Chapter 5

Fire History, Effects, and Management in Southern Nevada

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Introduction

Fire can be both an ecosystem stressor (Chapter 2) and a critical ecosystem process, depending on when, where, and under what conditions it occurs on the southern Nevada landscape. Fire can also pose hazards to human life and property, particularly in the wildland/urban interface (WUI). The challenge faced by land managers is to prevent fires from occurring where they are likely to threaten ecosystem integrity or human developments, while allowing fires to occur where they will provide ecosystem benefits. The Southern Nevada Agency Partnership (SNAP) Science and Research Strategy summarizes this desired outcome with Sub-goal 1.1, which is to manage wildland fire to sustain Southern Nevada's ecosystems (table 1.3; Chapter 1). This chapter provides information that will help land managers develop strategies to achieve this goal. It begins with a background section on fire history, spatial and temporal patterns of fire, and fire effects for the major ecosystem types of southern Nevada, (table 1.1; Chapter 1). Potential fire management actions are then discussed, the overall implications of the information to fire management are summarized, and the major knowledge gaps are described.

Fire History

Southern Nevada is situated within a broad ecotone between the Central Basin and Range of the Cold Desert ecoregion to the north and the Mojave Basin and Range of the Warm Desert ecoregion to the south, two regions more commonly recognized as the Great Basin and Mojave Desert (Chapter 1). The topography of this region is dominated by broad basins separated by isolated mountain ranges that contribute to local environmental gradients that translate into highly variable vegetation types and fire regimes. The predominant view of fire in this region is focused on the ecosystem types that dominate the majority of the landscape, namely Mojave Desert scrub and to a lesser extent blackbrush/shadscale (table 1.1; Chapter 1). This view is that fires were historically infrequent; all fires are detrimental because they have been historically infrequent and native species are not adapted to them, and significant management responses are often required to mitigate their negative effects on natural, cultural, and recreational resources. Although one or more of these assumptions is likely accurate across most of this region, there are some areas that likely do not align with this dogma because of their differing fire histories and recovery potential. The probability of fire in any ecosystem is a function of sufficient fuel, conducive weather conditions, and sources of ignition (van Wagtendonk 2006). These factors vary over the landscape in southern Nevada and back through history, so it is erroneous to think that the occurrence of fire has been relatively constant across space and time. Understanding the fire history of southern Nevada is essential for evaluating the causes of recent fire trends and placing them within the correct evolutionary context to better evaluate their effects, develop well justified fire management plans, and manage individual fires appropriately.

Prehistoric Fire Record

The deserts of North America were created during the early Pleistocene (approximately 2 million year ago) when uplifting mountain ranges established a rainshadow blocking storms moving eastward from the Pacific Ocean and northwestward from the Gulf of Mexico (Axelrod 1995). In the process, forests and woodlands retreated from lowlands and moved into higher elevation refugia. Since the last glacial period at the beginning of the Holocene approximately 10,000 years ago, the Mojave Desert and southern Great Basin Desert have continuously experienced arid to semi-arid conditions (Van Devender 1977; Van Devender and Spaulding 1979; Tausch and Nowak 2000; Tausch and others. 2004). Conditions since then have fluctuated, but generally trended toward increased aridity (Van Devender 1977; Van Devender and Spaulding 1979) that has likely caused an upslope shift in vegetation formations and their associated fire regimes (McKinley and others, in press; Tausch and Nowak 2000; Tausch and others 2004).

Indigenous humans also influenced fire regimes in pre-historic North America (Williams 2000), including Paiutes in southern Nevada and Shoshones in eastern Nevada (Stewart 1980). Fire was used for hunting game, clearing vegetation, growing food, opening pathways of travel, managing pest species, and facilitating the growth of vegetation with desirable properties (e.g. basket materials, tobacco, seeds for meal, game forage, fuel-breaks). The shift of vegetation formations upslope during the Holocene was likely mirrored by indigenous humans following natural resources necessary for survival in the Mojave and Great Basin deserts. In the Mixed Conifer, Piñon and Juniper, and Sagebrush Ecosystems of the Mt. Irish area of Lincoln County, fires were very frequent from 1550 to 1860 (mean fire return interval 4 years), but then declined precipitously between 1861 and 2006 (only two fires in 1883 and 1916) (Biondi and others 2011). This transition from frequent to infrequent fire was coincident with Euro-American settlement and the displacement of Native Americans in this region during the middle 1800s. Accordingly, fire most likely occurred primarily at higher elevations and in more mesic riparian areas, both because fuels were more conducive to fire spread and ignitions from Native Amreicans were likely more frequent in those areas. In addition, current lightning occurrence in southern Nevada is positively correlated with elevation due to the orographic effects of terrain on climate (Randerson and others 2004), a physical phenomenon that has likely occurred as long as the mountain ranges of southern Nevada have been in existence back through the Holocene and beyond.

With increased aridity, decreased productivity, and decreased human presence over prehistoric time, the spatial extent of fire across southern Nevada undoubtedly declined and became increasingly isolated within disjunct high elevation areas. A dominance shift from perennial grasses to woody non-sprouting shrubs (Spaulding 1990) suggests a change in fuelbed characteristics from one that is conducive to fire spread and adapted to periodic fire, to one which is less conducive and less resilient to fire (McKinley and others, in press; Miller and Wigand 1994). In addition, the change from perennial grasses to shrubs suggests a shift of summer rains of the North American monsoon away from southern Nevada, and with that shift a decline in the incidence of lightning from summer thunderstorms.

Historic Fire Record

The first accounting of extensive historic fire in southern Nevada was reported by Croft (1950) who estimated that 20 percent of the 161,875 ha (400,000 acres) of blackbrush (*Coloegyne ramossissima*) that occurred in the region burned during the late 1930s and early 1940s. This time period coincided with a 4-year period of high rainfall and strong El Nino-Southern Oscillation (ENSO) signature from 1938-1941, which was one of the highest rainfall years on record, eclipsed only by rainfall totals during 1983 and 2005 (Hereford and others 2006; National Climate Data Center, www.ncdc.noaa.gov). It appears that high rainfall temporarily increased fuel loading and continuity and facilitated fire spread. During the mid-century drought from 1942 to 1975 there were relatively few fires documented in southern Nevada, followed by a significant increase in fires when precipitation began to increase in 1976 (Brooks and others 2007; McKinley and others, in press). This has been attributed to low fuel loads and fuelbed continuity and infrequent lightning from summer monsoons during the middle decades of the 1900s, followed by increases in precipitation and lightning during the latter decades (McKinley and others, in press).

Ranchers and U.S. Bureau of Land Management (BLM) staff implemented a program of prescribed burning in the late 1930s and into the 1940s to increase forage production for livestock in blackbrush stands of southern Nevada (Croft 1950). Additional blackbrush burning likely occurred at least through the 1960s, because a policy review during that time by the Range and Forestry Office of BLM in Nevada recommended that blackbrush burning continue to increase forage production (Dimock 1960). However, it is unlikely that most of these mid-century fires spread extensively considering the low productivity of these systems and drought conditions during the middle 1900s.

Current Fire Record

Fire records for recent decades have been derived using point occurrence databases publically available for Federal lands in the Mojave Desert (Brooks and Esque 2002; Brooks and Matchett 2006; Brooks and Minnich 2006) and other desert regions in North America (e.g. Collins and others 2006; Knapp 1997; Schmid and Rogers 1988and others). Although these point occurrence databases include all reported fires, they can also contain a high degree of error, up to 30 percent for DOI lands (Brown and others 2002) and can therefore be highly misleading. There are also regional sources of fire perimeter data that can provide more precise information than that associated with point occurrences (McKinley and others, in press). However, these fire perimeter databases can also misrepresent area burned by as much as 18 percent (Kolden and Weisberg 2007) and are typically only available for short time intervals within specific geographic areas. In contrast, the current fire record results summarized below in this chapter were derived using Landsat satellite imagery to precisely document area burned by large fires (≥1,000 acres) between 1972 and 2007 in Lincoln and Clark Counties (McKinley and others, in press). Although these large fires only represent 1-2 percent of the total number of fires that occurred during that time interval in southern Nevada, they comprise 93 percent of the total area burned.

Approximately one million acres burned in Lincoln and Clark counties as a result of 116 large fires that occurred during a 36-year period from 1972 through 2007 (McKinley and others, in press). A chronology of these fires is graphically portrayed in figure 5.1 where shades from cool to warm colors represent the chronology of fire occurrence beginning with the oldest fires (blue) and ending with the recent fires (red). This figure shows that most of the older fires (pre-2005) occurred in Lincoln County, aside from a cluster of fires during the 1980s in the Spring Mountains of western Clark County.

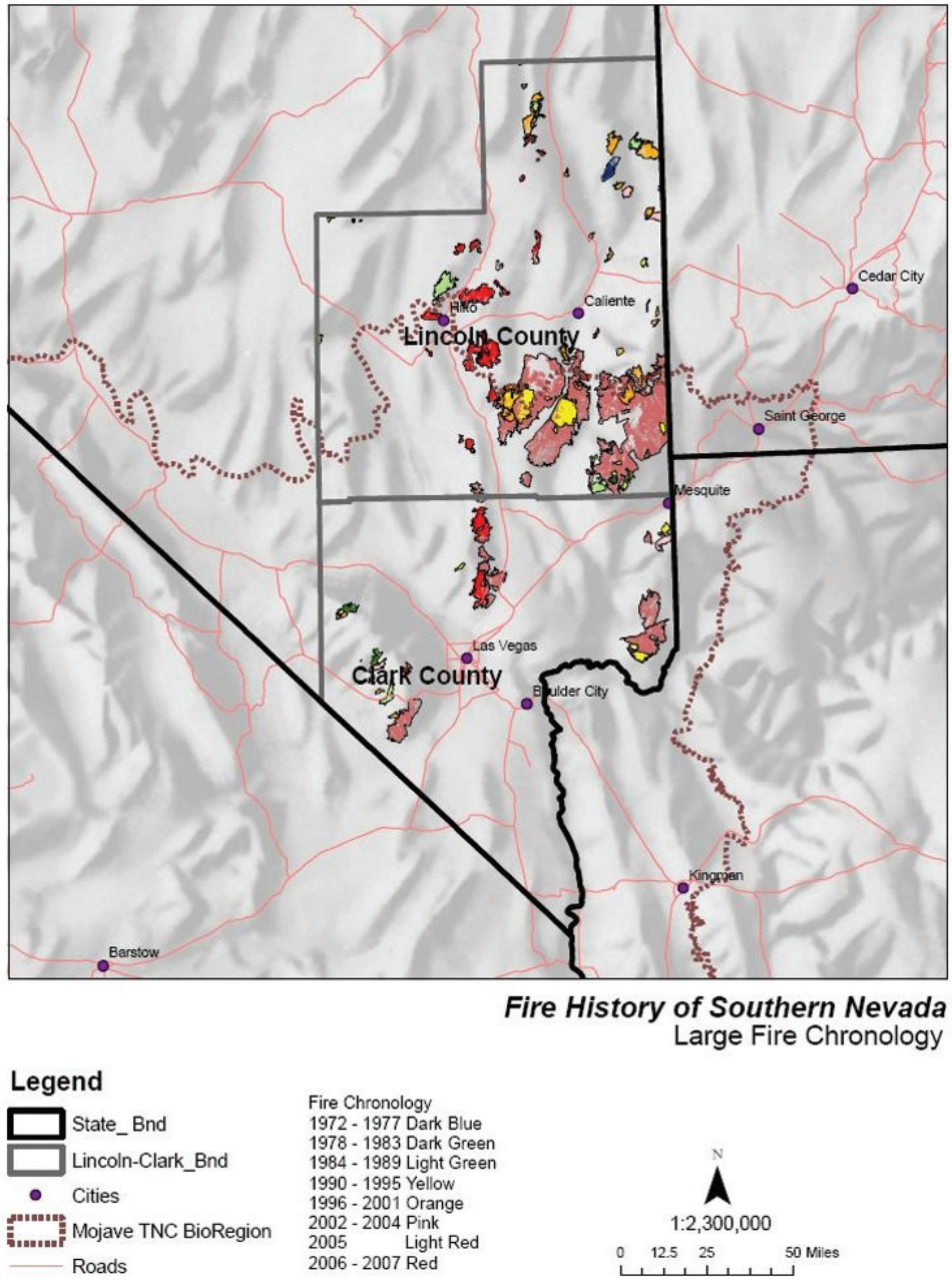

Figure 5.1—Large fire chronology 1972 through 2007 inclusive (≥1,000 acres). Shades from cool to warm colors represent the chronology of fire occurrence beginning with the oldest fires (blue) and ending with the recent fires (red) (reprinted with permission from McKinley and others, in press).

Figure 5.2 illustrates that most of the burned acreage (90%) occurred in areas that had not previously burned during the 36-year study period, 8 percent occurred in areas that had burned once before, and 2 percent occurred in areas that had burned two or three times before during this time period.

The number of fires and area burned from 1972 through 2007 may have been unprecedented in the historic record extending back to the late 1800s (McKinley and others, in press). In particular, the 2005 and 2006 fire seasons were extreme events that had

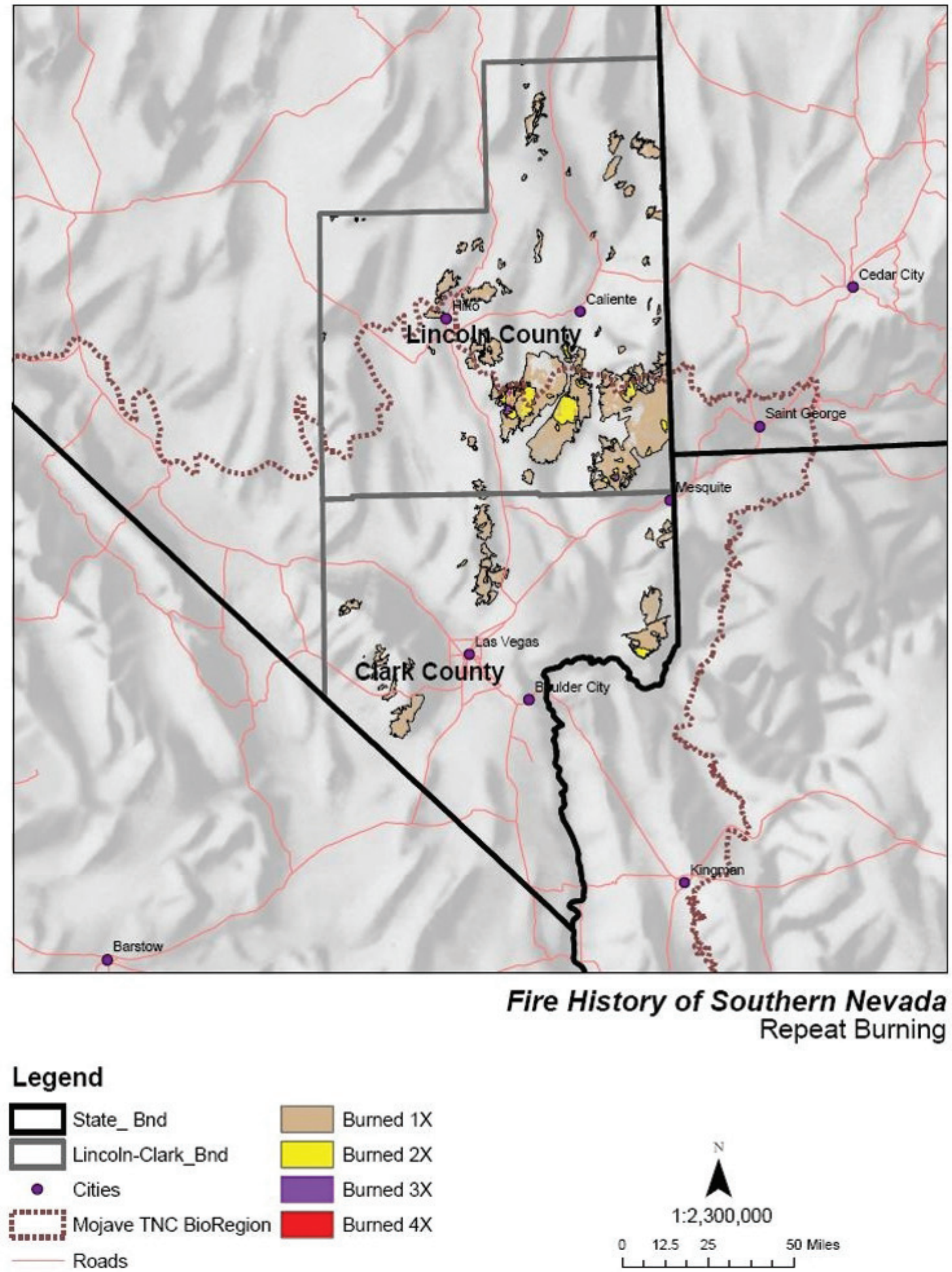

Figure 5.2—Fire frequency of large fires 1972 through 2007 inclusive (≥1,000 acres) (reprinted with permission from McKinley and others, in press).

a major influence on the fire statistics for both number of fires and total area burned. The number of fires and area burned in those 2 years were so statistically anomalous that it is appropriate to evaluate trends with and without them included in the analyses. When 2005 and 2006 were excluded from the analyses, the number of fires and total area burned from 1972 through 2007 actually showed an increase up to the mid-1990s followed by a downward trend (fig. 5.3), a pattern that has persisted through 2012 (Matthew Brooks, personal observation of fire frequency while conducting studies in the Mojave Desert). Although the conditions that led to the fires in 2005 and 2006 may only occur every century or more (see below), this does not mean large areas will not burn in the future.

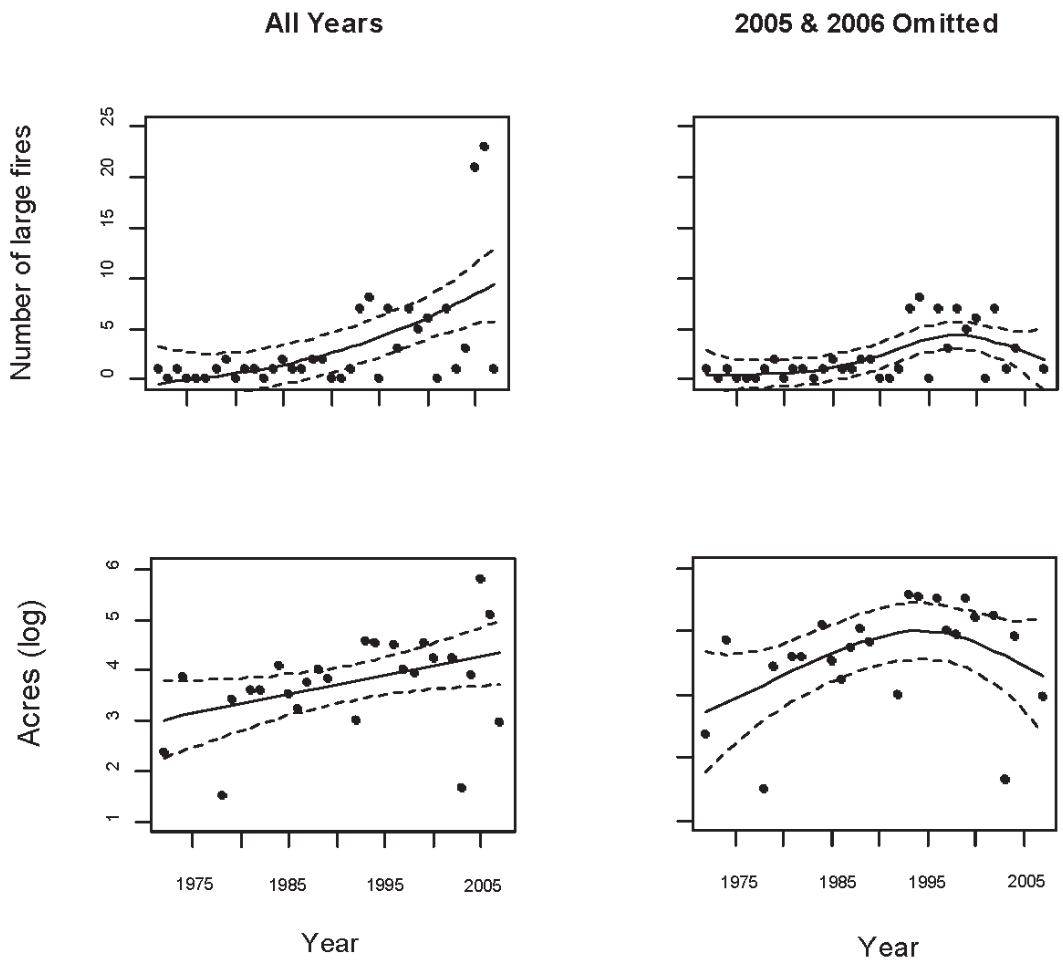

Figure 5.3—Patterns over time (1972-2007) for the number of large fires (≥1,000 acres), total area burned ($\log_{10}$ acres), mean fire size ($\log_{10}$ acres), and the proportion of burned area classified as high severity in Clark and Lincoln counties, Nevada. The shape of the relationship was derived from generalized additive models. Dotted lines are 95% confidence bands (modified with permission from McKinley and others, in press).

These trends must also be considered within the context of the 36-year sampling period (1972-2007). Previous studies from the Mojave Desert have reported both increasing numbers of fires (1980-1995, Brooks and Esque 2002) and no change in number of fires (1980-2004, Brooks and Matchett 2006). The differing conclusions of these two previous studies were entirely due to the different timespans of the datasets they used. As an extreme example, the conclusions published in Brooks and Matchett (2006) may have been very different if the data set included just 1 or 2 additional years (2005 and 2006). Thus, fire trends derived from only a few decades of data must be evaluated very cautiously, especially in a place like the Mojave Desert where fire occurrence is so episodic.

Temporal and Spatial Patterns of Burning

Evidence presented above from the historic and current fire record suggests that fire activity in southern Nevada is primarily associated with the warm (positive) phase of the multi-decadal Pacific Decadal Oscillation (PDO) cycle during which perennial fuels increase, and secondarily with the El Niño phase of the interannual El Niño-Southern

Oscillation (ENSO) cycle during which fine ephemeral fuels increase. It appears that the ENSO effect may not be sufficient alone to promote large fires, and may only kick in during the latter part of or soon after a multi-decadal period of high rainfall associated with the PDO (e.g. after 1993, and especially during 2005 and 2006). Although intentional burning by humans has at times added significantly to acres burned, these fires likely remain small when climatic conditions cause fuelbeds to be sparse.

Non-native annual grasses alter fire regimes worldwide through a process known as the grass/fire cycle (D'Antonio and Vitousek 1992; Brooks and others 2004). Species in the genera *Bromus* are specifically associated with changes in the temporal and spatial patterns of burning in upland areas of Mojave and Great Basin Deserts (Brooks 1999; Brooks and Matchett 2006; Brooks and Pyke 2001; Link and others 2006; Whisenant 1990). Although these species can undoubtedly alter fire regimes of southern Nevada, their influence is ultimately tied to the longer-term PDO and shorter-term ENSO cycles. Warm PDO phases are associated with exponential population growth of non-native annual grasses, such as that documented for *Bromus rubens* from the late 1970s through 1990 in southern Nevada (Hunter 1991). Increasing populations lead to high propagule production and dispersal into new areas (Brooks 2009), potentially increasing the regional scope of the grass/fire cycle. The El Niño ENSO phase is associated with years of extremely high rainfall that lead to episodic spikes in fuel loads created by invasive annual grasses and heightened fire hazards, especially in lower and middle elevation shrublands (Brooks and Matchett 2006). The hallmark of the grass/fire cycle is a landscape dominated by non-native annual plants, with low abundance of native woody species, and short fire return intervals. In southern Nevada, such landscapes are currently most common in Lincoln County (fig. 5.2), and fig. 5.4 illustrates how many of these landscapes look when burned.

Figure 5.4—A section of the 2005 Southern Nevada Fire Complex in the Tule Desert region of Lincoln County, Nevada. This is an area that had burned within the past few decades and was dominated by standing dead *Bromus* spp. biomass at the time of the fire (photo credit, Bureau of Land Management, Ely Field Office files).

One major factor that could decouple fire regimes from the PDO/ENSO cycles is related to the potential for the monsoon track to shift farther north and west from its current position at the southwest margin of the region (Hereford and others 2006). The vast majority of fires that occurred during the past few decades in the Mojave Desert have been associated with areas experiencing both high winter rainfall and high summer monsoonal rainfall and lightning, the former boosting fine fuel loads and the latter providing ignition sources and extreme wind conditions (J. Tagestad and others, in preparation). If the monsoonal track moves further into the Mojave Desert, then fires may follow, and if these fires move into areas where fire was historically infrequent and conditions are conducive to growth of non-native annual grasses, then a grass/fire cycle may emerge in those areas as well. In contrast, if the monsoon moves south and eastward, then fires may become less prevalent and the grass/fire cycle may wane as a significant land management threat.

Another factor that could affect future fire occurrence is increasing human population. More people likely mean more human ignitions. More people also mean more fossil fuel combustion that leads to increased rates of regional nitrogen deposition (see Chapter 2). Plant productivity, and therefore fuel production, especially non-native herbaceous fuels, are primarily influenced by precipitation and soil nitrogen levels (Brooks 2003; Rao and Allen 2010). Because elevated CO_2 increases water use efficiency of *Bromus* species, it has the potential to exacerbate the problem (Smith and others 2009; Chapter 2).

Alpine and Bristlecone Pine Ecosystems—These ecosystem types are located above 2,600 m in the Spring Mountains and the Sheep Range. Although lightning is relatively frequent in these ecosystems, fuels are sparse and continuity is low and most fires do not spread beyond single bristlecone pine (*Pinus longaeva*) or limber pine (*Pinus flexilus*) trees (Fryer 2004; Johnson 2001).

Mixed Conifer, Piñon and Juniper, and Sagebrush Ecosystems—These ecosystem types are predominantly located in the Spring and Sheep mountain ranges at elevations between 1,200 and 3,200 m. Lightning occurs frequently in these ecosystem types. These shrublands, woodlands, and forest stands are dominated by sagebrush (*Artemisia* spp.), chaparral shrubs, juniper (*Juniperus* spp.), piñon pine, and small stands of ponderosa pine, and have fire patterns that are driven mostly by long-term, decadal to century-scale PDO patterns of rainfall and perennial fuel accumulation (Brooks and Matchett 2006; Littell and others 2009). Continuity and amounts of native perennial fuels alone can be sufficient to carry fire under extreme fire weather conditions (i.e. high temperatures and wind, and low relatively humidity), and fire is a part of the natural disturbance regime. Years of high rainfall can produce additional herbaceous fuels that enhance fuelbeds and potentially further facilitate fire spread, but they also can increase live perennial fuel moisture levels that can decrease spread rate. Thus, the net effect of high rainfall on fire behavior at high elevations can be hard to predict.

Blackbrush/Shadscale and Mojave Desert Scrub Ecosystems—Fire patterns in these low and middle elevation shrublands, dominated by saltbush (*Atriplex* spp.), creosotebush (*Larrea tridentata*), and blackbrush, are affected most by short-term, interannual patterns of rainfall and ephemeral herbaceous fuels associated with the ENSO cycle (Brooks and Matchett 2006). Fire frequency is higher at the interface with higher elevation ecosystem types where fires often start and spread. Continuity and amounts of perennial fuels are insufficient alone to carry fire; pulses of herbaceous fuels following periods of high rainfall are necessary to fill the interspaces between perennials and allow fire to spread, and fire is not a major part of the natural disturbance regime. Fuels created by non-native annual grasses are more persistent than those from native forbs, and as such contribute more to fire spread potential (Brooks 1999). Altered fire regimes from invasive annual grasses (i.e. the grass/fire cycle) are most prevalent in these ecosystem types (Brooks and Matchett 2006).

Riparian and Spring Ecosystems—Fire patterns in riparian zones depend on where they occur on the southern Nevada landscape. The major riparian corridors that occur along the rivers of southern Nevada are situated at lower elevations within upland areas dominated by Mojave Desert scrub. Pre-historic Holocene and historic conditions of perennial grasses and shrubs as an understory beneath canopies of towering cottonwoods supported periodic low to moderate intensity surface fire that spread very rarely into the sparse surrounding uplands (Dwire and Kauffman 2003). Thus, fire is part of the natural disturbance regime, although at moderately long-return intervals (Petit and Naiman 2007). Indigenous humans also used fire during the Holocene to clear riparian terraces for agricultural purposes and to promote growth of basketry materials (e.g. rushes, reeds, and milkweed). The spread of fires was likely limited by barriers of vegetation gaps, standing water, and narrow bottleneck points along the floodplain where gaps occurred in the more consistently dry upland benches. Following the invasion of saltcedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*), ladder fuels and overall fuel loads increased, resulting in a more frequent and high intensity crown fire regime (Busch and Smith 1993; Lovich and others 1998; Busch 1995). In addition, the invasion of non-native annual grasses in both the riparian and surrounding uplands provides more continuous fuelbeds that likely allow fires to reach greater size now than in the past. In contrast, fire patterns in higher elevation riparian zones embedded within blackbrush, sagebrush, and mixed conifer uplands are affected more by the fire regimes of the surrounding ecosystem types than by the riparian fuels themselves. Similarly, patterns of natural fires at isolated springs are controlled by the fire regime of the surrounding vegetation. There is also evidence that indigenous humans, and then later Anglo settlers, used fire to reduce vegetation cover to facilitate hunting opportunities and enhance water flow rates.

Fire Effects

Fire and Fire Regime Characteristics

Although land management actions focus on individual fires, the ultimate influence of fires across landscapes and over time is attributed to fire regimes. The type (ground, surface, or crown fire), frequency (i.e., return interval), intensity (heat released), severity (ecological response), size, spatial complexity, and seasonality of fire define the fire regime within a given geographic area or vegetation type. Over very long time periods, the prevailing fire regime conditions can have a strong influence on the evolution of species. When fire regimes are altered (e.g. by plant invasions or land management practices) the recovery of the resident species following fire can be compromised and landscapes can be converted to new vegetation types that are better adapted to the new fire regime (Brooks 2008). Thus, the management of individual fires must be placed within the broader context of how they will affect the management of fire regimes.

It must also be understood that all fires are not the same. Unfortunately, the vast majority of published studies on fire effects report the effects of “fire” as if it is a univariate factor. Multiple aspects of fire behavior, seasonality, and spatial pattern can influence the effects of fire. Fire behavior can vary widely based on fuel, weather, and ignition characteristic (van Wagtendonk 2006). In addition, rate of spread, residence time, flaming zone depth, fuel energy content, fire type (ground, surface, crown), and other factors can influence fire intensity (energy released), which is a primary factor associated with fire severity (effects on the environment). The season during which a fire occurs can also significantly influence the survivorship of individuals and the responses of populations and communities (Fites-Kaufman and others 2006). Fires that occur when plants and animals are reproducing are often most damaging because they can potentially eliminate recruitment for the year (Fites-Kaufman and others 2006; Shaffer and others 2006and others). Survivorship of perennial plants is also lower when they are actively growing because below-ground carbohydrate reserves are depleted, whereas survivorship is higher when they are dormant because below-ground reserves are at their peak (Pyke and others 2010). Survivorship of animals may also be higher when they are hibernating or otherwise located below-ground and away from intense heating (Shaffer and others 2006). Fires also differ based on their spatial patterns of burning. Heterogeneous burns with unburned islands and abundant areas of low fire severity provide sources of propagules to recolonize burned areas, thus decreasing recovery time to pre-burn conditions. Homogenous complete burns with few unburned islands and uniformly moderate to high fire severity are often slower to recover to pre-burn conditions unless the resident species are fire adapted. Thus, one should never assume similar ecological responses to fire per se, but instead should expect similar responses among fires of similar characteristics considering the factors discussed above plus additional site and ecosystem type factors discussed below.

Site Characteristics

The predominant life forms of plants and animals at a site can influence their overall responses to fire. Plant life forms with meristem (perennating) tissue located aboveground tend to have the highest mortality rates because that life sustaining tissue occurs close to the flames and smoldering fuels and is exposed to the highest temperatures during fires (Pyke and others 2010). The general exception is when meristems are located far aboveground and away from flames with limited ladder fuels to carry fire up into their vicinity (e.g. some palm trees). In contrast, plant life forms with meristem tissue or dormant seeds located at or below the surface of the mineral soil tend to have the lowest mortality rates because they occur where temperatures are lowest during fires (Brooks 2002). Animal taxa that are more vagile or arboreal can avoid fire-induced mortality better that than those that are more sedentary (Shaffer and others 2006). However, susceptibility to mortality among the very young is more a factor of where their nests or dens are located, than the vagility of their species. For example, birds that are extremely vagile are more adept at avoiding extreme temperatures from fire, but nestlings have high mortality rates because they are typically situated within flammable nests located in the heart of the fuelbed and combustion zone. Thus, effects of fire on plants and animals can vary depending on the types of species. However, the longer term effects of fire on subsequent growing conditions or habitat characteristics are generally thought to have even greater effects on populations and communities.

The effects of fire can vary greatly depending on the elapsed time since the previous fire. Excessively short fire return intervals that do not allow time for individual plants to recover or new individuals to establish and reach reproductive maturity can reduce the abundance of maladapted species at a site. This is a significant mechanism by which vegetation type conversions occur as part of the grass/fire cycle (Brooks 2008). Excessively long fire return intervals that allow the abnormal accumulation of surface and ladder fuels can alter fire behavior and similarly reduce the abundance of maladapted species.

Relative concern about fire effects within a given area often hinges on the current or potential future dominance by invasive non-native plants. These species alone are often undesirable from a natural resource management perspective, but when their effects on fuelbeds and fire regimes are considered they often become significant fire management concerns as well (Brooks 2008).

Other factors such as historical and current land uses and weather patterns (especially precipitation) can influence the effects of fire at a particular site. Historically intense land use disturbance such as livestock grazing can produce similar effects as antecedent fire, including reduced perennial cover, reduced species diversity, and increased dominance by invasive plants. Thus, the landscape response to fire in these areas may be similar to that associated with short fire return intervals. Similarly, periods of drought before burning may reduce a species' abundance and resource reserves, and drought following burning may hinder its post-fire survival and/or establishment rates, in both cases rendering the species less tolerant.

Ecosystem-Specific Effects

Alpine and Bristlecone Pine Ecosystems—Bristlecone pine and limber pine trees are thin barked and lack adaptations to fire. These tree species can only survive low intensity surface fire or lightning strikes that do not result in complete girdling of the cambial tissue (Brooks and Minnich 2006; Fryer 2004; Johnson 2001). Many of these long lived trees have multiple scars indicating survival following lightning strikes that occurred repeatedly over their 1,000+ year lifespan, but apparently these ignitions did not spread to engulf trees and probably did not spread beyond the trees struck. Sparse surface fuels result in small fires and low fire intensities that likely had limited effects on small woody shrubs and herbaceous species in the understory. As a consequence of a warming climate, invasive annual grasses may move upslope and present fine fuel management problems in the future, and fire may facilitate this process (table 5.1).

Mixed Conifer Ecosystem—Interior ponderosa pine (*Pinus ponderosa* var. *scopulorum*) is one of the most fire-adapted conifer species in North America. Its adaptations include open crowns, self-pruning branches, thick bark, thick bud scales, tight needle bunches protecting meristems, high foliar moisture, and a deep rooting habit (Howard 2003). Widely spaced older trees display higher fire survival rates than more densely packed and younger trees. White fir (*Abies concolor*) is a thin barked tree species with branches and foliage from top to bottom that make it highly vulnerable to mortality from fire. Mountain mahogany (*Cercocarpus* spp.), Gambel oak (*Quercus gambelii)*, manzanita (*Arctostaphylos* spp.), and snowberry (*Symphoricarpos albus*) are all elements of the interior chaparral vegetation type often found in the understory of mixed conifer sites. All but mountain mahogany are extremely tolerant of fire (Brooks and others 2007).

Long intervals without fire allow the accumulation of understory and ladder fuels that generally result in higher severity crown fires that can threaten the persistence of isolated mixed conifer stands in southern Nevada (table 5.1). High severity events are often followed by initial dominance by understory chaparral species that may persist as an alternative vegetation type if conditions are no longer conducive to pine establishment. A warming climate may exacerbate this process.

Table 5.1—Typical fire management concerns associated with site and fire characteristics and guidelines for appropriate fuels management, fires suppression, and Emergency Stabilization and Rehabilitation (ES&R) actions in the major ecosystem types of southern Nevada.

Ecosystem type burned	Typical fire management concerns	Guidelines for appropriate management actions
Alpine and Bristlecone pine	*Fire* – Fires that burn more than a few acres and/or completely consume multiple trees. *Site* – Tree mortality and increased down woody fuels. Low tree recruitment post-burn and increased dominance of *Bromus* spp. could alter fire regime.	*Fuels* – Rarely warranted *Suppression* – Rarely warranted except to protect small and/or isolated stands if excessive fire spread is likely. *ES&R* – Rarely warranted unless perhaps to control invasive plants or following excessively large and severe fires.
Mixed conifer	*Fire* – High severity crown fire. Long fire return interval that allow excessive fuel accumulation. *Site* – Heavy surface and ladder fuel loads could lead to high severity crown fire. Wildland/urban interface (WUI) limits its fire management options.	*Fuels* – Periodically warranted in the WUI, egress routes, and where surface and ladder fuels have accumulated increasing the risk of large fires. Mechanical treatments may be initially needed, but subsequent treatments should utilize low severity, surface prescribed fire. Follow-up control of invasive plants may be needed during the first few post-treatment years. *Suppression* – Rarely warranted except in the WUI, egress routes, and where surface and ladder fuels have accumulated increasing the risk of large fires. Low severity surface fire with minimal crowning should be allowed as a natural fire regime component for vegetation management benefits. Follow-up control of invasive plants may be needed during the first few post-fire years. *ES&R* – May be warranted to stabilize slopes near the WUI or following excessively large and severe fires.
Piñon and Juniper	*Fire* – High severity crown fire. Short fire return interval that does not allow re-establishment of shrub understory or trees. *Site* – Dominant invasive annuals. Short fire return intervals (≤60 years).	*Fuels* – Rarely warranted except in the WUI and egress routes and in shrub-dominated areas exhibiting tree expansion. Prescribed fire may be the more cost-effective over large areas. Follow-up control of invasive plants may be needed during the first few post-fire years. *Suppression* – Warranted in the WUI, and to prevent large, high severity crown fires, but low severity surface fire in areas of undesirable tree expansion may be allowed as a natural fire regime component for vegetation management benefits. Follow-up control of invasive plants may be needed during the first few post-fire years. *ES&R* –Warranted to reestablish native plant community characteristics and to control invasive plants if inadequate perennial herbaceous species and shrubs exist for recovery. Herbicides may allow short-term control but long-term control requires a wider array of management actions, aerial seedings have low to moderate establishment rates.
	Fire – Large, homogenous fires *Site* – Dominant invasive annuals. Short fire return intervals (≤40 years).	*Fuels* – Rarely warranted, except in the WUI and egress routes and perhaps to increase the competitive ability of perennial herbaceous species and prevent invasive annual grass dominance. Prescribed fire rarely warranted except perhaps to create a stand-age mosaic in very old late-successional contiguous stands. Follow-up control of invasive plants may be needed during the first few post-treatment years. *Suppression* – Warranted in the WUI, and in areas with low abundance of perennial herbaceous species and high risk of conversion to annual invasive grasses. *ES&R* – May be warranted to reestablish the native community and to control invasive plants if inadequate perennial herbaceous species and shrubs exist for recovery. Herbicides may allow short-term control but long-term control requires a wider array of management actions, aerial seedings have low to moderate establishment rates.

(continued)

Table 5.1—(Continued).

Ecosystem type burned	Typical fire management concerns	Guidelines for appropriate management actions
Blackbrush/ shadscale	*Fire* – Large, homogenous fires. *Site* – Dominant invasive annuals. Short to moderate fire return intervals (≤100 years).	*Fuels* – May be warranted to reduce non-native annual grass fuels, perhaps centered on roads, to facilitate fire suppression. Follow-up treatments may be needed on an annual basis. *Suppression* – Warranted under most circumstances. Prescribed fire is never warranted except perhaps for experimental purposes. *ES&R* – May be warranted to reestablish the native community and to control invasive plants if inadequate perennial herbaceous species and shrubs exist for recovery. Herbicides may allow short-term control of non-native annual grasses, but long-term control requires a wider array of management actions including reestablishment of native perennial species. Aerial seedings have low establishment rates. Livestock closures are necessary post-fire to facilitate recovery of native perennial plants.
Mojave desert scrub	*Fire* – High severity, large, homogenous fires *Site* – dominant invasive annuals, short to moderate fire return intervals (≤100 years)	*Fuels* – May be warranted to reduce non-native annual grass fuels, perhaps centered on roads, to facilitate fire suppression. Follow-up treatments may be needed on an annual basis. *Suppression* – Warranted under most circumstances. Prescribed fire is never warranted except perhaps for experimental purposes. *ES&R* – May be warranted to reestablish the native community and to control invasive plants if inadequate perennial herbaceous species and shrubs exist for recovery. Herbicides may allow short-term control but long-term control requires a wider array of management actions including reestablishment of native perennial species. Aerial seedings have low establishment rates. Livestock closures are necessary post-fire to facilitate recovery of native perennial plants.
Riparian and Spring	*Fire* – High severity crown, large homogeneous fire. *Site* – Dominant invasive perennials. Heavy surface and ladder fuel loads could lead to high severity crown fire. Flood disturbance may have larger effects than fire especially in unregulated riverine systems (e.g. Virgin River)	*Fuels* – May be warranted to reduce non-native tamarisk and Russian olive fuels to reduce high severity fire. Prescribed fire may be the most cost-effective way to reduce invasive plan biomass over large areas, but follow-up control of surviving invasive plants will always be needed during the first few post-fire years, potentially followed by native plant revegetation. *Suppression* – May be warranted in the WUI or where tamarisk and native vegetation are intermixed, otherwise low to moderate severity fire may be allowed where native fuels predominate as a natural fire regime component for vegetation management benefits, or moderate severity fire may be allowed where tamarisk dominates to remove above-ground biomass as a prelude to other tamarisk control actions such as herbicide treatments followed by native plant revegetation. *ES&R* – May be warranted to control tamarisk, herbicides may allow short-term control but long-term control requires a wider array of management actions including native plant revegetation which may be accomplished by seedings or outplantings.

Piñon and Juniper Ecosystems—Single-leaf piñon (*P. monophylla*), Utah juniper (*Juniperus osteosperma*), Rocky Mountain juniper (*Juniperus scopulorum*), and western juniper (*Juniperus occidentalis*) are relatively thin barked and contain ladder fuels that facilitate consumption of their entire canopies, so they have low tolerance of fire (Brooks and Minnich 2006). However, stands can re-establish within about a century. Understory vegetation, such as interior chaparral, at higher elevations is very tolerant of fire that occurs at moderate return intervals. Sagebrush is intolerant of fire and rabbitbrush (*Chrysothamnus* spp.) is fire tolerant, but sagebrush communities can reestablish given longer fire return intervals (25-50 years) and relatively small fire size.

Excessively long intervals between fires may lead to canopy closure, seedbank depletion, fuel accumulation, and high fire severity, all of which reduce piñon-juniper resilience to fire (Allen and others 2008; Miller and others 2005, 2008) (table 5.1). If invasive annual grasses are present, they have the potential to dominate post-fire landscapes leading to short intervals (~2-5 years) between fires that do not allow time for trees to establish and grow to maturity.

Sagebrush Ecosystem—The sagebrush ecosystem is dominated by several different shrub species that are typically killed by fire, including big sagebrush (*Artemisia tridentata tridentata, A. tridentata wyomingensis* and *A. tridentata vaseyana*), low sagebrush (*A. arbuscula*), Bigelow sagebrush (*A. bigelovii*), and black sagebrush (*A. nova*). Other species such as rabbitbrush, snakeweed (*Gutierrezia sarothrae*), spiny hopsage (*Grayia spinosa)* and cliffrose (*Purshia neomexicana*) can resprout if only partially burned, and perennial grass species including wheatgrass (*Agropyron* spp.), bluegrass (*Poa* spp.), and needlegrass (*Achnatherum* spp.) usually survive topkilling.

High abundance of invasive annual grasses may lead to short fire return intervals that do not allow sufficient regeneration times for native sagebrush species to persist (table 5.1). These conditions also may lead to large homogenous fires that hinder seed dispersal of native perennial species back into burned areas and result in intense competition of the invaders with native seedlings for available resources.

Blackbrush/Shadscale Ecosystem—Blackbrush is the dominant shrub species in areas with shallow limestone-derived soils, and this species is easily killed and very slow to re-establish following fire (Brooks and others 2007). Shadscale (*Atriplex confertifolia*) and budsage (*Artemisia spinescens*) dominate on heavy, rocky soils and also are killed by fire. Other subdominant shrubs species include cliffrose, Mormon tea (*Ephedra* spp.), snakeweed, wolfberry (*Lycium* spp.), and spiny hopsage and all can resprout to varying degrees.

Similar to sagebrush ecosystems, high abundance of invasive annual grasses may lead to fire return intervals that that are shorter than the regeneration times of native blackbrush species (table 5.1). Invasive annual grass dominance may also lead to large homogenous fires that hinder seed dispersal of native perennial species back into burned areas and result in competition of the invaders with native seedlings for available resources. Although perennial plant cover may approach unburned conditions within the first four post-fire decades, species composition typically does not (Abella 2009; Brooks and others, in prep.; Engel and Abella 2011).

Mojave Desert Scrub Ecosystem—Creosotebush is the dominant species in upland areas and saltbush species are dominant in alkaline soils of lowland basin areas. Bajadas, the most common landform, are dominated by creosote bush and white bursage (*Ambrosia dumosa*); subdominants include desert thorn (*Lycium andersonii*), bladder sage (*Salazaria mexicana*), indigo bush (*Psorothamnus fremontii*), blackbrush, brittlebush (*Encelia farinosa*), and burro bush (*Hymenoclea salsola*). Most of these species have the capacity to survive fire if more than half of their aboveground biomass is left unburned (Brooks and Minnich 2006). Yucca species such as Joshua tree (*Yucca brevifolia*) and Mojave yucca (*Yucca schidigera*) often survive burning, but the Joshua tree often dies within the first few years after fire due to drought and herbivory stress on resprouts (DeFalco and others 2010).

Similar to sagebrush and blackbrush ecosystems, high abundance of invasive annual grasses may lead to excessively short fire return intervals that do not allow native Mojave desert scrub species to reestablish (Brooks 2011) (table 5.1). Although perennial plant cover may approach unburned conditions within the first four post-fire decades, species composition typically does not (Abella 2009; Brooks and others, in prep.; Engel and Abella 2011).

Riparian and Spring Ecosystems—Historical fire coupled with periodic flooding has resulted in riparian plant species adaptations that confer some degree of tolerance to fire. The invasion of saltcedar, Russian olive and invasive annuals have increased fuel loads, created ladder fuels where little previously existed, and resulted in the potential for larger, more intense, and more frequent fires than occurred historically (Dwire and Kaufman 2003) (table 5.1). Although some native riparian species may not survive this altered fire regime (e.g. cottonwood trees), others are clonal and are capable of surviving fire and resprouting (e.g., willows). However, the altered fire regime can also be accompanied by vigorous post-fire resprouting and seedling establishment of non-natives creating an intense competitive environment that can suppress native species (Dudley and Brooks 2006). Thus, the net effects on natives can be significant even though they may be generally tolerant of fire and other disturbances.

The degree to which fire may have affected the evolution of plant species in the vicinity of springs is related to the fire regime of the surrounding vegetation type. Current species assemblages may also have been affected by historical fire use by indigenous humans. In addition, some spring sites are dominated by non-native saltcedar, which can increase its dominance following fire if it is not actively managed.

Management Actions

Pre-Fire Fuels Management

Livestock grazing can reduce cover of perennial vegetation in the Mojave Desert (Brooks and Berry 2006), which may have led to reduced fuel continuity and landscape-scale flammability of fuelbeds in mid to upper elevation ecosystem types such as blackbrush/shadscale, sagebrush, and piñon and juniper after grazing began in the late 1800s (Brooks and Minnich 2006). Intensive grazing associated with livestock watering sites also can be associated with reduced cover of red brome (*Bromus rubens*) (Brooks and others 2006), which is the primary fine fuel that carries fire in the Mojave Desert (Brooks 1999; Brooks and Minnich 2006). Although livestock grazing has been shown to reduce flame lengths and fire spread rates in the Great Basin (Diamond and others 2009), it may only be a significant factor in early successional vegetation stands dominated by herbaceous species (Launchbaugh and others 2008). In addition, the intensity of grazing required to significantly alter fire behavior may actually facilitate the long-term dominance of non-native grasses that are often one of the most significant fuel management concern in the Mojave and Great Basin Deserts (Brooks and others 2007; Knick and Connelly 2011; Wisdom and others 2005). Moreover, extensive grazing over many decades on predominantly BLM lands in Lincoln County, Nevada, is coincident with the largest expanses of burned landscapes in the entire Mojave Desert (Brooks and Matchett 2006).

In contrast, nearby lands within the Desert National Wildlife Refuge to the west of these BLM lands have been protected from livestock grazing since the refuge was created in the 1930s and do not contain evidence of widespread fires (Brooks and others 2007).

Following the 2005 Mojave Desert fires, there was serious discussion about using herbicides to manage invasive annual fuels along the margins of dirt and paved roads to create fuelbreaks and reduce the window of opportunity for fires to spread and become large (e.g. Brooks and others 2005) (table 5.1), however it has rarely been implemented in part because its efficacy has yet to be evaluated in southern Nevada. Grass-specific herbicide used in small plot experiments has been effective at controlling the non-native Mediterranean split grass (*Schismus* spp.) promoting post-fire establishment of less flammable native forb species in the northwestern Sonoran Desert (Steers and Allen 2010). Even if herbicides are effective in the short-term, they would need to be applied at least every few years as a part of a regular maintenance program to maintain fuel-breaks. If this is done, then these corridors of managed fuels can be used to facilitate fire suppression efforts and potentially reduce the frequency of fire starts from vehicles travelling along roads that parallel treated areas. Mechanical thinning of fuels is a viable management option at very localized scales, due to relatively high cost and potential for undesirable side-effects. For example, narrow (e.g. <10 m wide) managed fuel zones along the margins of roads may be appropriate to minimize anthropogenic ignitions in sagebrush, blackbrush/shadscale, and Mojave Desert scrub ecosystems (table 5.1). Mechanical thinning of understory fuels in old mixed conifer stands may be necessary prior to the reintroduction of low to moderate intensity surface fires. This same approach of understory thinning followed by low to moderate intensity fire can be used in areas exhibiting piñon and juniper expansion into areas where the presence of these trees is not desirable. If fuel beds are already conducive to low to moderate intensity fire, then fire alone should be the preferred alternative.

It should be noted that fuelbreaks, thinnings, or any other type of fuels reduction project can also have negative effects, such as facilitating the spread of invasive plants (Merriam and others 2006). Accordingly, the cost and ecological effects associated with the creation and maintenance of managed fuel zones should always be weighed against their efficacy in slowing or stopping fires, the additional costs and efficacy of suppression efforts where fuelbreaks are not present, and the ecological effects of increased burned areas where fires attain larger size due to the absence of managed fuel zones.

Fire Suppression

Clearly, the most effective way to protect the majority of the low to mid elevation shrubland ecosystem types of southern Nevada from fire damage is to prevent fires from starting and/or spreading into large areas. This requires aggressive fire suppression efforts that may include aerial water and retardant drops and off-road travel by suppression equipment (e.g., engines, dozers). The use of these tactics should not be taken lightly, because phosphate-based retardants may promote dominance by invasive annual plants (Besaw and others 2011; Brooks and Lusk 2008) and off-road travel, especially the use of dozers, can damage both natural and cultural resources. Thus, the potential negative effects of aggressive fire suppression must be weighed against the potential negative effects of fires spreading to cover more area and the ability to mitigate negative suppression effects immediately following fires.

In contrast, wildland fire use may be the preferred alternative rather than fire suppression at higher elevation and tree dominated ecosystems in southern Nevada (table 5.1). Fire spread potential is minimal in the alpine and bristlecone pine ecosystem type, but if these conditions change in the future (e.g. due to climate change or plant invasions) then fire suppression may be necessary. Periodic low to moderate intensity fire is a desirable natural ecosystem process in the mixed confer ecosystem type, and to a lesser degree in the piñon and juniper and sagebrush ecosystem types. Fire suppression in these types should only be limited to what is required to protect the wildland-urban interface (WUI) and/or to limit the spread of fire into excessively large fires that could threaten the persistence of relatively small isolated vegetation stands.

Post-Fire Emergency Stabilization and Rehabilitation

Aerial seeding is often used to increase the recovery potential of native vegetation and decrease the dominance of invasive annual plants on post-fire landscapes in southern Nevada. Although very few studies exist from the Mojave Desert, numerous studies have been published regarding post-fire seeding in the Intermountain West. These studies indicate that establishment success of seeding projects depends on precipitation (Wirth and Pyke 2009) and that very high seeding rates may be required at the lower end of the precipitation spectrum (Thompson and others 2006). They also suggest that successful seeding can lead to lower invasive species abundance (Evans and Young 1978; Goodrich and Rooks 1999; Thompson and others 2006; Wirth and Pyke 2009), although unsuccessful seeding efforts may actually increase invasive plant abundance (Ratzlaff and Anderson 1995). Studies near completion from the Mojave Desert provide only scant evidence of establishment success following aerial seeding of post-fire landscapes in southern Nevada (Brooks and others. in prep.). These studies also indicate that, in general, perennial seedlings only appear in measureable numbers where both rainfall is high and density of invasive annual plants is low. Thus, seedings alone may not be the correct tool to control invasive annual plant populations in areas that they already dominate. Another study near completion reports that establishment of seeded species can be improved where mechanical pitting is done using hand tools and seeds are broadcast by personnel on the ground (DeFalco and others, in prep.). In general, aerial seedings have very low establishment rates in southern Nevada and Emergency Stabilization and Rehabilitation (ES&R) resources may be better applied in other ways (table 5.l).

Temporary closures for livestock grazing are also implemented to provide native perennial vegetation a chance to recover following fires. These closures typically last only a few years, and it is unclear if they are beneficial. They also appear difficult to enforce, as livestock are often observed within closure areas (Matthew Brooks, personal observation of post-fire landscapes during the winter following the 2005 fire season, Lincoln County, Nevada). Survival of residual native plants may be enhanced by protection from post-fire grazing, especially in blackbrush/shadscale and Mojave Desert scrub ecosystems where species generally have low capacity for post-fire recovery (table 5.1).

Knowledge Gaps

Understanding Fire Histories

A better understanding of fire histories of southern Nevada ecosystem types can be used to develop more effective management plans for these areas. Specific studies targeting key ecosystem types and locations are needed to test current hypotheses regarding assumed historic fire frequencies. These include dendrochronology studies of the mixed conifer zone in the Spring Mountains, and soil stratigraphy studies using charcoal lenses as proxies for fire events within watersheds dominated by single ecosystem types.

Climate and Fire Size and Frequency

Routine evaluations of the relationship of climate to fire size and frequency and how this relationship might change with climate warming are needed to develop effective fire management strategies. Precise descriptions of spatial and temporal patterns of burning only span a few decades of comprehensive records (e.g., agency reports and satellite imagery). Conclusions about fire trends can vary widely depending upon which time interval one evaluates within the current record. Re-evaluation of these data should be done at regular intervals (e.g., 5 year) to test the robustness of the current hypotheses regarding short-term ENSO and longer-term PDO effects on fire regimes.

Fire Effects on Plant Species and Vegetation Types

The effects of fire on plant species and vegetation types must be more thoroughly understood before predictive models can be useful to management. Within each ecosystem type the various effects of fire, fire regime, and local site characteristics need to be investigated further. This will require intensive data from numerous fires, and possibly the use of experimental fires. Even less information is available regarding the effects of fire on animals, but because so many sensitive species are associated with particular ecosystem types, a full understanding of fire effects on animals can only be realized after a more complete understanding of vegetation responses.

Post-Fire Management

Additional information is needed regarding appropriate management actions after fire. It is well established that aerial seedings of post-fire landscapes have very low establishment rates. However, much less is understood about other management actions designed to reestablish native vegetation. Also, little is known about the effects of post-fire grazing. For example, how does the duration and intensity of post-fire grazing by livestock affect vegetation resilience to fire and expansion of invasive annual grasses? How effective is livestock grazing at managing fuels created by invasive annual plants?

Fire Suppression Impacts

Considering that fire suppression may be the most effective fire management tool in low to mid elevation ecosystem types, there is a need to better understand the relative impact, both negative and positive, of aggressive fire suppression tactics (e.g. retardant drops and off-road travel) versus allowing fires to spread and burn more areas.

Semi-Arid Ecosystem Response to Wildfire

Because tree infilling and growth are ongoing processes in higher elevation conifer and piñon and juniper ecosystems, information is needed on the response of these semi-arid ecosystems to wildfire and fire and fuels treatments. Information also is needed on how fire and fuels treatments can be used for restoring and maintaining landscape heterogeneity of these diverse ecosystems.

Management Implications

Important take-home messages for land managers are that (1) the effect of an individual fire event should be evaluated within the context of the ecosystem type in which it occurs, the characteristics of the fire, and characteristics of the site; (2) fire suppression is the most cost effective way to manage fires across most of southern

Nevada, except in a few ecosystem types where fire is part of the natural disturbance regime and wildland fire use should be an option to achieve management objectives; and (3) the current range of post-fire mitigation tools used are either ineffective or their effectiveness is poorly documented. Like all aspects of land management, fire management must ultimately be placed in the broader context of all the other factors associated with managing landscapes in southern Nevada. In some cases decisions may need to be made regarding where to allocate limited resources, and in other cases conflicting objectives may need to be resolved between fire management and those focused on other management topics (especially natural and cultural resources). The information contained in this chapter should help all parties better understand issues associated with fire management.

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