Review of Fuel Treatment Effectiveness in Forests and Rangelands

and a Case Study From the 2007 Megafires in Central Idaho USA

Andrew T. Hudak, Ian Rickert, Penelope Morgan,
Eva Strand, Sarah A. Lewis, Peter R. Robichaud,
Chad Hoffman, Zachary A. Holden

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Abstract

This report provides managers with the current state of knowledge regarding the effectiveness of fuel treatments for mitigating severe wildfire effects. A literature review examines the effectiveness of fuel treatments that had been previously applied and were subsequently burned through by wildfire in forests and rangelands. A case study focuses on WUI fuel treatments that were burned in the 2007 East Zone and Cascade megafires in central Idaho. Both the literature review and case study results support a manager consensus that forest thinning followed by some form of slash removal is most effective for reducing subsequent wildfire severity.

Keywords: fire severity, fuel treatment, National Fire Plan, post-fire effects, wildland-urban interface (WUI)

Authors

Andrew T. Hudak, Research Forester, Rocky Mountain Research Station, Forest Service, U.S. Department of Agriculture, Moscow, Idaho

Ian Rickert, Squad Leader, Sawtooth Interagency Hotshot Crew, Sawtooth National Forest, Twin Falls, Idaho

Penelope Morgan, Professor, Department of Forest Resources, University of Idaho, Moscow, Idaho

Eva Strand, Research Professor, Department of Rangeland Ecology and Management, University of Idaho, Moscow, Idaho

Sarah A. Lewis, Civil Engineer, Rocky Mountain Research Station, Forest Service, U.S. Department of Agriculture, Moscow, Idaho

Peter R. Robichaud, Research Engineer, Rocky Mountain Research Station, Forest Service, U.S. Department of Agriculture, Moscow, Idaho

Chad Hoffman, Instructor, Department of Forest Resources, University of Idaho, Moscow, Idaho

Zachary A. Holden, Spatial Analyst, Northern Region, Forest Service, U.S. Department of Agriculture, Missoula, Montana

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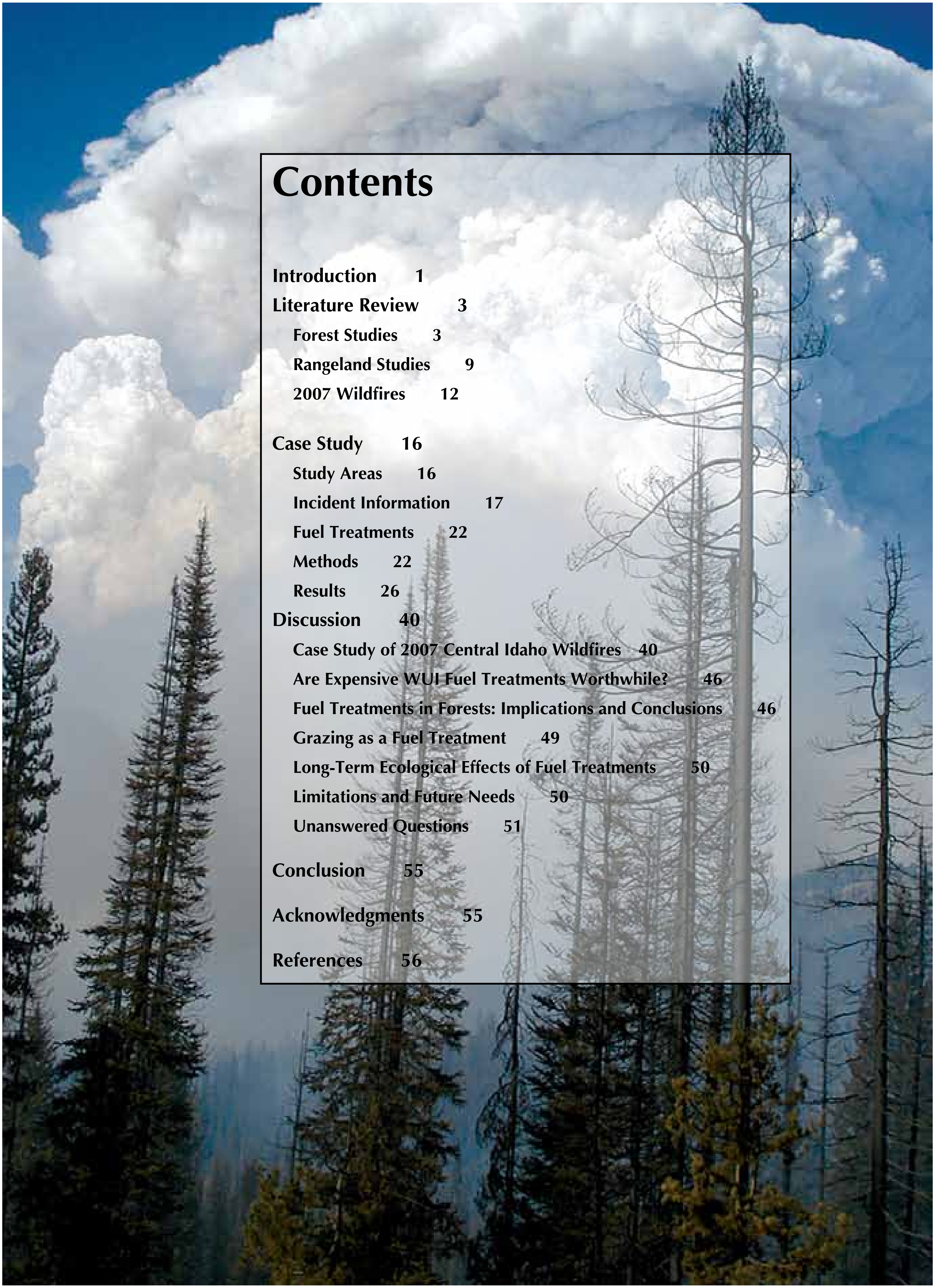

Contents

Introduction

More forest burned in Idaho in 2007 than in any year since the historic massive blazes of 1910. Fortunately, no lives were lost and all communities were protected, with only a limited number of structures lost (McCarthy and others 2008). Most of these fires were in southern and central Idaho. The 190,577 ha and 98,467 ha burned on the Payette and Boise National Forests (NF), respectively, were the largest ever recorded in a single year. The East Zone and Cascade complexes both exceeded 120,000 ha. They burned across McCall and Krassel Districts (Payette NF) and Cascade District (Boise NF) and eventually joined. Their massive size, extreme fire conditions, and rugged terrain precluded containment as a viable strategy after some initial attempts. Instead, fire crews switched to Appropriate Management Response (AMR) strategy of wildland-urban interface (WUI) and point protection (McCarthy and others 2008).

Since 2000, there has been a significant increase in the amount and cost of wildfire suppression efforts due to changes in weather conditions, fuel build up, and growth of residential development in the WUI (National Wildfire Coordinating Group 2009). The cost of fire suppression was $2.0 billion in 2006 and $1.9 billion in 2007, and the number of hectares burned was 4.00 million in 2006 and 3.78 million in 2007. The 10-year moving average of wildfire hectares doubled from 1.53 million in the 1990s to 2.89 million in the 2000s. The scale of recent fire activity in the West and northern Midwest has not been observed since the early 1900s, and this trend is expected to continue as anticipated climate change worsens the effects of extended droughts in various regions of North America (National Wildfire Coordinating Group 2009).

The national media has increased public awareness of recent research findings that fires in the western United States have grown in size and severity in the past 15 to 20 years (Bartuska and Conard 2007, Pelley 2007, Westerling and others 2006). Fire seasons have become longer west-wide (Westerling and others 2006), and land use change, fire suppression, and fuel accumulation have contributed to increasingly large, severe wildfires in the western United States in recent decades. In the face of broad and growing scientific consensus that we will continue to have large, severe fires (Helms 2007, Running 2006), fuel treatments have been promoted as an important tool for fire managers who seek a compromise between the ecological utility and socioeconomic constraints of wildfire (Brose and Wade 2002, Covington and others 2001, Fulé and others 2001).

Since the inception of the National Fire Plan in 2000, millions of dollars have been spent on fuel treatments in forests and rangelands to restore healthy ecosystems and to reduce hazardous fuel loads, especially in the WUI where people and property are likely to be threatened by wildfire. Under the National Fire Plan, 50 percent of treated lands must be in the WUI. Federal fuel treatments have grown to meet an annual target level of 1.2 million hectares treated, increased to 1.6 million hectares by including wildfire hectares that produce resource benefits, but this still amounts to less than half of the 3.2 to 4.0 million wildfire hectares that burn annually and the 4.0 to 4.9 million wildfire hectares that are projected to burn annually in the near future (National Wildfire Coordinating Group 2009).

In 2007, many large wildfires burned through fuel treatments, enabling an assessment of treatment effectiveness, which is the impetus for this and other reports (see Sidebar 3). The number of hectares treated is the current measure of success for hazardous fuel reduction projects, but this does not tell us whether and how much fuel treatments mitigate severe fire effects, what types of fuel treatments are most effective, or how long fuel treatments remain useful as fuels accumulate over time. Fuels may take 10 to 20 years to recover to pretreatment levels (Agee and Skinner 2005, Graham and others 2004), but this will vary by ecosystem, depending on rates of production, decomposition, and rates of plant establishment and mortality.

Fuel treatments include physical alteration of vegetation with the intent of reducing the probability of extreme fire behavior (Graham and others 1999, 2004). Treatments to reduce fire hazard often focus on thinning from below to reduce vertical (ladder fuels) and horizontal continuity of fuels as well as the amount of fuel on the ground. Mechanical treatments, such as mastication, chipping, piling by hand or machine, and compaction, and burning treatments, including piling and burning and broadcast burning, are designed to reduce the amount of fuel available to burn in subsequent wildfires (USDA FS 2005). Grazing is a common treatment in grasslands and shrublands. Treatments are designed to reduce the intensity and severity of fires (Graham and others 1999, 2004). All other fuel treatment goals such as improving firefighting efficiency, reducing risk, and mitigating fire severity are derivatives of moderating fire behavior.

In this paper, we focus on fuel treatment effectiveness in mitigating post-fire effects. Fuel treatment goals commonly include reducing wildfire risks to communities and the environment and improving ecosystem resiliency to wildfire effects (USDA USDOI WGA 2002, 2006). Treatment effectiveness can be judged in terms of three criteria: (1) Did the treatment reduce crown fire behavior to improve firefighter safety? (2) Did the treatment protect people and their property? and (3) Did the treatment mitigate severe fire effects to valued vegetation and soil resources? Resources include forests, soils, fish and wildlife habitat, water quality, and recreation. This report focuses on the third criterion and documents not just our assessment of immediate fire effects (also termed "fire severity" measures by Lentile and others [2006]), such as ash deposition and soil alteration, but also extended fire effects (also termed "burn severity" measures by Lentile and others [2006]) assessed one year post-fire, such as delayed tree mortality and vegetation response.

This paper consists of three main sections. First, we review literature from forests and rangelands, focusing on case studies where there was a quantitative assessment of fuel treatments that were implemented and tested in fires. We summarize findings from multiple studies, including assessments of fuel treatment effectiveness in 2007 wildfires. Second, we summarize field and remotely sensed data that we collected and analyzed ourselves from two WUI areas in central Idaho where fuel treatments were burned through by massive wildfires. Our intent in the case study was to not just qualitatively assess whether or not the fuel treatments effectively mitigated severe fire but to quantify a number of specific post-fire vegetation and soil effects and statistically compare them between treated and untreated sites. Third, we discuss the management implications of our findings, integrate what we found in the case study with what we learned from the literature review, and articulate the knowledge gaps that should be emphasized in future assessments of fuel treatment effectiveness.

Literature Review

Forest Studies

Despite the broad scientific consensus that removing or reducing fuels can alter fire behavior (Graham and others 1999, Martinson and Omi 2008, Pollett and Omi 2002), there are few studies that quantitatively assess this assertion in treated areas subjected to wildfires. Evidence for effective fuel treatments comes predominantly from model simulations of fire spread and crown fire potential. Martinson and Omi (2008) found that 26 of the 49 studies they reviewed relied on simulations and 13 of the 26 employed hypothetical treatments in addition to hypothetical fires. Thus, these modeling study results might be best viewed as hypotheses that await empirical testing. Tests have taken the form of natural experiments, where wildfires have serendipitously encountered fuel treatments. There have been 22 published analyses of such; 11 of them included a statistical analysis of the treatment effect, and 7 attempted to control for the influences of topography and weather. According to Martinson and Omi (2008), there are only four published studies that include both a statistical test and adequate control to discern a fuel treatment effect in an actual wildfire.

Advances made in quantifying differences in treatments and resulting fire effects are evident when field-based case studies are reviewed (see Forest Studies sidebar) in chronological order. We only include case studies where vegetation treatments were tested in wildfires. Although some of the vegetation treatments evaluated were not specifically designed as fuel treatments, they are mentioned because they had the same effect.

Forest Studies Literature

Cumming (1964) compared the effectiveness of prescribed burning treatments to untreated, adjacent stands in the New Jersey Pine Barrens. Over the 10-year period prior to the wildfire, State lands within the wildfire perimeter had one to five prescribed fire treatments while private lands had no management action, resulting in different (although unquantified) stand structures and fuel compositions. Fire damage consisted of the following classifications: killed (stem killed or no sprouts at root collar), severe (crown reduced by two-thirds or more), moderate (one-third to two-thirds crown reduction), light (one-third or less), and unburned or no damage. Wildfire-induced tree canopy mortality was less in areas with prior prescribed burning, particularly in those units that had been treated within three years of the wildfire. Oak mortality was two to four times higher in untreated versus treated stands, with 97 percent either killed or severely damaged in untreated stands but only 46 percent killed or severely damaged in treated stands. Compared to oaks, pines were more fire tolerant, with 79 percent killed or severely damaged in untreated stands compared to only 17 percent killed or severely damaged in stands treated with prescribed fire.

Van Wagner (1968) evaluated tree mortality in southern California in plantations subjected to wildfire that had undergone fuel treatments with the explicit intent of minimizing the severity of subsequent wildfires. He evaluated two plantations that were burned in 1959 and 1960 to determine treatment effectiveness for minimizing fire-induced tree mortality. The intent of this study was not to compare treated and untreated areas but to find the treatment method or fuel condition that effectively mitigated fire severity. Fuel treatment methods included brush removal within and around plantation perimeters, construction and maintenance of firebreaks, pruning of lower limbs, and scattering of slash from pruning. At the first site, pruning had been done three years prior to the fire, resulting in a canopy base height of 2.1 to 5.5 m and tree spacing of 3.0 to 3.7 m. Slash from thinning and pruning operations was scattered.

The second site underwent a similar treatment, yet canopy base height was only increased 1.8 to 2.4 m. Slash was scattered and a 12 m wide fuel break was created around the plantation. Under the extreme burning conditions on both sites, there was no significant relationship between tree mortality and diameter at breast height (dbh) or pre-fire canopy base height. Given that both sites were dominated by Coulter pine (*Pinus coulteri*), species composition was not regarded as an influential factor on fire severity. Plots with the largest average tree diameter also had the highest survival percentage (92 percent). Across both sites, as shrub density increased, so did tree mortality. Plots with steep slopes and low brush densities still showed high percentages of tree mortality. Although canopy base height appeared to be significantly improved by pruning, severity was ultimately a function of surface fuels and/or slope acting as the mechanisms to carry fire into aerial fuels. As slope and surface fuels increase, so does flame length, representing a common cause and effect relationship of intensity and severity found in other studies within this review.

Wagle and Eakle (1979) evaluated how a prescribed burn one year prior to a wildfire affected tree mortality and fuels in east central Arizona. Fuels were quantified as the depth of loosely arranged fine and coarse surface litter, compacted fine and partially decomposed litter, and total litter depth. Their study site was dominated by ponderosa pine (*Pinus ponderosa*) with adjoining untreated and treated units on the same slope. Only one live tree was found in the untreated stand, while 83 percent of the trees in the treated areas survived the wildfire. Most of the dead trees were less than 20 cm dbh. One year after the wildfire, understory vegetation was twice as abundant in the treated area (9 percent) than in the untreated area (5 percent).

Omi and Kalabokidis (1991) assessed the effects of a 1988 wildfire in lodgepole pine (*Pinus contorta*) and subalpine fir (*Abies lasiocarpa*) forests on the Targhee NF in Idaho. They compared fire severity in extensively managed sites versus intensively managed sites. Extensively managed areas were defined as having a mature overstory that was typically dominated by lodgepole pine, although species composition varied somewhat with elevation and included other tree species common to the Greater Yellowstone Ecosystem. Mountain pine beetle (*Dendroctonus ponderosae*) infestation had resulted in approximately 70 percent mortality of lodgepole pine throughout the extensively managed areas prior to the 1988 fire season. Intensively managed stands were naturally regenerated clearcuts composed of mostly lodgepole pine that had undergone harvest and slash disposal prior to 1988. Within intensively managed stands, tree densities increased with age, ranging from 15,000 trees/ha at a height of 0.9 to 5.0 m to 2000 trees/ha at a height of 0.3 to 0.9 m. Burn severity within extensively managed areas and intensively managed sites was determined using ocular estimates based on the following criteria:

- unburned: fire did not enter stand
- light: surface burn without crown scorch
- spotty: irregular crown scorch
- moderate: intense burn with crown scorch
- severe: high intensity burn with crowns totally consumed

Fuel loadings were generally higher in extensively managed, mature stands compared with intensively managed stands across large (greater than 7.6 cm diameter) wood, small (less than 7.6 cm diameter) wood, and litter categories. In 89 percent of plots in the extensively managed areas, wildfire damage was moderate or severe compared to only 20 percent of plots in the intensively managed areas. Of the 45 regeneration sites, 38 had ratings of unburned, light, or spotty while adjacent, mature stands had ratings of moderate and severe. The authors attributed differences in fire damage to differences in fuel loading and fuel depth. Mature stands were more combustible due to the presence of beetle-killed lodgepole pine compared with significantly lower amounts of activity fuels in the intensively managed sites. Although this study was restricted to variations in loading among size classes, additional explanatory variables such as differences in stand continuity, crown diameter, aspect, slope, and elevation between the two management schemes/stand types/topographic conditions were suggested as influencing factors.

Weatherspoon and Skinner (1995) evaluated fire severity in northern California in plantations or partially cut stands with fuel treatments, partially cut stands without treatment, and uncut stands without treatment. Using aerial photography that was calibrated by forest records or interviews, the authors quantified fire damage to trees, including damage from crown scorch. The authors defined five classes of burn severity called fire damage classifications (FDC):

- 0 (did not burn)
- 1 (light underburn, less than 10 percent of trees with greater than 50 percent crown scorch)
- 2 (moderate damage, 10 to 50 percent of trees with greater than 50 percent crown scorch)
- 3 (greater than 50 percent of trees with greater than 50 percent crown scorch but less than 50 percent with crowns consumed)
- 4 (extreme damage, greater than 50 percent of trees with crowns consumed)

The following variables were also evaluated: presence of grasses and forbs, elevation, aspect, site preparation method, damage in adjacent stands related to plantations, uniformity of damage, and dominant tree species. In partially cut, treated stands, the slash had been broadcast burned, machine piled (all piles were burned prior to wildfire), or lopped and scattered. Areas that had been treated with broadcast burns (low intensity surface fires) suffered the least damage in the subsequent wildfire (half were 0 or 1) compared with mechanically piled or lopped and scattered areas which had slightly higher fire damage ratings. By comparison, partially cut, untreated stands had an FDC of 3 or 4. Grass was more prevalent in pile and burn treatments than in broadcast treatments, while forbs were more prevalent in broadcast burn areas. Fire damage in treated plantations was also affected by fire damage in adjacent, untreated stands. As fire intensity and the resulting burn severity increased in untreated stands, it increased in the treated plantations. Treated plantations were relatively small (approximately 4 ha), and the authors stated that this likely affected the treatment's effectiveness, particularly during extreme burning conditions. Additionally, partially cut stands or plantations were under a selection system where the large, fire-resistant trees were removed. The broadcast burned units showed a trend of damage that decreased from the edge inward, compared to the spotty burn pattern found in machine-piled treatments and uniform fire damage found in untreated stands. Grasses played a significant role in the FDC, as reduced fine fuel loading and continuity affected the spatial patterns of severity. Fire damage across management schemes increased in the following order: uncut and untreated, partially cut with treatment, and partially cut without treatment. Uncut and untreated stands were mature stands with less activity fuels than plantations, and the closed canopy resulted in a micro-climate that moderated burning conditions. Stands that were partially cut and untreated likely had continuous, relatively high loadings of activity fuels and fine fuels, resulting in substantial crown scorch.

Choromanska and DeLuca (2001) assessed the effects of prescribed fire and wildfire on microbial biomass recovery and carbon and nitrogen transformations. In August of 1996, a high intensity fire partially burned into an area treated by a prescribed fire that was applied in May of 1996 on the Bitterroot NF in Montana. Both the prescribed fire unit and the adjacent, untreated stand consisted of a ponderosa pine overstory with thickets of Douglas-fir (*Pseudotsuga menziesii*) regeneration. Prior to the prescribed fire and wildfire, this area had not seen fire since 1916. Topography and weather conditions were very similar between the two treatments at the time of the wildfire. The intent of this study was to characterize and evaluate fuel treatment effectiveness and to compare nutrient cycling between a prescribed fire (PB), wildfire (WF), and the portion of the area treated by prescribed fire that burned in a subsequent wildfire (PBWF), using an untreated, unburned stand as a control. The prescribed fire was conducted with the intent of reducing fuel and eliminating the Douglas-fir thickets. Fuel estimates were 1.5 Mg/ha of fine material (0 to 8 cm diameter) and 0.2 Mg/ha of fuels greater than 8 cm diameter. The prescribed burn resulted in 42 percent consumption of fine fuels and no tree canopy mortality. After the wildfire, the WF area showed 100 percent tree mortality and 100 percent fine fuel consumption compared to the PBWF where there was 50 percent tree mortality and 70 percent fine fuel consumption. Fuel consumption in the soil organic horizons was 100 percent in the WF, 65 percent in the PBWF, and 42 percent in the PB. The PBWF area had significantly lower rates of net nitrogen mineralization and higher potentially mineralizable nitrogen in comparison to the WF area. Microbial recovery in the PBWF was faster than in the WF soils. Additionally, the PBWF area had double the carbon biomass and basal respiration of the WF area two years post-fire. Results show that prescribed fire treatment reduced wildfire burn severity in surface and aerial fuels, minimized losses of labile carbon and nitrogen, and improved soil microbial resilience to subsequent wildfire.

Pollett and Omi (2002) sought to empirically evaluate fuel treatments tested by recent wildfires. They selected sites with adjacent treated and untreated stands with accurate records and less than 15 years between treatment completion and subsequent wildfire. The four sites that met their selection criteria were the 1994 Webb fire in Montana that underwent a broadcast burn treatment in 1989; the 1994 Tyee fire in Washington that was treated with pre-commercial thinning in the 1970s and a subsequent underburn treatment in 1983; the 1994 Cottonwood fire in California that was treated with whole tree thinning in 1989 and 1990; and the 1996 Hochderffer fire in Arizona that underwent a thinning treatment in the 1970s and broadcast burn in 1995. The authors measured aspect, slope (%), tree basal area (m^2/ha), density (stems/ha), average diameter (cm), fire severity rating, crown scorch (%), and crown weight (kg), and they used severity ratings derived from Omi and Kalabokidis (1991). Sites that received mechanical fuel treatment reduced wildfire severity rating and percent crown scorch more than broadcast burning. Severity rating and crown scorch did drop significantly across all treatments, but the Webb fire (broadcast burn only) showed the least difference between treated and untreated stands. Mechanical treatments yielded more homogenous residual stand conditions than the broadcast burn treatment, resulting in lower density, larger trees, higher crown base heights, and reduced continuity in aerial fuels throughout the treatment area. The authors did not support the notion that more open stands would change the micro-climate and result in higher fire severity, as suggested by Weatherspoon and Skinner (1995). Pollett and Omi (2002) reasoned that treatments that reduced density and increased average tree diameter outweighed any increase in micro-climate effects, suggesting the degree of forest openness was not enough to sufficiently increase fire behavior and, thus, post-fire effects. However, they did state that under extreme burning conditions, fuel treatments may have minimal effect in mitigating fire severity. Nonetheless, all treatment types studied across sites significantly mitigated canopy scorch, with thinning treatments alone or in combination with prescribed fire reducing severity more than prescribed fire alone.

Martinson and others (2003) examined all natural and anthropogenic fuel alterations within the perimeter of the 2002 Hayman fire in Colorado, including past wildfires, prescribed fires, commercial tree harvests, pre-commercial stand improvements, plantations, and surface fuel treatments. Many of the treatments were not implemented with the intent of mitigating wildfire burn severity. The authors utilized a burned area severity map and verified results with field sampling, classifying severity as low, moderate, or high. The authors concluded that the extreme weather conditions and other abiotic factors that influenced burning conditions rendered most pre-wildfire stand treatments useless; although there were examples in which prescribed fires, old burns, thinning treatments, and timber harvests mitigated wildfire burn severity by changing fire behavior from crown to surface fire. In areas that were burned under moderate conditions, recent prescribed burns had lower wildfire severity than older prescribed burns, with a noticeable difference between units that had been burned multiple times. However, uncertainties in pre-fire forest structure made identifying specific attributes to predict treatment success impossible.

Skinner and others (2004) utilized the Cone fire of 2002 to evaluate multiple treatment types within the Black Mountain Experimental Forest in northern California. Forest structure within treatments was grouped into two distinct ecological groups: late-seral stage and mid-seral stage. In late-seral treatments, only ladder fuels were removed. In mid-seral stands, large, mature trees greater than 41 cm dbh and smaller trees (ladder fuels) were removed, creating stands composed of intermediate age classes. Six of each of these units, ranging from 77 to 142 ha, were created within the Experimental Forest. Each of the six units was split in half; one received a prescribed fire treatment, and the other received no additional management activity with the exception of grazing prior to the wildfire. Of these experimental units, two mid-seral stands and one late-seral stand were burned in the 2002 wildfire. Pre-treatment surface fuels varied greatly across sites; 0- to 7.6-cm material ranged from 3.6 to 20.8 Mg/ha (mean of 9.4 Mg/ha), and material greater than 7.6 cm ranged from 1.1 to 86.8 Mg/ha (mean of 24.2 Mg/ha). The harvest system in the mid-seral treatments increased surface loading by an average of 7.8 Mg/ha, particularly in the 0 to 7.6 cm size class, although this was also quite variable. Late-seral harvesting resulted in minimal fuel increases. The prescribed fire treatments essentially consumed all size classes less than 7.6 cm, except unburned islands, and significantly reduced larger surface fuels. Wildfire-induced tree mortality was highest in untreated stands (greater than 90 percent), and thinning only treatments that consisted of lop and scatter or that did not address surface fuels showed the second highest levels of mortality (40 to 60 percent). Similar to the results in Weatherspoon and Skinner (1995) and others, thinning treatments followed by prescribed fire two to four years prior to the wildfire showed the least fire-induced tree mortality compared with thin-only treatments. In addition, surface fuel loading and continuity again appeared to be key influences on post-fire tree mortality in ponderosa pine stands.

Based on the large area burned by the Rodeo-Chediski fires in Arizona, Finney and others (2005) evaluated the effectiveness of prescribed burning on reducing burn severity in treatments that were completed between 1993 and 2001. They used the differenced Normalized Burn Ratio (dNBR) that was interpreted from Landsat satellite imagery and the burn severity classes of Key and Benson (2005) to compare areas treated with prescribed fire versus untreated areas. Localized effects such as differences in fuel composition and consumption of different parts of the fuelbed were not addressed. Overall, burn severity was reduced where prescribed fires had occurred within the nine years prior to the wildfire. Burn severity was more consistently and significantly reduced in areas that were burned within four years before the wildfires. Burn severity was less where treatments were larger and in areas that had been burned multiple times. Because prescribed fires can yield patchy results, Finney and others (2005) suggested that increased treatment size may promote heterogeneity within the prescribed burn perimeter, thereby reducing overall burn severity across the treatment.

Raymond and Peterson (2005) measured fire damage across different treatment types when the Biscuit fire of 2002 burned through two separate study sites in the mixed conifer forest of the Oregon Coast Range. The first site was divided into three treatment plots (6 to 8 ha) composed of stands that were thinned, thinned then broadcast burned, or untreated. The second site had three plots that were thinned, thinned with coarse woody debris left, or untreated. The treatments were a combination of crown thinning and thinning from below. Douglas-fir, the dominant species on the site, was thinned to a relative density of 0.25 (proportion of standard volume for a given stand age and site quality), and evergreen broadleaf species were thinned to a spacing of 8 m. Fuel characteristics were quantified both pre- and post- treatment after the Biscuit fire. The authors measured 1-hr, 10-hr, and 100-hr fuel size classes, live woody and herbaceous fuels, fuel depth, and computed summary statistics. Additionally, aerial fuels were calculated from measured tree dbh, species, crown class, total tree height, and crown base height. Prior to the wildfire, thinning treatments across all sites decreased tree density, basal area, and crown bulk density and increased both crown base height and mean dbh. However, the effects on surface fuels differed with respect to treatment. There was more surface fuel in thinned-only stands, while the thin and burn treatment showed a net decrease in surface fuels. Fire damage was measured by crown scorch volume, height of crown scorch, and tree mortality. In thinned stands, crown base height and mean dbh were higher. Both crown scorch volume and crown scorch height increased in the following order: thin and burn, untreated, and thin only. Douglas-fir tree mortality two years post-fire was 80 to 100 percent in thinned stands, 53 to 54 percent in untreated stands, and 5 percent in thinned and burned stands. Untreated stands also had the greatest variance in fire damage. The increased surface fuel loading in thinned only treatments outweighed any benefit of increased crown base height in reducing burn severity. Patterns of tree damage varied among tree species; yet without treating surface fuels, the overstory trees all had relatively high scorch heights and mortality. The fire burned *around* the thinned and burned treatments where surface fuels had been greatly reduced but burned *through* thinned treatments. Raymond and Peterson (2005) also concluded that crown scorch volume, or the percent of crown volume scorched, is a better indicator of mortality than scorch height.

Cram and others (2006) evaluated fuel treatment effectiveness across multiple vegetation types after Arizona and New Mexico wildfires in 2002 and 2003. They sampled sites with fuel treatments and adjacent untreated areas with similar vegetation, wildfire conditions, slope, and aspect. All treated and untreated areas sampled were greater than 16 ha in size with no post-fire salvage cutting. Study sites were in the Oso and Borrego fires, which burned through mid- to high-elevation stands dominated by ponderosa pine with intermixed Douglas-fir and white fir (*Abies concolor*) that were treated by commercial harvest followed by prescribed fire. Two additional study sites were within the perimeter of the Rodeo-Chediski fire in Arizona, which burned at relatively low elevations in ponderosa pine and gambel oak (*Quercus gambelli*). The two sites consisted of a non-commercial lop, pile, and burn treatment and non-commercial lop and scatter treatment. Measurements that characterized the respective stands were basal area, density, dbh, tree height, crown length, height to pre-fire live crown, and canopy bulk density. Burn severity measurements included bole char height, crown scorch height, crown consumption height, percent crown scorch, and percent crown consumption. Burn severity was also quantified using the previously mentioned ocular estimates for crown damage developed by Omi and Kalabokidis (1991) in addition to the ocular estimates for surface damage modified after Ryan and Noste (1985). Results yielded a canopy fuel consumption threshold that consisted of canopy bulk density of 0.047 kg/m^3. Stands that underwent surface fuel treatments with canopy bulk density below this threshold showed no evidence of canopy fuel consumption. This threshold also applied to lop and scatter plots, although increased surface fuel load resulted in significant canopy scorch. The third lop and scatter plot that was above the crown bulk density threshold (0.084 kg/m^3) showed evidence of torching characterized by canopy fuel consumption. All untreated stands showed evidence of torching or crown fire and were above this threshold, supporting the conclusion that canopy bulk density is a limiting factor in torching and crown fire initiation (Rothermel 1991). Cram and others (2006) concluded that as tree density and basal area decreased and mean tree diameter increased, fire effects decreased. They stated that canopy bulk density was perhaps the best quantitative indicator of potential crown damage, suggesting that aggressive treatments are needed to significantly reduce canopy fuels to achieve this loading. Additionally, mechanical treatments followed by prescribed fire provided the best manipulation of surface and aerial fuels and resulted in the most significant mitigation of fire effects. Accordingly, treatments where slash was scattered still left stands susceptible to high severity fire effects, even in recent treatments four years old or less.

Omi and others (2006) evaluated fuel treatments within the boundaries of five large, recent wildfires: the Hayman fire (2002) in Colorado, the Aspen fire (2003) in Arizona, the Davis fire (2003) in Oregon, the Power fire (2004) in California, and the Fischer fire (2004) in Washington. Across these study sites, a full range of treatments were tested, including thinning from above and from below both with and without slash removal and in conjunction with prescribed fire or pile burning. These sites spanned a large range of environmental and topographic conditions. The majority of the study sites were in dry, mixed conifer forests dominated by ponderosa pine, Douglas-fir, or Jeffrey pine (*Pinus jeffreyi*). Canopy severity ratings were derived from Omi and Kalabokidis (1991), and fire effects on surface and ground fuels were evaluated based upon Ryan and Noste (1985). The authors measured canopy cover of understory plants and characterized soils to determine any possible relationship between moisture and nutrient (carbon and nitrogen) availability and non-native plant abundance. Results showed that treated areas had lower tree density (484 versus 1110 trees/ha), lower crown bulk density (0.07 kg/m^3 versus 0.10 kg/m^3), higher crown base height (8.0 m versus 4.1 m), and increased mean tree diameter (38.4 cm versus 28.6 cm) compared with adjacent untreated stands. With the exception of height to canopy (or canopy base height), differences in treated versus untreated stands were greatest in treatments that were intended to reduce canopy fuels. Across all study sites, treatments 10 years old or less were generally effective if they reduced surface fuels. However, the most effective treatments not only reduced litter and other surface fuels but also served as low thinning treatments by reducing canopy bulk density and increasing both height to canopy and mean tree diameter. The treatment type that most effectively achieved this result was mechanical thinning followed by some type of slash removal and surface fuel treatment within two years of the thinning. Treatments at the Hayman and Davis fires that involved both thinning and slash removal produced the most dramatic results of all study areas, with 80 percent less canopy scorch in treated areas compared to adjacent, untreated stands.

Moghaddas and Craggs (2007) used the 2005 Bell fire to evaluate a private land fuel treatment adjacent to an untreated stand on the Plumas NF in northern California. Species composition was dominated by Douglas-fir, with the remainder of the species composed of incense cedar (*Calocedrus decurrens*), ponderosa pine, sugar pine (*Pinus lambertiana*), white fir, and California black oak (*Quercus kelloggii*). The area was subjected to a selection harvest. Pre-treatment stand basal area was 59 m^2/ha with a stand density of 1181 trees/ha. Post-treatment tree density averaged 181.1 trees/ha, 9.2 m canopy base height, and 23.7 m^2/ha basal area. Surface fuels were characterized with an average depth of 3.6 cm and 11.9 Mg/ha for 1-hr, 10-hr, and 100-hr fuels combined. The treatment resulted in reduced vertical and horizontal continuity of aerial fuels. Compaction of surface fuels from the harvesting operation changed the surface area to volume ratio, while chipping unmerchantable tops at the landing mitigated increasing post-treatment surface fuel loading. Although the focus of the study was fire behavior and suppression efficiency, the authors measured severity in terms of percent crown volume scorched. Crown scorch was 75 percent at the southern edge of the treatment and decreased to less than 10 percent within 60 m as the fire moved to the interior of the treatment. The increased vertical and horizontal spacing of canopy fuels in the treated area greatly reduced passive crown fire and mitigated burn severity in the tree canopy. However, the treatment was situated on a ridgetop while the rest of the fire perimeter was on a relatively steep slope with a southerly aspect. Slope and aspect likely played significant roles in minimizing overall crown scorch and contributed to the spatial changes in severity across the treatment.

Strom and Fulé (2007) sampled treatments burned during the 2002 Rodeo-Chediski fire in Arizona that had been implemented with the explicit intent of mitigating fire effects. Fourteen sites were selected; all of which had adjacent treated and untreated stands with similar topography and no barriers to impede fire spread between them. Of the 14 sites, 12 were non-commercial thinning treatments where slash was piled and burned, and slash was either scattered or crushed in the remaining 2. All treatments were completed between 1990 and 1999. Post-fire tree measurements included tree species, condition, dbh, total height, canopy base height, bole char height (minimum and maximum), and a dwarf mistletoe rating (0 to 6) derived from Hawksworth (1977). Post-wildfire tree condition was also classified as either live, declining, or one of four stages of snags: recent, loose bark, clean, or broken above breast height (from Thomas and others 1979). Tree mortality was compared between treated and untreated stands. A subsample of increment cores was also collected to determine age and growth data of residual trees, while regenerating trees and shrubs were quantified by species, condition, and height class. Untreated stands had higher post-fire proportions of small trees, causing higher bole char, fire-induced increases in crown base height, and increased overall tree mortality, all of which contributed to a shift in the residual tree distribution toward larger trees. Treated areas had significantly higher live tree density and higher crown base height, with 50 percent mortality in treated stands compared to 95 percent in untreated stands. Due to similarities in pre-fire basal area and treatment-caused differences in fire effects, the authors concluded that the arrangement of canopy fuels—not the overall amount of fuels in the form of trees—was the determinant of burn severity. That is, fire effects differed greatly between stands with many small trees and those with a few larger trees. Initial measures of recovery were similar between treated and untreated areas, with the exception of the abundance of surface fuel and manzanita (*Arctostaphylos* spp.), both of which were more abundant in untreated areas. Ponderosa pine regeneration was patchy in both treated and untreated areas. Areas that were untreated prior to the wildfire would likely become shrubfields dominated by manzanita, gambel oak, and New Mexico locust (*Robinia neomexicana*). In contrast, areas that were treated prior to the wildfire would likely become dominated by ponderosa pine with larger trees, increased basal area, lower tree density, and a gambel oak understory in the coming decades. The authors concluded that fuel treatments are not only effective in mitigating fire severity but also contribute greatly to retaining the ecological functionality of southwestern ponderosa pine forests.

Martinson and Omi (2008) assessed the mitigating effect of fuel treatments on wildfire severity. The study site was on the Mississippi Sandhill Crane National Refuge, where slash pine is dominant and longleaf pine is present on flat topography. A prescribed fire in the prescribed fire-maintained Fontainebleau Unit on 18 April 1999 escaped by spotting across a railroad and became a wildfire that exhibited extreme fire behavior in untreated fuels on adjacent private land. The team opportunistically collected post-fire data in September 1999 to quantify fuel and fire severity differences between treated and untreated stands. The authors did not sample ground fuels because these were largely consumed by the fire. They used nine variable radius plots in each condition, with plot centers separated by 60 m and surrounded by a 60-m buffer to minimize edge effects. They ocularly assessed height to post-fire live crown and maximum scorch height. They found that trees in treated plots were 50 percent taller than trees in untreated plots and had twice the girth. Crown base heights were nearly twice as high in treated plots, while shrubs were twice as tall in untreated plots. Shrub density did not differ significantly. Maximum needle scorch height was nearly twice as high in untreated plots. Martinson and Omi (2008) concluded by noting the rarity of empirical, retrospective studies of fuel treatment effectiveness, and they reiterated the need to continue collecting empirical data from other natural experiments where wildfire burns through fuel treatments.

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Rangeland Studies

Prior to European settlement, fires in sagebrush communities typically burned in a patchy fashion, leaving unburned islands (Miller and Eddleman 2001). Sagebrush and grass were probably dominant with a strong perennial grass and forb component in the understory. Excessive livestock grazing by Euro-American settlers in the late 1800s and early 1900s caused major changes in plant communities within a few decades. By the early 1900s, an estimated 26 million cattle and 20 million sheep grazed in western rangelands. The grazing capacity of western lands had decreased by an estimated 60 to 90 percent by the 1930s (Miller and Eddleman 2001). From 1880 to 1912, when the number of cattle, sheep, and horses was particularly high, only 44 fires were reported in Great Basin rangelands, burning only 4500 ha (Miller and Narayanan 2008).

The Bureau of Land Management and the Forest Service reported that the annual area burned in wildfires is on a long-term upward trend (Davison 1996, Westerling 2006). The number of rangeland fires larger than 2000 ha has increased from 15 percent of all fires from 1960 to 1982 to 60 percent of all fires from 1983 to 2003 across the sagebrush (*Artemisia* spp.) steppe in the Snake River Plains and Northern Basin and Range ecoregions in the Great Basin (Kuchy 2008). In 2007, the Murphy complex burned over 240,000 ha of mainly sagebrush steppe in southern Idaho and northern Nevada; the Milford Flat fire burned over 120,000 ha of Utah rangelands in the same year. The predicted change in climate combined with the large fires that burned during the 2007 fire season and the need for reducing the likelihood of extensive fires in western rangelands have focused considerable attention on the effects of historical and current grazing regimes on fire fuels, fire effects, behavior, and post-fire soil and vegetation recovery. Because there are no explicit examples in the literature of wildfires that have burned across rangelands where fuel treatments have been conducted, this review (see Rangeland Studies sidebar) examines the potential effects of livestock grazing (removal of herbaceous fuels) on fuels, potential fire behavior, and fire effects. The geographic focus is on the sagebrush steppe ecosystems of the Great Basin, but examples are also included from the forest-rangeland interface. The case studies (arranged in chronological order) have focused on the degree to which grazing could alter fire behavior.

Rangeland Studies Literature

Rummel (1951) examined long-term effects in vegetation differences between grazed and ungrazed ponderosa pine-dominated plateaus in central Washington. Vegetation characteristics were compared on two plateaus that were similar in geologic origin, elevation, climate, and timber type. Both plateaus were unaffected by fire, and grazing was identified as the only broad-scale disturbance. Meeks Table had never been grazed by livestock while Devil's Table had been heavily utilized by livestock during the 40 years that led up to this study. Study plots were grouped into three classifications: (1) pinegrass-elk sedge (*Calamagrostis rubescens-Carex geyeri*) understory, open ponderosa pine overstory; (2) pinegrass-elk sedge understory, mixed ponderosa pine-Douglas-fir overstory; and (3) subalpine needlegrass (*Achnatherum nelsonii*)–Sandberg bluegrass (*Poa secunda*) open grassland type. Herbaceous and shrubby understory vegetation density and composition were determined for both mesas. Herbaceous weight of pinegrass was also determined. All trees less than 10.2 cm dbh were tallied by species and height class, while all trees greater than 10.2 cm dbh were recorded by species and dbh class. Grassland openings on the grazed mesa had been invaded by ponderosa pine while openings on the ungrazed mesa remained free of seedlings. Pinegrass cover was high in the grassland openings on the ungrazed mesa. Herbaceous and shrubby understory cover beneath open ponderosa pine stands averaged 35 percent on ungrazed mesas and 14 percent on grazed mesas. The pinegrass biomass (grass fuels) was 953 kg/ha on the ungrazed mesa and 269 kg/ha on the grazed mesa. On the ungrazed mesa, there were only 210 trees/ha that were less than 10.2 cm dbh; on the grazed mesa, there were 8132 trees/ha that were less than 10.2 cm dbh. The authors concluded that the vegetative ground cover and litter prevented the establishment of tree seedlings on the ungrazed mesa. In this ecosystem, the removal of understory vegetation by livestock facilitated tree propagation and growth. Livestock may have reduced fire potential by reducing grass and shrub biomass.

Madany and West (1983) examined the relative importance and interaction of fire cessation and livestock grazing on grazed and ungrazed mesas in southern Utah. Ungrazed study sites had been isolated from both fire and grazing. The study sites were located in Zion National Park in Utah. The area was dominated by ponderosa pine and gambel oak (*Quercus gambelii*), however, Rocky Mountain juniper (*Juniperus scopulorum*), pinyon pine (*Pinus edulis*), and bigtooth maple (*Acer grandidentatum*) were also present in the area. In addition to gathering historical land use information for this area, the authors collected sapling, shrub, forb, and graminoid cover data. A fire history was constructed from cross sections taken from 111 fire-scarred trees. A reduction in herbaceous cover and increases in woody species density were attributed to livestock grazing. While changes in vegetation composition and structure resulted in decreased fire frequency on the grazed mesa, ungrazed mesas retained savanna-like conditions despite the absence of frequent fires. The authors suggested that heavy grazing may lead to an increase in gambel oak stem density because oak more readily establishes when the grassy interspaces are disturbed. The dense sod associated with perennial grasses is likely the main controlling factor of ponderosa pine regeneration, however allelopathic interactions between grasses and pine seedlings, and competition for limited soil moisture, may also play important roles in pine regeneration. In summary, livestock grazing may help to accelerate functional and structural changes in ponderosa pine forests. As grazing depletes the herbaceous layer, it reduces fire frequency while simultaneously enabling pine seedling regeneration.

Zimmerman and Neuenschwander (1984) focused on the influences of livestock grazing on plant community structure, fire frequency, and fire intensity in the Douglas-fir/ninebark (*Pseudotsuga menziesii/Physocarpus malvaceous)* habitat type of northern Idaho. The area was grazed by cattle, sheep, and native ungulates since the 1920s, heavily from 1945 to 1967. Vegetation was sampled in 18 15- by 25-m plots in grazed and ungrazed areas (exclosures), including number of trees, tree basal area, shrub density, herbaceous plant cover and frequency, herbaceous biomass, and accumulation of downed woody fuels. Trees in size classes less than 40 cm dbh were significantly more numerous in the grazed plots compared to ungrazed plots, particularly in the 5- to 20-cm dbh classes. Douglas-fir seedlings were more abundant than ponderosa pine seedlings in grazed plots, indicating that long-term heavy grazing favors Douglas-fir rather than ponderosa pine. No significant effects were reported in the larger size classes of trees. Tree basal area was significantly higher in grazed compared to ungrazed stands, which was attributed to the larger number of young trees in the grazed stands. Total shrub density did not differ between grazed and ungrazed stands. Service berry (*Amelanchier alnifolia*), ninebark (*Physocarpus* spp.), and white spiraea (*Spiraea betulifolia*) had a greater density in grazed stands, while the remaining nine shrub species showed a higher density in ungrazed stands; however, only the density for redstem ceanothus *(Ceanothus sanguineus),* chokecherry *(Prunus virginiana),* and yerba buena *(Sarureja douglasii)* were significantly different. Total shrub cover was significantly lower in grazed stands—15.8 percent in grazed stands compared to 24.5 percent in ungrazed stands. Herbaceous biomass of bluebunch wheatgrass (*Pseudoroegneria spicata*), Idaho fescue (*Festuca idhoensis*), and pinegrass was significantly lower in grazed plots, while biomass of Columbia brome (*Bromus vulgaris*) and Kentucky bluegrass (*Poa pratensis*) was higher in grazed stands. Total forb cover was not significantly different between grazed and ungrazed stands. Downed woody fuel loadings of all size classes, including duff, was higher in grazed stands. Live herbaceous fuels were significantly lower (467 kg/ha) in grazed compared to ungrazed stands (719 kg/ha). The authors concluded that livestock grazing has the ability to alter vegetation structure and composition and also the composition of all fuel classes in the Douglas-fir/ninebark habitat type. The modifications result in a forest that is less likely to burn in frequent surface fires; that is conducive to vertical fire spread; and, in the absence of fuel treatments, promotes the occurrence of infrequent, high intensity fires.

Sapsis and Kauffmann (1991) measured fuel load, fuel moisture, fire weather, fire behavior, and consumed biomass during prescribed burning in sagebrush steppe in Oregon. Two fire treatments were conducted and compared: a fall burn (four plots) and a spring burn (five plots). The overstory was dominated by basin big sagebrush (*Artemisia tridentata* subsp. *tridentata*), and the understory was dominated by Idaho fescue and bluebunch wheatgrass. Each burn unit was at least 30 to 50 m in size and the burn treatments were randomly assigned; 1-hr, 10-hr, 100-hr, herbaceous, and downed woody fuels were measured prior to the treatments. Sagebrush cover was estimated by the line-intercept method. Shrub volume was measured and sagebrush biomass was calculated, as was the biomass of the herbaceous material. Post-fire fuels were estimated in a similar manner. The fall burn resulted in a longer flame length (4.1 compared to 1.7 m in the spring), higher rate of spread (1.6 compared to 0.3 m/s in the spring), and higher fire line intensity (6400 compared to 880 kW/m). Sagebrush foliar moisture was 186 percent in the spring and 97 percent in the fall. The consumption of fine fuel was not significantly different between treatments. However, consumption of 10-hr and 100-hr fuels was significantly higher in the fall burn. Biomass consumption was 93 percent in the fall burns compared to 84 percent in the spring burns. The largest difference in fuel consumption was found in the 10-hr fuels (85 percent for the fall burn and 52 percent for the spring burn) possibly due to the higher foliar moisture in the spring.

Link and others (2006) explored fire-plant-grazing interactions in a field in Saddle Mountain National Wildlife Refuge in Grant County, Washington. Relationships among fire ignition probability, cheatgrass (*Bromus tectorum)* cover, native perennial plants, and grazing are complex. The percent cover of vascular plant species, bare soil, soil cryptograms, litter, and cheatgrass was estimated in late August and early September 2002 in 176 plots. Fires were ignited on the upwind side of the plots and were considered a sustained fire if they grew to an area of 100 m^2. Fine-scale aerial photos taken in September 2002 were used to estimate cheatgrass cover in the plots. Fire ignition risk was 100 percent when the cover of cheatgrass was 45 percent or more, while the fire ignition risk dropped to 46 percent when cheatgrass cover was 12 percent or less. When ground cover of native perennials, litter, and soil approached 31 percent, the risk of ignition and sustainable fire was significantly reduced. Wind speeds greater than 5.7 km/hr increased fire risk, while fuel moistures greater than 7.5 percent decreased fire risk. Cheatgrass cover dramatically affected fire spread. The authors concluded that management strategies aimed at reducing cheatgrass cover and promoting perennial plants would reduce the risk of fire spread.

Nader and others (2007) presented techniques that have been explored to prevent the start and spread of wildfires in rangelands. The objective of fuel reduction is to change the fire behavior by changing the fuel bed depth, fuel loading, vegetative cover, and ladder fuels such that the flame length never reaches 1.2 m. Methods for fuels management were discussed, including mechanized treatment, herbicides, prescribed fire, hand cutting, and prescribed grazing. The authors compared the effectiveness and costs for the different methods in rangelands. Prescribed grazing was discussed in much detail, and the authors compared effects of grazing animal species, grazing intensity, season of grazing, animal condition, and desired outcome of the grazing treatment. The authors found that grazing is an appropriate tool when addressing small-diameter (1-hr and 10-hr) fuels. Grazing can impact these fuels by ingestion and trampling. Many factors affect the success of using grazing for fuels management, including species of livestock, the animal's previous grazing experience, time of year in relation to plant physiology, grazing intensity, grazing duration, plant secondary compounds, and animal physiological state. Grazing before seed set can change seed bank dynamics. Repeated grazing of perennial species can deplete root carbohydrates and cause mortality that can shift species composition. Each species of grazing animal has a unique utilization pattern. Cattle are effective in grass removal, while sheep and goats are effective on forbs and browse. Lactating and young animals are not recommended for fire fuel control because the animals may be required to eat below their nutritional needs. Care must be taken to select the appropriate combination of animal species, animal condition, season, duration, and intensity of grazing in order to reach desired fuel management objectives.

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2007 Wildfires

Wildfires were extensive in the summer and fall of 2007. Many burned through fuel treatments, providing the opportunity for assessment of fuel treatment effectiveness. Several reports already published by a number of teams are summarized (see 2007 Wildfire sidebar) both to increase their visibility and because they contribute to this synthesis of case studies. Compared to the previously reviewed journal articles, these reports communicate less about ecological fire effects and more about the influence of fuel treatments on fire behavior because they are targeted primarily at wildland firefighters who are more concerned with safety and are charged with protecting people and property.

2007 Wildfires Literature

Murphy and others (2007) assessed fuel treatment effectiveness at the 1243-ha Angora fire, which started from an unattended campfire southwest of South Lake Tahoe in California on the afternoon of 24 June 2007. The fire burned under some of the most severe fire danger conditions experienced in that area during the previous 20 years. The fire spread 6.4 km in three hours and burned over 250 structures on private property; most of the fire growth was under hot, dry, and windy conditions. U.S. Forest Service (USFS) lands constituted 89 percent of the area burned, about 50 percent of which was treated. Treatments covered 8 percent of the area on state and private land. About 300 urban lots and 93 ha of private property burned. The USFS Region 5 Fire Director sent a team to evaluate the effects of fuel treatments (mostly thinning with surface fuels piled and then burned) on fire behavior, fire suppression, structure ignition, and public safety/egress, as well as fire behavior in untreated areas and in other vegetation management treatment units (commercial thinning or salvage logging). Using on-the-ground and aerial reconnaissance; interviews with homeowners, firefighters, fire scientists, and fire behavior experts; and videos and photos taken prior to, during, and after the Angora fire, the team evaluated 16 fuel treatments (194 ha) on USFS land, 84 percent of which burned with surface fires. Of the 150 urban lots the team assessed, 80 percent burned with surface fire. About 164 ha of USFS fuel treatments burned with surface fire intensity. Many of the untreated stands were on steep ground with heavy fuel, and most burned in crown fires that consumed 95 to 100 percent of the tree crowns and surface vegetation. Untreated stands were mostly dense and multi-storied stands with abundant ladder fuels and moderate to heavy woody fuel and shrubs in the understory, including manzanita, bitterbrush (*Purshia tridentata*), and sagebrush. Untreated areas had trees with little commercial timber value, poor access, or adverse terrain for commercial logging or fire hazard reduction activities. Areas subjected to commercial logging, salvage logging, or other treatments not designed as fuel treatments mostly burned with high intensity crown fire. One partially treated unit where handpiles had been created but not burned exhibited fire behavior and effects similar to the untreated stands. Murphy and others (2007) found that most of the area fuel treatments reduced fire behavior from a crown fire to a surface fire. Area fuel treatments adjacent to subdivisions provided important safety zones for firefighters, which increased fire suppression effectiveness and helped firefighters save houses. Urban lot treatments reduced ember production, and reduced heat and smoke, allowing firefighters to be more effective. A large number of houses burned from firebrands that were generated from other burning houses rather than wildland fuel. Fuel treatment units on steep slopes burned at higher intensities than those on flat ground. Some fuel treatment units burned at high fire intensity because they were adjacent to and downwind from untreated units. Crown fire momentum carried high fire intensity partway into these treated areas before the more widely spaced crowns and reduced surface fuel load caused the fire to transition to the surface.

The lightning-ignited Antelope complex started 5 July 2007 and burned 9478 ha on the Plumas NF in California. Fites and others (2007) present findings and recommendations derived from evaluating the use and effectiveness of fuel treatments and fire behavior inside treated and untreated areas of dense, mixed conifer forest and shrubs. Their report is based on firsthand observation of fire behavior and suppression. Post-fire, the team quantified fire behavior and effects from field plots and Landsat satellite imagery. The fire burned through areas treated for fuel hazard reduction, untreated areas, and areas protected for California spotted owl (*Strix occidentalis*) and goshawk (*Accipiter gentillis*) habitat, as well as Riparian Habitat Conservation Areas. More than half of the area burned during the first two days when there were few firefighters working on the fire. Areas with fuel treatments had significantly reduced fire behavior, higher tree survival, and less impact on soils compared to untreated areas. Treated areas and recently burned areas had significantly lower burn severity than untreated areas, and burn severity was significantly higher in protected areas (owl and goshawk core and nest stands) than in other untreated portions of the landscape. Firefighters reported using treated areas in fire suppression. Treated areas provided safe escape routes for firefighters when other routes were not available. Even treated areas that burned intensely when few firefighters were present had reduced fire effects. Observations of fire behavior during the first two days suggested that large untreated areas allowed the fire to build momentum and contributed to increased rate of spread and intensity, making it more likely that suppression resources would be overwhelmed and treated areas would be threatened. As the wildfire burned in treated areas, it transitioned from crown fire or high intensity surface fire to moderate intensity surface fire. Fites and others (2007) recommended that more of the landscape be treated to reduce the likelihood of fires gaining momentum and increasing in behavior to a point where suppression and nearby fuel treatments become less effective. They also recommended treating protected areas to make them more resilient to fires and to prevent them from contributing to increased severe fire behavior across the landscape.

Dailey and others (2008) assessed fire behavior and effects on treated and untreated lands on the 2007 Moonlight fire, adjacent to the Antelope complex on the Plumas NF in California. Plume-dominated fire and long-range spotting exemplified intense fire behavior. From 3 September to 15 September, the wildfire burned 64,997 ha, including protected areas for California spotted owl and goshawk. Data from both randomly stratified field plots and Landsat satellite imagery were used in the assessment. Satellite dNBR data were instrumental in showing that most (68 percent) of owl core habitat areas had 75 to 100 percent canopy cover change, while about half (46 percent) of goshawk core areas had 75 to 100 percent canopy cover change, possibly rendering them inviable. Fire intensity and crown consumption were higher in the untreated protected areas than in treated areas. A smaller proportion of lands burned in the Moonlight fire had been treated compared to lands in the Antelope complex, rendering the fuel treatments less effective and making fire suppression more difficult. Daily and others (2008) recommended that larger portions of the landscape be treated to reduce the likelihood of fires gaining momentum and burning with high intensity into protected areas. They also suggested watershed-scale prescribed burns as a practical means to reduce fuel loads across broad areas of difficult terrain and in sensitive areas where other treatment options are limited.

Rogers and others (2008) assessed the Grass Valley fire, which burned in southern California in October 2007 under warm, dry weather and fuel conditions (relative humidity of 8 percent, live woody fuel moistures of 56 percent), Santa Ana winds of 29 to 64 kph, and gentle to steep slopes (less than 10 to greater than 60 percent). Many homes were threatened, and 199 structures were damaged or destroyed by the fire. Most of the area and many of the houses burned on the first day of the fire. The oak-shrub vegetation had an oak and pine overstory with dense white fir in the understory, was interspersed with chaparral dominated by manzanita, and had surface fuels of pine needles and oak leaves. Fuels had been treated on 30 percent of the 503 ha that were burned by the fire. Fuel treatments were designed to reduce crowning potential and ember production. Treatments varied but generally included removal of dead, dying, and diseased trees combined with thinning, pruning, chipping, and burning to reduce surface litter, woody fuel, and ladder and canopy fuel. More conifers than oaks were removed and more understory trees than overstory trees were removed, leaving widely spaced woodlands that were dominated by oaks and had discontinuous surface fuels. Along roads, removal of dead trees prior to the fire lessened the risk to firefighters working in and around structures, made evacuation routes safer, and reduced ember production and spot fires.

Other treatments were intended to make public evacuations safer while improving visibility and access for firefighters. The Grass Valley fire burned with lower flame lengths, slower rate of spread, fewer instances of transition to crown fire, and less spotting in treated areas than in adjacent untreated wildland fuels. As a result, firefighters were able to concentrate on evacuating people, protecting structures, and limiting fire spread. Fuel treatments improved visibility for the firefighters, but the fire burned more intensely within the residential area than in adjacent wildland fuels. Mass ember production from structures ignited adjacent and downwind structures in many cases. Lack of surface fire evidence in vegetation that surrounded burned homes provided strong evidence that house-to-house ignitions by airborne firebrands were responsible for many of the destroyed homes. Homes were close together and on steep slopes, and many had multiple wooden decks. The pre-fire removal of large-diameter dead trees from urban lots did little to reduce fire behavior once homes ignited. A Spotted Owl Protected Activity Center (PAC) burned in the fire. Trees survived in the portion of the PAC that had fuel treatments, while tree mortality was high in the untreated portion of the PAC. The authors identified three factors that contributed most to treatment effectiveness. First, the fuel treatments had been prioritized based on an integrated landscape look at hazardous fuels and terrain, fire weather and history, access, egress, and communities at risk. Second, treatments were planned and implemented to meet specific fire behavior objectives.

Third, treatments that were applied along roads and power lines and in urban areas all helped enhance suppression actions and enabled safe evacuation of the public.

Harbert and others (2007) interviewed local firefighters and fuel and vegetation managers, made field observations, collected relevant photographs, and reviewed fire operation and burn plan documents on three large 2007 fires east of the Cascade Range in Oregon: the Monument fire, the GW fire, and the Egley complex. Following is a summary of each fire.

The Monument fire burned 21,673 ha in July 2007 across a landscape with extensive but relatively low intensity fuel treatments that reduced severe fire effects. Most of the area affected by the Monument fire had previously been selectively logged. Since 1998, the Umatilla NF had implemented prescribed burns on 5487 ha within the fire perimeter. Areas that were previously underburned supported two-storied stands with light to moderate woody fuels and grass in the understory, while the untreated areas were dominated by multistoried stands with some plantations and meadows. Some areas had been prescribed burned more than once, and the interval between the most recent prescribed burn and the wildfire varied from 9 to 17 years. Few trees in treated areas died in crown fires, and the fire burned with less intensity and severity in treated areas. Of the fuel and vegetation treatments that had been applied on Umatilla NF lands prior to the Monument fire, only the underburn treatments significantly mitigated burn severity compared to the untreated areas.

The 2974-ha GW fire burned 2382 ha on the Deschutes NF and 591 ha of private timberland. Twenty-five percent of the USFS lands that burned had received prior fuel or other vegetation treatments. Dense, multi-story stands dominated where no treatments had been applied. The fire burned into intensive fuel treatments that were designed to reduce wildfire threat to Black Butte Ranch and was stopped with the help of favorable weather and effective fire suppression. The fuel and other vegetation treatments were useful for fire suppression in part because the Incident Commander knew about them and used them and several large, recent fires in fire operations.

The Egley complex burned 56,802 ha from 6 to 22 July 2007 under some of the most severe fire danger conditions experienced in the area in the previous 20 years. Fires in the complex threatened towns, private in-holdings, ranches, and Federal administrative sites. Harbert and others (2007) analyzed 39,872 ha burned on Malheur NF lands that supported open shrublands at low elevations, grass with sage and juniper, open ponderosa pine, and dense pine. Within the Egley complex, fuel and other vegetation treatments encompassed 42 percent of the burned areas assessed by the team. Treatment prescriptions were designed to accomplish ecosystem restoration and to improve firefighting effectiveness. Treatments generally included thinning with fuel treatment (fuels were usually piled and burned with underburning about five years later), commercial harvests, and underburning. These treatments were not intended to stop fire spread but to keep fires on the surface and enhance firefighter effectiveness. Where the Egley complex burned in treated areas, surface fire behavior predominated with only occasional torching, resulting in little firebrand production and overstory tree mortality and few spot fires. In areas where commercial thinning and piling had occurred and the piles were unburned at the time of wildfire, tree mortality was significant, particularly at the top of steep slopes. In contrast, tree mortality and burn severity were much lower in similar forests that had been treated with thinning and burning before the wildfire. Areas treated less than 12 years prior exhibited 15 percent less area with high and moderate fire severity effects. This was likely because the older treatments had more vegetation growth since treatment and, therefore, more available fuel. The more recent treatments also had more aggressive treatment prescriptions that removed more vegetation and fuel than the older prescriptions. In some areas, even well-designed and well-implemented fuel treatments were ineffective, as active crown fire caused significant tree mortality. The authors attributed this to extremely low live fuel moisture, very high winds, high slopes, or a combination of all three of these factors.

Harbert and others (2007) concluded that fuel treatments—both prescribed burning and more intensive fuel treatments, including thinning and burning—reduced fire intensity and severity. Intensive fuel treatments that were located along major ridgetop road systems were particularly useful in increasing fire suppression effectiveness. Thus, on all three fires, fuel treatments seemed to increase suppression effectiveness. Additionally, when Incident Management Teams had knowledge of treatments, they used the treated areas to plan and implement fire suppression. Harbert and others (2007) also evaluated treated and untreated areas with respect to burn severity inferred from satellite imagery using Burned Area Reflectance Classification (BARC) maps produced by the USFS Remote Sensing Applications Center and modified by local Burned Area Emergency Response (BAER) teams (Orlemann and others 2002). On the three fires studied, a higher proportion of hectares burned severely on untreated lands than on lands where fuel or other vegetation treatments had been applied prior to the fires. More recent treatments and higher intensity treatments reduced fire behavior and fire effects more effectively than older and less intense treatments.

Harbert and others (2007) had several key recommendations. They felt that land management agencies needed to develop and articulate a clear strategy for guiding treatments in order to reduce hazardous fuels, that monitoring must continue, and that information on the location and status of fuel treatments should be used in wildfire management strategies. The latter could be facilitated by providing maps of treated areas to Incident Management Teams. Further, Harbert and others (2007) recommended that quantitative data (not just anecdotal information and retrospective analyses) were needed, particularly regarding when and where fuel treatment performance was tested by wildfires.

Graham and others (2009) assessed the 2007 Cascade complex that burned through a variety of fuel treatments that were designed to protect over 70 summer homes and other buildings near Warm Lake in central Idaho. The fuel treatments modified fire intensity and allowed firefighters to protect all but two uninhabited structures. Beginning in 1996, treatments were designed to reduce the risk of wildfire affecting structures and other values at risk around Warm Lake. The prevailing winds were from the southwest, so priority areas for treatment were Boise NF lands immediately west of the residences along the western shore of Warm Lake. Both mechanical and prescribed fire treatments were used to reduce surface, ladder, and crown hazardous fuels. Mechanical treatments were usually hand-pile and burn treatments that thinned trees to a spacing of 3.0 to 4.6 m, pruned the lower limbs of residual trees up to 1.5 m high, and removed ladder and surface fuels. The covered piles were burned in the late fall or early spring and were monitored to ensure that at least 80 percent of the material was consumed. In a 61-ha mulch treatment, a vertical shaft machine was used to masticate fuels that were not subsequently burned. The cost to treat 3680 ha in the Warm Lake Basin in some fashion from 1996 to 2006 was over $1.65 million ($448/ha).

These WUI fuel treatments were tested by the 2007 Monumental and North Fork fires that merged near Warm Lake as part of the Cascade complex. These and at least 25 other wildfires were ignited by dry lightning from thunderstorms on 17 July on the Boise or Payette NFs in central Idaho. On 13 August, southwest winds pushed the Monumental fire into fuel treatments west and east of Warm Lake. On 14 August, west winds pushed the North Fork fire to the northwest of Warm Lake; from there, it spotted into the two northernmost treatment units on the west side of Warm Lake. Meanwhile, the Monumental fire burned farther into the treatment units east of Warm Lake. Both wildfires continued to progress, eventually merging on 17 August on both the northwest and northeast sides of Warm Lake. The merged fires continued to burn to the northeast and were not extinguished until snow arrived in early October. Graham and others (2009) provided a detailed narrative of the day-to-day fire behavior and spread as it burned through the Warm Lake vicinity. Intense crown fire behavior in the untreated forest along the river caused many spot fires into the treatment units, which fire crews could easily suppress. Knowledge that the fuel treatments reduced fire intensity to a manageable level helped Incident Command develop an AMR of strategic point protection. The treatment units provided an area of relative safety, from which firefighters could more safely and effectively conduct burnout operations and protect the structures and other values at risk in the Warm Lake community. The treatment units did not stop the fire progression but greatly modified fire behavior. Severe fire effects on vegetation and soils were more prevalent in areas that were not treated than in areas that were treated. Trees near the edge of treatment units were often scorched by the radiative heat from crown fires adjacent to the units or were burned by surface fires that spotted into the treatment units. The location of the fuel treatment units in relation to the fire progression, wind direction, roads, and topography was an obvious factor in fire suppression activities—one that greatly influenced the resulting pattern in burn severity in the post-fire landscape.

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Case Study

Study Areas

The Secesh Meadows community (45.245°; -115.822°) is located about 48 km north-northeast of McCall, Idaho, within the area that was burned by the 2007 East Zone complex (figure 1). The community stretches out along the Secesh River valley and is surrounded by steep, forested terrain on both sides of the riparian zone (figure 2). Several homeowners have implemented Firewise fuel treatments around their homes. A USFS campground is situated at the south end of the WUI area. The forest type is subalpine with lodgepole pine, Engelmann spruce (*Picea engelmannii*), and subalpine fir.

The Warm Lake community (44.645°; -115.688°) is located about 43 km east-northeast of Cascade, Idaho, within the area that was burned by the 2007 Cascade complex (figure 1). The community consists of over 70 structures, including summer cabins on leased Government land that surrounds Warm Lake. Lessees were unable to treat fuels near their homes on Government land. Two lodges, three public campgrounds, a youth camp, a church camp, and a USFS project camp were threatened by the wildfire. The forest type is mixed conifer with lodgepole pine, Douglas-fir, and ponderosa pine.

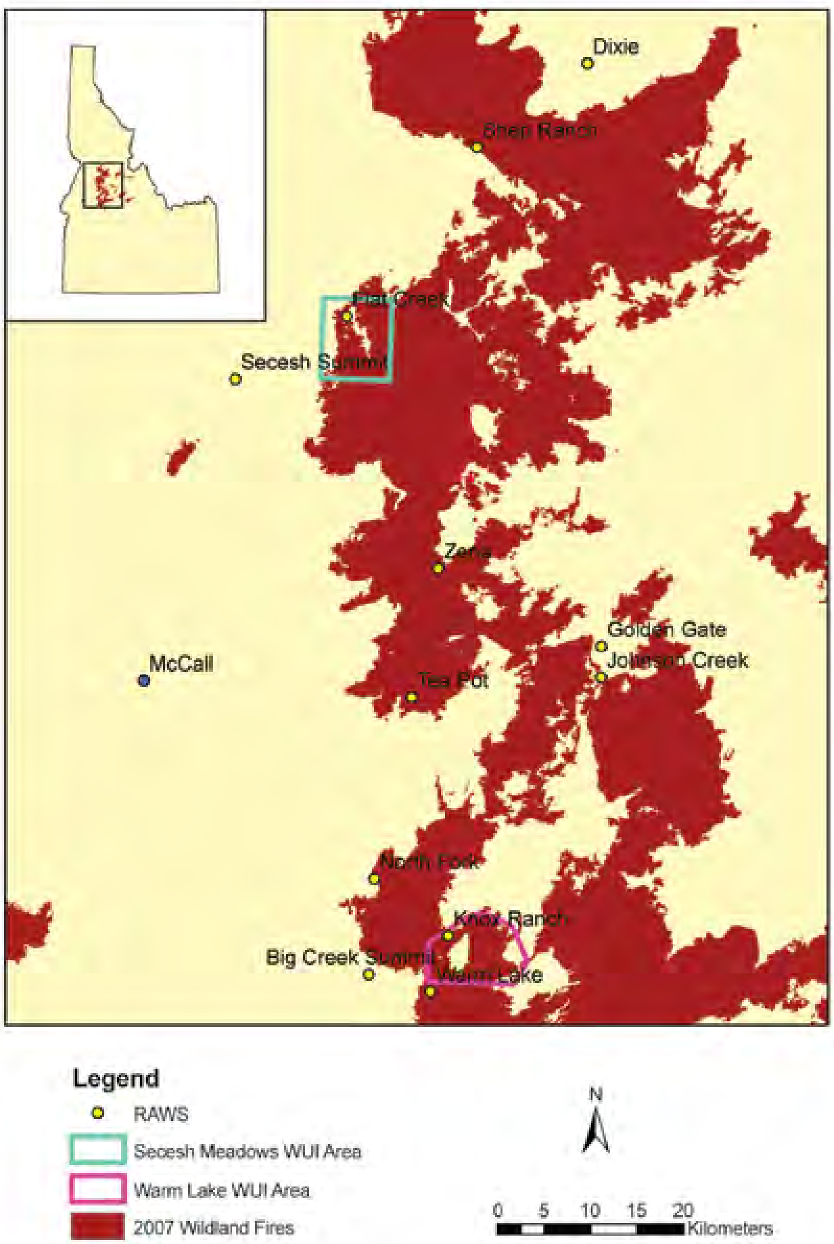


Figure 1. Location and extent of the East Zone and Cascade Complexes within central Idaho (inset). The city of McCall and the RAWS with weather data considered in this analysis are shown, as are the Secesh Meadows and Warm Lake study areas.

Figure 2. Secesh Meadows from the air, looking south toward Loon Lake and the Loon fire ignitions. *Photo: Roger Staats*.

Incident Information

On 7 July 2007, lightning strikes 40 km northeast of McCall, Idaho, ignited the wildfire(s) that grew and eventually merged into the 121,415-ha East Zone complex. The individual wildfires that merged were the Loon, Zena, Raines, Profile, and Horton fires (figure 3). The cost to fight these fires was over $32,500,000 (Independent Large Wildfire Cost Panel 2008), and the fires were contained on 30 September 2007.

Lightning ignited the first wildfire(s) of the Cascade complex 43 km east of Cascade, Idaho, 17 July 2007 at 1700. The North Fork, Monumental, and Riordan fires (figure 4) eventually merged into the 122,367-ha Cascade complex, with containment declared on 30 September 2007. The cost to fight the fires was over $40,700,000 (Independent Large Wildfire Cost Panel 2008). The East Zone and Cascade complexes eventually merged, and comprise a subset of the 2007 wildland fires that occurred in central Idaho (figure 1).

A point protection strategy was adopted to defend threatened structures in the local rural communities of Secesh Meadows (figure 5) and Warm Lake (figure 6). Difficult terrain, extreme fire weather, and exceptionally low fuel moisture (figure 7) all contributed to the strategy. The size of the wildfires and their severe fire behavior, added to the fact that 2007 was a very busy fire year and resulted in a shortage of firefighting resources, made point protection the only viable option to the incident management team (McCarthy and others 2008). The lack of distinct topographic ridgelines that were perpendicular to the prevailing winds and fire direction and close to the WUI made the WUI fuel treatment units important for effective fire suppression.

Figure 3. Loon Fire smoke plume on the first day the wildfire pushed into Secesh Meadows, looking southeast. *Photo: Roger Staats.*

Figure 4. Monumental Fire smoke plume in mid August 2007 east of Cascade, Idaho. *Photo: Ian Rickert.*

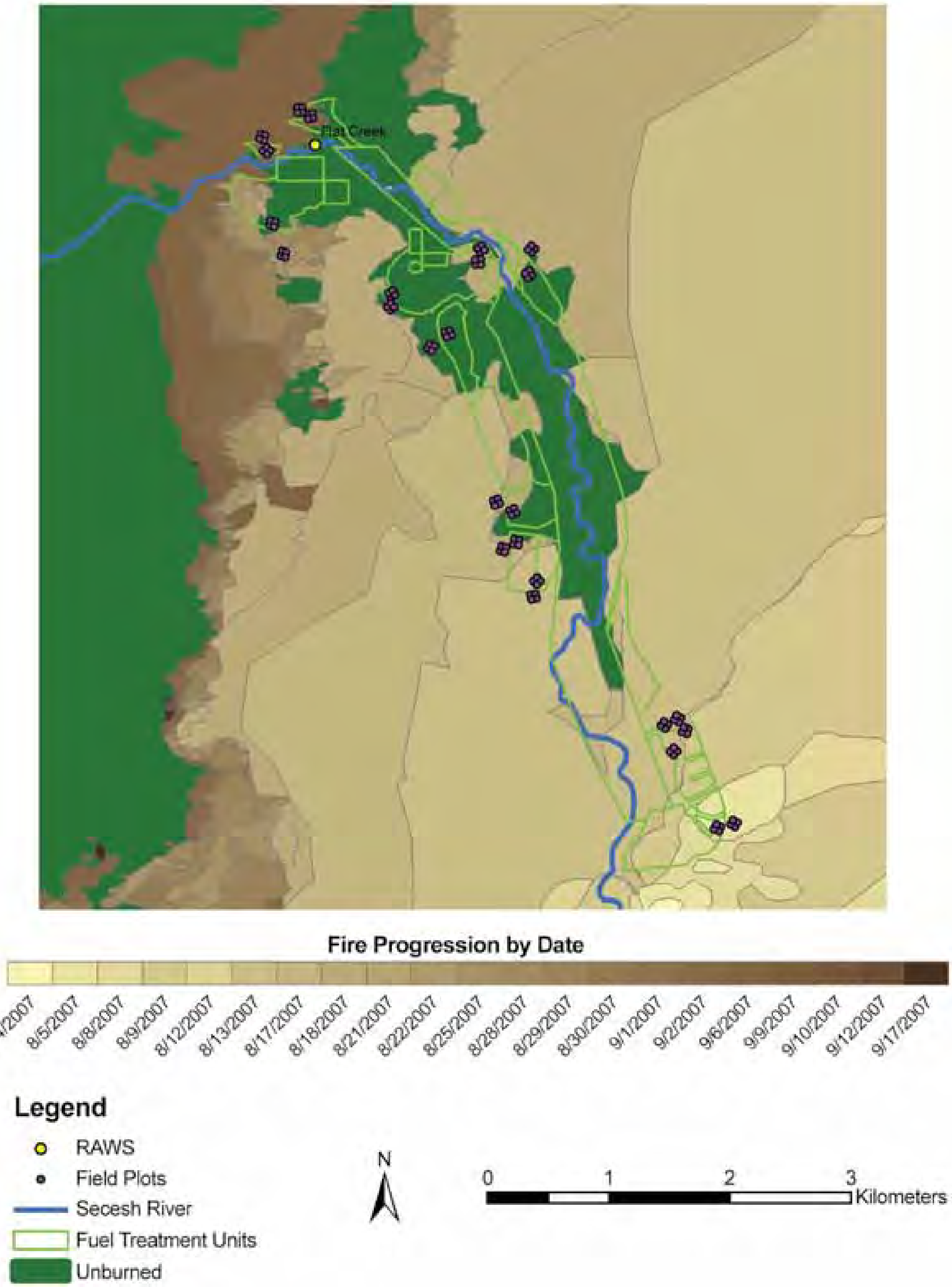


Figure 5. East Zone Complex progression map in the Secesh Meadows WUI area, with fuel treatment units and field plot locations from paired field sites (n = 13) overlaid.

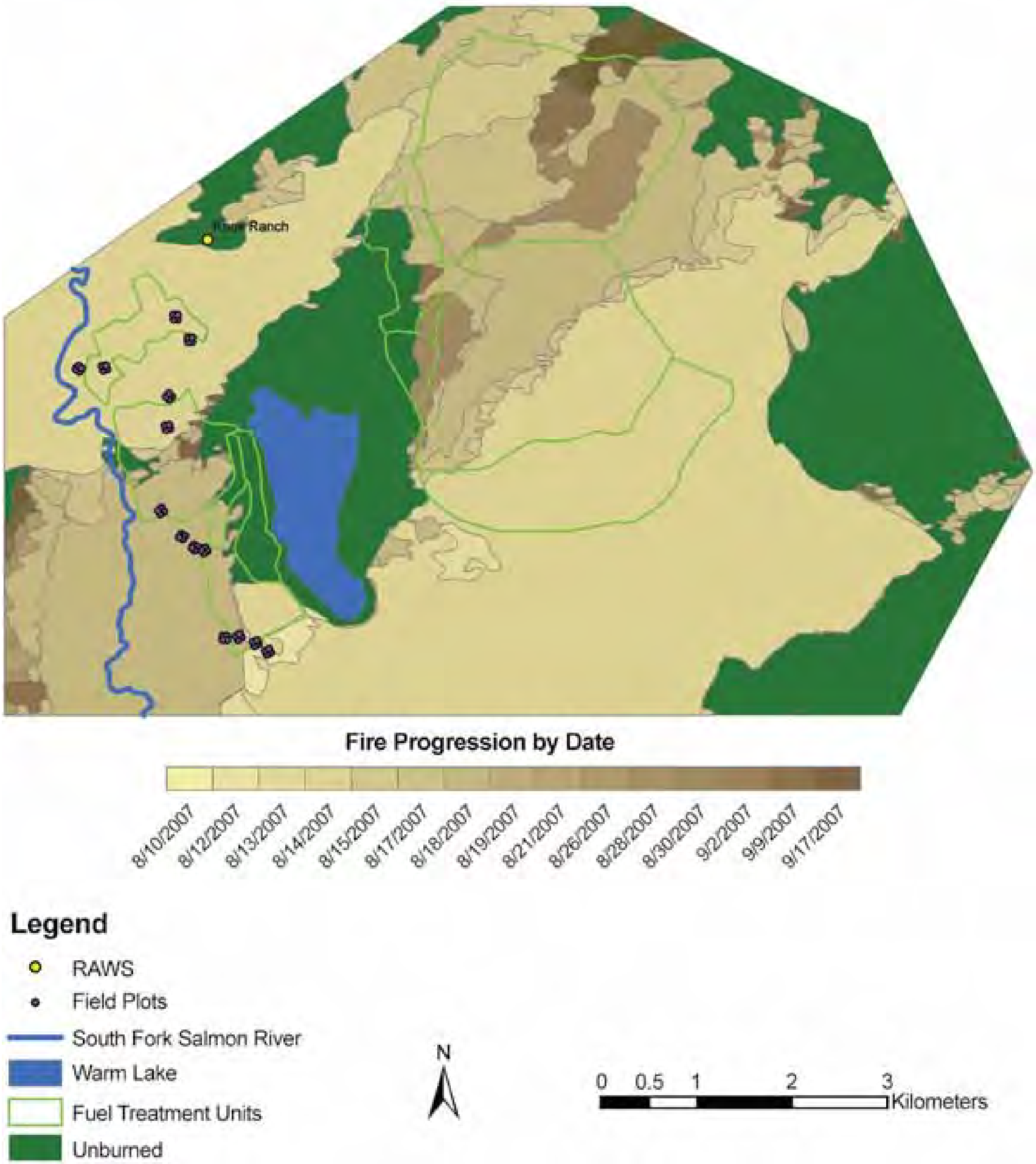
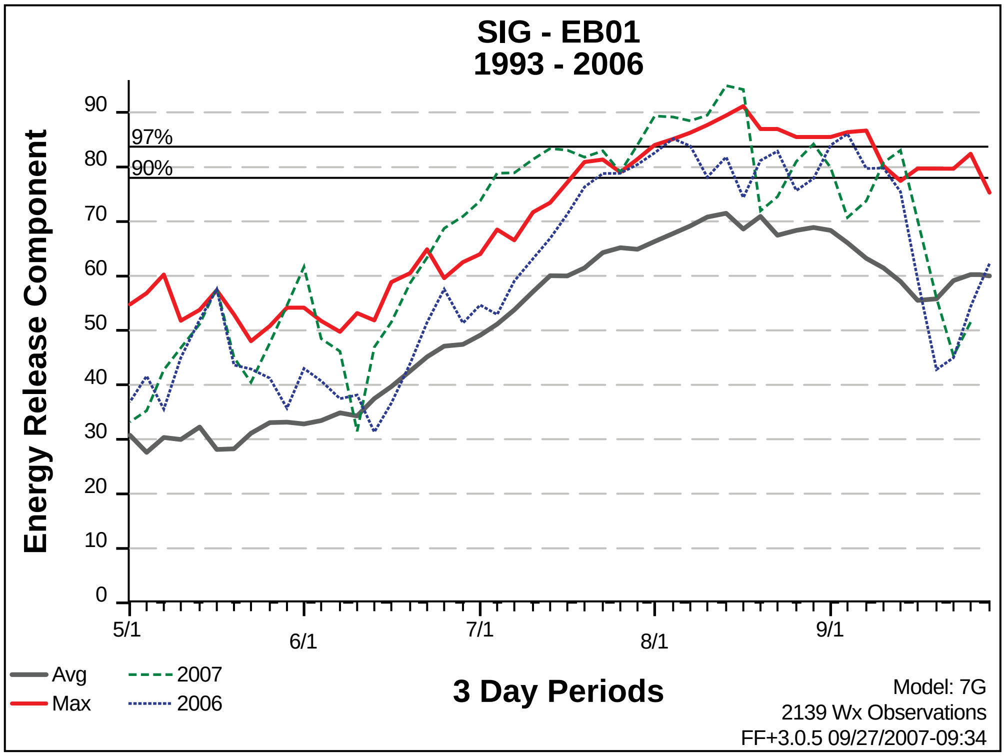


Figure 6. Cascade Complex progression map in the Warm Lake WUI area, with fuel treatment units and field plot locations from paired field sites (n = 7) overlaid.

(1)



(2)

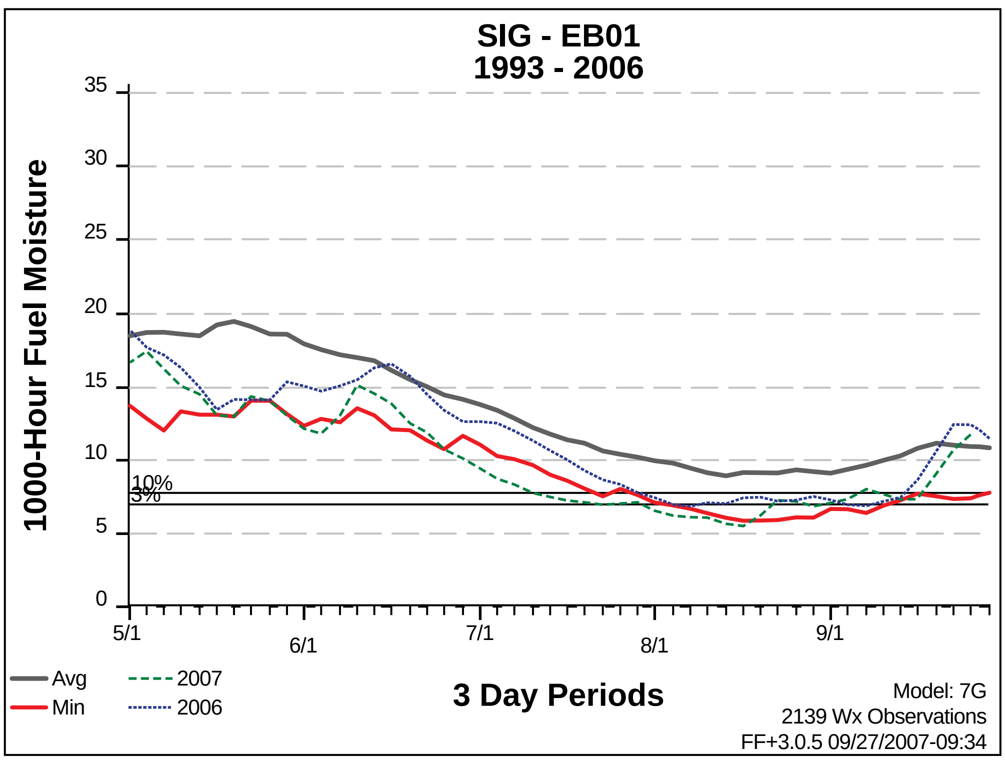


Figure 7. (1) Energy Release Component (ERC) and (2) 1000-hr Fuel Moisture in the west-central Idaho mountains where the 2007 East Zone and Cascade complexes occurred. From May through September, the extreme 2007 trends are plotted along with the 2006 trends, the 1993 to 2006 average trend, and the maximum ERC or minimum 1000-hr fuel moisture trends for the sake of comparison (http://gacc.nifc.gov/egbc/index.htm).

Fuel Treatments

Pile and burn fuel treatments were implemented in 2006 at Secesh Meadows. Thinned trees were piled in preparation for burning (figure 8). The piles in eight units had been burned prior to the wildfire; however in five units, the piles had not yet been burned as prescribed ("Rx Piles") but instead were burned in the wildfire ("WF Piles") (table 1). A wider variety of fuel treatments were implemented at Warm Lake over a period of 10 years, including pile and burn, mastication, and underburn treatments (table 1).

Methods

Initial field assessment

Fire effects on vegetation and soils were assessed initially in the field from 18 to 21 September 2007. We used paired field sites to compare fire effects in treatment units relative to adjacent untreated lands. NF managers helped locate fuel treatments that had burned through by wildfires. Once a fuel treatment unit was selected for sampling, five 1-m^2 subplots were used to measure charred and uncharred ground cover fractions, litter and duff depths, and overstory canopy closure;

Figure 8. Activity fuels prior to pile burning (*top left*), being ignited (*top right*), and burning prior to the wildfire (*bottom left and right*). *Photos: Paul Klasner.*

Table 1. Fuel treatment description, year (if applicable), and hectares treated for the (A) Secesh Meadows and (B) Warm Lake WUI areas.

Area description	Treatment year	Hectares
(A) Secesh Meadows WUI Area		
Firewise (private lands; 5 units)	2005	277
Pile and Burn (Rx piles; 8 units)	2006	156
Pile and Burn (WF piles; 5 units)	2006	63
Untreated		5774
Secesh River (50-m buffer)[a]		130
Total		**6400**
(B) Warm Lake WUI Area		
Mastication		
Warm Lake Highway	2004	61
Pile and Burn		
Warm Lake South	2000	76
Warm Lake North	2003	16
Warm Lake East	2005	38
Pile and Burn, followed by Prescribed Surface Fire		
Church Camp	2004	28
Paradise Valley	2005	72
Underburn		
Warm Lake Creek	1996	268
Chipmunk Creek	1997	461
Reeves Creek	1998	486
Kline Mountain	2006	122
Untreated		4502
South Fork Salmon River (50-m buffer)[a]		69
Warm Lake (50-m buffer)[a]		201
Total		**6400**

[a] The riparian zone near the water was excluded from the BARC map data analysis.

the five subplots were situated in a systematic pattern per site (figure 9), with the rule that no subplot should be closer than 5 m from the edge of the fuel treatment unit. The central subplot was situated in a random location near the center of the treatment unit, and the four other subplots were placed 30 m away from the central subplot in the four cardinal directions (figure 9). We recorded tree species, diameter at breast height (dbh), and condition in a 1/50-ha fixed-radius (8.0 m) plot situated at the center of the field site. An identical plot configuration was randomly placed in an adjacent untreated site that had a similar topographic position.

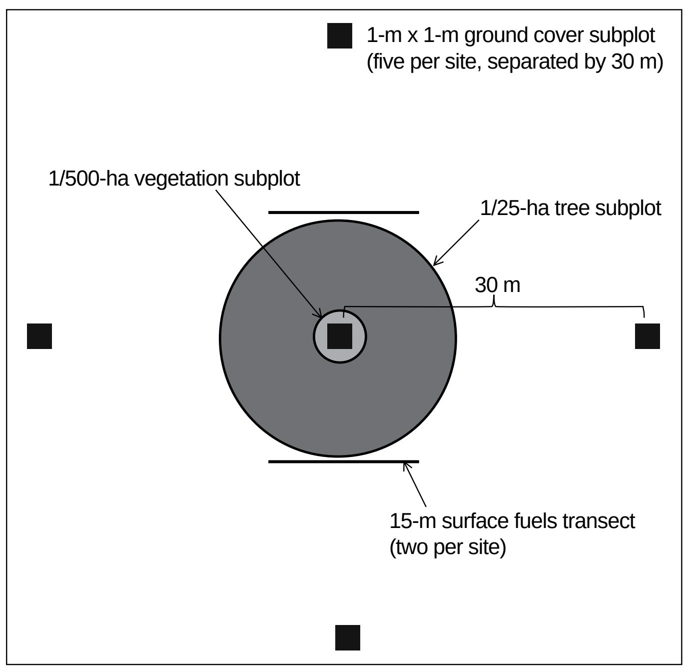


Figure 9. Systematic plot and subplot configuration at a field sampling site.

Extended field assessment

We conducted a more thorough field assessment in August 2008, one year after the fires. There were several notable differences between our initial and extended assessments. We expanded our field site count from 10 paired sites in 2007 to 20 paired sites in 2008, with the 2008 assessment including re-measurement of all of the 2007 sites. We learned late in the 2007 assessment that both sites in one of our site pairs were actually within a treatment unit, just on opposite sides of a road that was used as a firebreak. We resolved this complication in 2008 by reclassifying both sites as “treated” and establishing two new untreated sites to complement the treated sites, resulting in two paired sites. In other words, of the 20 field sites measured in 2007, 11 sites were treated and 9 were untreated. We expanded the 1/50-ha fixed-radius (8.0 m) plots from our 2007 assessment to 1/25-ha fixed-radius (11.3 m) plots in order to tally more trees (figure 9). In 2008, we also measured tree, crown base, maximum scorch, and bole char heights using a laser rangefinder. We measured one year post-fire vegetation re-growth in a 1/500-ha (2.5 m radius) subplot that was situated at the center of each field site (figure 9), where we ocularly estimated the percent cover of each plant functional type and listed the species encountered. We added Brown’s (1974) transects to our sampling protocol to measure surface fuels following a slightly modified FIREMON sampling protocol (Lutes and others 2006). The 30-m coarse woody debris (CWD) transect was divided into two 15-m sections situated on either side of plot center in order to preclude any directional bias (figure 9). Fine woody debris (FWD) was sampled along the central 2-m section of each 15-m CWD transect. We doubled our field site count, expanded the tree plot size, and measured additional variables in 2008 because we had more time to sample than in 2007 when the East Zone and Cascade complexes were still officially active and we needed to work expediently.

We used the tree diameter and sapling tallies in our 1/25-ha fixed-radius plots to estimate the plot-level basal area of trees and saplings. To retrospectively estimate the basal area of trees and saplings that were removed by either the fuel treatment or natural disturbance, we tallied the stumps in each plot within three diameter classes with the following midpoints: small (5.1 cm), medium (17.8 cm), and large (30.5 cm). We excluded any remaining bark on the stump from the diameter measurement. To convert the stump diameters to estimates of dbh, we multiplied by a ratio of 0.9 based on Bones (1960), who found dbh/stump ratios of 0.937 for lodgepole pine, 0.865 for subalpine fir, and 0.832 for Engelmann spruce in the Pacific Northwest. Thus, 0.9 served as a good average approximate ratio for the primary species encountered in this study. Rescaling the stump diameter measures to dbh allowed us to equitably compare the basal area of standing trees and saplings to those that were removed or were otherwise absent. Species could not be reliably called from just the stumps, but was recorded for the trees and saplings so that biomass could be estimated from the dbh measures and allometric equations (Jenkins and others 2003). The FWD and CWD fractions were combined for a total downed woody debris (DWD) estimate, which was converted to biomass following Brown (1974) for FWD and Harmon and Sexton (1996) for CWD.

Data collected at the five subplots per site (figure 9) were averaged to represent the site, which was our experimental unit. The paired site sampling design made pairwise comparisons a simple yet powerful method to assess the significance of differences between treated and untreated sites. Furthermore, separate tests were applied to each field variable measured to determine clearly which specific vegetation or soil measures were influenced by the fuel treatment and/or the wildfire and which were not. The Shapiro-Wilk statistic (Royston 1982, 1995) was used to test the normality of the field measures from both the treated and untreated sites, and the histograms were inspected. Most (70 percent) of the distributions were significantly non-normal (p-value less than 0.1, Royston [1995]); therefore, the non-parametric Wilcoxon signed rank test (Bauer 1972) and a significance level of $\alpha = 0.05$ was used to assess the significance of differences between paired sites. All statistics presented in this paper were run in R (R Development Core Team 2005) unless stated otherwise.

Satellite assessment

The NBR was developed as an index of burn severity that could be simply calculated by taking the difference between Landsat Thematic Mapper (TM) bands 4 and 7 and then dividing that quantity by their sum (Key and Benson 2005). Typically, this operation is applied to both pre- and post-fire images acquired at nearly the same time of year (to control for variable sun angle and vegetation phenology). Then, the post-fire NBR image is subtracted from the pre-fire NBR image to produce the dNBR, which is the most widely used index of burn severity (Key and Benson 2005, Hudak and others 2007). The USFS Remote Sensing Applications Center (RSAC) produces preliminary BARC maps from continuous dNBR values. The maps are delivered to incident management teams on important wildfires and are used as preliminary inferences of unburned, low, moderate, and high burn severity classes, as defined by Key and Benson (2005). If necessary, the thresholds that demarcate the four preliminary burn severity classes are adjusted by BAER teams working on the incident in order to more accurately represent conditions observed on the ground (Hudak and others 2007).

Because the East Zone and Cascade complexes were such high priority fires, RSAC produced three BARC maps using dNBR values derived from an 11 October 2004 Landsat 5 Thematic Mapper pre-fire image and either Landsat 5 TM or Landsat 7 Enhanced Thematic Mapper Plus (ETM+) post-fire images that were acquired 25 August (ETM+), 2 September (TM) and 26 September 2007 (ETM+) in the case of the East Zone complex and on 25 August (ETM+), 10 September (ETM+), and 26 September 2007 (ETM+) in the case of the Cascade complex. We used the latest ETM+ image in a BARC map data analysis of immediate post-fire severity because local wildfire activity within our two study areas had ceased by 26 September 2007. In this analysis, we calculated the areal percentage of hectares that burned at high severity on untreated lands, assumed this to be the percentage expected to burn at high severity in the absence of any fuel treatment, and then compared the expected percentage to the observed percentage of hectares that burned at high severity within each fuel treatment, as observed in the immediate post-fire BARC map. We considered each treatment unit as a replicate. Because the observed percentages were not normally distributed according to the Shapiro-Wilk statistic (Royston 1982, 1995), we tested the significance of differences between observed and expected percentages using Wilcoxon rank sum tests (Bauer 1972) and a significance level of $\alpha = 0.05$.

A problem with ETM+ imagery since 31 May 2003 is the failure of the scan line corrector mechanism on board Landsat 7, which causes data voids in the image parallel to the scan direction (perpendicular to the satellite path) that widen toward the edges of the scene. A majority filter is employed to fill the categorical data gaps in the classified BARC map and produce a more visually satisfying result, but this technique is inappropriate for filling the gaps in the continuous dNBR images. Unfortunately, a large proportion of our field sites (40 percent) fell into those dNBR data voids. To solve the problem, two successive ETM+ images that were collected on 10 July 2008 and 26 July 2008 were merged. The data gaps in the two scenes fortunately offset just enough to produce a composite one year post-fire image with continuous coverage over all of our field sites. A 13 July 2006 Landsat TM image provided the pre-fire NBR values needed to calculate dNBR. These one year post-fire dNBR data were more appropriate than immediate post-fire dNBR data for our paired site comparisons because the bulk of our field data were collected one year post-fire. After confirming that both the treated and untreated dNBR distributions were normal with the Shapiro-Wilk test (Royston 1982), we used paired t-tests and a significance level of $\alpha = 0.05$ to assess the significance of differences between treated and untreated sites.

Fire weather assessment

We were able to obtain weather data online (http://mesowest.utah.edu/index.html) from nine Remote Automated Weather Stations (RAWS) located in or near the East Zone or Cascade complexes, as well as from three portable RAWS (Flat Creek, Warm Lake, and North Fork) deployed near the wildfires to provide up-to-date local weather data to fire managers (figure 1). Variables recorded at RAWS on an hourly basis included air temperature, dewpoint temperature, relative humidity, precipitation, wind speed, and wind gust speed. We computed daily means from the hourly data that were available within the 63-day period from 18 July to 18 September 2007. Plotting the daily means against Julian Date revealed considerable daily variation among the 12 RAWS, which would be attributable to their wide geographic separation that encompassed a range of local weather conditions. Nevertheless, we averaged the daily means across the 12 RAWS to generate regional daily means for the 6 weather variables named above. The response variable (area burned per day; ha/day) was tallied from the East Zone and Cascade fire progression data that were obtained from the incident camp GIS team located just outside of Cascade, Idaho.

We employed ordinary least squares (OLS) regression to predict area burned per day from the weather variables, not for the purpose of prediction but to quantify the influence of the weather variables on fire activity (Hawkins 2004). Care is needed when using linear regression with time series data (Chambers 1992). Therefore, we examined the OLS model residuals for autocorrelation at time lags ranging from one to five days, and tested the significance of the autocorrelation using the Durbin-Watson statistic (Vinod 1973) in SAS. This test was imbedded in the SAS AUTOREG procedure (Gallant and Goebel

1976) that generated OLS models based on the six weather variables in all possible combinations (n = 63) and tested the residuals from every model for significant autocorrelation. In the models that exhibited significant autocorrelation, backward elimination of one- to five-day autoregressive (AR) error terms was employed until only the significant AR error terms remained. The best subset model from among the 63 possible models that were estimated using maximum likelihood was selected based on the lowest Akaike Information Criterion statistic (Burnham and Anderson 1998). This resulted in a best subset model that was parsimonious in that it consisted of only significant RAWS variables and significant AR error terms. All residual Durbin-Watson test statistics were non-significant after fitting the autoregressive error models, and the histograms of residuals were inspected to confirm normality.

Results

Our quantitative case study results consider all three components of the fire behavior triangle to some extent and are divided into six sections:

(1) the influence that fire weather had on landscape-level fire activity;

(2) whether mitigation of severe fire effects by fuel treatments was detected by satellite imagery that was acquired immediately post-fire and one year afterward;

(3) whether our paired site design controlled for topographic effects;

(4) a retrospective comparison of treatment effects on pre-fire fuel loads between treated and untreated field sites;

(5) a comparison of one year post-fire effects between treated and untreated field sites; and

(6) a comparison of immediate versus one year post-fire effects to compare site recovery between treated and untreated field sites.

Fire weather assessment

Temporal autocorrelation trends were evident not just in the individual RAWS records but in the means calculated across all 12 RAWS. Temporal autocorrelation indicates stability in weather patterns during the burning period, such as from inversions, and supports the fact that temperature can be more reliably predicted at a regional scale than cloud conditions or precipitation that vary more at finer scales. We found that daily burned area across the full regional extent of the East Zone and Cascade complexes could not be significantly predicted from autoregression models based on the daily mean weather time series variables averaged across the 12 RAWS ($R^2 = 0.10$, p-value = 0.213).

Fire weather conditions vary greatly at local scales. Therefore, we used the same approach to predict area burned within each of the local WUI areas using the daily means from just the local RAWS. The Flat Creek portable RAWS in the northern portion of the Secesh Meadows WUI area (figure 5) provided the most complete and proximal weather dataset for analysis. Trends in the weather variables recorded there most closely resembled the trend in area burned per day, particularly the spike on the big blowup day of 14 August (Julian Date 226, figure 10). The best subset autoregression model that predicted daily area burned within the Secesh Meadows WUI area was based on relative humidity, precipitation, and four- and five-day autoregressive terms (table 2). The next closest RAWS with a complete record was the Zena RAWS that was located approximately 21 km south-southeast; however, the best subset model based on dewpoint temperature and four- and five-day autoregressive terms was non-significant (table 2).

The Warm Lake portable RAWS was deployed just 1 km southeast of the Warm Lake WUI area, but the wildfire actually destroyed it on 17 August 2007 after only 10 days of recording, which was too short a period for meaningful analysis. The Knox Ranch RAWS was situated in the northern portion of the Warm Lake WUI area (figure 6), but weather records from this RAWS only began on 27 August 2007, almost entirely after the wildfire already had burned through the WUI area. The closest meaningful record was from the North Fork portable RAWS that was located about 9 km northwest, which explained much of the variation in area burned within the Warm Lake WUI (table 2) based on dewpoint temperature, wind gust speed, and a four-day autoregressive term. The closest RAWS with a complete record was the Tea Pot RAWS that was located approximately 24 km north-northeast; the best subset model was significant based on dewpoint temperature and a four-day autoregressive term (table 2).

Satellite assessments

The size of both the Secesh Meadows and Warm Lake WUI analysis areas was delimited at 6400 ha to include not just the treatment units but also the

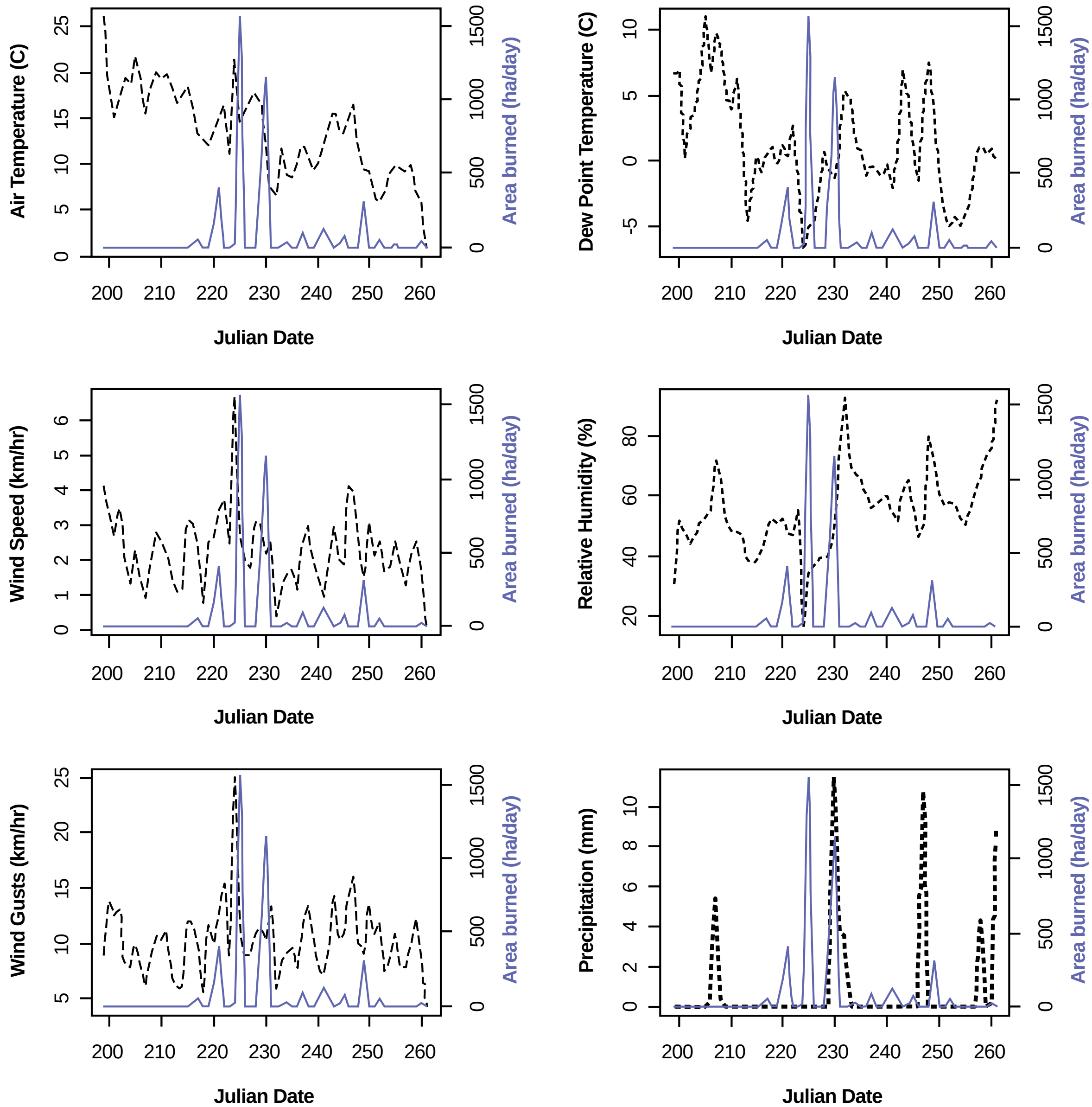


Figure 10. Daily averages of six weather variables recorded hourly at the Flat Creek Fire RAWS at Secesh Meadows from 4 August to 17 September 2007. Area burned per day (ha/day) is plotted as a solid blue line in each graph to assist in comparing trends.

Table 2. Best subset multiple linear regression models with autoregressive (AR) terms for predicting area burned in the (A) Secesh Meadows and (B) Warm Lake WUI areas from RAWS variables (daily means). Significant differences (α = 0.05) are indicated in **boldface**.

WUI area and RAWS	Significant model parameters[a]	Days sampled	AIC	Total R^2	P-value
(A) Secesh Meadows WUI Area					
Flat Creek Portable RAWS	Relative Humidity, Precipitation, AR4, AR5	63	**859.74**	**0.34**	**0.029**
Zena RAWS	Dewpoint Temperature, AR4, AR5	63	863.01	0.28	0.121
(B) Warm Lake WUI Area					
North Fork Portable RAWS	Dewpoint Temperature, Wind Gust Speed, AR4	31	**472.82**	**0.44**	**0.004**
Tea Pot RAWS	Dewpoint Temperature, AR4	63	**926.51**	**0.25**	**0.046**

[a] AR4 indicates significant four-day autocorrelation, and AR5 indicates significant five-day autocorrelation.

surrounding areas of relevance (table 1). At Secesh Meadows (figure 11), private lands with Firewise treatments comprised 277 ha, while treatments funded by the National Fire Plan (NFP) on Payette NF land comprised 220 ha (3.5 percent) of the WUI area (excluding a 50 m buffer zone on either side of the Secesh River). At Warm Lake (figure 12), NFP-funded treatments on Boise NF land covered 1645 ha (26.5 percent) of the WUI area (excluding a 50-m buffer zone on either side of the South Fork Salmon River and around Warm Lake). Based on the BARC map data at Secesh Meadows, a significantly lower areal percentage of hectares within pile and burn treatment units burned at high severity (provided the piles were burned as prescribed) compared to untreated lands (p-value = 0.008) (figure 13). Firewise treatments on private lands were most effective for significantly reducing high severity fire at Secesh Meadows (p-value = 0.007). At Warm Lake, there were too few treatment units to result in any significant differences by treatment type (table 1),

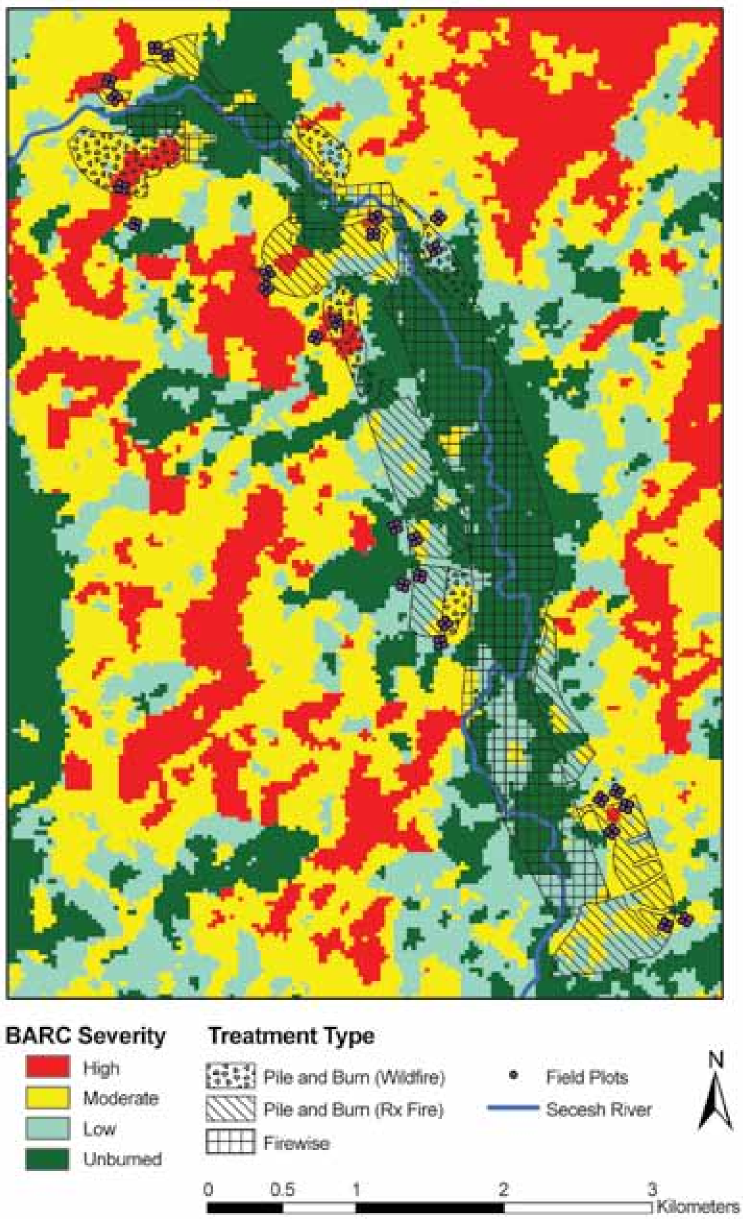


Figure 11. BARC classification of dNBR values derived from 11 October 2004 Landsat TM pre-fire and 26 September 2007 Landsat ETM+ post-fire images, indicating immediate post-fire burn severity across the Secesh Meadows study area.

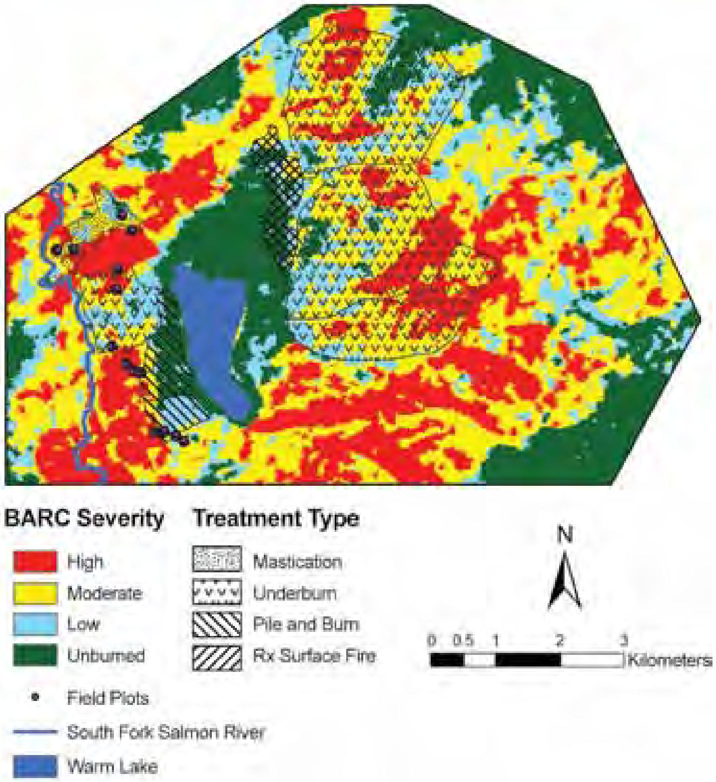


Figure 12. BARC classification of dNBR values derived from 11 October 2004 Landsat TM pre-fire and 26 September 2007 Landsat ETM+ post-fire images, indicating immediate post-fire burn severity across the Warm Lake study area.

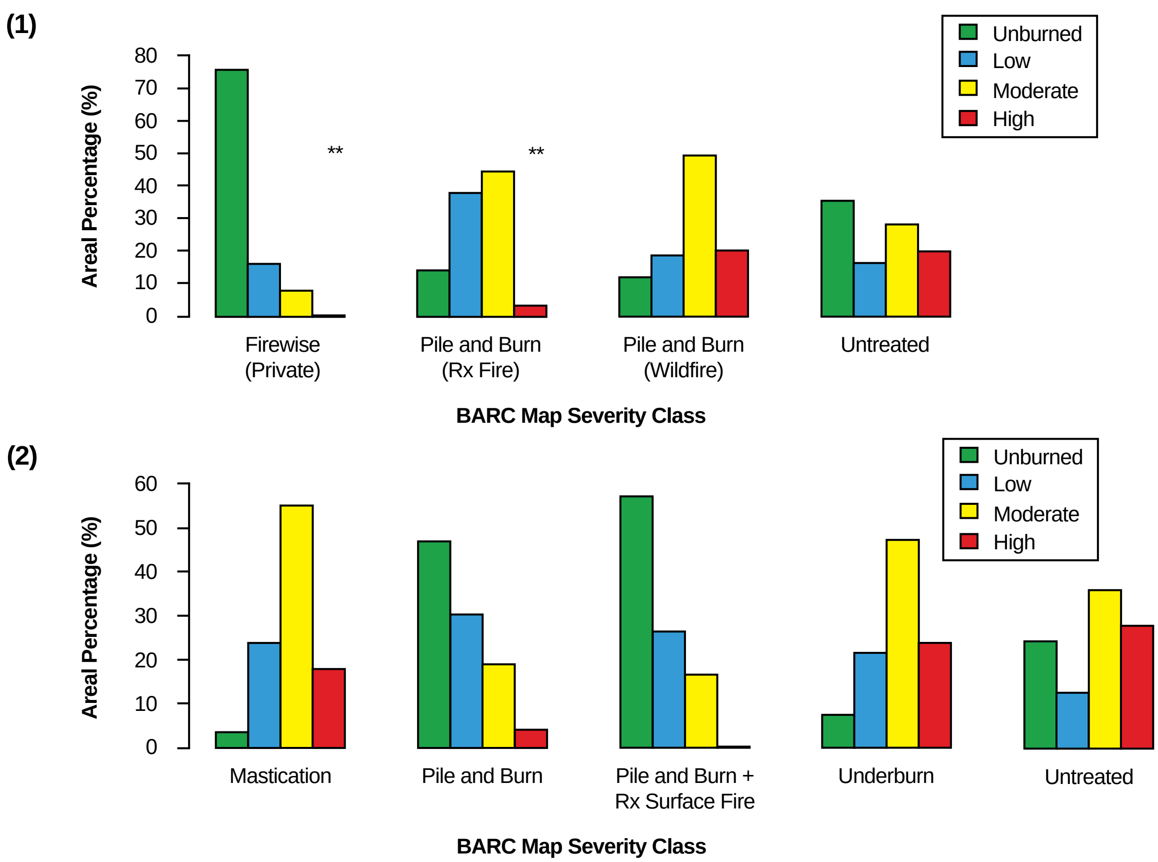


Figure 13. Areal percentages of immediate post-fire BARC map severity classes in different fuel treatment types at the (1) Secesh Meadows and (2) Warm Lake study areas. Significance: **, p-value<0.01.

but when the 10 units were considered together, there was a significantly lower percentage of high severity hectares inside treatment units compared to outside (p-value = 0.001). Based on the BARC map data, the combination of pile and burn and prescribed surface fire treatments was slightly more effective for mitigating high burn severity than piling and burning only; the mastication treatment was less effective; and the underburns were least effective (figure 13).

This evidence from the immediate post-fire BARC map data assessment that the fuel treatments mitigated fire severity was corroborated by the paired site comparison of one year post-fire dNBR. The dNBR was significantly lower at treated sites than at untreated sites at Warm Lake (p-value = 0.014), as well as at the "Rx Pile" treated sites at Secesh Meadows (p-value = 0.025). The "WF Pile" treated sites at Secesh Meadows exhibited the opposite trend, although this difference was not significant (p-value = 0.764) based on only four paired sites.

Topographic effects

The intent of the paired site design was to minimize the confounding influences of the fire weather and topography components of the fire behavior triangle to isolate the fuels component (figures 14 through 19). The spatial distribution of the RAWS was much too sparse to know if fire weather differed significantly between our treated and untreated site pairs. We can only assume that it did not, which is reasonable given

Figure 14. Secesh Meadows pile and burn treated site (left) where the fuel piles were burned prior to the wildfire ("Rx Piles") versus its paired untreated site (right) nearby, one month after the wildfire. The fuel treatment effect on fire severity at this site pair was positive. *Photos: Andrew Hudak.*

Figure 15. Secesh Meadows pile and burn treated site (left) where the fuel piles were burned prior to the wildfire ("Rx Piles") versus its paired untreated site (right) nearby, one month after the wildfire. The fuel treatment effect on fire severity at this site pair was neutral. *Photos: Andrew Hudak.*

Figure 16. Secesh Meadows treated site (left) where the fuel piles burned in the wildfire ("WF Piles") versus its paired untreated site (right) nearby, one month after the wildfire. The fuel treatment effect on fire severity at this site pair was negative. *Photos: Andrew Hudak.*

Figure 17. Warm Lake South pile and burn treated site (left) versus its paired untreated site (right) nearby, one month after the wildfire. *Photos: Andrew Hudak.*

Figure 18. Warm Lake Highway mastication treated site (left) versus its paired untreated site (right) nearby, one month after the wildfire. *Photos: Andrew Hudak.*

Figure 19. Warm Lake Kline Mountain underburn treated site (left) versus its paired untreated site (right) nearby, one month after the wildfire. *Photos: Andrew Hudak.*

that the distance separating the paired sites ranged from 100 to 350 m. Most site pairs were within sight of each other unless a hillslope intervened. While we could not test for differences in fire weather, we tested whether elevation, slope, and aspect as recorded in the field differed between treated and untreated site pairs. Wilcoxon signed rank tests of these variables revealed no significant differences at either Secesh Meadows or Warm Lake, with the exception of elevation at Secesh Meadows (p-value = 0.01). Although statistically significant, the mean difference between untreated and treated sites at Secesh Meadows was only 11 m. This was because the Secesh Meadows community is situated in a valley along the Secesh River where, in 10 of 13 cases, the untreated sites were situated upslope of the treated sites placed in WUI treatment units that were designed to protect structures along the valley bottom. Steeper slopes appeared to interact with fuel load and contributed to more severe fire effects at some paired sites; in other words, the site that had higher surface fuel loads sometimes burned more severely, whether treated (figure 16) or untreated (figure 19).

Treatment effects

The fuel treatments removed nearly all of the saplings and many of the trees at Secesh Meadows and Warm Lake (figures 20 and 21). We did not record whether or not stumps had been cut by a chain saw, but at both Secesh Meadows and Warm Lake, the density and basal area of stumps plus saplings and trees at treated sites was approximately twice that of saplings and trees at untreated sites, meaning the fuel treatments might have accounted for only half of the stumps we tallied (figures 20 and 21). Wilcoxon signed rank tests indicated that the differences in tree/sapling/stump density and basal area between treated and untreated sites were significant at Secesh Meadows but not at Warm Lake (table 3). Treatment effects on tree density, basal area, and biomass were not as pronounced at Warm Lake as at Secesh Meadows. Had the number of site pairs at Warm Lake (n = 7) matched that of Secesh Meadows (n = 13), it is likely that the density, basal area, and biomass of saplings would have significantly differed (table 3). It is likely that fewer stumps remained in the treatment units at Warm Lake because of the nature of the fuel treatments (for example, mastication and underburn) and because the 2000 to 2005 Warm Lake fuel treatments were one to six years older than the 2006 Secesh Meadows fuel treatments.

Estimates of tree and sapling biomass (figure 22) might better reflect *pre-fire* biomass than *post-fire* biomass, because the Jenkins and others (2003) allometric equations were based on live trees. Nevertheless, the relative differences are still useful and informative. Tree and sapling biomass was lower at treated sites than at untreated sites at both Secesh Meadows and Warm Lake (figure 22), although this treatment effect was significant only at Secesh Meadows (table 3). DWD biomass was higher at untreated sites at Secesh Meadows and, in contrast, at treated sites at Warm Lake (figure 22). However, these opposing trends were not significant for DWD or for the FWD or CWD components of DWD (table 3). These differences in *post-fire* surface fuels (DWD) does not preclude the possibility

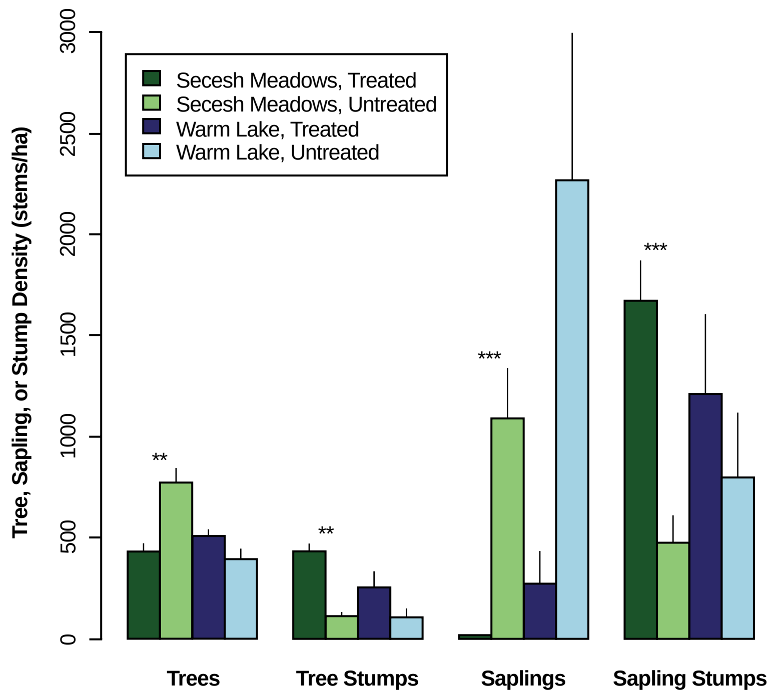


Figure 20. Mean (+SE, vertical lines) one year post-wildfire stem density of trees and saplings measured on treated versus untreated site pairs at Secesh Meadows (n = 13) and Warm Lake (n = 7). Stumps were also tallied to estimate stems removed from the site due to treatment effects on treated sites or natural disturbance and mortality effects on both treated and untreated sites. Significance: **, p-value<0.01; ***, p-value<0.001.

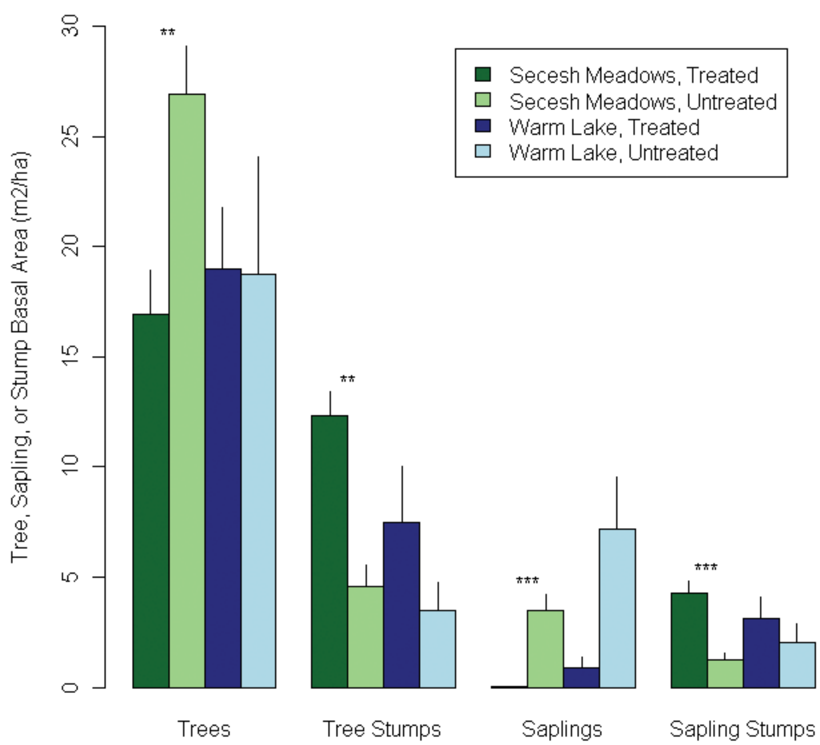


Figure 21. Mean (+SE, vertical lines) one year post-wildfire basal area of trees and saplings measured on treated versus untreated site pairs at Secesh Meadows (n = 13) and Warm Lake (n = 7). Stumps were also tallied to estimate basal area lost from the site due to treatment effects on treated sites or natural disturbance and mortality effects on both treated and untreated sites. Significance: **, p-value<0.01; ***, p-value<0.001.

Table 3. WUI treatment effects measured retrospectively at Secesh Meadows and Warm Lake. Significant differences (α = 0.05) indicated in **boldface**.

	Secesh Meadows				Warm Lake			
Variable description	Site pairs	Treated mean (SE)	Untreated mean (SE)	Wilcoxon p-value	Site pairs	Treated mean (SE)	Untreated mean (SE)	Wilcoxon p-value
Tree Density (trees/ha)	13	**432.7 (39.6)**	**775.0 (66.2)**	**0.00**	7	507.1 (29.2)	396.4 (48.6)	0.20
Tree Stump Density (stumps/ha)	13	**430.8 (40.7)**	**111.5 (20.9)**	**0.00**	7	253.6 (76.1)	107.1 (41.4)	0.07
Sapling Density (saplings/ha)	13	**13.5 (5.4)**	**1094.2 (245.9)**	**0.00**	7	275.0 (158.9)	2271.4 (732.7)	0.08
Sapling Stump Density (stumps/ha)	13	**1673.1 (203.5)**	**475.0 (134.8)**	**0.00**	7	1214.3 (389.0)	800.0 (318.1)	0.38
Tree Basal Area (m^2/ha)	13	**16.9 (2.0)**	**26.9 (2.2)**	**0.00**	7	19.0 (2.8)	18.7 (5.3)	0.94
Tree Stump Basal Area (m^2/ha)	13	**12.3 (1.1)**	**4.5 (1.0)**	**0.00**	7	7.5 (2.6)	3.5 (1.3)	0.38
Sapling Basal Area (m^2/ha)	13	**0.0 (0.0)**	**3.5 (0.8)**	**0.00**	7	0.9 (0.5)	7.2 (2.3)	0.08
Sapling Stump Basal Area (m^2/ha)	13	**4.3 (0.5)**	**1.2 (0.3)**	**0.00**	7	3.1 (1.0)	2.1 (0.8)	0.38
Tree Biomass (Mg/ha)	13	**71.1 (9.5)**	**110.8 (10.8)**	**0.01**	7	86.1 (19.4)	107.8 (40.2)	0.58
Sapling Biomass (Mg/ha)	13	**0.1 (0.0)**	**8.1 (1.8)**	**0.00**	7	2.2 (1.3)	16.5 (5.1)	0.08
Fine Woody Debris Biomass (Mg/ha)[a]	13	1.4 (0.5)	2.3 (0.9)	0.51	7	3.2 (0.8)	1.6 (1.0)	0.08
Coarse Woody Debris Biomass (Mg/ha)[a]	13	8.7 (2.8)	13.4 (3.1)	0.22	7	8.8 (3.1)	1.8 (0.8)	0.08
Downed Woody Debris Biomass (Mg/ha)[a]	13	10.1 (2.7)	15.7 (2.7)	0.19	7	12.0 (3.0)	3.4 (1.5)	0.08
Tree Height (m)	13	17.4 (0.6)	17.2 (0.7)	0.84	7	15.3 (1.0)	15.2 (2.0)	0.94
Crown Base Height (m)[b]	13	8.0 (0.9)	9.1 (0.9)	0.54	5	7.9 (1.2)	8.4 (1.3)	0.44

[a] These are post-wildfire forest floor fuel measures; pre-fire forest floor fuel loads are unknown.
[b] These are post-wildfire measures that exclude one site pair and one untreated site at Warm Lake where crown base height could not be assessed because no needles remained.

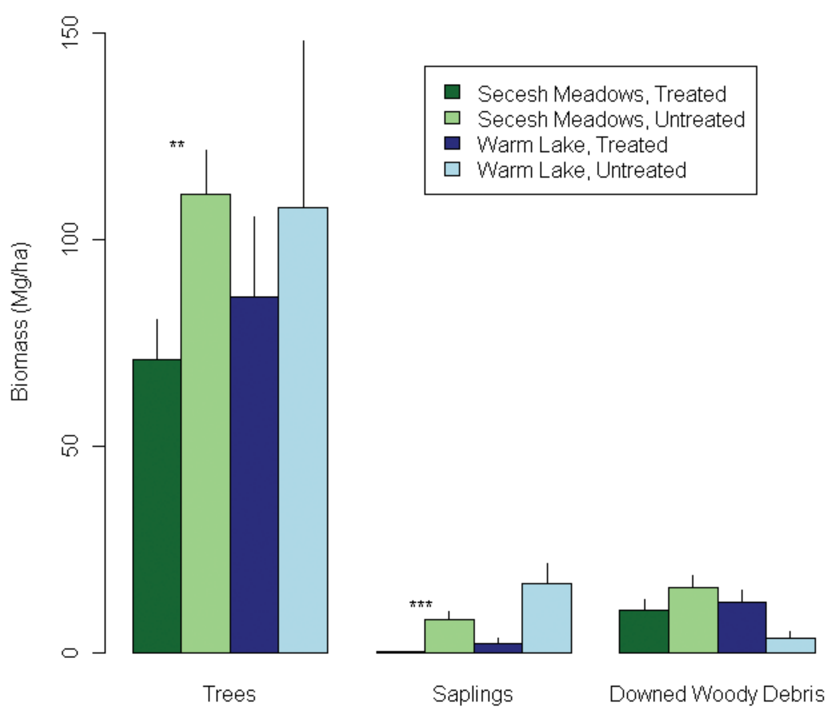


Figure 22. Mean (+SE) one year post-wildfire biomass of trees, saplings, and DWD measured on treated versus untreated site pairs at Secesh Meadows (n = 13) and Warm Lake (n = 7). Significance: **, p-value<0.01; ***, p-value = 0.001.

that *pre-fire* surface fuels did differ significantly between treated and untreated sites. This is because the wildfire consumed virtually all DWD, especially at more severely burned sites. Further obfuscating the surface fuel comparison is the fact that much of the DWD that was measured post-fire fell to the ground soon after the wildfire.

Tree and crown base heights that were measured *post-fire* did not differ significantly between treated and untreated sites at either Secesh Meadows or Warm Lake (table 3). However, *pre-fire* crown base height might have differed more between treated and untreated sites but could not be retrospectively assessed on high severity sites where the needles on the lower branches were consumed by the wildfire. For any tree crown in which all the needles had been consumed all the way to the top, or could not be inferred by the presence of charred cones, crown base height was recorded as "NA" so as not to bias the analysis.

Wildfire effects

Crown base height likely influenced maximum scorch and bole char heights because all three variables exhibited similar patterns (figure 23). At Secesh Meadows, where the fuel piles burned in the wildfire ("WF Pile," figures 23 through 26), both maximum scorch and bole char heights were higher on the treated sites than on the paired untreated sites, although not significantly based on only four site pairs (table 4). Maximum scorch and bole char heights were higher on untreated sites compared to treated sites at Secesh Meadows; in those treatment units where the fuel piles had been burned as prescribed ("Rx Pile," figures 23 through 26); and at the pile and burn, mastication, and underburn treatment units sampled at Warm Lake (figure 23). The only significant difference found was bole char height at Warm Lake (table 4). Note that for any scorched tree crown where no green needles remained, maximum scorch height was recorded as equal to the tree height, but this made for a less meaningful measure that did not differ significantly between treated and untreated sites at either Secesh Meadows or Warm Lake (table 4). Raymond and Peterson (2005) found that scorch volume was a better indicator of tree mortality than scorch height. We agree because areas of crown scorch often occur at variable heights.

The patterns in bole char height (figure 23) that were just described were repeated for tree mortality (figure 24). Indeed, tree mortality and bole char height were significantly correlated ($r = 0.68$, p-value<0.0001) across all sites. Tree mortality was

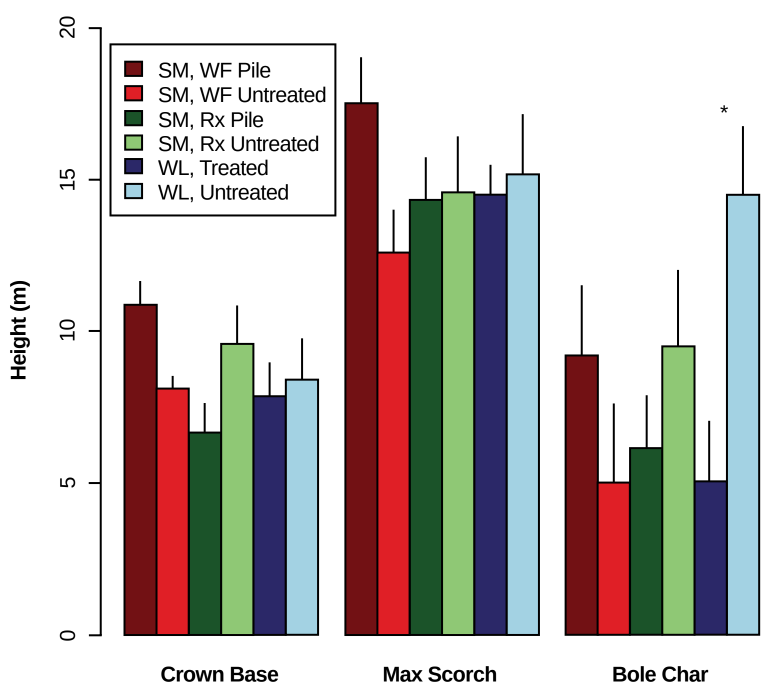


Figure 23. Mean (+SE) one year post-wildfire crown base, maximum scorch, and bole char heights on treated versus untreated site pairs at Secesh Meadows (SM) pile and burn treatment units, where the piles were burned by either the wildfire (WF, n = 4) or before the wildfire as prescribed (Rx, n = 9). Site pairs at Warm Lake (WL, n = 7) are also included. Significance: *, p-value<0.05.

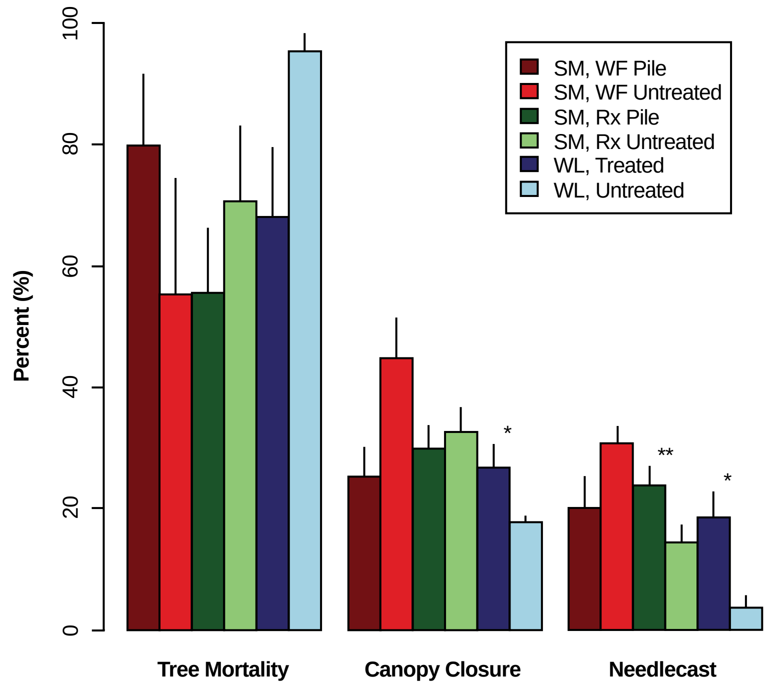


Figure 24. Mean (+SE) one year wildfire-induced tree mortality and post-wildfire canopy closure and needlecast on treated versus untreated site pairs at Secesh Meadows (SM) pile and burn treatment units, where the piles were burned by either the wildfire (WF, n = 4) or before the wildfire as prescribed (Rx, n = 9). Paired sites at Warm Lake (WL, n = 7) are also included. Significance: *, p-value<0.05; ** p-value<0.01.

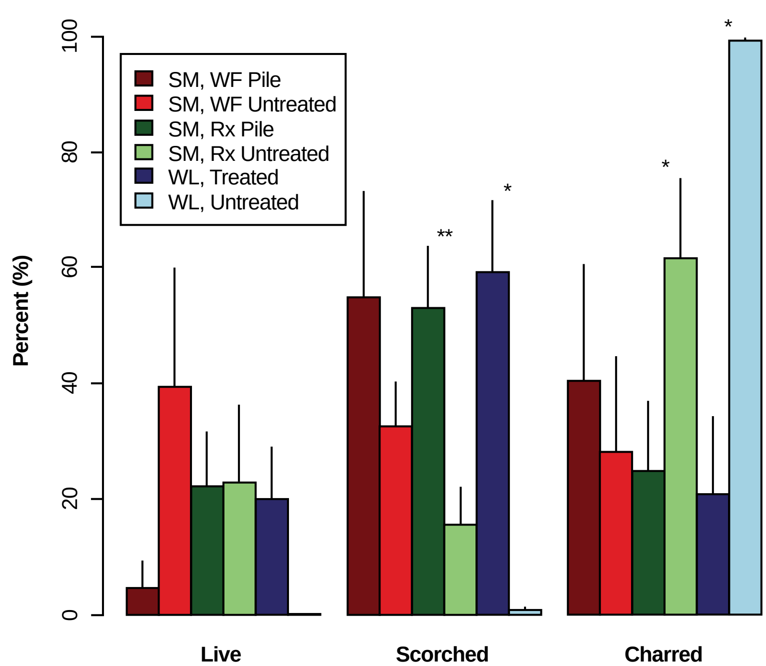


Figure 25. Mean (+SE) one year post-wildfire tree crown condition on treated versus untreated site pairs at (1) pile and burn treatment units at Secesh Meadows (SM), where the piles were burned by either the wildfire (WF, n = 4) or before the wildfire as prescribed (Rx, n = 9), and (2) at Warm Lake (WL, n = 7). Significance: *, p-value<0.05; **, p-value<0.01.

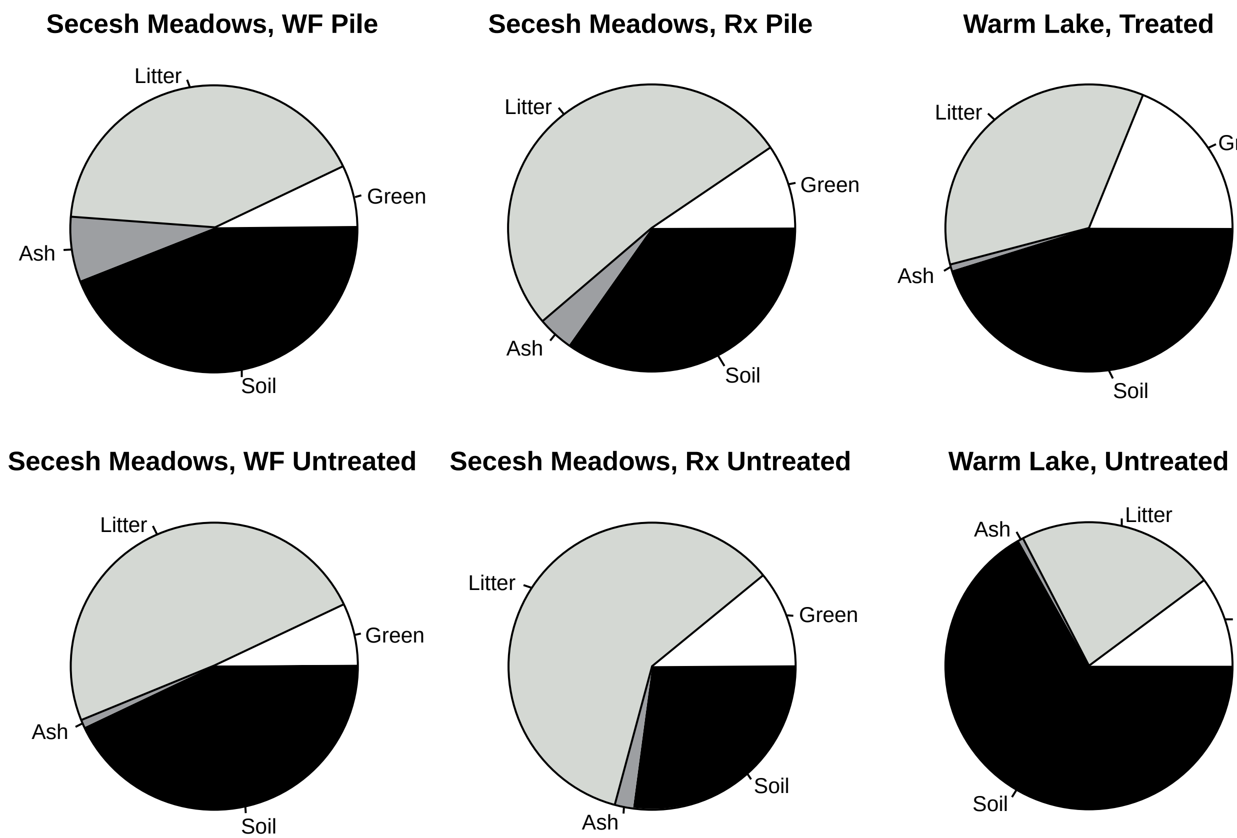


Figure 26. Mean one year post-wildfire ground cover fractions on treated versus untreated site pairs at Secesh Meadows, where the fuel piles from pile and burn treatments were burned by either the wildfire (WF, n = 3) or as prescribed (Rx, n = 10), and at Warm Lake (n = 7), where pile and burn, mastication, and underburn treatments were applied. There were no statistically significant differences ($\alpha = 0.05$).

higher on the treated sites at Secesh Meadows where fuel piles remained for the wildfire to consume (in other words, "WF Piles") than on untreated sites. Whereas, tree mortality was lower on fully treated sites at Secesh Meadows (in other words, "Rx Piles") and Warm Lake than on untreated sites (figure 24). However, Wilcoxon signed rank tests indicated that none of these differences was significant (table 4). Tree canopy closure was lower in treated sites than in untreated sites at Secesh Meadows—significantly in the "WF Pile" units but not significantly in the "Rx Pile" units. In contrast, tree canopy closure at Warm Lake was significantly higher in treated sites than in untreated sites (figure 24, table 4). We assert that differences in canopy closure were more influenced by treatment effects than by wildfire effects at Secesh Meadows, where treatment effects were more pronounced than at Warm Lake (see preceding subsection). This assertion is supported by observed differences in needlecast. In the "WF Pile" units at Secesh Meadows, needlecast was significantly lower in treated sites than in untreated sites. On the other hand, in the "Rx Pile" units at Secesh Meadows and at Warm Lake, needlecast was significantly higher on treated sites than untreated sites (figure 24, table 4). Needlecast can protect the soil against erosion, although the up to 35 percent coverage observed across all of our sites was below the 50 to 70 percent coverage deemed sufficient for erosion control (Pannkuk and Robichaud 2003). Furthermore, needlecast returns some essential nutrients to the soil. If the needles are consumed, as in a high severity fire, then much of the essential nutrients can be volatilized, depleting the site and slowing re-vegetation (Garrison and Moore 1998).

The observed trends in bole char height (figure 23) and tree mortality (figure 24) also were reflected in the percentage of charred tree crowns (figure 25), which significantly differed between treated and untreated sites for the "Rx Pile" treatment units at Secesh Meadows and at Warm Lake (table 4). The percentage of scorched tree crowns also significantly

Table 4. Wildfire effects measured one year post-fire on WUI fuel treatments at Secesh Meadows and Warm Lake. Significant differences ($\alpha = 0.05$) indicated in **boldface**.

	Wildfire (WF) burned fuel piles				Prescribed fire (Rx) burned fuel piles				Warm Lake fuel treatments			
Variable description	Site pairs	Treated mean (SE)	Untreated mean (SE)	Wilcoxon p-value	Site pairs	Treated mean (SE)	Untreated mean (SE)	Wilcoxon p-value	Site pairs	Treated mean (SE)	Untreated mean (SE)	Wilcoxon p-value
Max Scorch Height (m)	4	17.5 (1.5)	12.6 (1.4)	0.13	9	14.3 (1.4)	14.6 (1.8)	0.30	7	14.5 (1.0)	15.2 (2.0)	0.94
Bole Char Height (m)	4	9.2 (2.3)	5.0 (2.6)	0.25	9	6.1 (1.7)	9.5 (2.5)	0.20	7	**5.1 (2.0)**	**14.5 (2.2)**	**0.02**
Tree Mortality (%)	4	79.9 (11.7)	55.4 (19.1)	0.25	9	55.6 (10.8)	70.7 (12.4)	0.11	7	68.2 (11.3)	95.4 (3.1)	0.08
Canopy Closure (%)	4	25.2 (5.0)	44.8 (6.8)	0.13	9	29.9 (3.7)	32.6 (4.2)	0.50	7	**26.9 (3.6)**	**17.7 (1.0)**	**0.02**
Needlecast (%)[a]	4	20.0 (5.3)	30.9 (2.6)	0.13	9	**23.8 (3.1)**	**14.4 (3.1)**	**0.01**	4	18.6 (4.2)	3.6 (1.9)	0.13
Green Crown (%)	4	4.6 (4.6)	39.3 (20.6)	0.18	9	22.1 (9.6)	22.7 (13.7)	0.83	7	19.9 (9.2)	0.0 (0.0)	0.06
Scorched Crown (%)	4	54.8 (18.2)	32.5 (7.7)	0.63	9	**53.1 (10.5)**	**15.5 (6.3)**	**0.00**	7	**59.2 (12.2)**	**0.7 (0.5)**	**0.04**
Charred Crown (%)	4	40.6 (20.0)	28.1 (16.5)	0.88	9	**24.8 (12.1)**	**61.7 (13.8)**	**0.02**	7	**20.8 (13.5)**	**99.3 (0.5)**	**0.04**
Green Overstory (%)	4	17.5 (17.5)	38.0 (22.2)	0.18	9	25.7 (10.8)	22.8 (13.0)	1.00	7	24.0 (11.9)	0.0 (0.0)	0.06
Scorched Overstory (%)	4	46.2 (19.2)	42.0 (16.6)	0.88	9	**50.9 (10.7)**	**20.9 (7.5)**	**0.02**	7	**52.4 (13.8)**	**1.0 (0.7)**	**0.03**
Charred Overstory (%)	4	36.2 (22.0)	20.0 (16.8)	0.79	9	23.4 (12.0)	56.3 (14.1)	0.05	7	**23.6 (14.6)**	**99.0 (0.7)**	**0.03**
Green Understory (%)	4	24.8 (12.1)	26.2 (9.9)	0.88	9	29.8 (7.3)	36.6 (8.2)	0.44	7	39.4 (9.8)	27.3 (9.6)	0.58
Scorched Understory (%)	4	37.5 (13.0)	48.2 (8.5)	1.00	9	37.1 (6.0)	23.7 (4.3)	0.12	7	41.4 (7.8)	18.7 (10.5)	0.08
Charred Understory (%)	4	37.8 (17.8)	25.5 (10.7)	1.00	9	33.1 (8.2)	39.8 (11.6)	0.83	7	19.1 (11.2)	54.0 (11.2)	0.09
Green Surface Vegetation (%)	4	7.1 (3.2)	6.9 (2.3)	1.00	9	9.6 (2.5)	11.0 (3.7)	0.82	7	18.8 (9.2)	10.0 (3.7)	0.58
Litter/DWD (%)	4	41.6 (14.0)	49.4 (7.4)	0.63	9	51.8 (3.9)	59.9 (5.1)	0.13	7	35.4 (6.8)	22.8 (6.3)	0.30
Ash (%)	4	7.3 (4.9)	0.6 (0.6)	0.18	9	4.0 (3.7)	2.0 (1.1)	0.92	7	0.6 (0.4)	0.3 (0.2)	0.79
Mineral Soil/Rock (%)	4	44.0 (13.3)	43.1 (8.3)	0.88	9	34.7 (5.7)	27.1 (6.8)	0.13	7	45.2 (10.8)	66.9 (7.1)	0.16
Char (%)	4	55.9 (9.1)	68.4 (6.4)	0.63	9	62.6 (3.3)	71.5 (6.3)	0.20	7	52.1 (9.5)	59.3 (11.3)	0.47
Litter Depth (mm)	4	2.2 (0.6)	2.5 (0.3)	0.88	9	2.0 (0.3)	2.6 (0.5)	0.21	7	**2.3 (0.3)**	**1.3 (0.1)**	**0.03**
Duff Depth (mm)	4	2.5 (1.7)	1.8 (0.6)	1.00	9	1.7 (0.7)	3.0 (1.0)	0.23	7	1.2 (0.8)	0.1 (0.0)	0.10
Water Infiltration Rate at 1 cm Soil Depth (ml/min)[b]	4	4.5 (1.6)	3.4 (1.2)	0.88	9	8.5 (2.3)	4.8 (1.5)	0.30	6	10.8 (2.5)	9.7 (1.2)	1.00
Water Infiltration Rate at 3 cm Soil Depth (ml/min)[b]	4	2.5 (0.8)	3.6 (1.4)	0.63	9	2.6 (0.5)	3.7 (0.9)	0.57	6	5.7 (1.5)	7.9 (2.0)	0.44
Live Seedling Density (seedlings/ha)	4	2625.0 (1143.4)	125.0 (125.0)	0.18	9	1388.9 (570.0)	944.4 (536.5)	0.60	7	1714.3 (1159.1)	0.0 (0.0)	0.10
Vegetation Plot Cover (%)	4	11.8 (2.6)	10.2 (2.8)	0.79	9	17.6 (4.6)	24.4 (7.7)	0.44	7	26.3 (9.5)	24.0 (7.4)	0.74
Plot Species Richness (count)	4	9.0 (2.6)	5.2 (0.9)	0.18	9	6.1 (1.1)	8.2 (1.2)	0.11	7	6.0 (0.9)	5.9 (1.5)	0.80

[a] Excludes one treated and two untreated sites at Warm Lake where wheat straw was applied for post-fire rehabilitation prior to the 2008 field visits.
[b] Excludes one site pair at Warm Lake where soil conditions were too wet at the time of the 2008 field visits to measure water infiltration reliably.

differed in the same cases, while in no cases did the percentage of green tree crowns significantly differ (figure 25, table 4). Note that the percentage of charred, scorched, and green tree crowns must sum to unity; they are not independent. We estimated crown condition ocularly (Omi and Kalabokidis 1991) for every tree crown within the 1/25-ha tree plot and averaged them; whereas the live, scorched, and charred overstory (and understory) variables (table 4) were a single ocular estimate of the overstory (and understory) canopy condition across the entire field site. The high redundancy between the tree crown and overstory condition results (table 4) indicates that the trees in the tree plots are representative of the field sites and that the quicker overstory (and understory) estimates are sufficient for rapid response assessment.

The fractional ground cover components of green surface vegetation, litter/DWD, ash, and mineral soil/rock were also constrained to sum to unity. Unlike the tree crown condition variables, however, none of the ground cover variables differed significantly between treated and untreated sites (figure 26, table 4). The most dramatic contrast was at Warm Lake where, as has been indicated by the other field measures presented above, wildfire effects were more pronounced than at Secesh Meadows. The char fraction of surface materials did not significantly differ between any paired site groups (table 4).

The more marked contrast in burn severity between treated and untreated sites at Warm Lake may have contributed to one significant difference in a surface fuel variable--litter depth was significantly greater (p-value = 0.03) on treated sites than on untreated sites at Warm Lake (table 4). However, the significance of this result may have been driven by the mastication treatment, where a good share of the wood chips that resulted from the fuel treatment remained on the forest floor following the wildfire. Duff depths did not differ significantly between any paired site groups (table 4). We also compared the soil infiltration rates (Robichaud and others 2008) between treated and untreated sites but found no significant differences in infiltration rate for any paired site groups at either 1 cm or 3 cm depth (table 4).

We estimated there were 1300 more tree seedlings/ha on treated sites than on untreated sites—a significant difference when compared across all 20 site pairs (p-value = 0.03) but not significant within any particular paired site group (table 4). Neither total vegetation cover nor understory plant species richness assessed within our 1/500-ha vegetation subplots differed significantly between treated and untreated sites, either as a whole or for any paired site groups (table 4).

Site recovery

All of the field assessment results presented so far are based on data collected one year after the 2007 wildfires—in August 2008. However, we established half of our field sites during our initial, rapid assessment in September 2007. We compared treated versus untreated sites and used Wilcoxon signed rank tests to assess the significance of the differences in 2007, but the results did not differ appreciably from the 2008 results presented earlier. Only the differences in sapling density, basal area, and biomass; tree stump density; and tree mortality were significant, probably because only nine paired sites were available. With the exception of tree mortality, these significant differences reflected treatment effects rather than wildfire effects.

The real value of the 2007 initial assessment was to allow an assessment of site recovery, even if for only half the number of field sites as were characterized in 2008. Fewer field variables were measured in 2007 than in 2008, further limiting the number of comparisons that could be made. Many of the 2007 measures that were re-measured in 2008 did not change significantly, according to Wilcoxon signed rank tests. Such measures included variables that were related mainly to treatment effects (in other words, tree/sapling/stump density/basal area/biomass) and would not be expected to change after only one year (figure 27). Not surprisingly, percent tree mortality also did not change appreciably. However, a slight yet significant 4.1 percent increase in canopy closure was measured across both treated and untreated sites (table 5). This is counterintuitive, especially given that significantly more needlecast cover (5.7 percent) fell to the ground between the 2007 and 2008 assessments (table 5). Foliar regrowth could explain some of the observed increase in canopy closure, but given the high tree mortality rates observed, the more likely explanation was measurement bias. Different field crew personnel measured canopy closure in 2007 than in 2008, and spherical densiometer measurements are highly variable and, therefore, vulnerable to user bias. Overstory tree canopy condition changed little between 2007 and 2008, whereas understory canopy condition changed greatly from being predominantly charred in

Figure 27. Secesh Meadows pile and burn treated site (left) versus its paired untreated site (right) nearby, one year after the fire. Post-fire revegetation did not show a treatment effect. *Photos: Andrew Hudak*.

2007 to having equal proportions of live, scorched, and charred canopy in 2008 (figure 28). The increase in live understory canopy and concomitant decrease in charred understory canopy was significant, regardless of whether the sites had been treated (table 5).

Changes in ground cover from 2007 to 2008 did not differ between treated and untreated sites (figure 29) but were significant in either case (figure 30), with significant increases in green vegetation cover concomitant with significant decreases in surface litter/DWD and charred material (table 5). Grasses, forbs, and some resprouting shrubs were the main plant types recovering one year after the wildfire. Litter depth decreased 4.1 mm across all 20 re-measured sites (table 5), which may point to physical degradation and incorporation of organic material into the soil after a year of weathering. Although the trends in site recovery from 2007 to 2008 across treated and untreated sites were broadly similar, there was weak but significant evidence that soil water repellency and seedling density recovered more quickly on treated sites than on untreated sites (table 5), which may be indicative of less soil heating on treated sites than on untreated sites.

Discussion

Case Study of 2007 Central Idaho Wildfires

Field assessments

We are confident that the paired site design strengthened our inferences. One could argue that the large number of significant paired test results (n = 50; tables 3 through 5) increases the risk of falsely rejecting the null hypothesis of no difference between treated and untreated sites. However, at the significance level of $\alpha = 0.05$, this would be expected to amount to only $50 \times .05 = 2.5$ false positives, which would be unlikely to change our general conclusion that the fuel treatments did mitigate severe fire effects. From our standpoint, the cost of committing a Type I error by our analytical approach was greatly outweighed by the benefit of being able to point to specific fuel treatment and/or fire effects that may warrant greater attention by fuel and fire managers.

Martinson and Omi (2008) concluded that retrospective studies of fuel treatment effectiveness can be limited by such caveats as a firebreak between treated

Table 5. Comparisons of immediate and one year post-wildfire effects to assess site recovery at Secesh Meadows and Warm Lake. Significant differences (α = 0.05) indicated in **boldface.**

	Treated sites				Untreated sites				All sites			
Variable description	**Site pairs**	**2007 mean (SE)**	**2008 mean (SE)**	**Wilcoxon p-value**	**Site pairs**	**2007 mean (SE)**	**2008 mean (SE)**	**Wilcoxon p-value**	**Site pairs**	**2007 mean (SE)**	**2008 mean (SE)**	**Wilcoxon p-value**
Tree Mortality (%)	11	57.2 (13.6)	62.8 (9.7)	0.68	9	90.4 (3.2)	82.0 (10.1)	0.59	20	72.1 (8.4)	71.4 (7.2)	0.98
Canopy Closure (%)	11	**22.9 (2.5)**	**26.5 (3.7)**	**0.01**	9	**22.0 (2.9)**	**26.7 (4.6)**	**0.04**	20	**22.5 (1.9)**	**26.6 (2.8)**	**0.00**
Needlecast (%)[a]	11	14.0 (4.1)	20.9 (4.2)	0.07	7	7.9 (4.2)	11.5 (4.9)	0.24	18	**11.6 (3.0)**	**17.3 (3.3)**	**0.02**
Green Overstory (%)	11	26.8 (12.8)	20.8 (10.3)	0.79	9	8.9 (8.9)	11.2 (11.0)	0.37	20	18.8 (8.1)	16.5 (7.4)	0.93
Scorched Overstory (%)	11	39.1 (11.8)	46.1 (10.9)	0.57	9	13.6 (5.7)	12.4 (9.6)	0.27	20	27.6 (7.4)	30.9 (8.2)	0.87
Charred Overstory (%)	11	34.1 (12.7)	33.1 (11.8)	1.00	9	77.6 (11.2)	76.3 (13.6)	0.40	20	53.6 (9.7)	52.5 (10.0)	0.61
Green Understory (%)	11	**2.4 (1.6)**	**38.4 (7.8)**	**0.00**	9	**1.1 (1.1)**	**30.4 (9.3)**	**0.01**	20	**1.8 (1.0)**	**34.8 (5.9)**	**0.00**
Scorched Understory (%)	11	24.5 (5.6)	31.5 (5.7)	0.39	9	14.4 (4.0)	27.7 (9.7)	0.12	20	20.0 (3.7)	29.8 (5.2)	0.11
Charred Understory (%)	11	**73.1 (6.9)**	**30.1 (9.8)**	**0.01**	9	**84.4 (4.4)**	**41.9 (11.6)**	**0.01**	20	**78.2 (4.4)**	**35.4 (7.4)**	**0.00**
Green Surface Vegetation (%)	11	**0.8 (0.5)**	**16.3 (6.0)**	**0.00**	9	**0.8 (0.7)**	**13.5 (4.0)**	**0.00**	20	**0.8 (0.4)**	**15.0 (3.7)**	**0.00**
Litter/DWD (%)	11	**69.5 (7.0)**	**42.6 (4.8)**	**0.00**	9	**55.7 (9.4)**	**38.4 (8.3)**	**0.02**	20	**63.3 (5.8)**	**40.7 (4.5)**	**0.00**
Ash (%)	11	2.0 (1.4)	4.1 (3.0)	0.44	9	2.0 (0.7)	1.5 (1.1)	0.57	20	2.0 (0.8)	3.0 (1.7)	0.98
Mineral Soil/Rock (%)	11	27.7 (6.8)	37.0 (6.7)	0.28	9	41.5 (9.8)	46.6 (10.6)	0.30	20	33.9 (5.8)	41.3 (5.9)	0.10
Char (%)	11	**85.8 (5.5)**	**55.8 (6.0)**	**0.00**	9	**91.3 (3.4)**	**61.0 (7.9)**	**0.00**	20	**88.3 (3.4)**	**58.1 (4.8)**	**0.00**
Litter Depth (mm)	11	**6.5 (1.3)**	**2.3 (0.3)**	**0.00**	9	**6.2 (1.3)**	**2.2 (0.5)**	**0.01**	20	**6.3 (0.9)**	**2.2 (0.3)**	**0.00**
Duff Depth (mm)	11	1.9 (1.0)	1.5 (0.7)	0.64	9	1.7 (1.1)	1.8 (1.1)	0.10	20	1.8 (0.7)	1.6 (0.6)	0.97
Water Infiltration Rate at 1 cm Soil Depth (ml/min)[b]	9	**0.4 (0.1)**	**9.8 (2.6)**	**0.00**	6	4.0 (2.8)	7.6 (1.9)	0.22	15	**1.9 (1.2)**	**8.8 (1.7)**	**0.00**
Water Infiltration Rate at 3 cm Soil Depth (ml/min)[b]	10	**1.3 (0.5)**	**4.1 (1.1)**	**0.02**	8	**2.3 (1.0)**	**6.2 (1.4)**	**0.04**	18	**1.8 (0.5)**	**5.0 (0.9)**	**0.00**
Live Seedling Density (seedlings/ha)	11	**0.0 (0.0)**	**1545.5 (739.9)**	**0.02**	9	55.6 (55.6)	611.1 (551.4)	0.37	20	**25.0 (25.0)**	**1125.0 (476.9)**	**0.01**

[a] Excludes two untreated sites at Warm Lake where wheat straw was applied for post-fire rehabilitation between the 2007 and 2008 field visits.
[b] Excludes field sites at Warm Lake where soil conditions were too wet at the time of either the 2007 or 2008 field visits to measure water infiltration reliably.

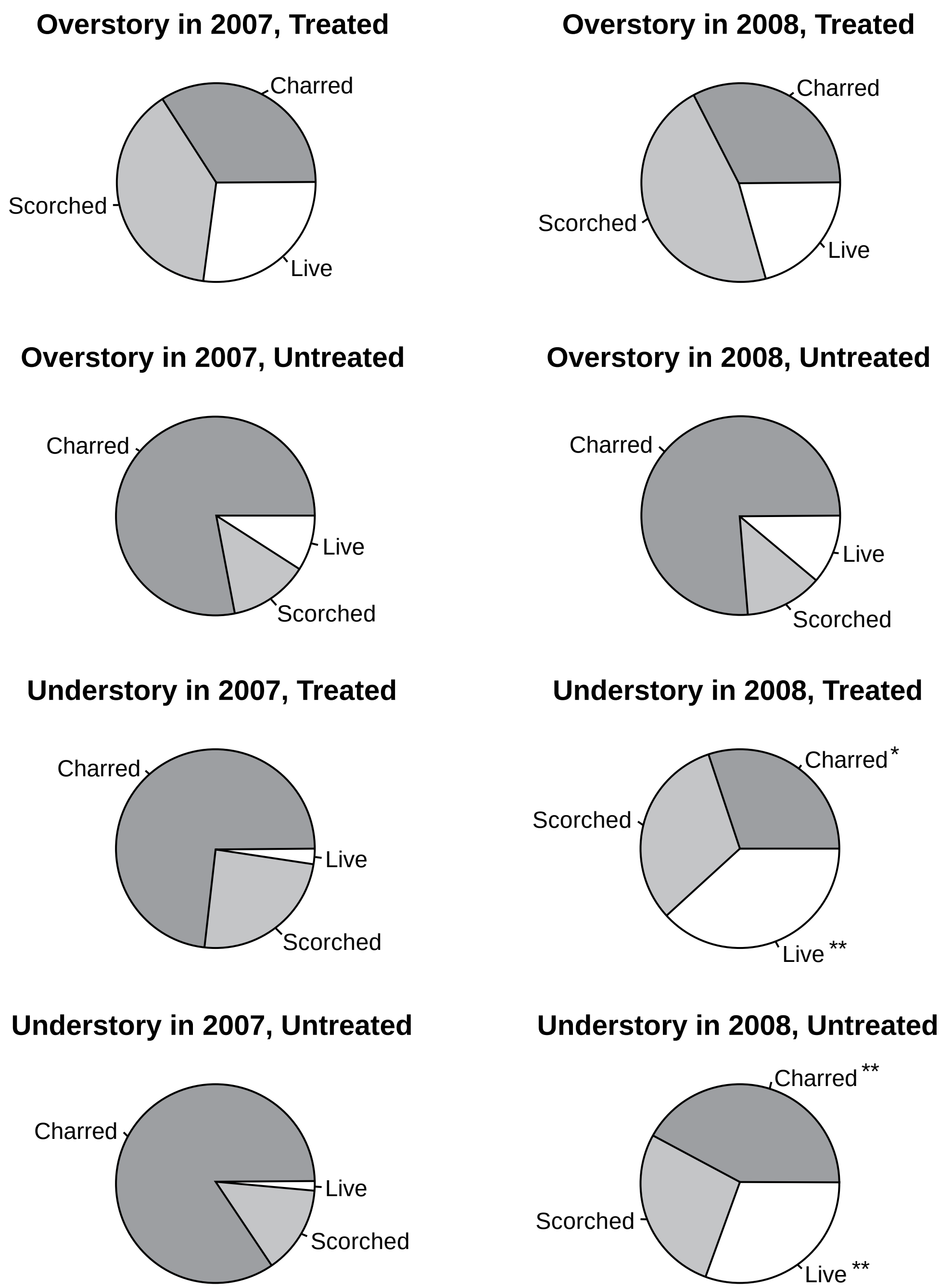


Figure 28. Mean post-wildfire overstory and understory canopy condition on treated (n = 11) and untreated (n = 9) sites, immediately post-fire (2007) versus one year post-fire (2008). Significant changes from 2007 to 2008 are indicated on the right: *, p-value<0.05; **, p-value<0.01.

Figure 29. Secesh Meadows pile and burn treated site (left) versus its paired untreated site (right) nearby, one year after the fire. Ground cover fractions changed but similarly between treated and untreated sites. *Photos: Andrew Hudak.*

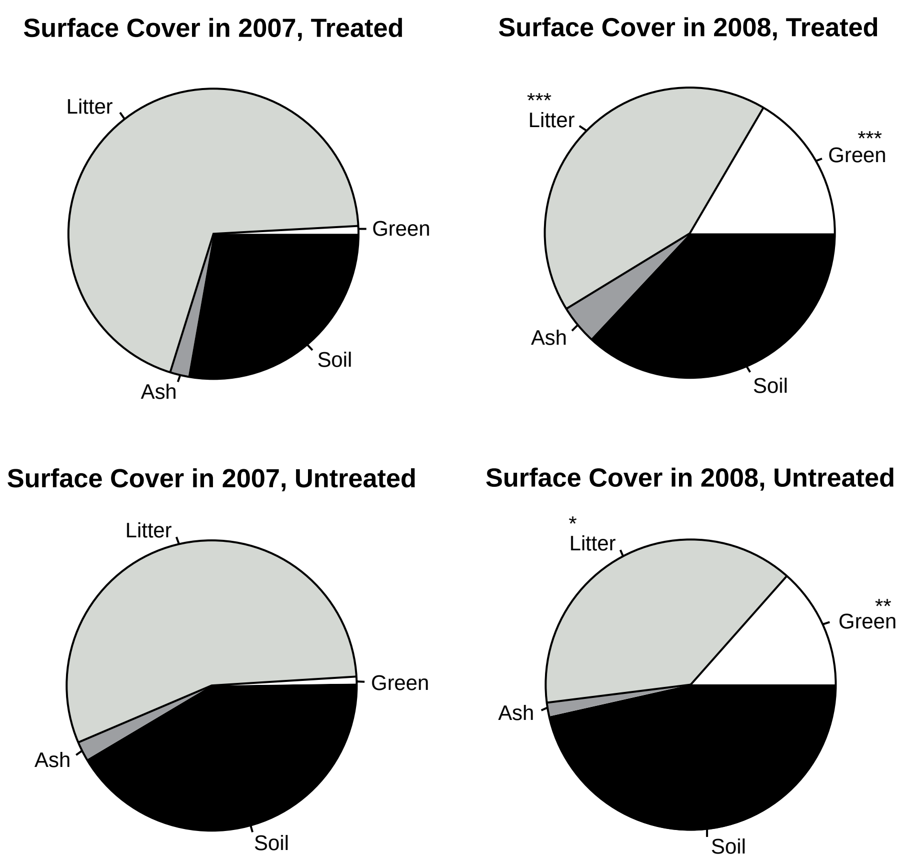


Figure 30. Mean post-wildfire ground cover fractions on treated (n = 11) and untreated (n = 9) sites, immediately post-fire (2007) versus one year post-fire (2008). Significant changes are indicated on the right: *, p-value<0.05, **, p-value<0.01; ***, p-value = 0.001.

and untreated sites (for example, a railroad track in their study) or pseudo-replicated samples that are caused by having multiple sites within a single treatment unit. We sampled two treatment units at Secesh Meadows twice, two treatment units at Warm Lake twice, and another at Warm Lake thrice (figure 12), but these were larger treatment units in which different site pairs were separated by either ample distance perpendicular to the direction of the advancing crown fire or by roads/powerlines that acted as firebreaks between different paired sites. Strom and Fulé (2007) also found that a paired sampling approach was powerful for focusing attention on the fuels component of the fire behavior triangle as fire weather and topography at a given pair of sites were otherwise very similar.

Fuel treatment effects on tree density, basal area, and biomass were more pronounced at Secesh Meadows than at Warm Lake (table 3), perhaps in part because the Secesh Meadows treatments were more recent (2006) compared to the Warm Lake treatments sampled (2000 to 2005). The nine treated sites at Secesh Meadows where the piles had been prescribed burned prior to the wildfire burned less severely than their paired untreated sites. Fuel piles that burned in the wildfire contributed to more severe fire effects than at untreated sites (figure 31)—a result noted in other assessments (Harbert and others 2007, Murphy and others 2007). At Warm Lake, all seven treated sites burned less severely than their paired untreated sites regardless of treatment type (figures 17-19), so all seven paired sites at Warm Lake were grouped together for the statistical comparison (table 4).

Our vegetation recovery results after a single year indicated little differential ability in the sites to recover from wildfire after one year, based on how and whether fuels had been treated (table 5). However, one year is not an adequate period of time to consider tree and understory plant recovery. Because vegetation recovery following fuel treatments can be rapid, most fuel treatments need repeated maintenance to remain effective (Graham and others 1999, 2004). Longer-term monitoring of fuel treatments is needed to understand the duration of treatment effectiveness in different ecosystems, whether or not a wildfire actually burns through.

Satellite assessments

The dNBR and potentially other burn severity indices may be useful for assessing fuel treatment effectiveness consistently across large areas. Wimberly and others (2009) found that thinning treatments that used prescribed fires to reduce fuel loads were more effective than thinning alone in reducing subsequent burn severity for three western wildfires. The data used by Wimberly and others (2009) were dNBR maps developed by the national-scale Monitoring Trends in Burn Severity (MTBS) project from 1984 to present Landsat imagery, and geospatial data layers from the LANDFIRE project to control for pre-fire variability in canopy cover, fuels, and topography, which can confound efforts to assess treatment effects from remotely sensed data. The MTBS project uses one year post-fire imagery because it provides a more accurate indicator of burn severity, defined as the ecological impact of the fire (Lentile and others 2006), than immediate post-fire imagery.

We found that one year post-fire dNBR, the most common remotely sensed indicator of burn severity, significantly differed between our treated and untreated sites. This and other indices derived from satellite images are sensitive to burn severity, as characterized by fire effects on vegetation and soils (Key and Benson 2005, Hudak and others 2007). Considering all 20 paired sites, one year post-fire dNBR correlated more strongly to bole char height ($r = 0.84$, p-value<0.0001), percent charred overstory canopy ($r = 0.85$, p-value<0.0001), percent charred tree crowns ($r = 0.82$, p-value<0.0001), and tree mortality ($r = 0.66$, p-value<0.0001) than to any other field measures. These field measures were especially high at the untreated sites at Warm Lake, again indicating that severe wildfire effects were more pronounced there than at Secesh Meadows (table 4). This result supports the finding of Hudak and others (2007) that vegetation canopy reflectance has more influence than soil reflectance on the dNBR signal integrated over a 30- by 30-m pixel area.

The immediate post-fire BARC map data analysis of treated versus untreated lands also showed that the fuel treatments significantly reduced high severity fires (figure 13). This agreed with similar findings by Harbert and others (2007) on three large 2007 fires in central Oregon. However, simple BARC map analyses such as these do not control for complicating factors such as topography and local fire weather as effectively as a paired site sampling strategy. For example, local USFS fire and fuel managers were doubtful that the weather data recorded at the Flat Creek portable RAWS (figure 10), situated between the northernmost treatment units at Secesh Meadows (figure 5), would well represent the

Figure 31. ***Left:*** Backing fire igniting unburned fuel piles at Secesh Meadows, which contributed to more severe fire effects. *Photo: Roger Staats*. ***Right:*** Contrasts in crown condition, tree mortality, and understory re-vegetation at Secesh Meadows one year post-fire, looking north up the hill toward the Pioneer Cemetery. *Photo: Andrew Hudak*. Area upslope to the left had residual fuel piles burned by the wildfire (WF Piles), while most of the piles in the area in the foreground and to the right were burned prior to the wildfire as prescribed (Rx Piles). More severe fire effects persisted one year later on the left than on the right as evidenced by greater crown scorching, higher tree mortality, and less re-vegetation.

entire WUI because of the way the wind swirled within the constricted valley during the fire. Similarly, eddies produced by the local topography were undoubtedly a factor at Warm Lake. On the other hand, the RAWS variables that we considered explained a large proportion of the variance in area burned per day in both WUI areas. The significant explanatory power of most of the spatial autoregressive models based on RAWS records that were collected within 25 km of the local WUI areas demonstrated that local weather is an important determinant of fire activity (table 2).

Fuel treatment effectiveness

The combination of mechanical treatment followed by prescribed surface fire was the most effective type of fuel treatment for mitigating severe fire effects (figures 12 and 13). While this result confirms many other findings (Cram and others 2006, Martinson and others 2003, Omi and others 2006, Prichard and others 2010, Raymond and Peterson 2005, Ritchie and others 2007, Skinner and others 2004, van Wagner 1968, Weatherspoon and Skinner 1995), it was probably influenced by several local variables that merit discussion. First, the effectiveness of the underburn treatments at Warm Lake depended on time since treatment. The areal percentage of high severity hectares was 16 percent in the 2006 Kline Mountain underburn unit but ranged from 18 to 34 percent in the three other underburns applied from 1996 to 1998. Even more dramatically, the areal percentage of low severity hectares was 40 percent in the Kline Mountain underburn but only 11 to 24 percent in the older underburn units.

Fire direction was another factor. The treatment units west of Warm Lake were most severely tested by the advancing wildfire, namely the Warm Lake Highway mastication treatment (tested by the North Fork fire that approached from the west); the Warm Lake South pile and burn treatment (tested by the Monumental fire that approached from the south); and the Kline Mountain underburn treatment in between, where steeper slopes and variable aspects appeared to interact with the converging wildfires. These treatments all worked as designed to slow the momentum of the advancing crown fire. The observed west to east gradient of high to low burn severity that was captured by the BARC map across the treatment units (figure 12) was the immediate post-fire expression of a similar gradient in fire intensity, as crown fires dropped down to surface fires as they moved through the treatment units. The many (more than 100) bucket drops from helicopters into the treatment units in close proximity to structures was also a factor in the high percentage of unburned hectares in pile and burn treatment units at Warm Lake and in Firewise-treated private lands at Secesh Meadows.

Whether or not these fuel treatments would have protected homes without the added fire suppression measures may not be as important as determining how fuel treatments can be optimally located and distributed across the landscape to allow fire suppression crews to gain control more effectively and safely. Maps of fuel treatment units should be made available to incident commanders to use as potential fire breaks, similar to roads and streams. Other assessments have promoted planning fire suppression efforts around fuel treatment units (Fites and others 2007, Graham and others 2009, Harbert and others 2007, Murphy and others 2007, Rogers and others 2008).

Are Expensive WUI Fuel Treatments Worthwhile?

We are rapidly learning about fuel treatments, though we need better monitoring and communication of successes and lessons learned. The interagency Joint Fire Science Program has funded multiple studies on fuel treatments, including some user guides (Graham and others 1999, 2004) and recent projects that are focused on the life cycle of fuel treatments. Fuels synthesis tools are also useful. While there is general agreement that removing and reducing fuels reduces fire intensity, not all agree that fuel treatments are effective, and many assessments of treatment effectiveness are qualitative or based on simulation models with little empirical data.

Rhodes and Baker (2008) found that fuel treatments only have a mean probability of 2.0 to 7.9 percent of being encountered by moderate or high severity fire within 20 years following treatment, and they argue that the millions of dollars spent on fuel treatments is, therefore, not justified. Rhodes and Baker (2008) argue that, even if those areas benefited from reduced burn severity, they are too small to counterbalance the adverse effects of fuel treatments on watersheds.

However, Rhodes and Baker (2008) oversimplified their analysis by assuming that ignitions, and fuel treatments, are randomly distributed across the 11 western states. Neither assumption is correct, as 50 percent of fuel treatments are mandated to occur in the WUI (but see Schoennagel and others 2009 and discussion below). Ignitions, all of which except lightning are human-caused, are more probable where human populations are denser, as in the WUI. Rhodes and Baker's (2008) findings are instructive for regional or national level planning but are less helpful to local fuel managers who design and implement fuel treatments. Simulation models provide valuable planning tools for decision makers. For example, Ager and others (2010) found that fuel reduction treatments on just 10 percent of the landscape resulted in a 70 percent reduction of large tree mortality to help preserve highly valued, old forest stands. Many fuel treatments will not be challenged by fire, and so they need to be well designed for other vegetation management objectives that are appropriate to the biophysical and socio-economic setting.

Schoennagel and others (2009) looked at the 44,000 fuel treatments implemented across the western United States and found that only 3 percent of the treated areas were within the WUI and only 8 percent within 2.5 km of the WUI, which falls far short of the 50 percent mandated by the National Fire Plan. Only 17 percent of the area within 2.5 km of the WUI is under Federal ownership. Therefore, Schoennagel and others (2009) concluded that the focus for treating fuels needs to shift from public to private lands.

The costs and benefits of fuel treatments—particularly in the WUI—are likely to continue to be debated. Our primary goal in this report is not to add to this debate but to look at cases where fuel treatments were tested by wildfire and ask, "Were these treatments effective?" Our qualified answer is "Yes." We now summarize the important implications of our findings.

Fuel Treatments in Forests: Implications and Conclusions

Fuel treatments mitigate fire effects. Of the case studies we reviewed that had been tested in wildfires, only Weatherspoon and Skinner's (1995) showed that fuel treatments were not effective. However, in their study, the untreated stands were simply more fire resistant than treated stands.

We can draw some broad conclusions from our case study and the diverse studies we reviewed. These conclusions apply most directly to the dry forests of the western United States where most of the studies have been conducted.

First, fire effects on the overstory trees were most effectively mitigated by treatments that addressed both surface and crown fuels through combination treatments such as thinning followed by a prescribed burn or by removing slash after thinning (Cram and others 2006, Martinson and others 2003, Omi and others 2006, Prichard and others 2010, Raymond and Peterson 2005, Ritchie and others 2007, Skinner and others 2004, van Wagner 1968, Weatherspoon and Skinner 1995). Thinning alone can certainly alter the amount and arrangement of crown fuels (Omi and others 2006, Pollet and Omi 2002), but the presence of abundant activity fuels or slash (Skinner and others 2004), grasses (Weatherspoon and Skinner 1995), and shrubs (van Wagner 1968) contribute to tree canopy damage or mortality when those fuels burn. Field observations and fire behavior modeling studies have demonstrated that increased surface and ladder fuels increase crown fire risk (Vaillant and others 2009). From our own case study, we found that the most effective treatments combined forest thinning and reduction of surface fuels (figures 12 and 13).

Second, prescribed burn treatments varied in their effectiveness. Pollet and Omi (2002) found that prescribed burning was the least successful of alternative fuel treatments in mitigating fire severity, and our BARC map data comparison of treatment types at Warm Lake confirm this finding (figure 13). However, Omi and others (2006) found that prescribed fire treatments that removed small-diameter trees in addition to surface fuel consumption resulted in a canopy structure similar to thinning from below and that this proved highly successful compared with thinning treatments that did not affect surface fuels. Given the variability in prescribed fires, repeated entry may be necessary to achieve desired fuel amount and composition (Martinson and others 2003). It is difficult to kill most medium-sized trees and many small trees by fire alone. Multiple rounds of prescribed fire are more effective in reducing burn severity of subsequent wildfires than single entry treatments (Finney and others 2005, Harbert and others 2007).

Third, treatments become less effective with time since treatment. This was the case with the 1996 to 2006 underburn treatments at Warm Lake (figure 12). Fuels accumulate because vegetation production outweighs decomposition; fires maintain a balance between fuel accumulation and consumption (van Wagtendonk and Moore 2010). The rate of forest fuels accumulation varies as a function of forest type, climate, and disturbance regime, particularly fire disturbance (Graham and others 1999). Keifer and others (2006) found that fuels in ponderosa pine stands in California accumulated to 84 to 88 percent of pre-fire levels 10 years after burning and 150 to 180 percent 31 years after burning. Skinner and others (2004) found that treatments conducted two to four years prior to wildfires showed the least tree mortality. Finney and others (2005) found treatments completed within four years before the fire reduced severity most consistently and more significantly than those completed nine years prior to wildfires. Omi and others (2006) took this a step further, stating that treatments that removed slash, which significantly reduced surface fuel loading, were effective for 10 years, while thinning-only treatments were only found to be effective if they were 1 year old or less. Although it is clear that fuel treatment effectiveness declines with time, more studies are needed. It is likely that longevity of treatments is site-specific and constrained by climate. Re-treatment or other maintenance of treated areas will be necessary for continued effectiveness. Landscape-scale prescribed burning and maintenance of treated areas must be part of long-term vegetation and fuel treatment strategies, and the need for maintenance treatments will continue to escalate as more lands are restored (Harbert and others 2007). Firewise communities (www.firewise.org) need to be part of developing fire smart landscapes—landscapes that are resilient to the effects of future fires. Collaborative efforts will be key, as will treatment of public lands immediately adjacent to private lands like we observed at Secesh Meadows (figure 11).

Fourth, we know little about the importance of spatial arrangement and spatial heterogeneity of fuels and fuel treatments. Finney and others (2005) and Weatherspoon and Skinner (1995) found that larger treatments generally mitigated fire effects more effectively than smaller treatments. Where post-treatment fuels were homogeneous, burn severity decreased with distance from treatment edge. Weatherspoon and Skinner (1995) found that in broadcast burn treatments, burn severity decreased with distance from treatment edge, while machine-piled and -burned treatments did not show this effect but rather a "spotty" burn pattern. The relatively uniform treatment tested by Moghaddas and Craggs (2007) showed a 65 percent decrease in crown scorch within 60 m from the edge of the treatment, although topography likely influenced this.

Omi and Kalabokidis (1991) did not find this edge effect in the relatively uniform fuels of regenerating lodgepole pine clearcuts. In our case study at Warm Lake, distance from the edge was an observable factor in the mechanical treatment units with homogeneous fuel conditions but not in the underburn units with more heterogeneous fuel conditions (figure 12).

Fifth, placement of treatments with respect to topography, wind, and existing fuels can influence treatment effectiveness. Treatments need to be carefully prioritized in a landscape-scale context (Rogers and others 2007). Both less intense, landscape-scale treatments (for example, prescribed burning and wildland fire use for resource benefit) and more intense treatments (for example, thinning combined with piling, masticating, or otherwise treating surface fuels) will be part of strategic choices. Tools such as ARCFUELS (http://www.fs.fed.us/wwetac/arcfuels/index.html, Ager and others 2006, Finney and others 2007) can help managers strategically place future treatments with respect to topography and existing vegetation, thereby minimizing the effects of extreme fire weather on fire behavior. Although placement was not a variable included in their discussion, the ridgetop placement of the thinning treatment evaluated by Moghaddas and Craggs (2007) should be viewed as an example of maximizing the effect of a fuel treatment with regard to topographic influences on burning conditions. This agrees with earlier findings by van Wagner (1968), who documented slope effects that contributed to increased fire behavior and severity; and Weatherspoon and Skinner (1995), who noted that aspect was a significant variable that contributed to fire severity. We also observed slope effects that contributed to more severe fire effects at Secesh Meadows and Warm Lake (figures 16 and 19). Additional studies are needed to prioritize the locations where treatments can be most effective and to test placement empirically.

Sixth, there is no magic formula. While thinning from below is a common treatment and thresholds in tree density, crown base height, crown bulk density, tree spacing, and other fuel composition descriptors exist for a given stand, there is no general prescription that will work in all or even most stands. The great variety of stand conditions, topography, wildfire burning conditions and other variables make it impossible to identify target thresholds for fuel treatment effectiveness. Targets have been suggested, such as the crown bulk density threshold isolated by Cram and others (2006) of 0.047 kg/m^3 or the threshold identified by Agee (1996) of 0.10 kg/m^3 for torching and crown fire propagation. However, the conditions that will make a given stand resilient to wildfire depend on many factors.

Seventh, fuel treatments are not designed to stop fires but rather to modify fire behavior. Firefighters can often use treated areas in effective fire suppression to limit fire spread. Fuel treatments can assist fire managers in burn operation strategies when treatments reduce fire intensity (Fites and others 2007). Fuel treatments are designed to be used together with fire suppression and Firewise principles to effectively reduce the likelihood that wildfires will burn homes. Fuel treatments can be designed to change fire behavior from crown fire to surface fire, thereby reducing spotting distances and convective and radiant heat (Murphy and others 2007). Short of removing all fuel, we cannot design fuel treatments that will not burn, especially under extreme fire conditions (Pollett and Omi 2002). However, we can readily design and implement sustainable, visually appealing fuel treatments that will be resilient when they burn and that will help fire managers charged with protecting key resource values while also providing for ecological restoration and health. It is worth repeating that fuel treatments need to be more intensive (more surface fuels removed and wider crown spacing) on slopes to achieve the same effect as on flat ground.

Eighth, whether fuel treatments exacerbate undesirable fire behavior has been a point of contention. Within treated areas, the lower tree canopy cover and higher light and nutrient availability on the forest floor may lead to increased grass or other conditions that favor rapidly spreading fires of high intensity (Agee 1996, Covington and others 1997). Increased wind velocity can decrease fuel moisture and increase flame lengths and rates of spread, thereby increasing the magnitude of post-fire effects (Agee 1996, Covington and others 1997, Weatherspoon and Skinner 1995). On the other hand, Faiella and Baily (2007) examined fuel moisture in thin and burn and burn-only ponderosa pine restoration treatments in Arizona and compared them to untreated sites. They found no significant differences in the moisture content of fuels in the 0 to 6, 6 to 25, and 25 to 100 mm size classes and concluded that concerns of decreased fuel moisture in treated ponderosa pine forests appear to be unwarranted. Though it is certainly possible for fuel treatments to increase fine fuel temperature and create a micro-climate that favors increased winds and lower relative humidity. Future research should explore these factors as well as the scale at which they affect fire behavior. In addition, managers must recognize that fuel treatments are not intended as one time actions and an increase in fine fuels can be planned for and dealt with using maintenance treatments. For example, a high loading of grasses following prescribed fire can be controlled by periodic grazing or maintenance burns.

Ninth, fuels are just one leg of the fire behavior triangle. Weather and topography affect fire behavior; in some cases they render the most robust fuel treatments useless (Bessie and Johnson 1995). Firefighting tactics and fuels management treatments are all based upon the recognition that fuels are the one aspect of the fire environment that humans can most readily alter. Researchers acknowledge that the interaction of weather and topography is difficult to adequately quantify in empirical studies. More recent studies have attempted to control those variables through statistical tests and sample design (Cram and others 2006, Martinson and Omi 2006, Omi and others 2006, Pollet and Omi 2002, Raymond and Peterson 2005). Future treatments will likely be more effective if topography and potential micro-climate effects are considered in treatment prescription and location (Agee 1996, Weatherspoon and Skinner 1995). Recent efforts to statistically test the variables should be applauded, but more definitive controls are needed to apply fuel treatment parameters with confidence. Whether treatments are effective or not will vary with fire weather conditions. We expect fuel treatments to be less effective in hot, dry, and windy conditions. In other words, fuel treatments may successfully dampen the behavior of fires that are fuel-driven, but wind-driven fires carry tremendous momentum and are much more difficult to control.

Tenth, there is much to be learned in fuel treatment design and implementation from the many years of experience gained by forest and rangeland managers who manage vegetation for other objectives. Vegetation treatments, if thoughtfully designed, can often accomplish multiple objectives. For instance, promoting the resilience of vegetation to future disturbance, especially in the face of climate change, may become increasingly important. Much of the vegetation surrounding the WUI is valued for recreation, aesthetics, and songbird habitat, and there are often ways to adapt treatment design to enhance those ecosystem services (Graham and others 1999, 2004).

Grazing as a Fuel Treatment

The removal of biomass during grazing (particularly heavy grazing) reduces fine fuels and decreases risk of fire occurrence and spread. Livestock grazing has the ability to alter the vegetation structure and composition of all fuel classes in the Douglas-fir/ninebark habitat type (Zimmerman and Neuenschwander 1984). Those modifications result in a forest that is less likely to burn and is less conducive to vertical fire spread. Disturbances that affect the heterogeneity of fuel loads (for example, grazing) affect the total area burned and the complexity of the perimeter in a fire. Fire behavior modeling in tallgrass prairie indicates that fires are smaller and have more complex shapes in heterogeneous landscapes (Kerby and others 2006). Grazing and browsing can be targeted to manage long-term vegetation structure and composition (Davison 1996, Nader and others 2007, Taylor 2006). Such targeted grazing often requires intensive management of livestock, including careful selection of appropriate animal species, animal condition, season, duration, and intensity of grazing, and does not often result in optimum livestock production.

Cheatgrass is a flammable annual grass that has significantly altered the fire regimes in the sagebrush steppe. It would be inadequate to discuss fire patterns in sagebrush ecosystems without including a discussion of cheatgrass and its interactions with grazing. Cheatgrass is an invasive annual grass that was introduced to North America in the Eighteenth century, likely as a contaminant in grain seed (Mack 1981). Cheatgrass forms a fine-textured, continuous fuel bed that is highly flammable when dry and that can support rapidly spreading fires (Klemmendson and Smith 1964, Link and others 2006). Cheatgrass typically cures by early June, expanding the fire season by nearly two months in sagebrush communities (Klemmendson and Smith 1964). Cheatgrass dominance has changed the fire regime in many areas of the sagebrush steppe enough to significantly alter succession, create a more homogeneous landscape, and decrease species diversity (Peters and Bunting 1994). Historically, grazing and agriculture were significant disturbances within the Snake River Plain; however, fire is now the primary disturbance that allows for and perpetuates the invasion of cheatgrass (Peters and Bunting 1994). Fire frequency changed from 35 to 110 years to 3 to 5 years due to cheatgrass invasion, and because of the continuity it provides, fires burned more uniformly, leaving less unburned vegetation (Whisenant 1990). However, cheatgrass does not do as well in black sage (*Artemisia nova*) communities (Miller and Eddleman 2001), nor does it readily dominate in more mesic and cool areas that are typified by mountain big sagebrush and low sagebrush (*Artemisia arbuscula*) (above 1500 m in the northern portion of the sagebrush biome and above 1600 m in the southern portion).

While the fire frequency has changed greatly in areas that are invaded by annual grasses such as cheatgrass, conifer expansions in the mesic portion of the sagebrush steppe have decreased fire frequency (Miller and Rose 1999). Post-settlement western juniper (*Juniperous occidentalis*) expansion is associated with an increase in domestic livestock, a reduction in fire frequency, and an increase in precipitation (Miller and Rose 1999). Increased sagebrush cover, resulting from intensive grazing and other vegetation modifications that are designed to increase forage for domestic livestock, may provide safe sites for juniper establishment and sapling growth. Interactions among grazing regimes, invasive plants, and a changing climate contribute to the complexity of fuels management in rangelands. The legacy of the land and potential future trajectories in vegetation composition must be carefully considered when deciding whether or not to apply fuel treatments in rangelands.

Long-Term Ecological Effects of Fuel Treatments

Fuel treatments can have long-term ecological effects, whether or not the treated area burns in wildfires. Choromanska and DeLuca (2001) showed that prescribed fire can minimize carbon and nitrogen losses from subsequent wildfires, improve soil microbe resistance, increase soil organic carbon, and increase basal respiration. Wagle and Eakle (1979) concluded that after wildfire, understory plants recovered more rapidly in areas treated with prescribed fire prior to the wildfire. Cram and others (2006) demonstrated that when wildfires burned in areas with prior treatments, there was less bare soil, more litter, and improved herbaceous plant recovery.

Fulé and others (2007) modeled different post-fire successional trajectories for treated and untreated areas in Arizona. Their results suggested that thinning and prescribed burning encouraged ponderosa pine, while untreated areas that burned in the same wildfires became dominated by manzanita, gambel oak, and New Mexican locust with few areas dominated by ponderosa pine.

Invasive plant species can increase in abundance following fuel treatments or wildfires, especially if disturbances are severe (Brooks and others 2004, Hunter and others 2006). Omi and others (2006) found higher cover of non-native plant species in areas that burned more severely. Where fuel treatments mitigated fire effects on canopy and ground fuels, native species richness was higher and species were more abundant than in areas that wildfires burned with high severity. However, post-fire invasive species are likely to be most abundant where they were abundant pre-fire, such as along roadsides or in other disturbed areas.

Limitations and Future Needs

These case studies cannot represent all types of fuel treatments or ecosystems, making it difficult to draw generalizations. Most case studies are based on thinning and prescribed burning in dry pine forests. There are few such studies in woodlands, shrublands, and grasslands (though the SageSTEP project [http://www.sagestep.org/] will soon provide useful information for sagebrush and juniper woodlands). Despite their limitations, simulation modeling and case studies will continue to be used to evaluate fuel treatments. More landscape-level studies that exploit remotely sensed data are also needed (for example, Dailey and others 2008, Fites and others 2007, Harbert and others 2007, Wimberly and others 2009).

Because there are few studies of the effectiveness of fuel treatments that are subjected to actual wildfires, our literature review reflected not only fuel treatments but also fuel modifications or management practices that altered vegetation composition and, thus, the fuelbed. It is important to recognize that many fuel treatments are similar to the vegetation manipulation treatments that have long been applied in forests and rangelands. That is an advantage because most treatments will not be tested by fires. Thus, it is important that fuel treatments be thoughtfully matched to the ecological conditions for long-term sustainability, as well as to be feasible and socially acceptable.

More consistent and specific quantitative data are needed to assess treatment effectiveness. For instance, burn severity is measured in many different ways. Weatherspoon and Skinner (1995) use 50 percent crown scorch as their measure of extreme fire damage, while Omi and Kalabokidis (1991) define severe damage as 100 percent crown consumption. Rating systems such as that outlined by Omi and Kalabokidis (1991) and Ryan and Noste (1985) for quantifying tree canopy scorch and consumption of surface and ground fuels have been used by many. The Composite Burn Index provides a generalized rating of post-fire conditions in the field and includes fire effects on both vegetation and soils (http://www.nrmsc.usgs.gov/science/fire/cbi/description). Other measures of burn severity include methods from Brown (1974), Ffolliott and others (1968), and Keeley (2009).

Unanswered Questions

Are fuel treatments less effective in stand-replacing fire regimes?

Most studies regarding fuel treatment efficacy focus on dry forest types where fire exclusion and climatic fluctuations have affected vegetation on a grand scale (Allen and others 2002, Pollet and Omi 2002, Westerling and others 2006). Therefore, fuel treatments that reduce burn severity and, in some cases, promote ecological restoration have been widely accepted in those regions. However, in subalpine forests, current fire regimes have been significantly less impacted by fire exclusion, meaning that fuel composition is more typical of historical conditions (Schoennagel and others 2004, Turner and others 2003). Climate fluctuations that result in periods of prolonged warm, dry conditions have contributed to the large, severe fires of recent decades in cold forests (Morgan and others 2008, Swetnam and Westerling 2007).

Therefore, if fire intensity and severity are dictated more by the weather leg of the fire triangle than by fuels, are fuel treatments likely to be less effective outside of dry forests? Weather affects the availability of fuels, not the abundance of the tree canopy fuels that are necessary to carry fire across the landscape. It stands to reason that reducing canopy fuel loading and tree spacing would reduce the potential for stand-replacing fires. Our case study results in the lodgepole pine-dominated forests of central Idaho suggest that fuels management in those forests can mitigate post-fire effects.

In subalpine forests, new ecological problems may result if stands are maintained outside of their natural range of variability (Schoennagel and others 2004), for example by promoting large areas of low severity burn through fuel treatments. This should be considered when determining treatment size, location, and prescription. Spatial data such as the historical burn severity data made available through the national-scale MTBS project (www.MTBS.gov) are available to evaluate patterns of burn severity across large areas (Wimberly and others 2009). Wilderness areas that tend to be at higher altitudes are logical places to focus such efforts. Such data could also be used to inform models of where on the landscape high severity burns are most likely to occur (Holden and others 2009)—information useful in both planning fuel treatments and fire management.

What about masticated fuels, biochar, and other innovative treatments?

We concur with Graham and others (2009) that the single mastication treatment (Warm Lake Highway) that was tested by an intense wildfire under extreme weather conditions worked to reduce the intense crown fire to a low intensity surface fire, although fire suppression measures undoubtedly played a role. Being a relatively new method, precious little is known regarding the effectiveness of mastication treatments. This will surely change, because as more and more mastication treatments are implemented, some will inevitably be tested by wildfires. In theory, masticated fuels are more likely to smolder than to flame, compared to natural fuel. Mastication usually results in a relatively compact layer of fuel. If it burns, long-term smoldering would likely result in deep soil heating (Haase and Sackett 1998, Hungerford and others 1991). Busse and others (2005) found that soil temperatures at a depth of 10 cm under burning residues from mastication of 7.5 cm or more surpassed the 60 °C threshold that is lethal for most living plant tissue.

There are other good reasons to implement fuel treatments besides reducing hazardous fuel accumulations and lowering the risk of catastrophic fire. Implementing fuel treatments can provide jobs and help people feel safer. Fuel treatments may improve habitat for wildlife species of interest. Treatments also may benefit forest carbon management beyond the treatment effects on aboveground biomass measured in this study. Biofuels may increase in marketability in the future. Another future management objective may be to increase carbon sequestration by mimicking the more active historical fire regime with periodic prescribed fires. North and others (2009) found that fire suppression causes surface and standing, small-diameter fuels to accumulate, which contributes to higher emissions from either fuels treatment activities or possible wildfire combustion. The same forest maintained with frequent fires, as was the historical norm, emits less carbon.

How far away?

The distance that fuel treatments should be from the values they are intended to protect is of particular importance in the WUI where fuel treatments are commonly justified as reducing wildfire threats to homes. Wildfires that transition to structure fires are becoming more common and costly, which is as a major challenge and hazard for wildfire management. Solving this problem requires bridging the gap between structural and wildland fire management and research.

Maranghides and Mell (2009) collected post-fire data following the 2007 Witch and Guejito fires in The Trails development of Rancho Bernardo north of San Diego, where 74 of 245 homes within the fire perimeter were destroyed and 16 were damaged. Fifteen of the 16 damaged homes were defended; it is likely they would have been destroyed had they not been defended. One out of every three homes was defended, which probably reduced losses from over 37 to 30 percent. Early findings, based on field work, are that 40 percent of homes on the edge of the development were destroyed compared to 20 percent in the interior. Direct flame impingement from structure to structure was not identified as a significant contributor to fire spread within The Trails. Nineteen of the destroyed structures were categorized as having possibly ignited due to the fire carrying through uninterrupted vegetation on the edge of the development; 20 destroyed structures were categorized as having ignited from embers, both at the edge and in the interior of the development; 35 destroyed structures were categorized as having vegetation near the structure that may have ignited from embers, if the structures were not directly ignited from embers. It is possible that all of the homes destroyed may have been ignited from embers. The majority of damage was caused not by embers preceding the main fire front but by embers from the main fire front. Structure ignitions peaked at 21 per hour when the main fire front reached the community, with 29 (40 percent) of the structures burning at the same time. Structure ignitions continued for nine hours following the arrival of the main fire front.

Understanding exactly how structures ignite from wildfires is critical to both fire managers and homeowners. Cohen (2000) concluded that the construction of the home and the fuels within 40 m of it determine ignitibility. Modeling and case studies suggest that if firebrands and/or flames do not come within approximately 40 m of structures, ignition is unlikely (Cohen 1999, 2000, 2004; Cohen and Stratton 2008). The most common source of home or structure ignition is firebrands from other homes, not direct flame impingement from a wildfire (Cohen 2000, Cohen and Stratton 2008). In many cases, homes act as the fuel that carries fire through communities in lieu of wildland fuels (Cohen and Stratton 2008).

These conclusions have several implications for fuels management. First, fuel treatments do not need to be very large in order to significantly lower the probability of structure ignition. Second, when a home ignites, it becomes the fuel that can continue the spread of fire to surrounding homes. Thus, Firewise communities that apply Cohen's findings will likely be more resilient than poorly constructed homes surrounded by a large fuel treatment. Third, because the home ignition zone is in close proximity to the structure, Cohen (2000) concluded that "the WUI fire loss problem can be defined as a home ignitibility issue largely independent of wildland fuels management issues."

There is no substitute for fire-resistant home construction and improvements within the WUI that ultimately lower home ignition potential. Fuels management and Firewise treatments can complement each other but will not replace one another. Applying fuels management outside of the home ignition zone may be more necessary where complex topography and the potential for extreme fire behavior can complicate exactly how far a Firewise-treated area should extend from the home. Fire intensity and spotting distance are affected by slope (Murphy and others 2007), and topographic features such as canyons increase fire intensity and can facilitate home ignition. In some situations, a larger Firewise-treated zone and more resistant structure are necessary to sufficiently minimize ignition potential. Certainly, the existence of the 2005 Firewise treatments on private lands at Secesh Meadows, not just the 2006 treatments on surrounding NF lands (table 1; figures 11 and 13), was a factor that enabled firefighters to protect all of the homes (figures 32 and 33). Similarly at Warm Lake, the fuel treatments in the WUI did not stop the fire progression, but they directly impacted home survivability, allowed for safer and more efficient fire suppression, and played an integral role in the point-protection management strategy (Graham and others 2009). The treatment units west of Warm Lake (figure 12) effectively slowed down the eastward momentum of the Monumental and North Fork crown fires and exemplified how strategically placed fuel treatments can help firefighters save structures in the WUI.

How does one consider spatial patterns of fuels and treatments?

Determining the optimal spatial pattern of fuel treatments across a landscape brings together all of the factors of fuel treatment efficacy discussed thus far (for example, Moghaddas and Craggs 1997, Ritchie and others 2007, Skinner and others 2004, Weatherspoon and Skinner 1995). Strategic locations, prescription, temporal thresholds, and fire regimes must all be considered to decide where and when to implement fuel treatments for maximum effect. Because of the cost and magnitude of testing various spatial patterns of fuel treatments, most studies are limited to modeling.

By simulating treatment shapes and arrangements, Finney (2001) determined that separate, partially overlapping treatments (similar to a checkerboard) was the most effective spatial pattern for slowing the growth of large fires. Within this pattern, fuel treatments were rectangular and oriented so that the short axis of the treatment units were parallel to the primary direction of fire spread. This spatial pattern resulted in lower intensity head fires progressing through the fuel treatments and primarily flanking fires in the untreated areas. This pattern also resulted in fire moving across the landscape in a uniform manner, minimizing the areas where two flame fronts converge and produce high intensities and rapid rates of spread (Finney 2001).

However, accounting for multiple directions of fire spread complicates how fuel treatments should be arranged. By locating treatments in areas of high fire

Figure 32. Structure protection at Secesh Meadows. *Photo: Roger Staats.*

Figure 33. Crews foaming to defend a home at Secesh Meadows. *Photo: Roger Staats.*

susceptibility, Parisien and others (2007) determined that clustered treatments were the most efficient in reducing the spread of fires burning from several directions. Using fuel treatments to connect lakes and other natural barriers further reduces fire spread potential (Parisien and others 2007). Temporal limitations further complicate the spatial arrangement of fuel treatments as there needs to be a balance between maintaining existing treatments and implementing new treatments to sufficiently reduce the threat of large fire events. Assuming that all individual treatments are effective in moderating fire behavior and are placed in optimal locations, Finney and others (2007) found that only 1 to 2 percent of the landscape must be treated annually to sufficiently lower rate of spread and intensity. Randomly located fuel treatments with identical prescriptions required approximately 4 percent of the landscape be treated to reduce large fire growth to the same level. This means that fewer maintenance treatments would be necessary to achieve the desired result. If 2 percent of the landscape is treated annually, then fewer than 5 percent of the treated areas would receive three or more treatments over five decades (Finney and others 2007).

Because most landscapes are heterogeneous, designing optimal fuel treatments is not simple. Fuel treatments must consider where and how fires are likely to spread in landscapes with variations in topography, vegetation, land uses, and land management objectives. Tools such as ARCFUELS can be useful in evaluating alternative treatment scenarios.

The paradox of fire suppression is that the more we suppress fires, the more intensely they may burn in the fuels that have accumulated. A related issue is the degree to which many valued ecosystem services benefit from fire. Landscape-scale fuels management must be thoughtfully designed to protect resources at risk and to promote other objectives such as water quality, wildlife corridors, and habitat diversity (figure 34).

Figure 34. Aerial view of fire severity patterns across the Secesh Meadows landscape soon after the wildfire. *Photos: Tim Sexton.*

Are fuel treatments less effective in extreme weather conditions?

Probably. This is a widely held assumption, but Omi and others (2006) found that the effectiveness of combined surface and canopy fuel treatments actually increased with weather severity, as indicated by the Burning Index of the National Fire Danger Rating System. More quantitative assessments are required.

What about climate change?

Fires will burn under conditions greatly altered by people—long fire seasons, extreme weather conditions, expanding urban interface, and many invasive species will test the resilience of western ecosystems. Decades of fire suppression across the West have promoted unnatural fuel accumulations in many areas, especially in some dry ponderosa pine forests that historically experienced frequent, low intensity fires that were ignited by lightning and Native Americans (Schoennagel and others 2004). Fuels are often abundant in mixed conifer and subalpine fir forest types, but the effects of fire exclusion are much less pronounced than in dry forests. Another perception that is gaining acceptance is that we are observing the effects of climate change, as evidenced by the growing number of large fires in the western United States (Westerling and others 2006). Whether the pervasive drought that has afflicted much of the West is part of a long-term trend remains to be seen. Both climate change and land use change will continue to interact and influence fire regimes, but the relative importance will vary with ecosystem and location.

Conclusion

Fuel treatments altered fire behavior and subsequent fire effects in multiple large fires in 2007, usually in desirable ways but not always. Our results confirm the widely held notion that mechanical thinnings are the most effective fuels treatment, provided the activity fuels are treated. Prescribed burn treatments may be the most cost effective maintenance treatments for keeping fuel accumulations in check over time and for lowering the risk of severe fires. Further quantitative research studies and more consistent protocols are needed to assess fuel treatment effectiveness, especially over the longer term to evaluate the duration of treatment effectiveness and the cumulative effect of multiple treatments in the same location.

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Station Headquarters
Rocky Mountain Research Station
240 W Prospect Road
Fort Collins, CO 80526
(970) 498-1100

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