FIRE IN UTAH

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Prior to Euro-American settlement in the mid-1800s, fire played an important role in the health and evolution of ecosystems by recycling nutrients, improving soil productivity, and by maintaining biodiversity, community composition, habitat structure, and watershed condition (DiTomaso, 2000; Griffin, 2002; Allen, 2002; Dombeck et al., 2004; Miller and Heyerdahl, 2008). While the value of fire in ecosystems has only been realized in the recent past, aboriginal Americans noted and made use of fire throughout their time in the region (Griffin, 2002; Allen, 2002). Historic accounts show that fire was used in localized areas to increase the availability of desirable plants, as a hunting strategy, and to remove available forage in the event that enemies attempted to cross tribal lands (Downs, 1966; Allen, 2002; Parker, 2002; Griffin, 2002). Accounts by friars Domínguez and Escalante, on their exploration into what would become the Utah Territory, reported intentional burning by local Paiute Indians to dissuade the party of explorers that was mistaken for a group of invading Comanche Indians (Griffin, 2002).

Intentional burning by Native Americans in the western United States caused few changes to the overall vegetation communities when contrasted to larger, naturally occurring fires ignited by lightning (Parker, 2002). The use of fire by native populations, even over relatively small geographic areas for cultural purposes, was vilified by Spanish explorers and Euro-American settlers moving into the region (Downs, 1966; Griffin, 2002). The practice of utilizing fire to improve the environment was eventually referred to as Paiute forestry by settlers and deemed punishable by early Western laws (Wuerthner, 2006).

Consequently, fire suppression policies were heavily enforced, resulting in modifications to fire behavior. The frequency, intensity, severity, and seasonality of fires changed (Bock et al., 1993; Davison, 1996). Simultaneously, these alterations increased the complexity and cost of fire suppression, forcing the government to bring administrative tasks under the jurisdiction of federal organizations, such as the United States Forest Service and the Bureau of Land Management. These organizations were tasked with the responsibility of fighting and suppressing wildland fires across the United States. Presently, these agencies are supported by billions of dollars, fleets of ground and aircraft equipment, and an army of manpower (Wuerthner, 2006). The financial burden of fire fighting and suppression has subsequently been incurred by United States taxpayers (Wuerthner, 2006).

Due to the alteration of natural fire regimes, significant changes to the vegetation structure, vegetation type, and the natural fire return intervals have occurred. Major ecosystems, including grasslands, sagebrush, sagebrush steppe, and upland forested regions have experienced some of the greatest alterations due to fire suppression policies. The Federal Wildfire Occurrence Dataset indicates that Utah was subject to nearly 24,000 fires between 1980 and 2007 (Figure 8.2.1). To date, 2007 was the largest fire year on record with more than 1,400 fires burning 620,730 acres. The 2007 season saw the largest recorded fire in Utah's history, the Milford Flat Fire, burning 363,052 acres of land near Milford, Utah (NICC, 2007). The increased frequency and intensity of fires has had a significant impact on the ecosystems of Utah.

EFFECTS OF FIRE ON GRASSLAND ECOSYSTEMS

The desert and mountain grassland ecosystems of Utah have been exposed to the effects of altered fire return intervals since settlement by Euro-Americans. Prior to widespread settlement, fire ignition was generally caused by lightning and only occasionally by aboriginal peoples (Rice et al., 2008; Griffin, 2002; Parker, 2002). Pre-settlement fire return intervals in desert grassland systems are debatable, but thought to have ranged from 35 to more than 300 years, and the higher, more mesic mountain grassland systems ranged from 10 to 110 years (Figure 8.2.2). This broad variation in return interval is due to changes in available seasonal precipitation and temperature (UDFFSL, 2007; Rice et al., 2008). Introduction of non-native species and open grazing practices during the late-1800s and through the 1900s in these ecosystems increased flammable fuel loads, enhanced fire susceptibility, decreased biodiversity, and shortened the fire return intervals (Rice et al., 2008). In addition to these changes, and because the system did not evolve with regular fire, plants in the desert and mountain systems have not adapted to repeated burning and have largely decreased in overall health and abundance (UDFFSL, 2007).

As settlements expanded in desert grassland areas, non-native species were planted to stabilize soils and improve depleted forage in overgrazed areas (Rice et al., 2008). In the late-1800s, species, such as red brome, were inadvertently introduced and began rapidly invading the already invasive-prone systems. Woody plants that occurred in desert grassland systems were found to be highly susceptible to

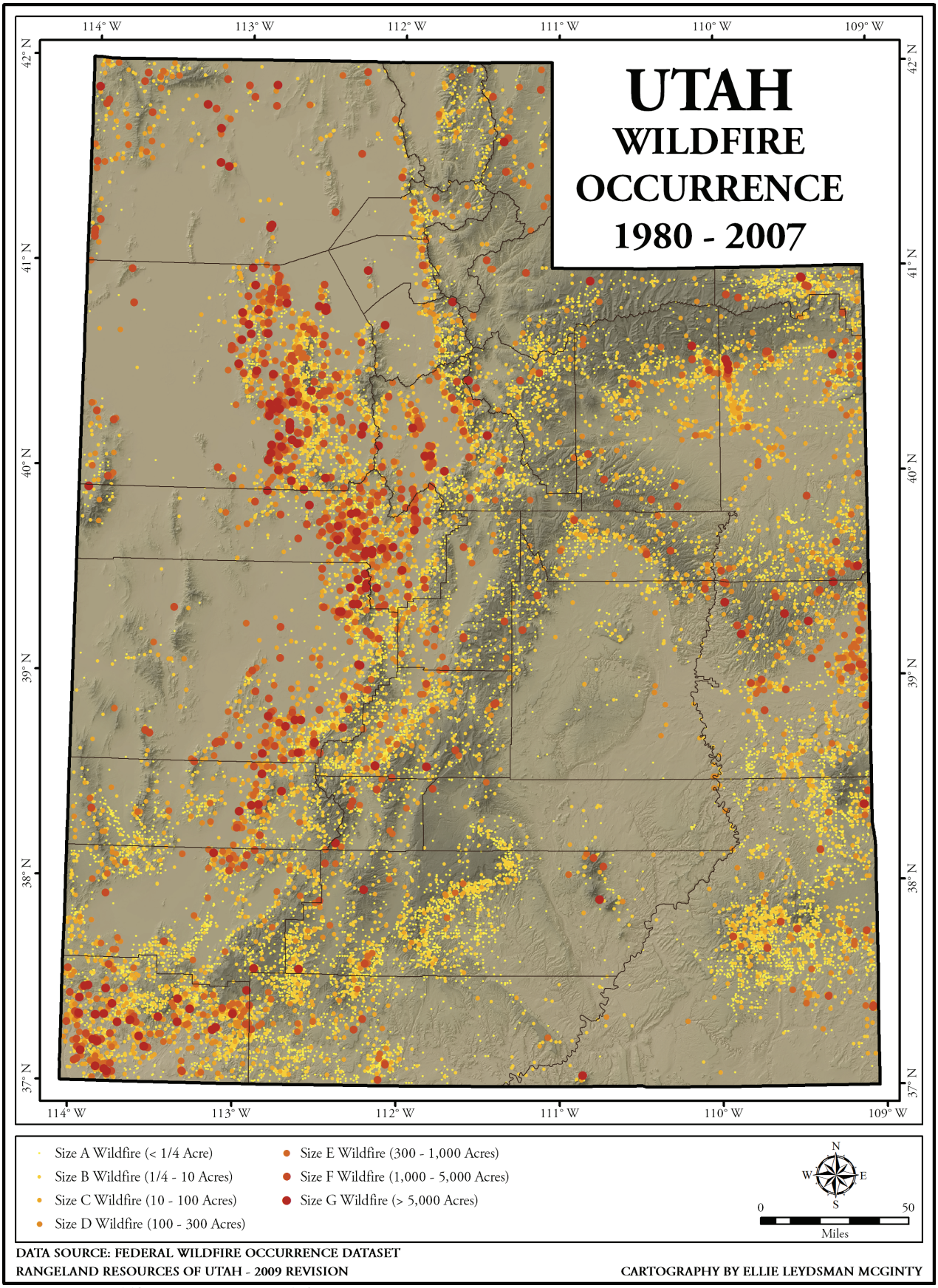

Figure 8.2.1. Wildfire occurrence in Utah from 1980 to 2007.

fire and the slow-growing species would rarely re-sprout once burned. Further, many of the slow-sprouting woody plants could take up to 10 years to begin producing seed, thus reducing the probability of site recovery from burning and intense grazing (Rice et al., 2008).

Mountain grasslands in the region are less susceptible to fire than those of the xeric desert lowlands; however, dry years bring an elevated risk of fire to the system. These regions have historically offered plentiful livestock forage in wet years and adequate forage even in drier years. Because these native systems often maintained dense stands of native grasses, advancement of non-native species due to fire has been slower than those systems in desert environments. Invasive species such as cheatgrass (*Bromus tectorum*) and medusahead (*Taeniatherum caput-medusae*) have made inroads into these communities, affecting native plant populations, livestock, and wildlife forage (Rice et al., 2008).

EFFECTS OF FIRE ON SAGEBRUSH AND SHRUBSTEPPE ECOSYSTEMS

Fire is a natural and essential component in native sagebrush steppe and semi-desert shrubland ecosystems. The frequency of fires in native vegetation communities is variable and depends on sagebrush, shrub, or woodland species, geographic location, climatic variables, and soil properties. In many semi-desert shrubland communities, the structure, characteristics, and lack of a continuous fuel source do not readily promote the spread of fire; therefore, the fire return interval typically ranges from 60 to 110 years (Pimental et al., 2005). Sagebrush steppe and mountain brush plant communities occurring at higher elevations and latitudes in the Great Basin desert have a shorter mean fire return interval of 30 to 100 years because the shrub cover is denser and the shrub architecture is more flammable (Brooks and Pyke, 2001). Native communities of mountain big sagebrush have a mean fire return interval of 12 to 25 years (Miller and Tausch, 2001) (Figure 8.2.2).

In many sagebrush steppe and semi-desert shrubland ecosystems, the behavior and characteristics of fire have been modified as a result of fuel reductions by livestock grazing and human-induced fire suppression (Brooks and Pyke, 2001). Large concentrations of livestock have significantly reduced the cover of native grasses and forbs, and consequently facilitated the establishment of invasive plant species. Invasive plants, such as cheatgrass, have increased the fire return interval and inhibited the germination and propagation of native annuals and perennials (Pimentel et al., 2005). Cheatgrass provides a dense and continuous fuel source that extends the seasonality and increases the frequency of fires (USGS, 2002). Consequently, it often converts arid low-elevation sagebrush-bunchgrass communities into annual-dominated grasslands (Davison, 1996). The change in natural fire regime and conversion to non-native annual grasses has had inadvertent impacts on wildlife species (USGS, 2002).

Historic grazing practices and fire suppression has also encouraged the expansion of woodlands into areas previously occupied by sagebrush and semi-desert shrubs. Historically, fire played an integral role in maintaining sagebrush-steppe communities by limiting conifer encroachment, but the rapid increase in domestic livestock and reductions in fire frequency created ideal conditions for the establishment of woodland seedlings, such as juniper and pinyon pine (Bock et al., 1993; Madany and West, 1983). Woodland species began increasing into low and mountain big sagebrush communities during the late-1800s when grazing by livestock reduced the fine fuels required for low-intensity fires and decreased the competition provided by native herbaceous species (Miller and Rose, 1999). The extent of mountain big sagebrush has been significantly reduced by recent woodland expansion because the fire return interval has increased to greater than 100 years in some regions (Miller and Tausch, 2001).

EFFECTS OF FIRE ON UPLAND FOREST ECOSYSTEMS

Historically, fire influenced the structure, composition, and dynamics of semiarid western, interior forests (Zimmerman and Neuenschwander, 1984; Belsky and Blumenthal, 1997). Western forest tree densities, particularly in juniper woodlands and ponderosa pine forests, were maintained by two natural phenomena: low-intensity surface fires and competitive exclusion of tree saplings by dense understory grasses (Belsky and Blumenthal, 1997). Modifications in western forest ecology have occurred as a result of post-settlement land-use change and management, heavy grazing by sheep and cattle, reduced return intervals for low-intensity ground fires that served to thin dense stands of younger trees, and favorable climate years for tree reproduction around the turn of the nineteenth century (Borman, 2005; Belsky and Blumenthal, 1997; Miller and Tausch, 2001).

Juniper and pinyon-juniper woodlands have experienced pronounced change in both the distribution and density across the Intermountain West. Prior to Euro-American settlement, juniper and pinyon pine species were primarily

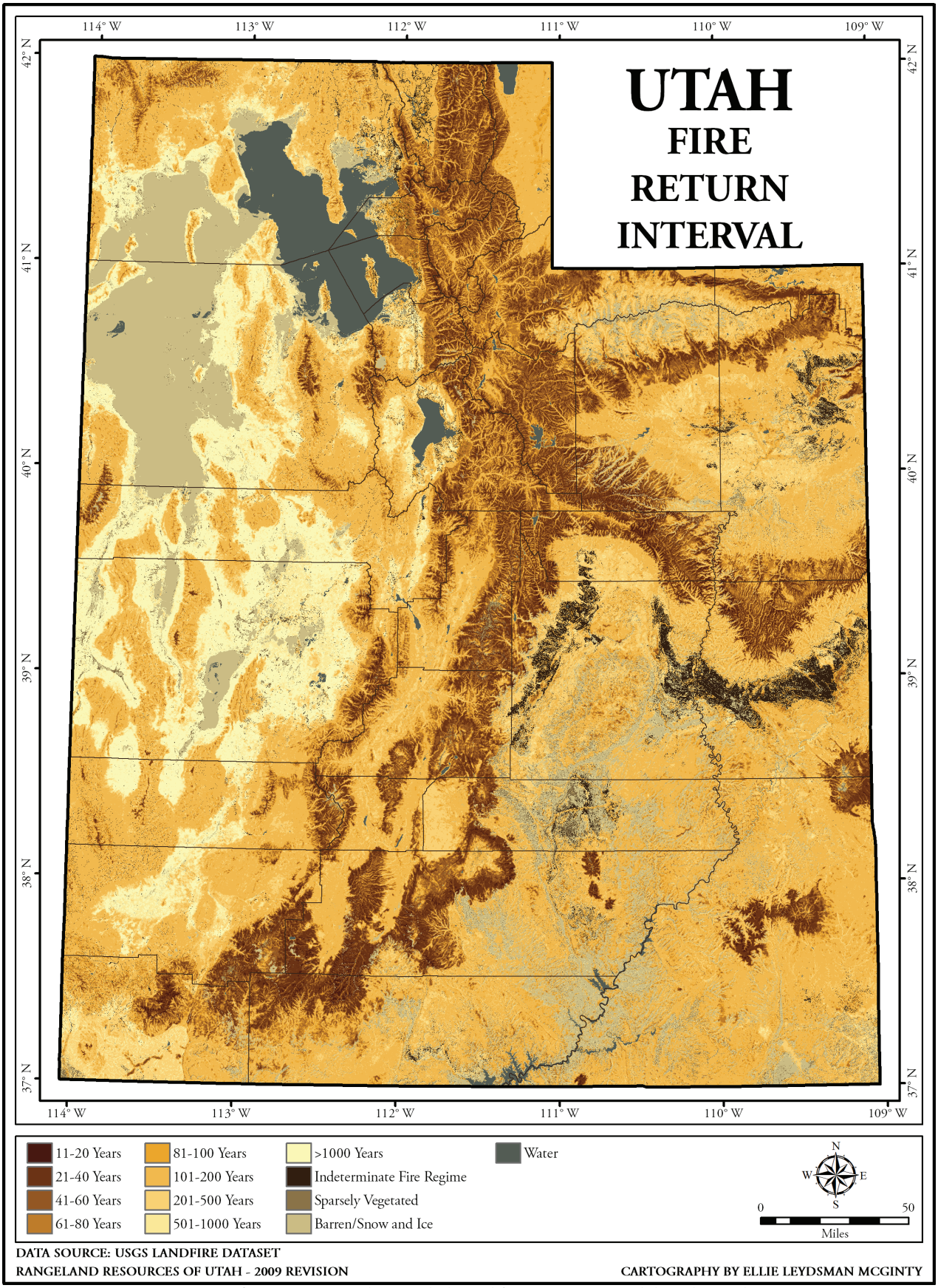

Figure 8.2.2. USGS LANDFIRE fire return intervals in Utah derived from vegetation and disturbance dynamics.

confined to rocky ridges or surfaces where sparse vegetation limited fire. Woodlands were characteristically open, sparse, and savanna-like from frequent low-intensity fires (Miller et al., 1995; Madany and West, 1983). However, juniper and pinyon-juniper woodlands throughout the Great Basin began to expand during the late 1800s and early 1900s. The expansion coincides with the introduction and increase of livestock, and the subsequent reduction in herbaceous species that served as fine fuel loads. Additionally, between 1850 and 1916, winters became milder and precipitation was greater than the long-term average. The wetter, milder conditions promoted vigorous growth in conifers (Miller and Rose, 1999; Miller and Tausch, 2001). The expansion of juniper and pinyon-juniper woodlands has been most dramatic in areas with deeper, well-drained soils; consequently, sagebrush steppe, semi-desert shrubland, grassland, aspen, and riparian plant communities are being invaded and displaced (Miller et al., 1995; Miller and Tausch, 2001).

In Rocky Mountain forests, the most extensive and heavily impacted communities have been those dominated by ponderosa pine and aspen. Historically, ponderosa pine forests were characterized by open stands of trees with a lush, herbaceous understory of perennials and varying densities of shrubs. Frequent, non-catastrophic fire was an important determinant in maintaining plant community structure and composition, particularly in dry southwestern ponderosa pine forests (Bock et al., 1993; Schoennagel et al., 2004). The historic mean fire return interval of ponderosa pine forests varied from 4 to 36 years (Schoennagel et al., 2004); however, research indicates that mean fire return interval in intermediate- and high-elevation ponderosa pine forests was longer, ranging from 30 to greater than 40 years (NPS, n.d.) (Figure 8.2.2). Aspen, a disturbance-dependent species, has also declined over much of its former range due to fire suppression and conifer encroachment (Bartos and Campbell, 1998).

Although higher-elevation ponderosa pine forests typically have longer fire return intervals and higher intensity fires than ponderosa pine forests in the southwest (NPS, n.d.), livestock grazing and fire suppression policies have promoted widespread stands of dense fire-sensitive and disease-susceptible trees (Belsky and Blumenthal, 1997). Intensive grazing by sheep and cattle was the primary agent in reducing the herbaceous vegetation and modifying vegetation structure (Madany and West, 1983; Touchan et al., 1995). The shift in vegetation structure encouraged the proliferation of trees, reduced flammability, and decreased fire frequency (Madany and West, 1983). Extensive fire prevention efforts from 1930 through 1960 intensified the effects (Borman, 2005). Presently, the dense stands of trees can provide fuels at intermediate heights that can carry fire up into continuous canopy fuels, thus increasing the probability of large, catastrophic, and stand-replacing fires (Schoennagel et al., 2004; Baker and Ehle, 2001).

FIRE AS A MANAGEMENT STRATEGY

Fire suppression efforts have interrupted the natural fire cycle in many intermountain rangeland environments (Weber et al., 2001). The frequency, intensity, severity, and seasonality of fire have been altered. Vegetation and wildlife communities have been modified; rangeland productivity has decreased; fuel loads have reached unprecedented levels; fire-tolerant, non-native plants have proliferated; and catastrophic fires have become common (Bock et al., 1993; Davison, 1996; Weber et al., 2001). Consequently, federal and state agencies are beginning to focus on management strategies that reduce fuel build-up and the risk of fire. One of the primary methods being implemented and evaluated is prescribed burning.

Prescribed burning is the controlled application of fire to wildland fuels to attain planned resource management goals (Johnson, 1984). Research conducted in the Rocky Mountains confirmed the widespread use of fire by native people to manipulate and improve vegetation communities (Kay, 2007). When prescribed burning policies are founded on ecological principles, prescribed burning can reduce wildfire risk and severity, control invasive plants, suppress woody species, improve forage and rangeland productivity, and enhance wildlife habitat and native plant communities (Brooks and Pyke, 2001; DiTomaso, 2000; Dombeck et al., 2004; Madany and West, 1983; Yoder et al., 2003). Prescribed burning can mitigate fire severity through the reduction of tree and shrub density and accumulated fuels (Pollet and Omi, 2002; Madany and West, 1983).

Prescribed burning is an ecologically sound way to improve wildlife habitat. Land management plans that integrate prescribed burning can enhance the habitat of game species and plants and/or animals of concern. It can open areas for increased movement, reduce ground litter, control brush encroachment, increase nutritional value, and diversify plant species (Anderson and McCuistion, 2008). Fire removes litter and dead standing herbage of low nutritional value and increases forage production. Consequently, herbivores can more efficiently select nutritious plant material (Bleich et al., 2005, Madany and West, 1983; Anderson and McCuistion, 2008).

Prescribed burning can be used to control invasive plants. However, the decision to use fire as a management tool must evaluate interrelationships between fire and invasive plants because fire may promote rapid recovery of invasive species and/or the establishment of other fire-tolerant invasive plants (Brooks and Pyke, 2001; DiTomaso, 2000). Information on the physiology, anatomy, life history, and seed dispersal and longevity of invasive plants is integral to the decision (Brooks and Pyke, 2001). The timing of prescribed burning is critical for success. In general, prescribed burns should be conducted following seed dispersal by invasive plants and senescence of native grasses and forbs (DiTomaso, 2000). December is the preferred month to avoid damage to native forbs in sagebrush steppe environments (Anderson and McCuistion, 2008). Additionally, rehabilitation work, such as seeding with mixes of native species, is often required after prescribed burns (Beck and Mitchell, 2000). Immediate revegetation with desirable and competitive plant species is a sustainable long-term method for suppressing invasive plants, while providing high forage production on rangeland (DiTomaso, 2000).

Although prescribed burning is gaining favor in many areas, it has some drawbacks. Prescribed burning is an inherently risky resource management tool because there is a threat that the fire may escape and spread, people may be injured, and equipment may be lost. Therefore, prescribed burning can impose unintended costs (Yoder et al., 2003; Johnson, 1984). Also, the smoke and pollution produced by prescribed burns may violate regulations, such as the Clean Air Act, and may impact surrounding communities (Davison, 1996). Because of air quality concerns and the need for correct fire-weather conditions, there is usually a narrow period of time in which prescribed burning can be conducted (Nader et al., 2007).