68*, 77–88. https://doi.org/10.1093/biosci/bix146

Stephens, S. L., Burrows, N., Buyantuyev, A., Gray, R. W., Keane, R. E., Kubian, R., ... & Collins, B. M. (2018). Temperate and boreal forest mega-fires: characteristics and challenges. Frontiers in Ecology and the Environment, 16(S1), S18-S26.

Stevens, J. T., Collins, B. M., Miller, J. D., North, M. P., & Stephens, S. L. (2017). Changing spatial patterns of stand-replacing fire in California conifer forests. *Forest Ecology and Management. 406: 28-36*, *406*, 28–36. https://doi.org/10.1016/j.foreco.2017.08.051

Stevens-Rumann, C., & Morgan, P. (2016). Repeated wildfires alter forest recovery of mixed-conifer ecosystems. *Ecological Applications*, *26*(6), 1842–1853. https://doi.org/10.1890/15-1521.1

Swain, D. L., Abatzoglou, J. T., Kolden, C., Shive, K., Kalashnikov, D. A., Singh, D., & Smith, E. (2023). Climate change is narrowing and shifting prescribed fire windows in western United States. *Communications Earth & Environment*, *4*(1), 1–14. https://doi.org/10.1038/s43247-023-00993-1

Swetnam, T. W., Allen, C. D., & Betancourt, J. L. (1999). Applied Historical Ecology: Using the Past to Manage for the Future. *Ecological Applications*, *9*(4), 1189–1206. https://doi.org/10.1890/1051-0761(1999)009[1189:AHEUTP]2.0.CO;2

Swetnam, T. W., & Lynch, A. M. (1993). Multicentury, Regional-Scale Patterns of Western Spruce Budworm Outbreaks. *Ecological Monographs*, *63*(4), 399–424. https://doi.org/10.2307/2937153

Taylor, A. H., & Skinner, C. N. (2003). SPATIAL PATTERNS AND CONTROLS ON HISTORICAL FIRE REGIMES AND FOREST STRUCTURE IN THE KLAMATH MOUNTAINS. *Ecological Applications*, *13*(3), 704–719. https://doi.org/10.1890/1051-0761(2003)013[0704:SPACOH]2.0.CO;2

Tubbesing, C. L., Fry, D. L., Roller, G. B., Collins, B. M., Fedorova, V. A., Stephens, S. L., & Battles, J. J. (2019). Strategically placed landscape fuel treatments decrease fire severity and promote recovery in the northern Sierra Nevada. *Forest Ecology and Management*, *436*, 45–55. https://doi.org/10.1016/j.foreco.2019.01.010

Uhl, C., & Kauffman, J. B. (1990). Deforestation, Fire Susceptibility, and Potential Tree Responses to Fire in the Eastern Amazon. *Ecology*, *71*(2), 437–449. https://doi.org/10.2307/1940299

United States: Agriculture Department: Forest Service, Irwin, L. L. (Larry L., Irwin, L. L. (Larry L., & Pacific Northwest Research Station (Portland, Or.). (1994). *Effects of Long-Term Grazing by Big Game and Livestock in the Blue Mountains Forest Ecosystem*. Agriculture Department. https://purl.fdlp.gov/GPO/LPS116250

Urza, A. K., Hanberry, B. B., & Jain, T. B. (2023). Landscape-scale fuel treatment effectiveness: Lessons learned from wildland fire case studies in forests of the western United States and Great Lakes region. *Fire Ecology*, *19*(1), 1. https://doi.org/10.1186/s42408-022-00159-y

US EPA, (2016, February 26). *Ecoregion Download Files by Region* [Data and Tools]. https://www.epa.gov/eco-research/ecoregion-download-files-region

US Forest Service. 2022. Confronting the wildfire crisis: a strategy for protecting communities and improving resilience in America's forests. FS-1187a. US Forest Service. 47 p.

Waring, R. H., & Franklin, J. F. (1979). Evergreen Coniferous Forests of the Pacific Northwest. *Science*, *204*(4400), 1380–1386. https://doi.org/10.1126/science.204.4400.1380

Westerling, A. L., Hidalgo, H. G., Cayan, D. R., & Swetnam, T. W. (2006). Warming and Earlier Spring Increase Western U.S. Forest Wildfire Activity. *Science*, *313*(5789), 940–943. https://doi.org/10.1126/science.1128834

Whitlock, C., & Knox, M. (2002). *Prehistoric Burning in the Pacific Northwest*. https://www.semanticscholar.org/paper/Prehistoric-Burning-in-the-Pacific-Northwest-Whitlock-Knox/d3341d8485d4fbc30d8b21ba0cc417e3c554ed9d

Whitlock, C., & Larsen, C. (2001). Charcoal as a Fire Proxy. In *Tracking Environmental Change Using Lake Sediments. Vol. 3: Terrestrial, Algal, and Siliceous Indicators* (Vol. 3, pp. 75–97). https://doi.org/10.1007/0-306-47668-1_5

Williams, J. N., Safford, H. D., Enstice, N., Steel, Z. L., & Paulson, A. K. (2023). High-severity burned area and proportion exceed historic conditions in Sierra Nevada, California, and adjacent ranges. *Ecosphere*, *14*(1), e4397. https://doi.org/10.1002/ecs2.4397

APPENDIX

GEOSPATIAL DATASET DESCRIPTIONS

BpS Name	BpS Model
California Montane Jeffrey Pine(-Ponderosa Pine) Woodland	10310_2_3_4_5_6_12
California Montane Jeffrey Pine(-Ponderosa Pine) Woodland	10310_7
Mediterranean California Dry-Mesic Mixed Conifer Forest and Woodland	10270_2_3_7
Mediterranean California Dry-Mesic Mixed Conifer Forest and Woodland	10270_4_5_6
Northern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest	10450_1_7_8_9
Northern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest - Larch	10452_10_19
Northern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest - Ponderosa Pine-Douglas-fir	10451_10_19
Northern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest - Ponderosa Pine-Douglas-fir	10451_20
Northern Rocky Mountain Ponderosa Pine Woodland and Savanna	10530_10_19
Northern Rocky Mountain Ponderosa Pine Woodland and Savanna - Mesic	10531_1_7_8_9
Northern Rocky Mountain Ponderosa Pine Woodland and Savanna - Xeric	10532_7_9
Northwestern Great Plains-Black Hills Ponderosa Pine Woodland and Savanna - Low Elevation Woodland	11791_29_30
Northwestern Great Plains-Black Hills Ponderosa Pine Woodland and Savanna - Savanna	11792_29_30
Southern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest and Woodland	10510_6
Southern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest and Woodland	10510_18_21
Southern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest and Woodland	10510_15_16_17_22_23_24_25
Southern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest and Woodland	10510_28
Southern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest and Woodland	10510_29
Southern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest and Woodland	10510_27
Southern Rocky Mountain Ponderosa Pine Savanna	11170_20
Southern Rocky Mountain Ponderosa Pine Savanna	11170_16_23_24
Southern Rocky Mountain Ponderosa Pine Savanna	11170_13_15_28
Southern Rocky Mountain Ponderosa Pine Savanna	11170_29
Southern Rocky Mountain Ponderosa Pine Savanna	11170_33
Southern Rocky Mountain Ponderosa Pine Savanna - North	11172_27
Southern Rocky Mountain Ponderosa Pine Savanna - South	11171_27
Southern Rocky Mountain Ponderosa Pine Woodland	10540_13_14
Southern Rocky Mountain Ponderosa Pine Woodland	10540_20
Southern Rocky Mountain Ponderosa Pine Woodland	10540_22
Southern Rocky Mountain Ponderosa Pine Woodland	10540_16_23_24
Southern Rocky Mountain Ponderosa Pine Woodland	10540_17
Southern Rocky Mountain Ponderosa Pine Woodland	10540_28
Southern Rocky Mountain Ponderosa Pine Woodland	10540_15_25
Southern Rocky Mountain Ponderosa Pine Woodland	10540_29
Southern Rocky Mountain Ponderosa Pine Woodland	10540_26
Southern Rocky Mountain Ponderosa Pine Woodland	10540_33

Southern Rocky Mountain Ponderosa Pine Woodland - North	10542_27
Southern Rocky Mountain Ponderosa Pine Woodland - South	10541_27

Table A.1. Landfire Biophysical Settings Models. 17 Landfire biophysical settings models aggregated to develop the spatial extent of Dry Western Coniferous Forests.

Application	Layer Name	Source
Dry Western Conifer Forests	LANDFIRE 2020 Biophysical Settings (BPS) CONUS	LANDFIRE, Earth Resources Observation and Science Center (EROS), U.S. Geological Survey
Existing Forest Cover	Forest Mask	Dillon et al., 2020
Aspect	LANDFIRE 2020 Aspect (Asp) CONUS	LANDFIRE, Earth Resources Observation and Science Center (EROS), U.S. Geological Survey
Slope	LANDFIRE 2020 Slope Degree (SlpD) CONUS	LANDFIRE, Earth Resources Observation and Science Center (EROS), U.S. Geological Survey
MFO/ Wildfirefire Perimeters 1984-2020	MFO/ Wildfire Perimeters 1984-2020	Eidenshink et al., 2007
Wildfire Severities (dNBR)	MTBS_Bsmosaics	Eidenshink et al., 2007
EPA Ecoregions	NA_CEC_Eco_Level2	USEPA, 2016

Table A.2. Geospatial Data Application and Sources. Geospatial data application and sources used in this study.