Fire and the Great Basin

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Fire regimes in Great Basin ecosystems have changed significantly since settlement of the region in the mid- to late 1800s. The following provides an overview of the nature and consequences of altered fire regimes, factors influencing the changes, and research and management questions that need to be addressed to maintain sustainable ecosystems. For more detailed information, please see the list of references and recommended links.

What are the consequences of altered fire regimes?

Major changes are occurring in Great Basin vegetation communities—(1) In forested systems, a decrease in fire frequency due largely to fire exclusion has resulted in a shift in species composition from early-seral, shade intolerant species to late-seral shade tolerant species. Increases in vertical stand structure (fuel ladders) and biomass (fuel loads) are resulting in more severe fires (Keane and others 2002). (2) In the pinyon-juniper woodland zone, decreased fire frequency due to fire exclusion, overgrazing through the mid-1900s, and climate change initially facilitated expansion and in-filling of pinyon and juniper trees. As stands mature and canopy closure occurs, the risk of higher-severity crown fires is increasing (Miller and others, in press). (3) In arid and semi-arid shrublands and lower-elevation pinyon-juniper woodlands, an increase in annual invasive grasses, coupled with higher fire frequencies, is resulting in progressive conversion of these ecosystem types to near homogenous grasslands dominated by invaders (Brooks and Pyke 2001).

Changes occurring in vegetation communities are decreasing biodiversity and placing species at risk—Populations of many sagebrush-associated species are in decline, and approximately 20 percent of the ecosystem's native flora and fauna are considered imperiled (Center for Science, Economics and Environment 2002). Fire exclusion in forests that historically experienced higher fire frequencies has resulted in increased landscape homogeneity and decreased patch diversity. Vegetation types that depend on more frequent and lower severity fires are decreasing in abundance, as are the plant and animal species associated with these types (Keane and others 2002). Aspen is declining in many mixed-conifer forests, placing species dependent on aspen habitat at risk. DeByle (1985) documented over 134 bird and 55 mammal species that regularly use aspen forest. Similarly, in mid-elevation sagebrush ecosystems, expansion and in-filling of pinyon and juniper trees within the woodland zone is resulting in the progressive displacement of sagebrush and its associated species (Wisdom 2005). In mid- to low elevation sagebrush ecosystems, salt desert ecosystems, and lower elevation pinyon-juniper woodlands, progressive conversion to exotic annual grasslands is resulting in the widespread deterioration of these habitats.

Changes in vegetation communities and fire regimes are affecting watershed function—In forest systems, fire exclusion and transition to late-seral plant communities can result in greater soil water loss via evapotranspiration and snow ablation that causes decreased stream flows (Keane and others 2002). In more arid pinyon-juniper and sagebrush ecosystems, soil water storage is variable and may or may not be affected by fire depending on soil characteristics and precipitation regimes (Neary and others 2005). Increases in fire extent and severity can result in accelerated rates of soil erosion and sediment production depending on precipitation following the fire. Increased levels of suspended sediment and turbidity often have the largest effects on water quality after fire (Neary and others 2005). Water temperatures can increase following fire due to removal of stream bank vegetation. This can increase biotic productivity and decrease levels of dissolved O_2. These changes in water quality can negatively affect aquatic organisms including fish. Ultimately, impacts on watershed function can be detrimental to users of water (communities, agriculture, and industry) dependent on wildlands for water of adequate quantity and quality.

Changes in vegetation composition and structure are resulting in the loss of ecosystem services and straining the capacity of management agencies—There has been a loss of forage and browse, reducing carrying capacity for both livestock and wildlife, as a result of shrub and tree encroachment. Similar losses are occurring as a result of larger and more severe fires. For example, in 2006, 500,000 ha (1.3 million acres) burned in Elko County, Nevada. This loss of habitat prompted the Department of Wildlife to release additional hunting permits and transplant animals (pronghorn antelope) out of the area. Large fires are resulting in increased fire suppression costs and emergency rehabilitation efforts with over $19 million spent annually on the restoration of sagebrush ecosystems alone. Between 1998 and 2002, the BLM seeded an average of 1.3 million kilos (2.9 million lbs) of seed/year on burned areas. Despite this restoration, success is often unsatisfactory.

Wildland fires are emitting air pollutants that can be harmful to human health and welfare—The *Federal Wildland Fire Policy* (U.S. Department of the Interior – U.S. Department of Agriculture 1995) and the *Clean Air Act as Amended 1990* (PL 101-549) established regulatory and management requirements for determining emissions and air quality impacts from both wildfires and prescribed fires. Smoke management and air quality programs are being implemented with support from research and land management agency programs. These programs directly affect use of prescribed fire in most fire dependent ecosystems within the West. Management of smoke from all burning rests with each state. This emphasizes the importance of interstate/intrastate coordination. Effective monitoring systems and predictive tools are necessary to facilitate fire management and to alert the public and air regulators of air quality threats.

Why are the changes occurring?

Climate has a major influence on fire frequency and severity, but effects vary for the different ecosystem types in the region—Drought or extreme fire weather results in more frequent and larger fires in both forests and shrublands (Keane and others 2002). High elevation subalpine forests are characterized by low frequency, high severity fires. Moister and cooler conditions in most years decrease fire frequency and allow woody fuel buildup. This results in high severity fires when climatic conditions are conducive to fires. Lower elevation forests, such as ponderosa pine forests (*Pinus ponderosa*), typically exhibit higher frequency, lower severity fires due to annual "drought" and higher levels of fine fuels. In arid to semi-arid shrublands, conditions that promote fine-fuel accumulation (wet falls, winters, and springs) or accumulation of fine fuels during the previous growing season increase fire probabilities (Westerling and others 2003). Climate change and warmer temperatures during recent decades have resulted in longer fire seasons, and a drying trend may be contributing to more frequent periods of extreme fire weather (Westerling and others 2006).

Fire exclusion has influenced both forest and rangeland ecosystems, especially areas that burned more frequently in the past—Fire exclusion in forests has resulted in increased woody fuels primarily in mid-elevation forests with mixed-severity fire regimes and in lower elevation ponderosa pine forests with relatively high fire frequencies (Keane and others 2002, Schoennagel and others 2004). Woody fuel accumulations, coupled with longer and warmer fire seasons, have resulted in larger and more severe fires in these forest types during recent decades. Increases in woody fuels in shrubland and woodland types are due not only to fire exclusion, but also to climate change and overgrazing by livestock, which disfavors herbaceous species (fine fuels) (Miller and others, in press). In many shrubland and woodland types, fuels are greater and fire frequency is higher on deeper soils and moister sites (Bauer 2006, Johnson and Miller 2006). In recent decades, fires on these sites also have tended to be larger and more severe than in the past. Vegetation types on shallow soils and harsher (arid or cold) sites with sparse fuels have always been characterized by longer fire return intervals and thus are relatively unaffected by fire exclusion.

Invasive alien plants are expanding throughout the region affecting fire frequency, size and severity—The most significant of these are fire-adapted annual grasses, such as cheatgrass (*Bromus tectorum*), which increase fine fuels, have high flammability, and increase the rate of fire spread (Link and others 2006). In many parts of the region, an annual grass-fire cycle now exists in which fire return intervals have decreased from about 60 to 110 years to as little as 3 to 5 years (Whisenant 1990). It is estimated that cheatgrass monocultures covered a minimum of almost 2 million ha (7,720 mi^2) or 5 percent in the 1990s (Bradley and Mustard 2005) – an additional 15 million ha (57,900 mi^2) have been estimated to be at high risk of cheatgrass invasion (Suring and others 2005). The invasion of cheatgrass into more arid salt desert ecosystem types is resulting in fires in ecosystems that did not burn previously (Brooks and Pyke 2001). More recently, weedy forbs, such as knapweeds (*Centaurea* spp.), are establishing and spreading through the region but their effects on fire regimes are poorly understood.

In the Great Basin area, human populations are growing at the highest rate in the nation increasing the risk of fire at the wildland urban interface—In the western United States, 38 percent of all new homes are constructed in the wildland urban interface. Human activities such as arson or trash burning cause many interface fires, especially under weather conditions conducive to wildfire like low relative humidity, high temperatures, and high winds (for example, Keeley and others 1999). Homeowners in these areas expect fire protection, but access and water are often limited and fire fighting tactics differ considerably for wildfires versus structures. Fire prevention programs focused on outreach and education, reducing hazardous fuels, and restoring fire-adapted ecosystems adjacent to the wildland urban interface are being emphasized, but require collaboration among state, local, and federal agencies as well as homeowners.

Ongoing land uses and management treatments are influencing fire regimes—Increased access and human use of wildland ecosystems by humans have increased fire starts across the region. Roads, off-road vehicle and recreational use, and livestock grazing have increased the spread of flammable invasive grasses (Wisdom and others 2005). Overgrazing by livestock has decreased native herbaceous species and given a competitive advantage to non-native invaders such as cheatgrass and woody species. Management treatments designed to reduce woody biomass in past decades have met with varying success, especially on rangelands. The sagebrush or pinyon and juniper targeted for removal have reestablished or regrown, introduced species seeded onto the sites have decreased biodiversity, and non-native species have invaded treated areas. The resulting changes in vegetation composition have altered both fuel structure and fire regimes, often with unknown consequences for these ecosystems.

Research and Management Questions

What are the relationships of past and present weather and climate patterns to past and present fire regimes?

What are the effects of climate change on changes in stand structure and composition and the resultant fire regimes?

What are the relationships among fuel and fire management and carbon balance?

In which ecosystem types and to what degree have fuels increased with fire suppression?

Where are forest and rangeland restoration treatments appropriate and how will fire respond to fuel-reduction treatments in different types?

Where and when is the influence of short-term (in other words, seasonal and annual) climatic variation expected to override the effectiveness of fuels treatments?

What are the abiotic and biotic thresholds of ecosystem recovery from wildfires and fuels treatments?

What is the landscape composition and structure (in other words, patterns described by patch size, shape, and type) necessary to maintain critical habitat for both plants and animals and thus biotic diversity?

What is the disturbance regime (natural and anthropogenic) needed to maintain a desired landscape-mosaic structure?

What are the effects of both wildfire and fuels management on ecosystem resistance and susceptibility to non-native invaders?

What are the most effective treatments and management actions for restoring ecosystems affected by fire exclusion?

What are the most effective post-fire treatments and management actions for restoring ecosystems?

What are the effects of changes in fire regimes on watershed function?

What are the effects of the size, severity, and pattern of disturbance resulting from wildfire or fuels treatments on watershed function (erosion, water quality and quantity, and biotic organisms)?

What are the effects of fire on riparian ecosystems, stream channels, and aquatic habitat?

What are the relationships among fire characteristics, post-fire climatic events, watershed processes, and aquatic habitat?

What are the relationships among drought, fire severity, and watershed function?

What are the effects of fire rehabilitation treatments on watershed functioning?

What are the most effective monitoring systems for facilitating fire attack strategies and resource allocation decisions and for tracking fire, fire severity, and smoke concentrations and dispersion?

What are the most effective models and software tools for predicting smoke concentrations and dispersion based on fire characteristics and weather?

What are the most effective methods for determining the contribution of live vegetation consumption, masticated fuels combustion, and duff smoldering by fire to smoke generation and carbon emissions?

Existing Programs and Resources

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