UTAH STATE UNIVERSITY

Wildfire in Utah

The Physical and Economic Consequences of Wildfire

Paul M. Jakus, Editor

1/15/2017

Project Authors:

Paul M. Jakus
Man-Keun Kim
Department of Applied Economics
Center for Society, Economy, and the Environment

Randy C. Martin
Ian Hammond
Department of Civil and Environmental Engineering
Utah Water Laboratory

Edd Hammill
Nancy Mesner
Jacob Stout
Department of Watershed Resources
Ecology Center

PRELUDE

On June 26, 2012 a lightning strike ignited a wildfire in the Manti-La Sal National Forest of central Utah's Carbon and Emery counties (Figure P.1). By the time the Seeley wildfire was contained three weeks later, some 48,000 acres of federal, state, and private land had been burned and $8.7 million in suppression costs expended (Styler 2012). According to the Monitoring Trends in Burn Severity (MTBS.gov) project, nearly one-third of the acreage was severely burned, damaging vegetation and soils for years to come. Severe burns vastly increase the erosion potential of burnt landscapes, and the steep lands of Huntington Canyon proved to be no exception.

Figure P.1: Smoke from the Seeley wildfire. *Source: Inciweb.nwcg.gov*

Subsequent thunderstorm events caused dangerous landslides as debris flows composed of water, boulders, gravel, sand, tree trunks, and ash scoured soils down to bare rock (Giraud and McDonald, 2013; Figure P.2). The flows also increased erosion in Huntington Creek and severely damaged State Highway 31, which was subsequently often closed due to debris and washouts. The debris-filled water also threatened drinking water supplies in Huntington City and cooling water used in the Huntington Power Plant. Runoff choked with rubble and sediment also affected water quality and wildlife habitat, killing fish as far as 50 miles downstream in the Price and San Rafael Rivers (Walker, 2013). In addition, fire-related danger closed six campgrounds in the Huntington Creek watershed, accounting for nearly 30% of campsites in the area (Huntington Creek Watershed Plan, undated).

Figure P.2: Debris flow from the Seeley wildfire. Source: Inciweb.nwcg.gov

Over the longer term, the Seeley fire has the potential to change runoff and snowmelt patterns on the burned area for several years after the fire (Anonymous, 2013). A number of mitigation and restoration activities were undertaken in the Huntington Creek watershed following this wildfire, at a cost of over $4 million. Had the fire burned above the water supply reservoirs located at elevations above the fire (e.g., Electric Lake), damage to water supplies, and mitigation costs, would have been far higher.

The Seeley fire was atypical of wildfire events in Utah; most wildfires are smaller and result in much less severe ecological and infrastructure damage. That said, the Seeley wildfire illustrates the numerous issues associated with wildfire. Wildfires not only result in the loss of trees and forage on burned ground, but can also have effects on ecological services associated with water supply and water quality. In addition, wildfires can affect the use of public range by ranchers on forested and non-forested lands. Smoke from wildfires affects air quality, causing downwind respiratory effects. Wildfires can also limit recreational services provided in areas that have burned and in areas that are affected by smoke.

References

Anonymous, 2013. The Seeley fire and potential impacts on Huntington Creek water Supply – Clay Springs fire and potential impacts on Oak Creek water supply. http://docshare01.docshare.tips/files/13409/134094069.pdf

Giraud, R. and G. McDonald. 2013. Damaging debris flows prompt landslide inventory mapping for the 2012 Seeley fire, Carbon and Emery counties, Utah. Survey Notes (Utah Geologic Survey) 45(3):1-3.

Huntington Creek Watershed Plan. Undated. http://www.emerycounty.com/Appendix-F.pdf

Styler, M. 2012. Utah Legislative presentation: 2012 Fire suppression and restoration costs. http://le.utah.gov/interim/2012/pdf/00001083.pdf

Walker, C. 2013. Utah Water Quality Task Force presentation: Impacts of forest fires on water quality. (May 22). http://www.deq.utah.gov/ProgramsServices/programs/water/nps/docs/2013/08Aug/052213Finalminutes.pdf

EXECUTIVE SUMMARY

- Between 2002 and 2015 Utah has annually averaged 1,283 wildfires burning 178,437 acres. In recent years Utah has experienced numerous fires in excess of 40,000 acres. Utah's worst fire year in recent memory was the 620,000 acres burned in 2016.

- *Fire-dependent* ecoregions are those in which fire of a specific type is needed to support native plants and wildlife. Ecoregions that are fire dependent are often resilient and strengthened as they recover from wildfires of the appropriate periodicity and intensity (fire regime). Much of Utah is composed of fire-dependent ecoregions.

- Fires that depart from a natural regime can cause ecosystem damage though severe burns. *Burn severity* distinguishes fire-intensity (energy release) effects on vegetation (above- and belowground loss of organic matter) separately from effects on soil. The Monitoring Trends in Burn Severity (MTBS) project provides satellite measurements of near-infrared and shortwave infrared spectra to calculate two metrics that have been found to be highly correlated with field evaluations of burn severity, the differenced Normalized Burn Ratio (dNBR) and the Relative differenced Normalized Burn Ratio (RdNBR)).

- The MTBS burn severity classification approach relies upon applying standard thresholds to the dNBR and RdNBR measures. Field researchers have documented that application of such thresholds can result in relatively large errors in burn severity classification. MTBS products can provide a general sense of burn severity, but errors are such that the metrics are best used in conjunction with validation in the field.

- The West Wide Wildfire Risk Assessment (WWRA) was completed in 2013; the report and its publicly available GIS data currently represents the most comprehensive and robust estimate of Fire Threat for the region. State of Utah personnel advised the project and the Division of Forestry, Fire and State Lands use WWRA products.

- The WWRA fire threat index provides an annual burn probability risk at a pixel resolution of 30 m^2. Burn probabilities were used to replicate annual wildfire activity 1000 times, providing an empirical distribution of acreage burned, vegetation burned, and risk to drinking water sources.

- The fire threat index was also evaluated with respect to land characteristics and ownership. All else equal, fire risk was higher for land that was steeper and north-facing. Certain vegetative types were also more prone to fire: upland mixed forests, upland shrubland, and land with greater than 60% shrub cover. Lands that are administered by federal agencies were more likely to have land with the characteristics listed above. There is little land managers can do to affect slope and aspect, leaving vegetative changes the key avenue for controlling fire risk.

- Due to computational constraints, two regions were selected to evaluate air and water consequences of wildfire. A 1.7 million acre cluster of urban counties (Davis, Morgan, Salt Lake, and Weber) were selected to represent the high population densities where much of Utah's population resides, yet is home to an important supply of water (the Weber River). The rural cluster was composed of Juab and Sanpete counties, totaling 3.2 million acres. This cluster provided variation in climate, precipitation, and biota that characterizes much of Utah; this cluster also has a land ownership pattern that closely resembles that of the state as a whole.

- Empirical distributions of annual burned acreage—along with the type of vegetation burned—were generated by simulating yearly fires 1000 times. Acreage burned in a typical year in the Urban cluster was 6,151 acres while in the rural cluster acreage burned was 24,306 acres. These simulated results are in line with average acreage burned as calculated from 1992-2015 wildfire history.

- The empirical distribution of burned vegetation was converted to kg of biomass using a GIS layer developed by the Oak Ridge National Laboratory. Conversion factors were then applied to determine annual fire-related releases of particulate matter and other pollutants.

- Total fire-related pollutant emissions were calculated for the urban and rural clusters, and then compared to emissions levels of pollutants emanating from primarily anthropogenic sources ("inventoried" sources). Emissions modeled were carbon monoxide, PM2.5, NOx compounds, volatile organic compounds, methane, carbon dioxide, nitrous oxide, and ammonia.

- With the exception of ammonia, urban cluster wildfire pollutant emissions are generally about 10% of inventoried sources; wildfire-associated ammonia releases are about 159% on annual inventoried emissions. In the rural cluster, the proportion of some wildfire pollutant emissions relative to inventoried sources is similar to urban regions (NOx, VOCs, methane), but releases of the other pollutants are much higher than emissions from inventoried sources. Emissions of ammonia are 1800% of anthropogenic sources.

- Increases in runoff flows (m^3/s), sediment concentrations (mg/l) and sediment loads (tons) were estimated using USGS curve numbers and the Modified Universal Soil Loss Equation (CNN-MUSLE model). Increased runoff can result in flooding and damage to river banks. Pollutants such as heavy metals and phosphorus are associated with sediment. In addition, increased sediment volumes (loads) may fill in downstream reservoirs as well as smother fish habitat.

- Modeled responses under a range of precipitation events in two watersheds, the Upper Weber River (located upstream of the Urban cluster) and the Lower Sevier River (part of the Rural cluster), suggest that forested landscapes are more susceptible than rangelands to higher runoffs and to increased sediment loads following severe fires. Predicted increases in sediment yields and runoff were highest in the deciduous forests of the urban watershed.

- Sediment and runoff responses to a 50 mm rain event, using the CNN-MUSLE method for each 30 m pixel in the state, were aggregated into average responses for HUC 10 (relatively small) watershed areas. Predicted runoff increases were greatest in several northeast watersheds, while the increases in sediment (and therefore other pollutant) concentrations were greatest in the extreme north of the state. Watersheds susceptible to the greatest increases in loads of sediment and other pollutants were distributed across the mountainous areas of the state, with most of the vulnerable watersheds in the northern half of the state.

- About 1/3 of all the drinking water withdrawal points and drinking water reservoirs fall within areas predicted to have substantially increased sediment (and other pollutant) concentrations following a severe fire, and almost 50% of these sites are in areas predicted to have increased sediment loads. Several drinking water infrastructures in the state's southeast appear to be particularly at risk.

- All Blue Trout fisheries were predicted to have at least modest increases in flow (flooding), sediment concentration and sediment load following a severe burn. As noted above, sediments typically deliver other pollutants of concern, such as metals and phosphorus. The Middle Provo River and portions of the Weber River were predicted to have higher risk of increased concentrations of sediment and sediment associated pollutants. The Duchesne River and Huntington Creek were at higher risk of flooding from high flows as well as higher sediment volumes, which can smother high quality fish habitat. Of all Blue Ribbon Reservoirs, Steineker Reservoir was predicted to have the highest risk of sediment delivery, which would impact sediment associated pollutants as well as increase the rate at which the reservoir is filled (i.e. decrease reservoir life expectancy.)

- Ranchers of southern Utah are heavily dependent upon public rangeland to profitably run their operations, and they have little access to private alternatives if wildfire displaces animals from the public range. For example, in our study region of Garfield, Grand, Kane, San Juan, and Wayne counties, private owners control only 7% of the region's land while 82% is held publicly.

- Wildfire on the public range causes affected ranchers to remove animals from the range and then decide if they wish to feed the animals with purchased hay, or simply sell the animals to avoid the higher feed costs. Our modeling suggests that ranchers engage in both actions.

- A cattle inventory model links the number of cattle in the five counties to wildfire activity from 1992-2015, finding that for each 1000 acres of wildfire some 40 feeder cattle are sold to reduce inventory. Past research suggests that ranchers sell such animals at a 40% discount relative to the value received for animals at full weight. The total regional economic losses (direct, indirect, and induced) of premature sale are about $500,000 in a median fire year and about $1.4 million in a mean fire year. These losses are recovered as the range recovers, which is typically a process of at least two years.

- Higher feed costs for animals displaced form the range reduce the profitability of ranching. Total losses in labor income due to reduced profitability (direct, indirect, and induced) is about $250,000 in a median fire year and $650,000 in an average fire year.

- Wildfire activity also affects recreation behavior as roads, campgrounds, and trails may close, smoke affects visibility and, possibly, health, and some may believe that recreation near a wildfire is dangerous and/or stressful.

- Statistical models linking visitation at each of Utah's five national parks to wildfire suggests that visitation at four (Arches, Bryce, Capitol Reef, and Zion) are all affected by wildfire activity within a 50 mile radius of each park's visitor center. Peak season visitation to the parks is expected to fall by 11,000 visitors during a median fire year, and by nearly 31,000 visits during a mean fire year.

- Changes are relatively small (less than 1%), but at an average expenditure of $73 per visitor per visit, the loss in expenditures by visitors is $780,000 during a median fire year and $2.3 million during a mean fire year. The total economic losses (direct, indirect, and induced effects) due to fire-related decreases in visitor spending are $1.2 million for a median fire year and 3.7 million for a mean fire year.

- Data from July 2004 through October 2015 were used to link wildfire activity within a 200-mile radius of Salt Lake City to monthly average PM25 concentrations. On average, wildfires increase PM25 concentration by 1.62 $\mu g/m^3$ during June, July, and August. It is associated with 0.5% increases in respiratory-related medical admissions and a 0.8% increase in cardiovascular admissions.

- A recently released USFS study found that some 54% if Utah's forests are fully occupied and that 21% of the state's forests are overstocked. Overstocked forests are at greater risk of increased mortality due to competitive stress, and also at greater risk of catastrophic fire. National wildfire suppression costs are highly variable and are rising over time. Analysis of 450 Utah wildfires found total suppression costs to rise with fire size and increase with rugged topography. Per-acre suppression costs fell with fire size and increased with rugged terrain.

- Numerous studies of the efficacy of fuels reduction treatments are in broad agreement that fuels reduction efforts, especially combined prescribed burn/mechanical treatments, can be very effective in modifying fire behavior to reduce the severity of wildfire. Simulation modeling indicates that fuels treatments can also achieve a number of co-benefits, including reduced suppression costs, though this is not a primary goal of most fuels reduction efforts.

- Fuels treatments are quite costly--approaching $175 per acre for prescribed fire, and possibly in excess of $1,000 per acre for mechanical treatments. Fuels reduction programs remain a relatively small portion of overall wildfire management budgets; funding for treatments is not sufficient to meet needed landscape-scale fuel reduction efforts.

- The cost of fuels treatments can be offset through the sale of biomass removed as part of the treatment process, but hauling costs for small diameter trees and chipped volumes are high relative to its value. Empirical research suggests that treatments tend to occur in proximity to existing wood processing facilities.

- Utah's forest timber harvest has fallen by 70% between 1992 and 2012; during the same time period the number of mills fell by nearly 66%. In 2012, Utah mills operated at 20% of capacity, suggesting scope to absorb a large volume of wood product generated by fuels reduction activities. The spatial distribution of mills, though, suggests few opportunities to sell removed biomass to the few remaining mills in central and southern Utah.

Table of Contents

CHAPTER 1: INTRODUCTION

This study is sponsored by Utah House Bill 464, enacted by the Utah Legislature during its 2016 Legislative session. The goals of the legislation were many and ambitious, including

- Documenting historical number and acreage of wildfires in Utah,
- Assess severity of wildfire in Utah with regard to public land ownership and management,
- Assess fire risks (the probability of wildfire) in Utah, and use numerical simulation to in two landscape-scale regions to gauge impacts of wildfire in these regions,
- Assess runoff and pollutant risks associated with wildfire,
- Assess air pollutant emissions,
- Assess the economic impacts of wildfire in Utah.

This report documents our efforts to satisfy the goals of the legislation. Chapter 2 provides an overview of Utah's ecoregions, vegetative zones, and land ownership. As is well-known, Utah is a mountainous, semi-arid state. Precipitation varies greatly across the state, ranging from less than 5 inches per year to over 10 times that amount at higher elevations. From its lowest point to its highest point, the elevation change in the state is about 11,000 feet. Variation in elevation and precipitation—which are correlated—has led ecologists to classify the state into different ecoregions that reflect differences in climate, biota, and landform.

In Chapter 3 we note that characteristics of an ecoregion are important because these help determine the local fire regime, where an appropriate fire regime is one whose timing, frequency, and intensity sustains ecosystem health and does not cause long-term damage. After reporting on Utah's wildfire history for the period 1992 through 2015, we provide greater technical detail on the terms related to fire regime, especially the term *burn severity* and how burn severity relates to ecosystem health. Statistics on burn severity in Utah from 1992 through 2015 are reported, where burn severity has been calibrated by the federally-sponsored Monitoring Trends in Burn Severity (MTBS) project. Statistics indicate growing severity of wildfire on land administered by the USFS. However, a growing literature that has field-validated MTBS products suggests that burn severity and the fire perimeters delineated by MTBS are subject to large errors. The consensus among fire scientists is that the MTBS products provide a general sense of burn severity, but errors are substantial enough that they not be used directly for empirical work without field-validation.

Wildfire risk in Utah is addressed in Chapter 4. We use the Fire Threat Index (FTI) developed for western states by a consortium of state, federal, and industry groups. The consortium's final report, entitled *West Wide Wildfire Risk Assessment* (WWRA), provided a number of high resolution GIS data layers, including an assessment of the annual burn probability (each pixel is 30 m^2). The FTI was used is two ways. First, we estimate a statistical model relating fire risk to pixel characteristics such as dominant vegetative cover, slope, and aspect (the direction of the slope, such as north-facing, west-facing, etc.) Each of these factors affects fire risk; we then evaluate pixel characteristics to land ownership. All else equal, we find that lands administered by federal agencies are more fire prone than lands administered by the state of Utah.

The second use of the WWRA product was to generate fire simulations for two selected regions of the state. The urban region was composed of Davis, Morgan, Salt Lake and Weber counties, while the rural region was composed of Juab and Sanpete counties. The urban region was selected to reflect an area where a great concentration of Utah's population resides and also included an important source of water (the Weber River Basin). The rural region was selected because it exhibited the great variation in climate/precipitation/biota common in Utah, and its land ownership and administration closely resembled that for the state as a whole.

Using the annual fire risk layer, simulated fires were used to develop estimates of burned vegetation and possible damage to important drinking water resources in each region. We then use the simulated distributions for burned vegetation to generate estimates of the various particulate matter and gasses that make up wildfire-related smoke in the two regions (Chapter 5). Wildfire-generated pollutants (such as carbon monoxide, PM2.5, NOx, volatile organic compounds, ammonia, carbon dioxide, methane, and nitrous oxide) are then compared to the inventoried levels of each particulate emanating from other sources. We find that, in the cluster of urban counties, a typical wildfire year adds relatively little (<10%) to inventoried particulates (with the exception of methane.) In contrast, a typical fire year in the rural region saw that short-term wildfire events can dramatically increase particulate concentrations above inventoried sources.

The impact of wildfire on water resources can be substantial, damaging water quality, increasing the risk of flash floods, and increasing the risk to water supplies. Chapter 6 applies a "curve number approach" to predict changes in runoff flow and sediment yield as a function of hypothesized precipitation events before and after a fire. Changes in flow and sediment yield were estimated for rangeland and forested areas located in the Lower Sevier River watershed (Rural cluster) and the Upper Weber River (located just upstream of the Urban cluster). A different modeling approach, the Soil and Water Assessment tool (SWAT) was also applied to the Upper Weber watershed, in which runoff flows and sediment loads were calibrated to known values. Changes in runoff and sediment were estimated by simulating burned acreage. Predicted changes in flows and sediment loads using the SWAT model were smaller than those predicted by the curve number approach, but the results were, in general, consistent with one another.

The economic impact of wildfire on rangeland resources was estimated in Chapter 7. If wildfire burns the publicly administered range, then those rangelands are generally not available for two years after the fire. Ranchers in southern Utah (Garfield, Grand, Kane, San Juan and Wayne counties) manage their operations in a region of the state where rangeland is at risk from wildfire, where alternatives to forage supplied on the public range are few, and where agricultural activity make up a large portion of the local economies. Hence, wildfire has the potential to impact not just individual ranchers, but can also spillover to the broader regional economy. We estimate a statistical model relating cattle inventory to wildfire and find that ranchers reduce herd size in response to wildfire. Inventory reductions in a median fire year are quite small, but a large wildfire year (representing the mean acreage burned) might result in premature sale of over 725 head.

Smoke-induced haze and possible health effects of wildfire can also affect Utah's largest export industry, tourism. Wildfire is primarily a summer phenomenon, and summer represents the peak visitation period for southern Utah's public lands. The most consistent and easily available dataset for tourism is aggregate monthly visitation to national parks. For each of Utah's five national parks (Arches, Bryce, Canyonlands, Capitol Reef, and Zion) we link wildfires occurring within a 50 mile radius of the park visitor center to monthly visitation (Chapter 8). The statistical model indicates that wildfire reduces visitation at every park except Canyonlands. For a median fire year, regional economic impacts are a loss of about $1.2 million in industry output; for a mean fire year losses in industry output are estimate to be $3.7 million.

Smoke-related effects on human health are investigated in Chapter 9. This chapter uses ten year's worth of data to estimate a model linking wildfire activity within a 200 mile radius of Salt Lake City to PM2.5 concentrations measured at an air quality monitor in the city's downtown. Simple modeling suggests a statistically significant increase in PM2.5 concentration due to wildfire.

Finally, Chapter 10 addresses the costs of wildfire and fuels reduction efforts, the efficacy of fuels reduction treatments, and the role of markets for merchantable woody biomass to improve the net cost. Using data from 450 Utah wildfires, we estimate total cost and cost per acre suppression models, establishing a statistical relationship between costs, fire size, and topography. Fuels reduction methods are reviewed, after which a review of the literature is used to evaluate the effectiveness of such efforts and how markets for wood products can affect the net cost of fuels reduction.

CHAPTER 2: UTAH'S ECOREGIONS, VEGETATION, AND LAND ADMINISTRATION

Paul M. Jakus
Dept. of Applied Economics and Center for Society, Economy and the Environment
Utah State University

To understand wildfire and its effects on Utah's population and its land, air, water resources, we must first review the physiography of the state and, given that Utah is a public lands state, who administers its land. By any measure, Utah exhibits great variation in its natural features. It has over 11,000 feet of elevation change, from its low point at 2,350 feet (715 meters) above sea level in Beaver Dam Wash in the southwest corner of the state to the 13,500 foot (4125 meter) heights of King's Peak in the Uinta Mountains of northeastern Utah. In between these limits lie over 70 mountain ranges and their associated valleys and plateaus. With the notable exception of the Uinta Mountains, nearly all of Utah's mountain ranges run north-to-south with mid- to high-elevation valleys between ranges (Gillies and Ramsey, 2009).

Much of Utah is characterized by dry, hot summers and cold winters, with precipitation coming in the form of both rain and snow. Annual average precipitation in Utah varies widely by both elevation and latitude (Figure 2.1), ranging from less than 5 inches (125 mm) per year to over 55 inches (1400 mm) annually. Snow pack dominates the precipitation at higher, alpine elevations, whereas many lower elevation valleys and plateaus—and those portions of the state at lower latitudes—receive the bulk of their limited precipitation in the form of rain. Even in the southern portion of the state, though, snow remains the most important source of water because high elevation snowpack acts as a natural reservoir.

Ecoregions in Utah

Classifying an area into "ecoregions" is an attempt to depict succinctly the great variation observed in temperature, precipitation, landforms, and biota. While numerous systems have been used, we will rely upon the system developed by Bailey (1995, 2009) and used by the US Forest Service. Bailey proposed a system that discriminated among regions by increasingly fine distinctions based primarily on differences in climate (because these differences affect vegetation). The initial system focused on three levels of macroscale, *Domain*, *Division*, and *Province*, which were then subdivided to the mesoscale.

Four domains are at the top level of the ecoregion hierarchy and are arranged according to temperature and precipitation: Polar, Humid Temperate, Dry (in which evaporation exceeds precipitation), and Humid Tropical. Each domain is further subdivided into 15 divisions, where the primary criterion is again climate. Divisions are subdivided into Provinces, which are based upon the macrofeatures of vegetation found in the province. Provinces are then subdivided by landform (terrain features such as mountains or plateaus) into Sections.

Utah lies wholly within the Dry domain of the Bailey ecosystem classification, which is distinguished by its relatively low precipitation and cold winter temperatures. Most of the state is covered by the Temperate Desert Division and Temperate Desert Regime Mountain (Ecoregion Codes 340 and M340). Utah's northern mountain region (including the Wasatch and Uinta ranges) is classified as Temperate Steppe Regime Mountains (Code M330). Finally, a swath of southern Utah lies within the Tropical/Subtropical Steppe Division (Code 310). [1]

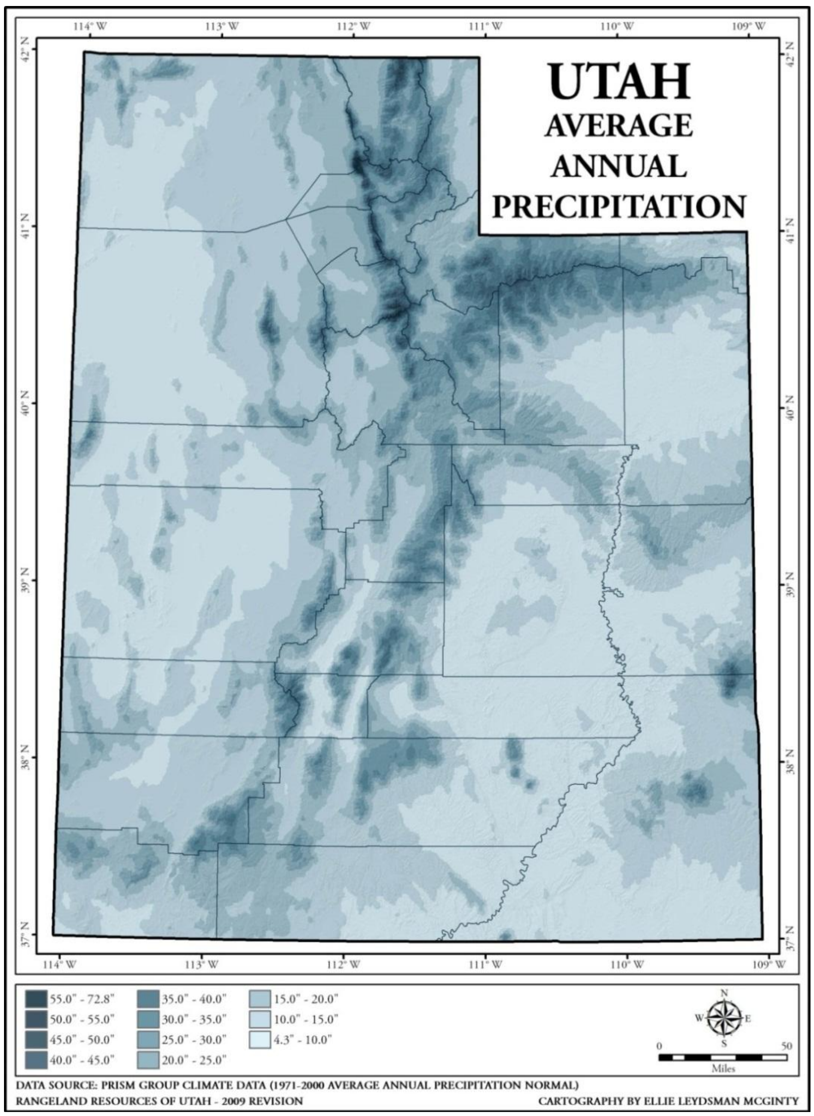


Figure 2.1. Average Annual Precipitation in Utah

[1] A small portion of the extreme southwestern corner of Utah, south of St. George, is in the Tropical/Subtropical Desert Division.

Source: Reproduced from Banner et al. 2009

The Temperate Desert Division is composed primarily of the Intermountain Semidesert and Desert provinces (Codes 341, M341), which contain most of Utah's portion of the Great Basin (Bailey, 1995). In Utah these provinces include the mountain ranges of central and southern Utah, as well as the West Desert. In addition, a stretch of northern Utah bordering Idaho is in the Intermountain semi-desert province (Code 342). These provinces are characterized by large seasonal variation in temperature and relatively low rainfall. Mean annual temperatures range from 40˚F to 55˚F. Precipitation varies with elevation, ranging from 5 to 20 inches annually. As with all ecoregion divisions in Utah, precipitation and vegetation vary according to elevation. For example, the low precipitation, low elevation portions of this division are dominated by sagebrush, whereas upper elevations have more abundant precipitation and conifer forests. Vegetation zones in Utah will be addressed in detail below.

The Temperate Steppe Division in Utah is composed entirely of the Southern Rocky Mountain-Open Woodland-Coniferous Forest-Alpine Meadow province (Code M331). As noted above, this province includes the northern Wasatch and Uinta mountain ranges. Mean annual temperatures are a bit colder than those of the Temperate Desert Division, ranging between 35˚F and 45˚F; valleys in this province will have slightly higher temperatures. Precipitation again varies with elevation, with mean totals between 10 and 20 inches, though very high elevations will have more than double this amount.

The famous red rock regions of southern Utah are covered by the Colorado Plateau Semidesert Province (Code 313). Though the Colorado Plateau is at an average elevation that is higher than other portions of the United States, this province is generally at a lower elevation than other ecoregion provinces found in Utah. Mean temperatures are on par with those of provinces found in the Temperate Desert division (40˚F to 55˚F). Mean precipitation is about 20 inches per year, though portions of the province may receive less than 10 inches annually.

One can observe that the Bailey Ecoregion classification system for Utah at the Section level results in seventeen different ecoregions (Figure 2.2), where the ecoregion provinces described above have been further subdivided by dominant landform. Recalling the link between topography, precipitation, and vegetation described earlier in this chapter, a comparison with the precipitation map (Figure 2.1) reveals a close correspondence between average annual precipitation totals and ecoregion section designation.

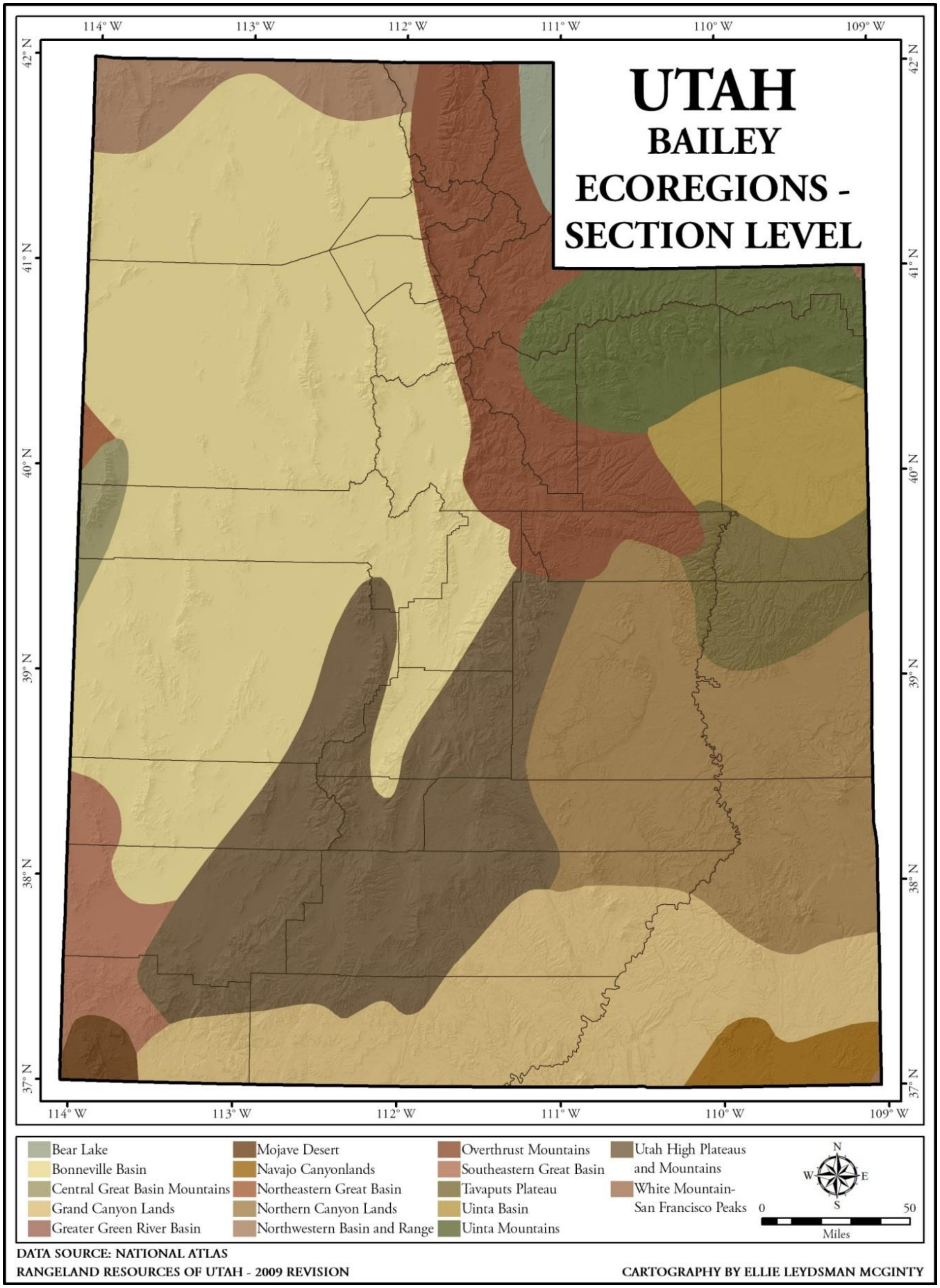


Figure 2.2: Bailey Ecoregions for Utah

Source: Reproduced from Banner et al. 2009

Vegetation in Utah

Within most of the ecoregions shown in Figure 2.2 one will observe great variation in temperature and precipitation; this variation is primarily a function of elevation. Banner (1992) classifies vegetation types into 13 different zones accounting for approximately 70% of the state's area.[2] Banner's description of vegetative zones matches up well with Bailey's ecoregions, though Banner's vegetative zones do not include other factors that influence designation of ecoregions, such as landform (Table 2.1).

The Uinta Mountains, the Overthrust Mountains, and the Utah High Plateaus and Mountains Sections, shown in Figure 2.2, all have mountaintops exceeding 11,000 feet. Banner classifies land above 11,000 feet as the alpine zone. The alpine zone consists mainly of tundra, which has few, slow growing species. The subalpine zone is composed of conifer forests mixed with long-lived pines; this zone was not heavily disturbed by human interventions.

Table 2.1. Major Vegetation Zones in Utah

Vegetation Zone	Area (1000 acres)	Elevation (feet above sea level)
Alpine	498	Above 11,000
Subalpine	1,250	9,000-11,000
Montane	1,744	5,500-9,000
Mountain Brush	955	5,000-8,000
Pinyon-Juniper Woodlands	8,948	5,000-8,000
Wheatgrass/Bluegrass Rangelands	387	5,000-6,000
Sagebrush Steppe	3,858	4,500-5,500
Great Basin Sagebrush	6,553	4,500-6,000
Saltbush/Greasewood	10,507	<6,000
Galleta-Threeawn Shrub Steppe	1,172	<6,000
Blackbrush Rangelands	1,439	3,000-5,000
Creosotebush Rangelands	148	<3,000
Tule Marshes/Wet meadows	328	along streams at any elevation

Source: Banner (1992)

The montane zone is comprised of Douglas-fir species in areas that have not experienced logging or wildfires; if timber activities or fires have disturbed this zone within the past 100 years, then aspen or lodgepole pines predominate. Aspen trees do not tolerate shade, so if they are left undisturbed (as for example, by active suppression of wildfire), they will eventually be crowded out by shade-tolerant species (McAvoy et al. 2012). Lodgepole pines require wildfire to open closed seed cones for propagation of the species. Ponderosa pines dominate in the drier montane regions of southern Utah. The thick bark of a ponderosa pine will often protect older trees from damage associated with normal

[2] Except where noted the bulk of the vegetative zone text draws upon Banner (1992).

wildfire patterns. In addition to timber harvest, the montane zone has been used extensively by livestock and as habitat by high-valued game species such as elk and deer.

The mountain brush zone is often found in transition between the montane zone above and the pinyon juniper zone below. It is characterized by tall shrubs such as Gambel oak and shrub oak, and numerous shorter shrubs and grasses. This zone is also important for grazing and winter range for game species. Following a wildfire, oaks in the montane zone will often develop quickly from sprouts.

One of the largest vegetation zones in Utah—accounting for about 60% of the state's forest cover—is pinyon-juniper (PJ) woodlands, which are often found in the drier regions of the state between 5,000 and 8,000 feet. Composed of singleleaf pinyon, Utah juniper, or both, the woodlands are interspersed with big sagebrush and relatively few grasses. This vegetative zone has been influenced by human interventions such as grazing, timber harvest, and fire control. In fact, tree densities in PJ woodlands have been increasing because of fire suppression and grazing activities (McAvoy et al. 2012). Greater ecosystem damage can occur in PJ woodlands because denser stands burn with greater intensity.

Wheatgrass-Bluegrass rangelands were more widespread in the past, but settlement of Utah's valleys, benches, and foothills have decreased the prevalence of this vegetative zone. This zone was important to settlers and later emigrants in that these lands could be converted to irrigated agriculture. Where this zone has been disturbed, though, non-native species such as cheatgrass have invaded. The sagebrush steppe vegetative zone is often found between the grass rangelands (that have been converted to agricultural or urban uses) and desert vegetative zones. Comprised primarily of sagebrush and big sagebrush, livestock grazing has affected this zone for over 100 years through land management efforts to increase forage production for livestock (Banner, 1992).

The Great Basin Sagebrush and Saltbush-Greasewood zones are often intermingled and, collectively, make up the most extensive vegetative zones in Utah. The Great Basin Sagebrush zone is dominated by sagebrush with a few selected grass species appearing according to elevation, moisture, temperature, aspect, etc. Though used extensively by ranchers as an intermediate region in which to graze livestock as they move between summer and winter feeding areas, the quantity of forage produced in the Great Basin Sagebrush zone is limited. The Saltbush-Greasewood vegetative zone is characterized by variation in soil moisture but relatively high salinity. Euhalophytes (e.g., rabbit brush, saltbush, and salt age) are found on soils whose moisture is less brackish whereas hydrohalophytes (e.g., greasewood and saltgrass) may be found on saltier soils. The Saltbush-Greasewood zone has been important historically for sheep grazing, though in recent years sheep range has been converted to cattle range. Some shrubs found in this zone are not tolerant of fire and are more susceptible to fire damage as invasive species such cheatgrass have encroached and serve as fuel for wildfire.

Blackbrush and Creosotebush Rangelands collectively occupy about 1.6 million acres in Utah. Neither rangeland produces much forage, so grazing impacts have been limited and restricted to the presence of ephemeral plants. Instead, major human changes to these zones have resulted from infrastructure projects such as building pipelines and highways. Finally, the Tule Marshes-Wet Meadows vegetative zone is found throughout the state at all elevations. These lands generally occur in the riparian areas adjacent to rivers, streams, and lakes. Due to the relative abundance of water, a large variety of plants can be found in this zone. In many riparian areas, though, saltcedar (tamerisk) has invaded and crowded out native species. Due to its high productivity and availability of water, many types of human activity have had a serious impact this vegetative zone, beginning with the early trappers and continuing through the 19th and 20th century activities of settlement, grazing, agriculture, construction of roads and water conveyance systems, as well as increasing urbanization.

Land Ownership and Administration

Administration and management of land in Utah is important because federal and state agencies may manage wildfire differently from one another. Indeed, even federal agencies differ from one another in how wildland fire is managed. Miller et al. (2012) note that, in the Yosemite National Park region, the National Park Service manages wildfire for resource (ecological) benefits whereas the USFS has adopted an immediate suppression strategy. The role of management strategy will be examined later in this report, but here we review the major land management agencies.

Like many states in the intermountain west, Utah's land ownership and administration is dominated by the federal government with relatively little ownership by private citizens (Table 2.2; Figure 2.3). The reasons for the large proportion of federal ownership are many and relate to the history of federal land disposal laws, the political economy of frontier era, and the suitability of land and water resources in Utah (and other western states) to provide the agricultural and mineral production needed for settlers to patent their land claims (Jakus et al., forthcoming).

Stambro et al. (Chapter 2, 2014) provide an excellent overview of the various land management agencies and their activities in Utah. The Bureau of Land Management (BLM) is the largest land manager in the state and is responsible for nearly 23 million acres. Much of BLM's land in Utah is relatively low elevation and composed primarily of semi-arid valleys and plateaus. Under the 1976 Federal Land Management and Policy Act, BLM manages its land for multiple use benefits, which can include management for grazing interests, mineral development, recreation activity, and preservation of wilderness.

Table 2.2: Utah Land Ownership and Administration

Entity	Acres	Share
Federal Agencies	35,019,955	64.5%
Bureau of Land Management	22,803,707	42.0%
US Forest Service	8,175,253	15.1%
US National Park Service	2,096,702	3.9%
Department of Defense	1,812,561	3.3%
US Fish and Wildlife Service	112,696	0.02%
Other Federal	19,001	0.003%
State Agencies	5,419,281	10.0%
School and Institutional Trust Lands	3,400,511	6.3%
Department of Natural Resources	2,015984	3.7%
Utah Department of Transportation	2,150	0.004%
Other State	636	0.001%
Private, county and Municipal	11,428,135	21.0%
Tribal	2,448,616	4.5%
Totals	54,315,952	100.0%

Source: Reproduced from Stambro et al. (2014)

The second largest land administration agency in Utah is the USFS, which manages more than 8.1 million acres. Most of this land lies at relatively high elevations and is covered by forests of various species depending on the ecoregion.. Five national forests (NF) are located wholly or predominantly in Utah: the Ashley NF in northeastern Utah, the Dixie NF of southern Utah, the Fishlake NF of central Utah, the Manti-La Sal NF of central and southeast Utah, and the Uinta-Wastach-Cache NF of northern Utah. Small portions of the Caribou and Sawtooth National Forests are also located in Utah. The USFS manages its land under the Multiple Use Sustained Yield Act of 1960 and the National Forest Management Act of 1976. In 1992 the USFS formally adopted an ecosystem approach to managing its multiple outputs. Like the BLM, the key products of USFS management are outdoor recreation and range resources, ecosystem services such as water quality and habitat for fish and wildlife, and timber harvest.

The National Park Service manages five national parks in Utah (Arches, Bryce, Canyonlands, Capitol Reef, and Zion), all of which are located in the band of red rock country spread across the southern portion of the state. In Utah the USNPS also manages six national monuments (Cedar Breaks, Dinosaur, Hovenweep, Natural Bridges, Rainbow Bridge, and Timponagos Cave), one national recreation area (Glen Canyon), and one national historical site (Golden Spike). All told the NPS manages nearly 2.1 million acres in the state. Unlike BLM and USFS, the NPS mandate is one of preservation of land for future generations but, in practice, this goal is balanced against the popularity of national park units as a source of recreation value and the associated commercial demands.

Two agencies manage nearly all of Utah's state-owned land. The School and Institutional Trust Land Administration (SITLA) manages 3.4 million acres of land, most of which had been ceded from the federal government at the time of statehood (1896). Established in 1994, SITLA has clear mandate for its land management efforts: to "…optimize and maximize trust land uses for the support of the beneficiaries over time."[3] Thus, SITLA's singular objective is to maximize revenues from its land. While its management actions remain governed by overarching environmental regulations such as the Clean Water Act and the Clean Air Act, its land management decisions are unencumbered by any complications associated with a multiple-use mandate or the need to satisfy ecosystem goals beyond compliance with standard environmental regulations. While SITLA's land is spread throughout the state, it is concentrated primarily in rural counties and interspersed mostly, but not exclusively, among land administered by the BLM.

The other large state land management agency is the Department of Natural Resources (DNR; 2.0 million acres). DNR is composed of the Division of Forestry, Fire, and State Lands (FFSL), the Division of Wildlife Resources, and the Division of State Parks and Recreation. FFSL, also established in 1994, is the largest land manger within the DNR and manages about 75% of DNR land. FFSL's objective is to manage its land under the public trust doctrine. FFSL interprets this doctrine as managing land to provide beneficial uses whilst satisfying long-term resource protection and conservation goals (Stambro et al. 2014, p. 88).

State lands tend to be interspersed with federal lands, which is the direct result of Utah 's Enabling Act; as a condition of statehood, the territory relinquished all lands not settled or otherwise claimed.[4] The federal government then returned Sections 2, 16, 32, and 36 of each township to the state for use of designated beneficiaries. Figure 2.3 illustrates the "checkerboard" nature of public land ownership in Utah. SITLA-administered that lands returned to state ownership under the Enabling Act can be seen in Figure 2.3 as the regularly recurring pattern of blue islands located within the larger domain of federally-administered land. The large concentrations of SITLA land are the result of subsequent land swaps. The most recent large state-federal land swap involved SITLA parcels located in the Grand-Staircase Escalante National Monument of Kane and Garfield counties that were traded for energy rich lands in Carbon, Emery, and Grand counties.

The intermingled public and private lands appearing in Figure 2.3 imply that large wildfires are likely to cross multiple administrative boundaries. For example, while the 2012 Seeley wildfire was located primarily on USFS land, the fire also burned on private land, state land, and land administered by the BLM. Where a wildfire ignites, the local pattern of land ownership, and the resources needed to manage the fire will all affect which agency (or agencies) respond to such a wildfire and how it is managed.

[3] Utah Administrative code R850-2-200 1-6, as quoted in Stambro et al. (2014, p. 72)
[4] Beginning in 1802 all territories seeking admission to the Union had similar enabling acts.

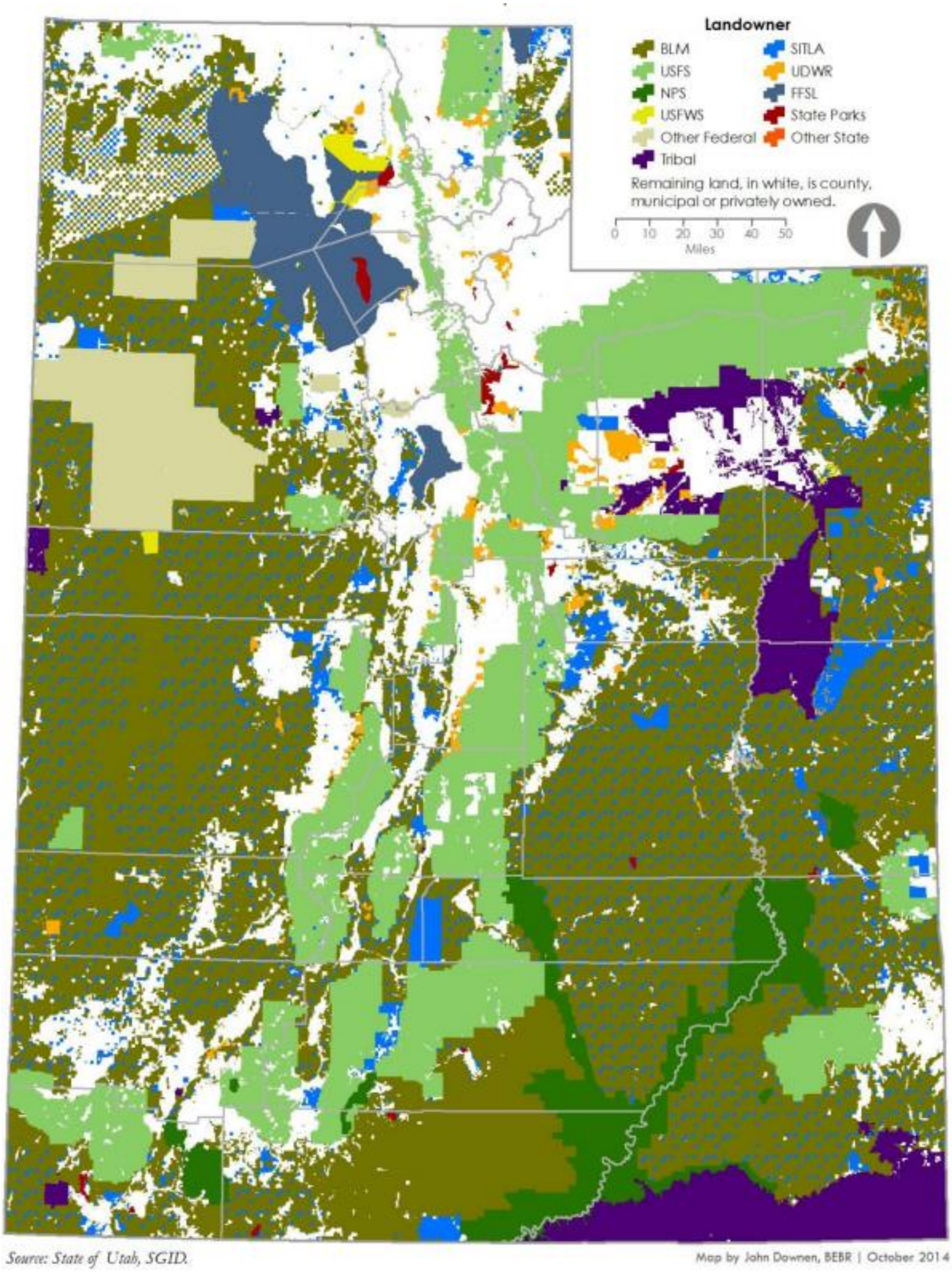


Figure 2.3: Land Administration in Utah

Source: Reproduced from Stambro et al. (2014)

References

Bailey, R.G. 1995. Description of the ecoregions of the United States. U.S. Department of Agriculture Forest Service Miscellaneous Publication No. 1391.

Bailey, R.G., 2009. Ecosystem Geography: from ecoregions to sites, 2nd edition. New York: Springer.

Banner, R.E. 1992. Vegetation types of Utah. *Rangelands*, 14(2):109-114.

Banner, R.E., B.D. Baldwin, and E.I. Leydsman McGinty, eds. 2009. Rangeland resources of Utah. https://extension.usu.edu/utahrangelands/files/uploads/RRU_Final.pdf

Gillies, R.R. and R.D. Ramsey. 2009. Climate of Utah. Section 5 in Banner, R.E., B.D. Baldwin, and E.I. Leysdman McGinty, eds. 2009. Rangeland resources of Utah. https://extension.usu.edu/utahrangelands/files/uploads/RRU_Final.pdf

Huntington Creek Watershed Plan. Undated. http://www.emerycounty.com/Appendix-F.pdf

Jakus, P.M. and five co-authors. Forthcoming, August 2017. Western public lands and the fiscal implications of a transfer to the states. *Land Economics*.

McAvoy, D., M. Kuhns, and J. Black. 2012. Utah forest types: an introduction to Utah forests. Utah state University Cooperative Extension Forest Fact Sheet NR/FF/011.

Miller, J.D., and five co-authors. 2012. Differences in wildfires among ecoregions and land management agencies in the Sierra Nevada region, California, USA. *Ecoregions*, 3(9):1-20. (Article 80) http://dx.doi.org/10.1890/ES12-00158.1

Stambro, J., and five co-authors. 2014. An analysis of a transfer of federal lands to the state of Utah. http://csee.usu.edu/htm/current-past-projects/an-analysis-of-a-transfer-of-federal-lands-to-the-state-of-utah/

CHAPTER 3: FIRE IN UTAH

Paul M. Jakus
Dept. of Applied Economics and Center for Society, Economy and the Environment
Utah State University

While ecoregions are quite useful in classifying ecosystems at a broad scale, Bailey (2009, p. 187) cautions against treating ecoregions as cleanly demarcated with regard to climate and vegetation, noting that the same vegetative type can appear in ecoregions that differ from one another in climate, soils, and topography. Thus, the ecosystem processes for the same species may differ across dissimilar ecoregions.

This distinction is important with regard to *fire regime*. A fire regime refers to the frequency, intensity, and pattern of wildfire on a landscape and subsequent effects on its ecosystem properties. Fire regimes are thus tightly bound to, and differ by, ecoregions. A species such as ponderosa pine is found in many western ecoregions with different fire regimes. *Fire-dependent* ecoregions are those in which fire of a specific type is needed to support native plants and wildlife (Bailey, 2010). Ecoregions that are fire dependent are often resilient and strengthened as they recover from wildfires of the appropriate periodicity and intensity.

Other ecoregions may be deemed as *fire-sensitive*. In general, these regions do not often experience frequent fire and vegetation does not have the ability to quickly respond to and recover from fire. Tropical rainforests are often held out as an example of a fire-sensitive ecoregion; Utah has no significant fire-sensitive ecoregions as defined by Bailey. Finally, other ecoregions are *fire-independent*—places where the lack of vegetation or ignition sources results in an absence of wildfire. In Utah, alpine vegetation zones are, for the most part, fire-independent.

Much of Utah is characterized by fire-dependent ecoregions, and wildfire plays an important part in sustaining the natural services delivered by an ecosystem. Thus, the fire regime that gives rise to an ecosystem must be maintained (Bailey, 2010). But natural fire regimes are often perturbed by the fire suppression efforts of federal and state land management agencies. Overly aggressive suppression efforts (extinguishing fires before they have reached their natural extent or intensity, or failing to manage fuel loads resulting in fires that burn too intensely or too extensively) can lead to wildfires that damage to ecosystem health. Here, wildfire departs from its natural regime and, when it occurs, can cause more harm than good. Other land management challenges, such as failure to control the spread of invasive species (such as cheatgrass) or pests (e.g., bark beetles), can also lead to departures from a natural fire regime.

Numerous federal and state agencies keep track of wildfire statistics for Utah, including the National Interagency Fire Center, the Bureau of Land Management, the US Forest Service, and Utah's Division of Forestry, Fire, and State Lands. Between 2002 and 2015, National Interagency Fire Center

Annual Reports have documented that nearly 18,000 documented wildfires have burned 2.5 million acres (Table 3.1). Over this 14 year period Utah has annually averaged 1,283 wildfires burning 178,437 acres. This works out to an average fire size of 138 acres.

Wildfires with burned areas equal to or greater than 5 acres make up only about 20% of all fires in any given year (or less) but usually account for over 95% of annual burned acreage. Fires of 1,000 acres or more often capture public attention as these fires generate wide-ranging noticeable effects such as smoke, sediment filled debris, road closures, recreation restrictions, and occasional evacuations of residents. In recent years Utah has experienced numerous fires in excess of 40,000 acres (Table 3.2).

Table 3.1: Total Wildland Acreage Burned in Utah, 2002-2015

Year	# of Fires	Total Acreage Burned
2002	1,243	237,427
2003	1,630	115,994
2004	1,530	76,654
2005	1,236	313,932
2006	1,844	340,572
2007	1,423	620,730
2008	999	28,490
2009	1,136	112,753
2010	1,050	64,781
2011	1,102	62,783
2012	1,534	415,267
2013	1,276	70,282
2014	1,035	28,255
2015	930	10,203
Totals	17,968	2,498,123

Source: National Interagency Fire Center Annual Reports

In 2007, Utah experienced its worst fire year on record, with over 600,000 acres burned. This total included the largest wildfire in Utah history, the 363,000 acre Milford Flat fire. Ignited on July 6, 2007 just northeast of Milford (in Beaver county), the fire burned 113,000 acres in Beaver county and 250,000 acres in Millard county before it was contained on July 19. The fire occurred in the lower elevations of the intermountain semidesert ecoregion province, and the burned vegetation was characterized by sagebrush and pinyon-juniper woodlands with the exotic and invasive cheatgrass providing surface fuel connectivity which undoubtedly contributed to fire spread.

Table 3.2: Wildland Fires Greater than 40,000 Acres in Utah, 2005 – 2015

Fire Name	Year	Size (Acres)	Estimated cost ($million)	Cause
Milford Flat	2007	363,052	$5.8	Natural
Clay Springs	2012	107,847	$6.9	Unknown
Westside Complex	2005	68,264	Not Reported	Natural
Jarvis	2006	50,738	$1.8	Human
Seeley	2012	48,050	$9.0	Natural
Wood Hollow	2012	47,387	$6.0	Unknown
Twitchell	2010	44,892	$18.9	Natural
Big Pole	2009	44,345	Not Reported	Natural
Neola North	2007	43,831	$9.1	Unknown
Dallas Canyon	2012	43,610	$2.0	Natural
Bull Complex	2006	43,571	$5.1	Natural

Note: Costs reported in constant 2015 dollars. Source: National Interagency Fire Center Annual Reports

While the Milford Flat fire accounted for nearly 60% of burned acreage in 2007, other heavy fire years in which relatively large amounts of acreage are burned can be characterized by numerous large fires. Some 415,000 acres burned in 2012, with four fires exceeding 40,000 acres in size (including the Seeley fire described in the prelude to this study).

Figure 3.1 and Figure 3.2 depict Utah's wildfire history over a longer time period (1992-2015), and restricts the data to only those fires >5 acres. These data are based on Short's 1992-2013 data (2015) and supplemented by recent data downloaded from the USGS National Fire Occurrence Data (2016).[5] In each of 1996, 2007 and 2012, wildfires burned over 400,000 acres across the state, but the average area burned in Utah based on the reference period 1992 to 1996 will have about 171,000 acres burned. As is evident in Figure 3.2, though, the variance in acreage burned across years is quite high: for example, burned acreage in 2007 was almost 61 times as large as that burned in 2015.[6]

Fire ignition and spread is highly dependent on fuel moisture. Fires are typically a late summer phenomenon, as Utah's soils lose moisture accumulated over the winter and spring. Seasonally warmer temperatures combine with the low summer precipitation common in Utah's ecoregions to dry vegetation and surface fuels. Splitting our 24-year wildfire history into eight-year increments, we can examine how seasonality in wildfire has changed in Utah (Figure 3.3). The eight-year segments are 1992-1999, 200-2007, and 2008-2015; coincidentally, each three time period contains one—and only one—of the three worst fire years (as measured by total acreage burned) over the aggregate time

5 The USGS database includes data collected from multiple agencies, each of which may report the same wildfire. This could happen if, for example, USFS fought fire on BLM-administered land (a double-count), or if both agencies provided personnel and resources to manage wildfire within a National Park (a triple-count). Short's database has been purged of all multiple records; we developed and implemented protocols to identify and remove multiple records for 2014 and 2015, the two most recent fire seasons for which complete information was available. We are in the process of purging multiple-counted fires in Utah for the 1984-1991 period.

6 The coefficient of variation (the ratio of the standard deviation divided by its average) for the 24-year period (1993-2015) is 0.97.

frame. No strong trend toward a longer fire season is immediately evident, though the 2000-2007 fire years seem more concentrated in the summer months. We performed a Kolmogorov-Smirnov test of equality between the cumulative monthly distributions of acreage burned for the 1992-1999 data against the distribution for the cumulative monthly average for the 2008-2015 data. The test failed to reject the null hypothesis that the cumulative monthly distributions were the same (p= 0.166). The power of this test—its ability to correctly reject the null hypothesis that the distributions are the same—is likely hampered by the relatively small sample size (eight years in each distribution). The double- and triple-counting of many fires in the pre-1992 period prevents us from using these data in the analysis as of this writing.

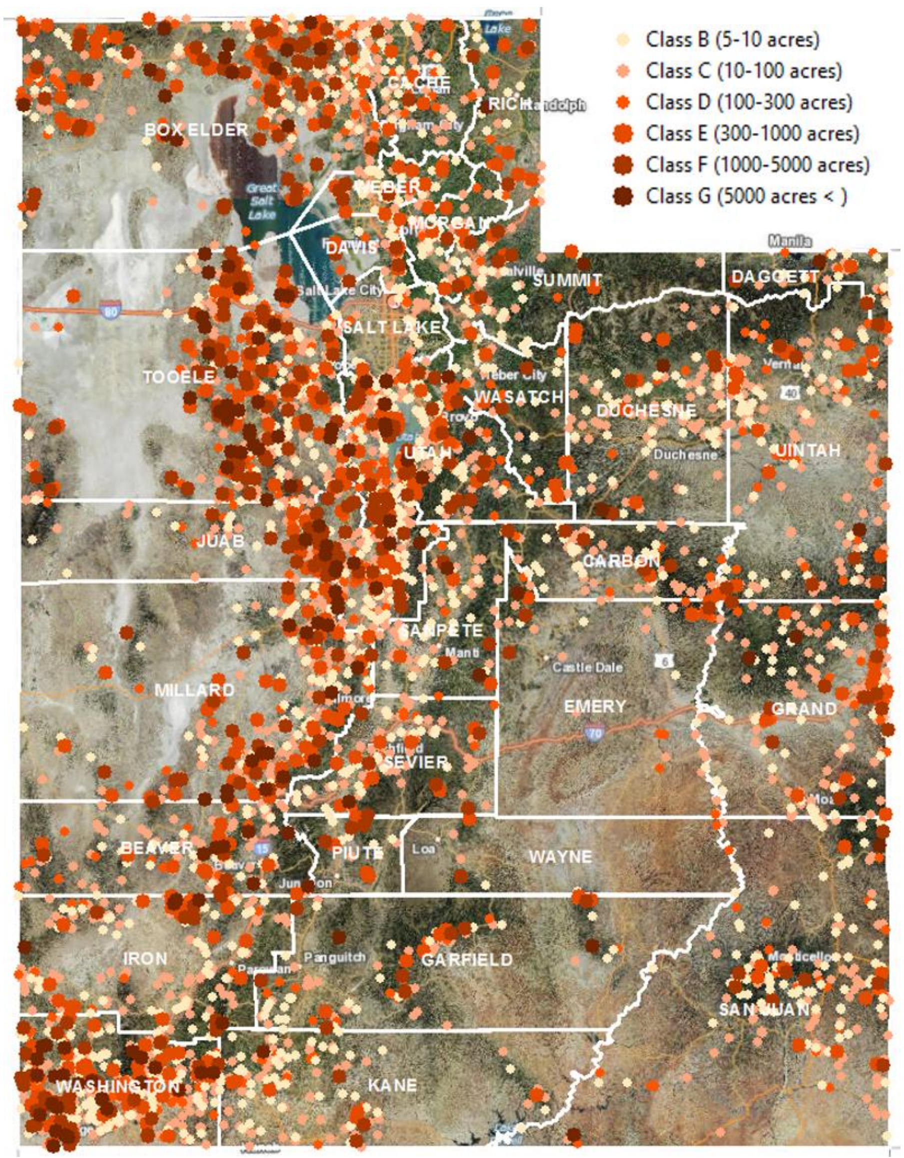


Figure 3.1: Wildfire Size and Ignition Point, 1993-2015 (Wildfires≥5 acres)

Sources: Short, K.C. (2015) and Federal Wildland Fire Occurrence Data

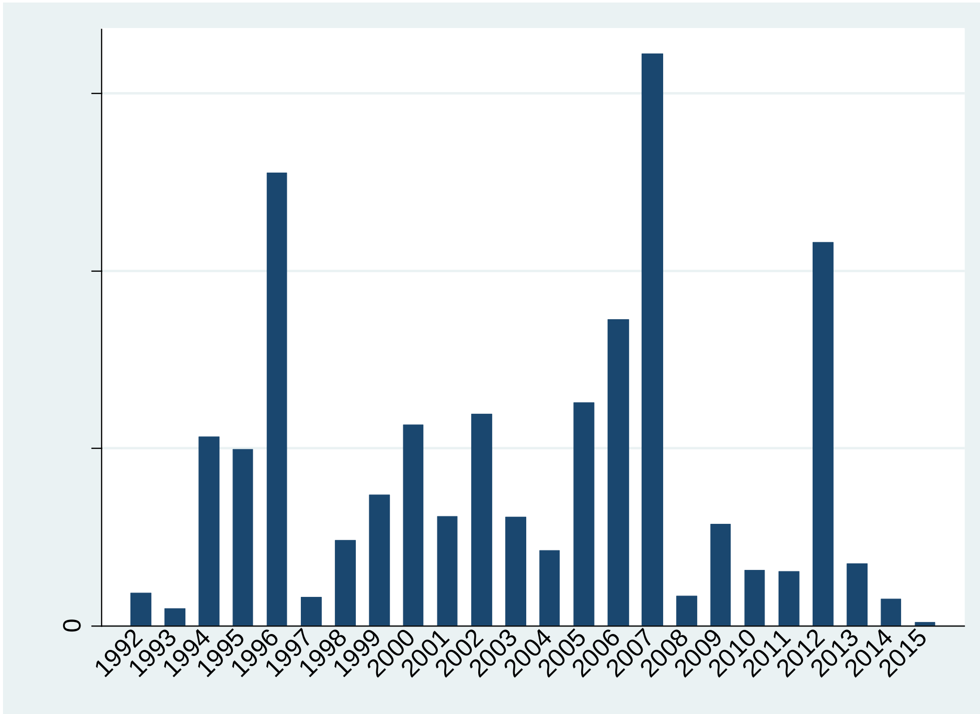


Figure 3.2: Acreage Burned, 1992-2015 (Wildfires ≥5 acres)

Sources: Short, K.C. (2015) and Federal Wildland Fire Occurrence Data

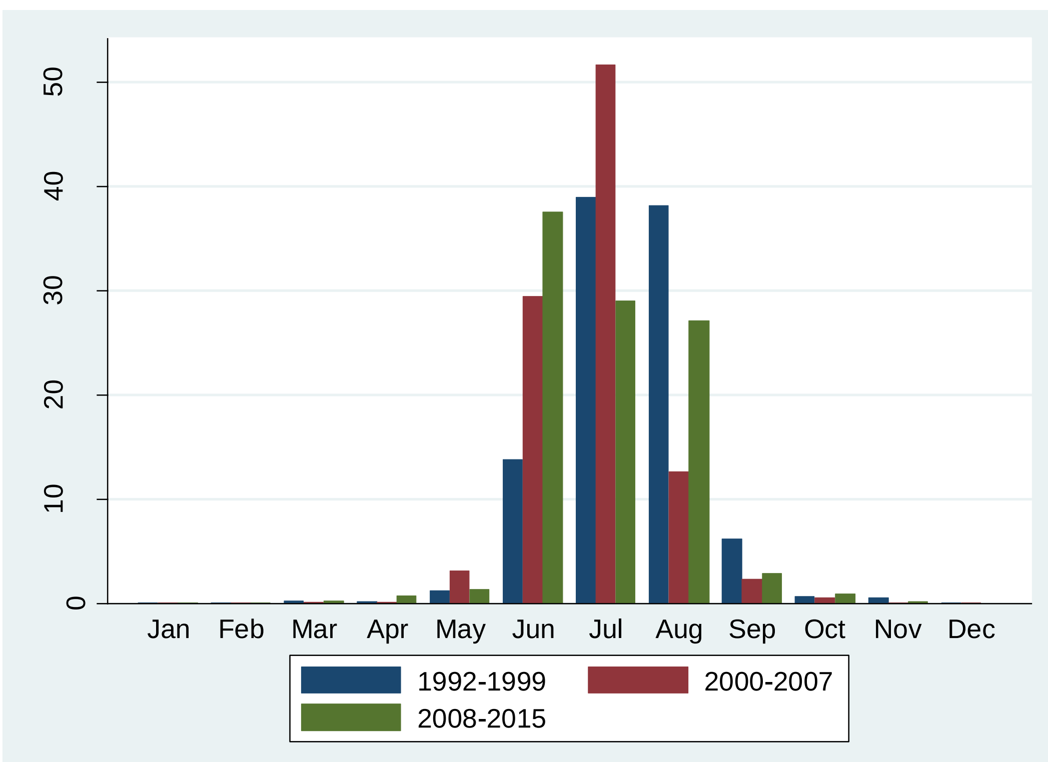


Figure 3.3: Proportion of Annual Acreage Burned, by Month (Wildfires≥5 acres)

Sources: Short, K.C. (2015) and Federal Wildland Fire Occurrence Data

Fire Intensity, Fire Severity, and Burn Severity

The number of wildfires and the total acreage burned are only two measures of possible damages caused by wildfire. In remarking upon the effects of wildfire on ecosystems, we have noted that different ecoregions have become dependent on wildfire to sustain ecosystem health. We have also stated that wildfires can differ in intensity and severity, and that wildfires whose intensity do not correspond to the natural fire regime can end up damaging rather than sustaining ecosystems. Thus far we have used these terms without the benefit of formal definitions.

Keeley (2009) notes that the terms *fire intensity* and *fire severity* refer to different things. Fire intensity is the amount of energy generated by a fire. Keeley discusses the various physical measures of energy release such as an energy flux (W per square meter) or heat transfer per unit length of the fireline (kW per meter), concluding that fire intensity is "…the energy released during the various phases of the fire and no single metric captures all of the relevant aspects of fire energy (p. 117)."

In contrast, *fire severity* refers to the effects caused to ecosystems by fire intensity. Keeley's (2009) review of the literature finds that most fire severity metrics have aimed at capturing the loss or decomposition of aboveground and belowground organic matter, which should correlate with above- and belowground heat pulses directly related to fire intensity. Fire severity can be measured according to a qualitative index or, using satellite imagery, by calculating differences in vegetation, which have found to have been relatively good correlation with loss of biomass. The success of such an approach, though, varies by vegetation type and ecosystem (Keeley, p. 119).

Burn severity is closely related to fire severity but will often distinguish fire-intensity effects on vegetation (above- and belowground loss of organic matter) separately from effects on soil.[7] Again, satellite imagery can be used to calculate an index called the *differenced Normalized Burn Ratio* (dNBR, discussed below) which many call a burn severity index; Keeley advocates calling this index the dNBR to distinguish it from onsite surface measurements of burn severity (p. 120). He notes that while dBNR correlates well with field-based measures of fire severity, it is not a good predictor of ecosystem response, which is the true measure sought by land managers.

Figure 3.4 summarizes the core of Keeley's analysis. Fire intensity is a measure of energy output, whereas fire severity and burn severity refer to the effects of fire intensity on vegetation and soil. These effects, in turn, feed into ecosystem response and recovery from fire. Fires that are too intense, or fires that cause extensive damage to soils and vegetation may fall outside the natural fire regime for an ecosystem, making it difficult for the ecosystem to recover from fire.

[7] Some metrics restrict the term *burn severity* to the effects of fire on soil.

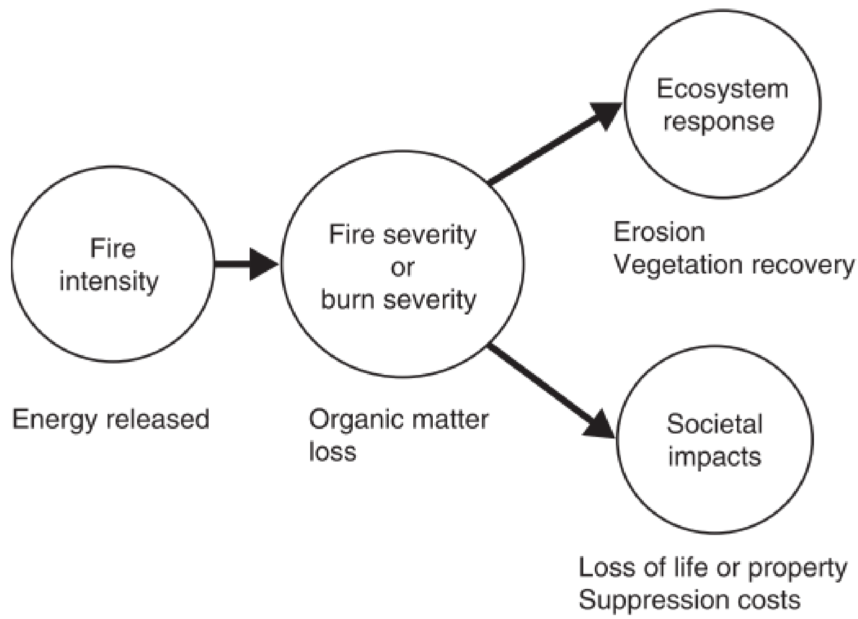


Figure 3.4: Fire Intensity, Burn Severity, and Ecosystem Response

Source: Reproduced from Keeley (2009)

A Closer Look at Burn Severity

The correspondence of remotely sensed data with fire severity led to a joint project between the US Forest Service and the US Geological Survey (USGS) to map the burn severity of every wildfire greater than 500 acres in the eastern U.S. and over 1000 acres in the western U.S. (Eidenshink et al. 2007). The goal of the project is to use satellite measurements to develop data, maps and reports to help land managers evaluate the severity of large wildfires. Data from the Monitoring Trends in Burn Severity (MTBS) project are available online for wildfires occurring from 1986 through 2014, and are updated annually.

The MTBS project defined burn severity as the "…degree to which a site has been altered or disrupted by fire; loosely, a product of fire intensity and residence time. (Eidenshink et al., p. 5)." Crucially, the outputs of the MTBS project focus on vegetative biomass and "…are not intended to be consistent with soil burn severity data…" collected from field-level evaluations conducted by the USFS and USGS. Thus, the burn severity metrics produced by MTBS depart from the metrics proposed by Keeley (2009) and others.

The MTBS burn severity metric is based on remotely sensed measurements of the near-infrared spectral range (0.76−0.90 μm wavelength, also known as the TM4 band) and the shortwave infrared spectrum (2.08−2.35 μm wavelength, also known as the TM7 band). Pre-fire and post-fire normalized burn ratios (NBR) are calculated using the equation below,

$$Normalized\ Burn\ Ratio = {(TM4 - TM7)}\Big/{(TM4 + TM7)}$$

For each pixel in an image, the pre- and post-fire NBR values are then subtracted to create the differenced normalized burn ratio (dNBR), the values of which are used to classify burn severity (Key 2006; Key and Benson 2006). Calculating the dNBR is straightforward; using a given dNBR value to assign a pixel to specific severity class relies heavily upon analyst interpretation in establishing the appropriate ecological severity thresholds. Further, it is sometimes difficult to use the dNBR measurement to identify the edges of wildfires. This is particularly true of wildfires in western landscapes (Eidenshink 2007, p. 11). The MTBS employs a consensus approach, with numerous analysts rating specific fires to develop threshold values for dNBR that correlate with burn severity. Threshold values from the reference wildfire deemed most appropriate to are then applied in analysis of subsequent wildfires "…occurring in similar conditions (p. 13)".

The threshold levels of satellite surface reflectance that are used to demarcate low, moderate, and high severity inherently depend on the vegetation and soil of a particular region, and sometimes on the land use history (harvest, previous burns, or recent beetle outbreak). The values provided by the MTBS program have not been calibrated for Utah's ecoregions, and therefore are presented as an approximation. However, the thresholds used by MTBS can be sensitive to specific ecosystem features.

Tables 3.3 provides data representative of data available from the MTBS project, whereas According to the evaluation protocols established by the MTBS project, forests and shrublands were the vegetative types that burned most severely during the 1992-2014 period. Of the over one million acres of forested land located within the perimeters of large Utah wildfires (≥1000 acres), some 221,000 acres (21%) burned with high severity. In contrast, more than twice as much shrubland was burned over the same time period, yet only 5% (114,000 acres) burned at high severity.

The MTBS data can be rearranged over time to examine possible changes in the type of vegetation subjected to high severity burns. Table 3.4 depicts high severity acreage by vegetative type, with the 1992-2014 time period is divided into three eight-year increments.. From 1992-1999, shrublands were the vegetation type making up the bulk of the 86,305 acres classified as having burned with high severity (69.1%). During the 2008-2014 period forests made up 96.1% the 77,772 acres classified as high severity burns.

Figure 3.5 shows the MTBS burn severity map for the 2012 Seeley fire in Carbon and Emery counties. MTBS estimates that one-third of the fire's acreage (14,944 acres) burned at high severity. In addition to the burn severity categories reported in Table 3.3, the map also lists two other categories reported by the MTBS data. The *increased greenness* category is difficult to interpret: it may result from special vegetation types, or may simply be a poor match for pre- and post-fire pixels. The *non-processing area mask* category refers to acreage that could not be assessed for burn severity due confounding influences; the primary problem with the satellite imagery is cloud cover.

Table 3.3: Burn Severity by Vegetative Type, Utah 1992-2014[a] (Acres)

Vegetative Type	High Severity	Moderate Severity	Low Severity	Unburned to Low Severity	Row Total
Shrubland	114,331	488,797	1,061,622	504,262	2,169,012
Forest	221,300	329,802	286,386	208,301	1,045,789
Herbaceous, Natural	23,111	139,370	275,696	79,949	518,126
Herbaceous planted/Agriculture	2,768	9,818	16,109	6,646	35,341
Developed	476	3,196	8,164	6,537	18,373
Barren/Sparsely Vegetated	327	578	2,111	5,209	8,225
Wetlands	642	1,348	4,020	1,410	7,420
Other	8	25	1456	119	297
Total	362,963	972,934	1,654,253	812,433	3,802,583

[a]MTBS data restricted to wildfires≥1000 acres *Source: MTBS.gov*

Table 3.4: High Severity Acreage Burned by Vegetative Type and Year [a]

	1992-1999		2000-2007		2008-2014	
Vegetative Type	Acres Burned, High Severity	% of High Severity Acreage	Acres Burned, High Severity	% of High Severity Acreage	Acres Burned, High Severity	% of High Severity Acreage
Shrubland	59,664	69.1%	51,947	26.1%	2,721	3.5%
Forest	11,396	13.2%	135,187	68.0%	74,718	96.1%
Herbaceous, Natural	12,323	14.3%	10,675	5.4%	112	0.1%
Herbaceous planted/Agriculture	2,331	2.7%	394	0.2%	43	0.1%
Developed	166	0.2%	231	0.1%	79	0.1%
Barren/Sparsely Vegetated	279	0.3%	45	0.0%	3	0.0%
Wetlands	146	0.2%	400	0.2%	96	0.1%
Other	0	0.0%	8	0.0%	0	0.0%
Total	86,305	100.0%	198,887	100.0%	77,772	100.0%

[a]MTBS data restricted to wildfires≥1000 acres *Source: MTBS.gov*

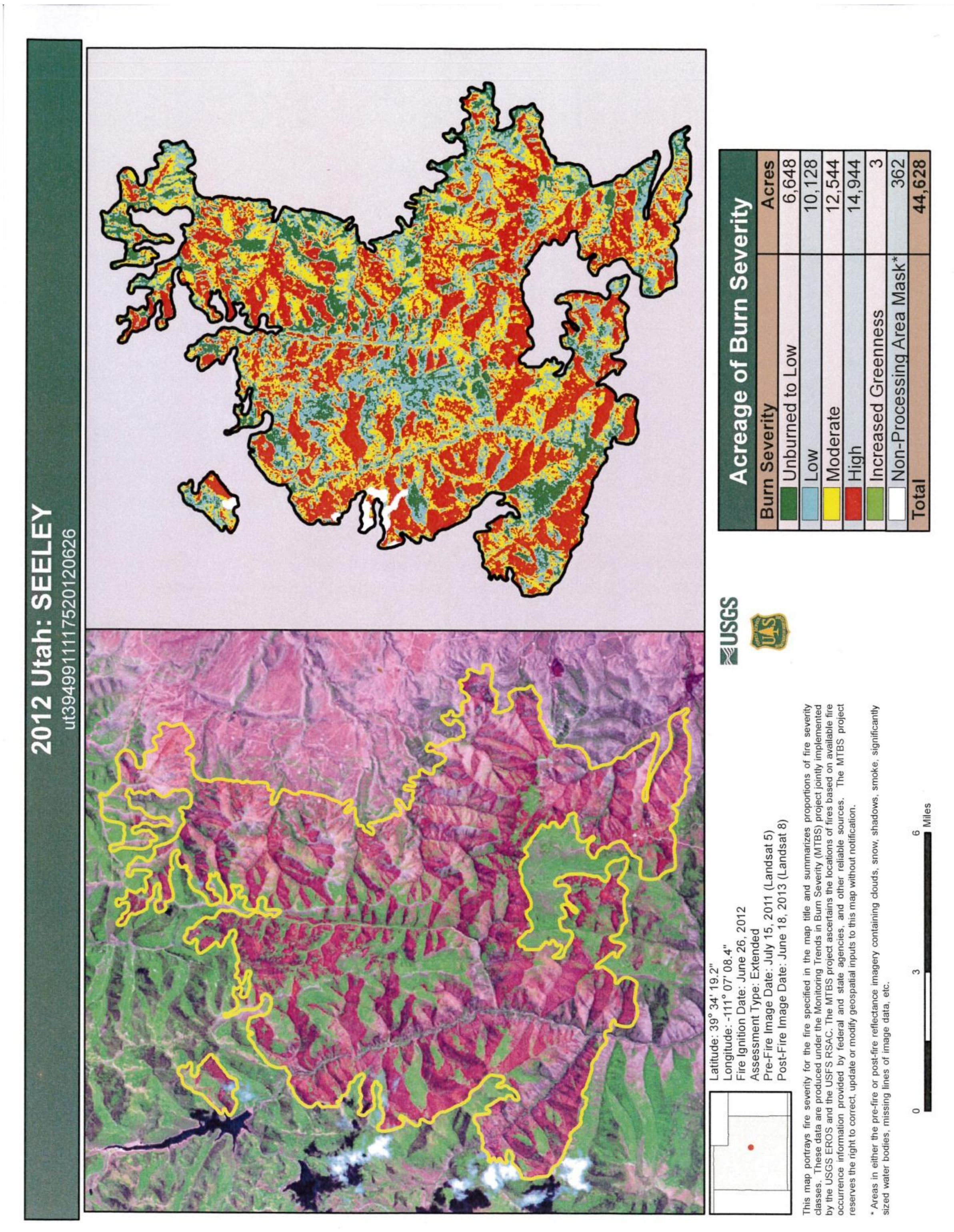


Figure 3.5: Burn Severity Map for the Seeley Wildfire (2012) Source: MTBS.gov

The MTBS database also permits us to examine burn severity by land ownership. Similar in construction to Table 3.4, Table 3.5 presents burn severity by land administration and by time period, where our 1992-2014 time period is divided into three eight-year increments. Miller et al. (2012) have noted that land management agencies may adopt different wildfire management strategies such that an analysis might allow us to examine if these alternative approaches have changed the burn severity of wildfire over time.

Table 3.5: High Burn Severity by Land Administration and Year[a]

			1992-1999		2000-2007		2008-2014	
Management Agency	Acres Managed (1000)	% of state total	Acres Burned, High Severity	% of High Severity Acreage	Acres Burned, High Severity	% of High Severity Acreage	Acres Burned, High Severity	% of High Severity Acreage
BLM	22,809	42.0%	39,654	45.9%	88,708	44.6%	9,178	11.8%
USFS	8,180	15.1%	13,828	16.0%	60,281	30.3%	60,788	78.2%
Other Federal	4,042	7.4%	2,720	3.2%	3,536	1.8%	2,221	2.9%
Tribal	2,450	4.5%	818	0.9%	6,172	3.1%	1	0.0%
State	5,421	10.0%	625	0.7%	1,914	1.0%	139	0.2%
Private	11,433	21.0%	28,660	33.2%	38,276	19.2%	5,448	7.0%
Total	54,335		86,305		198,887		77,775	
Federal, Total	35,031	64.5%	56,202	65.1%	152,525	76.7%	72,187	92.9%

[a]MTBS data restricted to wildfires≥1000 acres   *Source: MTBS.gov*

During the 1992-1999 time frame, the percentage of total acreage that burned with high severity roughly corresponded to the amount of acreage administered by land management agencies in the state. For example, BLM manages 42% of the state's land and accounted 46% of the high burn severity acreage; similarly, the USFS manages 155 or the state's area and accounted 16% of its high burn severity. All told, the proportion of federal land managed in Utah almost exactly equaled the proportion of severely burned land.

In contrast, the 2008-2014 period shows that the USFS has seen its share of high severity burned acreage grow to 78% of all high severity acreage in the state. The proportion of total high severity acreage associated with federal land administration has grown from 65% in 1992-1999 to almost 93% for acreage burned between 2008 and 2014. The data appear to suggest that federal wildfire management policies have, over time, resulted in more severe burns than management by state or local agencies. In fact, the data cannot be interpreted so simply.

Criticisms of the MTBS Burn Severity Classification Methodology

The semi-automated process of developing burn severity metrics by the MTBS project has generated a wealth of information about wildfire across the U.S. The dNBR measurements—calculated from Landsat images—are known to be correlated with burn severity, but concerns remain with regard to the reliance on analyst applications of dNBR thresholds to classify the burn severity categories for any given wildfire.

Kolden, Smith and Abatzoglou (2015) note that MTBS was developed for management needs and MTBS outputs (such as burn severity maps and acreages) have not undergone field validation for the vast majority of wildfires in the MTBS database. Among the concerns are (i) inaccuracies in mapping fire perimeters, (ii) there is no adjustment for seasonality (phenology offset) in the dNBR measurement, (iii) burn severity thresholds are subjective and highly variable, and (iv) the "…classification thresholds are neither ecologically quantified nor field validated.

While analyst interpretation of the dNBR values provides a general sense of burn severity, the thresholds used to assign a pixel to its particular burn severity class are subjective and highly variable. Thus, the resulting burn severity data do not enjoy a foundation strong enough for empirical work. Further, the dNBR metric is not as sensitive as other metrics (such as the relativized dNBR, or RdNBR) in detecting pre- and post-fire spectral differences in ecosystems with lower pre-fire vegetation densities common in much of semi-arid Utah.

Sparks et al. (2015) examined four fires as part of an assessment of the MTBS products, testing the MTBS wildfire classification against those developed using alternative methodologies. All of the fires were located in the Great Basin and had similar fire regimes. Two of these fires occurred in Utah; the burn severity map for one of those fires, the 2006 Hogups fire that burned just over 30,000 acres, is shown in Figure 3.6. The authors found that MTBS fire perimeters consistently overestimated fire size by 4% to 16.8%. The Hogups wildfire had the greatest error in fire size, and should have been assigned a size of less than 25,000 acres. The key problems were the inclusion of unburned areas and 'fingers' of land extending into the perimeter defined by MTBS.[8] Further, the problems encountered in discriminating between burned versus unburned areas raise similar concerns for thresholds used to delineate other classes of burn severity for a wildfire.

Perhaps the most important caveat about using categorized MTBS data is that the severity thresholds have not been calibrated for Utah's vegetation. Established field methods allow for such calibration of fire severity (Thode et al. 2011).

[8] Meddon, Kolden and Lutz (2016) and Kolden et al. (2016) note the importance of unburned areas within the perimeter of a wildfire because such regions can act as ecological refugia for wildlife and as a seed source for post-fire recovery.

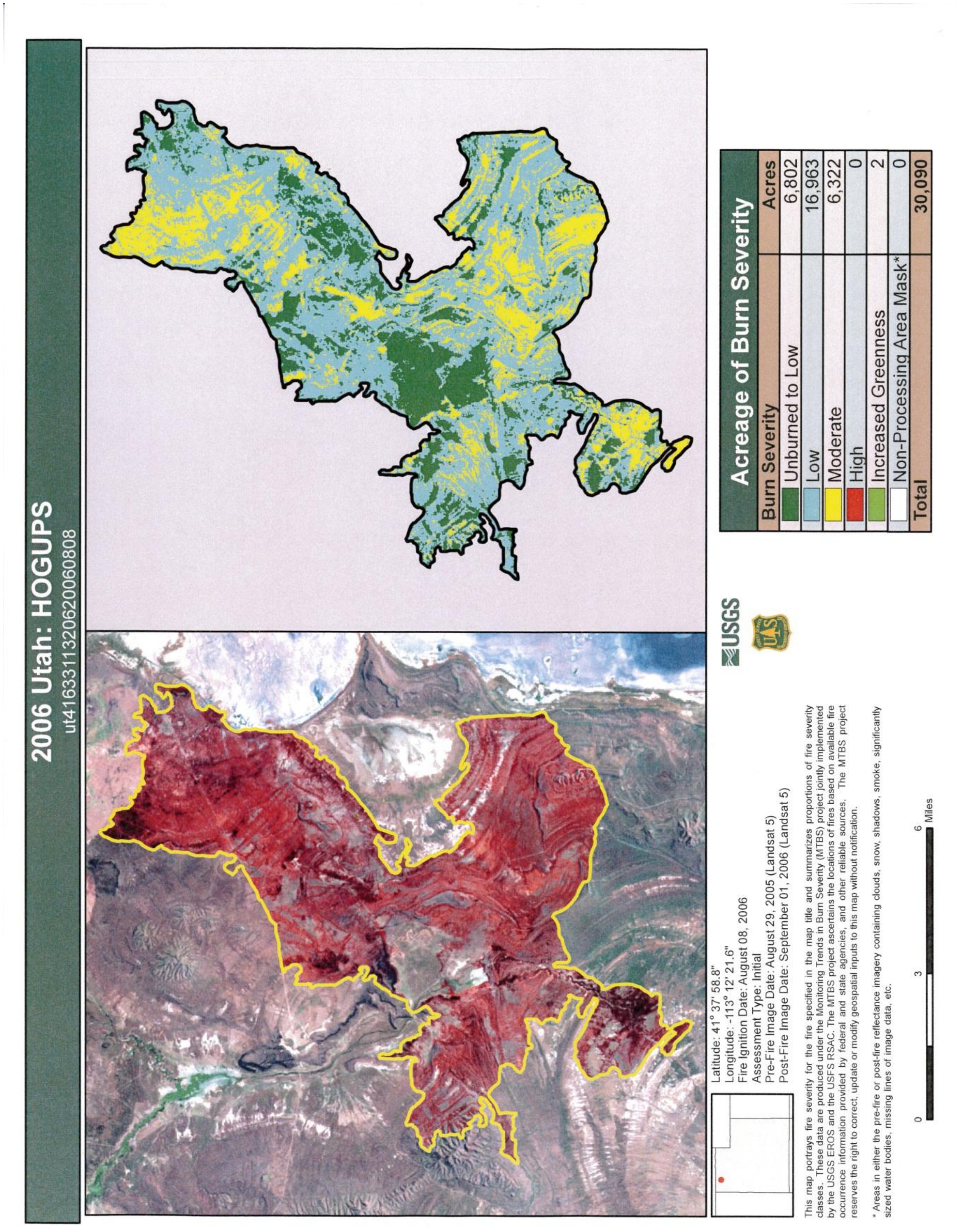


Figure 3.6: Burn Severity Map for the Hogups Wildfire (2006) *Source: MTBS.gov*

Summary

Wildfire in Utah is quite common. The state averages about 1,300 wildfires each year with annual acreage burned approaching 170,000 acres. Burned acreage is highly variable, though. In just the past decade total wildfire acreage has ranged from a relatively paltry 10,200 acres (in 2015) to well over 600,000 acres (in 2007). Acreage burned provides information about the extent of wildfire but the metric does not convey anything about the intensity and severity of wildfire. Wildfires can be quite beneficial to ecosystems if they conform to the fire regime that gave rise to the ecosystem. Wildfires that depart from the fire regime appropriate to an ecoregion can cause great damage to ecosystem health if they burn too intensely or continue for too long a period. Fire scientists have proposed numerous methods to capture the burn severity of wildfire; the most widely available information on burn severity is from the Monitoring Trends in Burn Severity (MTBS) project.

MTBS restricts its analysis of western wildfires to only those greater than 1,000 acres, and often smaller fires contribute a considerable portion to annual area burned. To measure of whether the area burned at high severity is increasing in Utah a program of reanalysis of Utah fires over the period of the Landsat data record would be required. Further, the methodology by which burn severity metric is produced by MTBS has been subject to criticisms. While there is little debate amongst scientists about the applicability of the dNBR and the RdNBR measures used by MTBS, the application of standardized threshold values for classifying burned pixels into burn severity classes by MTBS has proved problematic. Instead, determining the appropriate thresholds to classifying burn severity should be done based on ecosystem-specific field validation.

References

Bailey, R.G. 2010. Fire regimes and ecoregions. In, Miller, W.J., S. Ina, and L. Audin, eds. Cumulative watershed effects of fuel management in the western United States. Gen. Tech Rep. RMRS-GTR-231. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. pp.7-18.

Banner, R.E., B.D. Baldwin, and E.I. Leydsman McGinty, eds. 2009. Rangeland resources of Utah. https://extension.usu.edu/utahrangelands/files/uploads/RRU_Final.pdf

Eidenshink, J. and five co-authors. 2007. A project for monitoring trends in burn severity. *Fire Ecology* 3(1):3-21.

Keeley, J.E. 2009. Fire intensity, fire severity, and burn severity: a brief review and suggested usage. *International J. Wildland Fire*, 18:116-126.

Key, C.H. 2006. Ecological and sampling constraints on defining landscape fire severity. *Fire Ecology* 2(2): 178-203. doi: 10.4996/fireecology.0202178

Key, C.H., and N.C. Benson. 2006. Landscape assessment: ground measure of severity, the Composite Burn Index, and remote sensing of severity, the Normalized Burn Ratio. Pages LA1-LA55 in FIREMON: fire effects monitoring and inventory system. D.C. Lutes, R.E. Keane, J.F. Caratti, C.H. Key, N.C. Benson, S. Sutherland, and L.J. Gangi. USDA Forest Service General Technical Report RMRS-GTR-164CD. Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Kolden, C, A.M.S. Smith and J.T. Abatzoglou. 2015. Limitations and utilization of Monitoring Trends in Burn Severity products for assessing wildfire severity in the USA. *International J. Wildland Fire*, http://dx.doi.org/10.1071/WF15082

Kolden, C.A., and four co-authors. 2012. Mapped versus actual burned area within wildfire perimeters: characterizing the unburned. *Forest Ecology and Management*, 286:38-47. http://dx.doi.org/10.1016/j.foreco.2012.08.020

Meddens, A.J.H., C.A. Kolden and J.A. Lutz. 2016. Detecting unburned areas within wildfire perimeters using Landsat and ancillary data across the northwestern United States. *Remote Sensing of Environment*, 186:275-285. http://dx.doi.org/10.1016/j.rse.2016.08.023

Miller, J.D., and five co-authors. 2012. Differences in wildfires among ecoregions and land management agencies in the Sierra Nevada region, California, USA. *Ecoregions*, 3(9):1-20. (Article 80) http://dx.doi.org/10.1890/ES12-00158.1

Short, K.C. 2015. Spatial wildfire occurrence data for the United States, 1992-2013 [FPA_FOD_20150323]. 3rd Edition. Fort Collins, CO: Forest Service Research data Archive. http://www.fs.usda.gov/rds/archive/Product/RDS-2013-0009.3/

Sparks, A. and five co-authors. 2015. An accuracy assessment of the MTBS burned aresa product for shrub-steppe fires in the northern Great Basin, United States. *International J. Wildland Fire*, http://dx.doi.org/10.1071/WF14131

Stambro, J., and five co-authors. 2014. An analysis of a transfer of federal lands to the state of Utah. http://csee.usu.edu/htm/current-past-projects/an-analysis-of-a-transfer-of-federal-lands-to-the-state-of-utah/

Styler, M. 2012. 2012 Fire suppression and restoration costs. Presentation to Interim Session of the Utah Legislature. http://le.utah.gov/interim/2012/pdf/00001083.pdf

Thode, A.E., and three co-authors. 2011. Quantifying the fire regime distributions for severity in Yosemite National Park, California, USA. *International J. Wildland Fire*, 20:223-239.

CHAPTER 4: WILDFIRE RISK MODELING

Edd Hammill
Department of Watershed Sciences
Utah State University

Introduction

Estimating fire threat

Estimating the probability of a land parcel burning is a complex and laborious process due to the dynamic nature of wildfires. As a consequence of the difficulties associated with generating a specific fire risk estimate for the state of Utah, we opted to use the Fire Threat Index (FTI) contained within the West Wide Wildfire Risk Assessment (Sandborn Map Company 2013). The West Wide Wildfire Risk Assessment (WWRA) was commissioned in 2007 by the State of Oregon Department of Forestry, and the final report was completed in 2013. The final report contains fire threat data for the 17 Western states including Alaska and Hawai'i. The amount of data and effort included in the production of the report means that it currently represents the most comprehensive and robust estimate of Fire Threat for the region. The Fire Threat Index was a central output from the WWRA, and contains spatially-explicit probabilities of different land parcels burning at a fine scale spatial resolution (30m*30m).

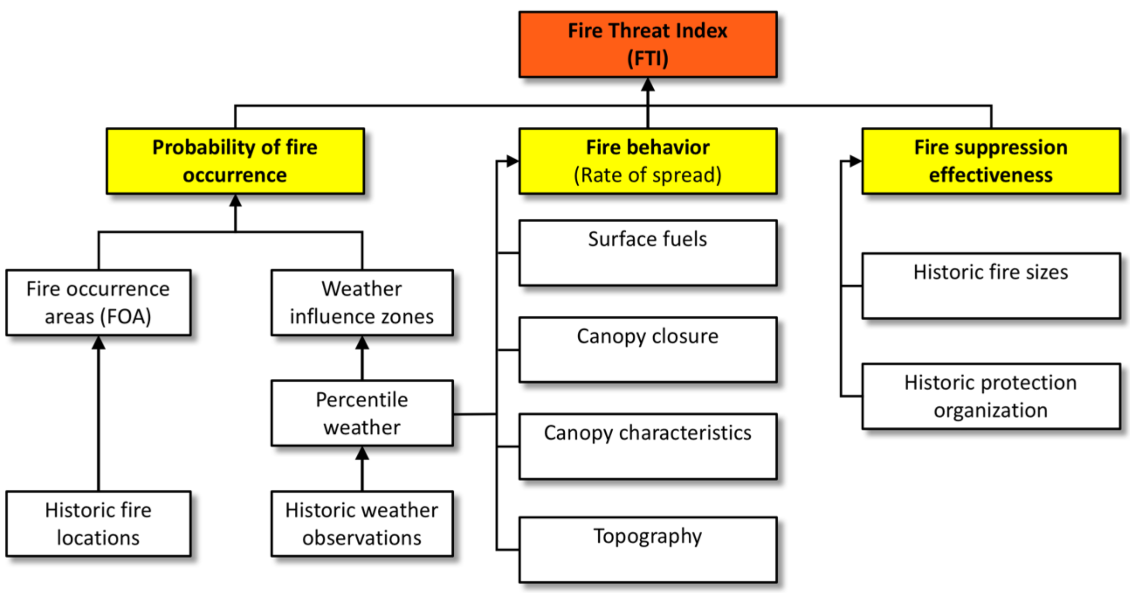


Figure 4.1: Data inputs used to generate the Fire Threat Index (FTI). The FTI represents a spatially precise estimate of the risk of a pixel (30m*30m spatial resolution) burning. Figure is adapted from the West Wide Wildfire Risk Assessment (Sandborn Map Company 2013).

Within the FTI, the probability of a fire occurring within a single 30m*30m pixel is essentially driven by three different processes: the probability of a fire igniting within the pixel ("probability of fire occurrence"), the likelihood of a fire spreading to a focal pixel from adjacent pixels ("Fire behavior"), and the likelihood of suppression activities being successful ("Fire suppression effectiveness"). Within these three main processes, a total of 11 different spatial explicit data layers are then used to calculate the eventual Fire Threat Index (Figure 4.1). The FTI can then predict the number of acres likely to be burned annually through Monte Carlo simulations (further details in the methods).

Benefits of the Fire Threat Index

The greatest positive aspect of the FTI is the amount of data and effort that were used in its calculation. Producing the FTI required 11 spatially-explicit data input layers, and took an experienced team of experts seven years to complete. Given the effort expended on the generation of the FTI, it would have been impossible to produce from scratch a better bespoke fire threat estimate for the current report.

A second benefit of the Fire Threat Index is that it already accounts for spatial auto-correlation amongst the 30m*30m pixels, an important consideration for spatial data (Hawkins 2012). The inclusion of "Fire Behavior" as a core predictor within the FTI means that it already includes the probability of a fire spreading into a focal pixel from elsewhere. As the chance of fire spreading from adjacent pixels has already been included in the FTI calculations, when using the FTI we can treat each pixel as independent when estimating its state (either burned or not) using Monte Carlo simulations.

Limitations of the Fire Threat Index, and Methods to Overcome

The FTI has two limitations that require addressing. The first is that it makes no calculations or estimates of the severity of fires that occur on different land parcels.[9] Due to this inability to estimate fire severity, in all of our calculations we opted for a "worst case scenario", meaning that if a 30m*30m pixel was determined to have been burned, all vegetation contained within the pixel was designated as lost, and all drinking water present within the pixel was affected.

The second limitation of the FTI is that it relies on some regional-scale parameters to generate local-scale fire threat estimates. The most pertinent of these regional-scale parameters for the current study is the inclusion of regional-scale "Weather Influence Zones" (WIZ). These WIZs can result in Fire Threat estimates being inflated or reduced at the local scale due to the influence of extremely high or extremely low risk areas that are also present within the WIZ. In the present analysis, comparison of fire threat values to historic burn rates indicated that the Fire Threat Index was over-estimating the probability of fires occurring within our focal area. As a consequence, we specifically contacted the authors of the WWRA and obtained scaling values to make our local-scale estimates of the Fire Threat more realistic (see methods for details).

Areas used in the analysis

We opted to use to two Clusters of counties within our analyses. These clusters were selected based on the range of different land uses and ecoregions they encompass, and represent many of the different land types contained in the state of Utah. The four-county Urban cluster (Davis, Morgan, Salt

---

[9] See Chapter 3 for discussion of fire severity measures.

Lake and Weber counties) totals over 1.7 million acres and accounts for 3.2% of the state total area. In contrast with the overall pattern of land ownership and administration in Utah (Table 2.2), the Urban cluster is dominated by private land ownership (64% versus 21% for the state). The Urban cluster contains some of the major metropolitan areas of the Wasatch Front, including Salt Lake City, West Valley city, West Jordan, Ogden, Layton, Taylorsville and South Jordan. The cluster contains the majority of the state's population and has a population density of 763 persons per square mile (294 persons per square km).[10] The USFS is the largest federal land owner while ownership by the School Lands and Institutional Trust Agency is minimal. Nearly all state lands in the Urban cluster are composed of state sovereign lands, wildlife refuges and state parks.

Table 4.1: Land Ownership and Administration in Wildfire Study Clusters

	BLM	USFS	State	Private	Tribal
Urban Cluster	0.2%	12.2%	22.8%	64.2%	0.4%
Davis	0.1%	9.6%	64.9%	24.0%	0.0%
Morgan	0.2%	4.2%	2.6%	93.0%	0.0%
Salt Lake	0.4%	19.5%	6.0%	73.2%	1.4%
Weber	0.0%	13.1%	21.7%	65.2%	0.0%
Rural Cluster	49.1%	15.9%	7.6%	25.4%	1.4%
Juab	66.0%	5.4%	8.4%	17.3%	2.1%
Sanpete	13.2%	38.2%	5.9%	42.6%	0.1%

Note: The Department of Defense (DoD) administers 6,233 acres in the Urban cluster; DoD and the US Fish and Wildlife Service administer 18,182 acres in the Rural cluster. No National Park units are present in either cluster. Source: Banner et al (2009)

The two-county Rural cluster (Juab and Sanpete counties) is roughly twice the size of the Urban cluster, encompassing 3.2 million acres, or about 5.9% of Utah's land area. The rural region is more representative of the pattern of land ownership in the state. Population density in the Rural cluster is much lower than the Urban cluster, at six persons per square mile (2.4 persons per square km). The dominant landowner is the federal government, led by the Bureau of Land Management (49% in the Rural cluster versus 42% for the state as a whole) and the US Forest Service (just under 16% for the cluster against 15% for the state). State-owned land (7.6%) is dominated by SITLA administration, just as it is for the state as a whole (10%). Private land ownership is slightly overrepresented (25% vs. 21%) whereas land administered by tribal authorities is underrepresented relative to the state (1.4% vs. 4.5%). We therefore believed that the Rural cluster would be a representative indicator of the impact of wildfire in the rural regions of the state.

10 Population density is based on land area; some 330 square miles of Davis county is covered by the Great Salt Lake.

Methods

Spatial extent

Due to computational limitations, we opted to limit our estimation of acres burned to the pre-selected group of counties described above. The WWRA fire risk data for the state of Utah were then clipped to these two different spatial extents (Figures 4.2a and 4.2b).

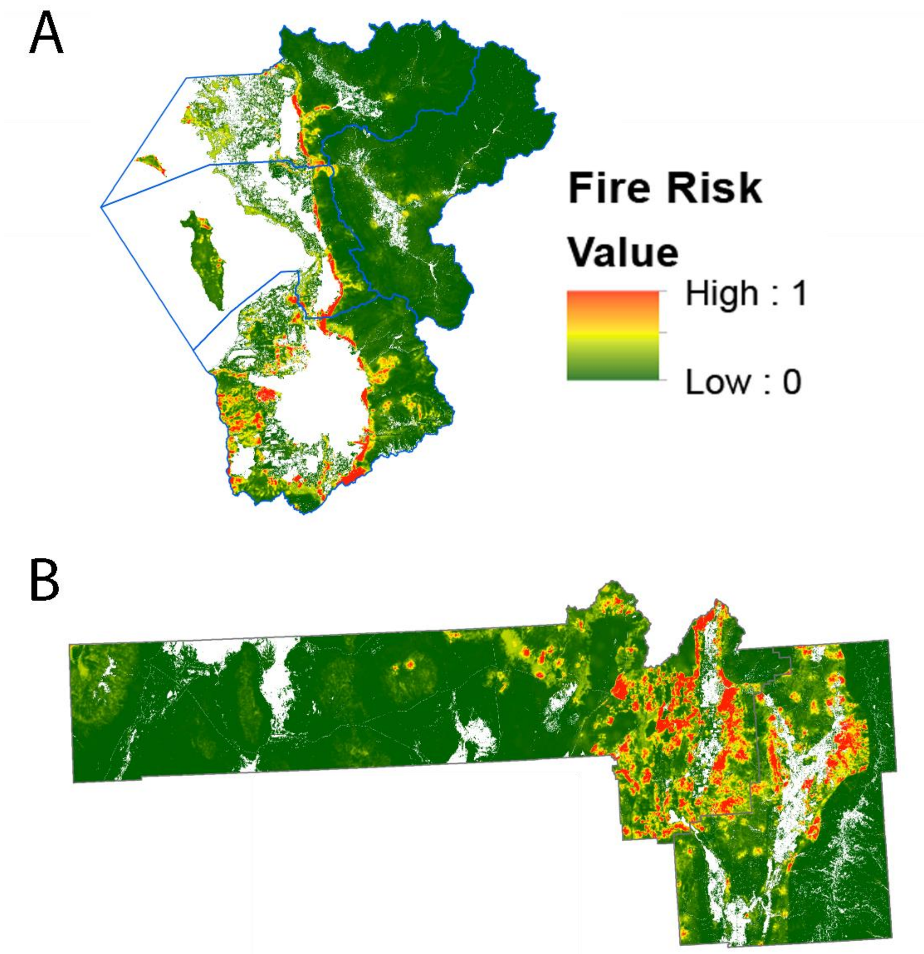


Figure 4.2: Spatial extents and raw fire risk values used in the analyses. **A.** Urban Cluster, consisting of Weber, Morgan, Davis and Salt Lake counties. **B.** Rural Cluster, consisting of Juab and San Pete counties White areas indicate locations with no risk of wildfire (open water, urban areas)

Fire Threat Estimates

All our analyses employed the Fire Threat Index (FTI) contained in the Western Wildfire Regional Assessment (WWRA), commissioned by the Oregon Department of Forestry (Sandborn Map Company 2013). This estimate employs a suite of variables including history of fire, land cover, surface fuels, topography and weather patterns (the full list of metrics used are given in Appendix 1 of the WWRA). The suite of predictive variables are then combined to estimate the likelihood of an acre burning. Crucially, the WWRA includes a spatial autocorrelation component. This means that for every pixel within the fire risk estimate, the likelihood of the adjacent pixels being on fire or not is incorporated into the likelihood of a focal pixel burning. We opted to use the WWRA fire threat estimate due to the number of different variables used in its calculation, its spatial precision, and the robustness of the analysis used in its generation. However, it is worth noting that the WWRA fire threat index represents a median value, where a weighted average of weather effects, differences in potential management regimes, and suppression possibility has been used. Therefore, much of the potential variation due to annual climatic differences or management efforts has been lost, and throughout we use the median value of fire threat.

Distribution of the factors influencing fire threat on Federal and State lands

In order to understand how different physical landscape attributes affect fire threat, we produced a logistic regression model to investigate how different vegetation cover types, slope, elevation, and aspect affect risk of fire. We then compare the distribution of these different physical attributes on state and federal lands. Logistic regression models are specifically designed to analyze how different factors alter probabilities of an event occurring.

Re-scaling the WWRA data to best predict Utah

While the fire risk data obtained from the WWRA gives spatially explicit estimates of the likelihood of fires occurring. However, these data are influenced by land areas outside the state of Utah, and scaled according to occurrences of fires outside of the state and weather influence zones (Figure 4.3). Initial comparisons of predicted burns and historical data revealed that the influence of high fire risk zones in Arizona and Nevada had led to the WWRA data over-predicting fire risk for the areas of Utah used in the current analysis. Acting upon the advice of a key WWRA wildfire modeler, we therefore back-scaled the data included in the WWRA, to exclude the influence of high risk zones outside of the state of Utah (Carlton 2016). This back scaling meant that the final fire risk estimates we used were based predominantly on the historical occurrence of only those fires within the state (Figure 4.4).

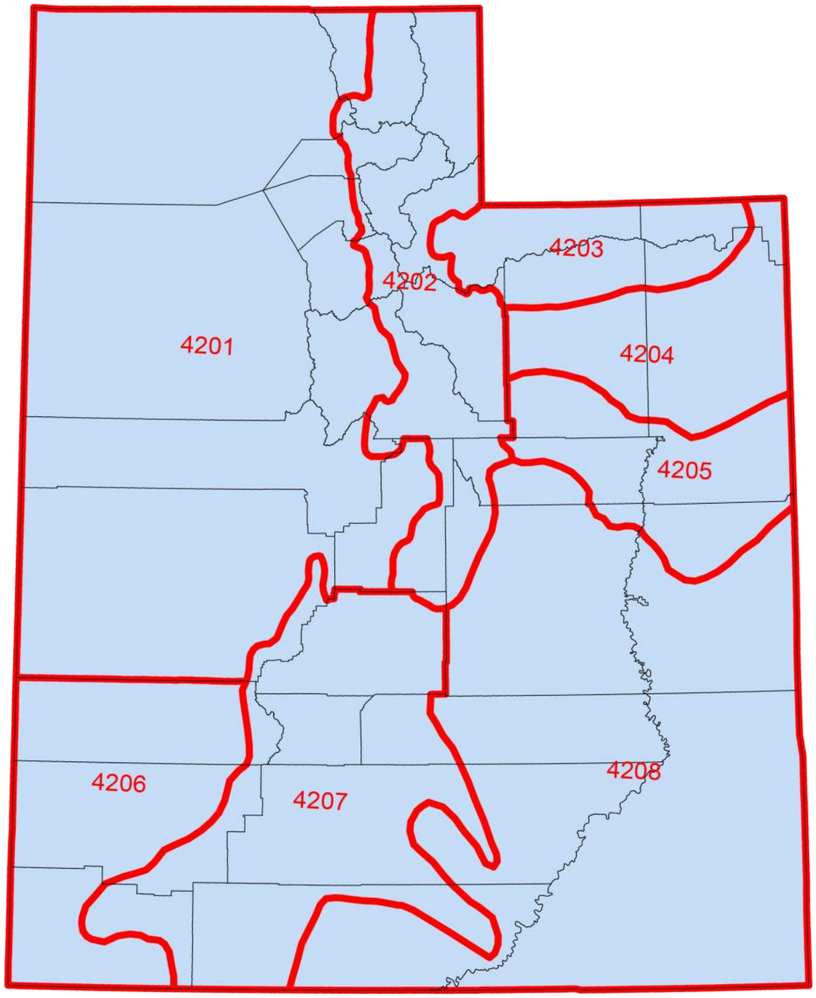


Figure 4.3: Weather influence zones within Utah. The two spatial extents used in the current analysis fall within weather influence zones 4201, and 4202. These weather influence zones are affected by high fire risk areas in other states, leading to an over-prediction of fire risk. We therefore scaled fire risk data to best reflect Utah's risk.

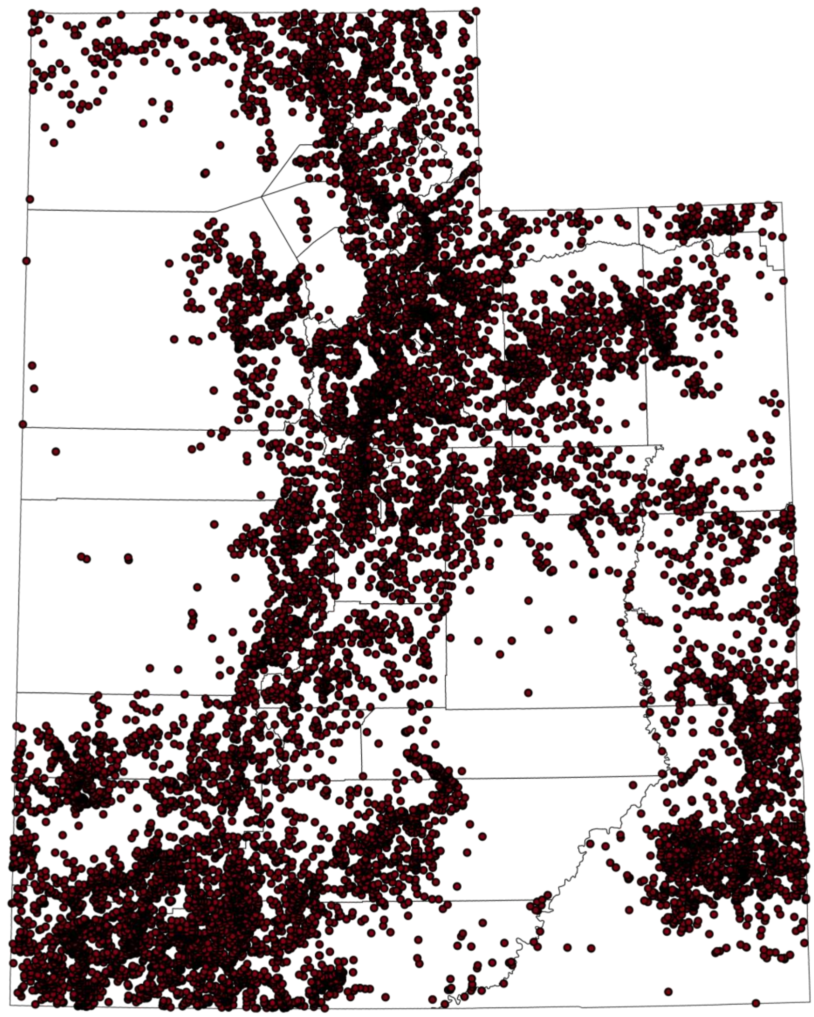


Figure 4.4: Incidents of fire within the state of Utah between the years 1998 – 2008. This history of fires was a central factor in the calculation of future fire risk.

Converting fire risk data into estimates of acreage burned

The FTI does not provide information as to the actual number of acres predicted to burn, or the amount of different vegetation types that will be affected. We therefore employed Monte Carlo numerical simulations to estimate of the number of acres burned from the WWRA data. Monte Carlo simulations represent an excellent method to estimate the effects of spatially-explicit, landscape level risks (Hammill et al. 2016), and have been previously employed in the calculation of fire threats (Carmel et al. 2009, Conedera et al. 2011). To run each iteration of the Monte Carlo simulation, each 30m*30m pixel in the spatial extent was assigned an individual random number between 0 and 1. If the random number assigned to a pixel was less than the pixel's FTI value, an acre was designated as "burned". We repeated this whole procedure 1000 times (new random number assigned to each pixel, an acre designated as burned if the random number is less than the fire risk estimate). For each of these 1000 simulations we then produced an estimate of the total acres burned, and the breakdown of burned acres in terms of different vegetation cover types, and the importance of the area for supplying drinking water. These 1000 different simulations then enabled us to produce a distribution of the estimated total number of acres burned, and report the results in terms of medians and 95% confidence intervals.

Results

The major physical attributes that contributed to fire threat were aspect, slope, and vegetation cover.

Influence of aspect on fire threat

We found that aspect is associated with significant changes in fire threat ($z = 16.05$, $P < 0.001$). However, we also found that fire threat was significantly affected by an interaction between vegetation cover and aspect ($z = 18.02$, $P < 0.001$), meaning that the effect of aspect on fire threat was different for different vegetation types. We therefore focused on how aspect affects fire threat on slopes covered by trees. The lowest fire threat was associated with a north facing aspect (Figure 4.5), the highest with a southern aspect, while East and West were intermediate between the two (Figure 4.5).

Looking at the distributions of different aspects on state and federal lands, we found that the area of land with a north facing aspect (the aspect associated with the lowest burn threat) was 58% greater on federal land than state land (Figure 4.6). State lands also had 37% more of their area with a south facing aspect (the highest fire threat) than federal land (Figure 4.6).

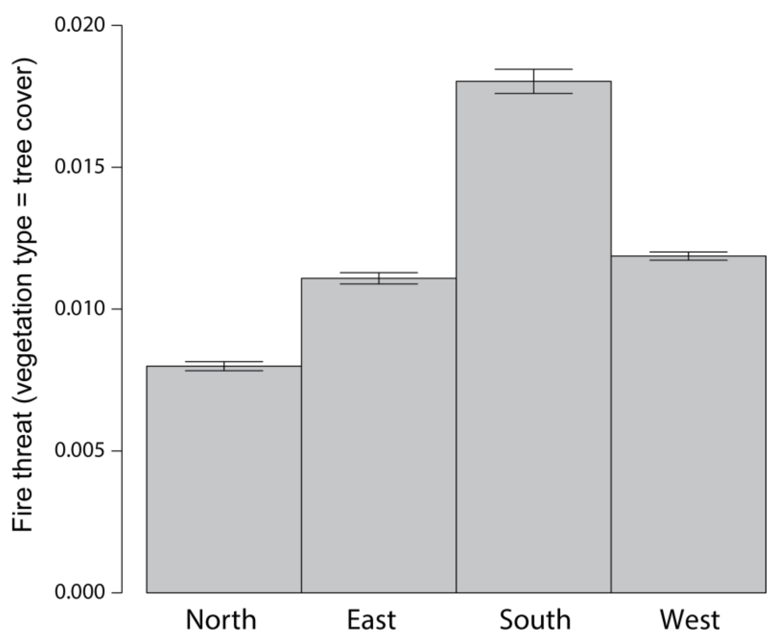


Figure 4.5: Fire threat associated with different aspects

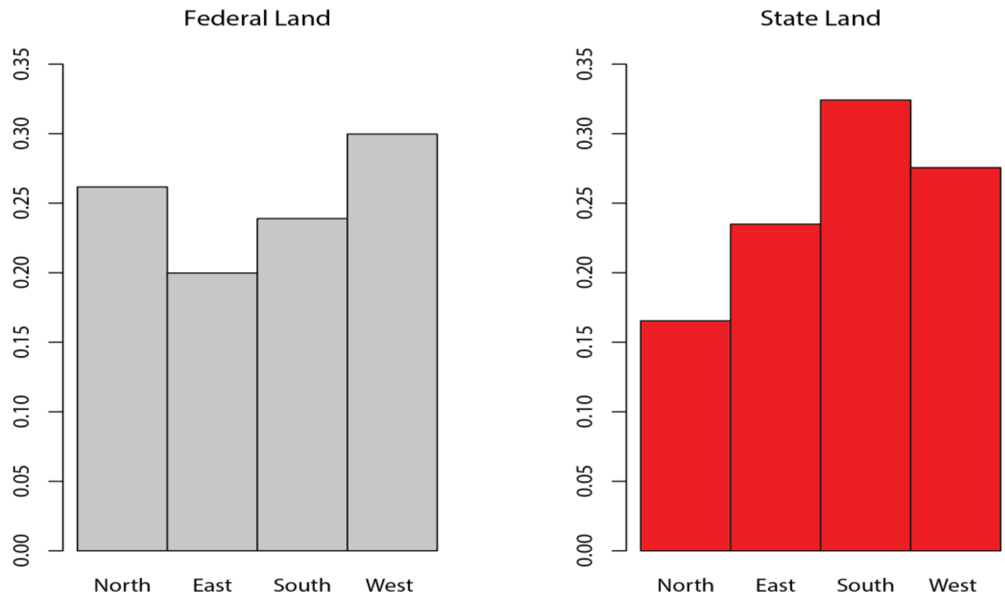


Figure 4.6: Break down of the proportion of Federal and State lands with different aspects.

Influence of slope percentage on fire threat

We found that steeper slopes were associated with increases in fire threat (z = 2337.38, $P < 0.001$, Figure 4.7a), as has been documented in previous investigations (Konoshima et al. 2010). As we found a significant interaction between slope and vegetation type, vegetation type was set as "tree cover". Looking specifically and the breakdown of different slopes on federal and state land, we found that Federally controlled areas had a higher proportion of land with steep slopes (slope greater than 50%, or greater than about 27°), and a lower proportion of land with a gentle or graduate slope (less than 50% - Figure 4.7b).

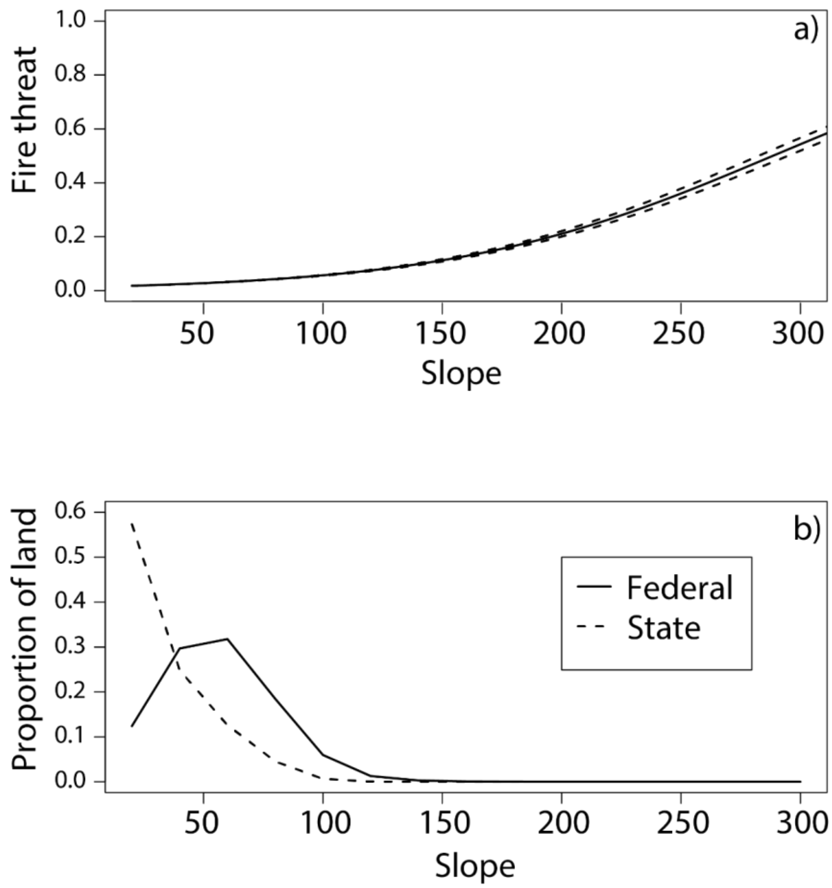


Figure 4.7: Relationships between **a)** slope (as a percentage) and fire threat, and **b)** landownership and proportion of land with different slopes

Influence of vegetation cover on fire threat

While different vegetation cover types were found to significantly affect fire threat, the pattern was far less clear than for either aspect or slope. We found a significant interaction between slope and vegetation cover, so for the remainder of the analysis, slope was set at 50%. Generally, developed forests, developed shrubland (forest or shrubland near urban areas), as well as higher herb and shrub covers were associated with increased fire threat (Figure 4.8a). Conversely, low density herb covers and higher density tree covers are generally associated with reductions in fire threat.

Looking at compositional differences between Federal and state lands, it would appear that Federal lands have relatively higher levels of cover types that area associated with low fire threat (e.g. tree covers between 20% and 80%, Figure 4.8b), and relatively lower levels of cover types associated with increased fire threat (e.g. shrub covers between 10% and 30%, and herb covers between 40% and 60%).

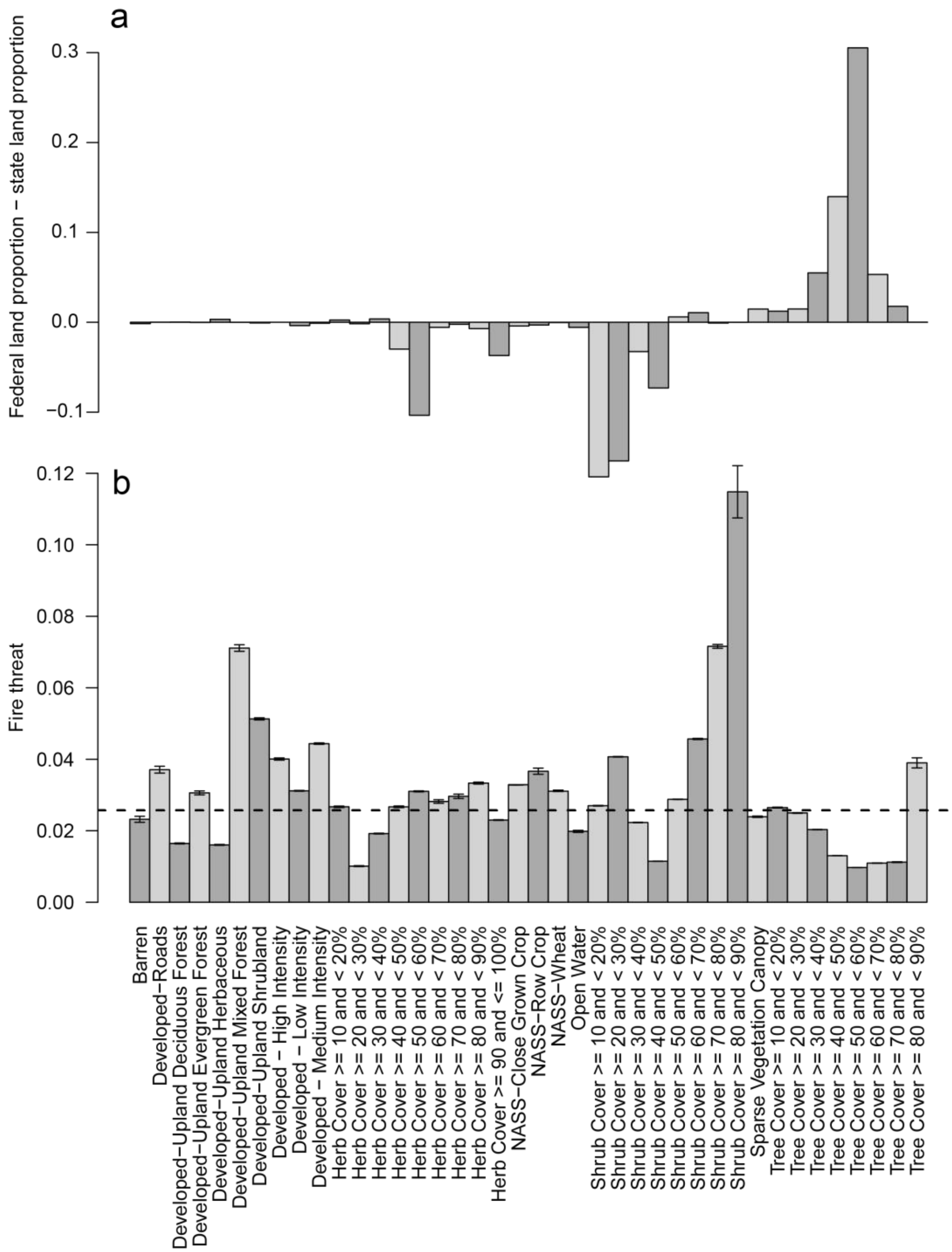


Figure 4.7: (a) Relative proportions of different vegetation cover types on Federal and State lands. Bars greater than zero indicate a vegetation type is more proportionally prevalent on Federal lands. Bars less than zero show vegetation types that are more prevalent on State lands. (b) Fire threat level associated with different vegetation cover types. Bars represent the fire threat associated with different vegetation cover types after the influence of slope and aspect has been removed.

Predicted Acreage Burned

Vegetation Cover Types

Urban Cluster – Weber, Morgan, Davis, Salt Lake

The scaled fire threat data predicted that 6,151 acres (95% confidence interval 6,079 – 6,227) would be burned per year in the Urban cluster. The number of acres of different vegetation cover types predicted to be burned per year are shown in Figure 4.9. The largest contributors to burns tend to be areas covered by shrubs and trees (Figure 4.9). However, this is likely due to the fact that these vegetation types make up substantial amount of the spatial extent used in this cluster (Figure 4.10). However, a comparison of Figures 4.9 and 4.10 would suggest that shrubs are over-represented among the burned acres (i.e. there are more burned acres covered by shrubs that we would predict given the number of overall acres covered by shrubs), while Tree covered areas seem slightly under-represented among the burned acres. This agrees with the data in Figure 4.8 that indicates shrub-covered areas are associated with higher fire threat, while tree covered areas are associated with lower fire threat.

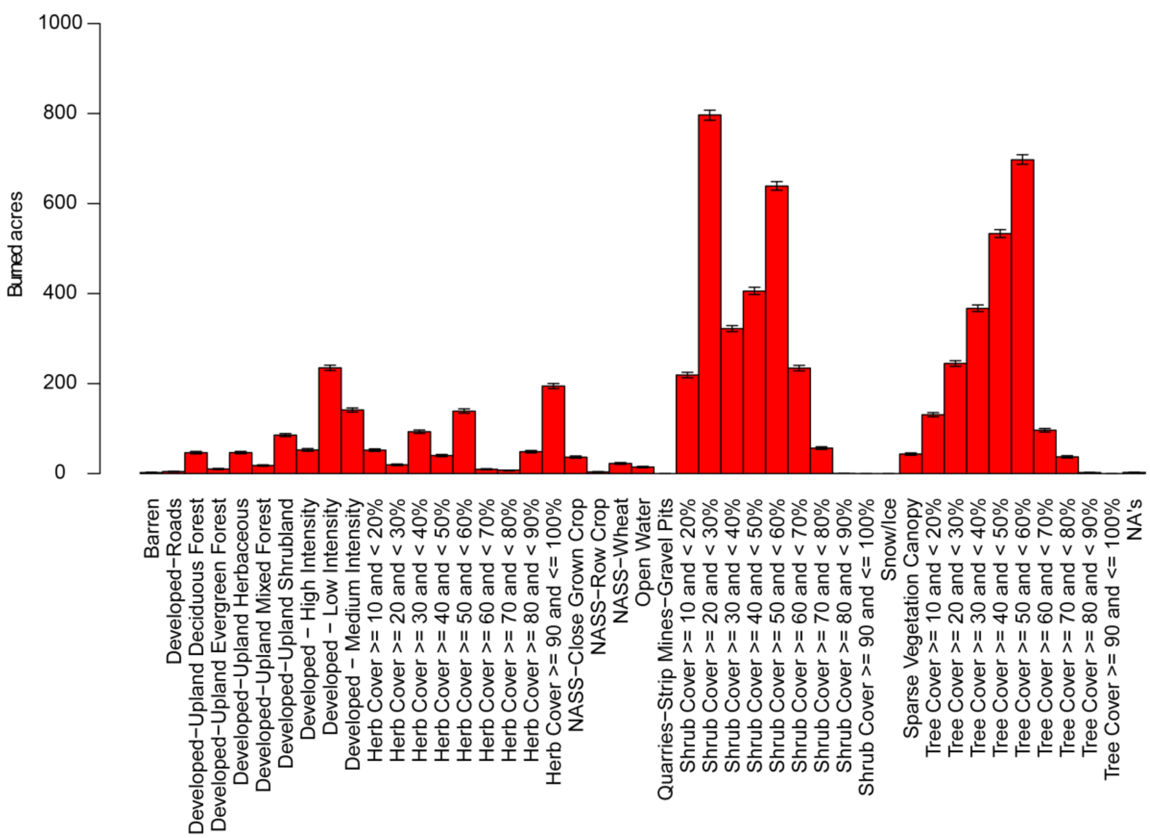


Figure 4.9: Number of acres of different vegetation cover predicted to be burned within the Urban cluster in a single year. Data were generated through 1000 Monte Carlo simulations using fire threat estimates from the WWRA. Bars represent the median values from 1000 simulations, errors represent the 95% confidence intervals.

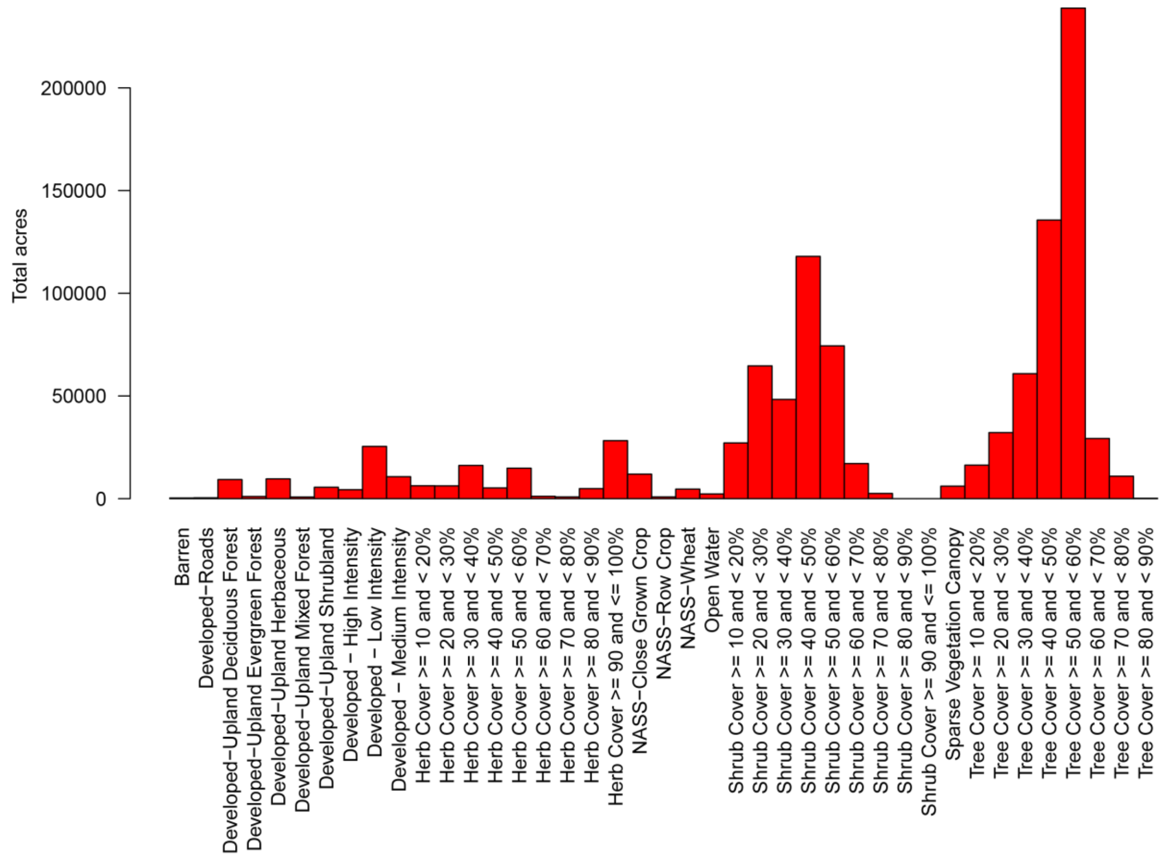


Figure 4.10: Total numbers of acres covered by each vegetation type in the Urban cluster

Rural Cluster–Juab, Sanpete

The scaled fire threat data predicted that 24,306 acres (95% confidence interval = 24,230 – 24,368 acres) would be burned annually in the Rural cluster. The highest amounts of burned acreage are made up of areas covered by shrubs, herbs and crops (Figure 4.11). Looking at the vegetation cover for the whole of the Rural cluster, herb cover, shrub cover, and tree cover make up the majority of the area, with crops covering a relatively small area (Figure 4.12). However, despite trees covering a relatively large area of the rural cluster, they appear under-represented in the burned acres, suggesting that they are burned less often than we would expect (as seen in the Urban cluster). Conversely, crops herbs, and areas with a large covering of shrubs (greater than 50%) make up a substantial portion of the burned acreage despite their limited overall coverage of the rural cluster, suggesting they burn relatively easily. This observation of herbs, crops and heavy shrub coverage being over-represented again agrees with the data in Figure 4.7, where these vegetation cover types were seen to be associated with high levels of fire threat.

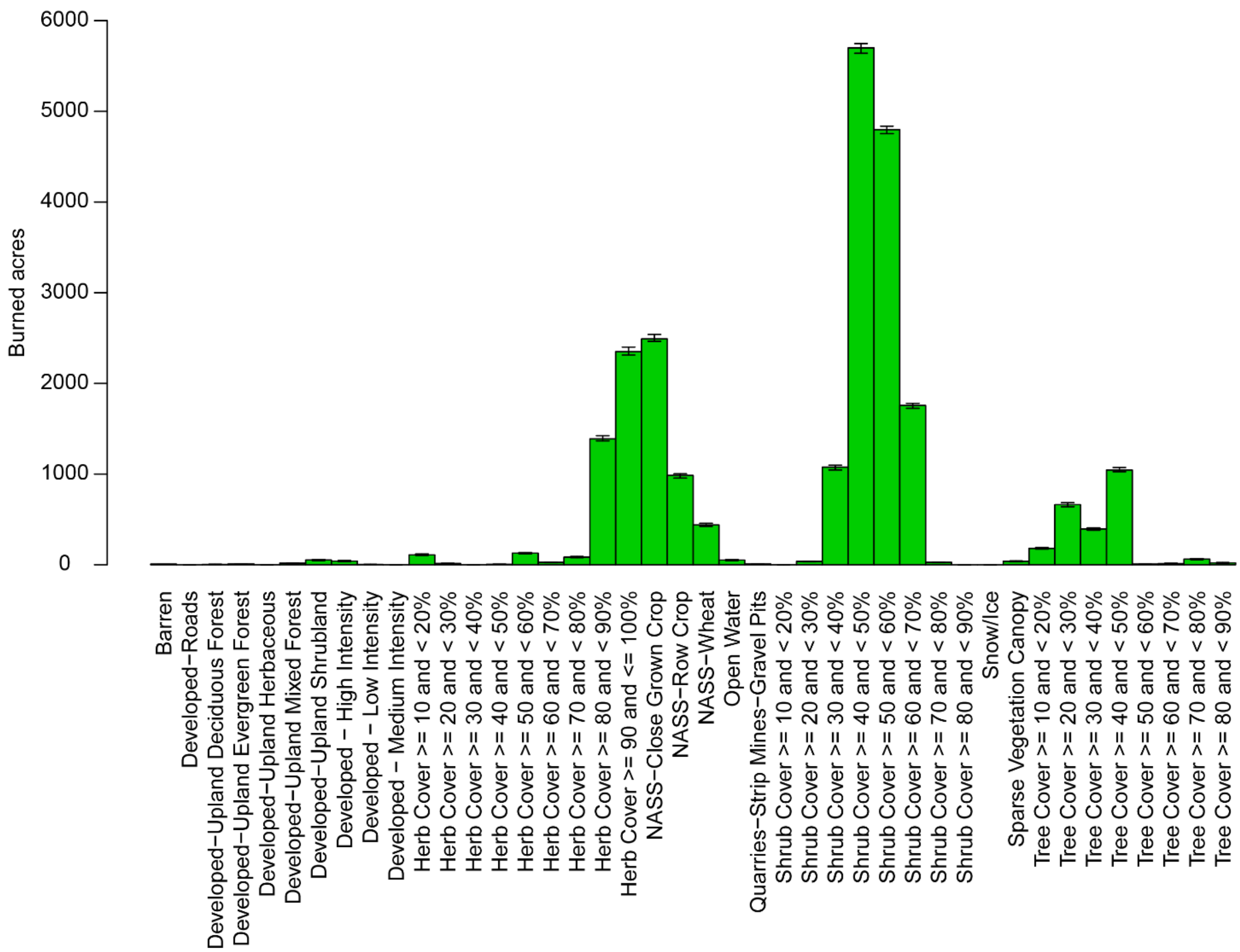


Figure 4.11: Number of acres of different vegetation cover predicted to be burned in the Rural cluster in a single year. Data were generated through 1000 Monte Carlo simulations using fire

threat estimates from the WWRA. Bars represent the median values from 1000 simulations, errors represent the 95% confidence intervals.

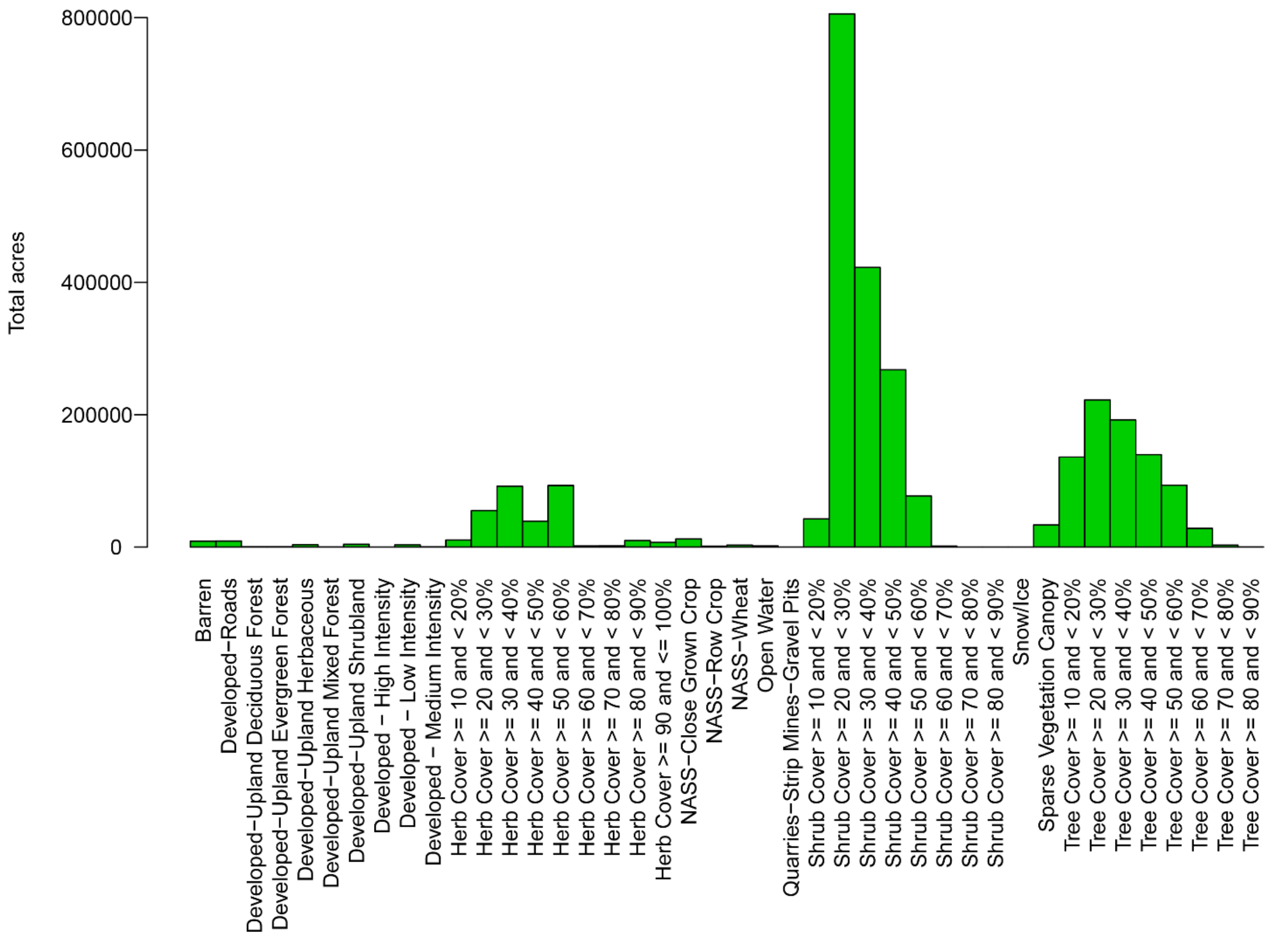


Figure 4.12: Total numbers of acres covered by each vegetation type in the Rural cluster

Importance of Land for Drinking Water

Further information relating to the potential impact of fire on drinking water availability is presented in Chapter 6. Here we present analyses investigating the presence/absence of fire on lands of differing levels of importance for drinking water.

Urban Cluster

The number of acres of areas with different levels of importance for drinking water are shown in Figure 4.13. The drinking water importance rank data were obtained from the US Forest service's "From the Forests to the Faucets" project (Weidner and Todd 2011), and are generated by combining data on water intake locations, population, and the mean annual water supply. Areas ranked as 1 (the least valuable for supplying drinking water) were burned more often than any other rank. This was not due purely to an over-representation of drinking water rank 1 areas in the spatial extent, as areas with a rank 1 are less common than areas ranked either 6 or 10 in the spatial extent (Figure 4.14).

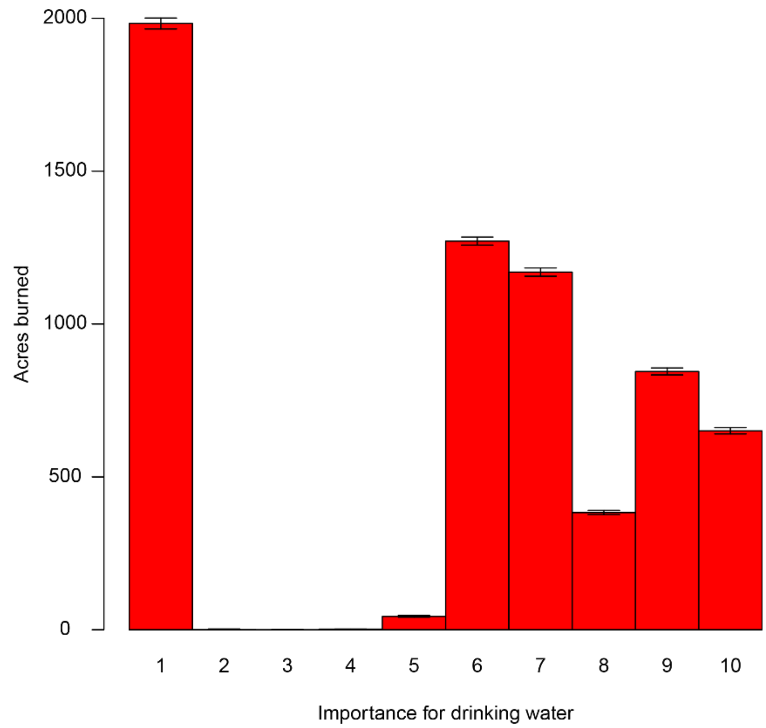


Figure 4.13: Lands are ranked 1 through 10 in increasing order of importance for drinking water in the Western Wildfire Regional Assessment. Figure illustrates the number of acres burned for each drinking water importance rank in the Urban cluster.

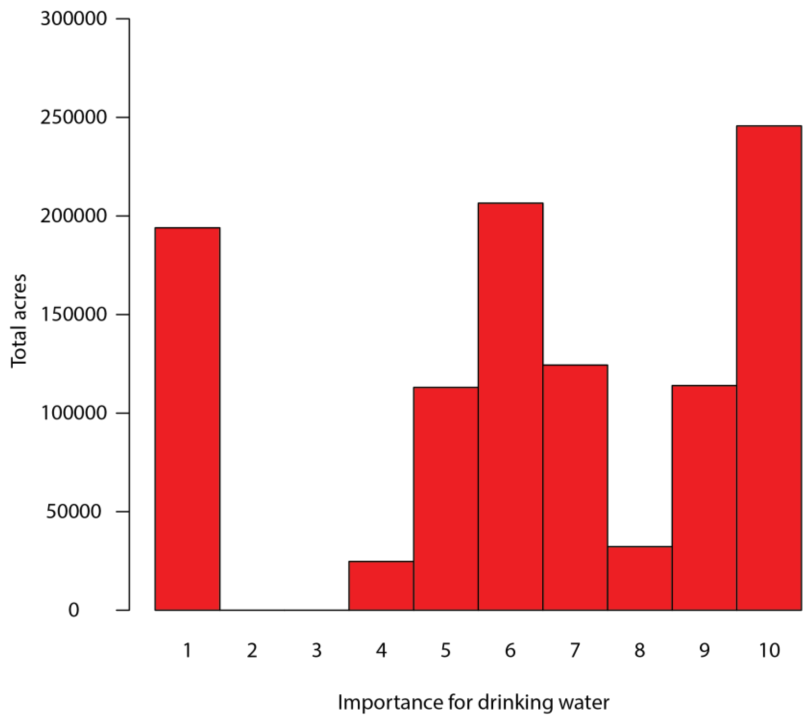


Figure 4.14: Number of acres within the spatial extent assigned to different drinking water importance ranks in the Urban cluster

Rural Cluster

The number of acres of land burned in the Rural cluster for each of the different drinking water importance ranks are shown in Figure 4.15. Like the Urban cluster, "1" is the most common drinking water importance rank within the cluster. However, unlike the Urban cluster there are far fewer areas with higher drinking water ranks (Figure 4.16). This is likely due to the lower human population and drier conditions of the Rural cluster.

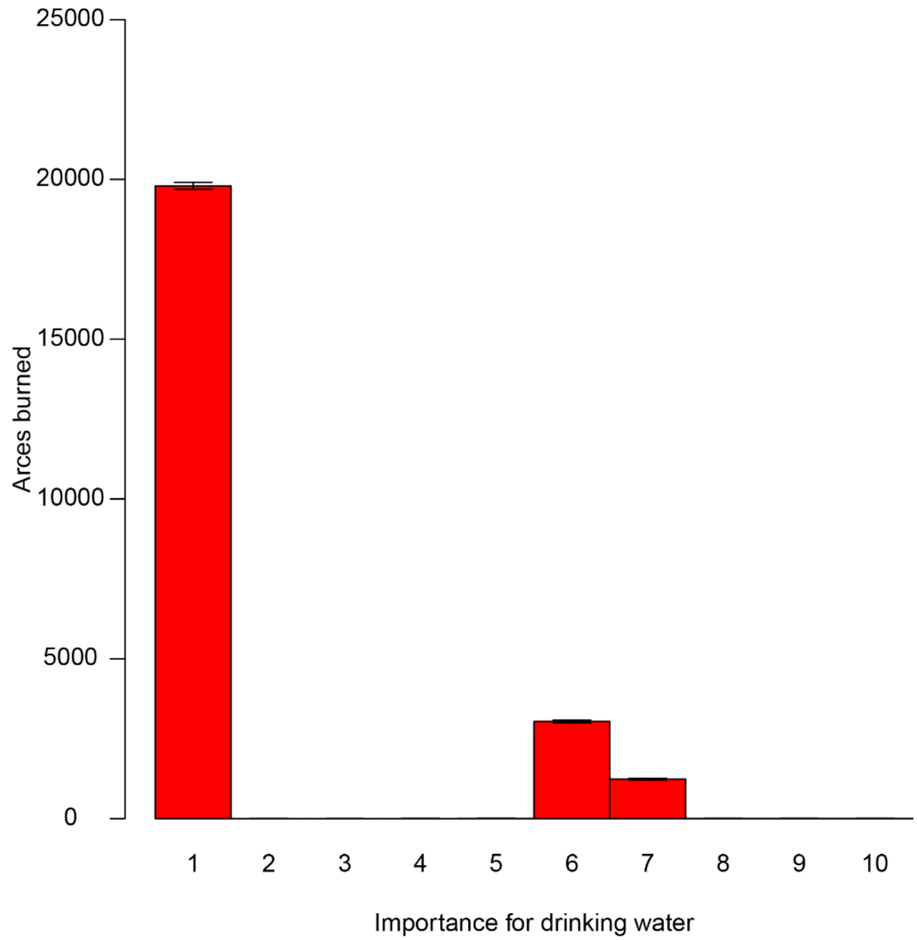


Figure 4.15: Number of acres burned of areas with different ranks for drinking water importance within the Rural Cluster.

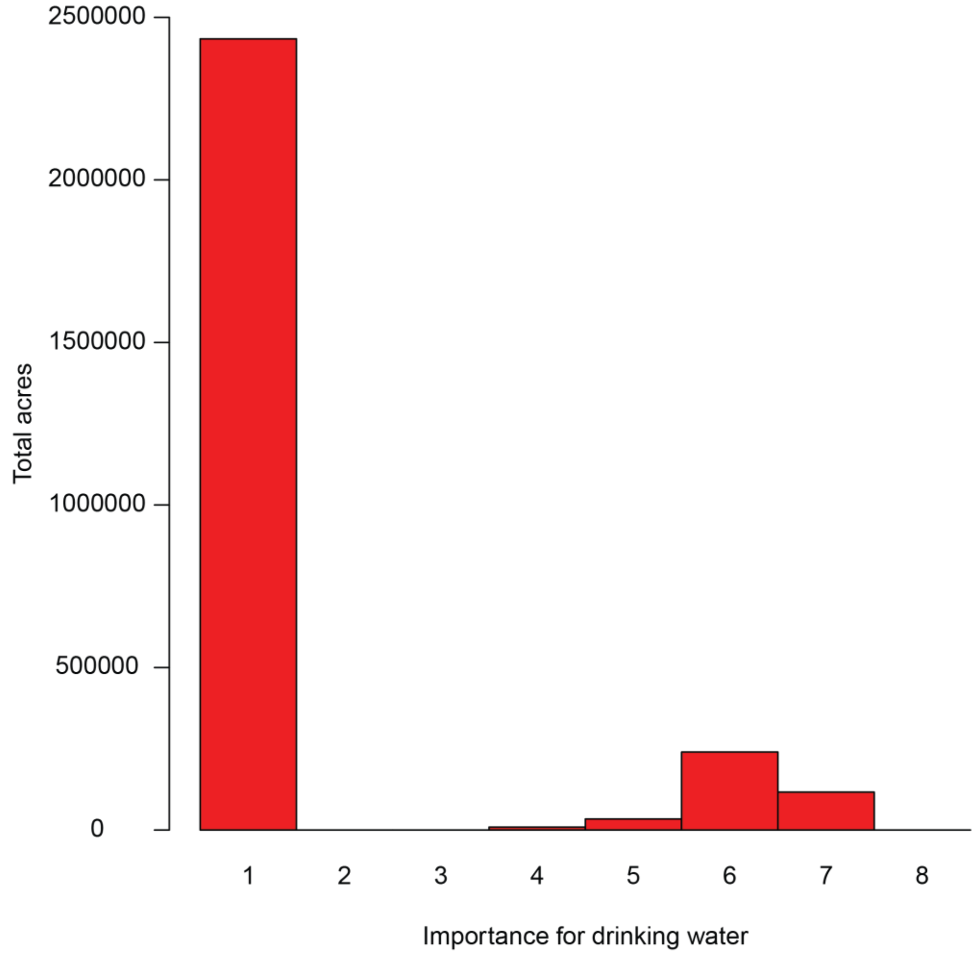


Figure 4.16: Number of acres within the spatial extent assigned different drinking water importance ranks in the Rural cluster

Summary

Influence of physical attributes on fire threat

Our analyses revealed that areas with south facing aspects and steep slopes have relatively high fire threats, as has been demonstrated in earlier investigations (Carmel et al. 2009, Haire and McGarigal 2009, Alexandre et al. 2016). The observed increase in fire threat observed on south facing slopes is believed to be a due to reductions in fuel moisture associated with increased solar exposure (Dillon et al. 2011). The topographical distribution of lands within Federal and State-managed areas indicates that Federal lands have an over-representation of steep slopes. This high number of steep slopes may partially account for the increased rates of fire on Federal lands. However, our analyses also show that Federal lands have a relatively greater proportion of north facing slopes than State lands, the aspect associated with the lowest fire threat. These unchangeable, physical environmental properties of Federal and State lands must be accounted for when management decisions are being made, as they may either directly influence the ability of managers to reduce fire threat, or affect accessibility.

With respect the relationship between vegetation types and fire threat, our analyses demonstrate that several vegetation cover types associated with lower fire threats (Tree cover between 30% and 80%) are more common on Federal and State land. In addition, certain vegetation cover types associated with increased fire threat are less prevalent on State land than Federal land (e.g. shrub cover between 10% and 30%). This over-representation of low fire threat vegetation on Federal lands would imply that the increased incidence of fire on Federal lands is not entirely due to the presence of fire-prone vegetation types. Future management activities conducted with the intention of reducing fire threat on Federal lands should take into account the fact these lands already possess vegetation types associated with lower levels of fire threat.

Impact of Fire threat reduction activities

In many cases, conducting management activities to reduce fire threat represents and economically viable management strategy. Pre-emptively reducing threat reduces the need for subsequent fire suppression, and activities such as the removal of woody debris may provide useful products for sale (Evans and Finkral 2009). However, threat reduction activities are relatively costly, meaning that they are often conducted over relatively small spatial scales in a targeted manner. Locations for threat reduction are selected on the basis of fire threat, perceived effectiveness of reduction, and proximity to high value areas (Watts and Hall 2016).

In addition to data from the WWRA, during the course of this project we were also provided with spatial data detailing the locations of efforts to reduce fire threat from the department of Forestry, Fire, and State Lands (FFSL). It had been our intention to combine the data on surface fuels from the WWRA with this fire threat reduction data to assess the impacts of reduction effects on fire threat. However, upon closer inspection of the data from FFSL and the surface fuel data within the WWRA, it was discovered that the time lines for the two data sets did not match up, with many of the fire threat reduction activities conducted by FFSL taking place after the surface fuel data were compiled (completed in 2005). In total, of the greater than 5 million acres within our spatial extent, threat reduction activities had been conducted on only 96 acres. Due this this extremely low number of areas receiving management activities to reduce fire threat, we did not feel confident in performing an analysis on the data.

Vegetation Cover of Acres Predicted to Burn

Our results indicate that although the majority of acres predicted to be burned were covered by shrubs and trees, this was predominantly a consequence of the high levels of these vegetation types in our study area. However, more developed areas, especially developed upland shrubs and upland mixed forest were predicted to burn more frequently. These developed areas represent a relatively small fraction of the spatial extent, and it is unclear from our analyses whether the predicted high incidence of fire is due to the vegetation cover itself, or its proximity to other developed areas.

Conclusion

Our results indicate that fire threat is influenced by a complex interaction between the physical characteristics of the environment (slope, aspect), and vegetation cover types. Our results reveal that in the spatial clusters we used for this study, Federal and State lands differ significantly in their composition. Federal lands tend to be steeper (a characteristic associated with increased fire threat) and also are more likely to be northerly facing (associated with lower fire threat). In addition, Federal and State lands differ in the communities of vegetation they contain, with Federal lands showing an over-representation of several land types associated with lower fire threats (Tree cover). When future decisions are being made with respect to management actions targeted to reduce fire threat, the influence of environmental factors that cannot be easily modified (slope, aspect) must be considered along with the influence of those that can (vegetation cover type)

References

Alexandre, P. M., S. I. Stewart, M. H. Mockrin, N. S. Keuler, A. D. Syphard, A. Bar-Massada, M. K. Clayton, and V. C. Radeloff. 2016. The relative impacts of vegetation, topography and spatial arrangement on building loss to wildfires in case studies of California and Colorado. *Landscape Ecology* 31:415–430.

Carmel, Y., S. Paz, F. Jahashan, and M. Shoshany. 2009. Assessing fire risk using Monte Carlo simulations of fire spread. *Forest Ecology and Management* 257:370–377.

Conedera, M., D. Torriani, C. Neff, C. Ricotta, S. Bajocco, and G. B. Pezzatti. 2011. Using Monte Carlo simulations to estimate relative fire ignition danger in a low-to-medium fire-prone region. *Forest Ecology and Management* 261:2179–2187.

Dillon, G. K., Z. A. Holden, P. Morgan, M. A. Crimmins, E. K. Heyerdahl, and C. H. Luce. 2011. Both topography and climate affected forest and woodland burn severity in two regions of the western US, 1984 to 2006. *Ecosphere* 2:1–33.

Evans, A. M., and A. J. Finkral. 2009. From renewable energy to fire risk reduction: A synthesis of biomass harvesting and utilization case studies in US forests. *GCB Bioenergy* 1:211–219.

Haire, S. L., and K. McGarigal. 2009. Changes in fire severity across gradients of climate, fire size, and topography: a landscape ecological perspective. *Fire Ecology* 5:86–103.

Hammill, E., A. I. T. Tulloch, H. P. Possingham, N. Strange, and K. A. Wilson. 2016. Factoring attitudes towards armed conflict risk into selection of protected areas for conservation. *Nature Communications* 7:11042.

Hawkins, B. A. 2012. Eight (and a half) deadly sins of spatial analysis. J*ournal of Biogeography* 39:1–9.

Konoshima, M., H. J. Albers, C. A. Montgomery, and J. L. Arthur. 2010. Optimal spatial patterns of fuel management and timber harvest with fire risk. *Canadian Journal of Forest Research* 40:95–108.

Sandborn Map Company. 2013. WEST WIDE WILDFIRE RISK ASSESSMENT.

Watts Jr, J. M., and John R. Hall. 2016. Introduction to fire risk analysis.

Weidner, E., and A. Todd. 2011. From the forest to the faucet: drinking water and forests in the US. Methods paper, Ecosystem Services and Markets Program Area, State and Private Forestry. USDA Forest Service.

CHAPTER 5: WILDFIRES AND AIR POLLUTANTS

Randal S. Martin and Ian Hammond
Dept. of Civil & Environmental Engineering and the Utah Water Research Laboratory
Utah State University

General Fire-related Air Quality and Health Impacts

It is well-known that wildfires can negatively impact local and regional air quality. The U.S. Environmental Protection Agency and other groups have prepared a guide for public health officials that summarizes recent research, fire-related smoke issues, and suggests protective measures and communication to the broader public (EPA, 2016a). Furthermore, owing to the increasing awareness and interest in the accurate quantification of the impacts of wildfires, the National Oceanic and Atmospheric Administration is coordinating a four year (2016-2019) multi-agency, multi-university field and laboratory study, FIREX. The stated goal of the program is to develop a comprehensive research effort to understand and predict the impact of North American fires on the atmosphere and ultimately support better land management (Warneke et al., 2015).

In brief, wildfire-derived air pollutants can include particulate matter (PM_{10} and $PM_{2.5}$), carbon monoxide (CO), and reactive gases [oxides of nitrogen (NO_x), volatile organic compounds (VOCs), inorganic acids, ammonia (NH_3)] (Urbanski et al., 2009; Kreidenweis, et al. 2010). Additionally, significant emissions of important greenhouse gases such as carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are often observed (Urbanski, 2103). It should be noted that wildfires can also be a necessary part of a healthy ecosystem (Mutch and Cook, 1996). They enable the growth of new plants, open up niches that may have disappeared and release carbon (C) back into the environment to be used by plants and animals.

Many of the reactive and toxic chemicals given off during wildfires can impact the overall atmospheric load of trace gasses and aerosols, as well as lead to the formation of secondary species, most significantly downwind ozone (Knorr et al., 2012). Further, Jaffe and Wigder (2012) estimated that 3.2% of all *global* tropospheric O_3 is likely due to wildfire emissions, and that percentage is likely to increase as wildfire activity increases. Additionally, some of these pollutants may condense out secondary organic aerosols (SOAs) or inorganic aerosols (e.g. ammonium nitrate, NH_4NO_3) – the latter of which is regionally important to the northern Utah region.

In most regions affected by wildfires, the worst air pollution day of the year is often caused by wildfires. Kenward et al (2103) showed cities within 50-100 miles of wildfires typically found their air quality dropping to 5-15 times worse than normal, and 2-3 times worse than the worst non-fire day. Kenward et al. compiled data from EPA's AirNow network for the 2011 Wallow Fire in Arizona and New Mexico (eventually >535,000 acres) and demonstrated that local $PM_{2.5}$ concentrations in Springerville, AZ soared to 310 μg/m^3 (AQI = 360; hazardous, "Maroon"). During the same event, $PM_{2.5}$ concentrations in Albuquerque, NM, 150 miles to the east, reached 68 μg/m^3 (AQI = 157; unhealthy, "Red"). Unhealthy (AQI "Orange") $PM_{2.5}$ concentrations were also noted in Taos, NM, over 350 miles away from the fires. Additionally, the University of Maryland-Baltimore County's Smoke Blog (UMBC, 2011) visually showed the plume from the Wallow Fire ultimately impacted regions throughout New Mexico, Colorado, Texas, Oklahoma, Arkansas and Missouri.

Similarly, as is demonstrated in Figure 5.1, local and regional wildfires can produce downwind $PM_{2.5}$ concentrations on the order of the same magnitude as northern Utah's notorious wintertime, "homegrown" $PM_{2.5}$ levels as measured at the Cache Valley's regulatory location. As can be seen, discrete episodes in 2005, 2012, and 2015 exceeded or approached the 24-hr National Ambient Air Quality Standard (NAAQS) of 35 μg/m^3. These periods can directly be traced to local or regional wildfire events.

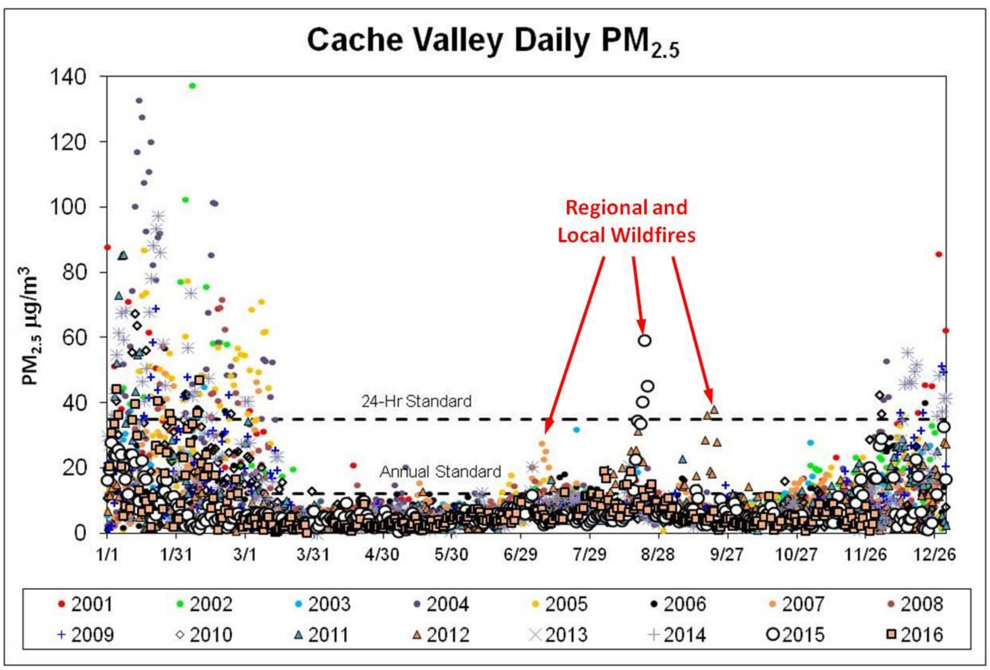


Figure 5.1: Cache Valley daily $PM_{2.5}$ concentrations 2001 – 2016 Source: UDAQ, 2016a

Looking more closely at the 2015 episode can give additional insight to the extent of downwind transport of fire-derived air pollutants. In August of 2015, wildfires in central Washington and Idaho, totaling several hundreds of thousands of acres, produced plumes which transported well into northern Utah. Figure 5.2 shows the EPA's Air Quality Index map for August 24, 2015. In brief, the AQI was established as an easier way for the public to relate concentration units (μg/m^3 or ppm) to simple 0-100 linear scale, where 100 is generally equal to the given NAAQS value. A color code (e.g. green, yellow, orange, red, purple, maroon) is also associated with given health breakpoints with "100" typically occurring at the yellow/orange interface. As can be seen, the areas immediately within the fire zones are color characterized as maroon/purple while the downwind transport is indicated by the red, orange, and yellow AQI colors. The obvious plume was carried to the southeast across Idaho and into northern Utah, still maintaining unhealthy levels of particulate matter.

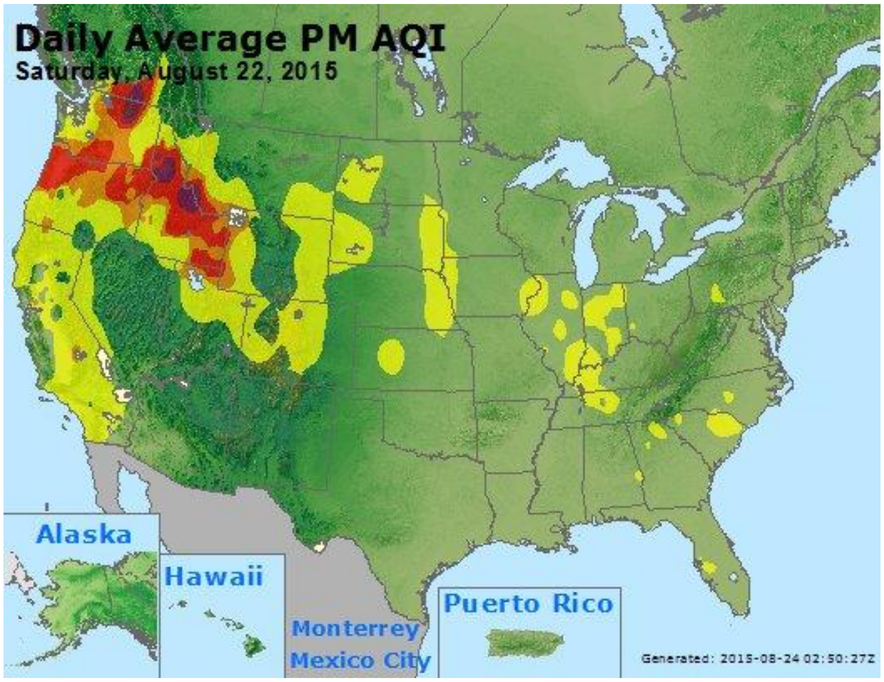


Figure 5.2: U.S. EPA's $PM_{2.5}$ Air Quality Index (AQI) for August 24, 2015 showing areas of unhealthy air quality (orange, red, and maroon) associated with western wildfires

Source: AirNow, 2016.

Although not explicitly derivable from Figure 5.2, it should be pointed out that the maroon/purple areas shown are indicative of hazardous and very unhealthy levels of air pollutants. The hourly and daily $PM_{2.5}$ data can be accessed from the Idaho Department of Environmental Quality. The observed maximum hourly $PM_{2.5}$ concentrations in the fire zones reached 325 $\mu g/m^3$ (AQI 375), 334 $\mu g/m^3$ (AQI 384), and 471 $\mu g/m^3$ (AQI 481), at Salmon, Plummer, and Pine, ID, respectively (IDEQ, 2016). All of these levels are within the AQI maroon "Hazardous" category. These AQI levels are similar to the extreme values frequently reported from mega cities around the world (WAQ, 2016).

Figure 5.3 shows the daily averaged $PM_{2.5}$ concentrations for several areas along the Wasatch Front, including Cache Valley, before, during, and after the impact of the 2015 WA/ID fires. It should be noted that during this period, UDAQ was transitioning between sampling locations in the Cache Valley and, as such, two Utah regulatory data sets were available (Logan and Smithfield). As shown, the $PM_{2.5}$ concentrations were generally in the 5-10 $\mu g/m^3$ range before and after the fire-impacted period, which is typical of summer-time Values (refer back to Figure 5.1). During the event, the concentrations rapidly approached the NAAQS, but plateaued along the Wasatch Front locations

(Lindon, Hawthorne, and Bountiful). However, the Cache Valley locations, and to some degree the Brigham City site, showed continued and elevated impacts for a few additional days. This neatly demonstrates the spatial variability even in fairly long-range plume transport. Interestingly, this spatial disparity is even demonstrated within the confines of the Cache Valley. The observed $PM_{2.5}$ values significantly increased relative to the more northern position of the sampling stations. The southernmost Logan site is 7.5 miles from the Smithfield site which is 12 miles from the Franklin, ID site.

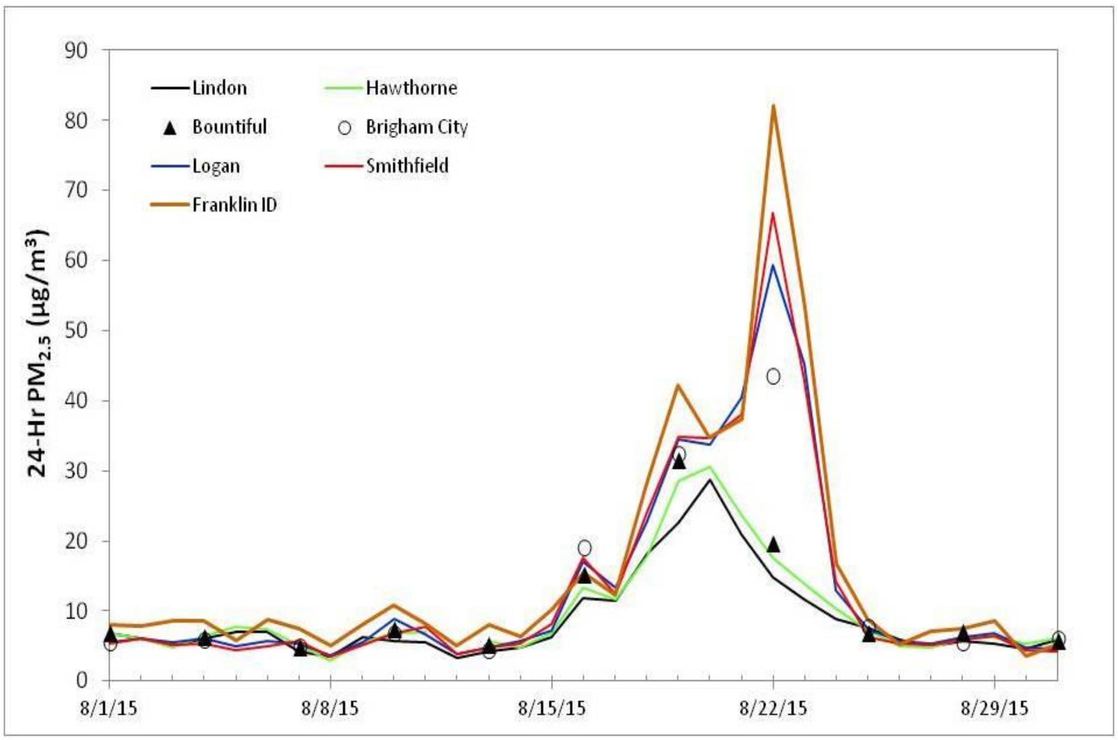


Figure 5.3: Daily PM2.5 concentrations along Utah's Wasatch Front and the UT/ID Cache Valley encompassing the August 2015 fire impacted period Sampling sites at Bountiful and Brigham City only collected PM2.5 filters every 3rd day.

Source: UDAQ, 2016a; IDEQ, 2016

The 24-hr $PM_{2.5}$ concentrations reached 18-26 μg/m^3 in the Salt Lake area (AQI = 63-80; moderate, "Yellow"), 44 μg/m^3 in Brigham City (AQI = 122; unhealthy for sensitive groups, "Red"), 59 μg/m^3 in Logan (AQI = 153; unhealthy, "Red"), 67 μg/m^3 in Smithfield (AQI = 157; unhealthy, "Red"), and 82 μg/m^3 in Franklin, ID (AQI = 165; unhealthy, "Red") (UDAQ, 2016a).

Figure 5.4 shows the average, the best and the worst 24-hr $PM_{2.5}$ concentrations attributable to the August 2015 WA/ID fires relative to the EPA's AQI color codes. As shown, the worst days extend well into the Unhealthy (red) category, while even the worst non-fire days were still within the Good (green) category. The averages across the fire-impacted impacted days were into the Moderate (yellow) category. Similar to the above demonstration, using reverse trajectory and meteorological modeling, Mallia et al. (2015) showed 2007 and 2012 wildfires in California and the Intermountain West contributed to enhancements of CO and $PM_{2.5}$ of 250 ppb (3-hr) and 15 μg/m^3 (24-hr) above local background levels, respectively, specific to the greater Salt Lake City area.

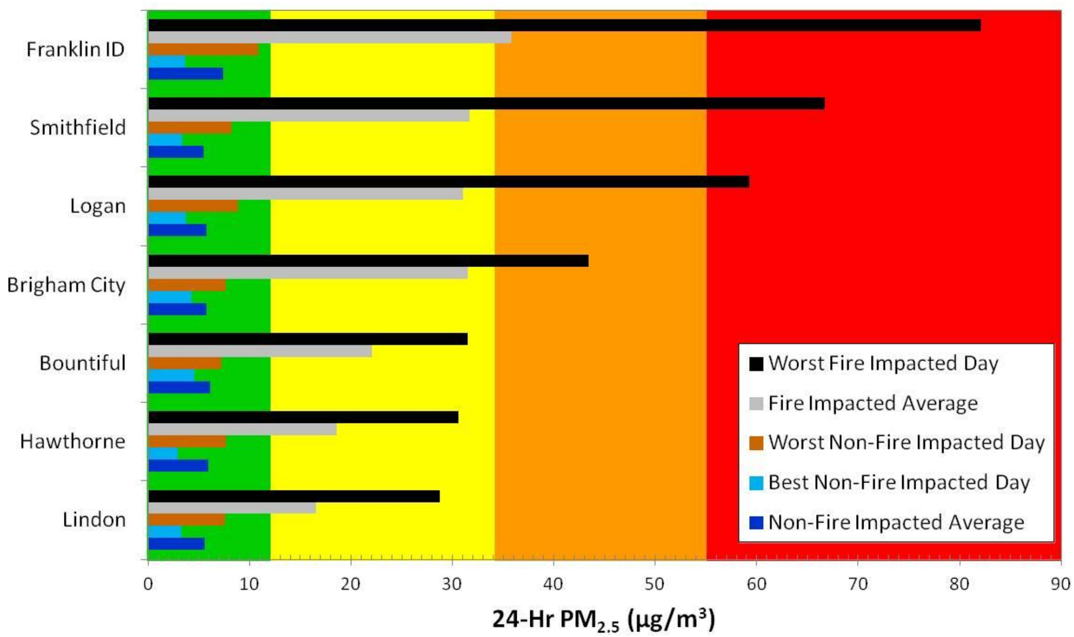


Figure 5.4: Comparison northern UT and southeast ID best and worst PM2.5 days associated with the August 2015 WA/ID fires overlaid onto the AQI color code system.

Furthermore, regional wildfires have also been shown to decrease visibility across wide swaths of the western United States via other sampling networks. The IMPROVE network, a $PM_{2.5}$ monitoring array located primarily within Class I areas (national parks & wilderness areas), has shown statistically significant $PM_{2.5}$ enhancements due to regional wildfires (Jaffe et al., 2008). Although the average enhancement seemed small (1.11 μg/m^3), it is important to point out that the average summer $PM_{2.5}$ concentrations within these areas were typically around 4-5 μg/m^3 (Jaffe et al, 2008).

The potentially added burden to the air pollutant levels attributed to wildfires can increase the risk for negative health effects and affect air quality attainment status. Several areas within Wasatch Front and in Utah's Uintah Basin have been recently recommended to the EPA as non-attainment for ozone (Herbert, 2016; UDAQ, 2016b). Similarly, airsheds along the Wasatch Front and Cache Valley are presently in non-attainment status for $PM_{2.5}$, which is strongly dominated by the secondary species NH_4NO_3 (UDAQ, 2014a; IDEQ, 2014).

In addition to potential regulatory air quality issues, wildfire emissions are a cause for health concern. Many of the emissions or secondary pollutants are toxic in nature or are able to penetrate into the lungs, causing adverse health effects which are typically attributed to "common" air pollutants.

The literature, as well as scientific/regulatory websites, is replete with studies describing the connection between air pollutants and public health (e.g. Kampa and Sastanas, 2007; NIH, 2016). Specific reviews of health impacts from exposure to wildfire smoke have shown increased respiratory morbidity, increased risk of respiratory and cardiovascular disease especially in sensitive populations (elderly, children, those, with chronic diseases), as well as ophthalmic, psychiatric, and multi-organ complications (Finlay et al. 2012; Liu et al. 2015; and Reid et al. 2016). Vedal and Dutton (2006) examined the likelihood of an increase in daily mortality in the Denver area due to acute wildfire smoke exposure, and although increases in cardiorespiratory deaths were observed, they were not statistically significant. Conversely, Richardson et al. (2012) estimated a wildfire-related health cost of $84.42 per exposed person per day from wildfires impacting the Los Angeles, CA region, although no estimate was given for the total exposed population. A similar study for wildfires impacting the Reno, NV area (Moeltner et al., 2013) estimated per exposed person costs between $54 and $467 dollars, varying as a function of acres consumed, distance from fires, and fire fuel load.

Estimated Air Pollutants Emissions from Fire Risk Assessment Model for Selected Urban and Rural Utah Counties

Deriving fire-induced emissions of air pollutants requires detailed estimates of the total area burned, the biomass type and extent, and reliable emission factors – the latter typically expressed in grams of pollutant per kilogram of dry biomass consumed. As discussed in other chapters of this document, for the purpose this analysis two contrasting Utah regions were selected for representative analysis. The four counties of Davis, Morgan, Salt Lake, and Weber were selected to represent a highly populated (Urban) area. The paired counties of Juab and Sanpete were chosen to represent a highly vegetative, less populated (Rural) area.

As was also previously presented within the document, the Western Wildfire Regional Assessment (WWRA), was used to estimate the spatial location, vegetative cover type and total acreage burned within each grid cell at a resolution of 30 m (Sandborn Map Company, 2013). These fire risk determinations were then overlaid onto a speciated biomass index dataset, also at 30 m resolution, as compiled by NASA's Oak Ridge National Laboratory's Distributed Active Center North American Carbon Program which provided the dried biomass (kg) per square meter for each of the specified vegetative or land cover types (ORNL DAAC, 2013). Figure 5.5 shows the modeled acreage burned for the selected Urban and Rural county clusters. As can be seen, the estimated area burned within the Urban counties totaled 6,150 acres, and was dominated by tree and shrub covered areas. A total of 24,306 acres was modeled to be burned within the Rural counties, also being dominated by tree and shrub covered areas. However, herb covered regions also contributed notably to the total burned biomass. Although not precisely scalable due to the different vegetative distributions, the larger burned acreage in the Rural area is indicative of large expected fire-derived emissions as well.

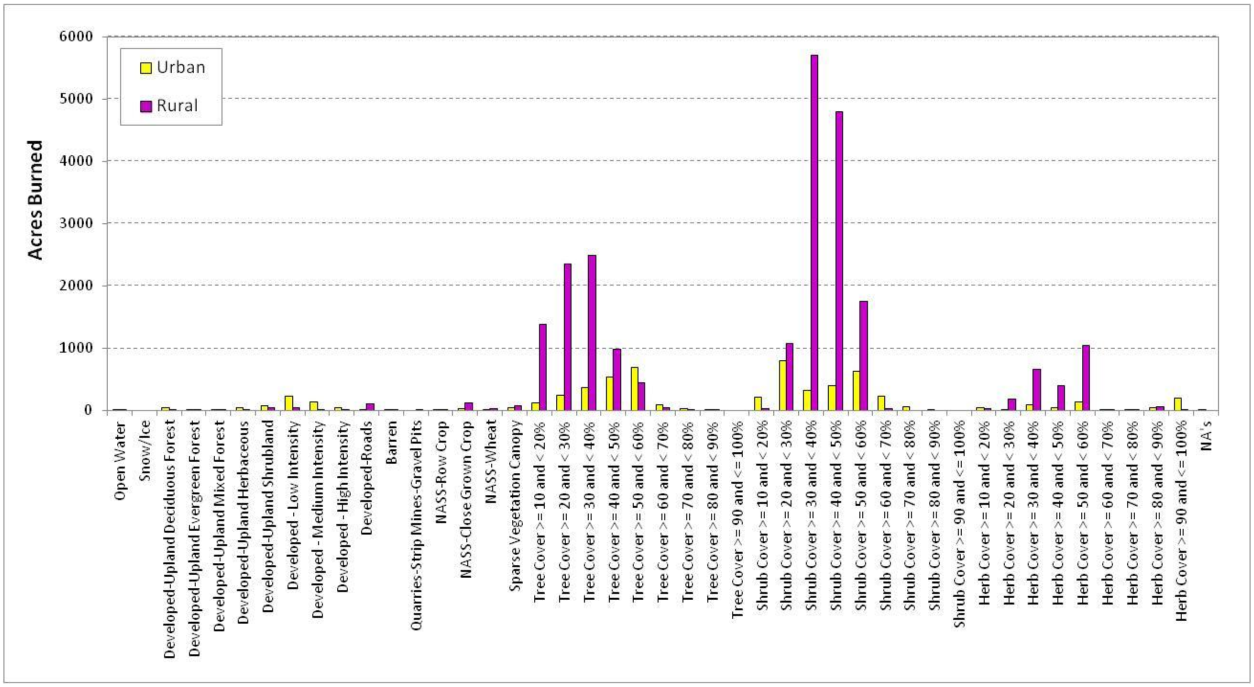


Figure 5.5: Modeled burned acreage per land cover type for the selected Urban (Weber, Morgan, Davis, and Salt Lake counties) and Rural (Juab, Sanpete counties) areas. (This figure corresponds to Figures 4.9 and 4,10.)

Several estimation algorithms are available deriving air pollutant emission rates as a function of biomass burned (Battye and Battye, 2002; Akahi et al. 2011; Yokelson et al., 2013). Additionally, the EPA's AP-42: Compilation of Air Emission Factors (EPA, 2016b), typically used for permitting and modeling emission estimates, includes relationships for biomass combustion-derived (e.g. wildfire) air pollutant emissions. However, more recently emission factors based on field observations and laboratory studies relevant to the western United Sates have been compiled by investigators from the United States Forest Service's Missoula Fire Sciences Laboratory. Urbanski et al (2009) and Urbanski (2013) compiled detailed emissions from differing vegetative regimes for typical pollutant categories (particulate matter and reactive gases) and also included over three dozen different volatile organic compound (VOCs). Table 5.1 summarizes these emission rates for western forests and rangelands for the regulated ambient pollutants, particulate matter less than 2.5 µm ($PM_{2.5}$), carbon monoxide (CO), and oxides of nitrogen (NO_x). Additionally, atmospherically important ammonia (NH_3) and summed VOCs emission rates are shown in Table 5.1. Furthermore, emission rates of important greenhouse gases (GHG) carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) are also shown. It should be noted that, as described by Stockwell et al. (2014), ultimate emissions can also be significantly affected by the severity of the burn. For example, Stockwell et al. showed that as the burn efficiency decreases (e.g. the fire goes from complete burn to smoldering) production of VOCs can increase significantly. However, since the WWRA fire risk model cannot accurately predict the fire severity, no adjustment will be attempted for variable combustion efficiencies.

Additionally, Knorr et al. (2012) examined the sensitivity of simulated fire burn and emission models to available satellite-derived data, and found the models are extremely sensitive to major model inputs including estimated fuel loads, combustion efficiency, and assumed emissions factors. As such, the emission loads modeled herein must not be taken as absolute, but rather as the best-available estimate with the current state of knowledge.

Table 5.1. Utilized emission rates for wildfire-derived air pollutants in grams of pollutant per kilogram biomass consumed.

<table>
<tr><th></th><th>CO (g/kg)</th><th>PM2.5 (g/kg)</th><th>NOx (g/kg)</th><th>VOCs (g/kg)</th><th>NH3 (g/kg)</th><th>CO2 (g/kg)</th><th>CH4 (g/kg)</th><th>N2O (g/kg)</th></tr>
<tr><td>Forest</td><td rowspan="2">138</td><td>11.7</td><td rowspan="2">1.7</td><td>13.2</td><td rowspan="2">0.85</td><td rowspan="2">1,597</td><td rowspan="2">7.2</td><td>0.16</td></tr>
<tr><td>Rangeland</td><td>9.7</td><td>1.69</td><td>0.32</td></tr>
</table>

Source: Urbanski et al., 2009 and Urbanski, 2013

It is of interest to note that the emission rates given in Table 5.1 show that not all of the biomass burned in a fire event is completely consumed. In other words, not all of the biomass is emitted to the atmosphere as gaseous or particulate pollutants and residual solid mass is left in place. If it is assumed that all of the original biomass is carbon (C), by taking ratios of the molecular weight of carbon to that of listed pollutants, the fraction of original biomass emitted to the atmosphere can be derived. Conversely, the mass of biomass left on the ground can also be estimated. For example, CO has a molecular weight of 28 g/gmole, while C has a molecular weight of 12 g/gmole. In Table 5.1, the emission rate for CO was given as 138 g/kg, therefore, the emission of C implicit to the CO would be 59.1 g/kg (12/28 = 0.429 or 42.9% of the original 138 g/kg). Similar analysis across the other carbon-containing pollutants results in an estimated C emission rate of 515.8 g/kg. This suggests that approximately one half of the biomass is volatilized to atmospheric products, leaving the remainder (0.48 kg of each original kg) as remnant, on-the-ground material.

Figures 5.6 - 5.13 show the modeled emissions for the above pollutants in terms of total tons of pollutants throughout the fire events. Note that the emission scale (y-axis) is shown in logarithmic scale, meaning each tick mark represents a change in order of magnitude. To examine the significance of the magnitudes of these predicted emissions, we compare wildfire emissions to the most recent emission inventories for the selected counties for CO, NO_x, $PM_{2.5}$, and VOCs (UDAQ, 2014b) and NH_3, CO_2, and CH_4 (NEI, 2011). The UDAQ/NEI emission values are reported in total tons per year, as opposed to total tons per fire as estimated via the previously described modeling algorithms.

The estimated emissions of carbon monoxide (CO) are shown in Figure 5.2. As can be seen, the UDAQ, primarily anthropogenic, emissions were calculated to be 17,822 and 175,333 tons per year for the Rural and Urban counties, respectively. This difference is in-line with population difference between the two regions: 35,585 vs.1,650,786 (2013, US Census Bureau). This trend is reversed with the modeled fire-derived CO emissions. In the Rural counties, 48,041 tons were emitted due to the fires, while 13,837 tons were emitted from the Urban counties, reflecting the differences in total acreage burned. For the Rural counties, the predicted CO from the fires would increase the local emissions by approximately 270%, while the Urban increase is only estimated at 8% additional to the anthropogenic emissions..

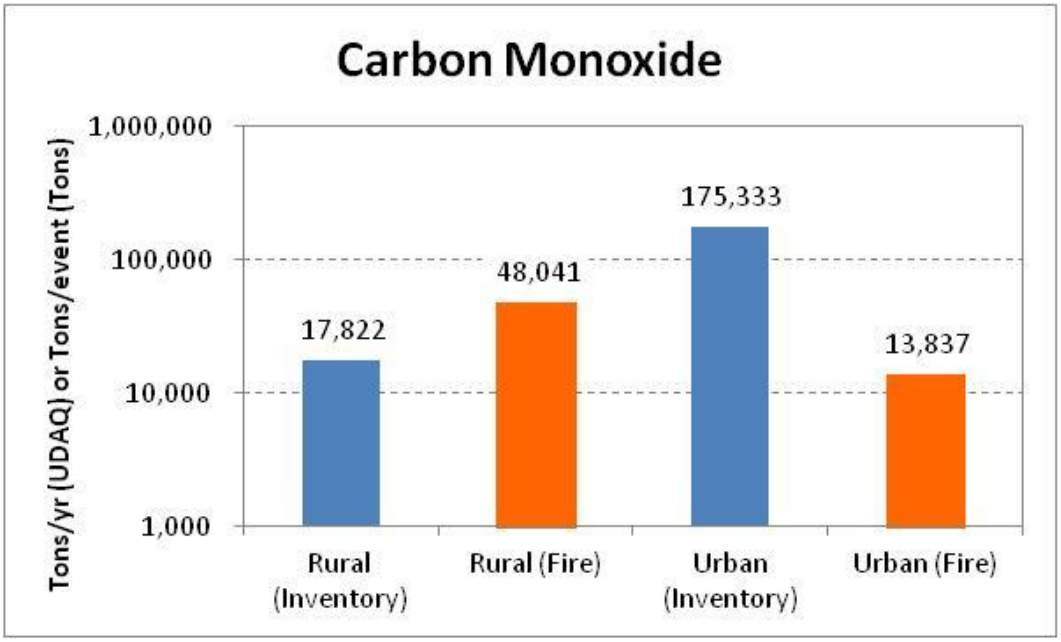


Figure 5.6: Fire-derived carbon monoxide (CO) emissions for the examined Rural and Urban counties compared to CO emissions from UDAQ (2014b) and NEI (2011) emission inventories.

Fire-modeled and anthropogenic emissions of direct $PM_{2.5}$ emissions are shown in Figure 5.7. As shown, the Rural fire $PM_{2.5}$ enhancement is about 250% of the UDAQ, primarily anthropogenic, emissions inventory (4,047 tons vs. 1,559 tons/year). For the Urban counties, the direct $PM_{2.5}$ emissions were found to be 8,534 tons/year and 1,165 tons for the UDAQ inventory and fire model, respectively. Again, the potential fires were modeled to show a more significant impact in the Rural as opposed to the Urban area.

As previously mentioned, $PM_{2.5}$ particles can not only be directly emitted from the various sources, but can also be formed via atmospheric photochemistry involving precursor species (NO_x, SO_x, VOCs, and NH_3), resulting in particulate forms of ammonium sulfate and ammonium nitrate. Additionally, NO_x and VOCs can also participate in local and downwind ozone (O_3) formation. Figures 5.8 through 5.10 show the compiled and modeled emissions for NO_x, VOCs, and NH_3. Oxides of sulfur (SO_x) are not typically associated with wildfires emissions. The UDAQ/NEI (anthropogenic) emissions of NO_x, VOCs, and NH_3 are given as one-to-two orders of magnitude greater than the comparable fire-derived emission estimates. While these modeled values do not appear to add significantly to the annual airshed burden of the given pollutants, previously referenced studies have shown appreciably enhanced O_3 concentrations downwind of active wildfires. As such, fire-derived emissions of NO_x and VOCs should be considered important, particularly in the modeled Rural counties.

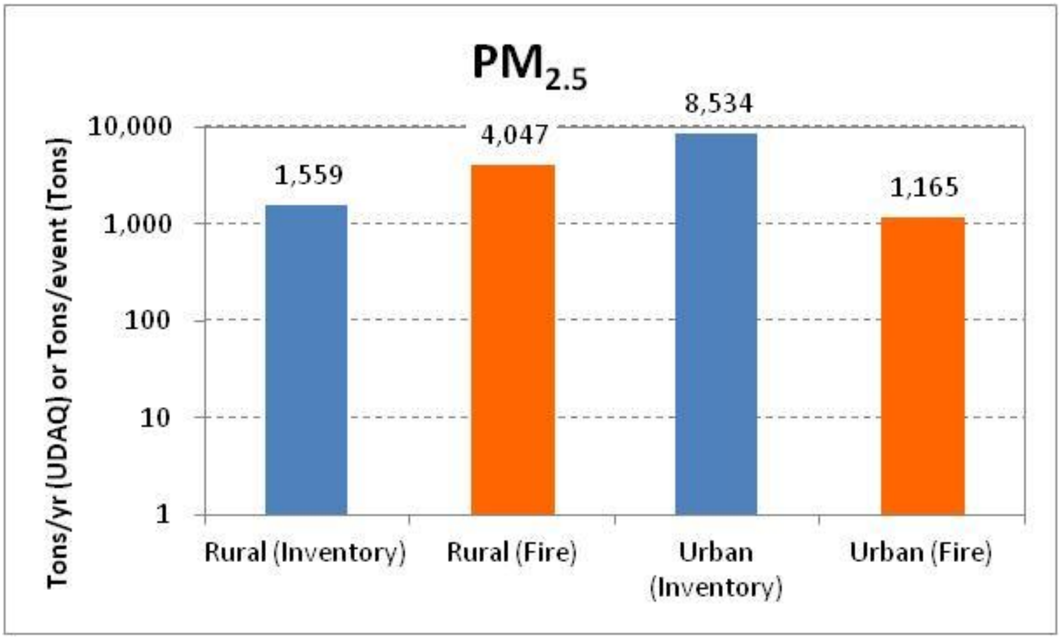


Figure 5.7: Fire-derived $PM_{2.5}$ emissions for the examined Rural and Urban counties compared to direct $PM_{2.5}$ emissions from UDAQ (2014b) and NEI (2011) emission inventories.

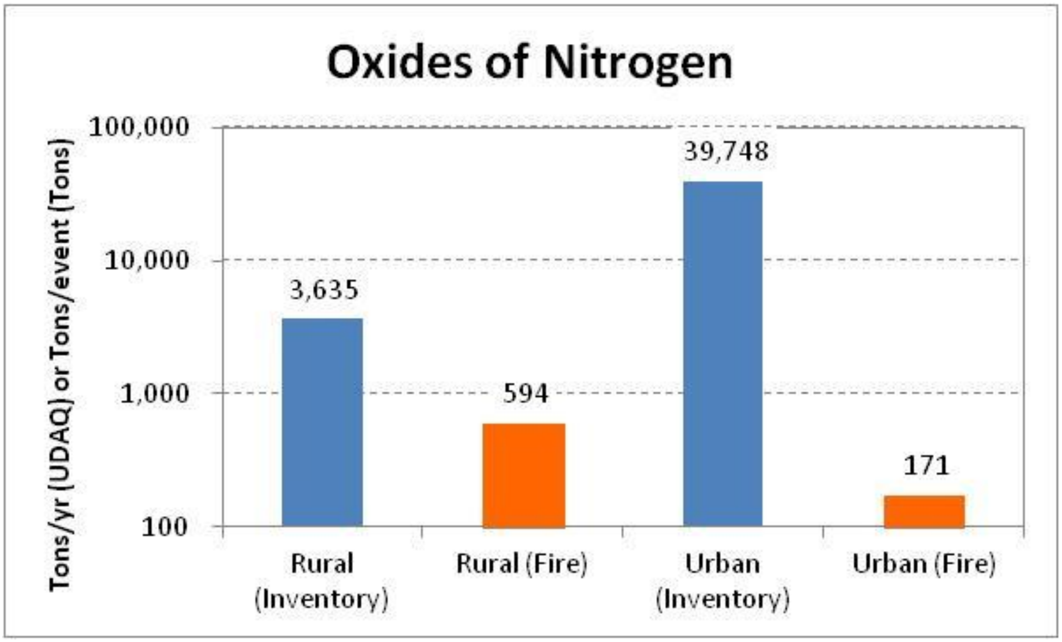


Figure 5.8: Fire-derived oxide of nitrogen (NO_x) emissions for the examined Rural and Urban counties compared to NO_x emissions from UDAQ (2014b) and NEI (2011) emission inventories.

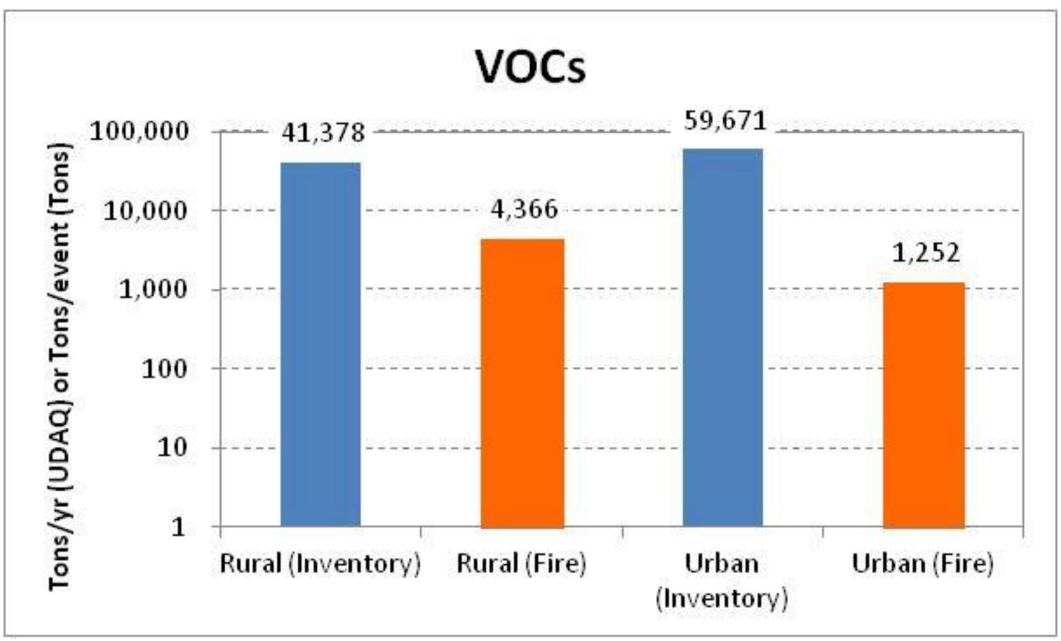


Figure 5.9: Fire-derived VOC emissions for the examined Rural and Urban counties compared to VOC emissions from UDAQ (2014b) and NEI (2011) emission inventories.

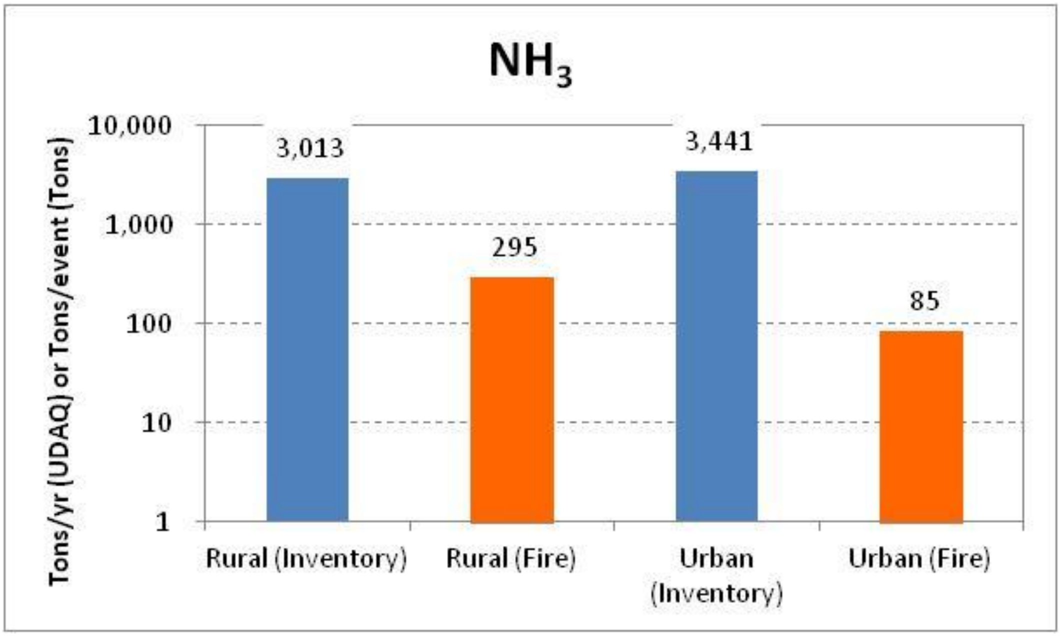


Figure 5.10: Fire-derived ammonia (NH_3) emissions for the examined Rural and Urban counties compared to NH_3 emissions from UDAQ (2014b) and NEI (2011) emission inventories.

As a moderately efficient combustion process of carbon-intensive fuels, wildfires can be expected to emit significant quantities of strong greenhouse gases (GHGs), which are scientifically known to have global warming potential (GWP), as discussed by the EPA (2016c). Specifically, these fired-derived gases include carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). The latter compound, nitrous oxide, should not be confused with oxides of nitrogen (NO_x) which is by definition the sum of nitric oxide (NO) and nitrogen dioxide (NO_2). The seemingly small difference between the N_2O and NO_x terms is an important distinction. N_2O, nitrous oxide (also known as laughing gas), is photochemically non-reactive in the troposphere, but readily absorbs energy and reradiates it back according to greenhouse theory. NO_x, oxides of nitrogen, on the other hand, is very reactive in the troposphere and is a major component of the tropospheric O_3 cycle.

GWP is defined as the relative greenhouse potential of various compounds when compared to the greenhouse warming and atmospheric lifetime of CO_2 or a multiplier to examine the increased potential warming for each molecule of the chosen compound. As such CO_2 has a GWP of 1.0, while the GWP of CH_4 and N_2O, assuming a 100-yr time scale, are given as 28-36 and 265-298, respectively (EPA, 2106c). Figures 5.11 through 5.13 show the inventory compiled emissions (tons/yr) and modeled fire emissions for CO_2, CH_4, and N_2O, respectively. As can been seen, CO_2 is by far the dominant emission, varying from 100's of thousand to over a eight million tons/yr or tons, even when compared to the previously discussed pollutants. This is expected as CO_2 is considered the ultimate, and often desired, product of carbon-fuel combustion. The Urban county fires did not appear to significantly add to the atmospheric burden for CO_2 and N_2O, approximately 2% and 5%, respectively. The modeled Urban fires did, however, show an additive CH_4 burden of about 160%. After CO_2, CH_4 was the next most abundant GHG emission, and the largest fraction was that derived from the modeled fire emissions in the Rural counties (2,521 tons) which was almost 20 times the emissions inventory value (134 tons/yr) and 3½ - 5½ times the Urban values. Similarly, the model Urban fire-derived CH_4 (726 tons) was about 1 ½ times the UDAQ/NEI emissions inventory (456 tons/yr). Modeled or compiled N_2O emissions were generally an order of magnitude or so less than the CH_4 emissions, with the Urban emissions inventory showing the greatest value (338 tons/yr) and the Rural fire-derived emissions at the next greatest level (59 tons).

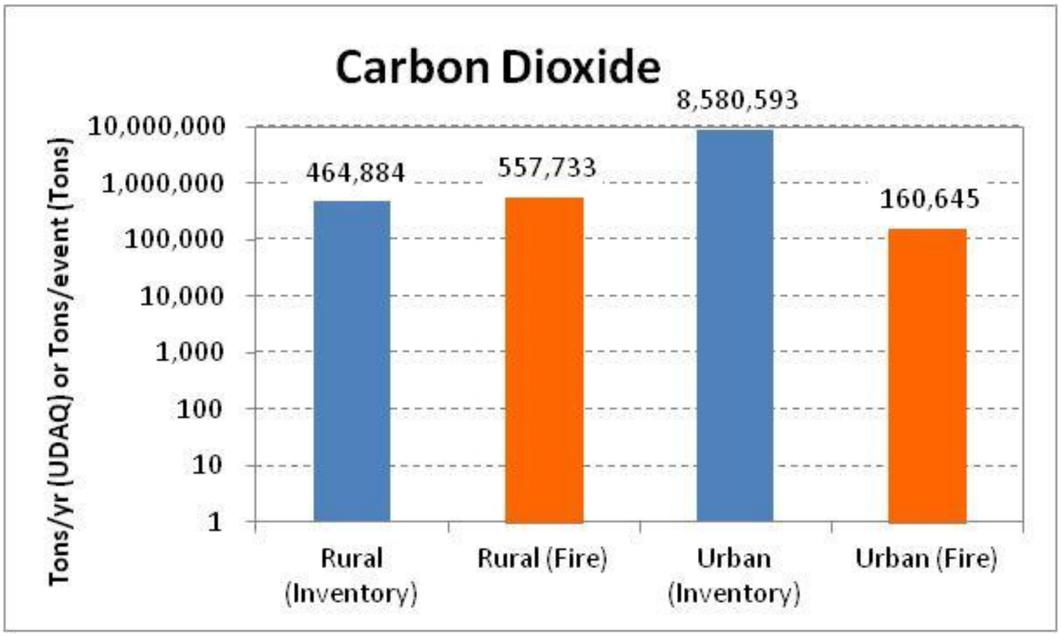


Figure 5.11: Fire-derived CO_2 emissions for the examined Rural and Urban counties compared to CO_2 emissions from UDAQ (2014b) and NEI (2011) emission inventories.

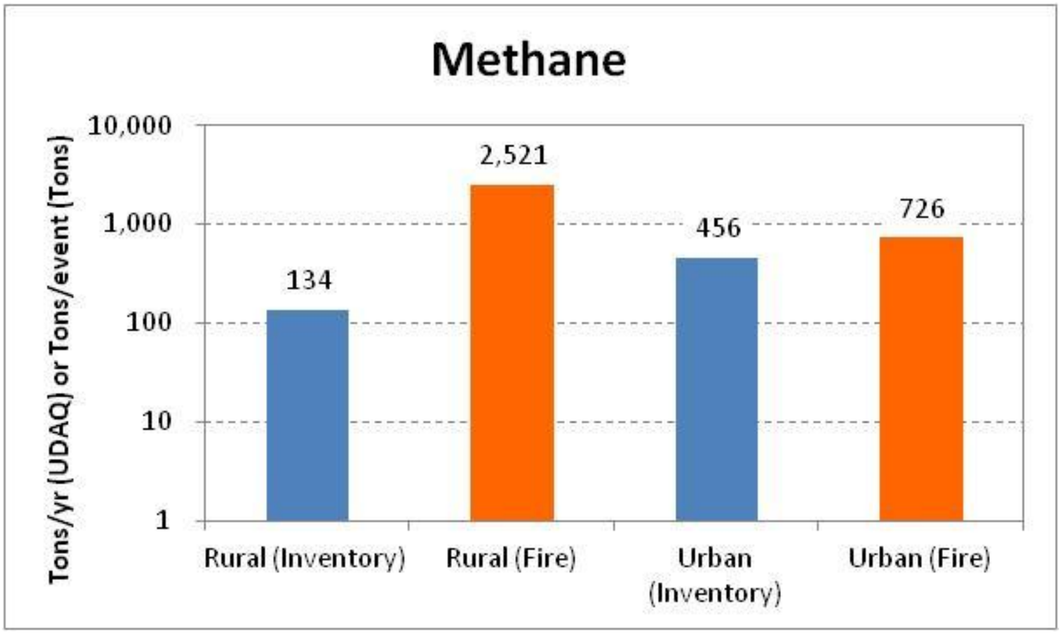


Figure 5.12: Fire-derived methane (CH_4) emissions for the examined Rural and Urban counties compared to CH_4 emissions from UDAQ (2014b) and NEI (2011) emission inventories.

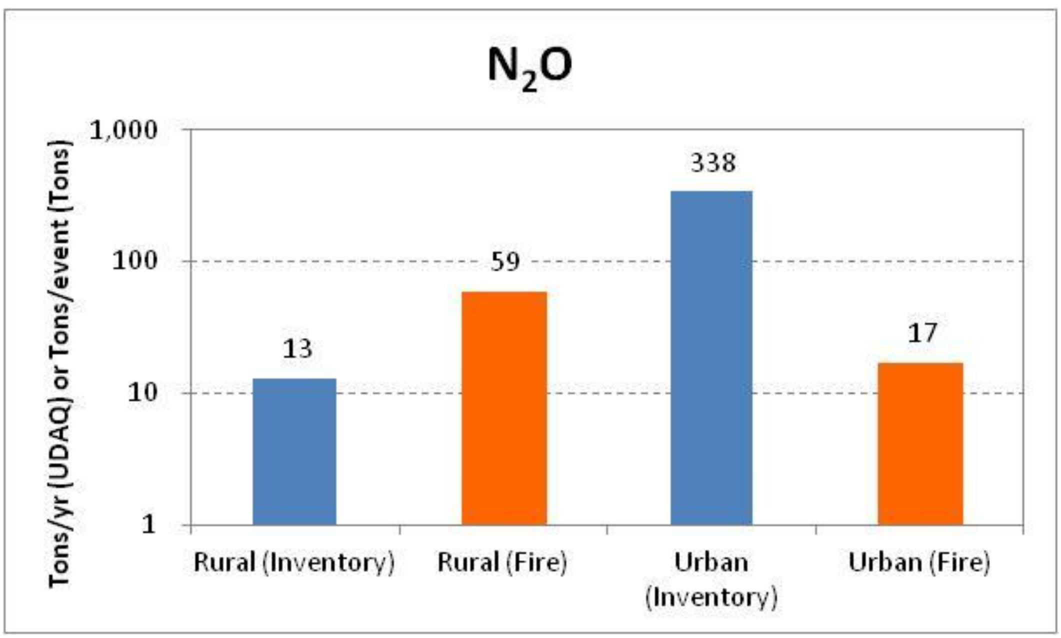


Figure 5.13: Fire-derived N_2O emissions for the examined Rural and Urban counties compared to N_2O emissions from UDAQ (2014b) and NEI (2011) emission inventories.

Summary

Wildfires can produce large volumes of locally and regionally concentrated air pollutants over a relatively short time periods, days to weeks, which can adversely affect the health and welfare of downwind populations and ecosystems. The exercise presented herein shows the modeled derived wildfire air pollutant emissions for predicted burns in two regions characteristic to Utah: Rural (Sanpete and Juab counties) and Urban (Weber, Morgan, Davis, and Salt Lake counties). Table 5.2 summarizes the model (tons per event) and emissions inventory compiled (tons/yr) total emissions. As shown, within the Urban region, with the exception of CH_4, wildfires generally added less than 10% of the examined pollutants to the airshed. This should not be misconstrued to mean these wildfires fires in urban areas are insignificant, as in-the-plume concentrations and short-duration (acute) exposures can have significant impacts to human health and the local ecosystems Table 5.2 also shows that within the Rural counties the short-term wildfire events dramatically added to the total atmospheric loading for CO (270%), $PM_{2.5}$ (260%), CO_2 (120%), CH_4 (1,884%), and N_2O (458%) when compared to the annual anthropogenic emission inventory. Further, it should be noted that the fire-derived pollutant emissions are to be taken as additional air quality burdens above the inventoried, anthropogenic-type emissions.

Table 5.2. Compiled Rural and Urban wildfire-derived air pollutant emissions compared to available emission inventories.

	CO	$PM_{2.5}$	NO_x	VOCs	NH_3	CO_2	CH_4	N_2O
Emissions Inventory (UDAQ, 2014b; NEI 2011)								
Rural (tons)	17,822	1,559	3,635	41,378	3,013	464,884	134	13
Urban (tons)	175.333	8,534	39,748	56,671	3,441	8,580,593	456	338
Modeled Wildfire Emissions								
Rural (t/yr)	48,041	4,047	594	4,366	295	557,733	2,521	59
Urban (t/yr)	13,837	1,165	171	1,252	85	160,645	726	17
Percent Fire Emissions Relative to UDAQ/NEI Emissions Inventories								
Rural (%)	270	260	16	11	10	120	1884	458
Urban (%)	8	14	>1	2	3	2	159	5

It should be noted that although it is tempting to scale up these emission calculations for the entire state of Utah, or perhaps the Intermountain West, this approach should be discouraged owing to the heterogeneous nature of the different areas, both intra- and interstate. Reasonable emission factors could only be accomplished with individualized and detailed modeling and assessment.

References

AirNow, 2016, AIRNow Archives, Air Quality Maps, Archive, Monthly Overview, August 2015, https://airnow.gov/index.cfm?action=airnow.mapsarchivecalendar, accessed Dec. 2016.

Akagi, S.k., R.J. Yokelson, C. Wiedinmeyer, M.J. Alvardo, J.S. Reid, T. Karl, J.D. Crounse, and P.O. Wennberg, 2011, Emission factors for open and domestic biomass burning for use in atmospheric models, *Atmos. Chem. Phys.*, 11, 4039-4072.

W. Battye and R. Battye, 2002, Development of Emissions Inventory Methods for Wildland Fires, Final Report, prepared for T.G. Pace, D205-01, U.S. EPA, Cntract 68-D-98-046, Feb. 2002.

EPA, 2016a, Wildfire Smoke: A Guide for Public Helath Officials, U.S. Enviornmental Protection Agency, U.S. Firest Service, U.S. Centers for Disease Control and Prevention, California Air Resources Board, reivsed May 2016.

EPA, 2016b, AP-42: Compilation of Air Emission Factors, 5th edition and proposed/revised/new emission factors, https://www.epa.gov/air-emissions-factors-and-quantification/ap-42-compilation-air-emission-factors, accessed Nov. 2016.

EPA, 2016c, Understanding Global Warming Potentials, Greenhouse Gas Emissions, U.S. Environmental Protection Agency, https://www.epa.gov/ghgemissions/understanding-global-warming-potentials, accesed Nov. 2016.

Finlay, S.E., A. Moffat, R. Gazzard, D. Baker, and V. Murray, 2012, Health impacts of wildfires, PLoS Currents, 2012 November 2,4; issue 205193, PMC 3492003, doi: 10.1371/4f959951cce2c.

Herbert, G.R., 2016, Utah 2015 8-hour Ozone Designation Recommendation, letter to EPA Region 8, http://deq.utah.gov/Pollutants/O/ozone/docs/2016-10-Utah-Governor-Ozone-Area-Recommendation.pdf , dated sept. 28, 2016, accessed Oct. 2016.

IDEQ, 2016, Real-Time Air Monitoring, Department of Environmental Quality, Dec. 2014, http://airquality.deq.idaho.gov/, accessed Dec. 2016.

IDEQ, 2014, Cache Valley Idaho PM2.5 Nonattaiment Area State Implementation Plan Amendment, State of Idaho, Department of Environmental Quality, Dec. 2014, https://www.deq.idaho.gov/media/1118467/cache-valley-pm25-nonattainment-area-sip-amendment.pdf, Jaffe, D.A. and N.L.Wigder, 2012, Ozone production from wildfires: A critical review, *Atmos. Env.*, 51, 1-10.

Jaffe, D., W. Hafner, D. Chand, A.Westerling, and D. Spacklen, 2008, Interannual varisations in PM2.5 due to wildfires in the western United States, *Environ. Sci. Technol*, 42, 2812-2818.

Kampa, M. and E. Castanas, 2007, Human helath effects of air pollution, *Env. Pollut.*, 151, 362-367.

Kenword, A., D. Adams-Smith, and U. Raja, 2013, Wildfires and Air Pollutanst: The hidden health hazards of climate Change, Climate Central, Princeton, NJ, http://assets.climatecentral.org/pdfs/WildfiresAndAirPollution.pdf , accessed Sept. 2106.

Knorr, W., V. Lehsten, and A. Arneth, 2012, Determinants and predictability of global wildfire emissions, *Atmos. Chem. Phys.*, 12, 6845-6861.

Kreidenweis, S.M., J.L. Collett, H. Moosmuller, W.P. Arnott, W. Hao, and W.C. Malm, 2010, Overview of the Fire Lab at Missoula Experiments (FLAME), abstract #A21B-0060, presented at the 2010 Fall Meeting of the American Geophysical Union, San Francisco, CA, Dec. 2010.

Liu, J.C., G. Pereira, S.A. Uhl, M.A. Bravo, and M.L. Bell, 2015, A systematic review of the physical health impacts from non-occupational exposure to wildfire smoke, *Environ. Res.*, 136, 120-132.

Mallia, D.V., J.C. Lin, S. Urbanski, J. Ehleringer, and T. Nehrkorn, 2015, Impacts of upwind wildfire emissions on CO, CO2, and PM2.5 conccnetrations in Salt Lake City, UT, *JGR-Atmos.*, 120(1), 16 January 2915, 147-166.

Moeltner, K., M.-K. Kim, E. Zhu, and W. Yang, 2013, Wildfire smoke and health impacts: A closer look at fire attributes and their marginal effects, *J. Environ. Econ. and Manag.*, 66(3), 476-496.

Mutch, R.W. and W.A. Cook., 1996, "Restoring fire to ecosystems: methods vary with land management goals, U.S. Forest Service, int_gtr341, http://www.fs.fed.us/rm/pubs_int/int_gtr341/int_gtr341_009_011.pdf, accessed November 2016.

NEI, 2011, National Emissions Inventory (NEI) Data, US Environmental Protection Agency, Air Emissions Inventories, https://www.epa.gov/air-emissions-inventories/2011-national-emissions-inventory-nei-data, accessed Oct. 2016.

NIH, 2016, Air Pollution: Your Environment, Your Health, National Institute of Environmental Health Sciences, https://www.niehs.nih.gov/health/topics/agents/air-pollution/, accessed Nov. 2016.

ORNL DAAC, 2013, NASA's Oak Ridge National Laboratory's Distributed Active Center North American Carbon Program Aboveground Biomass and Carbon Baseline Data, Version 2 (NBCD 2000), NBCD_MZxx_FIA_ALD_biomass.tif, https://daac.ornl.gov/NACP/guides/NBCD_2000_V2.html, accessed October 2016.

Reid, C.E., M. Brauer, F.H. Johnston, M. Jerrett, J.R. Balmes, and C.T. Elliot, 2016, Critical review of health impacts of wildfire smoke exposure, *Environ. Health Prespect.*, 124(9), 1334-1343.

Richardson, L.A., P.A. Champ, and J.B. Loomis, 2012, The hidden cost of wildfires: Economic valluation of health effects of wildfire smole exposure in Southern California, *J. of Forest Econ.*, 18, 14-35.

Sandborn Map Company, 2013, Western Wildfire Risk Assessment, commissioned by the Oregon Department of Forestry.

Stockwell, C.E., R.J. Yokelson, S.M. Kreidenweis, A.L. Robinson, P.J. DeMott, R.C. Sullivan, J. Reardon, K.C.Ryan, D.W.T. Griffith, and L. Stevens, 2014, Trace Gas Emissions from Combustion of Peat, Crop Residue, Biofuels, Grasses, and Other Fuels: Configuration and FTIR Component of the Fourth Fire Lab at Missoula Experiment (FLAME-4), *Atmos. Chem. Phys.*, 14, 9727-9754.

UDAQ, 2016a, Particulate PM2.5 Data Archive, State of Utah Department of Environmental Quality, Division of Air Quality, Air Monitoring Center, http://www.airmonitoring.utah.gov/dataarchive/archpm25.htm, accessed Dec. 2016.

UDAQ, 2016b, Utah Area Designation recommendations for the 2015 8-Hour Ozone National Ambient Air Quality Standard, State of Utah Department of Environmental Quality, Division of Air Quality, Sept. 2016, http://deq.utah.gov/Pollutants/O/ozone/docs/2016-10-Utah-Ozone-Designation-Recommendation-Staff-Analysis.pdf?v=2, accessed Oct. 2016.

UDAQ, 2014a, Utah: State Implementation Plan, Control Measures for Area and Point Sources, Fine Particulate Matter, PM2.5 SIP for the Salt Lake City, UT Nonattainment Area, Section IX, Part A.21, Dec, 2014, http://www.deq.utah.gov/Laws_Rules/daq/sip/docs/2014/12Dec/SIP%20IX.A.21_SLC_FINAL_Adopted%2012-3-14.pdf, accessed Oct. 2016.

UDAQ, 2014b, Statewide Emissions Inventory Program, Statewide Emission Inventories, 2014, State of Utah Department of Environmental Quality, Division of Air Quality, http://www.deq.utah.gov/ProgramsServices/programs/air/emissionsinventories/docs/2016/2014-State-Summary-by-Source.pdf, accessed Oct. 2016.

Urbanski, S.P., 2013, Combustion efficiency and emission factors for wildfire-season fires in mixed conifer forests of the northern Rocky Mountains, US, *Atmos. Chem. Phys.*, 13, 7241-7262.

Urbanski, S.P., W.M. Hao, and S. Baker, 2009, Chemical Composition of Wildland Fire Emissions, Chptr 4, in Developments in Environmental Science, Volume 8, A. Bytnerowicz, M. Arbaugh, A. Riebau, and C. Anderson, eds., 2009 Elsevier B.V., ISSN: 1474-8177/DOI:10.1016/S1474-8177(08)00004-1.

Vedal, S. and S.J. Dutton, 2006, Wildfire air pollution and daily mortality in a large urban area, *Environ. Res.*, 102, 29-35.

WAQ, 2016, World Air Quality, Air Pollution in World: Real-time Air Quality Index Visual Map, http://aqicn.org/map/world/#@g/18.6608/69.6094/2z, accessed Dec. 2016.

Warneke, C., J.M. Roberts, J.P. Schartz, R.J. Yokelson, B. Pierce, J.A. ded Gouw, K. Floyd, D.M. Murphy, R._S. Gao, G.J. Frost, M.K. Trainer, S.A. McKeen, J.B. Burkholder, J.S. Daniel, E.J. Williams, and D.W. Fahey, 2015, Fire Influence and Regional and Global Environments Experiment (FIREX): The impact of biomass burning on climate and air quality: An intensive study of Western North America Fires: NOAA Field and Laboratory Studies during 2016-2019, white paper, http://www.esrl.noaa.gov/csd/projects/firex/, accessed Sept. 2016.

Yokelson, R.J., I.R. Burling, J.B. Gilman, C.Warneke, C.E. Stockwell, J. de Gouw, S.K. Akagi, S.P. Urbanski, P. Veres, J.M. Roberts, W.C. Kuster, J. Reardon, D.W.T. Griffith, T.J. Johnson, S. Hosseini, J.W. Miller, D.R. Cocker III, H. Jung, and D.R. Weise, 2013, Coupling field and laboratory measurements to estimate the emission factors of identified and unidentified trace gases for prescribed fires, *Atmos. Chem. Phys.*, 13, 89-116.

Chapter 6: Impacts of Wildfire on Water Quality

Nancy Mesner and Jacob Stout
Department of Watershed Sciences
Utah State University

Introduction

Wildfire has multiple impacts on water quality (Tecle and Neary 2015). Loss of vegetation fundamentally changes runoff and infiltration characteristics of a local watershed (Ice, et al 2004). Intense fires may result in crusted or hardened soils that exacerbate these changes. Impacts are typically most critical in the months and years immediately following any fire, although the extent, intensity (heat energy produced) and severity (loss of vegetation) of the fire will determine total impact and the ability of the system to be restored to pre-fire runoff conditions. Fires may fundamentally modify a watershed's runoff characteristics, resulting in more frequent flooding, more major floods and more rapid runoff response following a storm event (Ice et al, 2004; Murphy, 2012). Post fire base (summer and fall) flows may also be diminished or lost due to reduced infiltration and poorly functioning riparian corridors (Ice et al 2004).

Nutrients, metals, salts and sediments released following a fire are of particular concern to water quality. Depending on basic watershed characteristics, the carbon, nutrients and other basic elements released when vegetation burns may be held on the landscape, entrained in surface runoff and delivered to receiving waters, transported by winds, or infiltrated into subsurface waters. Surface concentrations downstream of fire may increase by several orders of magnitude, resulting in short term exceedances of state and EPA drinking water and ambient water quality criteria (Tercle and Neary 2015). Nitrogen compounds such as ammonia are directly toxic to aquatic life. Under well oxygenated stream conditions, however, ammonia is typically transformed to other nitrogen compounds, such as nitrate and nitrite. Nitrates and nitrites in drinking water are toxic to humans and livestock at relatively high concentrations (Meixner and Wohlgemuth, 2004; Murphy, 2012). Phosphorus is not toxic but if entrained in surface runoff, both phosphorus and nitrogen compounds may contribute to eutrophication of downstream lakes and reservoirs. This past year, several Utah lakes experienced some of the impacts of over-fertilization, including harmful algal blooms resulting in noxious mats of green slime on surface waters (UDWQ 2016). More importantly, these blooms may release a suite of toxins that are deadly to livestock, pets and may even harm humans if contaminated water is used for irrigation (Penrod 2015.) Even those algae which do not directly produce toxins may result in drinking water taste and odor problems following chlorination and other treatments for drinking water (Murphy, 2012). In Utah, this phenomenon has resulted in the temporary closure of drinking water sources, such as Deer Creek Reservoir in 2001 (PSOMAS 2002) and Mt Dell Reservoir in 2011 (EPA 2016), until excess nutrients were addressed and conditions improved.

Wildfires also result in increased runoff, which erodes exposed land surfaces and stream banks, causing excess sediment delivery and buildup in downstream waters (Meixner 2004). Increased sediment fills in reservoirs, reducing their life expectancy and value for storing drinking or irrigation water and for recreation. Excess sediments also directly affect our fisheries. Algae and other microscopic plants that grow on rocky stream substrates provide food for small aquatic organisms (eg. aquatic insects) that are consumed by many important fish in Utah's rivers. The gravels and cobble on stream and river beds also provide critical habitat for these tiny aquatic organisms. Sediment buildup can also reduce the availability of gravels used for nesting sites ("redds") for high value fish or may directly suffocate developing eggs. Depending on location, Utah's soils may have high concentrations of salts or heavy metals including arsenic.

Methods

Threats to water quality were evaluated in several ways. We used an empirical modeling approach to predict runoff and sediment yield using the modified universal soil loss equation (MUSLE) coupled with runoff estimates derived from the SCS Curve Number Method (Neitsch et al., 2011; Homer et al, 2015). We estimated runoff and sediment yield for a range of precipitation events under normal vegetative conditions and under assumed conditions following a severe fire. Model results for different precipitation events for average unburned conditions were compared to post-wildfire conditions.

We aggregated our 30 m pixel results from our MUSLE empirical model into average values for HUC 10 watersheds to help and overlaid these with drinking water intakes, drinking water reservoirs, and valuable blue ribbon fisheries. This allowed us to identify those areas that are most sensitive to water quality impacts following fire.

For the purposes of this study, we modeled a "severe" burn, defined as a burn that eliminates all vegetation over the burned area (i.e. experiences 100% combustion efficiency.) This is consistent with the definition used in other parts of this project (risk of fire and impacts to air quality) For the water quality assessment, the carbon, nutrients and other remnant organic material from vegetation on the burned area was not incorporated into any of the runoff models.

Modified Universal Soil Loss Equation (MUSLE)

The SCS curve number method is a relatively straightforward approach that predicts runoff changes resulting from unique precipitation events (storms) for a specific watershed based on soil conditions and land cover (Homer et al, 2015).. The modified universal soil loss equation (MUSLE) was used to estimate sediment yields based on soil characteristics, land cover, topography and runoff, which was derived from the SCS curve number method (Neitsch et al., 2011).

The Modified Universal Soil Loss Equation (MUSLE) estimates sediment yield (*sed*) in metric tons per day:

$$sed = 11.8 \cdot \left(Q_{surf} \cdot q_{peak} \cdot area_{hru}\right)^{0.56} \cdot K_{USLE} \cdot C_{USLE} \cdot P_{USLE} \cdot LS_{USLE} \cdot CFRG$$

$$Q_{surf} = \frac{\left(R_{day} - 0.2S\right)^2}{\left(R_{day} + 0.8S\right)}$$

where Q_{surf} is in mm / day and R_{day} is precipitation in mm/day. S is the retention parameter calculated using an SCS curve number (CN) which is obtained from a table of previously derived curve numbers.

$$S = 25.4\left(\frac{1000}{CN} - 10\right)$$

q_{peak} , the peak discharge volume in cubic meters per second, is calculated as

$$q_{peak} = \frac{C \cdot i \cdot Area}{3.6}$$

where C is a coefficient (set to 1), i is rain intensity in mm per hour, area is in km^2, $area_{hru}$ is the watershed area contributing to runoff, in hectares, and 3.6 is a unit conversion factor. Other variables are described in Table 6.1.

.

Table 6.1: Variables used in MUSLE

Variable	Definition	Source
K_{USLE}	Soil erodibility factor	Homer et al, 2015
C_{USLE}	Cropping factor	Purdue University,2013; Homer et al, 2015
P_{USLE}	Conservation practice factor	Texas A&M, 2016
LS_{USLE}	Topographic factor	Utah AGRC, 2016
CFRG	Coarse fragment factor	Homer et al, 2015

K_{USLE}, C_{USLE}, P_{USLE}, LS_{USLE}, and $CRFG$ are all derived from the topographic, soil and land cover data, using ESRI's ArcMap 10.3.1. Topographic and land cover data were available at 30 m resolution, which was therefore the resolution for the MUSLE empirical model.

We applied the MUSLE model to 30 m pixels under normal and burned conditions. Normal conditions utilized land cover data from the National Land Cover Database (Homer et al, 2015). For burned condition modeling, we reclassified all range, forested and agricultural land cover types to barren land. This approach assumes that all vegetation in a pixel is completely burned and also leaves no residue. Modeling residue was outside the scope of this particular modeling effort. We did not modify any of the developed, water or wetland classifications. Our MUSLE model treats each pixel as an independent event and did not route sediment or water from one pixel to the next.

Targeted Rural and Urban Sub-watersheds

We focused our attention on two areas identified within this larger project. An urban cluster represented developing areas in higher gradient forested and range lands along the Wasatch front and back, and a rural cluster represented lower gradient rangeland in the central part of the state (Figure 6.1). For water quality purposes, we identified smaller sub-watersheds within or directly adjacent to these clusters for a more detailed analysis of runoff patterns from areas with different vegetation cover (range, deciduous forest, evergreen forest and agricultural lands.) Specifically, we focused on a portion of the Weber River watershed just upstream of the urban cluster, using flows from a USGS gaging station 10128500m above the urban cluster (USGS 2016). This small watershed is close to but outside the urban cluster chosen for this project, but we feel was still representative of these conditions in the downstream portions of the region. We focused on a portion of the Sevier River watershed within the rural cluster as well. These two watersheds provided a comparison of runoff and sediment yield responses from rangelands and forested landscapes typical of other areas across Utah.

In these designated sub-watershed areas (representing the rural and urban clusters) we modeled normal vegetation and post burn runoff, sediment concentration and sediment yields under hypothetical rain events ranging from 20 mm/day to 75 mm/day (~0.8 in/day to 3 in/day). We averaged the outputs for areas of common vegetation types within our targeted sub-watershed.

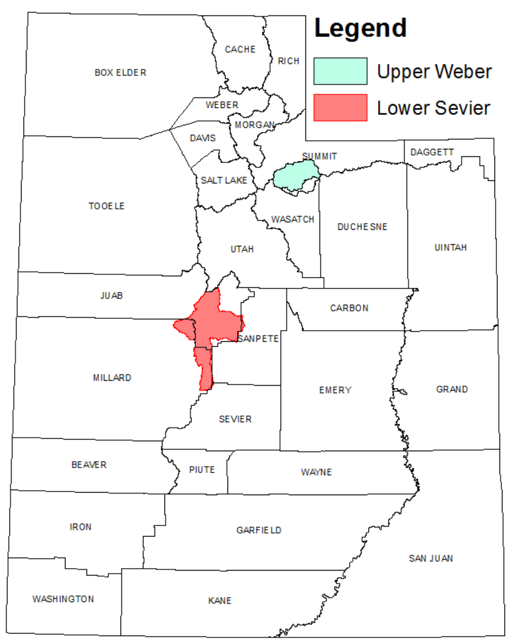


Figure 6.1: Suburban and urban clusters

Statewide Application of MUSLE model

Statewide, we modeled yields for each 30 m pixel following a 50 mm/day storm under unburned and burned conditions. We aggregated these results, averaging unburned and burned responses within HUC 10 sub-watersheds (Seaber et al, 1987). Of particular interest to this study, because of the potential economic consequences to the state, are those areas of overlap between drinking water resources or blue ribbon fisheries and areas predicted to have greater runoff or sediment yields following a fire. To identify areas of particular concern, we overlaid maps of HUC 10 aggregated responses with locations of drinking water inlets and reservoirs and with locations of blue ribbon water bodies.

Burn Ratios

For each 30 m range or forest pixel, we calculated a ratio of burn to normal estimates of runoff, sediment concentration and sediment yield. At a rudimentary level, this approach provides a statewide snapshot of sensitivity to fire impacts. This approach also allows a comparison of land covers in watersheds of different areas. By aggregating these ratios in larger watershed areas, we were able to estimate potential downstream impact to drinking water inlets and reservoirs, and to blue ribbon fisheries.

Soil and Water Assessment Tool (SWAT) model

The time available for this project limited our ability to develop more mechanistic models that could predict dynamic changes in nutrients, metals or salts following a fire. We successfully parameterized a mechanistic model (SWAT) for one sub-watershed in northern Utah and were able to predict flow and sediments under unburned and burned conditions for a single year with a significant storm event.

The Soil and Water Assessment Tool (SWAT) predicts runoff and water quality from a defined watershed area throughout a year (Texas A&M, 2016). This model is driven by topography, soil, land cover and weather and is highly parameterized, requiring a detailed knowledge of conditions within the watershed. Internal to the model is the MUSLE soil loss calculation (Neitsch et al., 2011). By using this for the storm event modeling, we could compare this model's outputs with the single storm modeling described above. We had hoped to have sufficient data for parameterizing and calibrating this model for several watersheds, but lack of field specific data prevented us from modeling the Sevier sub-watershed in our rural cluster in the southern part of the state and most of southern Utah. We limited our SWAT modeling effort, therefore, to upper Weber River Watershed adjacent to the urban cluster (described in the section above). This watershed also had the advantage of being relatively unmodified by development such as dams and diversions.

Assumptions and limitations of our methods

We only modeled discharge (flow) and total sediment released from unburned and burned areas. Our models compared these parameters from an unburned condition, based on surveyed 2011 vegetative cover available in GIS databases. To model burned areas, the physical conditions (eg. slope, soils, aspect) remained unchanged but we assumed 100 % combustion efficiency so that all vegetation was eliminated. Our approach did not incorporate any of the residual organic material retained from burned vegetation but retained on the landscape. This assumption may result in over-estimates of water and inorganic sediment and does not address nutrients, carbon and other elements that would be washed off the slopes with the sediment.

Our modeling approach also assumed that soil characteristics were not changed by the fire. Most fires burn at intensities less than 300 degrees C (Ice et al, 2004) At these temperatures, organic matter is not fully destroyed. This can increase the hydrophobicity of the soil, resulting in enhanced runoff and reduced infiltration. At more intense fires (burning above 300 degrees C), the soil's hydrophobicity is not as altered because all the organic matter is destroyed. This is the condition used in our models.

We modeled changes in water and sediment runoff from landscapes at a 30 m pixel basis. The model assumes that each pixel is independent of all others. We took the mean of the response per pixel over landscape areas of interest, resulting in average values for HUC 10 watersheds.

Major assumptions with this approach therefore were:

- All sediment evacuated from the surface is removed from the 30 m pixel/area of interest.
- No sediment release associated with burned or unburned conditions was stored within the watershed.
- Burning only changes land cover. Soil characteristics remain the same/have negligible change.
- The chosen watersheds are representative of other Utah watersheds with similar characteristics.

Results

MUSLE model predictions of changes in runoff characteristics (discharge, sediment concentration and yield) in response to increasing precipitation and to a severe burn (100% combustion efficiency) are shown in Figures 6.2 and Figure 6.3. Figure 6.2 shows predicted responses in range dominated areas and deciduous forested areas in the drainage of the Weber River adjacent to the urban cluster. Figure 6.3 shows the same predicted responses for the middle Sevier River drainage, located within the rural cluster These two sub-watersheds are characteristic of the conditions found in their respective cluster (mountainous terrain and valleys in the urban cluster and lower elevation with gentler slopes in the rural cluster). Average responses in all pixels dominated by range vegetation or by deciduous forest are shown in each figure. Each figure shows response in a non-burned condition and in a burned condition.

In all cases, the average discharge increased with increasing precipitation and in all cases the burned sub-watershed response was greater, and increased at greater rate with increasing precipitation. This same pattern was seen for sediment yield. Sediment concentrations for burned conditions were always greater than the unburned condition. In all cases, the concentrations decreased with increasing precipitation. Modeled concentrations from deciduous forests in the weber sub-watershed were appreciably higher than for range lands in the Weber area or for any of the Sevier sub-watershed run-off.

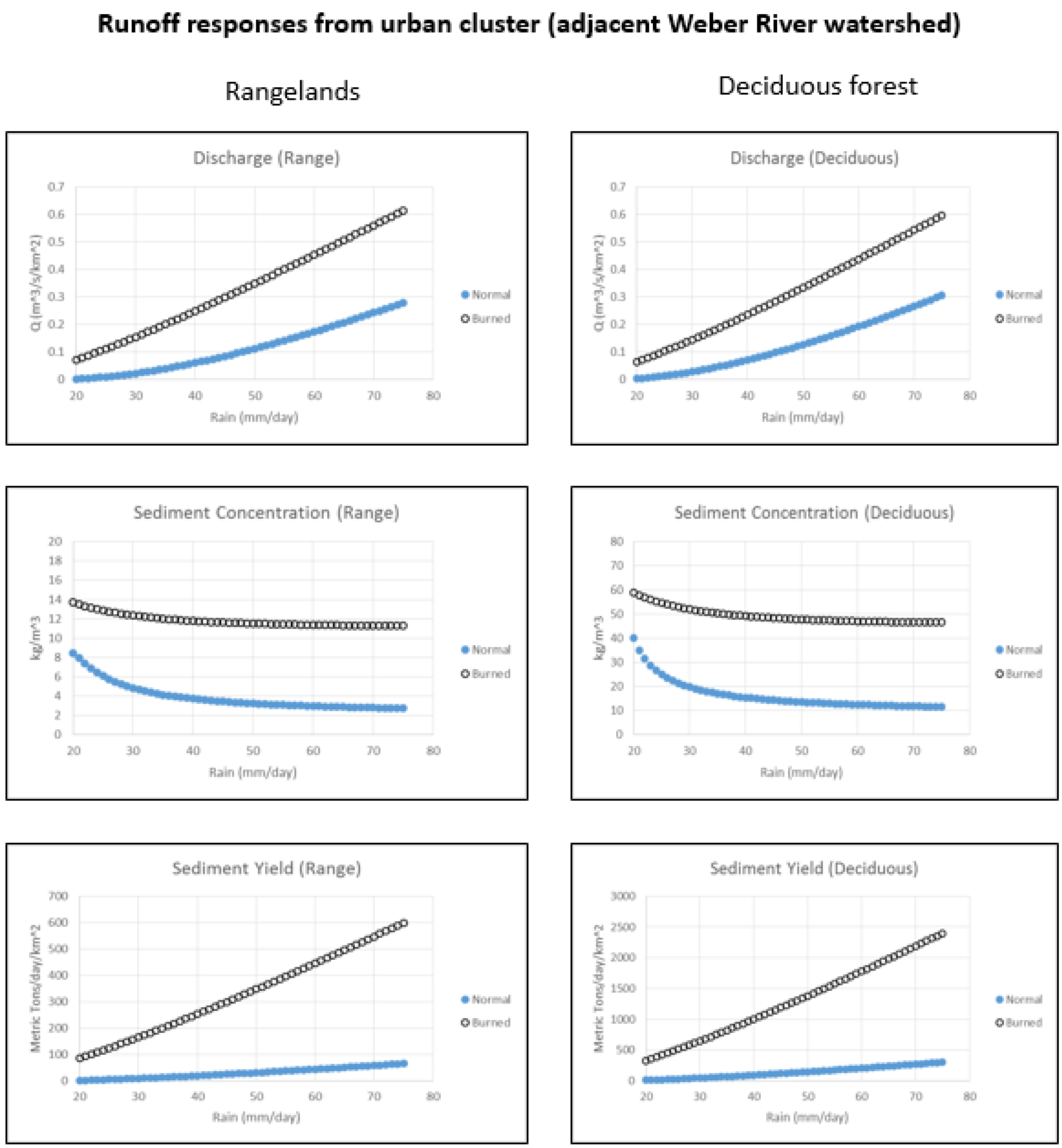


Figure 6.2: Change in runoff, sediment concentration and sediment yield in the upper-Weber sub-watershed with increasing rainfall intensities. Column on left shows cumulative values from range dominated pixels. Column on right shows cumulative responses of deciduous forest dominated pixels.

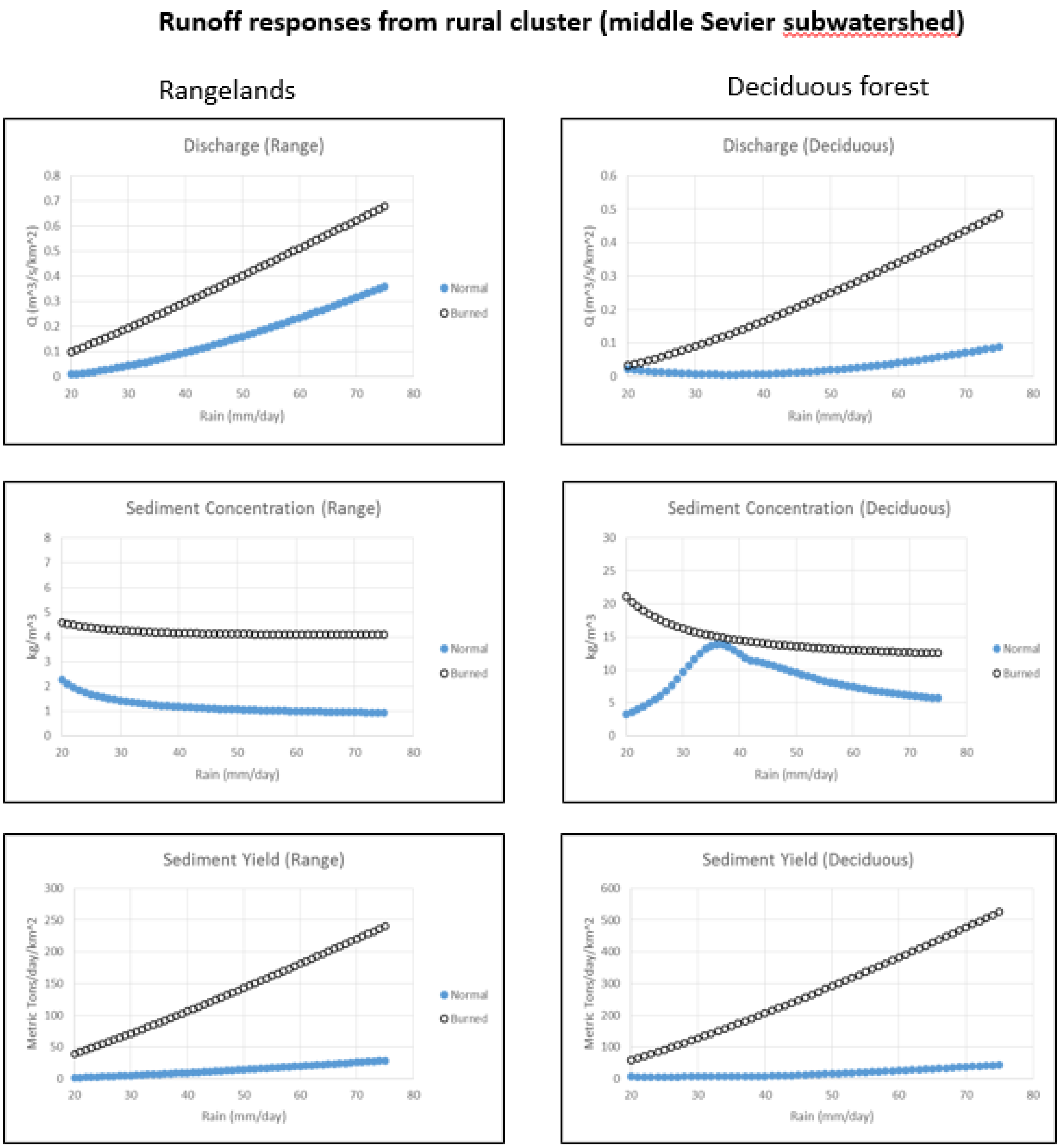


Figure 6.3. Change in runoff, sediment concentration and sediment yield in the mid-Sevier sub-watershed with increasing rainfall intensities. Column on left shows cumulative values from range dominated pixels. Column on right shows cumulative responses of deciduous forest dominated pixels.

The predicted B/N (burned conditions to normal conditions) ratios of runoff and sediment responses were also averaged for vegetation types typical of our urban and rural clusters (Table 6.2). The table contains ratios from results averaged over all precipitation events models (20 – 75 mm / day). Ratios were averaged for aggregated rangeland, evergreen forests and deciduous forests.

Runoff increases associated with fire vary from 3.4 fold in rural rangelands to a 16 fold increase in evergreen forests in the urban cluster. The increase in B/N ratios for sediment yield was greater in all cases, varying from 11 in rural rangelands to a 23 fold increase in the evergreen forests of the urban cluster. This is consistent with 7 to 20 fold sediment yield increases reported by Ice et al (2004).

Table 6.2 Burn to Normal ratios for modeled discharge, sediment concentration and sediment yield using the MUSLE model. Models were run for precipitation events from 20 to 75 mm/day and then averaged. Urban Cluster sub-watershed is the Upper Weber sub-watershed (adjacent to the Urban Cluster). A sub-watershed of the Sevier River was modeled for the Rural Cluster. Results from pixels dominated by different land cover were combined and averaged.

	Rangeland	**Evergreen Forest**	**Deciduous Forest**
A. Urban Cluster Subwatershed			
Discharge Q	5.4	15.8	4.2
Sed. Concentration	3.3	2.0	3.3
Sediment Yield	14.6	23.1	11.7
	Rangeland	**Evergreen Forest**	**Deciduous Forest**
B. Rural Cluster Subwatershed			
Discharge Q	3.4	5.0	10.9
Sed. Concentration	3.6	2.6	2.0
Sediment Yield	11.3	12.2	17.0

SWAT Model Results- Upper Weber River Watershed

We parameterized the SWAT model with land use and physical condition parameters for the Upper Weber Watershed. We calibrated the model from July through December 2014 with measured flows in the Upper Weber River gage 10128500 (USGS 2016). We achieved an excellent fit between our calibrated model and actual flows (Figure 6.3). The Nash Sutcliffe Efficiency number was 0.88, where 1 is a perfect calibration, and the slope of predicted to observed was 1.04 with an R^2= 0.927. The fit was especially tight during a large runoff event that occurred during this period.

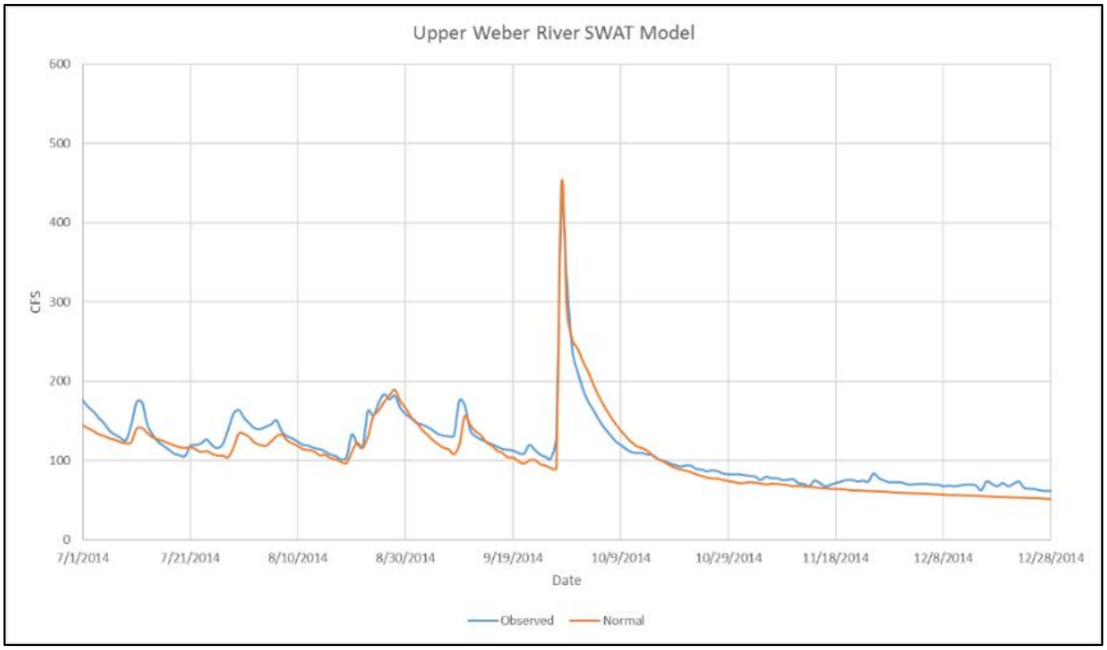


Figure 6.3. Comparison of observed flows in the Upper Weber Watershed with flows predicted by the calibrated model.

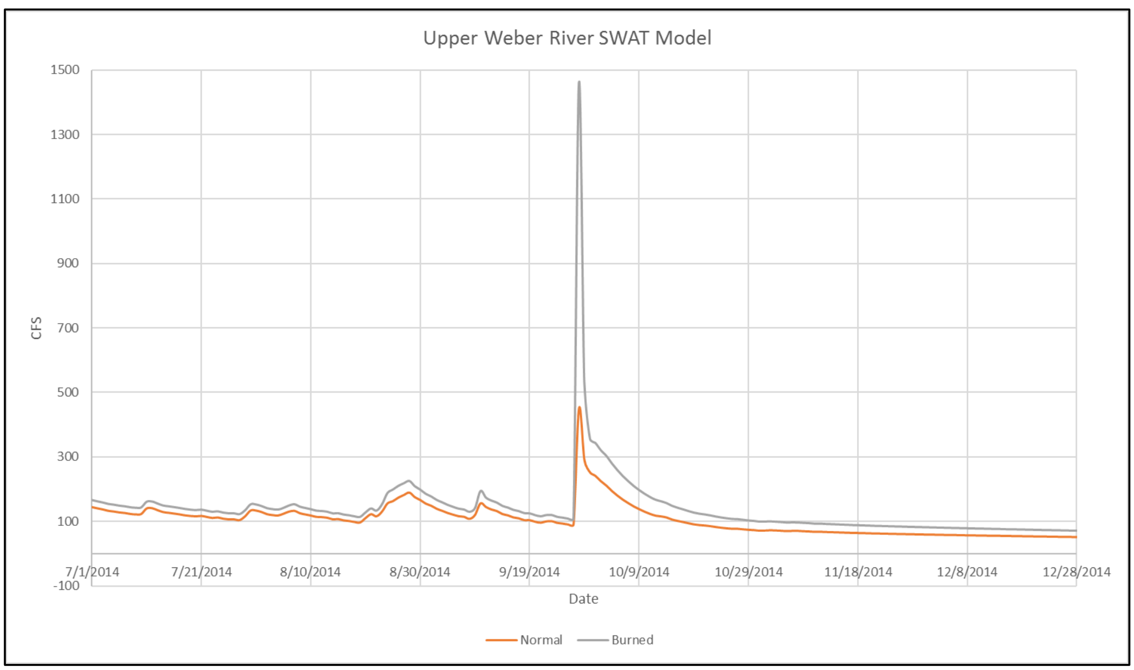


Figure 6.4. Measured discharge and estimated daily discharge after a severe burn (assumes loss of vegetation but no change in soil characteristics).

We ran the calibrated SWAT model for burned and unburned conditions, using the probabilistic burn model developed as part of this study (Figure 6.4). We modeled a severe fire only, which assumed all vegetation was eliminated but soils were not affected. Predicted discharge was slightly higher during the entire period, but showed a major spike during a storm event in late September, with a 3 fold increase in peak discharge.

During this late September storm event, a total of 53.3 mm of precipitation fell on September 28, dropping to 25.8 mm on September 29. We then predicted sediment concentrations for unburned and burned watershed conditions and calculated the B/N ratio for both days. The SWAT model predicted a 1.3 to 1.6 fold increase in sediment concentration following a burn, while it predicted a 5.2 fold increase in sediment yield on the first day of the storm and an increase of 2.4 on the second day. The higher change in sediment load (yield) on day one is likely due to a flushing effect characteristic for sediment carried during a storm. Concentrations tend to be higher as waters rise during a storm, with lower concentrations associated with the same discharge on the falling limb of a storm hydrograph.

Table 6.3. Burn to Non-burned ratios predicted by our SWAT model of a portion of the Weber River watershed (adjacent to the Urban cluster) for sediment concentration and sediment yield from a two day storm event.

	B/N ratios for SWAT model predictions	
	Sediment Concentration	Sediment Yield
9/28/2014	1.6	5.2
9/29/2014	1.3	2.4

Predictions of B/N ratios using MUSLE model

We applied the MUSLE Curve Number runoff model to individual pixels statewide, comparing a 50 mm storm event under normal and severe burn conditions. Figure 6.5 shows this response for runoff for each 30 m pixel. Figure 6.6 is a map of the same data, but showing average responses for each HUC 10 watershed area in the state. Figures 6.7 and 6.8 show average sediment yield and concentration on a HUC 10 basis.

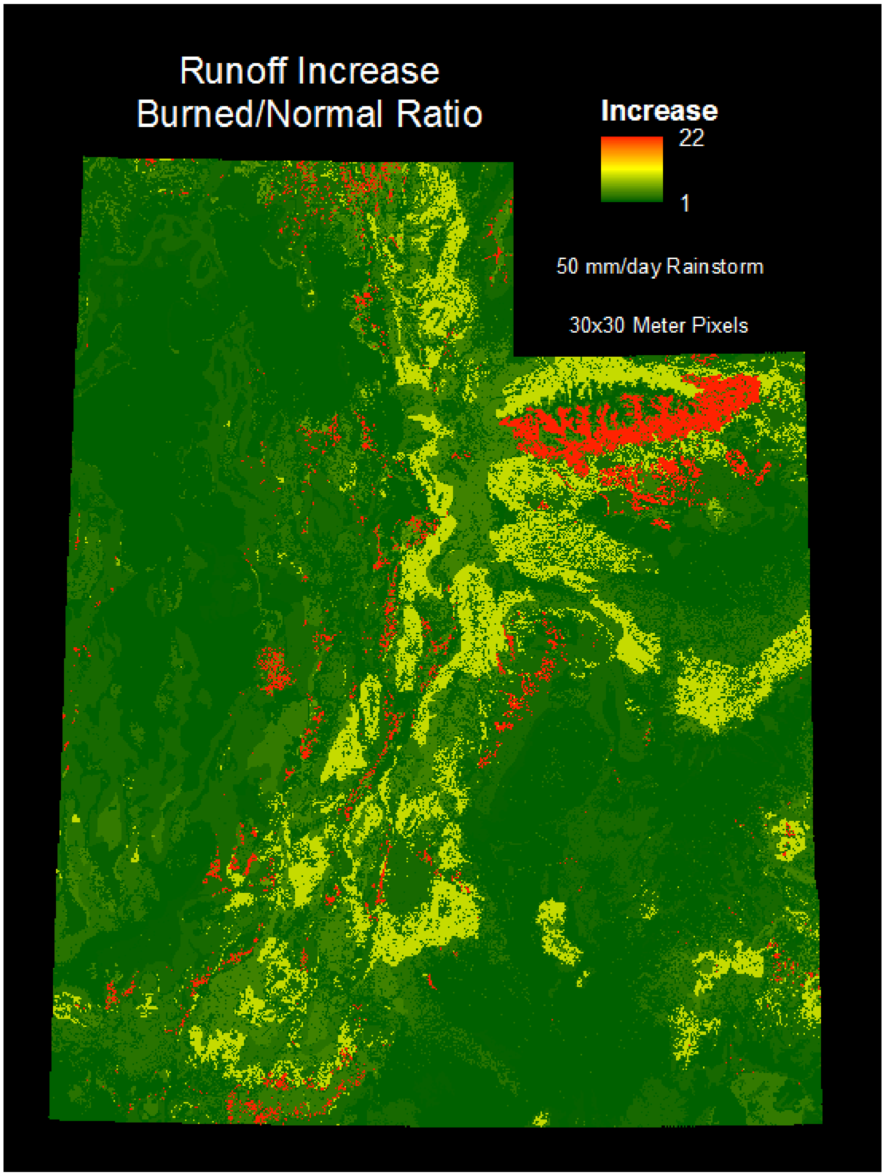


Figure 6.5: Burn/Normal ratio following a 50 mm/day storm. Map shows results from each 30 m pixel in the state.

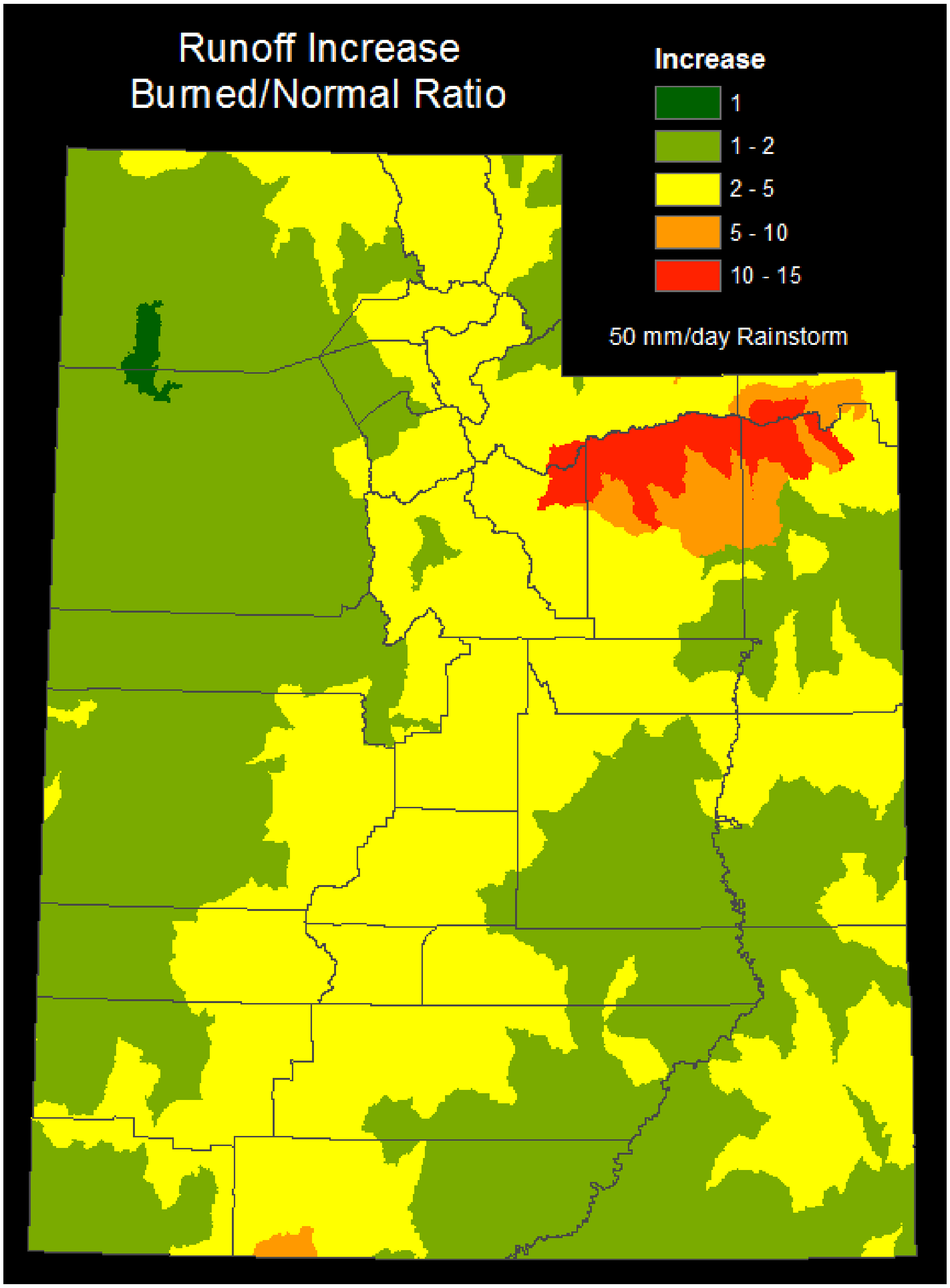


Figure 6.6: Burn /Normal ratio following a 50 mm/day storm. Map shows results averaged for each HUC 10 watershed in the state.

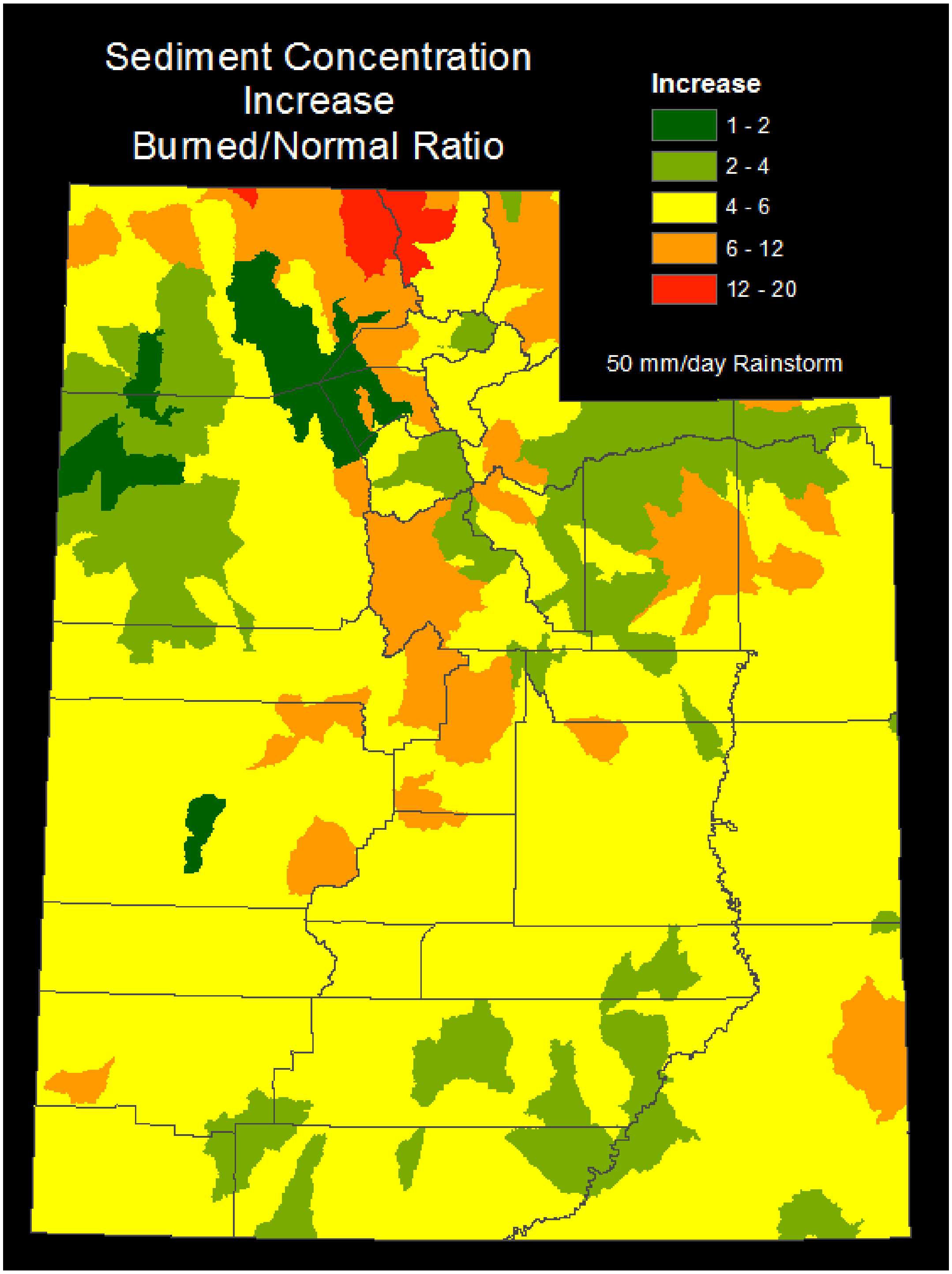


Figure 6.7. Burn /Normal ratios of sediment concentrations following a 50 mm storm. Results are for HUC 10 watershed units within the state.

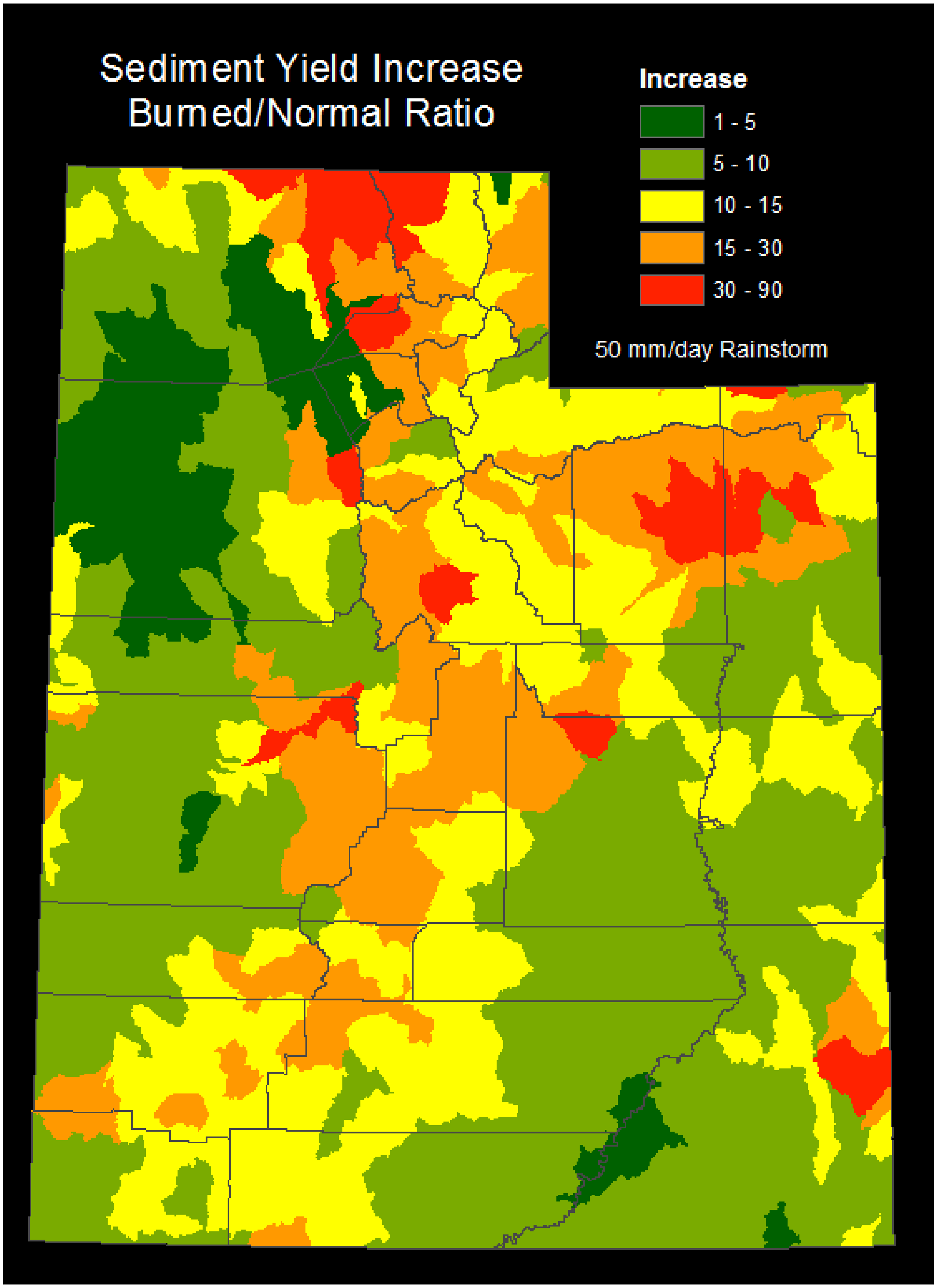


Figure 6.8: Burn /Normal ratios of sediment yields following a 50 mm storm. Results are averaged for HUC 10 watershed units within the state.

Yellow to green areas in Figures 6.5 – 6.8 have relatively low vulnerability to water quality impacts, while those at the red end of the scale are the most vulnerable to larger changes in runoff, sediment yield and sediment concentrations following an average storm of 50 mm/day. Conditions of steep slopes or more highly erosive soils will result in greater B/N ratios, although most of the red areas identified in these HUC 10 maps are actually agricultural areas. Agricultural areas have more intense vegetative cover than most of the natural landscapes in Utah, so the difference between non-burned and burned conditions are more pronounced. These areas are also typically targeted for greater protection.

The West Wide Wildfire Risk Assessment Report evaluated areas in Utah sensitive to drinking water impacts, using data from the USFS Forest to Faucets program (Weidner and Todd 2011). Utah's drinking water withdrawals and reservoirs cluster over these areas of highest sensitivity (Figure 6.9). To identify potential risk to drinking water, therefore, we overlaid locations of drinking water withdrawals and reservoirs onto maps of predicted increases in runoff, sediment concentration, and sediment yield. Several amenities fall within areas predicted to have 5 to 15 fold increases in runoff (Figure 6.10). Drinking water amenities fall within areas predicted to have 2 to 10 fold increases in sediment concentration (Figure 6.11) and within areas predicted to have 10 to 90 fold increases in sediment yield (Figure 6.12).

The analysis resulting in predicted increases in B/N ratios assumes a severe wildfire in each pixel across the state. Figure 6.13 shows the location of Utah's drinking water outlets and reservoir locations overlaid on a map of predicted fire threat across the state (Thompson et al, 2013; USDA 2016). Many of these drinking water structures fall within areas of highest fire threat. Figures 6.14 and 6.15 show the overlap of drinking water withdrawal points and reservoirs in our focus areas (urban and rural clusters). These maps also show the distribution of drinking water sensitivity (UDF, 2013). Over 15 drinking water inlets and one drinking water reservoir fall within areas of increased fire risk in the urban cluster (Figure 6.14). None fell within the rural cluster. The Drinking Water Appendix of this report contains a table of predicted changes in runoff, sediment concentration and sediment yield for all drinking water inlets and reservoirs in Utah.

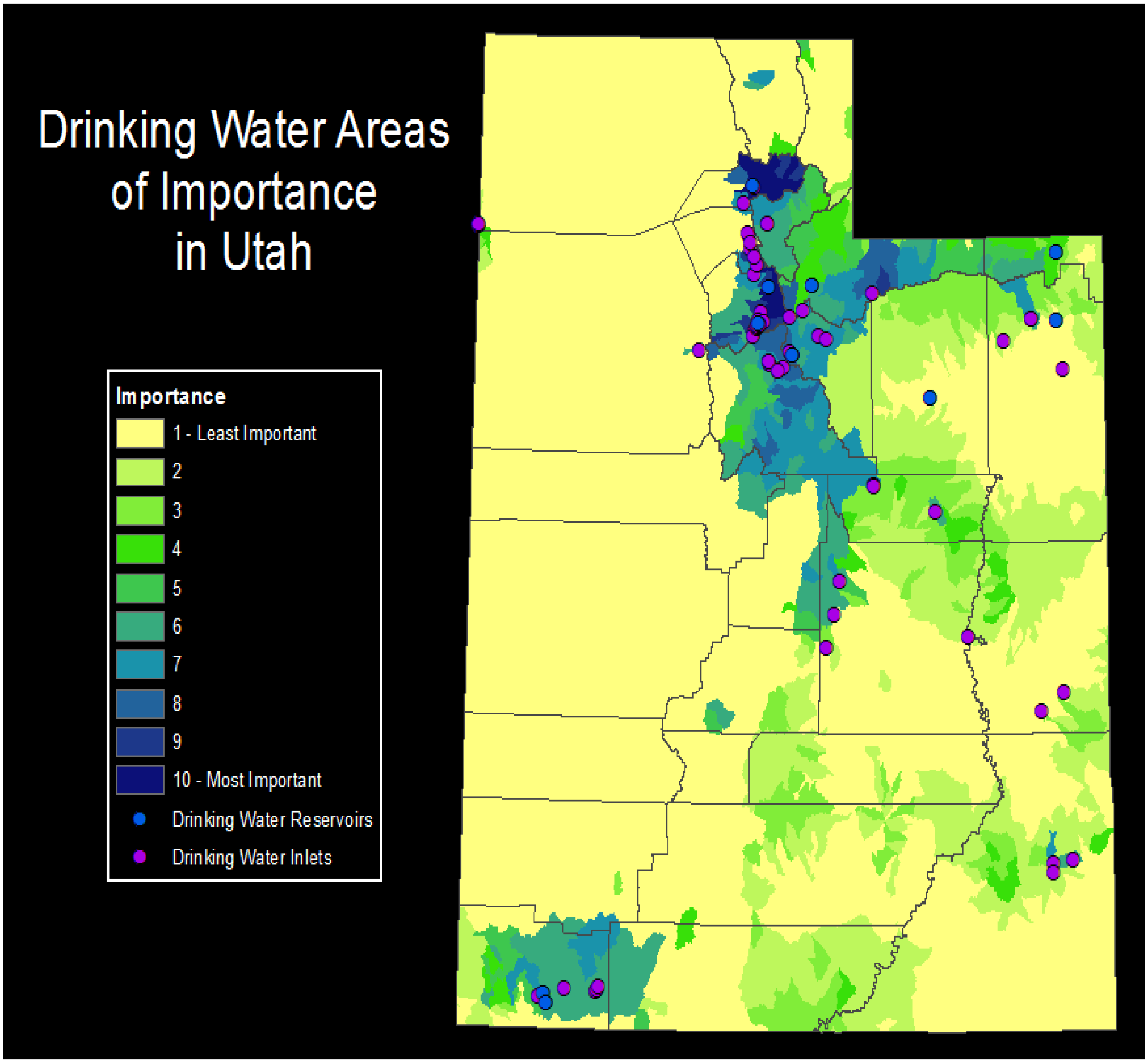


Figure 6.9: Location of drinking water structures and resources overlaid on the drinking water areas of importance as identified in the West Wide Wildfire Risk Assessment Final Report (USDA Forest Service, 2013.)

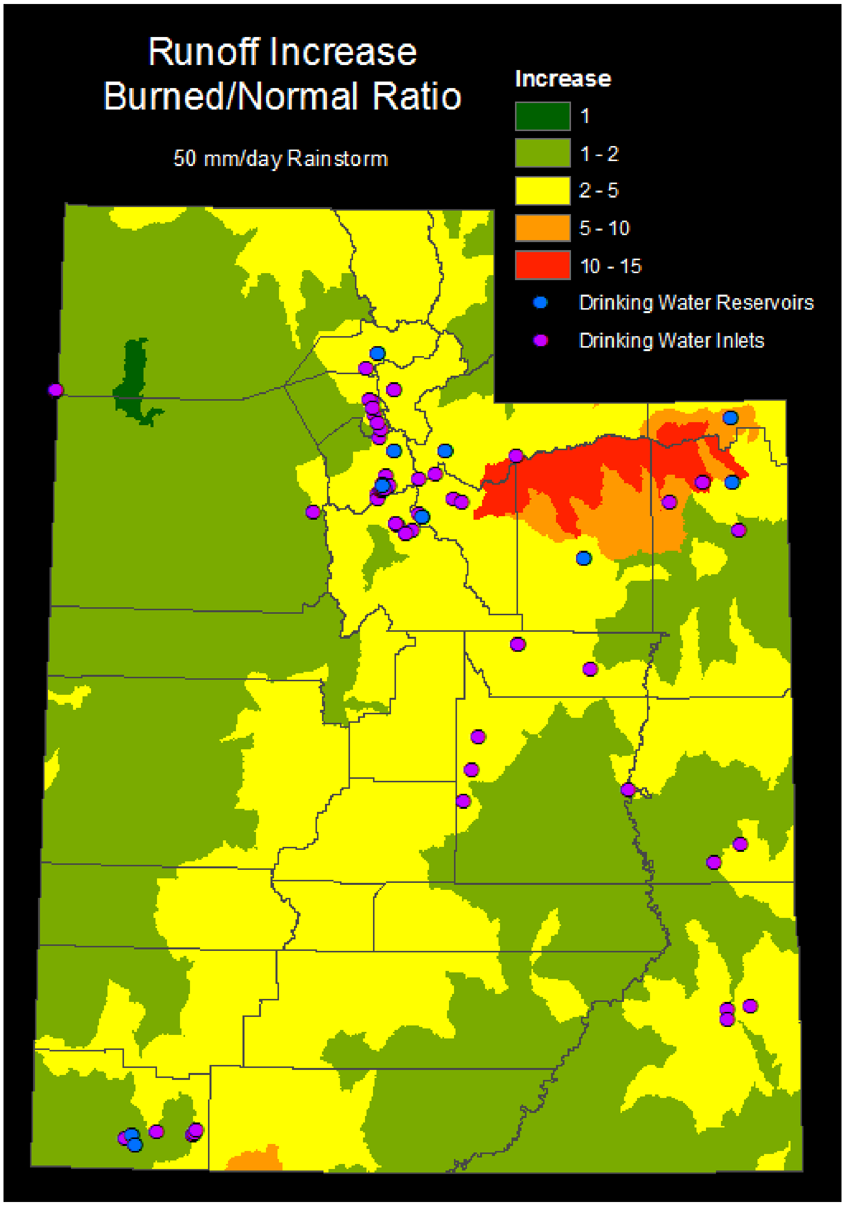


Figure 6.10: Drinking water withdrawal points (streams and reservoirs) overlaid on the Figure 6.6 map of predicted increases in runoff from a 50 mm storm, averaged over HUC 10 watersheds. Drinking water data provided by Utah Division of Drinking Water.

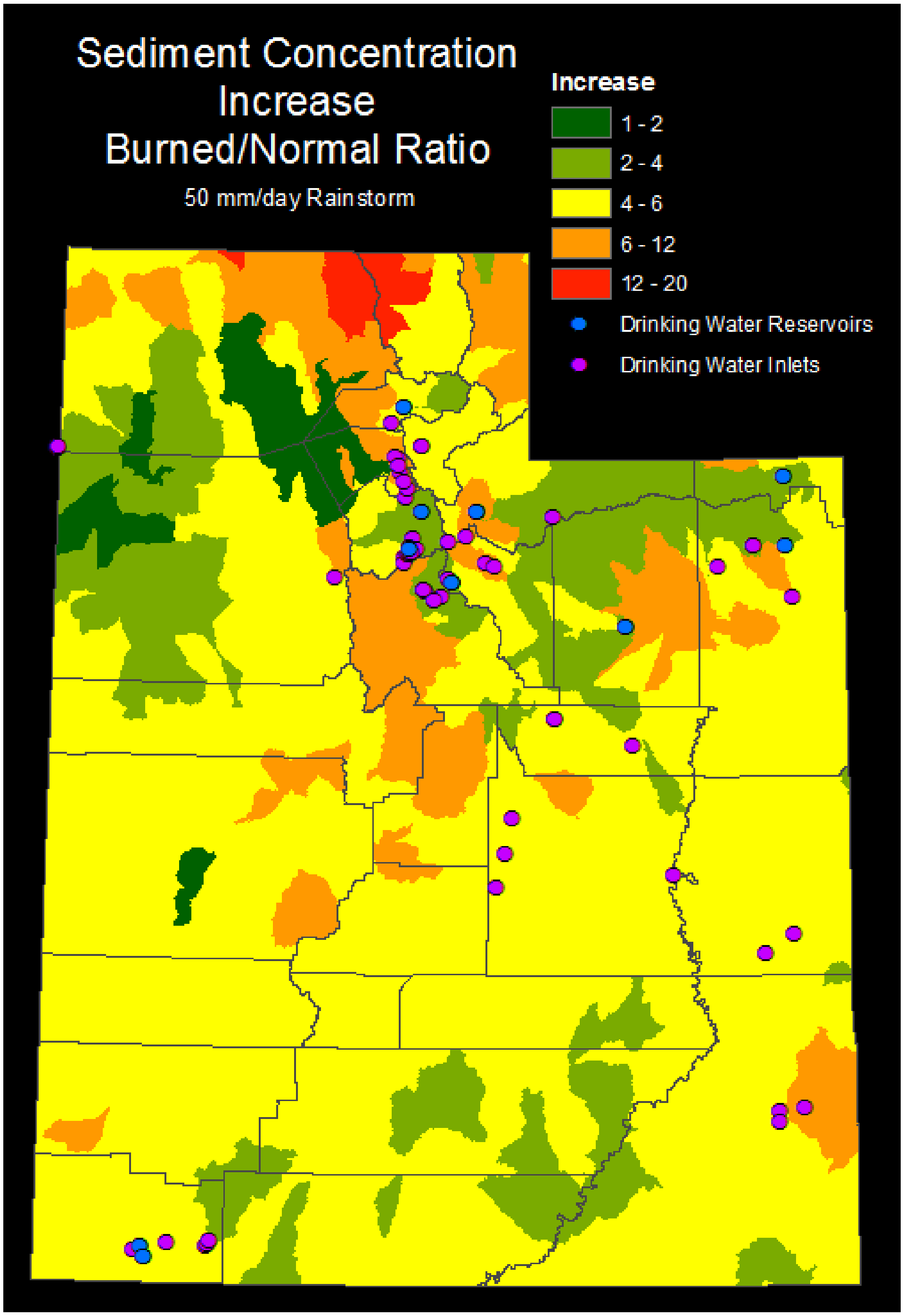


Figure 6.11: Drinking water withdrawal points (streams and reservoirs) overlaid on the Figure 6.7 map of predicted increases in sediment concentration following a 50 mm storm, averaged over HUC 10 watersheds. Drinking water data provided by Utah Division of Drinking Water.

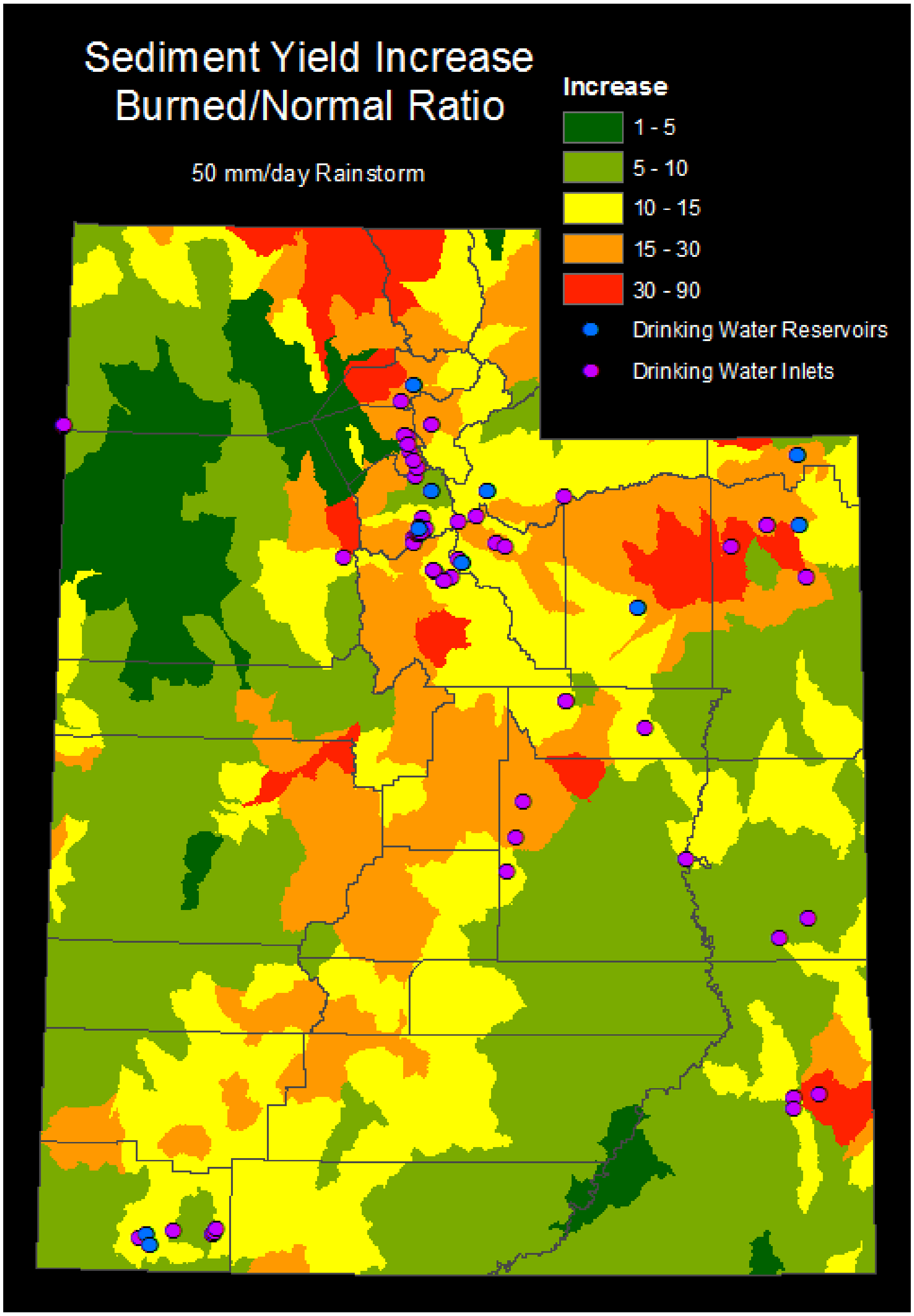


Figure 6.12: Drinking water withdrawal points (streams and reservoirs) overlaid on the Figure 6.8 map of predicted increases in sediment yield following a 50 mm storm, averaged over HUC 10 watersheds. Drinking water data provided by Utah Division of Drinking Water.

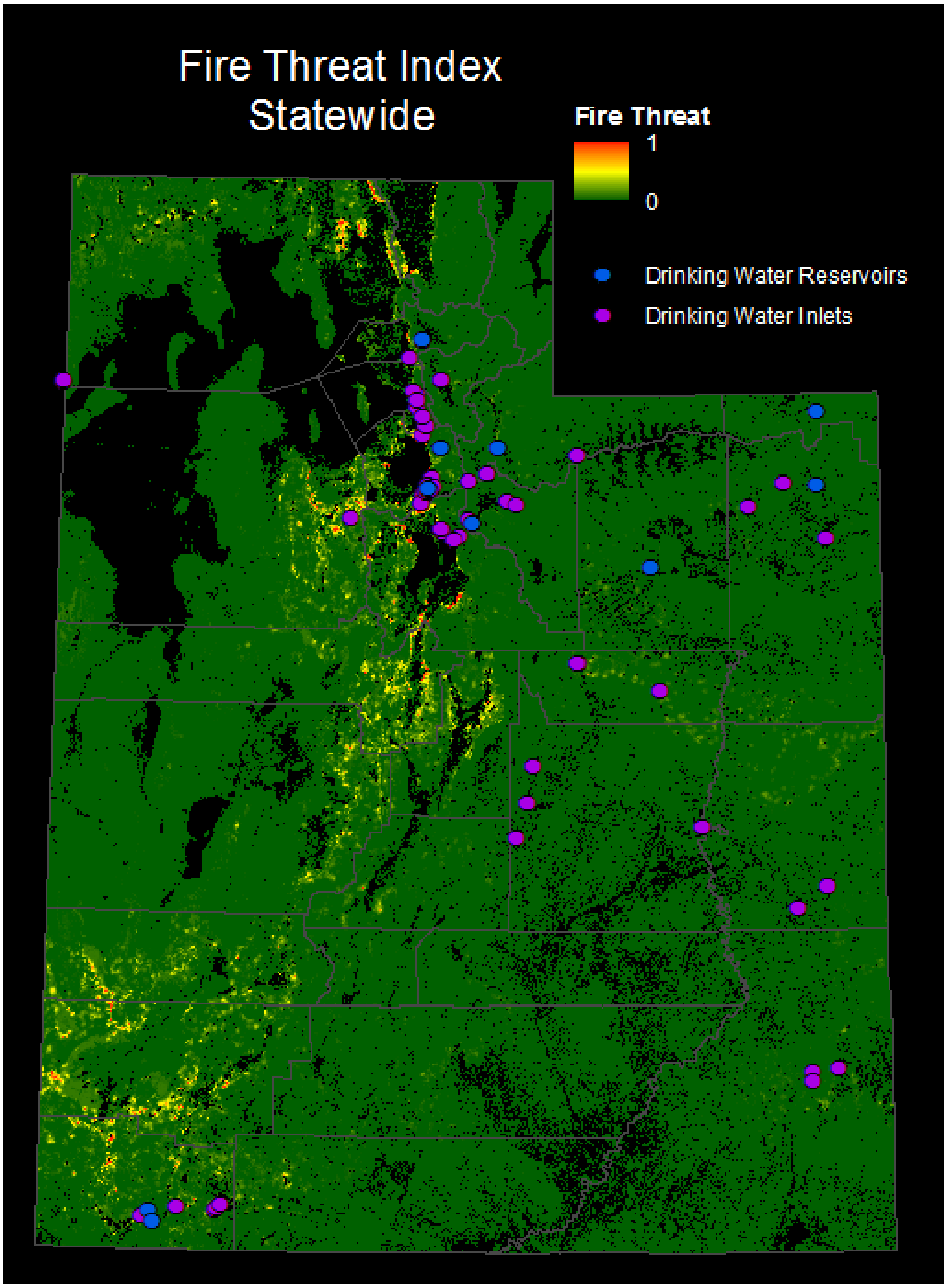
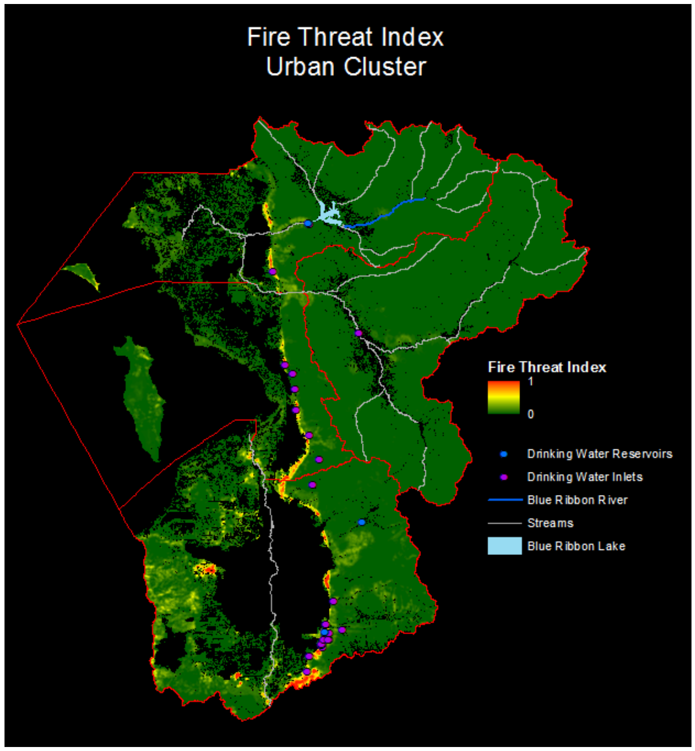


Figure 6.13: Location of drinking water structures and resources overlaid on map of Fire Threat Index as identified in the West Wide Wildfire Risk Assessment Final Report (USDA Forest Service, 2013).



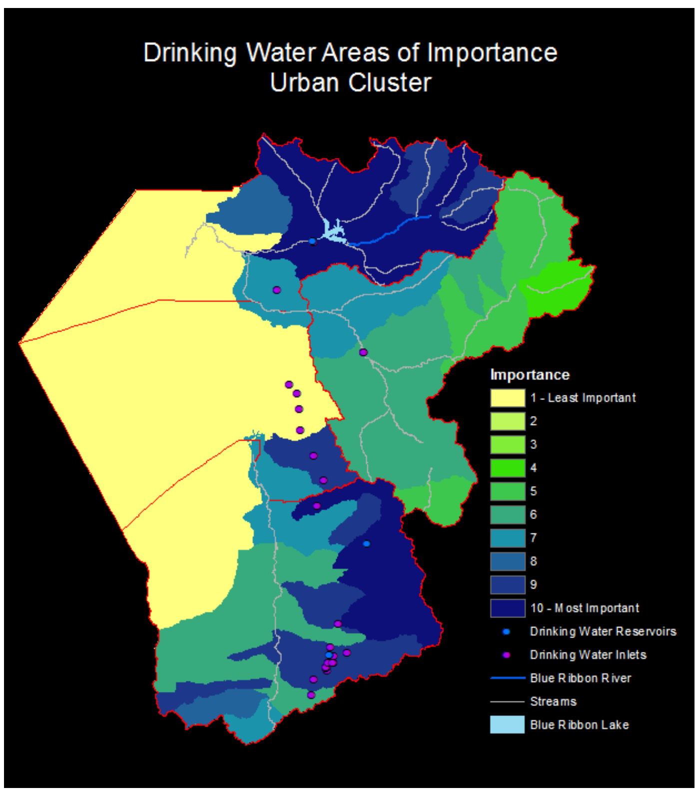


Figure 6.14: Urban cluster focus area for this study. Top map: Fire threat index in urban cluster (USDA Forest Service, 2013.) Bottom map: Drinking water structures (withdrawal points and reservoirs) overlaid on map of drinking water areas of importance (USDA Forest Service, 2013.)

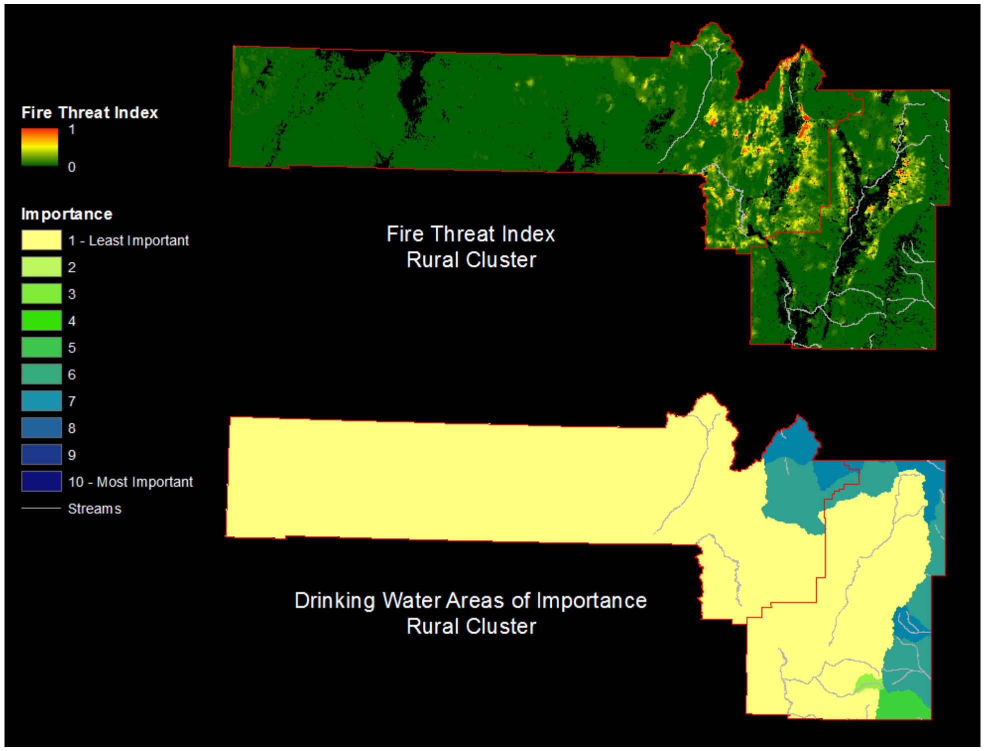


Figure 6.15: Rural Urban cluster focus area for this study. *Top map:* Fire threat index in urban cluster (USDA Forest Service, 2013). *Bottom map:* Drinking water structures (withdrawal points and reservoirs) overlaid on map of drinking water areas of importance (USDA Forest Service, 2013)

To evaluate risks to blue ribbon fisheries, we overlaid maps of these rivers, lakes and reservoirs over our distributions of predicted increases in runoff, sediment concentration and sediment yield. Figures 6.16 – 6.18 show the predicted water quality threat for these high value water bodies. Many of the fisheries are in areas of low risk (small B/N ratio). A few fisheries, however, including Bear Lake, fall into areas predicted to see 6 to 12 fold increases in sediment concentrations in post fire runoff, and to see 15-30 fold increases in sediment yield following a fire. Tables 6.4 and 6.5 provide a complete list of Utah's blue ribbon fisheries and predicted increases in runoff, sediment concentration and yield. Blue ribbon fisheries that fall within our rural and urban clusters are identified in this table.

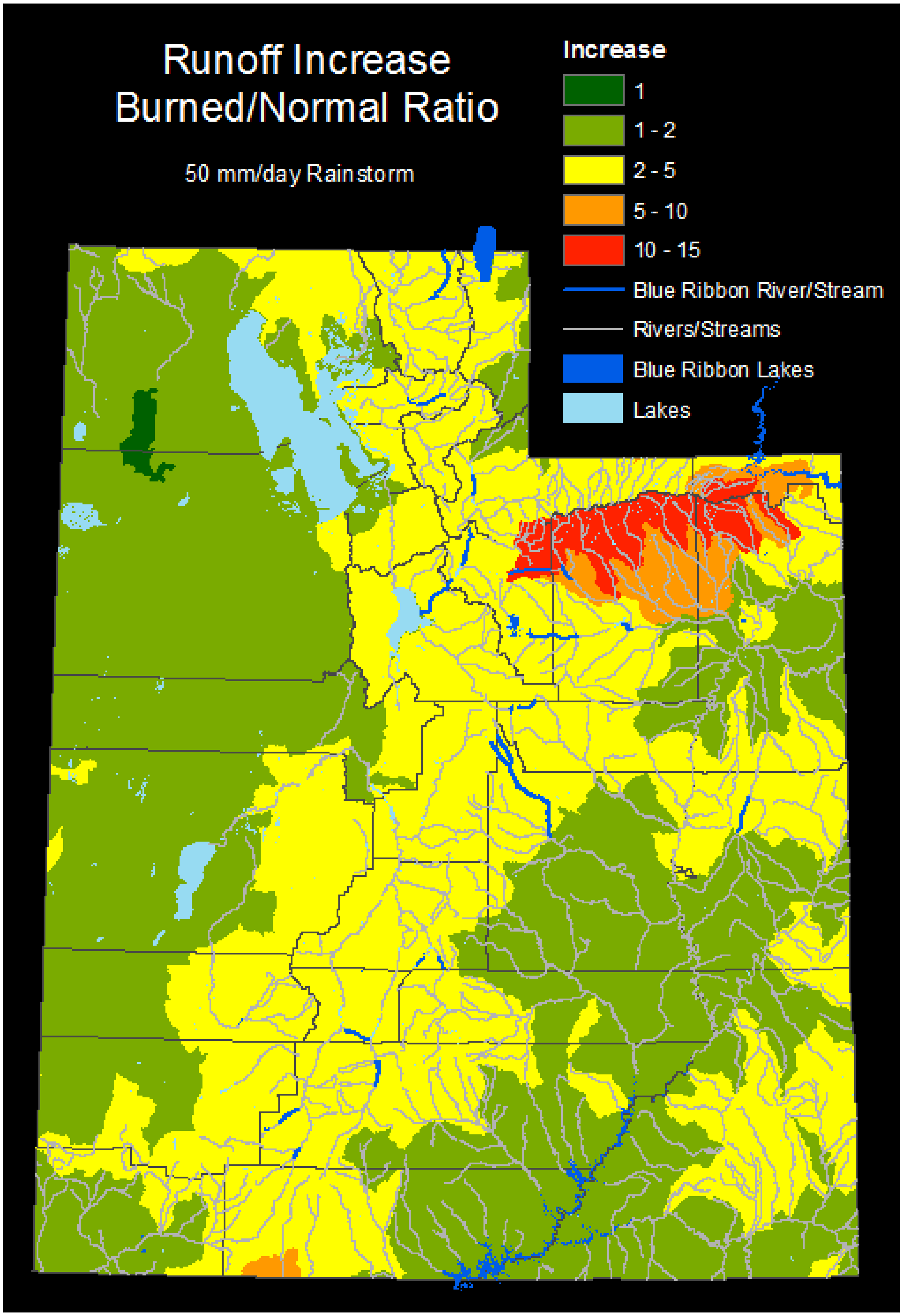


Figure 6.16: Blue ribbon fishery locations overlaid on the Figure 6.8 map of predicted increases in runoff following a 50 mm storm, averaged over HUC 10 watersheds. Blue Ribbon fisheries data provided by Utah Division of Wildlife Resources (UDWR 2016.).

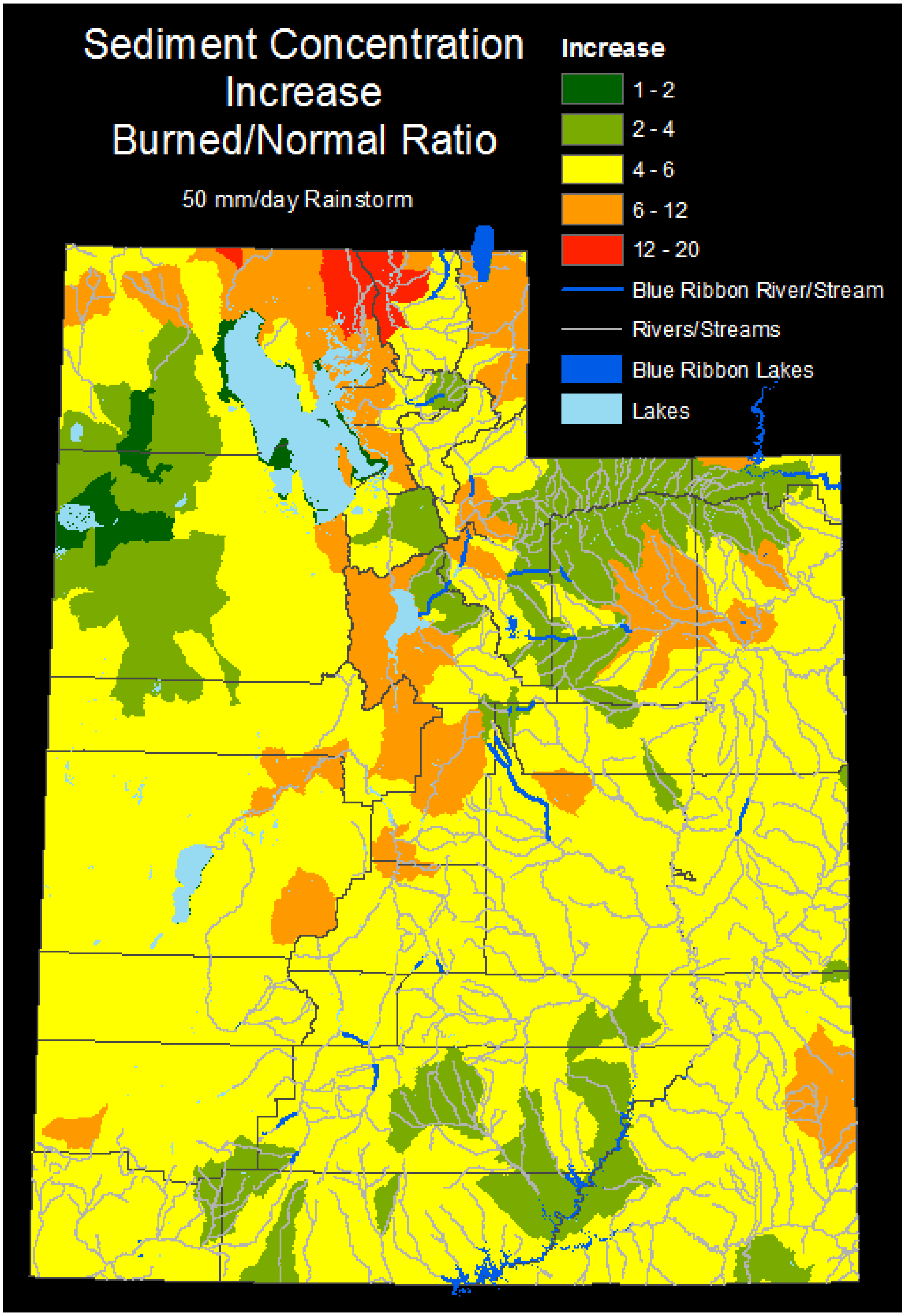


Figure 6.17: Blue ribbon fishery locations overlaid on the Figure 6.8 map of predicted increases in sediment concentration following a 50 mm storm, averaged over HUC 10 watersheds. Blue Ribbon fisheries data provided by Utah Division of Wildlife Resources.

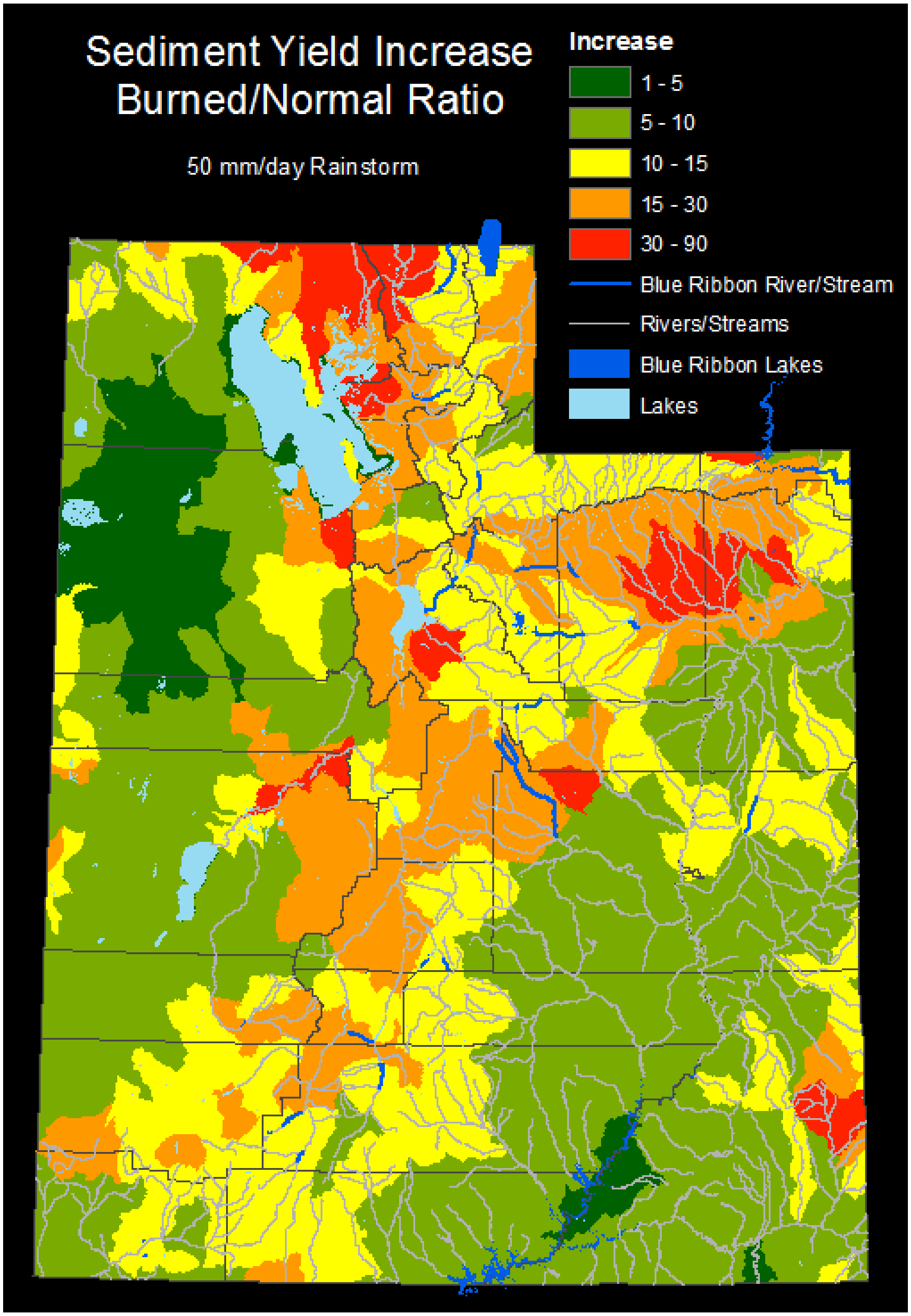


Figure 6.18: Blue ribbon fishery locations overlaid on the Figure 6.8 map of predicted increases in sediment yield following a 50 mm storm, averaged over HUC 10 watersheds. Blue Ribbon fisheries data provided by Utah Division of Wildlife Resources (UDWR 2016.)

Table 6.4: Predicted ranges of increased runoff, sediment concentration and sediment yield in Utah's Blue Ribbon Rivers after a severe burn and a 50 mm storm. Water bodies falling within the rural or urban cluster are highlighted. Blue Ribbon fisheries data provided by Utah Division of Wildlife Resources (UDWR 2016.)

Blue Ribbon River	Runoff	Sediment Yield	Sediment Concentration	
Asay Creek	2-5	10-15	4-6	
Duchesne River (Hanna to North Fork)	5-10	15-30	2-4	
East Fork Sevier (Black Canyon)	2-5	10-15	4-6	
East Fork Sevier (Kingston Canyon)	2-5	15-30	4-6	
Fremont River	2-5	10-15	4-6	
Green River	2-10	10-30	2-6	
Huntington Creek	2-5	15-30	4-6	
Left Fork Huntington Creek	**2-5**	**15-30**	**4-6**	Rural
Logan River	2-5	10-15	4-6	
Lower Fish Creek	2-5	10-15	2-4	
Lower Provo River	2-5	10-15	2-4	
Middle Provo River	2-5	15-30	6-12	
Panguitch Creek	2-5	10-15	4-6	
Right Fork Huntington Creek	2-5	15-30	4-6	
South Fork Ogden River	**2-5**	**10-15**	**2-4**	Urban
South Fork Ogden River	**2-5**	**10-15**	**2-4**	Urban
Strawberry River (Duchesne River to Starvation)	2-5	10-15	2-4	
Strawberry River (Soldier Creek Dam to Red Creek)	2-5	10-15	2-4	
Weber River (Echo to Wanship)	2-5	10-15	6-12	
West Fork Duchesne River	10-15	15-30	2-4	
West Willow Creek	2-5	10-15	4-6	

Table 6.5: Predicted ranges of increased runoff, sediment concentration and sediment yield in Utah's Blue Ribbon Lakes after a severe burn and a 50 mm storm. Water bodies falling within the rural or urban cluster are highlighted. Blue Ribbon fisheries data provided by Utah Division of Wildlife Resources (UDWR 2016.)

	B/N Ratio			
Blue Ribbon Lake	**Runoff**	**Sediment Yield**	**Sediment Concentration**	
Bear Lake	2-5	10-15	6-12	
Blind Lake	2-5	10-15	4-6	
Brough Reservoir	1-2	5-10	4-6	Rural
Calder Reservoir	2-5	10-15	4-6	
Duck Fork Reservoir	2-5	15-30	4-6	Rural
Fish Lake	2-5	10-15	4-6	
Flaming Gorge Reservoir	2-10	5-90	2-12	
Gunlock Reservoir	1-2	5-10	2-4	
Huntington Reservoir	2-5	15-30	4-6	Rural
Jordanelle Reservoir	2-5	10-30	4-12	
Kolob Reservoir	2-5	10-15	2-4	
Lake Canyon Lake	2-5	10-15	2-4	
Lake Powell	1-2	1-10	2-6	
Manning Meadows Reservoir	2-5	10-15	4-6	
McGath Lake	2-5	10-15	2-4	
Minersville Reservoir	2-5	15-30	4-6	
Panguitch Lake	2-5	10-15	4-6	
Paragonah Reservoir	2-5	10-15	4-6	
Pelican Lake	2-5	15-30	6-12	
Pineview Reservoir	2-5	15-30	2-4	Urban
Quail Creek Reservoir	1-2	10-15	4-6	
Sand Hollow Reservoir	1-2	10-15	4-6	
Scofield Reservoir	2-5	10-15	2-4	
Steinaker Reservoir	2-5	30-90	6-12	
Strawberry Reservoir	2-5	10-15	4-6	

Discussion

The MUSLE curve number models for urban and rural watersheds predicted similar responses in runoff, sediment concentration and sediment yield under a range of storm events. Average predicted sediment concentrations were considerably higher for forested lands compared to rangelands, although the normalized B/N ratio was comparable. Weber watershed includes steep mountainous landscape characteristic of portions of the urban cluster. Sevier watershed represents west desert valley and small range landscapes – both typical landscapes in the rural cluster and across much of the state. Although soils and other factors differ between these two watersheds, relative differences were likely due to differences in slope between the two modeled watersheds.

The SWAT model also resulted in increased discharge and yield under burned conditions, although the predicted B/N ratios for a 50 mm and a 25 mm precipitation event were about half those predicted by the MUSLE method. Unlike the MUSLE model, the SWAT model includes routing and deposition of sediment, a more realistic condition that resulted in lower predicted responses.

The statewide map of predicted runoff and sediment yield provides an indication of areas that are particularly sensitive to sediment runoff following severe fire events. Most of the state falls within relatively low B/N ratios. Small areas of the state appear to be at risk of high sediment runoff following severe fires. None of the state's drinking water intakes or reservoirs appear to be directly downstream of these most sensitive areas.

Our predicted B/N results for runoff, sediment concentration and sediment yield fall in the low to mid-range of values reported in the literature (Ice et al, 2004). As noted in the introduction, our models did not incorporate any of the organic material released during a severe fire, so relatively modest increases in sediment concentrations and yield relative to published field studies was expected. Our study does reveal relative responses across the state, in areas of differing watershed characteristics and vegetation and for storm events of varying intensities. This should help land, fire and water quality managers identify those areas within the state that are most at risk from a severe fire.

By overlaying maps of our predicted changes in runoff and sediment following a severe fire with locations of drinking water intakes and reservoirs and locations of blue ribbon fisheries, we were able to identify areas specifically at risk if a severe fire were to occur. We linked drinking water amenities with actual fire risk as well.

This study focused on the relative increases in runoff and sediment releases following a wildfire. Our models did not allow us to route predicted sediment or water across landscapes or within rivers. Rather, we provided an average value for aggregated pixels within HUC 10 (moderate sized) sub-watersheds of the state. Even with these limitations, we are able to determine which drinking water resources and blue ribbon fisheries may be most impacted by a severe fire. Drinking water intakes are sensitive to increased sediment concentrations, which may cause abrasion of pumps and pipes. Increased sediment yield fills in drinking water reservoirs, resulting in reduced storage. About 1/3 of all the drinking water withdrawal points and drinking water reservoirs fall within areas predicted to have substantially increased sediment (and associated pollutant) concentrations following a severe fire, and almost 50% of these sites are in areas predicted to have increased sediment yield. Drinking water infrastructure in the state's southeast appear to be particularly at risk.

Blue ribbon fisheries are directly affected by increased sediment, which reduces visibility in the rivers (affecting visual predators), but also smothers important food resources and breeding habitat. All Blue Trout fisheries were predicted to have at least modest increases in flow (flooding), sediment concentration and sediment yield (mass of sediment) following a severe burn. As noted above, sediments typically deliver other pollutants of concern, such as metals and phosphorus. The Middle Provo River and portions of the Weber River were predicted to have higher risk of increased sediment and other pollutant concentrations. The Duchesne River and Huntington Creek were at higher risk of flooding from high flows as well as increased risk of greater sediment volumes. Steineker Reservoir was predicted to have the highest risk of increased sediment loads.

We did not model other water quality contaminants that increase in runoff following. Many of these impacts are correlated with increased runoff and sediment, however, so we feel that these are reasonable indicators of potential risks associated with increased nutrients, toxic metals, and salt loads resulting from wildfires. Many pollutants, such as metals and phosphorus, adsorb to soil particles or are bound closely within minerals. These will likely increase when sediment concentrations and yields increase (Smith et al, 2011.) Soluble pollutants such as nitrogen compounds will likely increase with increased runoff, although dilution effects make predicting the actual response more difficult without directly modeling these.

Summary

Sediment yields and concentrations increase with rain events after a wildfire. The magnitude of this response depends on land cover, soil, and topographic conditions. It also varies with the severity of the wildfire and intensity of rain events. Small rain events may not have much erosive power in comparison to the larger rain events. The results from the study provide indications of the relative responses of runoff and sediment following a severe fire in regions of Utah with differing vegetation and physical watershed characteristics.

References

Homer, C. G., Dewitz, J. A., Yang, L., Jin, S., Danielson, P., Xian, G., ... & Megown, K. (2015). Completion of the 2011 National Land Cover Database for the conterminous United States-Representing a decade of land cover change information. *Photogramm. Eng. Remote Sens*, 81(5), 345-354.

Ice, George G. , Daniel G. Neary, and Paul W. 2004. Adams Effects of Wildfire on Soils and Watershed Processes. *Journal of Forestry* -Washington- · September 2004 issue. 20 pp.

Meixner, T., & Wohlgemuth, P. (2004). Wildfire impacts on water quality. *Journal of Wildland Fire*, 13(1), 27-35.

Murphy, S. F. (2012). Wildfire effects on source-water quality—lessons from Fourmile Canyon fire, Colorado, and implications for drinking-water treatment (No. 2012-3095). US Geological Survey.

Neary, D. G., Gottfried, G. J., DeBano, L. F., & Tecle, A. (2003). Impacts of fire on watershed resources. *Journal of the Arizona-Nevada Academy of Science*, 23-41.

Neitsch, Susan L., et al. Soil and water assessment tool theoretical documentation version 2009. Texas Water Resources Institute, 2011.

Penrod, Emma. 2015. "It's not just Utah Lake: Toxic algae plagues 20 waterways, including drinking water sources" | The Salt Lake Tribune. First Published Aug 07 2015 07:21PM • Last Updated Aug 08 2015 10:32 am http://www.sltrib.com/news/2815139-155/its-not-just-utah-lake-toxic

PSOMAS. 2002. Deer Creek Reservoir Drainage TMDL (2002) Prepared For: Utah Department of Environmental Quality - Division of Water Quality Dave Wham Project Manager Harry Lewis Judd Project Supervisor.
http://www.deq.utah.gov/ProgramsServices/programs/water/watersheds/approvedtmdls.htm. Accessed 12/18/2016.

Purdue University. 2013. SCS Curve Number Method.
https://engineering.purdue.edu/mapserve/LTHIA7/documentation/scs.htm. Accessed 11/24/2016.

Seaber, P.R., F.P. Kapinos and G.L. Knapp. 1987. Hydrologic Unit Maps. U.S. Geological Survey Water-Supply Paper 2294. 63 pp.

Smith, H. G., Sheridan, G. J., Lane, P. N., Nyman, P., & Haydon, S. (2011). Wildfire effects on water quality in forest catchments: a review with implications for water supply. *Journal of Hydrology*, 396(1), 170-192.

Tecle, A., Neary, D. G., Ffolliott, P., & Baker Jr, M. B. (2003). Water Quality in Forested Watersheds of the Southwestern United States. Journal of the Arizona-Nevada Academy of Science, 48-57.

Tecle, Aregae and Daniel Neary. 2015. Water Quality Impacts of Forest Fires. *Journal of Pollution Effects and Controls* 3:140. Doi: 10.4172/2375-4397. 100140.

Texas A&M. 2016. ArcSWAT. Computer software. SWAT Soil and Water Assessment Tool. Vers. 2012. Texas A&M University, 2016. http://swat.tamu.edu/software/arcswat/. Accessed Sept. 2016..

Thompson, M. P., Scott, J., Langowski, P. G., Gilbertson-Day, J. W., Haas, J. R., & Bowne, E. M. (2013). Assessing watershed-wildfire risks on national forest system lands in the rocky mountain region of the United States. *Water*, 5(3), 945-971.

USDA Forest Service. 2013. West Wide Wildfire Risk Assessment Final Report, Addendum I. Prepared by The Sanborn Map Company for: Oregon Department of Forestry Western Forestry Leadership Coalition Council of Western State Foresters Funded by: USDA Forest Service. 105 pp.

USDA Forest Service. 2016. Forest to Faucets website. http://www.fs.fed.us/ecosystemservices/FS_Efforts/forests2faucets.shtml. Accessed 12/15/2016.

U.S. EPA (2016) Watershed Quality Assessment Report. https://iaspub.epa.gov/waters10/attains_watershed.control?p_huc=16020204&p_state=UT&p_cycle=2010&p_report_type=#assessment_data. Accessed January 2017.

USGS 2016. National Water Information System: Web Interface. https://waterdata.**usgs**.gov/usa/nwis/uv?site_no=1012850. Accessed 1/1/2017.

UDWQ (Utah Division of Water Quality. 2016. Algal Blooms 2016. Downloaded 12/18/2016. http://deq.utah.gov/Divisions/dwq/health-advisory/harmful-algal-blooms/bloom-2016/index.htm

Utah Division of Wildlife. 2016. Blue Ribbon Fisheries Information. https://wildlife.utah.gov/hotspots/blueribbon.php. Accessed 12/10/2017.

Weidner, Emily and Al Todd. 2011. Drinking water and forests in the U.S. Methods Paper. 2011. Ecosystem Services & Markets Program Area. *State and Private Forestry*. October 2011 issue.

CHAPTER 7: WILDFIRE AND THE UTAH CATTLE INDUSTRY

Man-Keun Kim
Dept. of Applied Economics and the Center for Society, Economy, and the Environment
Utah State University

Introduction

Agricultural production makes up only about 1% of Utah's gross state product; if one includes the agricultural processing sector then agriculture makes up about 4% of the total Utah economy as measured by value-added (Ward, Jakus and Coulibaly, 2012). Four counties located along the Wasatch Front (Davis, Salt Lake, Utah, and Weber), plus three other counties with relatively large economies (Cache, Summit, and Weber), each enjoy a highly diversified economic structure and, collectively, comprise the bulk of Utah's overall economy. The economic structures of the remaining 22 counties are based primarily upon the natural resources found in those counties; key economic sectors include (i) production agriculture, (ii) oil, gas, coal, and other mineral production, and (iii) tourism and recreation. Thus, while production agriculture is a relatively small component of the Utah's overall economy, production agriculture is an important contributor to the economy of rural Utah. For example, production agriculture comprises 20% of the Beaver county economy and 12% of the Wayne county economy, again using value-added as the metric (Jakus et al. 2013).

In 2014, cash receipts from Utah's agricultural production were nearly $2.4 billion (USDA/NASS 2015). Some 78% of cash receipts come from livestock and livestock-related products. The largest source of livestock cash receipts was generated by cattle & calves, which account for more than $800 million in output. Indeed, cattle ranching is the largest sector in Utah's agricultural economy. Utah's ranching industry, though, is highly dependent upon access to the public range; public land administered by the Bureau of Land Management (BLM) and the U.S. Forest Service (USFS), along with lands administered by the School and Institutional Trust Lands Administration (SITLA), are an important source of livestock forage. According to Godfrey (2008), ranchers in Grand and San Juan counties obtained 47% of forage for their cattle from state and federal rangelands, while only 8% of forage was produced from private range and another 29% from pasture.[11] Similarly, ranchers in Garfield, Kane and Wayne counties obtained more than 65% of forage from state and federal lands, with less than 10% of forage sourced from private range and additional 20% from pasture (Godfrey, 2008, Table 10, p. 33).

After experiencing wildfire on a public grazing allotment, a rancher's access is typically restricted for a minimum two years (BLM, 1999, Bruce et al. 2007). Limited access to public grazing land—especially in regions of the state where ranchers are highly dependent on public range—can have

[11] The remaining animal forage requirement was purchased in the market or produced on the farm.

negative economic impacts on the economies of rural counties. This chapter quantifies the economic impact of wildfire on the livestock sector. We begin by empirically estimating a cattle inventory equation for five counties in southern Utah, a region of the state that is heavily reliant upon public rangeland and has few alternatives if access to the public range is restricted. Annual cattle stocks (from 1992 through 2015) are specified as a function of feeder price (most ranchers sell feeder cattle), the price of hay, and concurrent and lagged wildfire activity. Wildfire activity is measured as acres burned in any given year. The econometric model links wildfire to cattle stocks and is estimated to quantify the impact of wildfire on cattle inventory. Predicted changes in cattle inventory due to wildfire are then used to estimate the regional economic impact of wildfire on public range.

The Study Region

The area is dominated by public land ownership; Table 7.1 shows the percentage of land ownership and administration in each county. The study region totals nearly 15 million acres accounting for 27% of the state's area. Wilderness areas are included in the percentages for BLM and USFS, as are state lands not generally available for grazing (such as state parks). Including this acreage does not change the overall assessment of public land in the region. Net of wilderness, BLM administers 51.0% of the region, USFS administers 11.8%, and SITLA administers 6.9%.

Table 7.1: Land Ownership and Administration

	BLM	USFS	NPS	State	Private	Tribal
Garfield	45%	31%	14%	5%	5%	0%
Grand	66%	2%	3%	16%	4%	8%
Kane	63%	5%	18%	4%	10%	0%
San Juan	41%	9%	12%	5%	8%	25%
Wayne	57%	10%	19%	11%	4%	0%
Five County Region	51%	12%	13%	7%	7%	10%

Note: The Department of Defense administers 2,507 acres in Grand County *Source: Banner et al. (2009)*

The study region contains only a small number of large ranching operations. Across the five counties, the average number of farms raising 500 or more cattle is 11 (2012 Agricultural Census) while the number of farms with 500 and more cattle in the study region is only 6. The average farm in the study region owns 90 cattle while state average is 105 cattle (2012 Agricultural Census). It is clear from Table 7.1 that private range and pasture opportunities are scarce, and that access to such forage resources would be difficult if wildfire were to burn large portions of the public range. Another measure of the limited ability of ranchers to effectively respond to wildfire-related loss of rangeland is hay production per head. Across Utah's 29 counties, farmers produce about 2.6 tons of hay per head. Millard county produces the greatest ratio, at just under 3.6 tons/head. In contrast, counties in the five-county study region produce about 1.85 tons of hay per head of cattle. In addition, ranch size in five counties are relatively small (90 cattle per ranch) and the hay produced in these counties is not adequate to feed current herd size. Thus, producing hay in case of restricted access to public grazing land due to wildfire is not a viable option to maintain cattle inventory.

Methodology

This study utilizes the existing time series data on wildfire activities combined with cattle inventory data from National Agricultural Statistics Service (NASS), United State Department of Agriculture (USDA). Cattle inventory in county i in year t is set as the function of major input and output prices, and wildfire activities such that:

$$(1) \qquad cattle_{i,t} = \beta_0 + \beta_1 wf_{i,t} + \beta_2 wf_{i,t-1} + \beta_3 p_t^{feed} + \beta_4 p_t^{hay} + \beta_5 BSE + \varepsilon_t,$$

where $cattle_{i,t}$ is the number of cattle ("cattle, including calves" on December 31st) in county i in year t, $wf_{i,t}$ is the area burned in acres from wildfire in county i in year t. Concurrent wildfire activity (wildfire during the year that cattle inventory is measured) is included in equation (1) because most of wildfire occur during a summer season and cattle inventory is measured on December 31st in each year. Thus, ranchers would have had the opportunity to adjust cattle inventory in response to recent wildfires. The lagged wildfire is included to capture any longer term adjustment processes. Two price variables are also included in equation (1). p_t^{feed} is the (expected) price of feeder cattle, measured by feeder futures price nearby[12], whereas p_t^{hay} is the Utah price for alfalfa hay. The variable BSE is an indicator variable capturing the influence of the bovine spongiform encephalopathy news reports appearing from 2004 through 2007. The error term, ε_t, is assumed to be properly distributed.

The coefficients for concurrent and lagged wildfire activities, β_1 and β_2, quantify the impact of wildfire activities on cattle inventory. We do not know how much of historical burned acreage actually occurred on public rangeland, and a key assumption of our modeling strategy is that when wildfire occurs, it occurs on public range. Fire suppression activities tend to focus on protecting private property; given the relatively large proportion of public rangeland in the study area, it is not unreasonable to assume that much of the burned acreage occurs on the public range.

Data Sources

Cattle Inventory Data

Annual county-level cattle inventory data across counties are collected from USDA/NASS (2016). Figure 7.1 shows the year-end cattle inventory for each county from 1992 through 2015. Using 2015 as a reference year, the number of cattle (including calves) was 18,900 head in Garfield county, 3,600 head in Grand county, 8,800 head in Kane county, 15,300 head in San Juan county, and 18,500 head in Wayne county. The total number of cattle in these counties in 2015 was 65,100, which is about 8% of the total Utah cattle inventory.

---

[12] *Futures price nearby* is the futures price with the closest settlement date when several futures contracts exist in the market.

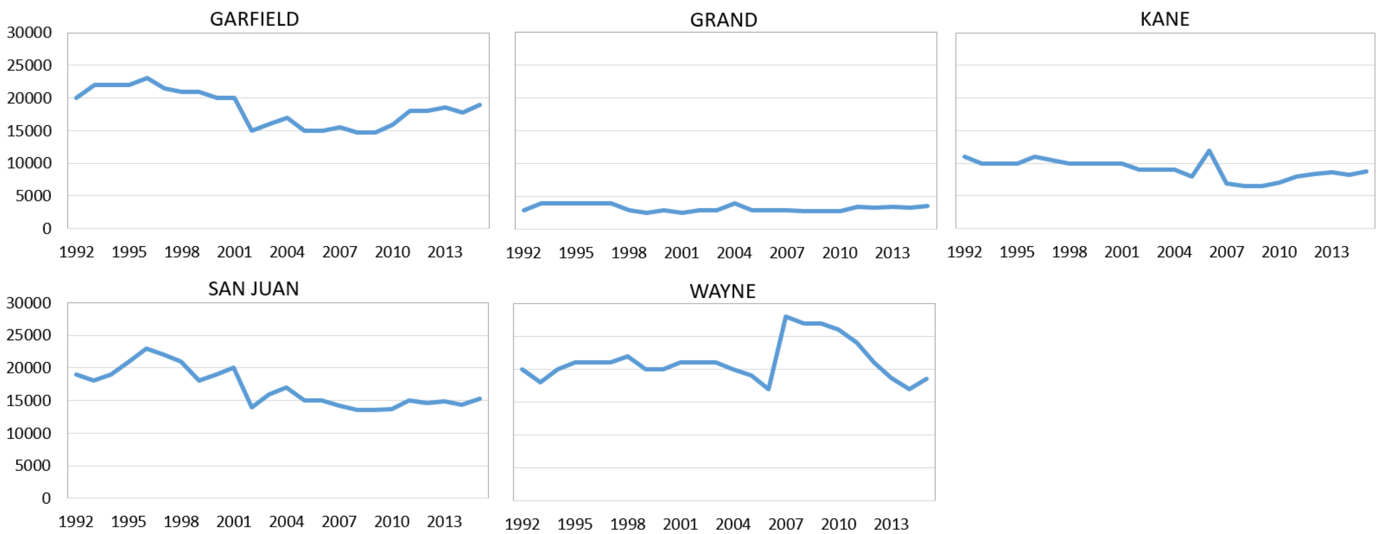


Figure 7.1. Number of Cattle and Calves in Study Region (head)

Source: USDA/NASS Quickfacts

Wildfire Data

The wildfire data set used for this portion of the study was described in Chapter 3. Short's cleaned wildfire occurrence data was combined with 2014 and 2015 wildfire data downloaded from National Federal Fire Occurrence database. This dataset contains 4,620 fires of 5 acres or greater in the state if Utah over the time range (1992-2015). Of these wildfires, 612 wildfires were identified as originating in the study region. Wildfires were assigned to the year during which they ignited. Wildfire activity for a given county/year pair is measured as total burned acreage for wildfires originating in that county during the specific year. The database did not keep track of wildfire perimeters so we could not track wildfires originating outside the study region but burning into it, nor could we adjust acreage of wildfires that spread outside the study region. Figure 7.2 shows a plot of the fire data, size of wildfire, and the location of the wildfire.

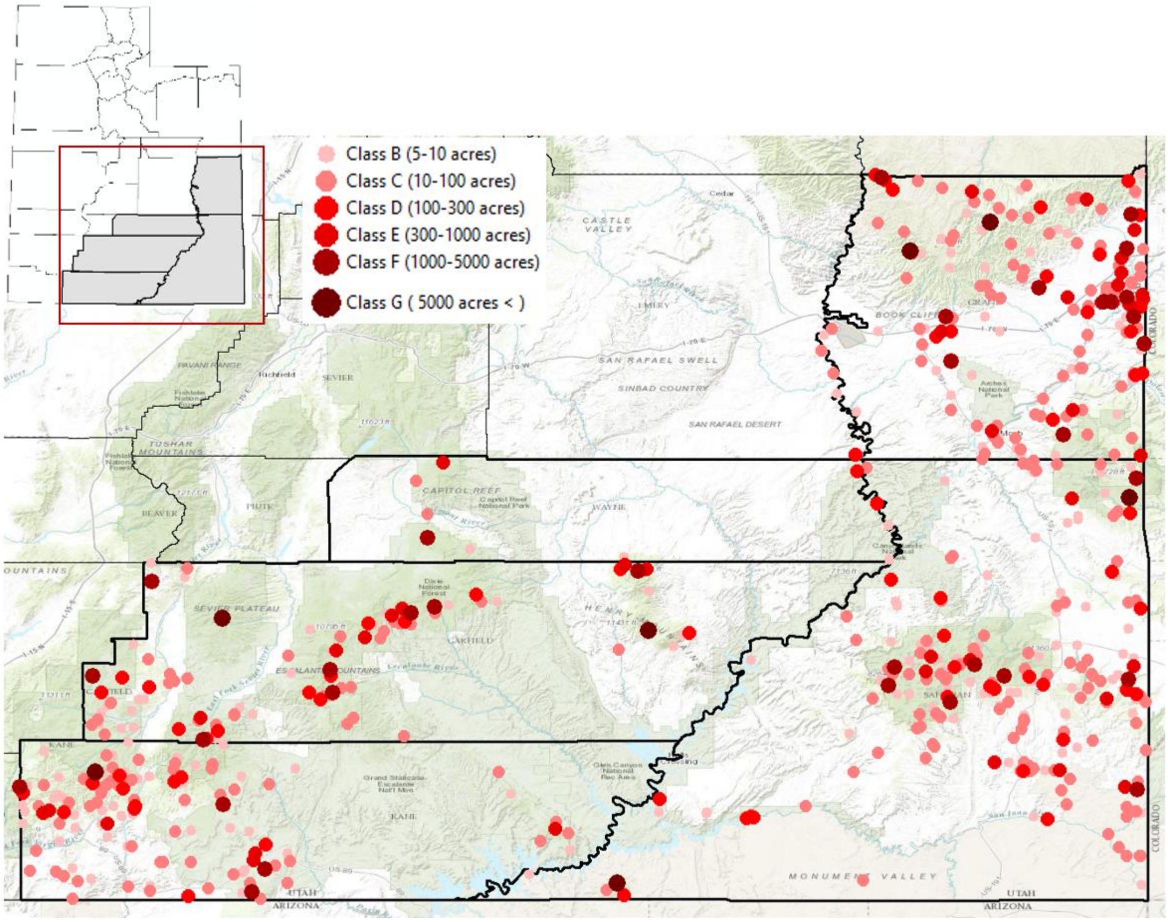


Figure 7.2. Plot of Wildfire Activities in Study Region (1992 – 2015)

Note: Red dots are origins of wildfires. Dark colors indicate the larger size of wildfires.

During 1992 - 2015 period, the average size of wildfires in the region was 509 acres. The largest wildfire in the dataset was the June 2002 Diamond Creek fire, which burned 88,421 acres near Arches National Park in Grand County. The distribution of wildfire size is heavily skewed with a very long right tail. Most wildfires (75%) burned less than 100 acres. Three wildfires burned more than 10,000 acres (Table 7.2 and Figure 7.3).

Table 7.2. Distribution of Wildfire Size in Study Region (612 wildfires)

	Classification	Number of fires	% of total	Cumulative
Less than 10 acres	Class B	166	27.1%	27.1%
Less than 100 acres	Class C	291	47.5%	74.7%
Less than 300 acres	Class D	71	11.6%	86.3%
Less than 1000 acres	Class E	44	7.2%	93.5%
Less than 5000 acres	Class F	33	5.4%	98.9%
More than 5000 acres	Class G	7	1.1%	100%

Source: Short et al. (2015) and Federal Wildfire Occurrence Database

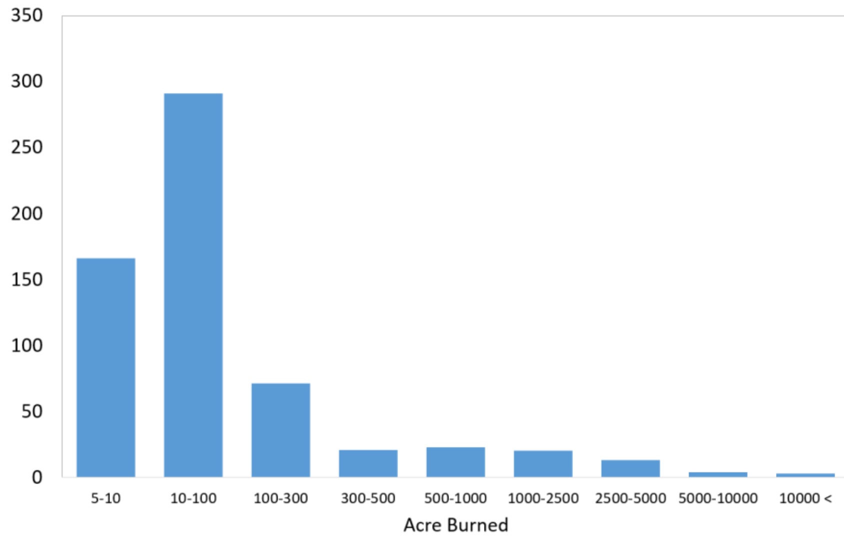


Figure 7.3. Wildfire Size Distribution in Study Region (612 wildfires)

Source: Short et al. (2015) and Federal Wildfire Occurrence Database

Price Data

Price for feeder cattle is compiled from Livestock Marketing Information Center (LMIC). LMIC reports monthly futures price data which is averaged for annual data. The price of hay is obtained from USDA/NASS (2016). Descriptive statistics for the key variables appear in Table 7.3.

Table 7.3: Descriptive Statistics

Variable	Mean	Median	Min	Max	St.Dev
Price, feeder cattle ($/cwt)	114.01	106.77	78.84	186.10	25.06
Price, hay ($/ton)	117.27	103.80	83.26	179.40	30.67
BSE	0.17	0	0	1	0.38
Garfield county					
Cattle inventory (head)	18,433	18,250	14,700	23,000	2,769
Wildfire, t (acres)	4,224	358	0	42,820	11,050
Wildfire, t-1 (acres)	4,230	358	0	42,820	11,048
Grand county					
Cattle inventory	3,275	3,000	2,500	4,000	508
Wildfire, t	5,894	624	0	99,384	20,140
Wildfire, t-1	5,894	624	0	99,384	20,140
Kane county					
Cattle inventory	9,100	9,000	6,500	12,000	1,476
Wildfire, t	887	324	0	8,061	1,662
Wildfire, t-1	892	324	0	8,061	1,660
San Juan county					
Cattle inventory	16,925	15,650	13,600	23,000	2,649
Wildfire, t	1,863	614	7	13,340	3,186
Wildfire, t-1	1,863	614	10	13,340	3,186
Wayne county					
Cattle inventory	21,217	21,000	17,000	28,000	3,067
Wildfire, t	121	0	0	2,179	457

Wildfire, t-1	160	0	0	2,179	485

Estimation and Loss in Cattle Inventory

Estimated models following equation (1) are shown in Table 7.4. Data are available for five counties over a 24 year period (1992–2015) yielding a panel of 120 observations. Panel data allow us to control for cross-county heterogeneity by allowing for a county-invariant parameter called a *fixed effect* (Baltagi, 2005). The fixed effect (FE) explores the relationship between wildfire activities and cattle inventory within a given county and allows for each county's individual characteristics (such as precipitation, vegetation, or terrain features) that influence its cattle inventory. Such differences across counties can be captured in the FE constant term (Greene, 2000). Alternatively, a random idiosyncratic component may affect cattle inventory in a county. This assumption gives rise to the random effects (RE) model. The RE model, unlike the FE model , the differences across counties are assumed to be randomly distributed across counties; hence, a county's cattle inventory over time is linked through the error term and not through a fixed individual specific constant term (Greene, 2000). Following standard panel data methods, we estimated both FE and RE models and report the results in Table 7.4.

Table 7.4. Cattle Inventory Models

	(1) Fixed Effect	(2) Random Effect
$acre_t$	-0.0359*	-0.0361*
	(0.08)	(0.07)
$acre_{t-1}$	-0.0243	-0.0246
	(0.227)	(0.218)
p_feeder	-16.492	-16.517
	(0.160)	(0.155)
p_hay	-14.895+	-14.893+
	(0.122)	(0.118)
Bse	-1356.6**	-1356.8**
	(0.02)	(0.02)
_cons	17799.7***	17803.6***
	(0.00)	(0.00)
N	120	120
R^2		
Within	0.1664	0.1664
Between	0.3117	0.3117
Overall	0.0421	0.0422
F or chi2 statistics	4.39	22.14
Prob > F or chi2	0.001	0.0005

Numbers in parentheses are P-values.
Significance levels are 1% (***), 5% (**), 10% (*), and 15% (+).

All of the model variables take the expected signs. The coefficients of the cattle feeder price and the hay price are negative, which is consistent with economic intuition. A higher feeder price encourages ranchers to sell their cattle early even though coefficients in both models are not statistically significant (P-values are 0.160 and 0.155, respectively). A higher hay price raises the cost of producing cattle (reducing profit), and also discourages ranchers from holding cattle. The hay price coefficients in both models are not statistically significant at confidence levels commonly used for hypothesis testing (P-values are 0.122 and 0.118, respectively).

As expected, all other things equal, concurrent wildfire activities have negative impact on cattle inventory and are statistically significant at 10% level: when wildfires burn more acreage in a given year then cattle inventories fall. The coefficient for the lagged wildfire acreage is negative but not statistically significant (P-values are 0.227 and 0.218, respectively). It appears that ranchers respond rapidly to wildfire by quickly adjusting their herd sizes concurrently. Using the estimates in Table 7.4, we can calculate the marginal impact of wildfire activities. If concurrent wildfire activity is increased by 110 acres, i.e. wildfire burned 110 acres more in a summer season, then a county's cattle inventory will fall about 4 head (= 0.036 x 110).

The overall goodness-of-fit measures are similar for both the FE and RE models, so other metrics must be used to distinguish which is the better model. We applied a Hausman test, which is formed using the null hypothesis that the RE model is preferred (Greene, 2000; Torres-Reyna, 2007). The χ^2 test statistic is 1.49 (Prob > chi2 = 0.47) which indicates that we fail to reject the null hypothesis that the RE model is preferred. Thus, our regional economic impact calculations are derived from the RE specification.

Regional Economic Impacts

Table 7.5 presents the current county estimates of farm income (cash receipts) from livestock (including poultry, hogs, and sheep) and crops. Production agriculture generates almost $74 million in farm receipts, making agriculture an important contributor to the region. The model presented in Table 7.4 statistically linked wildfire to cattle inventory; wildfire leads to reduced inventory associated with loss of rangeland access, which could have negative effects on the county economies in the study region.

Table 7.5. County Estimates: Farm Income (Cash Receipts, $1000 dollars)

County	Livestock & Products	Crops	Total
Garfield	6,231 (76%)	1,979 (24%)	8,210
Grand	2,052 (56%)	1,626 (44%)	3,678
Kane	11,135 (96%)	469 (4%)	11,604
San Juan	7,479 (45%)	9,155 (55%)	16,634
Wayne	18,641 (91%)	1,813 (9%)	20,454
Sum	45,538 (62%)	28,042 (38%)	73,580

Source: UDAF (2015) Utah Agriculture Statistics and Utah Department of Agriculture and Food Annual Report

Regional economic impacts measure "the net changes in new economic activity associated with an industry, event, or policy in an existing regional economy." (Watson et al., 2007). The event occurring in this study is the reduction of cattle inventory due to limited access to the public grazing land due to wildfire. The input-output (IO) analysis using IMPLAN (Impact analysis for PLANning) database may be utilized to measure the regional economic impact from changes in cattle inventory. The direct effects of wildfire-related loss of range are the changes in cattle inventory (in dollar value). Indirect effects of wildfire are the changes in inter-industry (support industries for cattle ranchers) purchases as they respond to different inventory levels. The induced effect then decreases economic activities for additional businesses (in the region) that support ranching business. This type of approach is called "demand-driven IO model" because this approach assumes that the changes in cattle inventory is the exogenous change in (final) demand in the regional economy.

Other studies, such as those by Leung and Pooley (2002), Fernandez-Macho, Gallastegui and Gonzales (2008), and Seung and Waters (2009), have argued that a supply-driven IO model it is more appropriate than a demand-driven IO model in situations where the output level is altered directly from a supply-side shock such as wildfire. That is, in a supply-driven IO model a "supply reduction" may occur rather than having a shock resulting from a shift in the demand curve. This is important because the supply effects are easier to estimate than the immediate demand effects because the change in (final) demand is unknown.[13] Using supply-driven IO multipliers, we can calculate both the backward and forward linkage effects of wildfire and cattle inventory change. The backward linkage is "a sector's relationship to upstream sectors (suppliers) that provide goods and services to cattle ranching & farming sector" (Seung and Waters, 2009), e.g., the reduction in cattle herd size may reduce its demand for inputs purchased from other sectors such as labor, feed, manufactured items (e.g., agricultural machinery, fencing, water infrastructure), and support services like those supplied by veterinarians, banks, insurance agencies, etc. The forward linkages are "a sector's relationship with its downstream demanders who purchase goods and services from the cattle ranching & farming sector" (Seung and Waters, 2009). Changes in cattle inventory may also reduce the output of meat processing (manufacturing sector) and wholesale sectors that purchase inputs from the cattle ranching sector.[14]

Estimation of Direct Impact from Simulation

We simulate prices and wildfire, using their historical distributions, and use the random effects model in Table 7.4 to quantify the direct impact of wildfire on cattle inventory. A simulation approach is used because the distribution of wildfire size is highly skewed. The steps in this approach are:

1. Randomly generate feeder and hay prices from their past history to generate empirical distributions of feeder and hay price. The random draws incorporates correlation across prices.
2. Randomly generate concurrent and lagged wildfire size in each county, again based on their past history. The random draws account for spatial correlation across the counties.
3. Based on the random draws of feeder price, the hay price, and the acreage burned in wildfires, the cattle inventory is estimated using the random effects model of Table 7.4.

[13] Consumers do not buy cattle directly. They purchase beef.
[14] Derivation of supply-driven IO or Social Accounting Matrix (SAM) multipliers is beyond the scope of this research. Refer to Kim (2015) for the supply-driven IO model and to Kim et al. (2016) for the supply-driven SAM model.

4. Compare the cattle inventory predicted in step 3 with predicted cattle inventory assuming **zero** wildfires.
5. Repeat the steps 1 through 4 one thousand times.

Table 7.6 presents the changes (loss) in cattle herd size as calculated in the empirical distribution described above. Selected percentiles are also reported in Table 7.6 for comparison. Figure 7.4 presents the approximate distribution of the reduction in cattle inventory. As expected the distribution of loss in cattle inventory has a long tail to the right which is similar to size distribution of wildfire in Figure 7.3. When there is a disastrous wildfire year, corresponding to once out of every ten years (the 90th percentile in Table 7.6), over 2100 head of cattle could be liquidated. This is equivalent to about 3% of the typical cattle inventory in the study region.

Table 7.6. Fire-related Loss of Range Reductions in Cattle Inventory in the Study Region (head)

	Garfield	Grand	Kane	San Juan	Wayne	Total
Min	0	0	0	1	0	7
25th percentile	14	23	11	22	0	127
Median	42	47	32	59	0	278
75th percentile	161	195	61	140	1	698
Mean	256	309	52	112	7	737
90th percentile	1,042	683	125	321	28	2,129
95th percentile	1,461	2,388	215	422	54	3,756
Max	2,571	3,587	361	803	130	6,260

Based on 1000 simulation results with random feeder price, hay price and concurrent and lagged wildfire activities with estimates in Table 2.

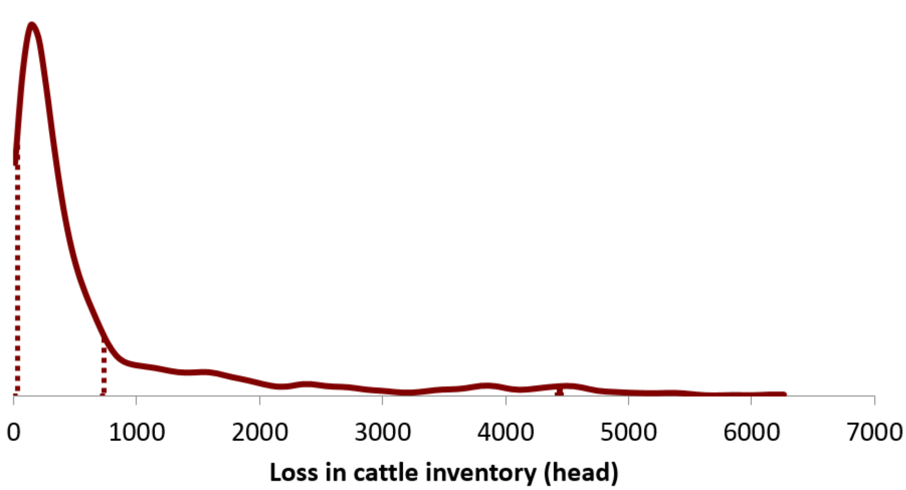


Figure 7.4. Frequency Distribution of Wildfire-related Range Loss Impact on Cattle Inventory (based on 1,000 random draws)

The value of cattle inventory loss (direct economic impact) is calculated as the change in the number of cattle (from Table 7.6) multiplied by the per head value of cattle. The average value of cattle over the 2012–2014 period is $1,243 per head (USDA/NASS 2015). Our economic impact modeling assumes that animals sold prematurely due to wildfire are sold at 60% of this value (a 40% loss).

Supply-driven Social Accounts Matrix Analysis

We construct a regional economy composed of all five counties, aggregating over 400 economic sectors into 10 aggregate sectors based on the IMPLAN database for the year 2013. While most of these sectors are highly aggregated, we maintain cattle ranching and farming as a separate, though smaller, economic sector. Before modeling the effect of wildfires, the gross regional product in the five counties was $1.43 billion and this level of economic activity supported an estimated 22,300 jobs. Major sectors include FIRES (finance, information, real estate, education, and other services) and government, which were estimated to support about 16,000 jobs. Cattle ranching & farming sector produces $35.78 million and supports 538 jobs in 2013.

The regional impacts based on results in Table 7.6 and the value of cattle are computed using supply-driven SAM multipliers are presented in Table 7.7. In the case of median cattle inventory loss (278 head), the total regional impact (direct + indirect + induced) is estimated to be $488,000 (Table 7.7). The total indirect impacts on industries are estimated to be $119,000. Value-added – employer compensation, proprietary income, other property income and indirect business taxes – were estimated to decrease by $90,000. As shown in Table 7.7, household sectors were partitioned into three categories according to income level (low, medium, and high) for brevity of presentation. Total household income is estimated to be reduced by $117,000 under (this) median case with about half of this impact affecting households in the medium income category. State and local government revenue is estimated to fall by $23,000 due to reduced taxes paid by industries and households.

Table 7.7: Regional Economic Impact of Loss in Cattle Inventory from Wildfire

	Median			Mean		
Sectors	(million dollars)					
Impact on industries (Indirect)	Backward[1]	Forward[2]	Total	Backward[1]	Forward[2]	Total
Other ag, forestry, fish and hunting	0.000	0.000	0.000	0.000	0.000	-0.001
Mining	-0.001	-0.007	-0.008	-0.003	-0.018	-0.021
Utilities	-0.008	-0.001	-0.009	-0.021	-0.003	-0.024
Construction	0.000	-0.010	-0.010	-0.001	-0.028	-0.029
Manufacturing	-0.002	0.000	-0.002	-0.006	0.000	-0.006
Wholesale trade	-0.005	0.000	-0.005	-0.013	-0.001	-0.014
Retail	-0.002	0.000	-0.002	-0.004	0.000	-0.004
Transportation & warehousing	-0.034	-0.003	-0.037	-0.091	-0.008	-0.099
FIRES[3]	-0.004	-0.001	-0.005	-0.011	-0.003	-0.014
Government	-0.039	-0.001	-0.040	-0.102	-0.003	-0.105
Total impacts on industries	-0.095	-0.024	**-0.119**	-0.252	-0.065	**-0.317**
Impact on value added (Indirect)	Backward	Forward		Backward	Forward	
Employment compensation	-0.009	0.000	-0.009	-0.024	0.000	-0.024
Proprietary income	-0.063	-0.002	-0.065	-0.167	-0.006	-0.173
Other property income	-0.013	0.000	-0.013	-0.033	0.000	-0.033
Indirect business taxes	-0.003	0.000	-0.003	-0.007	0.000	-0.007
Total impacts on value added	-0.088	-0.002	**-0.090**	-0.231	-0.007	**-0.238**
Impacts on household income (Induced)	Backward	Forward		Backward	Forward	
Low income household	-0.025	-0.001	-0.026	-0.065	-0.002	-0.067
Medium income household	-0.046	-0.001	-0.047	-0.123	-0.003	-0.126
High income household	-0.043	-0.001	-0.044	-0.114	-0.004	-0.118
Total impacts on household	-0.114	-0.003	**-0.117**	-0.302	-0.009	**-0.311**
State and local govt. revenue (Induced)	Backward	Forward		Backward	Forward	
	-0.022	-0.001	-0.023	-0.168	-0.009	-0.177
Indirect and induced regional impact (not including direct effects)	-0.319	-0.030	**-0.349**	-0.953	-0.090	**-1.043**
Total regional impact (including direct impact)[4]			**-0.488**			**-1.409**

1 The backward linkage is a sector's relationship with upstream sectors (suppliers) that provide goods and services used as intermediate inputs, which measures the change in output in sectors due to change in the output of cattle ranching sector.
2 The forward linkages is a sector's relationship with its downstream demanders who purchase goods and services from the cattle ranching sector
3 Direct impact is number of cattle reduction in head x $1243/head, value of cattle is the average of 2012, 2013 and 2014 and compiled from Utah Agricultural Statistics and Utah Department of Agriculture and Food 2015 Annual Report.
4 Direct impact is number of cattle reduction in head x $1243/head x (1 – 0.6), $1243/head is the value of cattle which is the average of 2012, 2013 and 2014 from Utah Agricultural Statistics and Utah Department of Agriculture and Food 2015 Annual Report. We assume that wildfire-induced herd-size adjustment occurs through early sales of cow and price discount due to off-season marketing is assumed to be 60% based on Kobayashi, Rolling and Taylor (2014).

Results for the case of cattle inventory reduction in response to a mean fire year (737 head) can be interpreted with the similar fashion; here, the total regional economic loss is estimated to be $1.409 million. Given the total number of households in the study region in 2015 was 13,554, this loss can be interpreted as about $36 per household for the median drop in cattle inventory and about $104 per household for the mean loss in inventory.

Income Loss Due to Increased Feed Purchase

When access to range is restricted due to wildfire, reductions in herd size are only one of the adjustments required of ranchers. Herd reductions mean that less forage is required, but there are still cattle remaining in inventory that can no longer be fed on the range. Instead, ranchers must purchase hay and other feed to sustain these animals until marketing. Given the limited amount of private land on which to rent pasture or range, the best available option is to purchase supplementary feed on the market. Purchased feed is more expensive than forage obtained from public rangelands, squeezing the profitability of a ranching operation whose public range has burned.

USU Extension data on stocking rates were used to estimate that 11.2 acres were required to support each cow-calf unit for the summer public range grazing season. For each simulated wildfire year, the predicted number of acres burned was used to determine the number of animals to be removed from the range, and then USDA supplementary feed cost estimates were applied. The USDA Economic Research Services estimated supplementary feed costs to $364 per cow per year in the Basin and Range region. This is a direct additional cost to ranchers to support animals displaced by wildfire.

For a median fire year, some 604 cow-calf units would be displaced from the range and must be maintained at a cost of just under $220,000. During a mean fire year, some 1,588 head would be displaced at an additional feed cost of just over $578,000. Regional economic impacts are presented in Table 7.8 for each type of fire year. In addition to direct income losses to ranchers, indirect labor income losses to those employed in other industries would total $27,321 in median fire year and $71,813 in a mean fire year.

Summary

Ranchers' access to public grazing land is restricted following a wildfire on public rangeland, typically for a minimum two years. Limited access to public grazing land has negative economic impacts on the regional economies, especially rural areas in Utah, where cattle production has been an important part of communities. We estimate a cattle inventory equation to quantify the impact of wildfire, which is the function of feeder price, hay price, and concurrent and lagged wildfire activity as measured in acres burned. Five counties, Grand, San Juan, Garfield, Kane, and Wayne, were selected due to the reliance of cattle producers in the region on public land sources of forage. Further, farm size in the study region is relatively small and hay production is not adequate to feed current herd size. Thus, purchasing hay in case of restricted access to public grazing land due to wildfire events is not a viable option to maintain cattle inventory.

We hypothesize that ranchers will reduce herd size when wildfire restricts access to the public range. Results show that concurrent wildfire activities have statistically significant effects on cattle inventory. Lagged wildfire activities also have negative impact but are not statistically significant. The fall in cattle inventory associated with a mean amount of acreage burned is estimated to be 771 head, or roughly 1.2% of cattle stock in 2015 in the region. The fall in cattle inventory associated with a median amount of burned acreage was estimated to be 278 head. Using supply-driven SAM multipliers, we calculate the backward linkages to upstream sectors that provide ranchers with goods and services, as well as the forward linkages to downstream demanders who purchase cattle from ranchers. We estimate the economic impacts of changes in cattle inventory to be $1.224 million for a median (small) fire year, or about $85 per household. For a mean (large) fire year economic impacts are predicted to be about $3.377 million, or about $233 per household.

Table 7.8: Regional Economic Impact of Direct Income Loss from Additional Supplementary Feed Purchase

Median

Sector	Industry output	VA	Labor income	Employment
		(2016 dollars)		(persons)
ag_forest	$218	$129	$35	0
ranching	$8	$3	$0	0
mining	$239	$156	$52	0
utilities	$1,881	$315	$174	0
construct	$1,180	$459	$366	0
manufact	$534	$199	$29	0
wholesale	$3,432	$1,841	$616	0
retail	$9,966	$5,793	$3,549	0
transport	$2,049	$775	$599	0
FIRES	$76,795	$42,338	$19,913	1
govt	$2,802	$2,574	$1,988	0
Total	$99,104	$54,582	$27,321	1

All the impacts are induced based on household income change
FIRES = Finance, insurance, real estate, education, and other services

Mean

Sector	Industry output	VA	Labor income	Employment
		(2016 dollars)		(persons)
ag_forest	$572	$339	$91	0
ranching	$21	$9	$1	0
mining	$629	$411	$136	0
utilities	$4,945	$827	$458	0
construct	$3,101	$1,207	$963	0
manufact	$1,404	$522	$77	0
wholesale	$9,021	$4,840	$1,619	0
retail	$26,194	$15,228	$9,328	0
transport	$5,386	$2,036	$1,574	0
FIRES	$201,856	$111,285	$52,341	2
govt	$7,366	$6,767	$5,225	0
Total	$260,495	$143,471	$71,813	3

All the impacts are induced based on household income change
FIRES = Finance, insurance, real estate, education, and other services

References

Baltagi, B.H. 2005. *Econometric Analysis of Panel Data*. Third edition, Wiley.

Bureau of Land Management (BLM).1999. *Emergency Fire Rehabilitation Handbook*. BLM Manual, Washington, DC.

Bruce, B., B. Perryman, K. Conley, and K. McAdoo. 2007. "Case Study: Grazing Management on Seeded and Unseeded Post-fire Public Rangelands," *The Professional Animal Scientist* 23: 285-290.

Fernandez-Macho, J., C. Gallastegui, and P. Gonzalez. 2008. "Economic Impacts of TAC Regulation: A Supply-driven SAM Approach." *Fisheries Research* 90: 225-234.

Godfrey, B.E. 2008. *Livestock Grazing in Utah: History and Status*, A Report for the Utah Governor's Public Lands Policy Coordination Office, Department of Applied Economics, Utah State University, available at http://apecextension.usu.edu/files/uploads/Environment%20and%20Natural%20Resources/Public%20Lands/Grazing%20Final%20Report.pdf

Greene, W.H. 2000. *Econometric Analysis*, Fourth edition, Prentice Hall.

Jakus, P.M., C. Silva, L. Coulibaly, and C.K. Chapman. 2013. Grazing on federal lands in Beaver, Duchesne and Wayne counties: an economic analysis of potential changes in grazing access. Working paper (October).

Leung, P., and S. Pooley. 2002. "Regional Economic Impacts of Reductions in Fisheries Production: A Supply-Driven Approach." *Marine Resource Economics* 16: 251-262.

Kim, M-K. 2015. "Supply Driven Input-Output Analysis: Case of 2010-2011 Foot-and-Mouth Disease in Korea." *Journal of Rural Development* 38(2): 173-188.

Kim, M-K., C.M. Ukkestead, H. Tejeda, and D. Bailey. 2016. "Benefits of an Animal Traceability System for a Foot-and-Mouth Disease Outbreak: A Supply-driven Social Accounting Matrix Approach," Poster presented at Traceability Symposium 2016, Calgary, Canada.

Pratt, M. and G.A. Rasmussen. 2001. Determining your stocking rate. Utah State Cooperative Extension Publication NR/RM/04. http://extension.usu.edu/files/publications/publication/NR_RM_04.pdf

Seung, C-K. and E.C. Waters. 2009. "Measuring the economic linkage of Alaska Fisheries: A Supply-driven Social Accounting Matrix (SDSAM) Approach." Fisheries Research 97: 17-23.

Short, K.C. 2015. Spatial wildfire occurrence data for the United States, 1992-2013 [FPA_FOD_20150323]. 3rd Edition. Fort Collins, CO: Forest Service Research data Archive. http://www.fs.usda.gov/rds/archive/Product/RDS-2013-0009.3/

Torres-Reyna, O. 2007. *Panel Data Analysis Fixed and Random Effects using Stata (v. 4.2)*, available at https://www.princeton.edu/~otorres/Panel101.pdf

USDA/ERS. 2016. Recent costs and returns: cow-calf, Basin and Range. https://www.ers.usda.gov/data-products/commodity-costs-and-returns/commodity-costs-and-returns/#Recent%20Costs%20and%20Returns:%20Cow-calf

USDA/NASS. 2015. 2015 Agricultural Statistics and Utah Department of Agriculture and Food Annual Report.

USDA/NASS. 2016. USDA/NASS QuickStats. https://quickstats.nass.usda.gov

Ward, R., P.M. Jakus and L. Coulibaly. 2013. The economic contribution of agriculture to the economy of Utah in 2011. Center for Society, Economy, and the Environment Research Report #3 (February).

Watson, P., J. Wilson, D. Thilmany, and S. Winter. 2007. "Determining Economic Contributions and Impacts: What is the Difference and Why Do We Care?" *Journal of Regional Analysis & Policy* 37(2): 1-15.

Chapter 8: Wildfire and National Park Visitation

Man-Keun Kim and Paul M. Jakus
Department of Applied Economics and the Center for Society, Economy and the Environment
Utah State University

Introduction

Utah's natural wonders have long attracted visitors from among Utah residents, residents of other states, and from other countries. Utah is home to five national parks (NPs), seven national monuments, two national recreation areas, one national historical site, and 43 state parks. Almost 8.4 million visitors were recorded at Utah's National Parks in 2015, with another 4.9 million visitors observed at the national monuments, recreation areas, and historical site (USNPS 2016). State parks accounted for another 4.2 million visitors in 2015 (Leaver 2016). Millions of acres of public lands in Utah (administered primarily by the Bureau of Land Management and the US Forest Service) are also open for dispersed recreation, but visitor counts for dispersed recreation are difficult to obtain. Finally, Utah has 14 ski resorts, 10 of which are located within one hour of the international airport in Salt Lake City. In the past five years, skier day counts at Utah's resorts have averaged over 4 million skier days per season, with a record 4.47 million skier days established during the 2015/16 season (SkiUtah.com, 2016).

Tourism, by any definition, is big business in Utah. In 2014, recreation activity in Utah resulted in expenditures of almost $8 billion and generated over $1 billion in state and local tax revenue. The tourism and travel industry is one of largest in the state, with expenditures by tourists employing almost 130,000 people and making up 9.3% of the state's workforce in 2014 (Leaver, 2016). Nonresidents accounted for the overwhelming majority of tourism expenditures (85%), making the tourism and travel industry Utah's largest export industry. The state of Utah has recognized the importance of this industry in recent years. The Utah Governor's Office of Economic Development has promoted Utah's recreation assets to national and international audiences through ad campaigns such as the "Mighty Five" (highlighting the five national parks) and its current campaign called "Road to the Mighty Five" (highlighting state parks and other places located near the national parks). Further, during the 16 day shutdown of the federal government in October 2013, the state of Utah provided funds to keep the five national parks open to the public (McCombs, 2014). The justification for this action was the importance of the national parks to the local and regional economies of southern Utah.

Just as the amount of winter snow affects the number of skier visits (and thus the economic impact of skiing in the state), it is possible that the visibility, safety, and health effects of seasonal wildfire may affect recreational visits to Utah's national parks and other public lands. Fires can lead to road and campground closures, create smoke that damages health and reduces visibility, and change the landscape in and around the national parks. In addition visitors may believe that visiting a national park with nearby wildfire activity may be dangerous. This chapter uses a statistical model to quantify the effect of wildfire on visitation to each of Utah's five national parks (NPs). We focus on national parks because of the availability of reliable and accurate long-term datasets on wildfire (from the USFS) and visitation (from the National Park Service). Our primary hypothesis is that wildfire negatively affects recreational visitation at Utah's NPs. Reduced visitation, in turn, means that tourism expenditures will fall, resulting in a cascade of employment and income effects throughout the regional economy.

Wildfire and Recreation

We are not the first to hypothesize such an effect of wildfire on recreation. Prior research on wildfire effects has focused on the response of visitors to onsite fire-related changes in the post-fire landscape (Table 8.1). Most of the previous literature has used survey methods to document the economic impacts from wildfire by measuring changes in the probability a site would be visited, the number of visits to a site, and the consumer surplus[15] (net welfare) derived from a recreational visit. Much of the literature employs stated preference methods (Vaux et al 1984; Englin et al. 2001; Loomis et al. 2001; Hesslen, et al. 2003), wherein photographs and hypothetical questions are used to estimate the welfare change associated with fire. This portion of the literature can be distinguished from the work of Love and Watson (1992), Englin et al. (1996), Hesseln et al. (2004), and Boxall and Englin (2008), all of whom used revealed preference methods (i.e., observations of actual—not hypothetical—recreation behavior) to examine visitation patterns immediately after a fire, and for years afterward, to calculate the effect of wildfire on the number and quality of visits to fire-damaged locales.

In contrast with the first seven studies listed in Table 8.1, Duffield et al. (2013) examine the contemporaneous effects of wildfire on visitation. The authors used aggregated visitation data and wildfire data to estimated changes in visitation to Yellowstone NP due to wildfire. The authors' specification was based on the travel cost model that links visits to a recreation site to the cost of getting to the site. Economic theory suggests that the number of visits will fall as the cost of travel to the site, which is a function of distance, increases. The vast travel cost literature has consistently found empirical support for this link. Economic theory also suggests that, all else equal, people will make more trips to higher quality recreation sites than to lower quality sites. Again, hundreds of studies have reported statistical support for this theoretical prediction. Aggregate visitor data such as monthly visitor counts prevent calculation of a travel cost variable as used in the studies cited above because such data do not report the distance traveled by each visitor. Instead, Duffield et al. use the price of gasoline as a proxy variable because it is highly correlated with travel cost. For their monthly visitation model, wildfire effects are captured by the total acreage of fires burning within 50 miles, 100 miles, and 200 miles of the park center during the month of visitation, as well as the preceding month. The authors' models found a statistically and negative effect of fire and lagged fire on monthly park visitation over the 1986-2011 study time frame.

[15] Consumer surplus is the value of visiting recreational sites which is defined as the difference between the visitors' willingness to pay and their actual expenditure.

Table 8.1. Selected Prior Research on Wildfire Effects on Recreation

Study	Brief summary
Stated Preference Studies	
Vaux, Gardner and Mills (1984)	Intense wildfires may have detrimental effects on recreation values
Loomis, Gonzales-Cabán, and Englin (2001)	Recreation values after a fire follow a nonlinear intertemporal path (Colorado).
Hesseln et al. (2003)	Hikers and bikers in New Mexico experience decreases in consumer surplus following either crown or prescribed fire.
Revealed Preference Studies	
Love and Watson (1992)	The 1988 Gates Park fire had relatively little impact on the choice to visit the North Fork or the South Fork, Montana.
Englin et al. (1996)	Nopiming Park, Manitoba; presence of historical fires along a canoe route were a disamenity to backcountry recreationists.
Hesseln, Loomis and Gonzales-Cabán (2004)	Compared economic effects of fire on hiking in Montana and Colorado suggests that the annual value of trips decreases after fire.
Boxall and Englin (2008)	Marginal per-trip welfare declines immediately after a fire, but recovers on a nonlinear path after ~35 years of regrowth (Nopiming Park, Manitoba).
Duffield et al. (2013)	Proximate wildfire has measurable and statistically significant concurrent effects on aggregate visitation at Yellowstone NP.

Methodology

Similar to Duffield et al., our study used recreation visitation data to Utah's national parks in conjunction with existing time series data on wildfire activity to estimate the statistical effect of wildfire on national park visits. Linear regression models of visitation to each of five national parks in Utah, i.e., Arches, Bryce Canyon, Canyonlands, Capitol Reef, and Zion, were estimated using the model shown in Equation 1:

$$\ln v_t = \beta_0 + \beta_1 wf_t + \beta_2 wf_{t-1} + \beta_3 p_t^{gas} + \sum_{m=1}^{11} \delta_m D_m + \varepsilon_t, \tag{2}$$

where $\ln v_t$ is the logged number of visitors in month t, wf_t is acre burned from wildfire in month t within 50 miles (80.5 km) radius to the park (visitor center or park entrance), p_t^{gas} is the real gas price adjusted by real personal income as a proxy of cost of traveling to the park, D_m are monthly indicator variables, and ε_t is the error term. Coefficients β and δ were estimated using the aggregate data. Coefficients for concurrent and lagged wildfire activities, β_1 and β_2, measure the relative change in the number of visitors for a given change in the wildfire activities (acre burned), i.e., $\beta \times 100$ % change in visitation (semi-log model). The results of the fire-visitation models were used to derive estimates of the direct expenditure change in the region.

Data

Data were collected from multiple sources, including the National Park Service, the National Wildfire Occurrence dataset, and standard sources of economic data such as the St. Louis Federal Reserve and sites maintained by the US Bureau of Census. Descriptive statistics may be found in Table 8.2.

Table 8.2: Recreation Model Data

	Mean	Median	Min	Max	Std. Dev.
Arches NP					
Visitation (# per month)	76,437	77,963	5,009	195,748	49,126
Wildfire, May (acres)	110	10	0	1,300	302
Wildfire, June	5,218	39	0	94,404	20,040
Wildfire, July	772	114	0	6,026	1,642
Wildfire, August	275	3	0	3,432	779
Wildfire, September	15	0	0	270	57
Bryce Canyon NP					
Visitation (# per month)	96,576	82,038	9,535	305,465	72,604
Wildfire, May (acres)	2,241	5	0	42,839	9,137
Wildfire, June	1,347	40	0	10,655	2,634
Wildfire, July	2,617	616	0	23,903	5,301
Wildfire, August	294	76	0	2,174	500
Wildfire, September	75	0	0	1,096	246
Canyonlands NP					
Visitation (# per month)	36,672	43,078	2,792	91,284	22,768
Wildfire, May (acres)	219	4	0	2,513	595
Wildfire, June	591	21	0	6,355	1,589
Wildfire, July	478	86	0	6,026	1,330
Wildfire, August	201	3	0	3,432	729
Wildfire, September	23	0	0	304	65
Capitol Reef NP					
Visitation (# per month)	52,967	58,850	4,604	135,543	35,337
Wildfire, May (acres)	206	0	0	4,406	938
Wildfire, June	394	0	0	3,733	1,031
Wildfire, July	1,739	25	0	32,053	6,806
Wildfire, August	205	0	0	1,865	556
Wildfire, September	21	0	0	338	73
Zion NP					
Visitation (# per month)	221,152	230,959	47,283	479,538	116,357

Wildfire, May (acres)	696	13	0	6,177	1,736
Wildfire, June	8,171	667	0	73,919	17,099
Wildfire, July	6,695	2,112	0	40,898	9,815
Wildfire, August	1,150	696	0	11,165	2,327
Wildfire, September	220	34	0	943	304

Visitation Data

The National Park Service maintains historical data about the monthly number of visitors to each national park (USNPS, 2016a). Reported statistics vary by park, with some parks reporting only the number of visitors, while other parks also report the number of overnight stays and the total number of hours on site. The metric common to all national parks was aggregate monthly visitation, so this measure was used as our visitation number, v_t. Data were collected for the five national parks for all months between May 1993 and December 2015 (273 observations for each park). Figure 8.1 presents the number of visitors in each national park during the sample period. Using 2015 visitation as a reference, the annual number of visitors was 1.40 million for Arches NP, 1.75 million for Bryce Canyon NP, 0.63 million for Canyonlands NP, 0.94 million for Capitol Reef NP, and 3.65 million for Zion NP, respectively. In 2015, the total number of visitors to all five national parks is 8.37 million. As shown in Figure 1, the data exhibit strong seasonality in visitation, with the peak season between May and September. The seasonality clearly evident in Figure 8.1 means that, econometrically, one can expect autocorrelation[16] in the model.

[16] Autocorrelation (also known as serial correlation) refers to the correlation of a time series with its own past (and future) values. For example the number of visitors to a National Park in July might be related to the number of visitors in May and June of the same year, as well as the number of visitors in July of the previous year. In this case estimated coefficients remain unbiased but are not efficient—they no longer have minimum variance (Greene, 2000). As a result, confidence intervals and hypothesis tests based on the t and F distributions are unreliable. Fortunately we can adjust for this problem to obtain estimated coefficients with desirable properties.

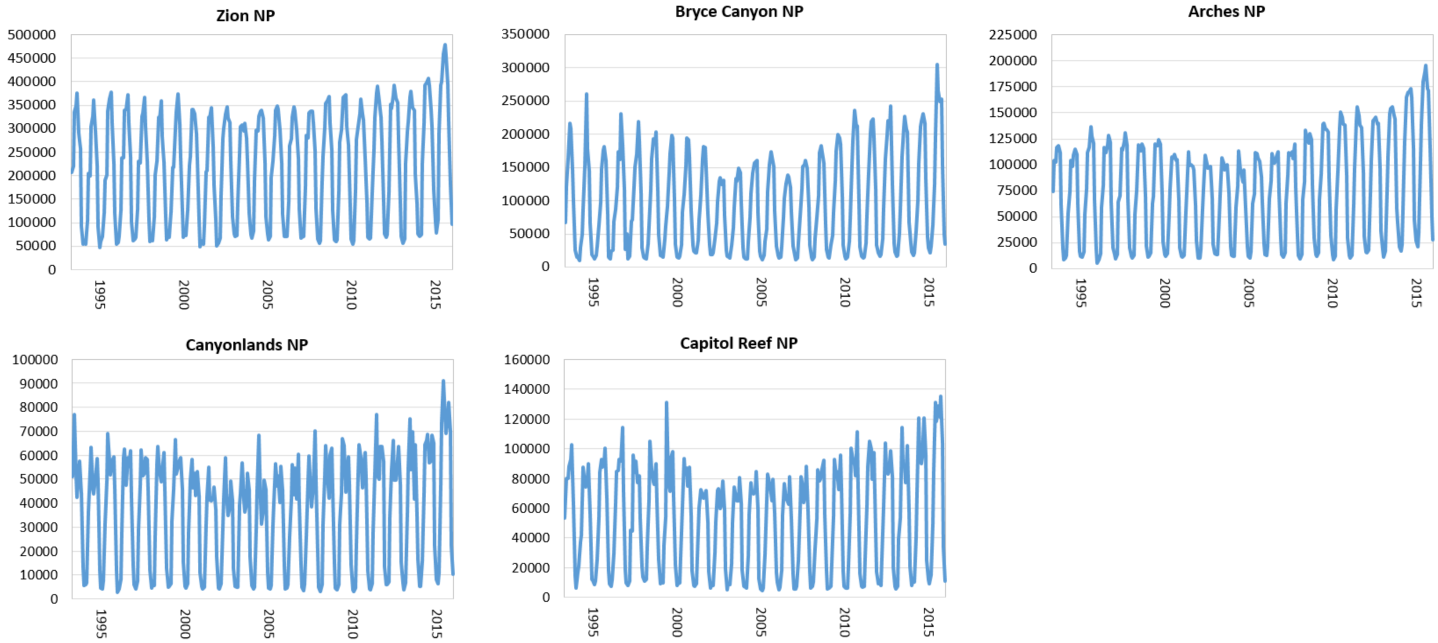


Figure 8.1. Number of Visitors in National Parks in Utah (persons)

Source: National Park Service (2016a)

Wildfire Data

The wildfire data set used for this portion of the study was described in Chapter 3. Short's cleaned wildfire occurrence data was combined with 2014 and 2015 wildfire data downloaded from National Federal Fire Occurrence database. This dataset contains 4,620 fires of 5 acres or greater in the state if Utah over the time range (1992-2015). We select only those fires occurring between May 1993 and December 2015, after which geo-location coordinates (longitude and latitude) for each wildfire are used to calculate distance between the fire origin and the visitor centers[17] of each NP. Any wildfire igniting outside a 50 mile radius of any national park's visitor center is eliminated, leaving a total of 990 wildfires as possibly influencing visitation at one or more national park (50 mile radii overlap for Zion NP and Bryce NP, and for Arches NP and Canyonlands NP). The total burned acreage of a fire was assigned to the month the fire started. For each park and for each month, all fires within the 50 mile zone are summed to create a variable measuring monthly fire activity in, or in close proximity, to national parks. Figure 8.2 shows a plot of the fire data and the locations of five national parks included. From 1993 to 2015, the average size of wildfires within a 50 mile radius of a national park was 670 acres. The largest fire close to a national park during the period of interest is the lightning-caused 88,421 acre Diamond Creek fire, which occurred in June 2002 near Arches NP.[18]

[17] Geolocations of a park visitor center or entrance was obtained from each national park's webpage and/or Google Maps.

[18] This fire was contained two months later, on August 22. While the fire occurrence data set reliably includes the start date of the 4,620 fires of interest to our study, almost 18% of fires do not have a reported contain date. Thus, our empirical analysis is limited to only the start date of fires.

The size distribution of the fire is highly skewed as most fires were relatively small. Some 71% of wildfires burned less than 100 acres, whereas there are only 11 wildfires that burned more than 10,000 acres (Table 8.2 and Figure 8.3). This brings us back to the issue raised in chapter 2 about how one measures an "average fire season". The mean acreage per fire is quite high, at 670 acres while the median acreage—the acreage burned that divides the fire distribution exactly in half, with 50% of fires being smaller and 50% being larger—is 26 acres. A single large wildfire, such as the Diamond Creek fire, can heavily skew the data. For Arches NP the mean size of fires in June was 5,218 acres whereas the median fire size was 39 acres (Table 8.2). We can use this variation in fire size to conduct sensitivity analysis.

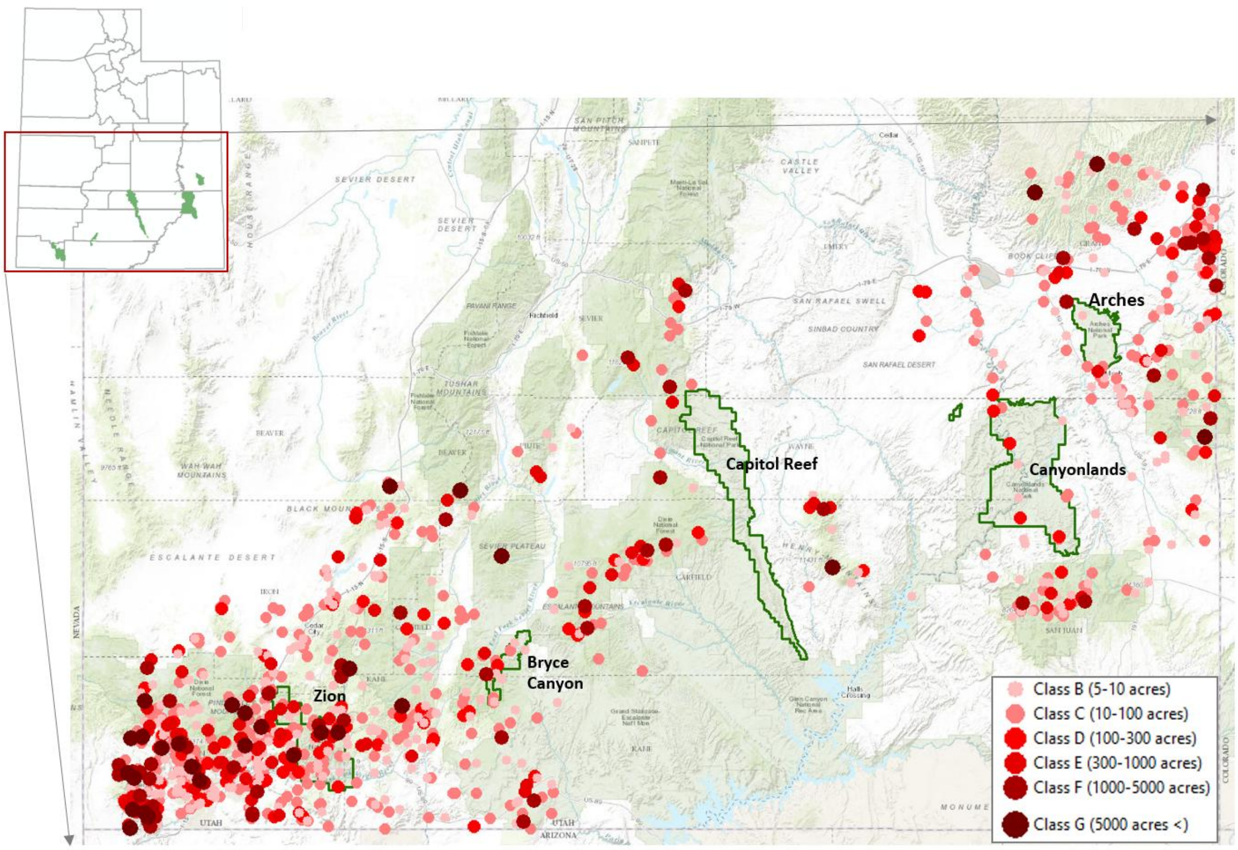


Figure 8.2. Plot of Wildfire Activities near National Parks in Southern Utah (May 1993 – December 2015)

Note: Red dots are origins of wildfires. Dark colors indicate the larger size of wildfires.

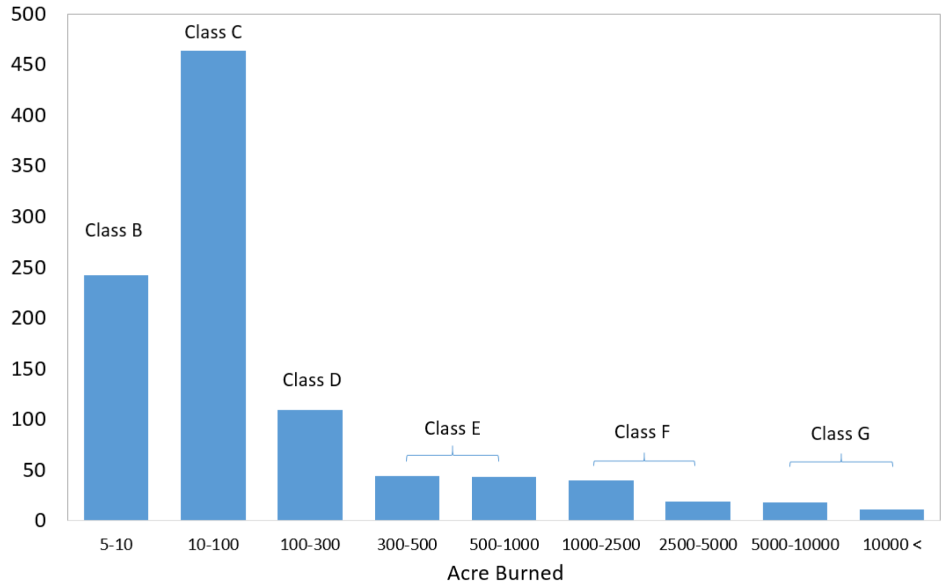


Figure 8.3. Wildfire Size Distribution within 50 miles Radius to National Parks in Utah (990 wildfires)

Economic Data

The gasoline price was obtained from U.S. Energy Information Administration (EIA) and adjusted for inflation to a 2015 "real gasoline price" and again adjusted by real personal income per capita. Recreation is considered a luxury good (an income elasticity greater than one) and is thus sensitive to broader economic forces that can affect income. We capture the influence of economic recession using indicator variables that take on a value of one during times of recession and zero otherwise. Two recessions occurred during our time frame: the first was the "dot.com" recession from April 2001 through November 2001, and the second was the "great Recession from January 2008 through June 2009. Beginning and ending dates for each recession were drawn from the Recession Indicators for the U.S. as calculated by the National Bureau of Economic Research and reported at the "FRED" website of the St. Louis Federal Reserve Bank (FRED, 2016).

The Utah Office of Tourism began a marketing campaign focusing on the five National Parks in Utah in April 2013 and has promoted out-of-state visitation to Utah through integrated communications, marketing, and travel trade initiatives. The "Mighty 5" campaign has been considered highly successful in bringing more visitors to Utah's National Parks. We include an indicator variable for the ad campaign in our empirical model to test if it can be distinguished from the broader national trend observed in recent years of increasing national park visitation.

Estimation Results and Loss in Visitation

Estimated monthly visitation models in equation (1) are shown in Table 8.3. The dependent variable is the natural log of the number of visitors (i.e., we are estimating a semi-log model). The key explanatory variables are total acres burned within 50 miles radius, $wf50$, of each national park for the current month and the previous month. Other explanatory variables include the income adjusted real price of gasoline (*adj_r_gas_p*), a simple time trend (*t*), and a variable indicating if the nation was in a *recession* during a particular month.

All models in Table 8.3 are satisfactorily explanatory ($R^2 > 0.95$) and most of variables are statistically significant at the 5% level or less. The Breusch-Godfrey test confirmed a problem with high order of autocorrelation: the error in predicting visitation in one month is correlated with the error for the same month in the previous year. We adjust for this problem by using Newey-West (1987) robust standard errors with 12 lags.

Table 8.3. National Parks Visitation Models (Semi-Log Model)

	(1) Arches	(2) Bryce	(3) Canyonlands	(4) Capitol Reef	(5) Zion
$wf50_t$	-0.000001433***	-0.000003365*	-0.000009225	-0.000006009***	-0.000000674*
	(0.002)	(0.074)	(0.561)	(0.000)	(0.094)
$wf50_{t-1}$	-0.000001864***	-0.000005209***	-0.000005104	-0.000005391***	-0.000000648
	(0.000)	(0.001)	(0.675)	(0.000)	(0.338)
adj_r_gas_p	-2.3718	-3.3074	0.4352	0.1124	-1.9242
	(0.384)	(0.419)	(0.895)	(0.978)	(0.357)
Mighty 5	0.2660***	0.2455***	0.3160***	0.3514***	0.09272
	(0.000)	(0.010)	(0.000)	(0.001)	(0.101)
t	0.001411***	0.0007718	-0.00003943	-0.0005035	0.001163***
	(0.000)	(0.154)	(0.905)	(0.223)	(0.000)
recession	-0.02624	-0.02479	-0.02584	-0.04479	-0.05957**
	(0.471)	(0.552)	(0.524)	(0.386)	(0.025)
Constant	9.3278***	9.8017***	8.4836***	9.0123***	10.991***
	(0.000)	(0.000)	(0.000)	(0.000)	(0.000)
depvar	ln(v_arch)	ln(v_bryce)	ln(v_canyon)	ln(v_capitol)	ln(v_zion)
N	272	272	272	272	272
R^2	0.989	0.980	0.988	0.983	0.986
F statistic	673.76	407.53	758.61	431.85	627.18

Note: Numbers in parentheses are P-values. Significance levels are 1% (***), 5% (**), and 10% (*).
Note: Results for monthly dummies are omitted to save space. Most of dummies are statistically significant.

The negative coefficient on the income adjusted real price of gasoline indicates that higher gasoline prices (increased travel costs) result in a fall in visitation but all of the coefficients are not statistically significant. The positive coefficient on the Mighty 5 dummy shows an indication of a successful marketing campaign. The positive coefficient on the time trend shows an increasing trend in national park visitation over time. The estimated parameter for recession indicates that, all else equal, a nationwide recession results in reduced visitation to Utah's national parks, but the coefficient was not statistically significant anywhere but Zion NP.

Turning to the wildfire coefficients, we find that wildfire activities have statistically significant negative impact on visitation in all of Utah's national parks except Canyonlands NP (model 3 in Table 8.2). Arches, Bryce, and Capitol Reef National Parks each show current and lagged effects of wildfires in close proximity to park entrances, whereas Zion NP exhibits reduced visitation for only current month wildfires (lagged effects are not significant). Current and lagged effects may occur because people can alter vacation plans in response to wildfire. For example, tourists may choose to forgo a visit to Zion NP and instead spend more time at, say, the Grand Canyon NP or Las Vegas upon hearing of wildfire activity in or near Zion NP. The semi-log form of the model allows us to easily calculate the relative change in visitation for a given change in an explanatory variable. For this model, a one unit change in an explanatory variable yields a $\beta \times 100$ % change in visitation. Thus, we can provide a numeric interpretation for the coefficients by considering the effect of a hypothetical 100 acre fire occurring near or in a national park. For example, a 100 acre fire within the 50 mile radius of Zion NP depresses current month visitation by 0.007% [(100 acres) × (-6.744×10^{-7}) × 100%)]. For Arches NP, the effect of a 100 acre wildfire is a 0.014% fall in the month concurrent with the wildfire and 0.019% fall in the month after the wildfire, for a total loss of about 0.033%. Similar calculations can be done for Bryce NP (0.086%) and Capitol Reef NP (0.114%).

Even using the calculations presented in the previous paragraph, we still don't know the predicted change in the number of visitors to a park. To investigate the impact of wildfire activity on park visitation we use a multi-step simulation approach, where monthly wildfire acreage burned is considered a random variable:

1. Generate random wildfires within the 50 mile radius of each national park based on the historical spatial distribution, timing, and size of wildfires. Intertemporal correlation among months is considered in the random draws.
2. For each park and its simulated monthly wildfires, calculate the effect of wildfire acreage on the number of visitors to each park using the model coefficients reported in Table 8.3. All variables other than wildfire acreage are fixed at their 2015 values.
3. Find the difference between the number of monthly visitors "with wildfire" (calculated in step 2) and the predicted visitors assuming zero wildfires in that month.
4. Repeat the steps one through three 1,000 times to generate an empirical distribution of wildfire effects on visitation at each park.

The skewed distribution of wildfire acreage results in a skewed empirical distribution for visitation losses. Hence, we report both median and median visitation losses arising from the 1,000 random wildfire draws. Wildfire activity is concentrated in the summer months, so the percentage changes in monthly visitation predicted by the model in Table 8.2 are assigned to visits occurring in the months of May through September (peak season). The implicit assumption is that off-peak season wildfires do not affect national park visits; given the relatively few fires occurring in off-peak months, this assumption seems warranted. Table 8.4 presents the changes (losses) in visitation in each national park.

Table 8.4. Visitation Losses due to Wildfire

National Park	Mean	% of peak season (May-Sep)	Median	% of peak season (May-Sep)
Arches	3,692	-0.41%	432	-0.05%
Bryce	12,802	-1.01%	4,662	-0.37%
Canyonlands	1,498	-0.38%	352	-0.09%
Capitol Reef	2,877	-0.45%	302	-0.05%
Zion	9,983	-0.46%	5,377	-0.25%
Sum	30,851	-0.60%	11,125	-0.22%

Note: % change in peak season visitation in 2015 (May through September)

The visitation losses follow expected patterns. Visitation losses are a function of the wildfire parameter estimates (Table 8.3), the amount of burned acreage, and baseline visitation. Large wildfire parameter effects—such as those for Capitol Reef—lead to large percentage changes in visitation, but the low baseline visitation (less than 1 million visitors in 2015) means that losses in visitor days are modest. The wildfire parameter estimates for Zion NP are relatively small (leading to small percentage effects), but baseline visitation (3.65 million visitors) is high enough to generate relatively large losses in visitor numbers.

Regional Economic Impacts

Changes (loss) in visitation have effects on the regional economies of the counties that surround the national parks including counties such as Garfield (Bryce Canyon NP), Grand (Arches NP), Wayne (Capitol Reef NP) and Washington (Zion NP), where the visitor spending is crucial in the local economy. This research utilizes the Input-Output (IO) approach to measure the impact of local economies from changes in visitation due to wildfire.

Economic impacts or contributions are based on visitors' expenditures associated with visiting national parks. Expenditures include food and beverages purchased at restaurants or grocery stores, gasoline and oil, purchasing sporting goods, lodging (hotel/motel/cabin/camping), equipment and rentals, and other transportation expenses. Expenditures affect the local and regional economy through the inter-relationships among different sectors or industries of the local economy. Multipliers can be described through the following definitions:

- Direct effects (or direct expenditures) are the changes in the industries associated with visitors (direct) expenditure. We have direct impacts from hotel/motel/cabin lodging, grocery purchases from the local stores, restaurants, gasoline purchase, equipment rentals, local transportation (bus, shuttles), etc.
- Indirect effects are the changes in inter-industry purchases as they respond to the new demands of the directly affected industries. The direct effect creates increases in economic activity for additional businesses (in the region) that support these direct industries.
- Induced effects are the increases in household income expenditures generated by the direct and indirect effects. In other words, induced effects are created as the new income

generated by the direct and indirect effects is spent and re-spent within the local economy.

- Total economic contribution is the sum of direct effects, indirect effects, and Induced effect, and multiplier is the ratio of the total effect to the direct effect.

Our economic impact analysis is based on direct expenditures by park visitors as gathered by the US National Park Service (2016b). For example, visitors to Arches NP spent $162.7 million in the year 2015, including $58.1 million for lodging, $9.2 million for local grocery purchases from the local stores, $34.9 million at restaurants, $11.4 million for the purchase of gasoline, $15.9 million on services provided by recreation industries, $11.4 million on local transportation (bus, shuttles), etc. Table 8.5 presents 2015 direct expenditures in million dollars as reported on the NPS Visitor Spending Effects website.

Table 8.5. Direct Expenditures in 2015 (million dollars, $2015)

	Arches	Bryce	Canyon-lands	Capitol Reef	Zion	Sum
2015 visitors (million)	1.399	1.746	0.634	0.941	3.649	8.370
Expenditures						
Gas	11.4	14.8	5.7	9.8	16.4	58.1
Groceries	9.2	8.7	2.6	3.8	11.0	35.3
Hotels	58.1	48.7	12.5	26.3	67.5	213.1
Recreation industries	15.9	13.7	2.9	2.8	4.1	39.4
Restaurants	34.9	26.6	7.2	12.7	47.8	129.2
Retail	17.9	14.3	3.7	4.5	23.7	64.1
Transportation	11.4	15.2	2.2	4.6	25.8	59.2
Camping	3.7	3.9	1.1	2.1	5.8	16.6
Sum	162.5	145.9	37.9	66.6	202.1	615.0

Source: US National Park Service (2016b)

We can use the changes in visitation reported in Table 8.4 to calculate the change in direct expenditures due to wildfire activities, $\Delta Expnd$, by the following:

(2) $$\Delta Expnd_i = \sum_j \Delta v_i \cdot \frac{expnd_{ij}}{total\ v_i},$$

where Δv_i is the change in visitation in national park i= Arches, Bryce Canyon, Canyonlands, Capitol Reef and Zion, $expnd_{ij}$ is the direct expenditure in a category j = gas, groceries, …, camping in Table 8.5, and $total\ v_i$ is the annual visitor numbers in each national park in 2015 reported in Table 8.5 as well.

The losses in visitor spending in the local economy are shown for each park on the basis of mean visitation losses (Table 8.6) and median visitation losses (Table 8.7). The loss of 3,692 visitors to Arches (Table 8.4, based on mean acreage burned) results in a loss of $429,287 in visitor spending (Table 6). For the median acreage burned, Arches lost 432 visitors (Table 4) for a total loss of $50,208 in visitor spending (Table 8.7). Similar calculations are presented for all national parks under both fire scenarios. Tables 9.6 and 9.7 show the aggregate loss in visitor spending across all national parks to be between $0.780 million (median visitation loss) and $2.345 million (mean visitation loss).

Table 8.6. Loss in Visitor Spending – Mean ($2015)

	Arches	Bryce	Canyonlands	Capitol Reef	Zion	Sum
Gas	30,079	108,527	13,453	29,957	44,868	226,884
Groceries	24,274	63,796	6,136	11,616	30,095	135,917
Hotels	153,298	357,112	29,501	80,394	184,672	804,977
Recreation industries	41,952	100,461	6,844	8,559	11,217	169,034
Restaurants	92,084	195,055	16,993	38,821	130,775	473,729
Retail	47,230	104,860	8,732	13,756	64,840	239,418
Transportation	30,079	111,460	5,192	14,061	70,586	231,378
Camping	9,763	28,598	2,596	6,419	15,868	63,244
Sum	428,760	1,069,870	89,448	203,583	552,921	2,344,581
% of visitor spending in 2015	0.26%	0.73%	0.24%	0.31%	0.27%	0.38%

The regional economic model that calculates the direct, indirect, induced, and total effects builds upon models using the IMPLAN software for the year 2013. The six counties that encompass the bulk of southern Utah—Garfield, Grand, Kane, San Juan, Washington, and Wayne—are aggregated into a single economic region that is home to all of Utah's national parks. The regional economy is further aggregated to 13 sectors from Implan's 435 disaggregated sectors. While most of the economic sectors reported in the tables below are highly aggregated, we maintain disaggregated sectors for those sectors that are assumed to be most impacted by wildfire-related losses in visitor spending, e.g., accommodation (hotels/motels/others), restaurants, recreation industries, which are broken out in detail. Other key visitor expenditure categories such as gas, groceries and retail, are aggregated into the retail trade sector.

Table 8.7. Loss in Visitor Spending – Median ($2015)

	Arches	Bryce	Canyonlands	Capitol Reef	Zion	Sum
Gas	3,518	39,524	3,162	3,150	24,165	73,519
Groceries	2,839	23,234	1,442	1,221	16,209	44,945
Hotels	17,929	130,055	6,933	8,453	99,461	262,832
Recreation industries	4,907	36,586	1,609	900	6,041	50,043
Restaurants	10,770	71,036	3,994	4,082	70,433	160,315
Retail	5,524	38,189	2,052	1,446	34,922	82,133
Transportation	3,518	40,592	1,220	1,478	38,016	84,825
Camping	1,142	10,415	610	675	8,546	21,388
Sum	50,146	389,632	21,022	21,405	297,795	780,000
% of visitor spending in 2015	0.03%	0.27%	0.06%	0.03%	0.15%	0.13%

The gross regional product for the six county area was $6.096 billion (total value-added); this level of economic activity supported an estimated 97,497 jobs. Major economic sectors include FIRES (finance, information, real estate, education, and other services), which supported 43,102 jobs and government, which were estimated to support 11,887 jobs. Retail trade produces $812 million and supports 11,473 jobs. The restaurant sector produces $398 million and supports about 8,125 jobs in 2013 whereas the accommodation sector produces $240 million and hires 3,069 employees.

The estimated regional economic impact of wildfire-related losses in visitor spending is shown in Tables 9.8 and 9.9. The total loss of industry output associated with decreased expenditures by visitors is $3.654 million (mean visitation loss, Table 8.8) and $1.216 million (median visitation loss, Table 8.9). Relative to the gross change in expenditures, losses in output correspond to an effective expenditure multiplier of 1.56, which is reasonable for a relatively small economic region; that is, every dollar spent in the national parks generates $1.56 in total economic output.

The loss in value-added (net regional output) resulting from decreased industry output was estimated to be $1.993 million (mean loss in visitation) and $0.662 million (median loss in visitation), respectively. A portion of the value-added impact is the loss of income accruing to labor: losses in labor income are estimated to be $1.152 million (mean loss in visitation), which includes losses of 42 full- and part-time jobs (Table 8.8). In the median visitation loss case, losses in labor income were $0.383 million loss and a loss of 14 jobs full and part-time jobs.

Tax revenues are also affected by losses in the level of output, labor income, and value added; under the mean visitation loss scenario state and local governments could expect to see losses of $0.268 million whereas the federal government could experience losses of $0.292 million. In case of median loss in visitation, the loss of tax revenue was estimated to be $0.129 million for state/local government and $0.142 million for federal government.

Table 8.8. Economic Loss of Decreased in Visitor Spending from Wildfires in National Parks (Mean Loss in Visitation)

Sector	Industry Output	Value Added	Labor Income	Employment
	(dollars)			(persons)
Agriculture	2,397	1,277	318	0
Mining	3,406	2,163	621	0
Utilities	38,738	8,932	4,825	0
Construction	38,446	13,541	10,410	0
Manufacturing	13,603	3,785	1,661	0
Wholesale	48,403	27,567	11,425	0
Retail trade	704,210	421,319	261,150	10
Transport & Warehousing	309,120	141,100	89,515	2
FIRES[1]	865,042	492,646	206,756	7
Recreation	172,901	96,682	56,482	3
Accommodation	870,957	475,886	274,469	10
Restaurant & Food Services	540,995	266,447	202,272	10
Government	45,656	41,916	32,077	1
Total[2]	3,653,874	1,993,261	1,151,981	42

[1] FIRES = Finance, Insurance, Real estate, Educational services, and other services
[2] May not sum to total due to rounding

Table 8.9. Loss of Decreased in Visitor Spending from Wildfires in National Parks (Median Loss in Visitation)

Sector	Industry Output	Value Added	Labor Income	Employment
	(dollars)			(persons)
Agriculture	797	424	106	0
Mining	1,140	724	208	0
Utilities	12,815	2,955	1,596	0
Construction	12,750	4,491	3,452	0
Manufacturing	4,556	1,268	556	0
Wholesale	16,191	9,221	3,822	0
Retail trade	234,494	140,295	86,960	3
Transport & Warehousing	111,203	50,759	32,202	1
FIRES[1]	287,343	163,643	68,679	2
Recreation	51,326	28,701	16,767	1
Accommodation	285,129	155,793	89,854	3
Restaurant & Food Services	182,618	89,942	68,279	4
Government	15,248	13,999	10,713	0
Total[2]	1,215,610	662,215	383,194	14

[1] FIRES = Finance, Insurance, Real estate and Educational services, and other services
[2] May not sum to total due to rounding

Summary

This chapter has quantified the effect of wildfire on recreation visitation at national parks in Utah. Using monthly data from May 1993 to December 2015, we empirically linked wildfire activities (measured as monthly acres burned within a 50 mile radius) to monthly visits to each national park. Results show that wildfire activities have negative and statistically significant concurrent and lagged effects on visitation (Arches, Bryce Canyon, Canyonlands, and Zion NPs but not Capitol Reef NP). We find that there is 0.1%~1.0% loss in aggregate visitation due to wildfire, that is, a seasonal loss of between 11,125 to 30,851 visitors relative to visitor numbers that would occur in the absence of wildfire. We also estimated the regional economic impacts of losses in visitor spending due to the decrease in visitation. The loss in direct visitor spending was estimated to be between $0.78 million and $2.34 million. Visitation and spending is related to the regional economies where national parks are located and supports regional businesses such as hotels and restaurants, and creates jobs in private sectors. The regional economic impact of wildfire activities is estimated to be a seasonal loss between $1.22 and $3.65 million. Counties where national parks are located may lose between 14 and 42 jobs depending on the extent of acreage burned in proximity to national parks. Wildfire-related reductions in expenditure also decrease the tax revenue for state and federal governments (seasonal losses between $0.19 million~$0.56 million).

References

Boxall P, and J. Englin. 2008. "Fire and Recreation Values in Fire-prone Forests: Exploring an Intertemporal Amenity Function using Pooled RP-SP Data." *Journal of Agricultural and Resource Economics* 33: 19–33.

Duffield, J.W., C.J. Neher, D.A. Patterson, and A.M. Deskins. 2013. "Effects of Wildfire on National Park Visitation and the Regional Economy: A Natural Experiment in the Northern Rockies." *International Journal of Wildland Fire* 22: 1155-1166.

Englin J, P. Boxall, K. Chakraborty, and D. Watson. 1996. "Valuing the Impacts of Forest Fires on Backcountry Forest Recreation." *Forest Science* 42: 450–455.

Englin J, J.B. Loomis, and A. Gonzalez-Caban. 2001. "The Dynamic Path of Recreational Values Following a Forest Fire: A Comparative Analysis of States in the Intermountain West." *Canadian Journal of Forest Research* 31: 1837–1844.

Federal Reserve Economic Data (FRED). 2016. "NBER based Recession indicators for the United states from the Period following the Peak through the Trough." https://fred.stlouisfed.org/series/USREC

Greene, W.H. 2000. *Econometric Analysis*, Fourth edition, Prentice Hall.

Hesseln H, J.B. Loomis, A. Gonzalez-Caban, and S. Alexander. 2003. "Wildfire Effects on Hiking and Biking Demand in New Mexico: a Travel Cost Study." *Journal of Environmental Management* 69: 359–368.

Hesseln H, J.B. Loomis, and A. Gonzalez-Caba. 2004. "Comparing the Economic Effects of Fire on Hiking Demand in Montana and Colorado." *Journal of Forest Economics* 10: 21–35.

Leaver, J. 2016. "The state of Utah's travel and tourism industry." http://gardner.utah.edu/wp-content/uploads/2016/05/TourismReport-v7.pdf

Love T.G. and A.E. Watson. 1992. "Effects of the Gates Park Fire on Recreation Choices." USDA Forest Service, Intermountain Research Station, Research Note INT-402. (Ogden, UT).

McCombs, B. 2014. "Feds: Utah move to reopen national parks dring government shutdown paid off." Deseret News, March 3, 2014.

Newey, W.K., and K.D. West. 1987. "A Simple, Positive Semi-definite, Heteroskedasticity and Autocorrelation Consistent Covariance Matrix." *Econometrica* 55: 703–708.

SkiUtah.com. 2016. "Utah Ski Areas Set Historical Skier Visit Number for 2015-16 Season." https://www.skiutah.com/news/authors/pr/utah-ski-areas-set-historical-skier

United States National Park Service. 2016a. "National Parks Visitor Use Statistics". https://irma.nps.gov/Stats/Reports/Park

United States National Park Service. 2016b. “2015 National Park Visitor Spending Effects”. Natural Resource Report NPS/NRSS/EQD/NRR-2016/1200. (April).

Vaux H Jr., P.D. Gardner, and T.J. Miller. 1984. “Methods for assessing the Impact of Fire on Forest Recreation.” USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, General Technical Report PSW-79.

Chapter 9: Wildfire, Urban Air Quality, and Health Impacts

Man-Keun Kim
Dept. of Applied Economics and the Center for Society, Economy, and the Environment
Utah State University

Introduction

Numerous studies have provided evidence of the direct relationship between poor air quality and human health problems, especially fine particles which can affect the heart and lungs, and cause serious adverse health effects. When people inhale PM2.5, fine particles with 2.5 micrometers in diameter or smaller, some particles are deposited along the respiratory tract while others penetrate deeply into the lung where they can enter the bloodstream (U.S. EPA 2003). These particles aggravate the severity of chronic lung diseases and impair airway functions, causing inflammation of lung tissue which results in the release of chemicals that impact heart functions and lead to changes in blood chemistry that produces clots which can cause heart attacks (U.S. EPA 2003). According to U.S. EPA (2009), exposure to PM2.5 is responsible for causing cardiovascular including heart attacks and its associated mortality. PM2.5 also causes increases in hospital admissions for breathing problems, including respiratory illnesses such as asthma, and is linked to other adverse respiratory, reproductive, developmental, and cancer outcomes.

Dockery, Schwartz, and Spengler (1992), Pope (2000), and Pope, Burnett, and Thun (2002) concluded that exposure to PM2.5 has been consistently linked with increased mortality from cardiopulmonary diseases, lung cancer, and numerous other respiratory illnesses and associated morbidity. Pope (2000) and Pope, Burnett, and Thun (2002) found that a $10 \mu g/m^3$ increase in ambient PM2.5 concentration was associated with approximately a 4% increased risk of all-cause mortality, a 6% increased risk of cardiopulmonary mortality, and an 8% increased risk of lung cancer mortality. Recent epidemiological studies also have shown that high levels of PM are closely correlated with substantial adverse health effects such as acute respiratory infections and mortality in the short-term (Chen et al. 2000; Sastry, 2002; Tham et al. 2009). Long-term exposure to the combustion-related PM and SO_2-related air pollution could lead to cardiopulmonary and lung cancer (Viswanathan et al. 2006).

Wildfire smoke, which can travel several hundred miles (Mustain, 2012), can be a major contributor to a deterioration in air quality and cause adverse health effects (Moeltner et al., 2013). Jayachandran (2009) examined the impact of the 1997 wildfires in Indonesia on fetal, infant, and child mortality and found that smoke from wildfires and (air) pollution led to 15,600 deaths amongst the target population (1.2% of the affected birth cohorts). Emmanuel (2000) investigated impact of the same wildfire on neighboring Singapore. Findings from the health impact study showed that a 30% increase in outpatients treated for haze-related conditions, a 12% increase in upper respiratory tract illness, and 19% and 26% increases in patients treated for asthma and rhinitis.

Burty et al. (2001) analyzed economic impacts of the catastrophic wildfires in northeastern Florida in June and July 1998; the Florida wildfires produced economic impacts of at least $600 million. Viswanathan et al. (2006) also analyzed the impact of major gaseous and particulate pollutants emitted by the wildfire of October 2003 in San Diego on ambient air quality and health of San Diego residents. The study showed that the increased PM concentration above the federal standard due to wildfire resulted in a significant increase in hospital emergency room visits for asthma, respiratory problems, eye irritation, and smoke inhalation.

Moeltner et al. (2013) investigated wildfire smoke and health impacts in Reno/Sparks, Nevada area over a 4-year period. They related the daily acreage burned by wildfires to daily data on air pollutants (PM2.5) and local hospital admissions. The results indicate that seasonal wildfire smoke can cause considerable health costs of several million dollars in a single, medium-sized city, as measured by inpatient treatment expenses for respiratory and cardiovascular patients.

Jones et al. (2016) estimated wildfire smoke health costs in the case of a Wallow mega-fire in southeastern Arizona on Albuquerque, New Mexico (300 miles away from the burn site) using US EPA benefits mapping and analysis program (BenMAP-CE). This study illustrated BenMAP-CE program can be applied to wildfire smoke damage assessment (in an urban area). They found substantial increases in respiratory and cardiovascular incidences associated with smoke exposure from the wildfire event.

This chapter attempts to investigate the relationship between PM2.5 concentration in the Salt Lake City (SLC) metropolitan area and surrounding wildfires using historical wildfire records and PM2.5 monitoring data. Results show that the level of PM2.5 has been affected by wildfire activities which are measured in acres burned. On average, wildfires near the SLC metropolitan area increased PM2.5 concentration by 1.62 $\mu g/m^3$ during June, July and August. These concentrations might be associated with a 0.5% increase in respiratory patients and a 0.8% increase in cardiovascular hospital admissions.

Data

The time frame for the analysis ranges from July 2004 to October 2015, for a total of 136 months. This is based on the availability of data on air quality in SLC. The goal of the analysis is to relate acres burned by wildfires within a 200-mile radius to an air quality monitoring station near the center of SLC.

Wildfire Data

The wildfire data set used for this chapter was described in Chapter 3. Short's cleaned wildfire occurrence data was combined with 2014 and 2015 wildfire data downloaded from National Federal Fire Occurrence database. This dataset contains 4,620 fires of 5 acres or greater in Utah over the time range (1992-2015). We select only those fires occurring between July 2004 and October 2015 within a 200-mile radius to SLC (1,621 fires are identified). Wildfire activities are measured in acres burned. The wildfire activities are subdivided into three groups (zones) based on the distance to SLC, i.e., less than 50 miles (zone 1), between 50 and 100 miles (zone 2), and between 100 and 200 miles (zone 3). Wildfire data are then aggregated (summed) to monthly data.

Table 9.1 presents basic statistics of wildfire data and Figure 9.1 presents annual wildfire activities from 2004 through 2015. As shown in Figure 9.1, serious wildfire activity occurred in 2007 and 2012.

Table 9.1: Individual Wildfire Data (July 2004 through October 2015)

	Number of fires	Mean (acres)	Median (acres)	Min (acres)	Max (acres)	Std. Dev. (acres)
Zone 1 (≤ 50 miles)	475	464	27	5	44,345	2,738
Zone 2 (51 – 100 miles)	603	1,003	35	5	48,986	4,682
Zone 3 (101 – 200 miles)	543	1,570	38	5	357,185	16,281

Note: The wildfire activities are subdivided into three groups (zones) based on the distance to Salt Lake City, i.e., less than 50 miles (zone 1), between 50 and 100 miles (zone 2), and between 100 and 200 miles (zone 3).

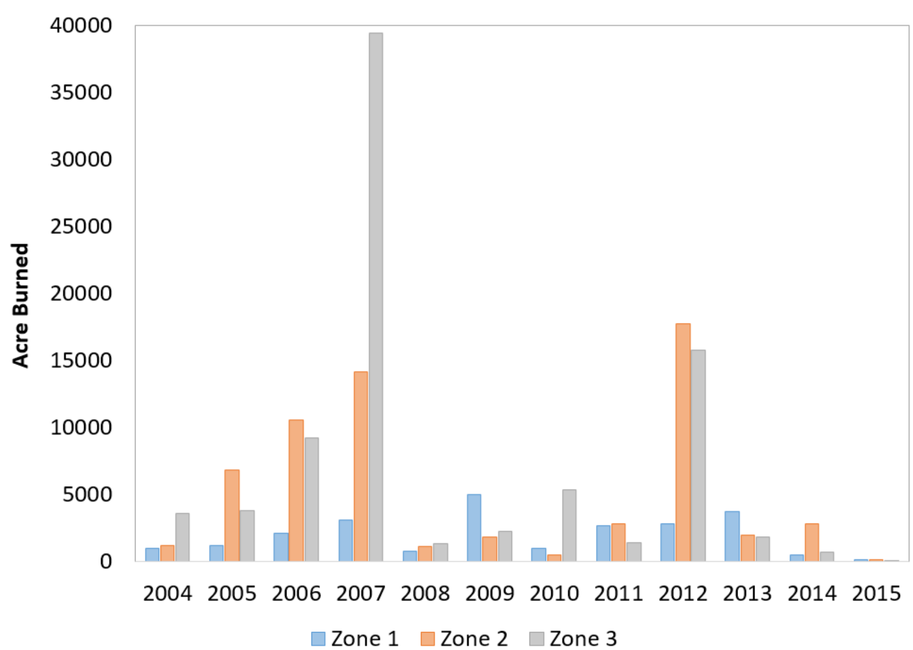


Figure 9.1: Annual Wildfire Activities in Acres Burned

Note: Wildfire activity is subdivided into three groups (zones) based on the distance to Salt Lake City, i.e., less than 50 miles (zone 1), between 50 and 100 miles (zone 2), and between 100 and 200 miles (zone 3). Annual wildfire activities are the sum of acre burned in a year.

PM2.5 Concentration Data

PM2.5 concentration data are compiled from Utah Department of Environmental Quality – PM2.5 Data Archive. There are several monitoring stations in SLC area. We collect PM2.5 data measured at Hawthorne monitoring station (1675 S. 600 E. Salt Lake City, near the intersection of I-15 and I-80).

Figure 9.2 presents *daily* average PM2.5 concentration for 2007 with the AirNow.gov air quality index (AQI) to understand the relationship between air quality and PM2.5 concentrations. Colored dotted lines in Figure 9.2 indicate the AQI demarcations for the verbal description of air quality. PM2.5 concentrations of ≤12 are considered good. When PM2.5 concentrations are between 12.1 – 35.4 $\mu g/m^3$, represented by the area above the yellow dotted line and below the brown dotted line (corresponding to an AQI range from 51 to 100), air quality is considered acceptable, however, there may be a "moderate health" concern for a small number of people for some pollutants. When PM2.5 concentrations are between 35.5 and 55.4 $\mu g/m^3$, represented by the area above the brown dotted line and below the red dotted line, the AQI ranges between 101 and 150, and air quality is considered unhealthy for sensitive groups such as those with respiratory ailments, older adults, and children. When the PM2.5 concentration is in the 55.5 – 150.4 $\mu g/m^3$ range, the region above the red dotted line, the AQI is between 151 and 200, and air quality is considered unhealthy for the general population. Everyone may begin to experience adverse health problems, while members of sensitive groups may experience more serious health problems (U.S. EPA AirNow). Note that Figure 9.2 depicts a clear seasonal pattern with increased PM2.5 levels during winter time when inversion patterns trap polluted air during the cold season.

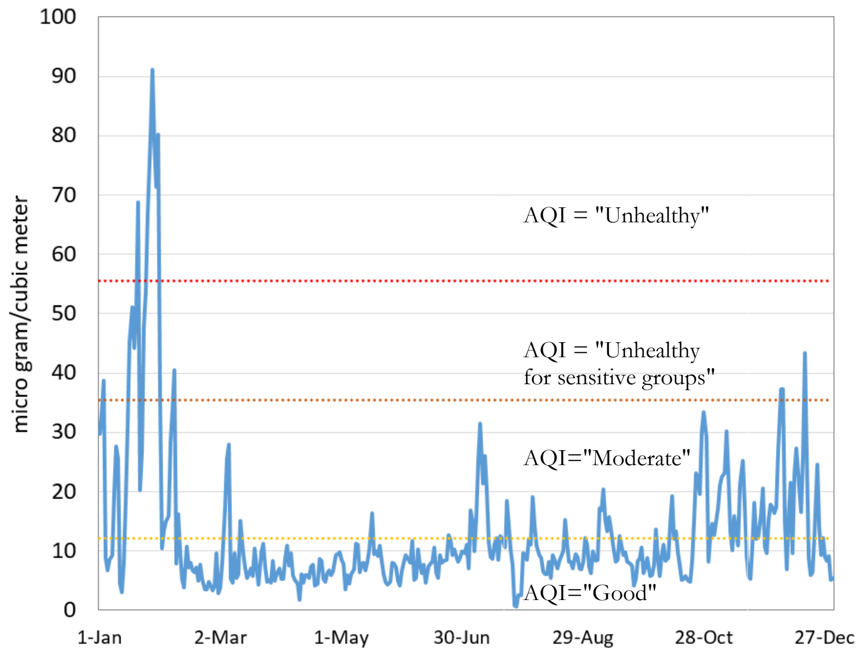


Figure 9.2: Daily Average PM2.5 Concentration measured in Salt Lake City (Hawthorne Monitoring Station) in 2007

Source: Utah Department of Environmental Quality, PM2.5 Data Archive

Wildfire and PM2.5 Concentration

Figure 9.3 provides a focus on the fire season and plots *monthly* wildfire activity (the sum of acres burned in a month) within a 100-mile radius (zones 1 and 2) of Salt Lake City's Hawthorne monitoring station during the months of May through September for the years 2004 through 2015. The blue bars measure acreage burned (in 1000 acre units) on the left axis, with the orange line measuring monthly average PM2.5 concentration during the same period ($\mu g/m^3$) on the right axis. Figure 9.3 depicts a clear temporal correspondence of elevated PM2.5 levels and acres burned. The correlation is apparent especially during the 2007 and 2012 fire seasons.

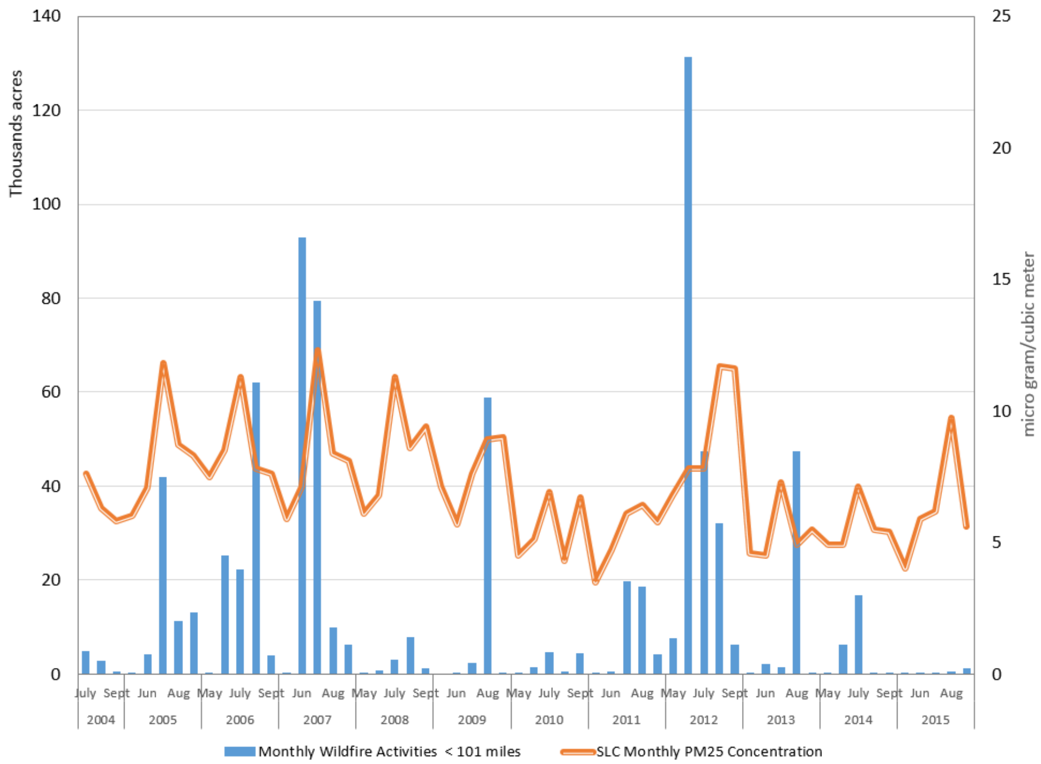


Figure 9.3: Monthly Wildfire Activities within a 100-mile Radius to Salt Lake City (left axis) and Monthly Average PM2.5 Concentration (right axis) during Fire Seasons between 2004 and 2015.

PM2.5 Modeling

Monthly average PM2.5 concentration in SLC in month t is set as the function of wildfire activities (acre burned) and meteorological variables. All individual wildfires of at least 5 acres in size within a 200-mile radius to SLC are included (1,621 wildfires are identified). The wildfire activities are subdivided into three groups (zones) based on the distance, less than 50 miles (zone 1), between 50 and 100 miles (zone 2), and between 100 and 200 miles (zone 3). The model is

$$pm25_t = \beta_0 + \beta_1 wf_{1,t} + \beta_2 wf_{2,t} + \beta_3 wf_{3,t} + \beta_3 tmp_t + \beta_4 prp_t + \beta_5 wnd_t + \beta_6 wnt + \varepsilon_t$$

Where $pm25_t$ is average PM2.5 concentration in month t, $wf_{1,t}$ is the acres burned in zone 1 in month t, $wf_{2,t}$ is acres burned in zone 2, and $wf_{3,t}$ is acres burned in zone 3. Meteorological variables are also included following Moeltner et al. (2013) such as average temperature (tmp), total precipitation (prp), and average wind speed (wnd). Winter dummy (wnt), which takes a value of one during the months November through February, is added to control for cold season inversions. Table 9.2 includes descriptive statistics for the variables used in the econometric model.

Table 9.2: Descriptive Statistics (July 2004 ~ October 2015, 136 observations)

	Mean	Std. Dev.	Min	Median	Max
PM2.5 Concentration[a] (μg/m³) - all sample	9.77	6.59	3.40	7.45	37.10
PM2.5 Concentration (μg/m³) – fire season[b]	7.11	2.12	3.50	6.85	12.30
Wildfire in zone 1 (d ≤ 50 miles)(acres)	1,620	5,432	0	7	45,101
Wildfire in zone 2 (50< d ≤ 100 miles)(acres)	4,445	15,301	0	22	122,326
Wildfire in zone 3 (100< d ≤ 200 miles) (acres)	6,270	37,293	0	10	408,417
Average Temperature (degree F)	54.42	17.68	20	53	85
Total Precipitation (inches)	1.10	0.83	0	0.92	3.73
Average Wind Speed (miles per hour)	7.95	1.37	5	8	12

Note: [a] monthly average concentration; [b] Fire season = May to September

Estimation Results

The PM2.5 model is reported in Table 9.3. The Breusch-Godfrey test confirmed a problem of high order autocorrelation. We adjust for this problem by using Newey-West (1987) robust standard errors with 12 lags. The key explanatory variables are wildfire activities in each zone (Wildfire_zone).

Table 9.3: PM2.5 Model Estimation Results

	PM2.5_hw
Wildfire_zone1	0.000117***
	(0.001)
Wildfire_zone2	0.000048**
	(0.024)
Wildfire_zone3	0.000007**
	(0.023)
Temperature	-0.1045***
	(0.000)
Precipitaton	-0.3369
	(0.497)
Wind Speed	-2.1125***
	(0.000)
Winter month	3.0894***
	(0.008)
Intercept	31.168***
	(0.000)
Depvar.	PM2.5_hw
N	136
F statistics	27.399
R^2	0.596

Numbers in parentheses are P-values.
Significance levels are 1% (***), 5% (**), and 10% (*)

As can be seen from the table, all the variables have the expected sign and are tatistically significant except precipitation. The positive coefficients on wildfire activities indicate the positive relationship between wildfire activities and PM2.5 concentration in SLC. Temperature and wind speed have negative effect, and the winter dummy variable has a positive coefficient as expected.

Table 9.4 reports model coefficients when calculated as elasticities (evaluated at mean values of other variables) and marginal effect for the wildfire activity in each zone. For example, PM2.5 concentration is increased by 0.019% when wildfires burned one percent more acres in zone 1, which is equivalent to 0.012 $\mu g/m^3$ for every 100 acres burned. The predicted PM2.5 concentration is 0.022% higher when wildfires burn one percent more acres in zone 2, which is equivalent to 0.005 $\mu g/m^3$ per 100 acres burned in zone 2. The magnitude of marginal effects from the PM2.5 model are slightly smaller than estimates of Moltner et al. (2013) which ranges between 0.08 and 0.30 $\mu g/m^3$ per 100 acres burned for Reno/Sparks, Nevada depending on distance and fuel types.

Table 9.4. Elasticities and Marginal Effects for PM2.5 Model

	Elasticity at means	Marginal effect ($\mu g/m^3$ per 100 acres burned)
Wildfire_zone1	0.0194***	0.0117
	(0.0068 ~ 0.0320)	(0.0041 ~ 0.0193)
Wildfire_zone2	0.0220***	0.005
	(0.0028 ~ 0.0411)	(0.0006 ~ 0.0090)
Wildfire_zone3	0.0048***	0.0007
	(0.0007 ~ 0.0088)	(0.0001 ~ 0.0014)

Numbers in parentheses are 95% confidence intervals.
Significance levels are 1% (***), 5% (**), and 10% (*)

To investigate the effect of variation in wildfire activities on PM2.5 concentrations in SLC, we use a multi-step simulation approach, where monthly wildfire acreage burned is considered a random variable:

1. Generate random wildfires in each zone based on the historical spatial distribution, timing, and size of wildfires. Intertemporal correlation among months is considered in the random draws.
2. Calculate the effect of wildfire acreage on the PM2.5 concentration using the model coefficients reported in Table 9.3. All variables other than wildfire acreage also are generated randomly based on historical data.
3. Find the difference between the PM2.5 concentration "with wildfire" (calculated in step 2) and the predicted PM2.5 concentration assuming zero wildfires ("base") in that month.

4. Repeat the steps one through three 1,000 times to generate an empirical distribution of wildfire effects on PM2.5 concentration and differences between "with wildfire" and "base".

Table 9.5 presents the simulation results with PM2.5 concentration of each case and (average) differences between "base" and "with fire" in PM2.5 concentration in a fire season (May to September). PM2.5 concentrations with fire in May and September are not much different from base cases, however, PM2.5 concentration during June to August show some differences ranging between 1.40 and 1.75 μg/m^3. In short, estimates in Table 9.5 indicate that PM2.5 concentration in SLC area is about 1.62 μg/m^3 higher on average during June, July and August due to wildfire activities within a 200-mile radius to SLC, which is roughly 28% higher than "base".

Table 9.5: Wildfire Effects on PM2.5 Concentration (μg/m^3) in Salt Lake City

	Average PM2.5 Concentration (μg/m^3)			
	Base	With Fire	Average Difference	% change
May	5.46	5.55	0.09	1.6%
June	4.79	6.18	1.40	29.2%
July	6.77	8.50	1.72	25.5%
August	6.04	7.79	1.75	28.9%
September	7.15	7.45	0.30	4.2%

Based on 1000 simulation results with random wildfire activities in each zone, random average temperature, total precipitation and average wind speed with estimates in Table 9.3.

Health Impact

As discussed in the introduction, the evidence of a direct relationship between poor air quality and human health problems, especially PM2.5, is quite strong. To accurately assess the medical costs of elevated PM2.5 concentrations from wildfire we would need to know daily hospital admissions and medical expenditure data, and then correlate those data with fire-related PM2.5 concentrations. Such data were unavailable in the short time frame available for this project. Instead, we assess the economic costs of elevated PM2.5 concentration using benefit transfer (BT). BT is the use of studies in the existing literature that share similar features, i.e., similar wildfires, a focus on PM2.5, and comparable health impact studies, but may have been conducted in other regions or at another time. BT transfers estimates from the "study site" to a policy site for which little or no data exist (Rosenberger and Loomis, 2001). In our case, Salt Lake City is the policy site. While BT is considered by economists to be a second-best strategy, it can be very useful in informing decisions at the site of interest. While the literature on the economic cost of adverse health effects from wildfire-smoke exposure is quite sparse (Kochi et al. 2010), we are fortunate to have available a recent study conducted at a study site that is broadly comparable to Salt Lake City (Moeltner et al. 2013).

Moeltner et al. (2013) estimated health impacts of wildfires by linking the number of hospital admissions for respiratory and cardiovascular problems to increased PM2.5 concentrations in the Reno/Sparks area of Nevada. The coefficient of PM2.5 for the respiratory admissions model (Table 6, p. 488 in Moeltner et al., 2013) is estimated to be 0.003 (standard error 0.001) which implies that expected respiratory hospital admissions increase by 0.3% for every 1 $\mu g/m^3$ increase in PM2.5 concentration. Similarly, the coefficient of PM2.5 for the cardiovascular admissions model (Table 7, p. 489 in Moeltner et al., 2013) is estimated to be 0.005 (standard error 0.001) which implies that expected cardiovascular hospital admissions increased by 0.5% for every each PM2.5 concentration increase of 1 $\mu g/m^3$.

As shown in Table 9.6, during June, July, and August, the elevated PM2.5 concentration due to wildfire causes additional 0.49% respiratory patients' hospital admissions on average and 0.81% cardiovascular patients' hospital admissions with every 100 acres of wildfire burn. Obviously, the number of admissions can add up to substantial amounts in an intense fire season such as 2012. Adjusted to $2015 by a medical care price index, Moeltner et al. estimate the treatment costs of $3,202 ($2015) for respiratory patients and $2,060 for cardiovascular patients.

Table 9.6: Changes in Patients Admissions and Treatment Costs per 100 Acres Burned

	Admissions (% increase in patients)	
	Respiratory patients	Cardiovascular
May	0.03	0.04
June	0.42	0.70
July	0.52	0.86
August	0.52	0.87
September	0.09	0.15

Note that additional medical treatment expenditures may not be the true economic costs of wildfire smoke health impact because they do not fully capture the disutility of illness (Richardson, Loomis and Champ, 2013) and do not include opportunity costs of time spent visiting hospitals or medical care, and the value of lost wages due to time spent sick (Jones et al., 2016). Willingness to pay (WTP) to avoid a wildfire smoke health impact is the way to estimate the true costs. Richardson, Champ, and Loomis (2012) found a WTP of $93 per (exposed) person per day for a large wildfire in southern California. Jones et al. (2016) also estimated WTP in the case of Wallow mega fire in Albuquerque, New Mexico using a survey data and found a WTP of $131 per person per day, which is larger than the WTP from Richardson, Champ, and Loomis (2012).

Summary

This chapter has quantified the effect of wildfire on PM25 concentration in the Salt Lake City metropolitan area using monthly data from July 2004 to October 2015. We empirically linked wildfire activities (measured as monthly acres burned within a 200-mile radius which is subdivided into three zones with distance) to monthly average PM25 concentration monitored in SLC.

Results show that wildfire activities have positive and statistically significant effects on PM25 concentration. We find that on average, wildfires increase PM25 concentration by 1.62 $\mu g/m^3$ during June, July, and August. It is associated with 0.5% increases in respiratory-related medical admissions and a 0.8% increase in cardiovascular admissions.

References

Burty, D., D. Bercer, J. Prestemon, J. Pye, and T. Holmes. 2001. What is the price of catastrophic wildfire? *Journal of Forestry* 99: 9-17.

Chay, K.Y. and M. Greenstone. 2003. The impact of air pollution and infant mortality: evidence from geographic variation in pollution shocks induced by a recession. *The Quarterly Journal of Economics* 118(3): 1121-1167.

Chen, L., W. Yang, B.L. Jennison, and S.T. Omaye. 2000. "Air Particulate Pollution and Hospital Admissions for Chronic Obstructive Pulmonary Disease in Reno, Nevada." *Inhalation Toxicology* 12(4): 281-298.

Dockery, D.W., J. Schwartz, and J.D. Spengler. 1992. "Air Pollution and Daily Mortality: Associations with Particulates and Acid Aerosols." *Environmental Research* 59(2):362-373.

Emmanuel, S. 2000. "Impact to Lung Health of Haze from Forest Fires: The Singapore Experience." *Respirology* 5: 175-182.

Jayachandran, S. 2009. "Air quality and Early-life Mortality." *Journal of Human Resources* 44:916-954.

Kochi, I., G. Donovan, P. Champ, and J. Loomis. 2010. "The Economic Cost of Adverse Health Effects from Wildfire Smoke Exposure: A Review." *International Journal of Wildland Fire* 19: 803-817.

Moeltner, K., M-K. Kim, E. Zhu, and W. Yang. 2013. "Wildfire Smoke and Health Impacts: A Closer Look at Fire Attributes and Their Marginal Effects." *Journal of Environmental Economics and Management* 66: 476-496.

Moore, D., R. Copes, R. Fisk, R. Joy, K. Chan, and M. Brauer. 2006. "Population Health Effects of Air Quality Changes Due to Forest Fires in British Columbia in 2003: Estimates from Physician-visit Billing Data." *Canadian Journal of Public Health* 97(2): 105-108.

Mustain, A. 2012. "Smoke from Western Wildfires Reaches Atlantic Ocean." Live Science, available at http://www.livescience.com/21358-wildfire-smoke-map.html; Accessed January 7, 2017.

Newey, W.K., and K.D. West. 1987. "A Simple, Positive Semi-definite, Heteroskedasticity and Autocorrelation Consistent Covariance Matrix." *Econometrica* 55: 703–708.

Pope, C.A. 2000. "Epidemiology of Fine Particulate Air Pollution and Human Health: Biologic Mechanisms and Who's at Risk?" *Environmental Health Perspectives* 108 (Suppl 4):713-723.

Pope, C.A., R.T. Burnett, and M.J. Thun. 2002. "Lung Cancer, Cardiopulmonary Mortality, and Long-term Exposure to Fine Particulate Air Pollution." *The Journal of the American Medical Association* 287 (9):1132-1141.

Richardson, L.A., P.A. Champ, and J.B. Loomis. 2012. "The Hidden Cost of Wildfires: Economic Valuation of Health Effects of Wildfire Smoke Exposure in Southern California." *Journal of Forest Economics* 18(1): 14-35.

Richardson, L.A., J.B. Loomis, and P.A. Champ. 2013. "Valuing Morbidity from Wildfire Smoke Exposure: A Comparison of Revealed and Stated Preference Techniques." *Land Economics* 89(1): 76-100.

Rosenberger, R.S., and Loomis, J.B. 2001. Benefit transfer of outdoor recreation use values: A technical document supporting the Forest Service Strategic Plan (2000 revision). Gen. Tech. Rep. RMRS-GTR-72. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Tham, R., B. Erbas, M. Akram, M. Dennekamp, and M.J. Abramson. 2009. "The Impact of Smoke on Respiratory Hospital Outcomes during the 2002–2003 Bushfire Season, Victoria, Australia." *Respirology* 14(1):69-75.

Sastry, N. 2002. Forest fires, air pollution, and mortality in Southeast Asia. Demography 39(1):1-23.

U.S. Environmental Protection Agency (U.S. EPA). 2003. Particle Pollution and Your Health, Available at https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1001EX6.txt , accessed January 6, 2017.

U.S. Environmental Protection Agency (U.S. EPA). 2009. Integrated Science Assessment for Particulate Matter (Final Report). U.S. Environmental Protection Agency, Washington, DC, EPA/600/R-08/139F, 2009. Available at https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?deid=216546

Viswanathan, S., L. Eria, N. Diunugala, J. Johnson, and C. McClean. 2006. An Analysis of Effects of San Diego Wildfire on Ambient Air Quality. Journal of the Air and Waste Management Association 56(19):56-67.

Chapter 10: Fire Suppression and Fuels Reduction Efforts

Paul M. Jakus
Department of Applied Economics and the Center for Society, Economy and the Environment
Utah State University

Introduction

Numerous research papers and government reports have reported that, in the last three decades, wildfires have become larger in extent and more severe in their ecological impacts. Further, the wildfire season has gotten longer, with fires beginning earlier in the year and extending past the historical conclusion of the fire season. The causes have been numerous. A century of a fire management policy that focused on wildfire exclusion has upset natural fire regimes that govern ecoregions (Gorte 2013). Elimination of wildfire contributed to the buildup of unnaturally high levels of biomass ranging from fine materials such as grass and needles, to more coarse forms of biomass such as seedlings or small trees. Commercial logging often removed the economically valuable pines and larger, more fire resistant trees and left more fire sensitive species standing. Management policies that have reduced logging and grazing on public forests and rangelands has contributed to increased quantities of small diameter biomass—in some cases, public lands have become overgrown with biomass.

Livestock grazing has different effects on wildfire depending on the nature of the grazing region. In forests characterized by frequent, low intensity fires, grazing can lead to the removal of fine grasses that helped maintain the ecologically appropriate fire regime. Low intensity wildfire does not return as per its standard regime, leading to buildup of coarse biomass and fires of greater intensity. In contrast, grazing in rangelands can have an inhibitory effect on the spread of wildfire. In a study of Great Basin rangeland infested by the invasive and highly flammable cheatgrass, Diamond et al. (2009) found that targeted cattle grazing reduced the biomass of cheatgrass, thus reducing flame length and spread of wildfire. Davies et al. (2015) examined winter grazing in big sagebrush shrub-grasslands of southeastern Oregon, finding that grazing reduced fine fuels and increased fuel moisture, both of which lowered flame height and slowed fire spread. Strand et al. (2014) also note the combined effect of grazing with weather, topography, and overall vegetation composition.

Finally, changes in climate have contributed significantly to the length of the fire season and severity of wildfire. Increasing temperatures and shifting precipitation patterns are changing key factors that influence the natural fire cycle (Westerling et al. (2006) and Runner (2006)). Warmer spring and summer temperatures associated with climate change have led to earlier snow melt, which decreases soil and vegetative moisture earlier than in decades past. In the future, the mix of precipitation is expected to change to more rainfall and less snowfall, further exacerbating the problem of drier soils and vegetation in the summer and fall. Already, Westerling et al. (2006) report, the length of the average fire season has been extended by 78 days. Recent research by Abatzoglou and Williams (2016) has found that anthropogenic sources of climate change have increased aridity of forest resources to such an extent that it has contributed an additional 10.4 million acres of wildfire from 1984 through 2015.

Forest Resources in Utah

The United States Forest Service recently completed an inventory of Utah forest resources, using field data to evaluate changes in the status of forested land in Utah between 2003 and 2012 (Werstak et al. 2016). Preliminary indications from the Monitoring Trends in Burn Severity project suggest that in recent years the most severe fires have occurred on USFS administered land (Chapter 3), so the forest inventory report is quite timely, as it demonstrates that many of the factors that contribute to the lengthening fire season and possible increasing severity of fire are present in Utah's forests.

Net Annual Growth in Utah Forests

Werstak et al. (2016) measure forest change by examining net annual growth, defined as the difference between annual growth in tree volume minus the volume lost through natural causes of tree death. Natural causes include death from insects, disease, suppression by overstory, and advanced tree age, as well as sudden deaths due to epidemic outbreaks of insects or disease, wildfire, or extreme weather events such as hurricanes, tornados, or ice storms. Tree removals due to logging are treated separately from mortality.

Annual gross growth of trees ≥5 inches diameter at breast height (dbh) is estimated at just over 207 million cubic feet (ft^3). Annual mortality of trees ≥5 inches dbh was just under 257 million ft^3, for net annual growth of −54 million ft^3. This figure does not reflect major differences in annual growth by land status. Reserved USFS lands—those lands protected from wood products utilization through statute or administrative designation—have a higher per acre mortality rate than unreserved lands (about 48 ft^3 $acre^{-1}$ vs. 28 ft^3 $acre^{-1}$). Per acre mortality on both reserved and unreserved lands administered by other public agencies or by private individuals had mortality of less than 10 ft^3 $acre^{-1}$.

The leading causes of mortality in Utah forests were insects, disease, and fire. Insects are, by far, the leading cause of tree mortality on both reserved and unreserved lands. Fire is the second leading cause of mortality on reserved lands (about 3.5 ft^3 $acre^{-1}$) and the third leading cause on unreserved lands (just over 2 ft^3 $acre^{-1}$). Regardless of cause of death, dead trees left standing in the in the forest will increase the fuel load available for fire unless they are removed.

Removals (Harvest) in Utah Forests

Tree removals for wood utilization are also reported by Werstak et al. (2016). The authors note that removals could be of either live or dead trees. The most recent removal data reported in the study was for 2007, when just over 16 million ft^3 of timber was removed, of which some 1.2 million ft^3 was left in the forest as slash. Over 70% of removals taken from the forest (14.9 million ft^3) was from non-growing forest resources, almost half of which was destined to be used as fuelwood.

Sorenson et al. (2016) provide the most up-to-date summary of trends in Utah's timber harvest and timber industry. Some 75% of Utah's 3.7 million acres of forested land is administered by the US Forest Service, with the remainder administered by private and tribal authorities (16%) or other federal and state agencies (9%). Measured in thousand board feet (MBF), harvest from Utah's forests has fallen by 70% over 20 years, from 64,674 MBF in 1992 to 19,356 MBF in 2012. Harvest from all ownerships (private, tribal, and public) fell, but the largest drop in harvest was from national forests, which decreased by 80% (from 49,989 MBF in 1992 to 10,117 MBF in 2012).[19]

Stand Density in Utah's Forests

Decreased removals may be correlated with increased stand density; a stand density index (SDI) measures the degree of crowding in a forested area and is based on "...the quadratic mean diameter of trees and the number of trees per acre (Werstak et al. 2016, p. 38)." Werstak et al. report that Utah's forests are "well-stocked" (p. 39) given the SDI measurements for various forest types, noting that 54% of Utah's forests are fully occupied. However, some 21% of Utah's forests are "...overstocked, meaning that self-thinning mortality is imminent or currently occurring (p.39)." Overstocked forests are subject to a variety of ills: first, increased mortality could occur because the competitive stresses of dense forest stands can make the forest more susceptible to insects and disease damage. Second, greater tree density puts a stand at greater risk of catastrophic fire.

The Costs of Wildfire Suppression

Past forest and range management policies, including fire exclusion, invasive species management, livestock stocking rates, and forest removals, have resulted in a buildup of fuels in many firesheds. At the same time a changing climate has lengthened the fire season and affected the form and timing of precipitation such that soil and vegetative moisture dissipates more quickly. Despite the fact that the vast majority of wildfires are suppressed quickly and successfully (or burn out on their own), all of these factors have combined to increase the frequency and extent of very large wildfires. A concomitant outcome of these larger wildfires is an increase in suppression costs, particularly since the turn of the millennium.

---

[19] As one might expect, a harvest decline of 67% would likely decrease the number of mills needed to process this timber. Indeed, the number of wood processing manufacturers in Utah fell from 51 to 18. Only 58% of Utah's timber harvest was processed by in-state mills (Sorenson 2016).

National Suppression Costs

National acreage burned and federal wildland fire suppression costs from 1985 through 2015 for USFS and U.S. Department of the Interior agencies (primarily BLM, NPS, and BIA) were obtained from the National Interagency Fire Center (NIFC, 2016). Cost data were adjusted for inflation to constant $2015 using the Government Expenditures Price index obtained from the Federal Reserve Economic Data website of the St. Louis Fed. Table 10.1 shows standard descriptive statistics for the data. The coefficient of variation (CV) is the ratio of the standard deviation of each variable to its mean and allows for comparison of variation across different measures. The relatively small value of CV for the number of fires suggests that while we observe year to year variation, the number of fires across the full 31-year time period is not that volatile. In contrast, the CVs for total acreage burned and suppression costs show substantial volatility: in some years the nation experiences a large (small) amount of acreage burned along with high (low) suppression costs.[20]

Table 10.1: Wildland Fire Suppression Costs, USFS and Department of Interior, 1985-2015

	# Fires	Total Acres Burned	Suppression Costs ($2015 million)	Cost per Acre ($2015)
Mean	75,022	5,062,539	$1,293.7	$290.84
Min	47,579	1,329,704	$425.2	$123.07
Max	96,385	10,125,149	$2,511.5	$613.26
Standard Deviation	12,357	2,729,957	$595.9	$129.61
Coefficient of Variation	16.5%	53.9%	46.1%	44.6%

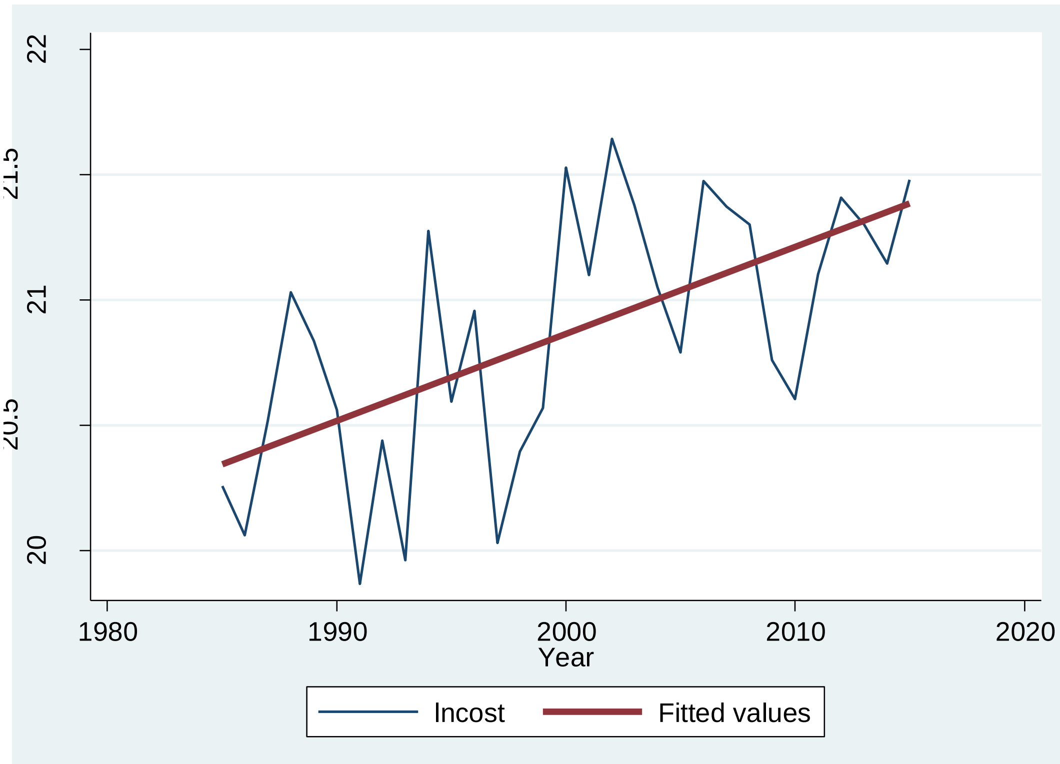
Suppression costs in constant $2015. Source: NIFC.gov

Figure 10.1 depicts suppression expenditures over time, and the clear upward trend in suppression costs might be indicative of a non-stationary data-generating process.[21] Indeed, the simple correlation between suppression costs (in constant $2015) and year is 0.60. A non-stationary process would mean that our year-to-year measures of suppression costs are not independently generated, yielding a host of modeling complications. We use the augmented Dickey-Fuller test to examine whether the time series follows a stationary process. Multiple versions of the augmented test—without and without a trend, and using one, two or three time lags in suppression costs—all yielded test statistics that reject the null-hypothesis of a unit-root in favor of the alternative hypothesis that the time series is stationary.

[20] Wildfire expenditures began to accelerate after 2000, but costly fire seasons occurred in previous years as well: after adjusting for inflation to constant $2015 the cost of suppression was over $1 billion in 1988, 1989, 1994, and 1996.

[21] Stationary processes have a constant mean and variance over the time series.

Figure 10.1: Time Trend in National Suppression Expenditures, 1985-2015



Note: On the Y-axis, ln(21) is equivalent to $1.32 billion in suppression expenditures.

Regression analysis can be used to examine the relationship between acreage burned and general time trend of expenditures. Results for national suppression expenditures appear in Table 9.2, where Models #1 through #3 are based on the natural log of suppression costs and the natural log of acres burned. The log-log specification allows the regression coefficient on acres burned to be interpreted as an elasticity. Time-series data may be serially correlated, but a Breusch-Godfrey test ($\chi^2 = 0.59$) failed to reject the null hypothesis of no serial correlation. Autoregressive specifications were also tested; results again favored a simple ordinary least squares approach.

The models reported in Table 10.2 show a strong relationship between acreage burned and suppression costs. Model #1—a comparison of suppression costs to acres burned in any given year—is depicted in Figure 10.2. Using model #2 as the preferred specification because it includes a time trend, the coefficient of acreage burned suggests that, at the national level, a 1% increase in acres burned results in a 0.44% increase in suppression costs. The coefficient of the year variable can be interpreted as a growth rate (the functional form for this variable is semi-log). That is, the model predicts growth in wildfire suppression costs at 1.8% per year (as depicted in Figure 9.1). Robust regression resulted in nearly identical parameter estimates as the weighting procedure identified no heavily influential observations (Model #3).[22]

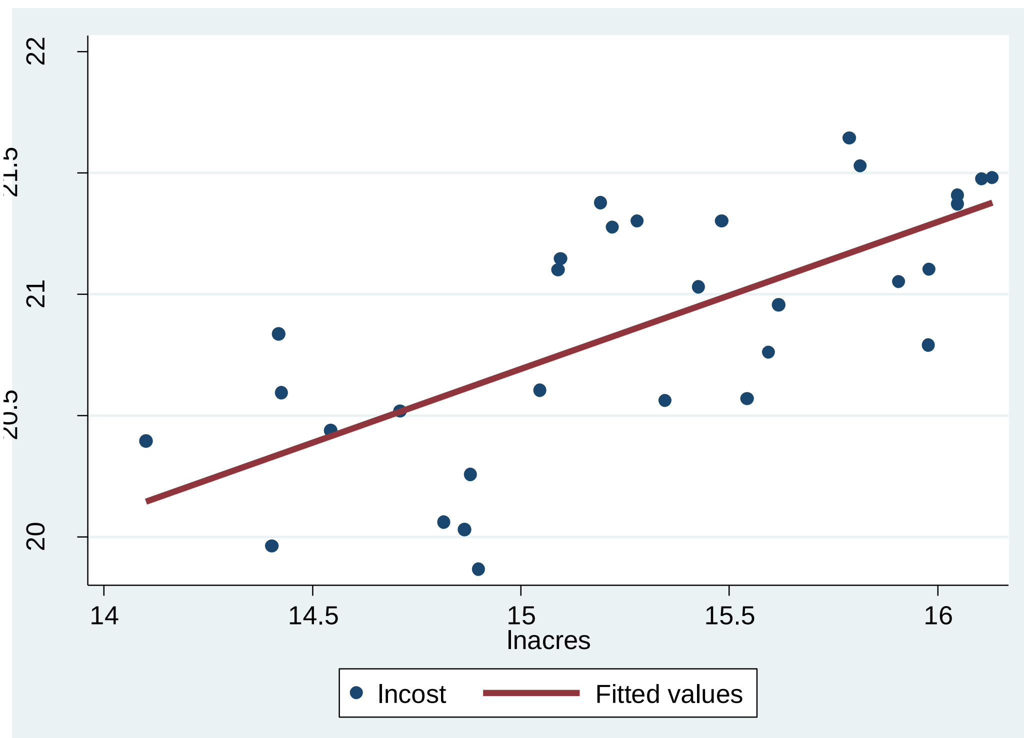
[22] No observation had a weight less than 0.75.

Table 10.2: Regression of Federal Suppression Costs by USFS and DOI, 1985-2015

Variable	Model #1 (OLS)	Model #2 (OLS)	Model #3 (Robust weighted)	Model #4 (OLS)
	Ln(Cost)	*Ln(Cost)*	*Ln(Cost)*	*Cost per Acre*
Fire size, ln(acres) or acres (#4)	0.607 (0.001)	0.441 (0.001)	0.432 (0.008)	−3.53×10-5 (0.001)
Year		0.018 (0.017)	0.018 (0.076)	5.151 (0.030)
Intercept	11.585 (0.001)	−21.452 (0.115)	-21.722 (0.244)	−9,833.148 (0.037)
Adjusted R^2	0.481	0.513		0.308
F-statistic			13.77 (0.001)	

Costs adjusted to constant $2015. P-values in parentheses, based on robust standard errors

Figure 10.2: Federal Suppression Costs by USFS and DOI, 1985-2015 (Model #1)



Note: On the X-axis, ln(15) is equivalent to 3.27 million acres burned. On the Y-axis, ln(21) is equivalent to $1.32 billion in suppression expenditures.
Source: NIFC.gov

The final column in Table 10.2 (Model #4) relates per acre suppression costs to annual acreage burned and a time trend. Per acre costs fall as acreage burned increases; evaluated at the mean cost per acre and annual acreage burned, a 1% increase in acreage burned reduces per acre costs by 0.61%. This result accords well with economic intuition, as large fires, which make up the bulk of acreage burned, require the commitment of fixed fire suppression resources (trucks, aircraft, etc.) and an increase in fire size will add only marginally to the total cost of suppression. The time trend in Model #4 indicates that per acre costs are, on average, increasing at just over $5.15 per year.

Suppression Cost Literature

The models presented in Table 10.2 are consistent with the growing literature that has examined wildfire suppression costs. Much of the empirical literature examines suppression cost using wildfires or wildfire regions as observational units, as opposed to the national-level model presented above. This allows researchers to explore a wider set of explanatory variables in a modeling effort. In their review of the literature, Ellison, Mosely and Bixler (2015) identified three broad categories of variables that have been linked to suppression expenditures. First, physical and environmental factors include measures such as total acres burned, topography, climate, and vegetative characteristics of the area in which the burn occurred. Socio-environmental factors generally focus on value-at-risk, such as human lives, structures, and infrastructure, which often receive the highest priority for suppression efforts (Ellison et al. 2015, p. 6). Finally, suppression efforts (and, hence, costs) are driven by management decisions. In addition to the previously mentioned variables, suppression decisions can be influenced by political pressures or media coverage of a fire.

In an analysis of 33 years of national fire suppression expenditure data, Calkin et al. (2005) found year-to-year acreage burned and expenditures to be increasingly erratic beginning in 1987. Although suppression costs were increasing with acreage burned, the cost per acre fell with larger fires (as seen in model #4 in Table 9.2). The authors also note that wildfires increase in a region when a dry year (the one in which fires occur) had been preceded by a relatively wet year.

Gebert, Calkin, and Yoder (2007) provide a prototypical example of a econometric suppression cost model. The authors linked suppression expenditures for 1,550 fires to detailed characteristics of the fire (slope, aspect, elevation, vegetation, etc.), values at risk, and factors affecting management. Per acre costs were negatively related to the fire size and positively related to slope. Though the distance to the nearest town was statistically insignificant, suppression expenditures increased as the value of housing stock within a 20 mile radius of a fire's origin increased.

In a major effort to model suppression costs, Hand et al. (2014) examined nearly 5,700 fires occurring from 2006 through 2012 in eight Forest Service regions. Costs per acre were found to be negatively related to fire size, distance to a wilderness boundary, and if the fire were managed by a DOI agency. Costs per acre were positively related to the energy release component (a measure of potential fire intensity), population within 20 miles of the fire, and vegetative types such as brush and timber. Numerous versions of the suppression cost model were estimated, each of which accounted for about 40% of the variation in cost per acre.

Fire size is related to the duration of a fire (days from ignition to control), and fire duration is related to suppression efforts. Thus, fire size and duration is akin to a chicken-and-egg problem; technically, the variables are endogenous. This question is addressed by Hand, Thompson, and Calkin (2016). Using a sample of 712 fires that exceeded 300 acres in size, the authors find expenditures are correlated with fire size (ϱ=0.56) and duration (ϱ=0.17). Fire duration and fire size were also significantly correlated (ϱ=0.34). After accounting for endogeneity of duration and size on a suppression expenditure model, the authors find that the elasticity of expenditures with respect to fire size increased from 0.69 to 0.94, i.e., a 1% increase in fire size leads to a 0.94% increase in expenditures. Thus, suppression expenditures appear to rise at a much faster rate in the endogenous model than in the authors' exogenous model (0.69%) or the simple model reported in Table 9.2 (0.61% in model #1 and 0.44% in Model #2).

The value of housing stock or the population within a zone near a fire's origin is related to an issue raised by Gorte (2013) in his evaluation of rising costs of wildfire. The wildland urban interface (WUI) is defined as an area where expanding populations and the associated demand for primary (or second) homes begin to encroach on land that is highly susceptible to fire. Technically, WUIs are those places where the housing density is greater than one structure per 40 acres and more than 50% covered in wildland vegetation, or the same density of structure with less than 50% cover, but located within 1.5 miles of an area greater than 3 square miles that is more than 75% covered in wildland vegetation (Martinuzzi et al. 2010). Although only 9.9% of the U.S. geographic area is defined as WUIs, These areas host approximately one-third of the nation's housing stock and population.[23] Gorte (2013) reports that saving human lives is the top priority of wildfire suppression efforts, whereas protection of property and natural resources are to be treated as co-equal second priorities. In reality, protecting structures (homes) receives priority over protecting land and other natural resource. Protection of structures may require specialized fire-fighting assets due to, for instance, the need for aerial retardant drops to slow the rate of fire spread, thus raising the cost of suppression. Gude et al. (2013) report additional empirical evidence for California fires; for every 1% increase in the number of homes located within 6 miles of an active fire, the daily suppression costs increase by 0.08%. Hand et al. (2016) use multiple perimeters (within 5 miles, 5-10 miles, and 10-20 miles); in each zone a 1% increase in housing stock increases suppression expenditures by roughly 0.03%.

Donovan et al. (2011) focus on the effect of media coverage and political pressure in affecting wildfire suppression costs. Media effects are measured by the number of stories—including front page stories—and words written about a fire in newspapers located in towns of various sizes (from populations of 30,000 to over 250,000). Political influence was gauged by the number of years in office of the appropriate members of the region's congressional delegation, as well as each delegate's membership on important fire-related congressional committees. The authors find that, in addition to the standard biophysical measures that influence suppression costs, the length of tenure in Congress by the local representative resulted in increased per-acre sup-

[23] In Utah, some 48% of houses and 45% of the population are located in a WUI (Martinuzzi et al. 2010, p. 15 and p. 19).

pression costs by $22 ($2015). A story appearing in a newspaper in a town of over 250,000 people increased per-acre suppression costs by $2,521 ($2015).

Suppression Cost Modeling for Utah Wildfires

Incident status summaries (ICS 209 reports) from 1999 through 2016 were used to analyze suppression costs for wildland fires in Utah. ICS 209 reports are required for large fires, but the reporting requirements are such that many small fires are also included.[24] Of the 918 fires on the initial list, we chose to focus on large fires (≥100 acres) whose ignition origin and final cost estimates were reported.

Initial analysis suggested that some observations were subject to measurement error. After adjusting costs to constant $2015, per acre suppression costs ranged from $0.79 per acre to $13,495 per acre. Total suppression costs ranged from $522 (a nominal cost of $500 was reported for a few fires), to over $2.6 million. Robust regression was used to identify heavily influential observations; six observations (all less than 10,000 acres) were eventually deleted, including the fires providing the extreme values for per acre costs. Table 10.3 provides the descriptive statistics for the final dataset of 450 fires.

Given our selection criteria for the dataset, the mean fire size was 4,840 acres. Some 210 fires (46.7%) were between 100 and 1000 acres, and another 199 fires (44.2%) were between 1000 and 10,000 acres in size. Very large fires comprise only a small portion of the dataset: 30 fires (6.7%) were between 10,000 and 40,000 acres, and 11 fires (2.4%) were in excess of 40,000 acres.[25] Suppression costs averaged over $825,000 per fire, with a mean cost per acre of nearly $575. As is common with cross-sectional data, the coefficients of variation are quite high (relative to the time series data presented for national fire suppression costs in Table 9.1).

Table 10.3: Wildland Fire Suppression Costs, Utah ICS 209 Fires, 1999-2016 (n=450 fires)

	Total Acres Burned	Suppression Costs ($2015)	Cost per Acre ($2015
Mean	4,840	$826,063	$573.81
Min	101	$3,108	$2.20
Max	363,052	$23,600,000	$10,651.72
Standard Deviation	19,506	$1,810,096	$1,066.51
Coefficient of Variation	403.0%	219.1%	185.9%

24 We thank Kara Stringer of the USFS for providing the ICS 209 data. The 209 reporting requirements can be downloaded at https://www.predictiveservices.nifc.gov/intelligence/ICS-209%20When%20to%20Report%20Wildland%20Fire%20Incidents.pdf

25 The vast majority of wildfires are less than 100 acres but large fires make up bulk of acreage burned. Our cost analysis is restricted to the largest fires occurring in Utah.

Total Suppression Cost Models for Utah

Total suppression cost models for Utah wildfires show that costs increase with fire size and rugged topography (Table 10.4). P-values are based upon robust standard errors that are adjusted to reflect heteroscedasticity (changing variance) inherent in cross-sectional data. The log-log specification again allows a simple interpretation of the ln(acres) coefficient: depending on the specification, a 1% increase in fire size results in an increase in suppression costs by somewhere between 0.51% and 0.55%, estimates that are comparable to those reported in the literature. Figure 9.3 depicts the relationship between fire size and suppression costs (Model #1 of Table 9.4).

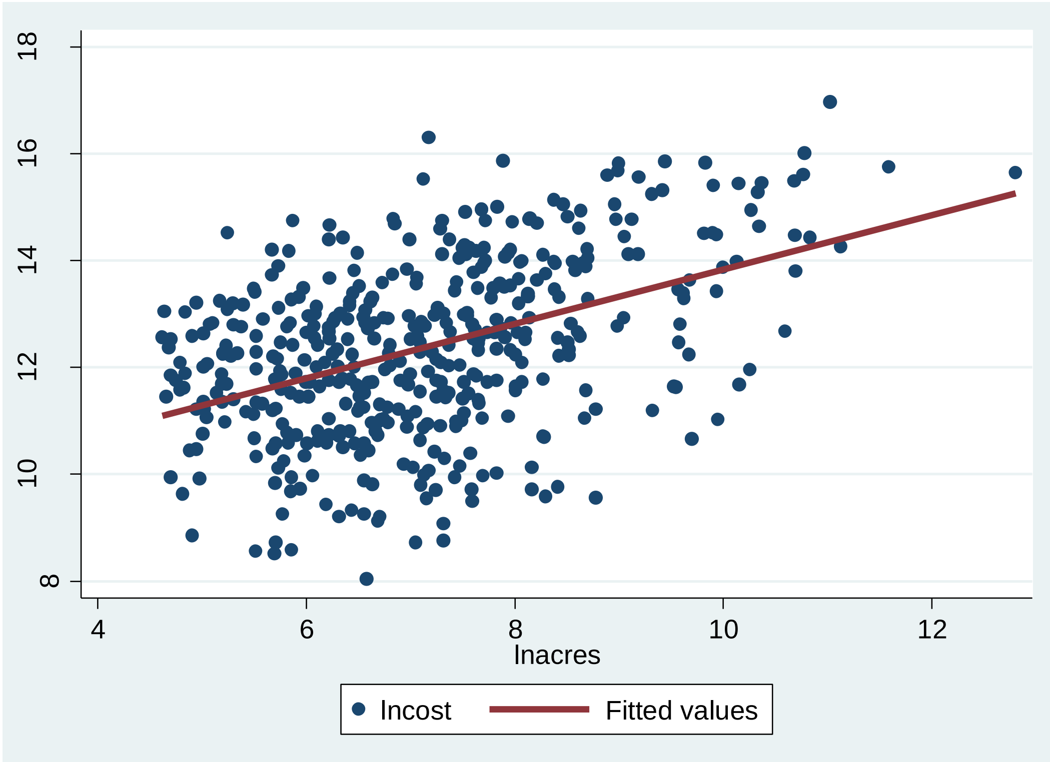
Table 10.4: Total Suppression Cost Models, Utah, 1999-2016 (n=450)

	Model #1	Model #2	Model #3
Fire size, ln(acres)	0.509 (0.001)	0.540 (0.001)	0.546 (0.001)
Topographic Index		0.420 (0.001)	0.421 (0.001)
Year			0.0156 (0.246)
Intercept	8.749 (0.001)	−34.218 (0.001)	−65.664 (0.020)
Adjusted R^2	0.202	0.280	0.281

Dependent Variable: ln(Suppression Cost). P-values in parentheses are based on robust standard errors

The topographic index was developed based on 8100 square meter pixels in the county of each fire's origin. Pixels with large changes in elevation have larger flattened areas than pixels with little change in elevation. The ratio of the flattened pixel area to the original area was calculated for all pixels in a county, with the mean value of all pixel ratios in a county used as a measure of topographic variability. Box Elder county exhibits the least topographic variability (index=100.7) whereas Carbon county is the most rugged (index=104.4). Based on the national suppression cost models, we would expect costs to rise as terrain in a county becomes more rugged and more challenging. Indeed, Models #2 and #3 show that suppression costs increase for fires originating in more topographically variable counties; evaluated at mean logged suppression cost (ln[12.383]=$238,600) an additional unit of topographic variation in the county of origin adds almost $125,000 to suppression costs. Finally, Model #3 finds no discernable time trend in total suppression costs. While the coefficient is positive—implying growing suppression costs, all else equal—the P-value of 0.246 means the coefficient is not statistically significant. Overall, the models explain about 28% of the variation in total suppression costs.

Figure 10.3: Wildfire Suppression Costs for 450 Utah Wildfires



Note: On the X-axis, ln(8) is approximately 2,981 acres; on the Y-axis, ln(14) is $1.2 million.

Per Acre Suppression Cost Models for Utah

Models of per acre suppression costs appear in Table 10.5. Costs per acre decline with the size of a fire, a result that is in accord with the national modeling reported by other authors. Evaluated at mean fire size (4,840 acres), an additional 1000 acres in fire size would reduce per acre costs by $5, from $574 to $569. Topographic variability adds considerably to per acres costs. Every one unit increase in topographic variability of the county of origin adds about $204 to per acre costs. Again, although the coefficient of the time trend (year) is positive (indicating rising per acre costs), it is statistically insignificant (Model #3). Our sample of 450 fires does not provide evidence of growing per acre costs over time. Overall, the per acre cost of suppression models explains only about 5% of the total variation in per acre costs.

Table 10.5: Cost per Acre Suppression Costs models, Utah, 1999-2016

	Model #1	Model #2	Model #3
Fire size, Acres	−0.005 (0.064)	−0.005 (0.084)	−0.005 (0.084)
Topographic Index		204.175 (0.001)	204.288 (0.001)
Year			6.387 (0.571
Intercept	598.684 (0.001)	−20,186.2 (0.001)	−33,019.91 (0.128)
Adjusted R^2	0.009	0.049	0.048

Dependent Variable: Cost per Acre P-values in parentheses are based on robust standard errors

Suppression Efforts as Part of Fire Management Strategy

Suppression costs are only part of wildfire management efforts. In addition to controlling and extinguishing fires, land management agencies must also prepare for fire, mitigate the damages of fire, and engage in fuels reduction activities (Hoover and Bracmort, 2015). Preparedness activities attempt to create an environment for safe and cost-effective fire management efforts. These include actions ranging from drills for response teams, to working with populations residing in the WUI to prepare an evacuation plan and create a defensible perimeter around a home. Post-fire mitigation actions focus on rehabilitating fire-damaged landscapes. As seen in both the Prelude and Chapter 6 of this report, much rehabilitation activity must focus on stabilizing soils and restoring normal hydrologic and biologic ecosystem processes. Finally, fire management also includes fuels reduction efforts designed to lower the probability of ignition and severity of fire and return a landscape to its appropriate fire regime. We address fuels reduction later in this chapter, but first we note the effect of volatility in suppression and other wildfire management expenditures on the other functions of agencies tasked with suppressing wildfires.

According to Hoover and Bracmort's (2015) analysis of appropriations for USFS and DOI wildfire management budgets for FY2006 through FY2015, suppression costs ranged from just under 34% of total wildfire management appropriations (FY2006) to almost 64% (FY2008).[26] Further, a USFS report (2015) noted that ex ante appropriations for wildfire suppression were often insufficient in heavy fire years, resulting in diversion of funds from accounts appropriated for other purposes to cover suppression and other wildfire related costs. Over time, wildfire management costs have grown from 16% of the USFS annual budget in FY1995 to 52% of the budget in FY2015. Given the expected growth in wildfire by the year 2025, it is projected that some 67% of the USFS budget will be consumed by wildfire management activities by then.

[26] These figures assume that "additional appropriations" and "FLAME" account appropriations in Table 4 of Hoover and Bracmort (2015) are used for suppression.

The wildfire management budget has grown by not only increasing wildfire appropriations, but also by reducing the budgets for other national forest management programs. Ingalsbee and Raja (2015) note that the rising costs of wildfire management could inexorably alter the mission of the USFS. USFS data do not contradict this contention (USFS 2015). National Forest System funding fell by 32% in real terms between FY1995 and FY2015. This has affected the ability of the USFS to fulfill its obligations. Programs to improve and manage forest vegetation and watersheds have fallen by 24% in real terms since FY2001, while wildlife and fisheries management budgets have fallen by 18% over the same period. Investments in facilities, roads, and deferred maintenance have fallen by 68%, 46%, and 95%, respectively. Land management planning programming funds have fallen by 64%, even as each forest unit is required to regularly update its Land Management Plan. Thus, the costs of wildfire include hidden opportunity costs that extend far beyond the cost of control and suppression.

Fuels Reduction Treatments

Some 54% of Utah's forests are considered fully occupied by biomass; 21% of forests are considered overgrown. As noted in previous sections of this report, the reasons for this are many, but two primary drivers have been (i) past forest management policies focused on preventing all wildfire (fire exclusion) and (ii) reductions in wood utilization (logging). Exclusion of fire led to a buildup of fuel loads, especially surface biomass, small diameter trees, and dead wood. As noted by Agee and Skinner (2005), "Fires that once spread as surface fires [are] now more intense, and capable of jumping into the canopy of forests as crown fires." A voluminous literature examining the cost and efficacy of fuels reduction methods exists. Here we review the key findings of that literature by relying upon a selected group of research papers. Studies have been selected on the basis of applicability to Utah or because they have become highly-cited, foundational investigations.

Though some fuels reduction studies have been conducted on rangelands, the bulk of the literature focuses on fuels treatments in forests. That said, the goal of fuels reduction efforts on both rangelands and forests is the same: to create more fire resilient ecosystems. Agee and Skinner (2005) outline four key principles guiding fuel reduction efforts in forests. First, reducing surface fuels (often the most hazardous fuels according to Stephens et al. (2012)) will reduce potential flame length. Doing so makes fire suppression easier and lowers the probability of initiating a crown fire. Second, treatments should increase the height to the live crown by removing *ladder fuels*, the vegetation that allows a fire to climb from the floor and into the tree canopy—Stephens et al. (2012) characterize ladder fuels as the second most hazardous fuel type. Again, the goal is to reduce the likelihood of starting a crown fire. Third, the crown density should be thinned (decreased), making tree-to-tree crown fires less probable. Fourth, improving the fire resiliency of a forest will require that larger, commercially-valuable, fire-resistant trees be maintained in the forest. In the event of a high-intensity fire, this achieves the goal of reducing the tree mortality rate and helps restore the historic structure of the stand.

Fuels reduction treatments come in many forms. Perhaps the most well-known fuel treatment is *prescribed fire*. Prescribed fire is effective at reducing surface fuels and increasing the height to canopy, but forest managers are constrained by multiple forces (social, economic, and administrative) that limit its use (Stephens et al. 2012). Further, although a prescribed burn may initially reduce surface fuel biomass, it can also increase mortality of smaller trees and, within a decade, the forest may return to the same or greater amount of surface fuels (although with a reduced ability of a fire to reach the crown). Prescribed fire is often used in conjunction with alternative fuel reduction methods.

The alternatives to prescribed fire are collectively known as *fire surrogates* and either reduce biomass or convert biomass into smaller pieces. For example, *thinning* would remove trees that serve as ladder fuel and thus reduce the risk of crown fires (Parker and Bennett 2008). Low thinning, or thinning from below, will leave larger more fire-resistant trees in the stand, creating a better stand structure. However, thinning can increase the amount of surface fuel biomass that, unless removed from the forest, can increase fire risk. A related technique to reduce ladder fuels is *pruning*, in which branches of a tree are removed, rather than an entire tree (Holmberg and Bennett 2008). Pruning also increases surfaces fuels unless the pruned branches are disposed of.

Mechanical treatments (including thinning and pruning) convert brush, smaller trees, and slash into smaller pieces that are either left on the surface or are disposed of by other methods. Grinding, mowing, masticating, or chipping fuels requires a vehicle to which a cutting tool is attached. Mechanical treatments are often used in conjunction with prescribed fire. In addition to reducing ladder fuels, the chipped biomass can create a dense bed of surface fuel. In some ecosystems the bed may be left in place if the dense layer of biomass limits oxygen availability and thus reduces the probability of ignition or slows the spread of fire (Bennett and Fitzgerald 2008).

On rangelands *targeted grazing* may be used to achieve fuels management objectives. In addition to the studies cited in the introduction to this chapter (Diamond et al. 2009; Stand et al. 2014; and Davies et al. 2015), range scientists are identifying the conditions under which grazing can alter vegetation such that the natural fire regime can return to rangelands. For example, a key problem in the Great Basin range is cheatgrass, which increases a fire's rate of spread. Schmelzer et al. (2014) conducted pasture-scale experiments suggest that fall grazing can reduce cheatgrass biomass and the amount of cheatgrass seed, as well as reducing the quantity of fine fuels that contribute to ignition and spread of fire.

This discussion of fuels management should make it clear that the goal of fuels reduction is not the elimination of fire from the landscape. Reinhardt et al. (2008) assert that the key objective of vegetative management is "...creating conditions in which fire can occur without devastating consequences...". Thus, an effort to reduce fuels will not necessarily make suppression easier, lower suppression costs, slow the rate of spread, or restore ecosystem health; these outcomes may, of course, occur as a co-benefit of fuels management, but they are not the primary goal.

Fuels Reduction Costs

Rummer (2008) provides an excellent overview of the factors that influence the costs of alternative fuels reduction methods, distinguishing between *in situ* techniques and methods of biomass removal. *In situ* fuel reduction practices, such as prescribed fire and mastication, directly consume vegetation or accelerate its decay with biomass remaining in place. Such methods are useful when it is not economical to market the biomass to be removed. Removal approaches (e.g., thinning) can be sorted into two types, either *removal and disposal* or *removal and utilization*. A disposal approach is used when the biomass cannot remain because of fire risk but is not marketable; a utilization approach is used if the biomass is merchantable. The production of merchantable biomass as part of a fuels treatment program will generate revenues that help to lower the net cost of treatment. Choice of a fuels treatment approach will also depend on the goal of the vegetative management effort, i.e., to remove surface fuels, ladder fuels, etc. (Reinhardt et al. 2008; Stephens et al. 2012).

The per acre costs of any treatment method are composed of fixed and variable costs (Rummer 2008). Fixed costs include depreciation and insurance on machinery, as well as the costs of project planning, permitting, and move-in operations. Variable costs include labor, fuel, repair and maintenance, and disposal of slash. Rummer (2008) notes that per acre costs are driven largely by fixed costs, which are spread over the entire area to be treated. Hence, one should expect declining average total per acre costs as the area of the treated region increases.

Representative treatment costs in the published literature are presented in Table 10.6. Two things are immediately evident. First, the range in per acre costs for similar treatments on nominally similar vegetation is quite high. Second, prescribed fire is less expensive than mechanical fuels reduction. Hartsough et al. (2008) note that while mechanical treatment costs per acre on forested land are higher than those of a prescribed burn, the sale of merchantable material from fuels reductions can greatly reduce the net cost of these approaches.

The table does not include per acre costs from a widely cited paper by Berry and Hesseln (2004) because the authors do not report them. Instead the authors provide econometric models explaining the variation in per acre fuels treatment costs in Pacific northwest forests. For prescribed fire, operations conducted in a WUI increased per acre costs by 43%; operations in a designated protection area raised per acre costs by 35%. However, per acre costs fell by 0.18% as the size of the area treated grew by 1%. Other factors, such as slope, management objectives, and methods for handling slash also affected costs. The Berry and Hesseln (2004) model for mechanical treatments found similar effects for WUIs, designated protection areas, and treatment area.

Table 10.6: Cost per Acre for Fuel Reduction Methods ($2015)

Method	Vegetation/Location	Cost/Acre	Source
Prescribed Fire	Sagebrush, Great Basin	$20	Taylor et al., 2012
	Forests, multiple USFS regions	$87	Calkin & Gebert, 2006
	Forests, Blue Mtns, OR	$169	Hartsough et al., 2008
	Forests, SW Plateau, AZ	$164	Hartsough et al., 2008
	Forests, SW Plateau, AZ	$98 – $196	Herjpe & Kim, 2008
Mechanical			
Thinning, pre-commercial	Forests, Private and commercial, OR	$250 – >$600	Parker & Bennett, 2008
Slashbusting & Grinding	Forests, Private and commercial, OR	$250 – >$600	Bennett & Fitzgerald, 2008
Mowing & Mastication	Forests, Private and commercial, OR	$40 – >$600	Bennett & Fitzgerald, 2008
General	Forests, multiple USFS regions	$309	Calkin & Gebert, 2006
General	Forests, Blue Mtns, OR	$1,914	Hartsough et al., 2008
General	Forests, SW Plateau, AZ	$915	Hartsough et al., 2008
General	Forests, SW Plateau, AZ	$587 – $875	Herjpe & Kim, 2008

Calkin and Gebert (2006) model treatment costs per acre using data gathered across a number of USFS Regions. All else equal, prescribed fires in a WUI increased per acre costs by 34%. For every 1% increase in the area treated per acre costs fell by 0.35%. The presence of a threatened or endangered species in the treatment zone increased per acre costs by 66%. For mechanical treatments, operations in a WUI increased costs by 62%. Per acre treatment costs fell by 0.30% for every one percent increase in the area treated.

The Efficacy of Fuels Reduction

A rather large literature has evaluated the efficacy of fuels reduction treatments. These studies have examined how treatments have affected both the behavior of fire, but also the co-benefits of fire. Three general approaches have been used. Controlled experiments, such as the National Fire and Fuel Surrogate study, set up an experimental design in which researchers attempt to control all factors other than those of interest (fuel reduction strategies). Natural experiments, in contrast, are those in which some (or most) of the experimental conditions are controlled by nature and are assumed to have been applied in a random manner that mimics a controlled experiment. Such studies involve comparison of the impact of a wildfire on land that had been previously treated with land within or adjacent to the fire perimeter that had not been treated. Finally, simulation modeling involves calibrating dynamic relationships, such as the long-term ecosystem response when treatments are (or are not) successful. Simulation models have the advantage of examining outcomes over a much longer time period (>100 years) than controlled or natural experiments currently allow.

A Controlled Experiment: The National Fire and Fire Surrogate Study

The National Fire and Fire Surrogate study (FFS) follows an experimental design to examine how prescribed fire and its surrogates affect the vegetation, fuel loads, soils and hydrology, wildlife, entomology, pathology, and cost and utilization of forested regions of the U.S. The response of 12 forested sites in the U.S.—including seven sites in western states—to differing mixes of mechanical treatment and prescribed burns was monitored over time (McIver et al. 2013; Stephens et al. 2012). Treatments included prescribed fire, different mechanical techniques, or both, conducted at the same time at nearby locations of a forest, allowing the research teams to follow the size and duration of different treatment effects. The study examined 40 environmental variables one year after treatment, and a subset (30) of those variables two to four years after treatment. Seven variables captured treatment effectiveness on the fuel bed and forest overstorey whereas the remaining 33 variable assessed on soils, the forest floor, vegetation, fauna, and the total ecosystem.

The FFS study found that prescribed fire and fire surrogates were effective in meeting immediate fuel-reduction goals (McIver et al. 2012, p. 69). One year after treatment, stands were likely to be more resilient to wildland fire. The most effective treatment in reducing potential fire intensity was mechanical reduction of biomass (or conversion of biomass to smaller size) followed by prescribed fire. Combined fire and mechanical treatment reduced large tree density and basal area, while increasing snag density. Height to the live crown increased, while the mass of woody fuels and the forest floor decreased.

Though there were few unintended effects of fuels reduction, desirable ecosystem effects were quite subtle, if measurable at all (McIver et al. 2013). The most significant one-year ecosystem effects were on vegetation, with combined fire and mechanical treatments resulting in reductions in carbon and nitrogen and increases in the richness of herbaceous and exotic species. On the forest floor, both carbon an nitrogen were reduced, as was the carbon:nitrogen ratio.

In assessing the two-to-four year effect of treatments, the authors (McIver et al. 2013, pp.75-76) note that the reduction in fuels and desirable changes in stand structure following treatment can be relatively short-lived. Thus, once a fuels reduction effort is initiated, managers must plan on repeating the effort at regular intervals in the future, with prescribed fire "...necessary to restore dry forest systems in the long run."

In the absence of a strict experimental design, researchers can return some time after treatment to evaluate metrics known to relate to fire behavior. Fulé et al. (2012) provide a meta-analysis of the broader literature post-treatment that evaluated forest structure measures for western dry forests. The study gathered information from 139 fuel treatment publications covering seven western states, plus another study conducted in western South Dakota, near the Montana border. The bulk of the studies were from dry forests in Arizona or California; 54 studies provided information sufficient for meta-analysis.

Three treatment approaches were considered: prescribed burning alone, thinning alone, and combined thinning/prescribed burn. The metric evaluated related to surface fuels (fine woody debris, live coarse woody debris, and dead coarse woody debris) and forest structure (tree density, basal area, and canopy cover).[27] Prescribed burns were found to be most effective in reducing surface fuels, with the thinning/burn treatment yielding a smaller effect and thinning alone resulting in greater density of surface fuels. Tree density and basal area were all improved under any treatment approach, but were most effective under the combined thinning/burn treatment. Thinning alone was slightly more effective than the combined thin/burn treatment in improving the canopy cover measure. The authors pose the question of whether fuels treatments can return a forest to its natural behavior. After listing a number of cautionary statements, the authors assert that "...findings to date indicate that thinning and/or burning treatments do have effects consistent with the restoration of natural fire behavior (p. 76)."

Natural Experiments

This evaluative technique does not follow an experimental design such as that used by the FFS. Instead, fuels treatments are applied when and where land managers designate, using the reduction methods managers deem appropriate. Fuels treatments may or may not have been repeated before the return of wildfire. When a naturally-occurring wildfire passes through the landscape, the effects of wildfire on treated and untreated areas is evaluated, often using a quasi-experimental post-fire design.[28]

In 2002 the Rodeo-Chedski fire burned over 450,000 acres in east central Arizona. At the time, it was the largest wildfire ever to occur in the southwestern United States. Just under 40% of the fire's area occurred on National Forests (the Apache-Sitgreaves and the Tonto). Stevens-Rumann et al. (2012) reviewed the literature that examined the effect of the wildfire on treated and untreated areas of the forests, with previous studies finding that untreated stands burned more severely than treated stands and had greater tree mortality. Fuels reduction efforts that had been completed as much as 20 years prior to the wildfire resulted in decreased crown fire hazard. Further, treated sites had greater rate of tree (pine) regeneration than untreated sites.

Nine years after the Rodeo-Chedski fire, Stevens-Rumann et al. (2012) evaluated the effect of fuels treatment on post-fire stand structure at seven paired treated-untreated sites established in the aftermath of the fire. Using the paired comparisons, the authors conclude the pre-wildfire fuels reduction treatments had reduced the severity of wildfire. Tree mortality was higher on untreated sites; surface fuel loads at untreated sites were roughly twice as large as at treated sites though untreated sites did not exceed recommended load densities. Stand structure was also influenced by fuels treatments. Basal area of live trees was higher on treated sites than untreated, whereas snag (standing dead trees) basal area was larger on untreated sites. The authors con-

[27] Basal area measures the total square feet of live (dead) tree stems per unit of area.
[28] Here, "naturally-occurring wildfire" refers to a wildfire that was not intentionally ignited as part of an experimental design.

clude that for forests in the southwestern U.S., "...extensive fuel reduction treatments are needed to reduce the potential for high-severity fires (p. 1115)" and that treatments can result in beneficial differences in stand recovery.

The 2011 Wallow fire was ignited in east central Arizona, eventually spreading into New Mexico as the fire burned nearly 550,000 acres. Waltz et al. (2014) compared the severity of wildfire and forest resiliency areas that had been treated in the 10 years prior to the fire with areas that had been left untreated. All study sites were located in warm-dry mixed-conifer portions of the Apache-Sitgreaves National Forest, which host ponderosa pine, Douglas-fir, white fir, Gambel oak, and some aspen. Further, the treatments were initially applied because they were located in WUIs. The primary pre-fire fuel reduction treatment method was pre-commercial mechanical thinning, followed by fuel removal by either burning or disposal. Sites were evaluated one year after the fire.

Pre-fire treated areas generally had lower tree densities and basal areas than untreated areas when measured in the aggregate or by any of seven tree species. Measured across all species, pre-fire treated areas had roughly 20% of the tree density and 43% of the basal area of untreated sites. Post-fire, these figures rose to 29% and 64% for tree density and basal area, respectively. This indicates a greater burn severity in untreated sites than in treated sites, as basal area loss and tree mortality were lower on treated sites. Further, burn patches of high severity were smaller on treated units than untreated. Based on these measures the authors conclude that the fuel treatments can reduce fire severity and "...increase specific resiliency metrics of forested ecosystems, even within short-term time frames (p. 50)."

Simulation Methods

Both controlled experiments and natural experiments generally evaluate relatively small treatment areas consisting of specific vegetative attributes over relatively short periods of time. Simulation methods allow analysts to scale these results to the landscape-level and provide analyses over much longer time frames. Simulation models specify mathematical equations (whose parameters are based upon the results from controlled and natural experiments or from other study types) and can be used that imitate the properties of forest and range ecosystems, and model ecosystem succession as they evolve and transition from one state to another over the course of many decades. Due to the complexity of simulation modeling, the summary of each representative study is a bit more detailed than with controlled or natural experiments.

Huggett et al. (2008) used a simulation approach to assess the effect on wildfire hazard of thinning operations at a landscape scale, treating nearly 2 million of a possible 3.7 million acres of ponderosa pine or Douglas fir forests in Colorado. Treatment types include an even-aged thin-from-below approach by removing vegetation, starting with the smallest diameter trees, and continuing until a stand density index consistent with target wildfire hazard risks was achieved. Thinning-from-below results in an even-aged stand, as only the larger trees remain. Other treatments removed trees of all sizes—resulting in an uneven-aged stand—again with the goal of satisfying wildfire target risks. Uneven-aged treatments had two variants: one selected a greater proportion of larger trees for removal, and the other selected a greater proportion of smaller trees. Finally, "limited" treatments allowed only a 50% maximum reduction in basal area in achieving the target hazard index, whereas "unlimited" treatments did not have such a constraint.

Fire hazards were measured by the torching index (TI) and the crowning index (CI). The TI measures the wind speed necessary to create a crown fire by moving surface fire up ladder fuels and into the crown. CI measures the wind speed needed to move fire from tree-to-tree in the crown. The goal of the thinning treatments is to increase the required wind speed for these types of fire, so that larger values of TI and CI represent less hazardous conditions. Forests with TI>25 mph and CI>25 mph or TI<25 mph and CI>40 mph were considered at low or very low risk. The simulation model thinned forests to achieve these targets, except when limited to the 50% maximum basal area removal.

Results show that the even-aged thin-from-below treatment was the most effective means of reducing wildfire hazard on the greatest acreage under both the limited and unlimited basal area removal options. This approach was also the cheapest on a per acre basis (roughly, $1000, not including biomass removal). Given the focus on TI and CI as measures of wildfire hazard, the even-aged thin-from-below approach implied a focus on the smallest diameter ladder fuels. Removing only small diameter ladder fuels means the volume of biomass removed was much smaller relative than the alternative thinning methods. If treatment costs are to be offset by sales of larger, merchantable timber and chipped biomass, then an even-aged thin-from-below method might not be least cost.

The effects of fuels treatments on ponderosa pine forest ecosystems were simulated by Taylor et al. (2015). Untreated, the ecosystem can transition through five possible stages, or states. If a forest initially begins as even-aged, closed canopy forest with a high fuel load (State A), it can either burn intensely and severely with a stand replacement fire (State E) or, if it avoids wildfire, it may transition to a mixed-age closed canopy forest with moderate fuel loads (State B). In this state the forest is no longer at as great a risk of severe, stand replacement fire. If it once again avoids fire, the forest will transition to State C, with an uneven-aged forest, an open canopy and low fuel loads. If State C enjoys regular, low intensity fire (or fuels treatment) it will eventually transition to the restored State D, with trees of all ages, an open canopy and a light fuel load.[29]

Taylor et al. (2015) model these transitional states over a 200 year period, starting with State A, both with and without hazardous fuels reduction treatments. In any given year *t* the transition between states is governed conditions at time *t-1* and by parameters for, among other factors, wildfire occurrence and the effectiveness of fuels treatments, each of which have an associated probability distribution. The full 200-year time period covering the ecosystem state-and-transition succession was simulated 10,000 times for two initial states, and for both treatment

[29] Successional State E, the post-crown fire state, may transition back to State A or to an alternatively stable state of grass and shrub land (Taylor et al. 2015).

and non-treatment conditions. In both simulation efforts, the researchers kept track of the number of wildfires (both beneficial and degrading), per acre suppression costs, and the final ecosystem state.

Regardless of the initial ecosystem state A (even-aged forest) or B (mixed-aged forest), approximately 54% of the simulations under the no treatment condition ended with the forest in ecosystem State E, having suffered a stand replacement fire at some point in the 200 year modeling period. State D, the restored forest, was achieved in only 0.2% of simulations beginning at State A and 1.03% of simulations when starting at State B. An average of 2 high severity fires occurred in each 200-year simulation regardless of whether the initial State was A or B.

Fuels treatments applied at State A were designed to transfer the ecosystem to State B after 20 years using pre-commercial thinning and prescribed fire. Treatments applied to State B (prescribed fire and thinning) were expected to cause a transition to State C after 20 years. Some 22% of simulations with the treated even-aged State A ended in State E whereas 19% of simulations initiated at State B ended in State E. State D, the restored forest, was achieved in 1.03% of simulations starting at State A and 37.25% of simulations initiated at State B. A typical simulation averaged less than one high severity fire over the 200 year period.

Taylor et al. (2013) conducted a similar analysis for the successional phases of the Wyoming Sagebrush Steppe (WSS) and Mountain Big Sagebrush (MBS) ecosystems prevalent throughout the Great Basin. WSS systems are characterized by three ecological states. WSS-1 is a healthy sagebrush system, which is maintained by appropriate wildfire and fuels treatment. Unhealthy disturbance (the authors cite excessive spring grazing) can cause a transition State WSS-2, an overgrown "decadent" state with fewer perennial grasses and more annual grasses. A system in decadent state WSS-2 can return to healthy WSS-1, but only with rehabilitation treatments that have uncertain success. Failure to return to WSS-1 will lead to the final state, WSS-3, where the ecosystem is dominated by invasive annual grasses and in which wildfires frequently occur.

The MBS ecosystem has more phases. State MBS-1a is a healthy MBS sagebrush ecosystem, again having mostly perennial grasses and few annual grasses. With regular wildfire and treatment, the ecosystem stays in MBS-1a. Exclusion of fire or fuels management will cause a transition to state MBS-1b, with the introduction of pinyon-juniper and mature shrubs. State MBS-1-b can be reversed to healthy sagebrush with wildfire and appropriate treatments, but the effectiveness of these treatments is uncertain. If treatment or wildfire are absent, the system moves to MBS-2, a closed canopy pinyon-juniper forest with invasive annual grasses dominating perennials. One can return to MBS-1a only with aggressive actions. If wildfire occurs or treatments are absent, the MBS-2 ecosystem transitions to an annual grass dominated landscape subject to large, frequent wildfire.

With no treatment, 200-year simulations beginning in States WSS-1 or WSS-2 had a 93% probability of ending in State WSS-3. Simulation beginning in State WSS-3 always stayed in that state. Treatments proved very successful when applied in simulation beginning in States WSS-1 or

WSS-3: 100% of the simulations initiated in WSS-1 under treatment conditions stayed in that healthy state; 99% of the WSS-3 simulations under treatment conditions ended in WSS-1 at the end of the simulation period. In contrast, only 49% of the simulation initiated at State WSS-2 ended in healthy WSS-1, while the remainder (51%) transitioned to State WSS-3.

Turning to the MBS ecosystem simulations, some 53% of simulations starting in MBS-1a with no treatment remained in that healthy state, with 36% transitioning all the way to State MBS-3. In contrast, 100% of the treatment simulations initiated at State MBS-1a stayed in State MBS-1a. For simulations beginning in State MBS-1-b, 91% transitioned to State MBS-3 without treatment, whereas 1.7% returned to healthy State MBS-1a. Ninety-two percent of no-treatment simulations initiated with an ecosystem in MBS-2 ended in State MBS-3; if treatments are applied, some 45% end in State MBS-1a and 53% end in State MBS-3. Finally, 100% of no treatment simulations starting in MBS-3 stayed in that final, unhealthy state. In contrast, the fuels treatment and rehabilitation simulations found 92% of simulations ending in State MBS-1a and only 2% remaining in State MBS-3.

The simulated ecosystem outcomes suggest that, if the goal is to return an ecosystem to a healthy state, then fuel reduction and rehabilitation efforts are most effective when applied either very early in the successional time frame or very late. This ignores the cost and effort needed to transition from very poor states (WSS-3 and MBS-3) to healthy states (WSS-1 and MBS-1a). The simulations also tracked the net cost of wildfire suppression expenditure savings, net of treatment costs. The low treatment and suppression expenditures incurred when preventing transition out of healthy states, relative to expenditures to return unhealthy ecosystems to healthy states suggest early treatment interventions are more cost effective.

Fuels Reduction Efforts and Wildfire Suppression Costs

Reinhardt et al. (2008) have stated that the objective of fuels reduction efforts is to "…alter the fuel condition so that wildfire is less difficult, disruptive, and destructive (p. 1998)." Given this objective, fuels reductions program may or may not result in fewer fires, a reduction in burned acres, or lower suppression costs. That said, it is possible that some or all of these co-benefits may occur even when the object of fuels reduction is aimed at changing fire behavior. Some researchers have attempted to gauge the effect of fuels reduction on suppression costs. As noted above, any savings in suppression costs due to fuels reduction efforts should be compared to the cost of treatment.

We begin the discussion with the simulation models of Taylor et al. (2015) and Taylor et al. (2013). Turning first to the Taylor et al. (2015) ponderosa pine simulation model, an average of 2.34 wildfires occurred over the 200 year time period (10,000 simulations) when the initial ecosystem was in State A (even-aged, closed canopy forest with a high fuel load). Nearly all of these fires (2.1) were high-severity. The net present value of suppression costs was $176 per acre (constant $2015). By comparison, the State A simulations with fuel treatments saw more wildfires, an average of 2.54 per simulation, only 0.6 of which were of high-severity. Treatments were repeated on average about every 16 or 17 years. Suppression costs with treatment averaged \$120 per acre (some \$56, or 32%. lower than the no treatment model). Treatment costs averaged \$164 per acre, which are greater than suppression cost savings. If one narrowly defines the benefits of treatment as avoided suppression costs, the benefit-cost ratio for treatment is 0.34 (\$56 ÷ \$164).[30]

For the pondersosa pine simulation models that are initiated in ecosystem State B (mixed-age forest with a closed canopy and moderate fuel loads), an average of 2.8 wildfires occurred in the no treatment simulations, of which 2.0 were of high severity (Taylor et al. 2015). Suppression costs averaged \$213 per acre (\$2015). The State B simulations with fuels treatment experienced an average 5.6 wildfires, of which 0.7 were of high severity. Treatments were repeated about every 22 years. Treatment costs averaged about \$157 per acre, with suppression costs averaging \$82 per acre (a reduction of roughly 62%). Again, treatment costs lower the per-acre cost of wildfire suppression but does not fully offset it, resulting in a benefit cost ratio of 0.83.

An analogous set of calculations were tracked in the sagebrush ecosystem simulations of Taylor et al. (2013). For an initial ecological state of WSS-1 (healthy Wyoming sagebrush steppe), some 15.1 wildfires occur over the 200 year simulation when no treatments were applied, resulting in per acre suppression costs of \$384 per acre (\$2015). When treatments are applied, wildfire occurs an average of only 1.8 times; suppression costs with treatments are \$61 per acre, a savings of \$323. This is balanced against per-acre treatment costs of \$24. Treatments in the WSS-1 state satisfy a benefit-cost criterion (13.3), as suppression cost savings easily outweigh treatment costs. The same cannot be said of treatments applied when the initial ecosystem is WSS-2 (decadent sagebrush) or WSS-3 (dominated by invasive annual grasses). In these cases, per-acre treatment and rehabilitation expenses are relatively high (\$224 and \$2,773, respectively) and are not successful as often. Per-acre suppression cost savings (\$146 and \$153) are not enough to offset treatment costs, yielding benefit-cost ratios of 0.7 (WSS-2) and 0.06 (WSS-3).

Treatment and rehabilitation efforts are more successful on early stage Mountain Big Sagebrush ecosystems than late-stage ecosystems. For initial ecosystem state MBS-1a (healthy sagebrush), a comparison of the treatment-no treatment simulations shows fewer fires and great success at maintaining the system in its healthy state. The net present value of suppression cost savings is \$120 per acre against treatment and rehabilitation costs of \$21, yielding a benefit:cost ratio of 5.7 (constant \$2015). Similarly, when the initial ecological state is MBS-1b (early PJ with shrubs and perennial grasses), per-acre wildfire suppression cost savings are substantial (\$403) whereas per-acre treatment costs are relatively low (\$49), yielding a mean benefit:cost ratio of 9.0.[31]

[30] This narrow criterion, of course, ignores the benefit of services provided by a healthy ecosystem.
[31] Taylor et al. report mean values across 10,000 simulation runs. The mean benefit:cost ratio reported by the authors (9.0) is the mean of 10,000 ratios, one for each simulation. This can differ from a single ratio calculated from mean suppression cost savings (\$409) and mean treatment costs (\$49).

Similar to the WSS ecosystems, late-stage treatments in the MBS ecosystems were more expensive and less successful in improving ecosystem health. Treatments applied to MBS-3 (dominated by invasive annual grasses) reduced fire activity from 22.0 fires over the 200-year period to an average of only 7.5 fires. Per-acre suppression costs were reduced by $607 per acre ($2015), but treatment and rehabilitation was very expensive ($3,168), yielding a benefit: cost ratio of 0.2.

In another simulation effort, Thompson et al. (2013) predict the effect of large-scale treatment of land in the Deschutes National Forest. Fuels reduction treatments were projected onto nearly 67,000 acres of a 145,000 acre landscape, or roughly 46% of the total area. Wildfire ignition and growth were simulated over 10,000 replications; the "without treatment" simulations calibrated well to forest conditions and wildfires in recent history. The authors kept track of fire size and suppression costs per acre across the "with" and "without" simulations. The mean size of fires originating in treated areas was 22% smaller, whereas the cost per acre of suppression increased by just over 2%. This result accords well with earlier sections of this chapter, which presented empirical models showing a declining cost per acre as fire size increases; we should expect per-acre costs to rise as fire size gets smaller. As a result of smaller fires size, total suppression costs per fire fell by nearly 16%. Over the entire study area, mean fire size after landscape-scale treatment fell on average by 4.7%, mean costs per acre fell by 0.5%, and mean costs per fire fell by 6.7%.

Offsetting the Cost of Fuels Reduction: Markets for Removed Biomass

As depicted in Table 10.6, fuels reduction treatments are not inexpensive. Prescribed fires costs can approach $175 per acre, whereas mechanical treatments can approach—or exceed—$1,000 per acre. The bulk of rangeland and much of the forested acreage in the western United States is owned and managed by the federal government. In recent years the annual combined USFS/DOI wildfire management budget has ranged between $3 billion and $4 billion, of which roughly 15% is used for hazardous fuels reduction programs, or about $450 to $500 million in any given year (Hoover and Bracmort 2015). Given the sheer size of the federal estate and the need to periodically repeat fuels treatment, the budget is clearly insufficient to achieve fuel reduction goals (Nielsen-Pincus et al. 2013).

Mechanical treatment methods generate biomass that may be sold to help subsidize fuel treatments and extend the limited federal budget. Two key obstacles with this approach are that (i) fuels treatments generally produce low-valued biomass (small diameter trees and chipped volume) and (ii) the contraction of the wood processing industry over the past 25 years (Herjpe and Kim 2008; Neilsen-Pincus et al. 2013).

A simulation model was used by Barbour et al. (2008) to examine where fuels treatments would occur and how much biomass would be removed under differing federal fuels reduction budgets. The area of study was timberland located in the 11 contiguous western states, plus South Dakota; federal and non-federal timberland totaled 83.8 million acres, of which 93% was federally owned.[32] This includes 3.6 million federally owned and 0.2 million non-federal acres in Utah. The federal fuels treatment budget constraints considered were $150 million, $300 million, and $1,500 million for each year of the five-year simulation period.

Barbour et al. (2008) adopted a "least-expensive highest-hazard area first" approach to fuels treatment. The authors established a fuels treatment goal of increasing the torching and crowning indices, using thresholds similar to those used by Huggett et al. (2008). For each state, priority was assigned to each forest type. The highest priority type in Utah was ponderosa pine, followed in order by lodgepole pine, Douglas-fir, fir-spruce, and pinyon-juniper. Eligible stands located in the WUI were ranked higher in priority than non-WUI stands. Of Utah's 3.8 million acres of timberland, some 577,300 acres were deemed eligible for fuels treatment, with just under 40,000 acres of eligible land located in a WUI. For comparison, total eligible land in the 12 states was 17.6 million acres with 1.5 million acres in WUIs.

The proximity of eligible land to sawmills capable of handling the removed biomass was then determined using 2005 information on mill location. The cost of hauling was included in treatment costs, as was the offsetting value of merchantable material. Table 10.7 summarizes the results from the Barbour et al. (2008) study. At a federal budget level of $150 million, very little (<1%) of Utah's eligible, treatable land receives treatment, as fuels treatment efforts are concentrated in Idaho, Arizona, and New Mexico (70% of treated acreage). Doubling the annual budget results in a similarly small percentage (2.2%) of Utah's acreage being treated. Only when the annual fuels budget reaches $1.5 billion does the acreage treated in Utah become substantial, with just under 60% of eligible acreage receiving treatment. With this much larger acreage comes a very large increase in biomass removal, 36% of which has commercial value.

Table 10.7: Simulated treatment area and biomass production in Utah under different budget constraints

	Fuels Reduction Budget		
	$150 million	$300 million	$1,500 million
Area Treated (acres)	2,600	12,700	342,700
% of eligible acreage treated	0.5%	2.2%	59.4%
Total biomass removed (1000 tons)	15	75	3,286
Total merchantable product (1000 tons)	9	31	1,174
Saw timber (1000 tons)	8	12	317

Note: 577,300 acres eligible for treatment in Utah Source: Barbour et al. (2008)

[32] Timberland was defined as land capable of producing 20 ft^3 of industrial wood per acre per year.

Herjpe and Kim (2008) examine the role of sawmill capacity and utilization rates of biomass removals from fuels reduction treatment on five national forests in portions of Arizona, Colorado and New Mexico. The authors begin by noting that the costs of mechanical thinning could be substantially offset if treatments produce material that can be sold to and used by the local wood processing industry. They cite literature that has concluded that regional markets without processing facilities can make mechanical fuels treatments too costly to undertake.

The forests of interest are dominated by ponderosa pines, and most of the fuels reduction efforts in the region have focused on that ecosystem. The regions of two national forests (Apache-Sitgreaves and San Juan) have maintained some sawmill capacity, but the regions of three other forests have not (Coconino, Gila, and Kaibab). Some 105,000 acres had been treated with either prescribed burns (57%) or mechanical removal (43%) in the year the study was conducted (FY2005).

For fuels treatments occurring in FY2005, the region that had retained the greatest sawmill capacity, the Apache-Sitgreaves National Forest, was also the region with the greatest utilization rate (wood processed per acre treated) of removed biomass. Utilization rates in forests with no milling capacity were very low, as the transportation costs of delivering relatively low-valued biomass to distant mills was often prohibitive. The authors conclude that regional mill capacity is an important factor is reducing the net cost of fuels reduction treatments.

A small literature addressing fuels treatments and proximity to processing mills has emerged, and a paper by Neilsen-Pincus, Charnley, and Mosley (2013) serves as a nice representative. The authors correlated the locations of nearly 8,500 fuels reduction treatments in Washington and Oregon with the location of processing mills, allowing them to gauge, among other measures, the influence of mills on the amount of merchantable material removed and on forest managers' choice of treatment locations. Sawmills with the ability to process small-diameter trees and biomass were included in the analysis, whereas mills that specialized in processing bark, products, posts, poles, and log furniture were not.

Some 8,451 treatments totaling over 812,000 acres for the years 2005 through 2010 were evaluated. Fifty-three percent of the treatments were mechanical, and about 38% of the acreage was conducted in WUIs. Of the total treated acreage, two-thirds (66.5%) occurred within 50 minutes driving distance of a mill or biomass facility, with only 33.5% occurring more than 50 minutes away. Dividing ranger districts at the median distance (43 minutes), ranger districts within this perimeter treated almost 7500 more acres per district than those outside the perimeter; these district also treated nearly three times the acreage in WUIs than district further from milling and biomass facilities.

Regression analysis was used to identify the threshold distance beyond which proximity to processing facilities had no statistical influence on districts' treatment decisions. Analysis showed the threshold to be 40 minutes, or about 25 miles. Neilsen-Pincus et al. (2013) caution against directly transferring this figure to other regions. Proximity thresholds estimated in other regions with differing transportation infrastructure and a differing mix of merchantable products have ranged up to 80 miles or more.

Biomass Processing Opportunities and Challenges in Utah

The literature clearly indicates that two key factors in improving the financial margin (net cost) of fuels reduction treatments is to (i) have treatments produce a sufficient quantity of merchantable material and, (ii) do so in relatively close proximity to processing facilities such that hauling costs are not prohibitive. Recent trends in the Utah forest products industry suggest that large portions of the state may have limited ability to offset the cost of fuels reduction by processing removed biomass.

Timber harvest is measured at regular intervals by the USDA and the quantity harvested has tended downward since 1992 (Sorenson et al. 2016). Between 1992 and 2012 timber harvest in Utah fell by 70%, from 64,674 MBF to 19,356 MBF. During the same time period, the number of processing facilities fell by 65%, from 51 to 18. The location of wood processing mills in 2012 are shown in Figure 10.4.

The loss of processing capacity is not unrelated to the fall in timber harvest; though based on only four observations over time, the simple correlation between reported harvest for the state and the number of its processing mills is 0.89. Other evidence in support of a connection between harvest and mill numbers can be found by looking at Garfield and Kane counties. As recently as 2002 these two counties were producing 8,966 MBF per year, almost 23% of the state total for that year. By 2007, Garfield county was near it 2002 output but harvest had been essentially eliminated in Kane county (60 MBF). In 2012, harvest in Garfield county had fallen to 965 MBF and there was no measurable harvest in Kane county.

Returning to the Barbour et al. (2008) simulation results, fuels treatment costs were related to hauling costs to nearby mills. Their analysis relied upon sawmills listed in Prestemon et al.'s (2005) list of mills with capacity to handle small-diameter trees, of which six were located in Utah. Comparison of the Prestemon list with Figure 10.4 reveals that three of the Prestemon mills have closed since 2005. Two of these mills were located in Garfield county. Utah treatments selected under the "lowest-cost high-hazard" selection criterion under the $150 million and $300 million annual budgets were located in southern Utah. With two mills closed, those areas are unlikely to be selected for treatment if the model were replicated in 2016, all else equal.

The contraction of the forest products industry over recent decades offers both opportunity and challenge for fuels reduction programs in Utah. As of 2012, Utah has retained capacity to mill 66,100 MBF, yet is milling only 13,200 MBF. This equates to a capacity utilization rate of only 20%, the lowest of the Four Corners states (Arizona, Colorado, New Mexico, and Utah).[33] Clearly, the forest products sector is capable of absorbing a large volume of merchantable biomass generated by fuels reduction treatments.

[33] Not all mills are capable of processing biomass generated by mechanical treatments.

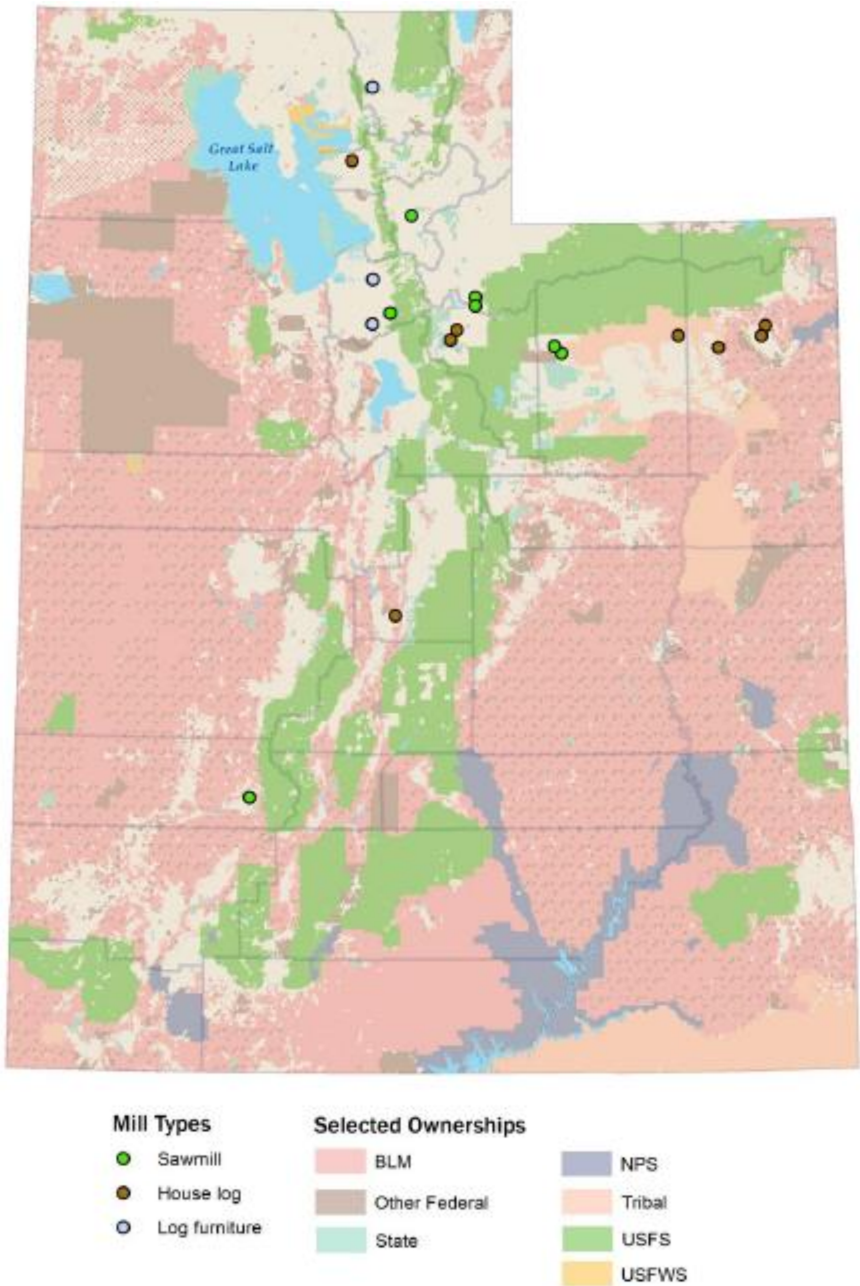


Figure 10.4: Mill location and type, Utah (2012) Source: Sorenson et al., 2016

An important challenge to Utah in reducing the net cost of fuels treatments can be seen in Figure 10.4. Given the relatively high cost of hauling merchantable wood products from treatment sites, mills must be located relatively nearby (roughly, 80 miles or less). Northern and eastern Utah appear to at least come close to satisfying this criterion. In contrast, central and southern Utah have so few mills as to make hauling materials great distances economically prohibitive.

Summary

A recently released USFS study found that some 54% if Utah's forests are fully occupied and that 21% of the state's forests are overstocked. Overstocked forests are at greater risk of increased mortality due to competitive stress, and also at greater risk of catastrophic fire. National wildfire suppression costs are highly variable and are rising over time. Analysis of 450 Utah wildfires found total suppression costs to rise with fire size and increase with rugged topography. Per-acre suppression costs fell with fire size and increased with rugged terrain.

Numerous studies of the efficacy of fuels reduction treatments are in broad agreement that fuels reduction efforts, especially combined prescribed burn/mechanical treatments, can be very effective in modifying fire behavior to reduce the severity of wildfire. Simulation modeling indicates that fuels treatments can also achieve a number of co-benefits, including reduced suppression costs, though this is not a primary goal of most fuels reduction efforts.

Fuels treatments are quite costly--approaching $175 per acre for prescribed fire, and possibly in excess of $1,000 per acre for mechanical treatments. Fuels reduction programs remain a relatively small portion of overall wildfire management budgets; funding for treatments is not sufficient to meet needed landscape-scale fuel reduction efforts.

The cost of fuels treatments can be offset through the sale of biomass removed as part of the treatment process, but hauling costs for small diameter trees and chipped volumes are high relative to its value. Empirical research suggests that treatments tend to occur in proximity to existing wood processing facilities.

Utah's forest timber harvest has fallen by 70% between 1992 and 2012; during the same time period the number of mills fell by nearly 66%. In 2012, Utah mills operated at 20% of capacity, suggesting scope to absorb a large volume of wood product generated by fuels reduction activities. The spatial distribution of mills, though, suggests few opportunities to sell removed biomass to the few remaining mills in central and southern Utah.

References

Abatzoglou, J.T., and A.P. Williams. 2016. Impact of anthropogenic climate change on wildfire across western U.S. forests. *Proceedings National Academy of Sciences*, 113(42):11770-11775.

Agee, J.K., and C.N. Skinner. 2005. Basic principles of forest fuel reduction treatments. *Forest Ecology and Management*, 211:83-96.

Bennett, M., and S. Fitzgerald. 2008, Reducing hazardous fuels on woodland property: mechanical treatments. Oregon State University Extension Bulletin EC 1575-E.

Berry, A. and H. Hesseln. 2004. The effect of wildland-urban interface on prescribed burning costs in the Pacific Northwestern United States. *J. Forestry*, 102(6):33-37.

Calkin, D.E., and three co-authors. 2005. Forest Service large fire area burned and suppression expenditure trends, 1970-2002. J. Forestry, 103(4):179-183.

Calkin, D. and K. Gebert. 2006. Modeling fuel treatment costs on Forest Service lands in the western United States. *Western J. Applied Forestry*, 21(4):217-221.

Davies, K.W., and three co-authors. 2015. Winter grazing can reduce wildfire size, intensity, and behavior in a shrub-grassland. *International J. Wildland Fire*, 25:191-199.

Diamond, J.M., C.A. Call, and N. Devoe. 2009. Effects of targeted cattle grazing on fire behavior of cheatgrass-dominated rangeland in the northern Great Basin, USA. *International J. Wildland Fire*, 18:944-950.

Donovan, G.H., J.P. Prestemon, and K. Gebert. 2011. The effect of newspaper and political pressure on wildfire suppression costs. *Society and Natural Resources*, 24:785-798.

Ellison, A., C. Moseley, and R.P. Bixler. 2015. Drivers of wildfire suppression costs: literature review and annotated bibliography.
https://ewp.uoregon.edu/sites/ewp.uoregon.edu/files/WP_53.pdf

Fulé, P.Z. and three co-authors. 2012. Do thinning and/or burning treatments in western USA ponderosa or jeffrey pine-dominated forests help restore natural fire behavior? *Forest Ecology and Management*, 269:68-81.

Gebert, K.M., D.E. Calkin, and J. Yoder. 2007. Estimating suppression expenditures for individual large wildland fires. *Western J. Applied Forestry*, 22(3):188-196.

Gorte, R. 2013. The rising cost of wildfire protection.
https://headwaterseconomics.org/wildfire/homes-risk/fire-cost-background/

Gude, P.H., and three co-authors. 2013. Evidence for the effect of homes on wildfire suppression costs. *International J. Wildland Fire*, 22:257-548.

Hand, M.S. and five co-authors. 2014. Economics of wildfire management: the development and application of suppression expenditure models. New York: Springer.

Hand, M.S., M.P. Thompson, and D.E. Calkin. 2016. Examining heterogeneity and wildfire management expenditures using spatially and temporally descriptive data. *J. Forest Economics*, 22:80-102.

Hartsough, B.R., and seven co-authors. 2008. The economics of alternative fuel reduction treatments in western United States dry forests: financial and policy implications from the National Fire and Fire Surrogate study. *Forest Policy and Economics*, 10:344-354.

Hjerpe, E.E., and Y-S. Kim. 2008. Economic impact of southwestern national forest fuels reductions. *J. Forestry*, 106(6):311-316.

Holmberg, J., and M. Bennett. 2008. Reducing hazardous fuels on woodland property: pruning. Oregon State University Extension Bulletin EC 1576-E.

Hoover, K. and K. Bracmort. 2015. Wildfire management: federal funding and related statistics. Congressional Research Service Report R43077.

Houtman, R.M., and six co-authors. 2013. Allowing a wildfire to burn: estimating the effect on future fire suppression costs. *International J. Wildland Fire*, 22:871-882.

Ingalsbee, T. and U. Raja. 2015. The rising cost of wildfire suppression and the case for ecological use of fire. Chapter 12 in, DellaSala, D.A., and C.T. Hanson, eds. The Ecological importance of mixed-severity fires: nature's phoenix. New York: Academic Press.

Martinuzzi, S. and five co-authors. 2010. The 2010 wildland-urban interface of the coterminous United States. https://www.fs.fed.us/nrs/pubs/rmap/rmap_nrs8.pdf

McIver, J.D. and 24 co-authors. 2013. Ecological effects of alternative fuel-reduction treatments: highlights of the National fire and Fire Surrogate study (FFS). *International J. Wildland Fire*, 22:63-82.

Neilsen-Pincus, M., S. Charnley, and C. Mosely. 2013. The influence of market proximity an national forest fuels treatments. *Forest Science*, 59(5): 566-577.

North, M., and seven co-authors. 2015. Constraints on mechanized treatment significantly limit mechanical fuels reduction extent in the Sierra Nevada. *J. Forestry*, 113(1):40-48.

Parker, B. and M. Bennett. Reducing hazardous fuels on woodland property: thinning. Oregon State University Extension Bulletin EC 1573-E.

Prestemon, J.P. and six co-authors. 2005. Primary wood processing mill locations in the continental U.S. USDA Forest Service, Washington, D.C. Available at www.srs.fs.usda.gov/econ/data/mills

Reinhardt, E.D., and three co-authors. 2008. Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. *Forest Ecology and Management*, 256:1997-2006.

Rummer, B. 2008. Assessing the cost of fuel reduction treatments: a critical review. *Forest Policy and Economics*, 10:355-362.

Running, S.W. 2006. Is global warming causing more, larger wildfires? Science, 313:927-928.

Schmelzer, L. and eight co-authors. 2014. Reducing cheatgrass (Bromus tectorum L.) fuel loads using fall cattle grazing. *The Professional Animal Scientist*, 30:270-278.

Sorenson, C.B. and six coauthors. 2016. The Four Corners timber harvest and forest products industry, 2012. Resource Bulletin RMRS RB-21. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Stambro, J.C., and five co-authors. 2014. An analysis of a transfer of public lands to the state of Utah. https://csee.usu.edu/htm/current-past-projects/an-analysis-of-a-transfer-of-federal-lands-to-the-state-of-utah/

Strand, E.K., and three co-authors. 2014. Livestock grazing effects on fuel loads for wildland fire in sagebrush dominated ecosystems. *J. Rangeland Applications*, 1:35-57.

Stephens, S.L., and seven co-authors. 2012. The effects of forest fuel-reduction treatments in the United States. *BioScience*, 62(6):549-560.

Stevens-Rumann, C. and three co-authors. 2013. Pre-wildfire reduction treatments result in more resilient forest structure a decade after wildfire. *International J. Wildland Fire*, 22:1108-1117.

Taylor, M.H., and three co-authors. 2013. The economics of fuel management: wildfire, invasive plants, and the dynamics of sagebrush rangelands in the western United States. *J. Environmental Management*, 126:157-173.

Taylor, M.H., and four co-authors. 2015. The economics of ecological restoration and hazardous fuel reduction treatments in the ponderosa pine forest ecosystem. *Forest Science*, 61(6):988-1008.

Thompson, M.P., and four co-authors. 2013. Quantifying the potential impacts of fuel treatments on wildfire suppression costs. *J. Forestry*, 111(1):49-58.

U.S. Forest Service. 2015. The rising cost of wildfire operations: effects of the Forest Service's non-fire work. https://www.fs.fed.us/sites/default/files/2015-Fire-Budget-Report.pdf

Waltz, A.E.M., and five co-authors. 2014. Effectiveness of fuel reduction treatments: assessing metrics of forest resiliency and wildfire severity after the Wallow Fire, AZ. *Forest Ecology and Management*, 334:43-52.

Werstak, Jr., C.E., and 12 co-authors. 2016. Utah's forest resources, 2003-2012. Resource Bulletin RMRS-RB-20. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://www.fs.fed.us/rm/pubs/rmrs_rb020.pdf

Westerling, A.L., and three co-authors. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. *Science*, 313:940-943.