Greater Sage-Grouse & Wind Energy Development

A Review of the Issues

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EXECUTIVE SUMMARY

At the request of the Renewable Northwest Project, this report summarizes major issues associated with wind energy development and greater sage-grouse in the western U.S. Specifically, we developed a brief summary of the major threats to sage-grouse, reviewed the pertinent literature on response of sage-grouse and similar species to wind energy development, provided several stipulations to avoid and minimize impacts to sage-grouse when planning and developing wind energy facilities, and provided a framework for mitigation though habitat enhancements that might be used to offset potential impacts to sage-grouse associated with wind energy development.

Sage-grouse are native game birds closely tied to sagebrush dominated habitats in the western United States and Canada. Sage-grouse originally occurred in 12 states and three Canadian provinces, but have been extirpated from Nebraska and British Columbia. Due primarily to alteration or elimination of sagebrush habitat, sage-grouse currently occupy only approximately 56 percent of their pre-European distribution, and overall abundance has decreased by up to 93 percent from presumed historic levels.

Based on a review of the literature, the current primary threats to sage-grouse include habitat loss, with fire and invasive plant species being major current sources of habitat loss; habitat fragmentation, especially as it relates to energy development; improper livestock grazing; West Nile virus; and climate change. Recreational hunting is currently not considered a threat to maintaining healthy sage-grouse populations.

Wind energy development may affect sage-grouse directly through collisions with wind energy facilities such as wind turbines and transmission lines, although it is unlikely that direct mortality would ever be substantial enough to cause population declines in most cases. The primary concern with wind energy development in sage-grouse habitats is indirect impacts that may occur if wind energy development displaces grouse and/or reduces survival.

Unfortunately, well-designed studies examining the potential impacts of wind energy development on sage and related prairie grouse species have not been completed, although telemetry studies are ongoing to evaluate effects of wind energy development on greater prairie chickens in Kansas and greater sage-grouse in Wyoming. Both of these studies as well as other observational studies indicate that sage-grouse and prairie grouse may continue to use habitats near wind-energy facilities; however, research conducted on greater sage-grouse response to oil and gas development has found that population declines due to oil and gas development may not occur until anywhere from two to ten years post-construction. Therefore, long-term data will likely be required to fully assess how sage-grouse respond to wind energy development.

This report provides several stipulations that may be used to avoid or minimize impacts when selecting sites for wind energy development as well as during the construction and operational phases of a wind energy facility. Finally, we provide a framework for developing a holistic sage-grouse management or mitigation plan for a given project area, primarily through habitat enhancements, and provide background information for each step for use by those developing the plan.

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Introduction and Background

The Renewable Northwest Project asked WEST, Inc. to summarize major issues associated with wind energy development and greater sage-grouse (*Centrocercus urophasianus*; hereafter referred to as sage-grouse) in the western U.S. Specifically, we were asked to develop a brief summary of the major threats to sage-grouse, review the pertinent literature on response of sage-grouse and similar species to wind energy development, summarize stipulations useful for avoiding and minimizing impacts to sage-grouse when planning and developing wind energy facilities, and provide a framework for mitigation though habitat enhancements that might be used to offset potential impacts to sage-grouse associated with wind energy development. The section entitled “Development of a Holistic Management or Mitigation Plan” was prepared by Dr. Matt Holloran of Wyoming Wildlife Consultants, LLC; the remainder of the document was prepared by Mr. Greg Johnson of WEST, Inc.

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Threats to Sage-Grouse

Sage-grouse are native game birds closely tied to sagebrush (*Artemisia* spp.) dominated habitats in the western United States and Canada. According to Schroeder et al. (2004), sage-grouse originally occurred in 12 states and three Canadian provinces, but have been extirpated from Nebraska and British Columbia. Due primarily to alteration or elimination of sagebrush habitat (Aldrich 1963; Connelly and Braun 1997), sage-grouse currently occupy only approximately 56 percent of their pre-European distribution (Schroeder et al. 2004), and overall abundance has decreased by up to 93 percent from presumed historic levels (Braun 2006). These decreases are the result of habitat loss, fragmentation, and degradation.

Because of limited survey effort, there are no rigorous estimates of sage-grouse population sizes prior to the late 1950s (Braun 1998). It is thought that sage-grouse populations in the 1960s and 1970s were two to three times higher than current populations (Connelly et al. 2004). Based on lek count data, significant long-term population declines occurred throughout the species range between 1965 and 2003. The annual rate of decline was higher from 1965 to 1985 (–3.5 percent) than it was from 1986 and 2003 (–0.37 percent). Because of these declines, there is still significant concern over the long-term viability of sage-grouse because of continuing loss and fragmentation of habitat as well as new threats such as West Nile Virus (Connelly et al. 2004).

From 1999 through 2003, the U.S. Fish and Wildlife Service (USFWS) received eight petitions to list various population segments, potential subspecies, or the entire species as threatened or endangered under the federal Endangered Species Act. In 2004, the USFWS determined that three of the petitions to list the greater sage-grouse as threatened provided substantial information that listing might be warranted, and a comprehensive range-wide status review was initiated. In 2005, the USFWS determined that the sage-grouse was "not warranted" for listing (Idaho Sage-grouse Advisory Committee [ISAC] 2006). In December 2007, the USFWS was required by court order to conduct another status review of sage-grouse. On March 5, 2010, the USFWS determined that the sage-grouse was "warranted, but precluded" and classified the sage-grouse as a candidate species; however, this decision was immediately appealed by several environmental organizations and the final outcome of this appeal has not been determined as of the date of this report.

The status of greater sage-grouse and current threats have been thoroughly reviewed in three recent documents. These include Connelly et al. (2004), USFWS (2008) and 24 chapters on greater sage-grouse conservation released in 2009 (Marti 2009). The purpose of this review is to briefly summarize the major existing threats to greater sage-grouse. The three documents cited above were relied on extensively for this review, and they are the primary citations included in the review. For additional information on threats to sage-grouse, as well as individual citations included within each of those documents, the reader is referred to those documents.

Habitat Loss

Habitat loss has undoubtedly been the largest factor affecting sage-grouse populations in western North America, as sage-grouse currently occupy only approximately 56% of their pre-European distribution (Schroeder et al. 2004). Historically, sagebrush was estimated to have covered 296 million acres in western North America (Schroeder et al. 2004). However, much of this has been lost due to agriculture, urban expansion, and other forms of development. Connelly et al. (2004) estimated that 61.5 million acres within their sage-grouse assessment area of nearly 800,000 square miles is composed of agricultural lands, and that the presence of agricultural lands has directly and indirectly influenced 284.7 million acres (56 percent) of sage-grouse habitat within their assessment area. The amount of habitat lost to agriculture varies by region, but has been especially prevalent in the Columbia Basin of Washington and Oregon and the Snake River Plain of Idaho. For example, it has been estimated that 99% of basin big sagebrush (*Artemisia tridentata tridentata*) habitat in the Snake River Plain has been converted to cropland, and that 60% of the original shrub steppe habitat in Washington has been converted to primarily agricultural uses. Large-scale conversion of sagebrush to agriculture in north-central Oregon has eliminated sage-grouse from that region (USFWS 2008).

After conversion to agriculture, urban and suburban sprawl likely represent the second most important cause of habitat loss. Over 60% of all counties in the Rocky Mountain West are experiencing rural sprawl. In some Colorado counties, for example, up to 50% of sage-grouse habitat is under rural subdivision development, and it is estimated that 3 to 5 percent of all sage-grouse historical habitat in Colorado has already been converted into urban areas (Braun 1998). According to the USFWS (2008), given the current demographic and economic trends in the Rocky Mountain West, it is expected that the rates of urbanization will continue to increase, resulting in further habitat fragmentation and degradation.

Fire

Fire eliminates or substantially reduces sagebrush within burned areas, and big sagebrush (*Artemisia tridentata*), the most widespread species, are killed by fire and do not re-sprout after burning. Wyoming big sagebrush (*A. t. wyomingensis*) likely requires well over 100 years to reestablish after fire, and mountain big sagebrush (*A. t. vaseyana*) takes 35 to at least 100 years (UFWS 2008). Therefore, large wildfires completely eliminate sage-grouse habitat for decades. According to Baker (2009), historical fire intervals in sagebrush communities were >200 years in little sagebrush (*Artemisia arbuscula*), 200–350 years in Wyoming big sagebrush, and 150–300 years in mountain big sagebrush; however, some range ecologists disagree with Baker's conclusions and have data that indicate shorter fire intervals (Alan Sands, The Nature Conservancy, pers. comm.).

From 1980 to 2007, the number of fires and the total area burned increased across the greater sage-grouse's range. The approximate area burned on or adjacent to BLM-managed lands increased from 346,000 acres in 1998 to 2.0 million acres in 1999 and to 3.5 million acres in 2006 (USFWS 2008). According to the USFWS (2008), fire is especially of concern and may even affect the persistence of greater sage-grouse in the Great Basin (western Utah, Nevada, southwestern Idaho, and eastern Oregon) because burned areas in this region are especially prone to invasion by exotic annual grasses. Wildfires have removed approximately 4.25 million acres of sage-grouse habitat since 2003, or approximately 2.5 percent of current habitat. Although fire occurs throughout the range of greater sage-grouse, it disproportionately affects the states of Idaho, Nevada, Oregon, and Utah. Fires destroyed 30 to 40% of sage-grouse habitat in southwestern Idaho in a 5-year period (1997–2001), including 500,000 acres that burned from 1999 to 2001, which significantly impacted the largest remaining contiguous patch of sagebrush in the state (USFWS 2008). Between 2003 and 2007, Idaho lost about 660,000 acres of sage-grouse habitat, which is about 7 percent of the total estimated habitat in the state. During the period 1999–2007, about 2.5 million acres of sagebrush habitats were burned in Nevada, which represents approximately 11% of current sagebrush in the state (USFWS 2008). A panel of sage-grouse experts in Idaho ranked what they perceived as all major threats to sage-grouse in the state, and wildfire was ranked the top threat to sage grouse (ISAC 2006).

Invasive Plant Species

Annual grasses, primarily cheatgrass (*Bromus tectorum*) in Wyoming big sagebrush communities and medusahead (*Taeniatherum caput-medusae*) in more mesic sagebrush communities, have caused extensive loss of sage-grouse habitats in the western U.S. (Connelly et al. 2004). These two species impact sagebrush ecosystems by shortening fire intervals to as frequent as every 5 years in some cases, which perpetuates their persistence and intensifies the impacts of fire (USFWS 2008). Although annual grasses occur throughout the range of sage-grouse, they are considered more of a problem in the western states (e.g., Washington, Oregon, Idaho, Nevada, Utah) than Rocky Mountain states (Montana, Wyoming, Colorado; Connelly et al. 2004); however, a recent large (100,000-acre) fire near Worland, Wyoming also became infested with cheatgrass (USFWS 2008).

In 2000, the BLM reported that noxious weeds and annual grasses occupied 29.4 million acres of public lands in Washington, Oregon, Idaho, Nevada, and Utah, and that 36.5 million acres were occupied by noxious weeds and annual grasses throughout the range of the sage-grouse (USFWS 2008). Cheatgrass has the potential to displace native species in approximately 80% of the Great Basin within 30 years (Connelly et al. 2004). According to the USFWS status review (USFWS 2008), invasive plants are a major threat throughout the range of the sage-grouse, and one of the highest extinction risk factors for sage-grouse based on the ability of invasive plants to out-compete sagebrush, the inability to effectively control them once they become established, and the synergistic interaction between invasive plants and other risk factors such as wildfire and infrastructure construction, which can lead to establishment of invasive plants in otherwise intact sagebrush ecosystems.

Although not exotic species in North America, pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) woodlands can encroach upon sagebrush habitats and render them unsuitable or less suitable for sage-grouse. Pinyon-juniper woodlands occupy at least 44.6 million acres within the range of sage-grouse, and the extent of juniper woodlands has increased 10-fold in the Intermountain West since European settlement. It is been estimated that up to 90 percent of existing pinyon-juniper woodlands currently occupy areas previously dominated by sagebrush prior to the late 1800s (USFWS 2008). According to Connelly et al. (2004), approximately 41% of the Great Basin has either a moderate or high risk of sagebrush displacement by pinyon-juniper within 30 years. In their 2005 status review, the USFWS considered pinyon-juniper invasion a high extinction risk for sage-grouse in the western portion of its range, but a low risk in the eastern portions of its range.

Although the expansion of pinyon-juniper has largely been attributed to decreased fire intervals in big sagebrush habitats, a recent analysis by Baker (2009) found that historical mean fire intervals were sufficiently long to allow trees to widely invade sagebrush, yet millions of acres of mature sagebrush without trees were present when Europeans first entered the West. Based on this, Baker (2009) concluded that fire was not likely the primary factor preventing tree invasion into sagebrush, and factors other than fire exclusion must be the primary causes of post European settlement tree invasions into sagebrush habitats. Other possible causes of increased pinyon-juniper invasion include livestock grazing, increased CO^2 levels in the atmosphere, and other influences of climate change that favor establishment of tree species (USFWS 2008).

Habitat Fragmentation

Habitat fragmentation is defined as the separation or splitting apart of previously contiguous, functional habitat components of a species (USFWS 2008). Fragmentation can result from direct loss of habitat that leaves the remaining habitat in non-contiguous patches, or from alteration of habitats that leaves the altered patches unusable to a species, known as functional habitat loss. Functional habitat losses include disturbances that change habitat or remove one or more habitat functions, physical barriers that preclude use of otherwise suitable areas, and activities or structures that prevent sage-grouse from using suitable habitat patches due to behavioral avoidance. In their analysis of anthropogenic features on the landscape, or the 'human footprint', Leu and Hanser (2009) estimated that high-intensity human footprints covered 4.8% of current sage-grouse range, intermediate-intensity covered 47.0%, and low-intensity covered 48.3%. In their analysis, examples of high-intensity human footprints included areas radiating from major urban areas such as Denver, as well as areas dominated by agriculture such as the Columbia Plateau in Washington, and the Snake River Plain in Idaho. Examples of low-intensity areas were national parks and wilderness areas (Leu et al. 2008). Fragmentation of sagebrush habitats has been considered a primary cause of sage-grouse population declines because sage-grouse require large expanses of contiguous sagebrush. Sources of habitat fragmentation within the range of the sage-grouse include energy development (oil and gas as well as renewable), power lines, roads and highways, communication towers, fences, and suburban or rural sprawl.

It has been estimated that there are 500,000 miles of high-voltage transmission lines within the U.S. and an unknown amount of smaller distribution lines (Manville 2002). Connelly et al. (2004) estimated that there is a minimum of 5,904 square miles of land (less than 1 percent of their assessment area) in transmission power line corridors within their assessment area. Power lines not only cause direct mortality to sage-grouse through collision mortality, but restrict use of adjacent areas due to increased predation rates from ravens and raptors perching and nesting on power lines, avoidance of tall structures, and habitat alterations caused by access roads along powerlines. Johnson (2009) conducted an extensive review of literature on powerline impacts to sage-grouse and found that although response of sage-grouse to power lines varies widely, population level impacts could occur out to three miles from powerlines, with one study showing lek decreases out to 5 miles. The extent of power lines within currently occupied sage-grouse habitats is anticipated to increase substantially in the future given the increasing development of energy resources and urban areas (USFWS 2008).

Connelly et al. (2004) reported that 9,510 new communication towers have been constructed in recent years within sage-grouse habitats. Although the USFWS (2008) stated that the impacts of communication towers on a rangewide basis are unknown, a recent study found that there was a steady downward pattern of trends of lek counts as the number of communication towers increased at both 5 km and 18 km scales (Johnson et al. 2009a). Fences may cause direct mortality when sage grouse collide with them but also result in habitat fragmentation when they are used by perching raptors or provide predator corridors when access roads occur along fences. Over 625 miles of fences were constructed annually in sagebrush habitats from 1996 through 2002, and over 31,690 miles of fences were constructed on BLM lands within the range of sage-grouse between 1962 and 1997 (USFWS 2008).

Impacts associated with roads include direct habitat loss, direct mortality, barriers to migration, facilitation of predators and spread of invasive plants, as well as other indirect influences such as noise (USFWS 2008). Interstates and major paved roads covered approximately 22,835 square miles within their assessment area (Connelly et al. 2004), while densities of secondary paved roads were more than 3.24 mi/mi^2. Roads provide corridors for both mammalian predators as well as corvids to move into previously unoccupied areas. The presence of roads also increases human access and associated disturbances. The expansion of road networks also has been documented to facilitate exotic plant invasions. Male lek attendance was found to decline within 1.9 miles of a gas haul road when traffic volume exceeded just one vehicle per day. In this same study area, it was found that nest initiation rates for females bred on leks close to roads were lower than females bred on leks removed from roads, and the presence of gas haul roads affected population recruitment (USFWS 2008). During a study of 804 leks within 62.5 miles of Interstate 80 in southern Wyoming and northeastern Utah, no leks occurred within 1.25 miles of the interstate and only 9 leks occurred between 1.25 and 2.5 miles away from the Interstate (Connelly et al. 2004). Railroads covered 53 mi^2 of the sage-grouse assessment area in Connelly et al. (2004), but were estimated to influence an area of 71,000 mi^2 , or 9% of the assessment area, using a zone of influence of three miles. Railroads likely have similar impacts as highways.

Energy Development

Energy development impacts sage-grouse and sagebrush habitats through direct loss of habitat from well pads, seismic surveys, roads, powerlines and pipeline corridors, as well as indirect loss of habitat. Behavioral avoidance of noise, gaseous emissions, changes in water availability and quality, and human presence lead to indirect habitat loss (USFWS 2008). Based on a review by Naugle et al. (2009), energy development causes both male and female sage-grouse to abandon leks if they are disturbed by raptors perching on power lines near leks, by vehicle traffic on nearby roads, or by noise and human activity associated with energy development. Direct mortality through collisions with power lines and vehicles, and increased predation by raptors is also associated with energy development. In addition, man-made ponds created by some forms of energy development (e.g., coal bed methane) increase populations of mosquitoes that are known to transmit West Nile virus to sage-grouse. Sage-grouse also avoid suitable habitat as development increases. For example, oil and gas development that results in more than one pad per square mile results in impacts to breeding populations, and conventional well densities of eight pads per square mile results in complete exclusion of sage-grouse (Naugle et al. 2009). Impacts to leks associated with oil and gas development (e.g., drilling rig, producing well, haul road) have been detected at distances of between 3–4 miles. Because sage-grouse exhibit high site fidelity and continue to use areas disturbed by energy development, a time lag of between three and four years has been detected between onset of the development and population impacts (Naugle et al. 2009).

According to the USFWS (2008), extensive oil and gas reserves occur throughout much of the sage-grouse's range, but especially in portions of North Dakota, South Dakota, Montana, Wyoming, Colorado, Idaho, and Utah. As an example of how future energy development may impact sage-grouse, approximately 80% of greater sage-grouse "core population areas" in Wyoming, which contain a substantial portion of the entire sage-grouse population in its eastern range, are currently leased for oil and gas production. In their 2005 status review, the USFWS identified energy development as the most significant extinction risk to the greater sage-grouse in the eastern portion of its range (Colorado, Wyoming and Montana).

Livestock Grazing

Native vegetation communities within the sagebrush ecosystem developed largely in the absence of significant grazing presence. Excessive grazing by domestic livestock during the late 1800's and early 1900's, along with severe drought, significantly impacted sagebrush ecosystems such that long-term effects from this overgrazing still persist today (USFWS 2008). Livestock grazing is the most widespread type of land use across the range of the sage-grouse, and virtually all sagebrush dominated areas are managed for livestock grazing (Connelly et al. 2004). Livestock grazing may affect sage-grouse in a variety of ways. Several studies, although not all, have shown that sage-grouse nest success increases with increased grass and shrub cover for protection from predators. Therefore, reduction of grass heights and cover due to livestock grazing in sage-grouse nesting areas has been shown to negatively affect nesting success by reducing cover and therefore increasing nest predation (USFWS 2008).

Livestock consumption of forbs may reduce food availability for sage-grouse, and potentially could affect nutrition of females, and thereby nest initiation rate, clutch size, and subsequent reproductive success (USFWS 2008). Domestic livestock grazing reduces water infiltration rates and cover of herbaceous plants and litter, compacts soil, increases soil erosion, and increases invasion of exotic plants, all of which combine to change in the proportion of shrub, grass, and forb components in the grazed area. The effects of direct competition between livestock and sage-grouse depend on condition of the habitat and the grazing practices, and thus the effects vary across the range of the greater sage-grouse. Poor livestock management in mesic sites, vital habitats for sage grouse chicks, results in a reduction of forbs available to sage-grouse chicks and may affect chick survival. Livestock may also compete directly with sage-grouse for food and other resources. Although cattle are primarily grazers, they will utilize forbs and even browse sagebrush. Domestic sheep prefer foraging on forbs, and also browse sagebrush. In addition to changes in habitat, nest destruction by livestock trampling has been documented and the presence of livestock can cause sage-grouse to abandon their nests (USFWS 2008).

Not all aspects of livestock grazing are detrimental to sage-grouse, as research has shown that sage-grouse preferred grazed meadows over ungrazed meadows in late summer because grazing had stimulated the regrowth of forbs. In addition, grazing by sheep and goats may have some use for controlling invasive weeds and woody plant encroachment onto sagebrush rangelands (USFWS 2008).

Another indirect impact on sage-grouse associated with livestock grazing has been the practice of eliminating sagebrush to increase forage for livestock. By the 1970s, over 5 million acres of sagebrush had been mechanically treated, sprayed with herbicide, or burned to improve grass production for livestock forage (USFWS 2008).

West Nile Virus

West Nile virus (WNv) was introduced into the northeastern United States in 1999 and has spread across North America, causing millions of wild bird deaths (USFWS 2008). The virus is largely transmitted from mosquitos to birds, but bird-to-bird transmission has also been documented in several species, including greater sage-grouse. The first documented sage-grouse fatalities from WNv occurred in Wyoming in 2003, but greater sage-grouse WNv deaths have now been detected in 10 states and 1 Canadian province (USFWS 2008). Greater sage-grouse have high susceptibility to WNv and experience high levels of mortality when exposed. Most WNv transmission in sage-grouse occurs in July and August. Greater sage-grouse congregate in mesic habitats such as riparian areas and near springs in the late summer, which increase their exposure to mosquitoes. Sage-grouse mortality from WNv occurs when survival is otherwise typically high for adult females, and therefore is likely an additive source of mortality that reduces average annual survival. Four studies in the eastern half of the sage-grouse range (Alberta, Montana, Wyoming) found that survival in affected populations declined 25% in July and August. A 2003 outbreak in Wyoming resulted in extirpation of the local breeding population, as five leks affected by the disease becoming inactive within 2 years. Female mortality in Montana increased from the typical level of 1–5% to 16 percent during July and August due to WNv mortality, and in South Dakota, summer mortality from WNv was estimated to be 21–63% of the population (USFWS 2008). Originally, it was thought that greater sage-grouse exposed to WNv would suffer 100% mortality. However, more recent studies have found small percentages of affected populations have birds with antibodies to the virus, indicating they survived exposure. Sublethal affects may still impact birds that survive WNv exposure (USFWS 2008). Based on their review of WNv effects on sage-grouse, Walker and Naugle (2009) found that WNv can simultaneously reduce juvenile, yearling, and adult survival, and persistent low-level WNv mortality in combination with severe outbreaks may lead to local and regional population declines. Small, isolated, and peripheral populations, especially those at lower elevations, and populations experiencing large-scale increases in distribution of surface water, such as those near coalbed methane developments, may be at higher risk.

Recreational Hunting

Hunting is not considered a significant threat to sage-grouse. However, this section was included because the continued practice of hunting greater sage-grouse even as they undergo population declines and have been petitioned for listing under the ESA is widely questioned. Unlike most other factors that impact sage-grouse, hunting does not affect habitat and therefore the potential for long-term impacts is eliminated. Greater sage-grouse are legally hunted in 10 of the 11 states where they occur, with Washington being the only state with a closed season (sage grouse are listed as state-threatened in Washington). In Canada sage-grouse are considered an endangered species and hunting is not allowed. Various authors have suggested that sage-grouse could sustain harvest rates of up to 30 percent, but current harvest levels are managed to be below 5-10% of the population (USFWS 2008). Later hunting seasons have been implemented by all states to reduce hunting pressure on females, and bag limits and season lengths are conservative compared to the past (USFWS 2008). Annual harvest estimates range from 10 birds in South Dakota to 10,378 in Wyoming, and the total estimated annual harvest of sage-grouse in 2007 was 28,180 birds (Reese and Connelly 2009). In their review, the USFWS (2008) concluded that no studies have demonstrated that regulated sport hunting is a primary cause of the decline in greater sage-grouse observed, although it is not certain that hunting has not been a contributing factor to sage-grouse declines. The effects of harvest likely depend substantially on the quality of habitat. Even though Washington State and Canada prohibit sage-grouse harvest, populations have not recovered in those areas. Data obtained from harvested sage-grouse also provide population data such as an index to annual chick production, which are useful for understanding the dynamics of sage-grouse populations (Christiansen 2008). In addition, hunting creates a constituency of sage-grouse advocates whose license fees provide funding for sage-grouse management and who are interested in seeing the needs of grouse populations are met (Christiansen 2008).

Climate Change

Potential effects of climate change on sagebrush habitats, and therefore sage-grouse, were recently reviewed by Miller et al. (2009). Their review found that average temperatures may increase by 2.8 to more than 6 ^{0}C for the Great Basin as well as many of the areas that support sagebrush in the western U.S. It has been predicted each 1 ^{0}C increase in temperature would result in a loss of 33,570 mi^2 of existing sagebrush habitat, and that only 20% of existing sagebrush would remain if the mean temperatures increased by 6.6 ^{0}C. Most of the remaining sagebrush would occur in the northern extent of its range and at higher elevations. Although the loss of sagebrush due to climate change would be caused primarily through increasing the distribution of other woody vegetation, the increase in exotic plant species and associated changes in fire cycles would also substantially reduce sagebrush habitats. Increasing temperatures would also favor expansion of frost-sensitive woodland vegetation northward, which could result in additional displacement of sagebrush habitats. Cheatgrass, an exotic annual grass, has increased the frequency and intensity of wildfires compared to historical levels and has resulted in large-scale conversion of sagebrush shrublands into exotic annual grasslands. The invasion of cheatgrass and its resultant replacement of sagebrush habitats through increasing the fire frequency is a significant threat to sage-grouse populations. Cheatgrass has been most successful at replacing sagebrush in the warmest, driest areas, including sagebrush communities at lower elevations and on south and west facing slopes. Sagebrush communities at higher elevations and on north and east facing slopes have been at lower risk of cheatgrass invasion, but increasing temperatures associated with climate change could increase the risk of cheatgrass invasion in these areas as well (Miller et al. 2009). In addition to increasing fire frequencies associated with cheatgrass invasions, wildfire is expected to increase substantially in sagebrush because of global warming (Baker 2009).

The USFWS (2008) also reviewed potential effects of climate change on greater sage-grouse and concluded that current climate change projections indicate the potential for negative affects on sage-grouse through changes in sagebrush communities. Their review found that the current area of sagebrush would contract significantly, and that increases in fire frequency under simulated climate projections would leave big sagebrush more vulnerable to fire impacts. As an example, they discussed a study conducted by the World Wildlife Fund, who developed a bioclimatic envelope model for big sagebrush and silver sagebrush in the states of Montana, Wyoming, and North and South Dakota, which suggested that large displacement and reduction of sagebrush habitats will occur under climate change as early as 2030 for both species of sagebrush examined.

Another significant emerging threat of climate change to sage-grouse is West Nile Virus (WNV), which is transmitted to sage-grouse from mosquitoes. Walker and Naugle (2009) reviewed the literature on sage-grouse and WNV and found that WNV transmission in sage-grouse is reduced and delayed during years with lower summer temperatures. The review also found data which suggested that high temperatures associated with drought increases WNV transmission. Temperature can also affect exposure of sage-grouse to WNV because it influences habitat use, as sage-grouse concentrate near water during hot weather and become more susceptible to mosquitoes. The authors concluded that increasing temperatures associated with climate change may increase impacts to sage-grouse from WNV.

The U.S. Department of Interior (USFWS 2008) examined scenarios for extinction of sage-grouse for 100–200 years into the future. The predicted changes in CO^2 emissions and temperature increases and the associated impacts to sage-grouse were found to be large enough that they would overwhelm any current population trajectories associated with habitat loss and alteration. Based on results of this review, it is clear that climate change presents a significant threat to maintenance of healthy sage-grouse populations throughout the West.

Effects of Wind Energy Development on Sage-Grouse: A Review of the Literature

Direct Effects

The most likely cause of direct impacts to greater sage-grouse associated with wind energy development includes mortality caused by collisions with wind turbines, distribution lines, and transmission lines. Other direct impacts may include vehicle collisions and increased levels of poaching. For sage-grouse, the highest collision probabilities appear to occur when structures are located in areas where grouse typically fly between foraging and loafing habitats (Science Applications International Corporation [SAIC] 2001). If the locations of such areas are known, impacts can be reduced by avoiding them when siting wind energy facilities. In most cases, it is unlikely that direct mortality would result in population declines of sage-grouse at wind-energy facilities.

Empirical data and theoretical considerations indicate that species such as sage-grouse, which have high wing loading and low aspect, have a relatively higher risk of colliding with obstacles. These birds are characterized by rapid flight, and the combination of heavy body and small wings restricts swift reactions to unexpected obstacles (Bevanger 1998). A literature review completed by Bevanger (1998) suggests that bird species can be grouped into six categories determined by differences in aerodynamic performance. Sage-grouse are grouped into the "poor flyers" groups, as are all galliformes. According to Rayner (1988), the species considered "poor" flyers have probably never experienced strong pressure to enhance their flight efficiency. Because of their lack of flying efficiency, sage-grouse as well as other gallinaceous birds may potentially be more likely to collide with obstacles.

It is possible that for the reasons mentioned above, upland gamebirds are one of the more common collision victims found during post-construction fatality surveys at wind energy facilities. Johnson and Stephens (2010) reviewed publicly available literature from 21 studies that quantified avian mortality at wind energy facilities in western North America and found that upland gamebirds ranked third, behind only passerines and raptors, comprising 9.6% of all identified fatalities. Although ring-necked pheasant (*Phasianus colchicus*; 45 fatalities), gray partridge (*Perdix perdix*; 38) and chukar (*Alectoris chukar*; 18) were the most common upland gamebird fatalities found, there were also five sharp-tailed grouse, one greater prairie-chicken, and one greater sage-grouse among the 1,247 fatalities represented by 128 species (WEST, Inc., unpublished data).

In addition to collisions with wind turbines, greater sage-grouse also may collide with other structures such as distribution lines, transmission lines, and meteorological towers associated with wind energy development. Documentation of direct mortality of sage-grouse resulting from collisions with transmission lines has been minimal. Two adult females and one adult male sage-grouse were found beneath a telephone line five miles north of Beaver, Utah (Carter et al. 1939). A second report provided on February 14, 1977 documented the locations of eight sage-grouse carcasses (on four separate occasions) where mortality was believed to be the result of collision with a power line. In the account the individual suggests that the flying elevation of the sage-grouse is a main concern for collisions. The observer reported seeing two flocks fly over the power lines, one approximately ten feet above the line and the second so close that he was unable to tell if the birds passed above or below the lines (unpublished letter 1977). Although data on sage-grouse collisions with power lines are minimal, one study showed that a significant proportion of annual mortality can be caused by power line collisions. Beck et al. (2006) monitored survival of 15 radio-collared juvenile sage-grouse in the Medicine Lodge area of Clark County, Idaho and 43 juvenile sage-grouse in the Table Butte area of Clark and Jefferson counties, Idaho in 1997 and 1998. Although all mortality documented in the Medicine Lodge area was attributed to predation, 33 percent of the juvenile mortality (two of the six fatalities) in the Table Butte area was attributed to collisions with power lines (Beck et al. 2006).

Several other instances have been found where sage-grouse have collided with power lines in Idaho (Jack Connelly, IDFG personal communication) and Montana (Craig Fager, MFWP, personal communication). In Montana, several dead sage-grouse were found underneath a power line in close proximity to a lek, and it was presumed birds flying into the lek during low light may have collided with the line. The actual occurrence of sage-grouse collisions with transmission lines is difficult to evaluate. Collision mortality may be higher at distribution lines than transmission lines because distribution lines are not as high. A majority of power lines are located in remote areas with little human presence; therefore, reported losses must be considered a superficial measure of its actual occurrence (Thompson 1978; Longridge 1986; Faanes 1987).

Many meteorological (met) towers at wind energy facilities are supported by guy wires. Although guy wires used to support communication towers have been implicated in higher collision mortality rates for many species of birds (e.g., Longcore et al. 2005), no instances of sage-grouse colliding with guy wires were found in the literature. However, the one sage-grouse fatality found at the Foote Creek Rim wind energy facility in Wyoming was found near both a wind turbine and a met tower, and it was not possible to determine which structure it actually collided with.

Although some sage-grouse collision mortality may occur at wind energy facilities, it is unlikely that this would result in any population consequences given that sage-grouse can apparently tolerate up to 5–10% annual mortality (USFWS 2008). Based on the number and species composition of bird fatalities found at 11 existing wind energy facilities in the Columbia Plateau region of eastern Washington and Oregon, Johnson and Erickson (2008) estimated that over 2,000 upland gamebird fatalities, primarily ring-necked pheasants (*Phasianus colchicus*), chukar (*Alectoris chukar*) and gray partridge (*Perdix perdix*) could occur annually with 6,700 MW of wind energy development in eastern Washington and Oregon. Given estimated population sizes of these three upland gamebirds in the study area (370,900 combined) based on breeding bird survey data (Blancher et al. 2007), however, this would represent only 0.55% of breeding populations, which is insignificant from a population standpoint.

Indirect Effects

Much debate has occurred recently regarding the potential impacts of wind-energy facilities on prairie grouse. It is currently unknown how prairie grouse, which are accustomed to a relatively low vegetation canopy, would respond to numerous wind turbines hundreds of feet taller than the surrounding landscape. Some scientists speculate that such a skyline may displace prairie grouse hundreds of meters or even miles from their normal range (Manes et al. 2002, U.S. Fish and Wildlife Service [USFWS] 2003, NWCC 2004). If birds are displaced, it is unknown whether, in time, local populations may become acclimated to elevated structures and return to the area. The USFWS argued that because prairie grouse evolved in habitats with little vertical structure, placement of tall man-made structures, such as wind turbines, in occupied prairie grouse habitat may result in a decrease in habitat suitability (USFWS 2004). Several studies have shown that prairie grouse avoid other anthropogenic features, such as roads, power lines, oil and gas wells, and buildings (Robel et al. 2004, Holloran 2005, Pruett et al. 2009). Much of the infrastructure associated with wind energy facilities, such as power lines and roads, is common to most forms of energy development and it is assumed that impacts would be similar. Nevertheless, there are substantial differences between wind energy facilities and most other forms of energy development, particularly related to human activity. For example, operating gas wells require one visit per day, whereas each wind turbine typically requires only two maintenance visits per year. While results of these studies suggest the potential exists for wind turbines to displace prairie grouse from occupied habitat, well-designed studies examining the potential impacts of wind turbines themselves on prairie grouse have not been completed. Although the data collected during some studies indicate that prairie grouse may continue to use habitats near wind-energy facilities, research conducted on greater sage-grouse response to oil and gas development has found population declines due to oil and gas development may not occur until four years post-construction (Naugle et al. 2009), and results of another study of oil and gas development suggested that there is a delay of 2–10 years before measurable effects on leks manifest themselves (Harju et al. 2010). The purpose of this review is to summarize what little information is available on prairie grouse response to wind energy.

Preliminary data are available from ongoing telemetry research projects being conducted on greater prairie chickens in Kansas and Minnesota and greater sage-grouse in Wyoming. The sage-grouse study is being conducted at the site of two proposed and one existing wind energy facility in Carbon County, Wyoming (Johnson et al. 2009b). The existing PacifiCorp Seven Mile Hill facility was completed in 2008 and consists of 79 GE 1.5-MW turbines. The study consists of monitoring leks near each facility as well as monitoring habitat use and demographics of 75 radio-collared female sage grouse. Three leks occur within one mile of the existing Seven Mile Hill facility. Results of lek counts conducted the year prior to and during the first breeding season following construction of the project are provided below:

Lek ID	**Distance to nearest turbine**	**2008 male count**	**2009 male count**
Missouri John	0.85 mi	74	62
Pine Draw	0.38 mi	33	20
Commo 1	0.83 mi	23	21
Total		130	103

The three leks closest to turbines ranged from 0.38 to 0.85 miles from the nearest turbine, and averaged 0.69 miles. All three leks remained active during the first breeding season after turbines were erected. The total number of males counted on all three leks was 130 in 2008, prior to construction of the facility. The maximum number dropped to 103 in 2009, the first breeding season after construction. However, there was also a slight decline of sage grouse on leks in the entire region between 2008 and 2009, possibly due to weather or some other natural factor, so the decrease on these three leks may or may not be attributable to the presence of wind turbines. Twenty-five female sage grouse were trapped off of these three leks and radio-collared in the spring of 2009. Nine of these birds (36%) nested within one mile of a turbine, the closest of which were located 130, 278, 388 and 486 m from the nearest turbine. Only one of the nine nests (11%) initiated within 1 mile of turbines was successful, and none of the four nests closest to turbines was successful. Overall nest success for the entire marked population was 40%. Without any pre-construction data, however, it is not known where birds from these leks nested before wind development or how successful nests within one mile of turbines were prior to construction. Radio locations of marked females from the time of capture in early April 2009 to

September 30, 2009 do not indicate significant displacement of these birds from the existing wind energy facility, as numerous locations have been recorded within and adjacent to the turbine development area. However, due to high site fidelity exhibited by sage-grouse, displacement may not occur until a few years after construction.

In addition to data collected on leks and radio-collared birds, anecdotal observations of sage-grouse use near turbines have been made. While night-lighting in the area to capture females in early April 2009, three males were observed night-roosting within 100 m of turbines, the closest of which was approximately 60 m from a turbine. It was not noted whether or not the turbines were running. In addition, while conducting fatality monitoring of the Seven Mile Hill turbines, WEST biologists routinely have observed adult sage grouse within the vicinity of (e.g., < 200 m) of operating turbines.

Incidental observations have also documented greater sage-grouse use of other wind energy facilities. Groups of 5 and 8 individuals were documented on separate occasions during the post-construction fatality study at the Elkhorn facility in Oregon, during July and November, respectively (Jeffrey et al. 2009). Greater sage grouse have been also been infrequently observed at the Wild Horse project in Washington, including one documented nesting attempt within 100 m of a project turbine (Erickson et al. 2008). At the Foote Creek Rim wind energy facility in Wyoming, one sage grouse and a brood were documented within the turbine development area (Young et al. 2001). Due to high site fidelity by sage-grouse, observations of sage-grouse near wind energy facilities do not necessarily imply that these facilities have no impact on sage-grouse populations or population viability.

Kansas State University has been conducting a study of greater prairie-chicken response to the 67-turbine Horizon Meridian Way wind energy facility in Cloud County, Kansas. The study includes conducting lek counts as well as monitoring several radio-collared females. Preliminary results indicate that lek activity may have decreased within the turbine development area following construction, although most leks located prior to construction are still present. One lek, although reduced from 12 to 4 birds following construction, was still active, with the birds booming on the turbine pad itself, or within 10 m of the turbine. One of these birds was even observed perched on the turbine steps. Based on a rigorous statistical analysis of 70 nesting locations within 25 km of the wind energy facility, it was determined that nesting female greater prairie chickens were not displaced by turbines. In fact, results of the statistical analysis indicated that females were nesting closer to turbines than would be expected at random.
Greater prairie chicken nests were located as close as 74 m from a turbine, 82 m from a transmission line, and 22 m from a road (McNew et al. 2009). These data were collected in 2009, the first year after construction of the wind energy facility, and may not reflect how prairie chickens respond to wind energy development several years after construction.

Greater prairie chicken response to a three-turbine wind energy facility in Minnesota has also been monitored through conducting lek counts and following radio-marked female greater prairie chickens. Researchers documented six active greater prairie-chicken leks within two miles of the turbines, with the nearest lek located within 0.6 miles of the nearest turbine (USFWS 2004). Subsequent research at this facility based on 40 nest locations also found that nesting females were not avoiding turbines. As was the case with the Kansas study, females were actually found to be nesting closer to turbines than expected. Based on extensive research of the prairie chicken population in the vicinity of this wind energy facility from 1997 to 2009, it was concluded that the distribution and location of leks and especially nests was determined by the presence of adequate habitat in the form of residual grass cover, not the presence of vertical structures such as trees, woodlots, power lines, and wind turbines (Toepfer and Vodehnal 2009). The statistical tests used for these analyses were not provided in the abstract prepared by Toepfer and Vodehnal (2009).

In addition to these ongoing telemetry studies, we are aware of two other publicly-available studies that examined response of prairie grouse to wind energy development. The Nebraska Game and Parks Commission (NGPC) monitored both greater prairie-chicken and plains sharp-tailed grouse leks following construction of the 36-turbine Ainsworth wind-energy facility in Brown County, Nebraska (NGPC 2009). Surveys for leks were conducted four years post-construction (2006-2009) within a 1- to 2-mile radius of the facility, an area that covered approximately 25 mi^2. The number of leks of both species combined in the study area was 13, 12, 9 and 12 in the first four years post-construction. The number of greater prairie chickens counted on leks increased from 70 to 95 during the 4-year period, whereas the number of sharp-tailed grouse decreased from 66 to 56. The total number of both species combined increased from 136 to 151 individuals. No pre-construction data were available on prairie grouse leks near the site, so it is not possible to determine what changes in lek numbers, locations or size may have occurred post-construction. However, densities of lekking grouse on the study area at the Ainsworth facility were within the range of expected grouse densities in similar habitats in Brown County and the adjacent Rock County (NGPC 2009). The leks ranged from approximately 0.42 to 1.65 miles from the nearest turbine, with an average distance of 0.88 miles.

Greater prairie chicken leks surveys were conducted at the Elk River wind energy facility in Butler County, Kansas, within the southern Flint Hills, beginning three years prior to and continuing for four years post-construction (Johnson et al. 2009c). The facility consists of 100, 1.5 MW turbines. During the year immediately proceeding construction of the project (2005), 10 leks were present on the project area, with 103 birds on all leks combined. The number of leks and mean lek size decreased annually after construction, and by 2009, four years after construction, only one of these 10 leks remained active, with three birds on the lek. The 10 leks were located from 88 m to 1470 m from the nearest turbine, with a mean distance of 587 m; eight of the ten leks were located within 0.5 miles of the nearest turbine. Although this decline may be attributable to development of the wind energy facility, greater prairie chicken populations have declined significantly in the Flint Hills due to the practice of annual spring burning (Robbins et al. 2002). During the same time frame that leks were monitored at the Elk River facility, the estimated mean number of greater prairie chickens per square mile in the southern Flint Hills declined by 57% from 8.2 in 2005 to 3.5 in 2009. In Butler County, the estimated number of birds per square mile declined by 66%, from 20.4 in 2005 to 7.0 in 2009 (Kansas Department of Wildlife and Parks, unpublished data). This regional decline is attributed primarily to the practice of annual spring burning and heavy cattle stocking rates, which remove nesting and brood-rearing cover for prairie chickens (Robbins et al. 2002). Within the Elk River project area itself, half of the turbines are located on a ranch that previously never burned their pastures for consecutive years; however, pastures on this ranch were burned in the springs of both 2008 and

2009. This change in annual spring burning practices where half the turbines are located, combined with annual spring burning on adjacent ranches, likely contributed significantly to the decline of prairie chickens noted in the project area since the wind energy facility was completed. Research being conducted by Kansas State University at a nearby proposed wind energy facility has shown a dramatic decrease in greater prairie chicken populations in that area as well over the last few years, in the absence of wind energy development (Lance McNew, Kansas State University, pers. commun.). It is likely that the practice of annual spring burning has contributed significantly to the decline of lekking prairie chickens in the Elk River project area, similar to that occurring throughout Butler County and the southern Flint Hills. Therefore, it seems unlikely that the decline of prairie chickens on the Elk River site is due entirely to the presence of wind turbines (Johnson et al. 2009b).

Anecdotal evidence regarding sharp-tailed grouse lek locations in relation to wind turbines in North and South Dakota indicate that leks greater than 0.5-miles from the nearest turbine continue to be active after construction of the wind energy facility. There is also evidence that in the first year or two after construction that sharp-tailed grouse will continue to lek within much closer distances, but it is unknown if this will continue through later generations (Jill Shaffer, USGS, pers. commun.).

Due to potential time lag effects (Naugle et al. 2009, Harju et al. 2010), data spanning two or more grouse generations will likely be required to adequately assess impacts of wind-energy development on prairie grouse. However, because prairie chickens and sharp-tailed grouse are not as long-lived as sage-grouse, many, if not most, of those birds using leks near the Ainsworth, Nebraska facility in 2009 were likely not alive when the facility was constructed in 2005. Because these 2nd generation birds continue to use habitats near the wind energy facility, it can be concluded that prairie grouse population declines are likely not associated with this facility.

Another grouse with a lek mating system, the black grouse (*Lyrurus tetrix*), was found to be negatively affected by wind power development in Austria (Zeiler and Grunschachner-Berger 2009). The number of displaying males in the wind power development area increased from 23 to 41 during the 3-year period immediately prior to construction, but then declined to nine males four years after construction. In addition to the decline in displaying males, the remaining birds shifted their distribution away from the turbines. One lek located within 200 m of the nearest turbine declined from 12 birds one year prior to construction to no birds four years after construction.

Mitigation for Wind Energy Development Impacts on Sage-Grouse

Mitigation for environmental impacts is usually sequenced, with the first step being to avoid impacts, followed by minimizing impacts where avoidance is not possible, and lastly, on or off-site mitigation (habitat enhancement or protection) for any identified impacts that cannot be avoided (Johnson et al. 2007). To avoid impacts, the location of wind resource areas being considered for development should be evaluated in terms of the site's potential for impacts to sage-grouse. Sage-grouse populations and habitat at the site should be thoroughly evaluated on a regional scale to determine how important the site is for maintaining sage-grouse populations. If development of a project is in high value habitat with the potential for significant risk to the population, development should be avoided until research is conducted to show what actual impacts are. For sites where it is determined that significant risks to the population are not likely, the following best management practices to avoid or minimize impacts and habitat enhancement measures to compensate for (mitigate) impacts are recommended.

Best Management Practices

Construction

Several steps can be taken to minimize and avoid impacts to sage-grouse during construction of the wind energy facility. Impacts to breeding grouse can be avoided by buffering leks when siting all facilities, including turbines, access roads, substations, O&M facilities, and power lines. Similar buffers should be considered for winter concentration areas as well as known nesting and brood-rearing areas, when feasible. Due to a lack of research, the appropriate distance for these buffers is currently not known. Developers should work with state and federal natural resource agencies early in study design to develop mutually agreeable buffers to avoid and minimize impacts to sage-grouse.

As is generally recommended for all types of construction activities, when wind energy facility construction occurs within four miles of a lek, construction activities should not occur during the breeding period, which ranges from March 15 to June 30. All temporarily disturbed areas associated with construction should be restored to perennial native cover, including big sagebrush, and managed to control the spread of noxious weeds and other invasive plants. Reclamation plans should be developed in consultation with state and federal land management agencies as well as the local sage-grouse working group.

Every precaution should be taken to prevent fires during construction activities. A Fire Prevention and Control Plan should be developed prior to construction. Firefighting equipment should be maintained on site during construction activities as required in this plan so that any fires that may occur can be quickly suppressed. Vehicles used for construction and maintenance should have exhaust systems that exhaust on the top to reduce the potential for starting fires.

The number and length of access roads should be limited to that absolutely necessary to construct the project. Permanent maintenance roads should be constructed using the minimum footprint required. Consideration should be given to closing permanent maintenance roads to limit human presence.

Design Considerations

All permanent met towers should by unguyed monopole structures to prevent collision mortality as well as reduce the potential for raptor perching. Lattice supported met towers should not be used. If guyed met towers are required, the guy wires should be marked with bird flight diverters (BFDs). For all overhead power lines, use of BFDs should be considered to decrease the potential for collision related mortalities. Because sage-grouse are rarely documented to collide with power lines, quantitative data on effectiveness of BFDs to reduce sage-grouse mortality are lacking. However, several studies have shown that use of BFDs reduces mortality of other species. Avian Power Line Interaction Committee (APLIC 2006) guidelines should be used to limit collision mortality of sage-grouse. Because sage-grouse are known to collide with fences, no new fencing should be erected as part of the project, other than that required to exclude human access to the substation.

Currently, few studies have shown the efficacy of perch deterrents in preventing use of power line support structures towers as hunting perches by avian predators. One study demonstrated that the initial effect of perch deterrents was a reduction in avian predator use of overhead transmission line towers via shorter perching duration on the deterrents versus other available perching substrates (Lammers and Collopy 2007). The results also indicated that the probability of the presence of avian predators after construction and retrofitting was reduced, and that the probability of an avian predator perching on the towers was reduced; however, interpretation was confounded by small sample size and overlapping confidence intervals. Overall, the study suggests that perch deterrents did not appear to have an effect on the observed numbers of avian predators of concern to greater sage-grouse (Lammers and Collopy 2007). A study of the effectiveness of using five types of raptor perch deterrents on power poles along 11 km of transmission line within Gunnison sage-grouse (*Centrocercus minimus*) habitat in Colorado found that none of the perch deterrents was effective at deterring raptors or corvids from perching on poles because insulators and insulator covers provided safe perch sites (Prather and Messmer 2007).

Maintenance

All areas temporarily disturbed during construction should be periodically inspected to ensure desired vegetation is becoming established and that noxious weed infestations are not present. Remedial measures should be implemented immediately to improve vegetation establishment where warranted and to treat any noxious weed infestations. To minimize disturbance, maintenance roads should be closed to public vehicular access.

When scheduling maintenance activities within 1.0 miles of a lek following construction of the facility, activities should be scheduled outside the mating season when feasible and should only occur between 9 a.m. and 6 p.m. to avoid disturbing birds on leks if work is necessary during the breeding season.

Development of a Holistic Management or Mitigation Plan

Because it is unlikely that direct mortality to sage-grouse (*Centrocercus urophasianus*) from wind energy facilities would significantly inhibit populations, especially if facility infrastructure is sited to avoid direct impacts, mitigation through habitat enhancements is generally recommended where it has been documented that indirect effects of a wind energy facility may be reducing local sage-grouse populations. Habitat enhancements can be conducted onsite, or in the vicinity of the wind energy facility itself, or off-site, preferably in an area not significantly influenced by wind energy or other forms of development, but still close enough that sage-grouse from the same population as those at the wind energy facility are involved.

This portion of the document is meant to suggest steps required to develop a holistic sage-grouse management or mitigation plan for a given project area, and provide background information for each step for use by those developing the plan. The details of a particular plan will need to be developed on a site-by-site basis based on site-specific sage-grouse population and habitat information. Wind energy facilities may present anthropogenic barriers to sage-grouse; therefore management planning should proceed from the need to provide sage-grouse populations with large, functional, connected habitat patches across landscapes. Literature reviews of many of the pertinent issues have already been completed, and we relied largely on these sources throughout the following sections.

The Wyoming interagency vegetation committee (2002) outlined evaluation criteria that should be considered when proposing vegetation treatments in sagebrush-dominated habitats; Bohne et al. (2007) expand on this list of criteria with sage-grouse specific recommendations. The following list represents a conglomeration of these 2 sets of recommendations modified to reflect the focus of this document:

1. Determine and map sage-grouse seasonal habitats in proposed project area; establish the relative importance of the seasonal habitats occurring in proposed project area (e.g., do any limiting seasonal habitats exist?).
2. Establish the demographic parameters (e.g., nesting success, chick survival, adult survival) influencing sage-grouse population growth within the proposed project area; establish the current condition (e.g., habitat quality assessment) of sage-grouse seasonal habitats occurring in proposed project area.
3. Establish clear and concise management objectives based on sage-grouse population and vegetative condition information.
4. Establish potential management options to fulfill intended objectives, and identify potential impacts of proposed prescription(s) to sage-grouse.
5. If habitat treatments (e.g., prescribed fire, herbicide application, mechanical manipulation) are prescribed:
   (A) Establish site potential to respond to treatment in a desirable manner;
   (B) Develop a post-treatment management plan that will ensure desired vegetative responses can be achieved and maintained; and
   (C) Assess the presence of undesirable plant species (e.g., cheatgrass, invasive noxious weeds, conifers) and the risk of these species increasing as a result of the proposed treatment.
6. Ensure short-term and long-term post-prescription monitoring of project.

1. Determine and map sage-grouse seasonal habitats in proposed project area; establish the relative importance of the seasonal habitats occurring in proposed project area (e.g., do any limiting seasonal habitats exist?).

Connelly et al. (2003) suggest that habitat selection assessments progress through the following spatial scales: (1) first-order selection or the geographic range of the sage-grouse population of interest. (2) Within this geographic range, second-order selection of habitat should be examined based on subpopulations (e.g., grouse associated with a lek or lek complex). (3) Third-order selection is used to further refine habitats used by subpopulations by defining seasonally selected habitats (e.g., nesting habitat). (4) Finally, assessment can be made at the fourth-order selection level that quantifies food and cover attributes at particular use sites. Johnson (1980:69) describes this hierarchical nature of selection as: “a selection process will be of higher order than another if it is conditional upon the latter.” As an example, selection of food items will be of a higher order than selection of a feeding site because selection of a particular site determines the array of food items available to be selected.

To accurately establish sage-grouse habitat selection at the first- and second-orders, or landscape spatial scales, the migratory nature (e.g., seasonal movements) of the population inhabiting the project area must be well understood. It is important to note that migratory populations may use very large areas on an annual basis; Connelly et al. (2003) suggest that migratory populations may use an area the size of the state of Rhode Island. Third- and fourth-order selection reflects habitats used by individuals (versus populations). Generally radio-marked sage-grouse are required to accurately assess habitat selection of population(s) inhabiting a project area at all spatial scales, including the migratory nature of these populations. However, at higher order scales (e.g., second-, third- and forth-order), assuming habitat selection is relative to biologically important locations on a landscape (e.g., nesting habitat generally within a certain distance of a lek; early brooding habitat generally within a certain distance of a nest) and correlating vegetative conditions of habitat patches to seasonal habitat requirements of sage-grouse can provide estimates of habitats used by subpopulations and seasonally suitable patches throughout a project area.

The relative importance of a given seasonal habitat may be dictated by quantity (e.g., critical winter habitat may represent a small proportion of the available sagebrush habitats within a population's available landscape), quality (e.g., potentially suitable early brood-rearing habitats may be wide-spread, but sub-optimal forb cover may generally exist in these areas; see step 2 below), or juxtaposition (e.g., suitable early brood-rearing sites may not be spatially associated with suitable nesting sites at scales appropriate for effective availability). It is important to note that movement corridors between seasonally suitable sites should be considered distinct seasonal habitats, especially for migratory populations moving long distances between 2 or more seasons (Connelly et al. 2003). Assessing potentially limited seasonal habitats to a population will need to be accomplished on a site-by-site basis based on information amassed completing steps 1 and 2 in this document. Although the optimal proportions of distinct seasonal habitats on a landscape are unknown, sage-grouse productivity is generally increased if individuals are able to space themselves widely across the available landscape (Holloran and Anderson 2005). Those developing management or mitigation plans should view the used landscape holistically from the need to provide large, functional, connected habitat patches across that landscape for sage-grouse.

Connelly et al. (2003; http://gf.state.wy.us/downloads/pdf/sagegrouse/grousehabitatbook.pdf) provide a detailed description of sage-grouse capture and radio-tagging, and vegetation monitoring techniques, including the vegetation variables important to include in vegetation surveys. The following sections provide information on landscape scale habitat considerations, including the importance of habitat diversity, mosaics, juxtaposition and connectivity, and information on sage-grouse habitat selection by season.

LANDSCAPE CONSIDERATIONS

Sage-grouse are considered a landscape-scale species as populations generally inhabit and rely on large, interconnected expanses of sagebrush (*Artemisia* spp.; Connelly et al. 2004). Connelly et al. (2010) suggested that conclusive data establishing minimum sizes of sagebrush-dominated landscapes necessary to support viable populations of sage-grouse are unavailable. However, Leonard et al. (2000) estimated that sage-grouse populations in Idaho used an annual range of at least 2,764 km^2. Doherty (2008) suggests that a sagebrush-dominated landscape 314 km^2 in size may be a biologically defensible estimate of the area necessary to maintain breeding habitat around a given lek, but the size of a landscape capable of supporting breeding habitats of an interspersed population (e.g., an area with multiple leks spaced <10 km) may exceed 1,000 km^2. Investigations from Idaho and Wyoming suggest that relatively large blocks of sagebrush habitat (>4,000 ha) are critical to successful reproduction and over-winter survival (Leonard et al. 2000, Walker et al. 2007).

The sage-grouse habitat management guidelines (Connelly et al. 2000*b*) defined 3 potential migratory patterns that sage-grouse populations may exhibit: (1) non-migratory, where sage-grouse do not make long-distance movements between or among distinct seasonal ranges; (2) one-stage migratory, where sage-grouse move between two distinct seasonal ranges (e.g., distinct winter areas and integrated breeding and summer areas); or (3) two-stage migratory, where sage-grouse move among three distinct seasonal ranges (e.g., distinct winter, breeding and summer areas). Sage-grouse individuals belonging to one or more of these types of populations may reside in the same geographic region during one or more seasons. Connelly et al. (2000*b*) suggest that identifying the migratory nature of a population is important for establishing the relative importance of habitats occurring in a given area. For non-migratory populations, Connelly et al. (2003) suggest that seasonal habitats should be well interspersed with no major anthropogenic barriers (e.g., reservoirs) between habitats. Although migratory populations may use a large area, there are specific seasonal habitats used by the population that may be spatially isolated; corridors of sagebrush-dominated habitats should connect these seasonal ranges (Connelly et al. 2003).

Connelly et al. (2010) report that sage-grouse populations typically occupy habitats with a diversity of species and subspecies of sagebrush interspersed with a variety of other habitats (e.g., riparian meadows, agricultural lands, grasslands, sagebrush habitats with some conifer or deciduous trees); these habitats are usually intermixed in a sagebrush-dominated landscape and are often used by sage-grouse during certain times of the year (e.g., summer) or during certain years (e.g., above normal snow pack). Connelly et al. (2010) caution that the natural variation in vegetation, the dynamic nature of sagebrush habitats, and the variation in the habitats selected by sage-grouse across a landscape imply that characterizing habitats using a single value or narrow range of values (e.g., 15 to 25% sagebrush canopy cover for breeding habitats; Connelly et al. 2000*b*) may be inappropriate; additionally, seasonal habitats often overlap (Connelly et al. 1988), thus the differing seasonal requirements of sage-grouse may dictate that multiple attributes in a particular site are important (e.g., winter habitat may also provide brood habitat, thus the shrub overstory and the forb understory are important). Interspersion and juxtaposition of the differing cover types used by sage-grouse on an annual basis likely have a great influence on the effectiveness of a given part of the landscape to provide sage-grouse with useable habitat (Connelly et al. 2010).

SEASONAL HABITAT SELECTION

Sage-grouse population persistence has been linked to the availability of sagebrush habitat; the dependence of the species on sagebrush through all seasonal periods has been well documented (Connelly et al. 2004). Aldridge et al. (2008) predicted that areas where sage-grouse populations had persisted (compared to areas where populations were extirpated; estimated from historical and current species distribution maps [Schroeder et al. 2004]) across the range of the species were at least 25% and preferably 65% of the landscape within a 30 km radius was dominated by sagebrush. Wisdom et al. (2010) corroborated these results, estimating that when sagebrush habitats occupied <27% of a landscape delineated by 18-km radius circles, populations had a high risk of extirpation while a high probability of population persistence was estimated when >50% of the landscape was dominated by sagebrush. The sage-grouse habitat management guidelines (Connelly et al. 2000*b*) describe productive arid breeding habitats (lek attendance, nesting, and early brood rearing occur in breeding habitats) as those with 15 to 25% sagebrush canopy cover, sagebrush 30 to 80 cm tall, and ≥15% canopy of grasses and forbs ≥18 cm tall; productive mesic breeding habitats are described as areas with 15 to 25% sagebrush canopy cover, sagebrush 40 to 80 cm tall, and ≥25% canopy of grasses and forbs ≥18 cm tall. Arid and mesic winter habitats are described as areas with 10 to 30% sagebrush canopy cover 25 to 35 cm tall; sagebrush exposure above the snow is a critical component of suitable wintering sites (Connelly et al. 2000*b*). Mesic and arid sites need to be defined at local scales based on annual precipitation, vegetative species (shrub overstory and herbaceous understory) composition, and soils.

LEK

Sage-grouse are polygamous and exhibit consistent breeding behavior on ancestral strutting grounds (leks) annually (Patterson 1952). Leks are situated in areas with minimal shrub cover adjacent to relatively dense sagebrush stands where strutting male exposure is maximized, but escape, thermal, and feeding cover is readily available (Patterson 1952, Gill 1965). In southeastern Idaho, Dalke et al. 1963 documented leks on open or cleared areas 0.04 to 4 ha in size. Scott (1942) observed that lek sizes generally ranged from 0.25 to 16 ha, and Hofmann (1991) reported a mean size of 36 ha for large leks in a study in central Washington. Although the vast majority of leks occur in naturally formed open areas within the sagebrush (including stream channels, ridges, and grassy meadows; Patterson 1952, Dalke et al. 1963), sage-grouse males have been documented using as leks a 5-year old burn, gravel pits (Connelly et al. 1981), stock ponds, sheep bedding grounds, plowed fields, and roadside areas cleared during road construction (Dalke et al. 1963).

During the breeding season, males display in early morning and evening hours, traveling up to 2.1 km (Ellis et al. 1987) from the lek to day-use feeding and resting areas. Male day-roost locations in northeastern Utah were generally 0.5 to 0.8 km from the lek, and 82% of male day roost locations in central Montana were between 0.3 and 1.8 km from the lek (Wallestad and Schladweiler 1974). Minimum core day-use areas of males in northeastern Utah were 0.25 km^2, and grouse were reported to often walk to day-use sites from leks (Ellis et al. 1989). In central Montana, sagebrush canopy cover at 51% of male day-roost locations was between 20 and 40%, and no day-roost locations were recorded in areas with ≤10% sagebrush canopy cover (Wallestad and Schladweiler 1974). Male day-roost locations in Utah were in areas with taller sagebrush and greater sagebrush canopy cover relative to random locations (Ellis et al. 1989).

The most important characteristic for leks may be their proximity and configuration with nesting habitat (per theories of lek evolution and mating behavior; Gibson 1996). In non-migratory populations, leks generally occur within nesting habitat, and may be situated near the center of seasonal ranges (Eng and Schladweiler 1972, Wallestad and Pyrah 1974, Wallestad and Schladweiler 1974). Doherty (2008) suggested that landscapes with active leks contained twice the amount of nesting habitat (as predicted by models generated from habitat variables at 0.35, 0.10 km, and local scales; see further explanation of models below in nesting section) as available locations in northern Wyoming. However, lek locations for migratory populations typically do not exhibit this pattern; female dispersal routes between wintering and nesting areas, rather than vegetation type, may influence lek locations in these instances (Dalke et al. 1963, Bradbury et al. 1989, Wakkinen et al. 1992, Gibson 1996).

PRE-NESTING

Sage-grouse females retire into the vicinity of their nest location within a few days of being bred, and remain relatively sedentary until they nest (Patterson 1952). Little information is available documenting pre-nesting habitat selection. However, diet analysis on females during the pre-laying period (March and early April) in Oregon indicated that hens preferentially selected forbs over sagebrush as food (Barnett and Crawford 1994).

NESTING

Sage-grouse nesting habitat is often a broad area between winter and summer range (Klebenow 1969). Holloran and Anderson (2005) reported that a majority (67%) of female sage-grouse nested within 5.5 km of the lek-of-capture in central and southwestern Wyoming; 85% nested with 8 km. The authors suggested that nest distributions were spatially related to lek location within 5 km of a lek in the relatively contiguous habitats found in these regions. However, there was no difference between nest-to-lek and random point-to-lek distances in southeastern Idaho, suggesting nests in this study were placed without regard to lek location (Wakkinen et al. 1992).

Distances between consecutive-year nests (females followed through consecutive nesting seasons) suggest female fidelity to nesting areas. Median distance between consecutive-year nests was 415 m throughout central and southwestern Wyoming (Holloran and Anderson 2005), and median distance moved between consecutive-year nests was 740 m in southeastern Idaho (Fischer et al. 1993). Additionally, although sample sizes were low (n = 3), yearling females nested in the same general area as their mother (Lyon 2000), suggesting fidelity for a specific area could carry over to subsequent generations. Preliminary information from a study currently (2009) being conducted in northwestern Colorado supports generational fidelity to nesting areas (T. Apa; CDOW, personal communication).

Selection of specific habitat features within a landscape by nesting sage-grouse has been extensively documented. Hagen et al. (2007) conducted a meta-analysis of sage-grouse habitat selection data collected from throughout Wyoming, Montana and Canada, and determined that nesting females consistently select areas with more sagebrush canopy cover and taller grasses than available (i.e., random) locations. Holloran et al. (2005) investigated nesting habitat selection throughout central and western Wyoming, and reported that females selected areas with increased sagebrush canopy cover and residual (e.g., standing dead) grass cover and height. Consistent results from studies conducted throughout the species range include (Klebenow 1969, Wallestad and Pyrah 1974, Wakkinen 1990, Fischer 1994, Sveum et al. 1998*b*, Aldridge and Brigham 2002, Holloran et al. 2005): (1) nests were located under larger sagebrush bushes with more obstructing cover relative to within patch characteristics; (2) selected nesting habitat had more sagebrush canopy cover and taller sagebrush compared to available habitats; and (3) other relatively consistent differences included: increased sagebrush density, taller live and residual grasses, increased live and residual grass cover, and decreased bare ground at selected nesting sites compared to randomly-selected sites.

Doherty (2008) suggested that sage-grouse select nesting habitat at multiple spatial scales; in northern Wyoming, sage-grouse selected for sagebrush canopy cover at the site scale, for patches of high-density sagebrush and flat topography at 0.10 km scale, and against conifer and grassland at 0.10 km and riparian cover at 0.35 km scales. Holloran et al. (2010) additionally suggested that yearling sage-grouse females select nest sites at the scale of their life-time nesting area (based on fidelity exhibited by adult females, a female will nest within a 272-ha area over its lifetime).

EARLY BROOD-REARING

Sage-grouse chicks are precocial and move immediately following hatch to search for food (Patterson 1952); females generally rear their broods for the first 2 to 3 weeks in immediate vicinity of their nest (Connelly 1982, Berry and Eng 1985). Early brood-rearing areas (between 2 weeks post-hatch and prior to July 8) were located between 1.6 and 3.2 km of the nest in central Wyoming (Heath et al. 1998), and between 0.2 and 5.0 km of the nest during the first 4 weeks post-hatch in western Wyoming (Lyon 2000). In southwestern Wyoming, 80% of early brood locations were within 1.5 km of the nest (Slater 2003). During June and July in central Montana, brood use areas averaged 86 ha (Wallestad 1971).

In central Montana, 100% of the brood observations during June were in sagebrush-grassland habitat type (Peterson 1970). Between 75 and 80% of brood locations from June 1 through June 15 were in areas with 1 to 25% sagebrush canopy cover; and between 50 and 55% of the locations during the same period were in areas with 10 to >25% sagebrush canopy cover based on visual assessment of habitat (Wallestad 1971). In south-central Wyoming, 68% of sage-grouse brood locations (brood transects were conducted from early June through August, but results were not presented by age of brood) were in sagebrush-grass or sagebrush-antelope bitterbrush (*Purshia tridentata*) habitat types (Klott and Lindzey 1990). Use of low sagebrush (*Artemisia arbuscula*) and mixed sagebrush (low and big mixed sagebrush) habitats in southeastern Oregon during the early brooding period was greater than expected based on the percentage of these habitats available to females (Crawford et al. 1992). Brooding females during the early brood-rearing (hatch through 6 weeks) stages in south-central Washington preferentially selected for big sagebrush (predominantly Wyoming big sagebrush [*A. t. Wyomingensis*)-bunchgrass habitats and against grassland habitats (areas devoid of sagebrush); 70% of locations were within big sagebrush-bunchgrass habitats (Sveum et al. 1998*a*).

During early brood-rearing, sage-grouse throughout Wyoming, Montana and Canada selected areas with more forb and grass cover, taller grasses, and less sagebrush canopy cover than random sites (Hagen et al. 2007); the authors considered habitat selection from hatch through 6 weeks post-hatch early brood-rearing. A study conducted throughout Wyoming documented grouse broods selecting relatively dense sagebrush stands between hatch and 2 weeks post-hatch, presumably for thermal and predator protection of young chicks (Thompson et al. 2006). Brood-use sites within big sagebrush-dominated habitat type in southeastern Idaho had decreased big sagebrush density within a 122-m^2 area and canopy cover within 7.6 m, and increased percent frequency of yarrow (*Achillea millefolium*), lupine (*Lupinus spp.*), dandelion (*Krigia dandelion*), and salsify (*tragopogon porrifolius*) within 7.6 m relative to random locations within the same habitat type (habitat use relative to brood age was not discussed (mean brood ages between 1 to 8 weeks), however brood locations compared in use vs. available analysis were in big sagebrush-dominated habitats, suggesting early stages of brood-rearing; Klebenow 1969). When broods were found in grass-forb open areas in south-central Wyoming, use sites had more shrub cover relative to random openings, and dandelion, knotweed (*Polygonum* ssp.), yarrow, and salsify were more abundant at sage-grouse selected brooding sites relative to random sites (Klott and Lindzey 1990). Forb cover in general was higher at sites used by sage-grouse broods during early brood-rearing periods compared to random sites in Washington (Sveum et al. 1998*b*) and southern Canada (Aldridge and Brigham 2002). In southeastern Idaho, Fischer (1994) reported higher Hymenoptera (ants, bees, wasps) abundance and higher Orthoptera (grasshoppers, crickets) frequency, but no difference in abundance of Coleoptera (beetles) at brood use vs. random sites.

LATE BROOD-REARING

Sage-grouse broods remain in sagebrush habitats until range desiccation induces them to move to habitats still supporting succulent vegetation (Peterson 1970, Wallestad 1971, Neel 1980, Fischer et al. 1997). Brooding females may remain in upland habitats if suitable microsite conditions (e.g., swales, ditches, and springs) are found (Wallestad 1971, Fischer et al. 1996, Hausleitner 2003) or if weather conditions result in forbs remaining succulent in these habitats throughout the summer (Holloran 1999). The beginning of late brood-rearing coincides with forb desiccation but also with changes in chick diets from predominantly insects to forbs (Patterson 1952, Klebenow and Gray 1968, Klebenow 1969, Peterson 1970, Drut et al. 1994*a*). Late brood-rearing habitats are generally used from July to early September (Patterson 1952, Dalke et al. 1963, Gill and Glover 1965, Savage 1969, Wallestad 1971, Connelly et al. 1988).

Sage-grouse in southeastern Idaho moved as far as 82 km from breeding and nesting to summer ranges (Connelly et al. 1988), and Fischer et al. (1997) recorded movements up to 62 km from nesting to summer habitats. Klebenow and Gray (1968) observed grouse migrating 8 to 24 km to summer ranges; grouse in this population summered at higher elevations ranging from 1,600 m to over 2,150 m. Wallestad (1971) reported that some broods only traveled short distances to summer habitats, whereas others moved as far as 5 km.

Stand structure and food availability are characteristics most frequently associated with habitat selection by sage-grouse during the summer (Klebenow 1969, Autenrieth 1981, Aldridge and Brigham 2002). Sage-grouse may use a variety of sagebrush habitats and other habitats (e.g., riparian, wet meadows and irrigated agricultural fields adjacent to sagebrush habitats) during summer. Dunn and Braun (1986) reported grouse select feeding habitat near edges of sagebrush-dominated cover types with more horizontal and vertical cover and less variation in shrub densities and size compared to random; hens with broods selected during day-roosting sites with more horizontal cover and greater variation in sagebrush canopy cover than random sites, but fed in open homogeneous areas during morning and afternoon periods (Braun 1986). Sveum et al. (1998*a*) found that morning and afternoon feeding locations differed from midday roosting and random locations by having taller grass and less shrub cover and height. Klott and Lindzey (1990) reported that broods using large openings and meadows foraged on the edges and avoided the centers.

Hagen et al. (2007) suggested that sage-grouse throughout Wyoming, Montana and Canada selected areas with increased forb and grass cover, but similar sagebrush canopy cover to random sites during late brood-rearing periods (>6 weeks post-hatch). Peterson (1970) reported that forb canopy cover averaged 33% at brooding sites in Montana; in Colorado, broods used wet meadows with mean 41% forb canopy cover (Schoenberg 1982); Sveum et al. (1998*a*) reported that females selected areas with 19 to 27% forb canopy cover for late brood-rearing in Washington; in Idaho, Apa (1998) reported sites used by broods had twice as much forb cover as did random locations; and Hausleitner (2003) reported females selected brood-rearing sites with higher forb canopy cover and less bare ground than random sites. Researchers in southeast Alberta suggested that 12-14% forb canopy cover may represent the minimum needed for brood habitat (Aldridge and Brigham 2002).

FALL

Fall is a transitional period for sage-grouse (Wambolt et al. 2002), during which sage-grouse diets change from a variety of forbs, insects, and sagebrush to predominantly sagebrush (Rasmussen and Griner 1938, Patterson 1952, Leach and Hensley 1954, Gill 1965, Wallestad et al. 1975). Fall habitats used by sage-grouse can vary widely, based on availability, elevation, topography, water, distance between summer and winter habitats, and weather conditions. These habitats are generally used from as early as late August to as late as mid-December (Patterson 1952, Dalke et al. 1963, Gill and Glover 1965, Savage 1969, Wallestad 1971, Connelly 1982, Connelly et al. 1988). In Idaho, movements from fall sagebrush habitats to winter range were generally slow and meandering, beginning in late August and continuing into December (Connelly et al. 1988). During periods of early, severe winter snowstorms sage-grouse may begin migrations to winter habitats, but at the onset of milder weather later in the fall may return to sites adjoining late brood-rearing habitat (Patterson 1952).

During early fall, habitats may include upland meadows, riparian areas, greasewood (*Sarcobatus vermiculatus*) bottoms, alfalfa (*Medicago sativa*) fields, irrigated native hay pastures, and sagebrush (Batterson and Morse 1948, Patterson 1952, Gill 1965, Gill and Glover 1965, Savage 1969, Wallestad 1971, Connelly 1982). As vegetation in these habitats desiccates and frost kills herbaceous vegetation, sage-grouse form larger flocks and begin using sagebrush habitats more often (Batterson and Morse 1948, Patterson 1952, Gill and Glover 1965). During early fall in Colorado, sage-grouse abandoned irrigated native hay meadows in response to the cessation of irrigation, mowing of hay, and killing frosts (Gill and Glover 1965). Wallestad (1971) found that sage-grouse used higher-density sagebrush habitats in the fall than during the late brood-rearing period as sage-grouse broods that had occupied bottomland vegetation types (greasewood and alfalfa fields) shifted back into sagebrush in late August and September; this shift coincided with the transition to a winter diet of sagebrush (Wallestad 1975). During the fall in Colorado, sage-grouse used the same upland sagebrush habitats used for breeding; however, their use in the fall appeared random, and not tied to lek location, as it was during the breeding season (Gill 1965).

WINTER

Sage-grouse may travel many kilometers or only short distances between seasonal ranges (Eng and Schladweiler 1972); migratory populations often travel 80 to 160 km to winter ranges (Patterson 1952), while sedentary populations increase flock size and move from meadows into nearby sagebrush habitats during winter (Autenrieth 1981). A precipitation event (usually snow) or a drop in the temperature initiates migration, which begins in late August (in advance of snow accumulation) and continues until December (Dalke et al. 1960, Berry and Eng 1985, Connelly et al. 1988). In certain areas and during certain winters when snow depths are sufficient to cover most sagebrush plants, suitable winter habitat (e.g., areas where plant exposure above the snow is maintained) may be the most limiting seasonal habitat (Patterson 1952, Beck 1977), with sage-grouse over a broad summering area congregating on smaller, traditional wintering grounds (Beck 1977, Berry and Eng 1985).

Selection of wintering habitats by sage-grouse is influenced by snow depth and hardness, topography (i.e., elevation, slope, and aspect), and vegetation height and density (Batterson and Morse 1948, Gill 1965, Greer 1990, Schroeder et al. 1999). The primary requirement of wintering sage-grouse is sagebrush exposure above the snow (Patterson 1952, Hupp and Braun 1989, Schroeder et al. 1999, Connelly et al. 2000*b*, Crawford et al. 2004). Winter ranges are characterized by large expanses of dense sagebrush (>20% sagebrush canopy cover) on land with south to west-facing slopes of <5% gradient (Eng and Schladweiler 1972, Beck 1977). Doherty et al. (2008) reported that sage-grouse selected large expanses of sagebrush with gentle topography and avoided conifer and riparian habitats during the winter in northern Wyoming. Robertson (1991) reported that sage-grouse in Idaho selected areas with increased Wyoming big sagebrush canopy cover and average height compared to available habitats during the winter. Sage-grouse selected wintering areas in Colorado having greater sagebrush cover and heights (2-3X taller) at use versus random sites (Schoenberg 1982). Connelly (1982) reported that wintering grouse in Idaho selected areas with taller sagebrush than random sites; movement information additionally suggested that sage-grouse moved to areas supporting taller sagebrush as snow depth increased. In central Montana, sage-grouse foraged during winter in big sagebrush with a mean canopy cover of 28% and were observed most often in areas with >20% sagebrush canopy cover (Eng and Schladweiler 1972). Dalke et al. (1963) reported that wintering sage-grouse in Idaho preferred black sagebrush (*A. nova*) when these shrub species were available (generally on wind-swept ridges). In Utah, wintering sage-grouse preferred shrub-dominated habitats with medium to tall (40 to 60 cm) shrubs and shrub canopy cover between 20 and 30%, and avoided areas characterized by shrub heights between 40 and 49 cm and sagebrush canopy cover <14% (Homer et al. 1993).

During severe winters, flat area usage diminishes after snow pack exceeds 30 cm, and drainages and steeper southwest facing slopes are used (Autenrieth 1981, Hupp and Braun 1989). Drainages are sheltered from the wind and often contain taller sagebrush stands, snow drifts (used for roosting to escape extreme cold), and closed shrub canopies, which combined provide food and reduce thermoregulatory costs (Hupp and Braun 1989, Homer et al. 1993, Heath et al. 1996). Because sagebrush exposure is critical for feeding, wind scoured ridge-tops provide suitable foraging areas until wind velocities exceeding 15 to 25 km per hour force grouse off these areas (Eng and Schladweiler 1972, Beck 1977).

2. ***Establish the demographic parameters (e.g., nesting success, chick survival, adult survival) influencing sage-grouse population growth within the proposed project area; establish the current condition (e.g., habitat quality assessment) of sage-grouse seasonal habitats occurring in proposed project area.***

If radio-marked sage-grouse are a component of pre-management data collection, the demographic data supplied by the sample can be effectively used to model population growth and assess the relative affect of individual vital rates (e.g., nesting success) on that growth; a radio-marked sample is required to effectively monitor adult survival. Populations can additionally be monitored through lek counts, brood surveys, and wing surveys; Connelly et al. (2003) provide a detailed description of these monitoring techniques. The demographic(s) limiting populations may also be deduced by establishing potentially limiting habitats within a proposed project area (addressed in step 1).

Establishing the condition of habitat patch characteristics that are correlated with productive sage-grouse seasonal habitats may be an effective technique to assess the quality of existing seasonal habitats across a project area. A relatively recent paradigm shift that has occurred for describing vegetation dynamics on rangelands is explained through state-and-transition models in conjunction with ecological site descriptions (discussed at length in step 6A below), and developing vegetation monitoring protocol within the framework of these theories would benefit monitoring results. The purpose of state-and-transition models is to produce a management-focused theory hypothesizing potential vegetation dynamics for an ecological site in a non-linear framework (Stringham et al. 2003). This framework is based on potential alternative vegetation states on the site, potential transitions between vegetation states, and recognition of natural and management actions resulting in transitions (Briske et al. 2005). However, quantified information establishing thresholds between states, management alternatives that effectively transition habitats between states, and management alternatives maintaining habitats in a desired state is limited for sagebrush ecosystems. Additionally, ecological site descriptions as currently developed are focused on gathering information pertinent to livestock producers, not wildlife. Although these theories have yet to be supported with quantified information for sagebrush systems, they are being promoted by many land managers and scientists (for example Pyke 2010) as effect description and management tools.

Current conditions within the proposed project area should include information on past management history of the site, and spatial and temporal history of past habitat treatments (including naturally occurring events [e.g., wildfire]). Government and ranch grazing and treatment records in conjunction with historical aerial photographs (often available from BLM, NRCS, State wildlife management offices (e.g., Game and Fish Departments), or University GIS clearinghouses) can be used to assess historic management of a particular landscape.

The following sections provide information on landscape and vegetation characteristics that influence sage-grouse productivity and adult survival.

LEK

Across the range of the species, Johnson et al. (2010) suggested lek trends (numbers of males occupying leks) were positively associated with increased proportions of tall stature sagebrush (big sagebrush species) and all sagebrush within 5 and 18 km. Walker et al. (2007) found strong support for models predicting lek persistence in northern Wyoming with positive effects associated with the proportion of sagebrush habitat within 6.4 km. Smith (2003) reported that sagebrush within 1.5 km of active leks in North and South Dakota was taller than sagebrush around inactive leks, and that active leks in North Dakota had increased sagebrush density, forb cover and bare ground within 1.5 km, and the percentage of habitat converted from sagebrush (tilled) within 4 km was less, compared to inactive leks in North Dakota.

PRE-NESTING

Barnett and Crawford (1994) reported that during a year when forbs comprised 45 to 50% of female diets compared to a year when forbs were approximately 18% (sagebrush made up the remaining bulk of the diet in both years) of female diets during March and the first week of April in Oregon, productivity estimates as measured by chicks per female were higher. Additionally, increased spring total and food forb cover, and tall (>18cm) grass cover was correlated with increased overall nest initiation rates, renesting rates, and nesting success rates in southeastern Oregon (Coggins 1998).

NESTING

The relationship between shrub and herbaceous characteristics at nest sites and nest success may be considered relatively unclear, especially when investigating nest success over relatively small spatial and temporal scales. Studies in southeastern Idaho (Fischer 1994), south-central Wyoming (Heath et al. 1998), central Wyoming (Holloran 1999), west-central Wyoming (Lyon 2000), and southwestern Wyoming (Slater 2003) failed to find any vegetative differences between successful and unsuccessful nests. However, when combining results from multiple studies over larger scales (anecdotally accomplished here), it generally appears that grouse nesting success is influenced predominantly by the herbaceous understory. However, although conditions in the herbaceous understory may increase the probability of a successful hatch, female sage-grouse across the range of the species consistently select relatively dense sagebrush stands for nesting (Hagen et al. 2007), therefore tall, dense herbaceous cover in dense sagebrush stands increases the probability of a successful nest. Vegetation consistently higher at successful compared to unsuccessful sage-grouse nests throughout the range of studied populations included: live and residual grass height, live and residual grass cover, forb cover and visual obstruction (Wakkinen 1990, Gregg et al. 1994, Sveum et al. 1998*b*, Popham 2000, Aldridge and Brigham 2002, Hausleitner 2003, Moynahan 2004, Holloran et al. 2005, Rebholz 2007).

Nest success in sage-grouse ranges from 15 to 86% (Schroeder et al. 1999). Heath et al. (1996) maintained that the chance of a sagebrush nest successfully hatching will increase 30% if it is within herbaceous vegetation exhibiting 20% canopy cover and heights of 15 to 30 cm. The residual herbaceous component is important during the initial stages of incubation because nests are initiated prior to the growing season for most grasses and forbs (Crawford et al. 1992, Heath et al. 1996). In Utah, nesting success was highest in areas with sagebrush >46 cm tall, with canopies >50%, and "where a good understory of grasses and weeds were present;" the presence of a good herbaceous understory interspersed throughout sagebrush stands increased the probability of a successful hatch relative to sagebrush stands of equal density without the understory (Rasmussen and Griner 1938). Sagebrush canopy cover was greater at successful vs. unsuccessful sage-grouse nests in Montana (Wallestad and Pyrah 1974). Sveum et al. (1998*b*) reported that successful nests in Washington had increased residual herbaceous cover compared to unsuccessful nests. In Oregon, tall (>18 cm) residual grass cover and medium height (40 to 80 cm) shrub cover were greater at successful vs. unsuccessful nests (Crawford et al. 1992, Gregg et al. 1994), and a combination of shrub and herbaceous screening cover were important for nest success in Idaho (Connelly et al. 1991). Successful nests in southern Canada had taller grasses, taller palatable forbs, and decreased grass cover relative to unsuccessful nests (Aldridge and Brigham 2002). In California, percent rock cover, total shrub height, and visual obstruction were greater at successful than unsuccessful nest sites (Popham 2000). Hausleitner (2003) reported that successful nests in northwestern Colorado had increased average forb and grass cover and taller grasses compared to unsuccessful nests. Moynahan (2004) and Rebholz (2007) noted greater canopy cover of live grasses at successful compared to unsuccessful nests in Montana and northwestern Nevada, respectively.

EARLY BROOD-REARING

A key factor associated with sage-grouse productivity is brood-rearing habitat availability (Crawford et al. 1992). Low chick recruitment has been proposed as a factor limiting sage-grouse population stability (Connelly and Braun 1997), and most chick mortality occurs prior to the flight stage (2 to 3 weeks) when decreased mobility increases vulnerability to predation and starvation (Patterson 1952, Autenrieth 1981). Sage-grouse chicks require protein-rich foods, including insects and forbs, for survival (1 to 10 days post-hatch) and optimal development (10 to 45 days post-hatch; Johnson and Boyce 1990). Sage-grouse productivity in Oregon was higher in areas where chick diets consisted of 80% forbs and insects compared to where chicks ate primarily (65%) sagebrush (Drut et al. 1994*a*).

Thompson et al. (2006) combined early brood-rearing (hatch through 2 weeks post-hatch) data collected from 3 sites in central and southwestern Wyoming between 1999 and 2003, and found that the proportion of females with confirmed chicks 14 days post-hatch was positively related to Coleoptera (beetles) abundance and total herbaceous cover at selected sites, and that the number of juveniles per female (estimated from wing barrel collections during fall harvest) was positively associated with the abundance of Hymenoptera (ants, bees and wasps) and grass cover.

In central Montana, percent crop volume from chicks 1 to 4 weeks old (n = 26) was 70% vegetation and 30% insect matter (primarily beetles and ants; Peterson 1970); however, the author reported that crop volume of insects in chicks <7 days old (n = 2) was 60%. The most preferred plant species collected from crops of chicks 1 to 2 weeks old (n = 8) were dandelion, salsify, pepperweed (*Lepidium* spp.), and gumweed (*Grindelia squarrosa;* Peterson 1970). In Idaho, chicks <7 days old (n = 4) had 48% crop volume of vegetation and 52% crop volume of insects (primarily ants and beetles), while chicks 7 to 14 days old (n = 4) had crop volumes of 90% vegetation (primarily Tapertip Hawksbeard (*Crepis acuminata*), Harkness Gilia (*Linanthus harknessii*), and dandelion) and 10% insects (primarily ants and beetles; Klebenow and Gray 1968). In southeastern Oregon, forbs (primarily Hawksbeard, milkvetch (*Astragalus* spp.), dandelion, clover (*Trifolium* ssp.), and broomrape [*Orobanche* spp.]) and beetles were selected as food items above available levels (Drut et al. 1994*b*).

LATE BROOD-REARING

Interestingly, little information correlating chick survival with the condition of late brood-rearing habitats exists. Aldridge (2000) suggested that broods in Canada did not move from sagebrush uplands to mesic habitats during wet years, and chick productivity may have been enhanced in these situations; conversely, mesic areas supporting succulent vegetation throughout the summer may be limiting in dry years, ultimately causing low recruitment. In years when suitable summering habitats were limiting in western Wyoming, drought conditions concentrated birds which potentially resulted in increased predation rates and increased adult hen late summer mortality (Heath et al. 1997). Results from Wyoming in 1998 and 2004 (Holloran 1999, 2005) suggested that sage-grouse preferred to remain within sagebrush-dominated habitats throughout the summer, and resorted to concentrating on mesic areas only after upland forb desiccation; this information suggests that the distribution and extent of suitable summering habitats could be important to sage-grouse summer survival.

FALL

No studies examining habitat suitability during migratory or fall transitional periods could be found. Fall habitat selection information discussed above suggests that juxtaposition to and continuity between summering and wintering areas of sagebrush-dominated habitats are important to provide migrating sage-grouse with suitable transition-range habitats. Jensen (2006) suggested that migrating sage-grouse in Wyoming spent little time in particular locations, and that selection of transition range habitats was not based on microsite conditions (i.e., vegetative conditions within 15 m of use site) but may be based on conditions at larger spatial scales.

WINTER

During the winter, sage-grouse rely almost exclusively on sagebrush exposed above the snow for forage and shelter (Rasmussen and Griner 1938, Patterson 1952, Remington and Braun 1985, Robertson 1991, Schroeder et al. 1999, Connelly et al. 2000*b*, Crawford et al. 2004). Sage-grouse populations could be restricted to <10% of the sagebrush-dominated lands in any given area during the winter (Beck 1977). Sage-grouse populations will utilize critical winter habitat once every 8 to 10 years, these locations providing food and thermal protection when increased snow pack has covered most surrounding areas (Heath et al. 1996). Holloran and Anderson (2004) and Moynahan (2004) report that severe winter weather (heavy snow and extreme cold) in areas with limited suitable wintering habitats correspond to sage-grouse population declines. Connelly et al. (2000*b*) stress that across a landscape winter habitats need to allow access to exposed sagebrush under all snow conditions.

Remington and Braun (1985) suggested that sage-grouse selectively feed on Wyoming big sagebrush during the winter due to its relatively high crude protein (nitrogen) content and reduced monoterpene levels compared to other big sagebrush sub-species. But, Welch et al. (1991), comparing food selection by pen-reared (captured wild) sage-grouse, found that birds preferred mountain big sagebrush (A. t. *vaseyana*). Sage-grouse may express preference while selecting both foraging plants and sites, but are capable of shifting their eating habits when either sagebrush quantity or quality becomes limiting (Remington and Braun 1985, Welch et al. 1991); again, sage-grouse distribution in the winter is affected primarily by sagebrush exposure rather than differences in nutritional quality of forage (Hupp and Braun 1989).

ADULT SURVIVAL

Zablan et al. (2003), using band-recovery data from over 6,000 banding individuals in Colorado, estimated 59% annual survival for adult females, 78% for yearling females, 37% for adult males, and 63% for yearling males. In Wyoming, 67% annual survival for females and 59% for males was estimated from over 3,000 banded individuals (Schroeder et al. 1999 after June 1963). Moynahan (2004) investigated factors influencing monthly survival of female sage-grouse in Montana, and reported that breeding status (nesting or non-nesting), environmental condition, and exposure to hunting resulted in variable seasonal survival probabilities. Environmentally, severe winter weather (heavy snow and extreme cold) and the emergence of West Nile virus (Naugle et al. 2004) reduced sage-grouse survival during an annual winter and fall period, respectively, whereas drought conditions (throughout the year) resulted in increased annual survival (Moynahan 2004). In contrast, Heath et al. (1997) suggested that drought conditions resulted in female sage-grouse concentrating on limited available summering habitat, facilitating prey search for both hunters and natural predators in Wyoming. Holloran and Anderson (2004), studying a sage-grouse population that experienced severe winter weather regularly in northwestern Wyoming, reported that changes in long-term lek counts correlated well with winter precipitation levels, suggesting that winter survival may influence adult survival.

3. ***Establish clear and concise management objectives based on sage-grouse population and vegetative condition information.***

The establishment of clear and concise objectives is a necessary and critical first step to developing an effective management or mitigation plan. An effective objective is one that can be used to develop suitable monitoring (e.g., specific enough to suggest specific monitoring needs), and should be based on, and specific to, information obtained by completing steps 1 and 2 above. For example, a suitable objective may be as follows: "increase forb diversity within delineated patches (see relevant figure of project area) to enhance insect populations within in these patches and increase sage-grouse early brood chick survival (e.g., hatch through 2 weeks post-hatch)." This objective suggests that sage-grouse population monitoring indicated chick survival during the early stages of incubation may be limiting population growth within a given area, the patches suitable for early brood-rearing within this given area are known, and vegetative monitoring suggested that forb diversity and associated insect populations were suboptimal within these suitable patches. This objective additionally suggests that sage-grouse early brood chick survival, forb diversity, and insect abundance are variables that should be monitored post-management. Finally, this objective focuses potential management options. Avoid establishing objectives such as "improve habitats for sage-grouse to increase populations."

To effectively manage sage-grouse populations in a pro-active manner, habitat enhancements have 3 requirements: habitat manipulations need to occur in areas where habitat deficiencies exist, the type of manipulation needs to result in habitats that are enhanced for the season of interest, and individuals need to use enhanced areas to reap potential benefits. Management planning in these areas revolves around the idea that to enhance sage-grouse populations within a given landscape, the carrying capacity of that landscape needs to be increased. Areas that sage-grouse are selecting but habitat conditions are such that individuals using those habitats do not successfully reproduce or survive have been termed "attractive sink" habitats (Aldridge and Boyce 2007); these represent precisely the areas where habitats need to be enhanced.

The following section provides information pertinent to effectively establishing objectives and developing a management or mitigation plan.

Kiesecker et al. (2009) recommend planning on landscape spatial scales following the Environmental Impact Assessment (EIA) process. EIA is a systematic process that examines the environmental consequences of planned anthropogenic developments which emphasizes predicting and preventing environmental damage from the development. Those employing the EIA process seek to minimize impacts through the application of a mitigation hierarchy: avoid, minimize, restore, or offset. Kiesecker et al. (2009) suggest however that no quantitative guidelines exist to guide this decision-making process. The authors propose that landscape-level conservation planning is the process of locating, configuring, and maintaining areas managed to maintain viable populations of impacted biota. A conservation portfolio (the end product of conservation planning) that includes a selected set of representative areas should encompass the full distribution and diversity of impacted systems; the key feature of a conservation plan is a clear statement of a biodiversity vision that incorporates the full range of biological features, their current distribution, and the minimum viability needs each biological target requires for long-term persistence. By initially avoiding and minimizing impacts to biological targets, then ensuring that damaged ecosystems are restored, and finally offsetting any remaining residual impacts, Kiesecker et al. (2009) contend that a conservation planning (and implemented) framework can be provided that is consistent with sustainable development; considering planning at landscape scales throughout this process is essential, as it ensures that biologically and ecologically important features remain the core conservation targets.

Wisdom et al. (2005) recommend planning on multiple spatial scales with the multi-scale approach needed that links the scale of individual sagebrush stands with scales of the seasonal, year-round, and multi-population ranges of sage-grouse. The authors further recommend that consideration of connectivity across scales is essential; the sustained use of a comprehensive suite of passive and active restoration treatments over extensive areas is needed; and that a comprehensive set of species that depend on the sagebrush ecosystem needs to be targeted for restoration planning and monitoring.

Pyke (2010) outlines a triage technique for initially prioritizing management actions within a given landscape where habitat patches are grouped into three categories: ones where immediate care and intervention should be received as these actions will benefit these sites which may have significant damage or damage may be imminent if management does not change; the other two groups are at opposite ends of the care spectrum with ones needing no immediate intervention (future intervention may enhance these sites but is not currently needed), and the latter having terminal damage that cannot be rectified even with intervention. Assessments of the status or health of each patch relies on a manager's knowledge of the ecosystem's current status relative to the level of ecosystem threats (the drivers of ecosystem change) and the probability of ecosystem recovery from those threats; the ecological site description and accompanying state-and-transition model concepts discussed above provide a baseline for assessing land status. Bestelmeyer (2006) cautions that there should be 2 kinds of classification thresholds to develop robust state-and-transition models; these thresholds should be based either on preventive management (e.g., the processes resulting in conditions that make systems vulnerable to deterministic change) or restoration (e.g., transitions to the dominance of desired species along with preservation of the desired species). The author concludes by stating that restoration failures result when degradation thresholds are addressed but pattern and process thresholds (those required for system stability) are not.

Pyke (2010) continues his triage discussion by suggesting that land assessments can be used to develop ecosystem intervention grids, or grids that provide decision levels for prioritizing management actions and restoration options. As an example, a potential grid for sagebrush grassland ecosystems may include extent of departure from the reference state and the potential for patches to recover (e.g., resilience) after management changes or restoration activities ensue (areas with higher annual precipitation and greater soil depth provide approximations of increasing resiliency for most sagebrush grassland ecosystems). Intervention grids may also contain additional information such as cost-benefit analyses. These grids are meant to simplify relationships into decision categories, but may also be represented in continuous probability scales and modeled to formulate decision tools.

Land status evaluations should be done spatially over a larger landscape than the proposed project area as sage-grouse are considered a landscape-scale species (Connelly et al. 2000*b*) and populations using portions of a project area may rely on habitats situated outside it boundaries. Pyke (2010) suggests using GIS technology to assess the probability of restoration success if assessments are done in a spatially balanced and consistent manner. GIS data layers may include sage-grouse seasonal habitats, climate, soils, topography, and sage-grouse habitat use information. Pyke (2010) continues by citing where managers demonstrated a prioritization model for sagebrush restoration that used data layers derived from environmental conditions present by big sagebrush subspecies (separately), potential for connecting existing stands of sagebrush, locations of viable sage-grouse populations, and potential for cheatgrass (*Bromus tectorum*) invasion to impede success; the author suggests that the approach followed insured that the local restoration projects were considering regional factors which increased the probability of successfully achieving both restoration and improved sage-grouse habitat.

4. ***Establish potential management options to fulfill intended objectives, and identify potential impacts of proposed prescription(s) to sage-grouse.***

It is extremely important to note up front to this portion of the document that the enhancement or restoration of sagebrush-habitats is not a trivial task. There is tremendous uncertainty as to the vegetative and sage-grouse population outcomes of habitat manipulations. Although managers often justify habitat manipulations with potential long-term benefits, the long-term affects to habitats and sage-grouse of most of the available habitat manipulation options are unknown. The Sage and Columbian Sharp-tailed Grouse Technical Committee (2009) argues that there is little scientific evidence supporting the use of habitat treatments for sage-grouse conservation, but that there is considerable information documenting negative effects of treatments on sage-grouse.

The following sections provide information on habitat management options that may be employed to address objectives established in step 3; the potential impacts to sage-grouse of each proposed action are included within each section.

SAGEBRUSH TREATMENT

FIRE:--The rate of sagebrush reestablishment into a treated site depends on grazing management and intensity following the treatment, weather conditions (amount and timing of precipitation) directly following the treatment, the extent and condition of the sagebrush seedbank within the treatment boundaries, the number of live shrubs that remain following the treatment (smaller treatment areas have accelerated reestablishment of sagebrush due to the close proximity of a seed source), and the amount of litter on the soil (Johnson 1969, Harniss and Murray 1973, Knight 1994). Burning tends to result in the greatest reduction of sagebrush cover and has the most protracted effect on sagebrush when compared to other treatments (Watts and Wambolt 1996). Recovery from a burn to a 20% sagebrush canopy exceeds 35-40 years in Wyoming big sagebrush habitat types, 25 years in basin big sagebrush types, and 15-25 years in mountain big sagebrush sites (Harniss and Murray 1973, Wright and Bailey 1982, Bunting et al. 1987, Winward 1991, Watts and Wambolt 1996). Watts and Wambolt (1996) reported that Wyoming big sagebrush canopy cover had reestablished at levels below original levels 30 years post-burn. Although sagebrush in a burn in Idaho were approaching pre-burn densities 30 years post-burn, the majority of the plants in the burned plots were <6 inches (Harniss and Murray 1973). In southeastern Idaho, shrub structural features, including percent cover of Wyoming and three-tip big sagebrush and total shrub height, did not recover to pre-burn levels 14 years post-fire (Beck et al. 2009). Nelle et al. (2000) found that Mountain big sagebrush had reestablished to 8% canopy cover 14 years post-burn (versus 18% canopy cover on unburned plots) in Idaho. Wambolt and Payne (1986) reported reduced Wyoming big sagebrush canopy cover relative to a control in southwest Montana 18 years post-burn. Lesica et al. (2007) also reported slow recovery rates of Wyoming big sagebrush in southwestern Montana, observing less than 2% recovery after 23 years. Cooper et al. (2007) investigated paired burned and unburned sites in eastern Montana and estimated full recovery of Wyoming big sagebrush cover on burned sites to be >100 years.

In a recent analysis combining fire-scar data with recovery rates verses investigating recovery rates solely (as the studies cited above), Baker (2010) reported that the best available estimates of fire rotation (i.e., the expected time to burn once through a land area equal to that of a landscape of interest) averaged >200 years in low sagebrush, 200-350 years in Wyoming big sagebrush, and 150-300 years in mountain big sagebrush. Baker (2006) concluded that fire exclusion likely has had little effect in most sagebrush communities, and that the reintroduction of fire into these systems is currently not a restoration need (i.e., fire suppression has not influenced sagebrush-dominated ecosystems).

Fall burning has the least effect on cool season grasses, spring burning the least on warm season grasses, while early to mid-summer burns are the most damaging to the overall vegetation (as they occur during the growing season; Wright 1974, Bunting et al. 1987). Repeated burning in a given area can cause a shift to a community dominated primarily by warm-season species from communities dominated by a mixture of cool and warm-season species, and may reduce perennial abundance while increasing the abundance of annuals (Knight 1994).

Wetter habitats supporting mountain or basin big sagebrush (*A. t. tridentata*) typically respond favorably (herbaceously) to burning, while xeric (≤25cm annual precipitation) Wyoming big sagebrush habitats generally have little potential for herbaceous improvement post-burn (Clifton 1981, Bunting 1989, Bunting et al. 1987, Fischer et al. 1996, Miller and Eddleman 2001). Total herbaceous cover typically increases post-burn, but most of the increase results from stimulated grass production while forb production remains constant or declines below pretreatment levels (Martin 1970, Harniss and Murray 1973, Clifton 1981, Sime 1991, Sturges 1983, Connelly et al. 1994, Nelle 1998).

The increases in grass production immediately following the disturbance are typically followed by declines, regardless of sagebrush reestablishment. Thilenius and Brown (1974) reported that total average grass production changed from 458 pounds/acre pre-treatment to 1263 pounds/acre 3 years post-treatment to 361 pounds/acre 11 years post-treatment. The declines in grass production from 3 to 11 years post-treatment occurred despite minimal sagebrush reestablishment (18% canopy cover pre-treatment to 3% canopy cover 11 years post-treatment; Thilenius and Brown 1974). Thus, the stimulation in grass production could be related to the fire itself, not to the removal of sagebrush competition, as an increase in grass production can be observed following steppe fires outside the range of sagebrush (Daubenmire 1975).

In general, rhizomatous grasses recover quickly after a fire, bunchgrasses (e.g., crested (*Agropyron cristatum*) and bluebunch wheatgrass [*A. spicatum*]) recover in 1 to 2 growing seasons, while other bunchgrasses take from 4 to 6 seasons to recover (Wright 1986). Grass species that are harmed by fire include: Idaho fescue (*Festuca idahoensis*) and prairie junegrass (*Koeleria macrantha*); species that are unharmed by fire include: bluebunch wheatgrass, needle-and-thread (*Stipa comata*), and thurber needlegrass (*Stipa occidentalis*); and species which are favored by fire include: cheatgrass, Kentucky bluegrass (*Poa pratensis*), and crested, thickspike (*Elymus lanceolatus*), and western wheatgrass (*Pascopyrum smithii*; Payne and Bryant 1994). West and Hassan (1985) reported generally the same responses excepting that the standing crop (kg/ha) of needle-and-thread declined and Indian ricegrass (*Oryzopsis hymenoides*) remained unchanged 2 years post-burn in a Wyoming big sagebrush stand in Utah.

Reports of increases in forb cover post-burn result from the stimulation of a few forb species (e.g., dandelion and lupine may increase, while others are damaged; Mueggler and Blaisdell 1958, Pyle and Crawford 1996). Harniss and Murray (1973) reported that burning resulted in increased lupine (which accounted for most of their forb biomass in burned plots) up to 30 years post-burn; and West and Hassan (1985) reported that 2 years post-fire, the standing crop (kg/ha) of fleabane (*Erigeron* spp.), buckwheat (*Eriogonum* spp.), phlox (*Phlox longifolia*), pussytoes (*Antennaria rosea*), prickly-pear (*Opuntia polycantha*), and vetch (*Vicia americana*) declined while mustard (*Descuirania* and *Sisymbrium* spp.) and buttercup (*Ranunculus testiculatus*) increased.

Burning can cause a number of undesired effects in the vegetation and environment that are not associated with other treatments. Probably the most conspicuous difference is the controllability of other treatment options relative to fire: remnant sagebrush plants can be effectively maintained with the boundaries of a treatment (either through treatment pattern or a higher mower height); small, patchy treatment patterns can be produced; seedbank integrity is not affected; and treatment effectiveness is not weather-dependent so treatments can be conducted when least disruptive to wildlife. Chemical treatments often leave sagebrush skeletons that accumulate snow and thereby increase soil moisture (Mueggler and Blaisdell 1958, Omphile 1986). Since nearly all of the aboveground plant material is initially removed by burning, the soil is often exposed to accelerated erosion (Pechanec et al. 1965). Species such as rabbitbrush (*Chrysothamnus* spp.), horsebrush (*Tetradymia* spp.), and snakeweed (*Gutierrezia sarothrae*) are capable of sprouting and can increase significantly following fire (Harniss and Murray 1973, Wright and Bailey 1982, Winward 1985, Miller and Eddleman 2001). Fire can injure forb and woody species (e.g., antelope bitterbrush), and burning an area repeatedly can result in the elimination of species that require more time to recover from fire (Mueggler and Blaisdell 1958, Blaisdell et al. 1982). However, repeated brush management treatments (mechanical methods, chemical methods, or burning), particularly on semi-arid rangelands, alter plant succession, decrease ecosystem stability, and result in a loss of plant, animal, and habitat diversity (McKell 1975, Boyd et al. 1997).

HERBICIDES:--Important factors in determining the success of an herbicide spray treatment include: chemical type, season, time of day, method of application, and rate of spraying, with careful timing of application minimizing the effect on desirable forbs (Johnson 1958, Laycock 1979*b*). The expected treatment-life for sagebrush-dominated areas treated with herbicides is >15 years and <25-30 years (Braun 1998). Big sagebrush seedlings are most numerous when approximately 40-60% of original plants are killed; on areas protected from post-treatment grazing there are fewer seedlings compared to grazed areas; and seedlings are more abundant where overall sagebrush kill is light (Johnson 1958). Studying the response of mountain big sagebrush to a 2,4-D spray in southeastern Wyoming Sturges (1983) indicated that herbaceous productivity was 2.4 times greater the first 3 years post-treatment and remained 1.9 times that of untreated vegetation 10-11 years post-treatment; however, total aboveground plant biomass production on treated experimental units was less than on undisturbed experimental units, as the increase in herbaceous production did not fully compensate for the loss of sagebrush production. Increased grass cover tends to replace the forb and browse elements, and chemical treatments generally result in a 20-50% and 80-95% reduction in relative forb and browse production and canopy coverage, respectively (Carr 1968, Martin 1970, Kearl and Freeburn 1980).

Spike 20P is a clay pelleted product containing 20% tebuthiuron which is absorbed by the roots of sagebrush and trans-located to the shoots, inhibiting photosynthesis and killing plants in 1-3 years (Baxter 1998). The product is sensitive to the amount of clay and especially organic matter in the soil and dissociates best in soils with low pH (Johnson et al. 1996, Baxter 1998). At low application rates (sagebrush thinning rates), tebuthiuron has little or no effect on grasses, forbs, or brush species such as bitterbrush, winterfat (*Krascheninnikovia lanata*) or serviceberry (*Amelanchier* spp.; Baxter 1998). In semi-arid environments, moisture does not wet the entire soil profile, limiting tebuthiuron penetration and reducing the amount leached from the soil profile (Johnson and Morton 1989). In sagebrush-dominated, semi-arid environments, tebuthiuron and its metabolites were detected in current growing season foliage ≥5 years post-application and was detected in the soil >11 years post application (>90% was detected in the top 30 cm during the first 5 years, and 55-73% was detected between 60 and 90 cm of the soil profile 9 years post-treatment; Johnson and Morton 1991). Additionally, small amounts of tebuthiuron persisted in the surface 7 cm of soil ≤10 years post-application, preventing sagebrush seedling establishment (Johnson and Morton 1989).

Big sagebrush was controlled with tebuthiuron at application rates of 0.6-1.1 kg/ha (silver sagebrush was not completely controlled), while grasses were not significantly reduced with tebuthiuron ≤1.1 kg/ha (Whitson and Alley 1984). Grass production in areas with ≤12% big sagebrush cover did not significantly increase post-application, suggesting that a point of limiting return was reached regarding improvement of graminoid production through sagebrush reduction (Johnson et al. 1996). Additionally, plant community and small mammal diversity increased with low tebuthiuron application rates, but declined to below control levels as tebuthiuron application rates increased, again suggesting limiting returns (Olson et al. 1994). Tebuthiuron at rates of 0.2 kg/ha increased forb production, while rates of 0.4 and 0.6 kg/ha resulted in unchanged forb populations, indicating that lower rates of tebuthiuron may improve the availability of forbs (Doerr and Guthery 1983). However, forbs displayed small declines in percent composition with increased tebuthiuron rates and within treatment areas relative to controls and low application treatments (Whitson et al. 1988, Olson et al. 1994). Insect abundance and diversity were not significantly affected by low application tebuthiuron treatments (Doerr and Guthery 1983). McDaniels et al. (2005) provide an interesting example concerning forage production following tebuthiuron application to Wyoming big sagebrush sites in northwestern New Mexico: in treated sites grass yield remained near peak levels for about 14 years until sagebrush canopy cover reached about 6%; it was projected to take approximately 40 years before Wyoming big sagebrush canopy cover would be equivalent in treated and untreated sites, but grass yield was similar in only about 20 years. Olson and Whitson (2002) reported that thinning effects of tebuthiuron application in Wyoming lasted at least 14 years, that grass species diversity was not influenced by treatment, and the authors noted an increase in the biomass of rabbitbrush and an increase in Japanese brome (*Bromus japonicus*) and cheatgrass presence in thinned plots.

MECHANICAL TREATMENT:--Chaining, relative to other sagebrush control measures, has fewer long-term detrimental effects on sagebrush obligate bird species because seedling and young plants escape damage, allowing more rapid regeneration (Castrale 1982, Braun 1998). Additionally, mowed sites generally have increased arthropod population densities and decreased richness during the year of disturbance (Christianson et al. 1989). Hessary and Gifford (1979) reported that there appears to be a general trend for best overall vegetative (e.g., general cover) responses to treatment on loam soils (though significant decreases in production were also indicated on this type of soil in some sample plots); plowing was the least successful sagebrush treatment studied; the best cover responses on the various range improvement practices were found on contour furrowing treatments on sandy clay loam and loam textures soils; and a general trend indicated that production increases were slightly higher for more recent sagebrush ripping and sagebrush chaining treatments than for older ones. Watts and Wambolt (1996) investigated Wyoming big sagebrush control treatments: burning, spraying with 2,4-D, rotocutting, and plowing, along with controls (no treatment, rest from livestock) in southwestern Montana 30 years post-treatment. Burning had the longest-term effect on sagebrush cover; after 30 years, burned plots had lower sagebrush cover than controls; sagebrush that was sprayed, plowed and rotocut equaled the untreated areas after 18, 10 and 18 years, respectively. Fairchild et al. (2005) concluded that chaining in one direction with a light, unmodified anchor chain maintained in a "U" shape (verses the more aggressive "J" configuration), was shown to be an effective treatment for thinning Wyoming and mountain big sagebrush and enhancing shrub leader-growth.

Summers (2005) studied the effects of 6 mechanical treatments on the herbaceous production within Wyoming big sagebrush sites in Utah: (1) disk plow followed by a land imprinter, (2) 1-way chaining using an Ely chain, (3) 1-way pipe harrow, (4) 2-way pipe harrow, (5) meadow aerator (fall), and (6) meadow aerator (spring); each plot was seeded with a mixture of native and introduced grasses, forbs, and four-wing saltbush (*Atriplex canescens*). Two years post-treatment, the density of grasses present at time of treatment was significantly lower for the disk and imprinter than for the control and other mechanical treatments; the other 5 mechanical treatments had no effect on established grass density. Density of seeded grasses for the disk and imprinter treatment was over 3 times greater than that for any of the other treatments. Total perennial forb cover, including established and seeded species, was not significantly different from the control for any mechanical treatments; the disk and imprinter, 1-way harrow, and spring-aerator treatments each had significantly lower cover of established perennial forbs than the control and establishment of seeded forbs was very low for all sites.

Yeo (2009) studied the effects of mechanical shrub crushing (e.g., Lawson aerator) in a Wyoming big sagebrush / bluebunch wheatgrass vegetation type in east-central Idaho. The author reported that Wyoming big sagebrush was negatively impacted by crushing, but regeneration rates 4 years post-treatment suggested impacts were not as pronounced as those reported for prescribed fire. Accompanying crushing with seeding of bluebunch wheatgrass resulted in substantial increases in the cover of this species within 4 growing seasons. In areas grazed by livestock post-treatment squirreltail (*Elymus elymoides*) increased substantially, whereas in areas protected from livestock needle-and-thread increased substantially. There was no response by forbs to treatments or protection from cattle grazing within 4 growing seasons post-treatment.

SAGEBRUSH SPECIES SPECIFIC INFORMATION:-- The predominant sagebrush taxa and those sagebrush species that are most important to sage-grouse are three subspecies of big sagebrush: Wyoming big sagebrush, mountain big sagebrush and basin big sagebrush; two low or dwarf forms of sagebrush: little or low sagebrush and black sagebrush; and plains silver sagebrush (*A. cana cana*) which occurs primarily in the northeast portion of the sage-grouse range (Connelly et al. 2004, Miller et al. 2010). The following summary of sagebrush species' characteristics and response to management were taken from reviews completed by the Bureau of Land Management (2002) and the U.S. Forest Service Rocky Mountain Research Station (1999):

<u>Wyoming big sagebrush</u>:

Species Response to Fire: Wyoming big sagebrush is readily killed by fire; essentially any fire passing through a plant will cause mortality. However, some plants in Montana were reported to have survived the burning of lower branches. Wyoming big sagebrush does not sprout, and establishes after fire from seed produced by remnant plants that escaped fire or from plants adjacent to the burn; remnant plants are the principal means of post-fire reproduction; the seedbank is usually consumed in a fire. Wyoming big sagebrush seed is not disseminated for great distances so off-site sources are probably less important than seed produced on-site; shrubs surviving within the perimeter of a disturbed area provide a more important seed source than those on the perimeter. The vast majority of Wyoming big sagebrush seed produced during fall is gone by spring when few seeds persist.

Species Response to Mechanical Treatment: Mechanical or chemical treatments are generally more effective in Wyoming big sagebrush sites because there often is insufficient fine fuels to allow for prescribed fire in the eastern portions of the species' range. Invasion of annual exotic herbaceous species in western portions of the species' range provide more than adequate fine fuel-loads to carry fire. The species' response to mechanical treatments generally mirror those described for response to fire; however, mechanical treatments often result in increased residual plants following treatment, thus recruitment of sagebrush into mechanically treated sites is generally accelerated relative to burned sites.

Species Response to Herbicides: This taxa is readily killed by herbicides.

Mountain big sagebrush:

Species Response to Fire: Mountain big sagebrush is highly susceptible to injury from fire; plants are readily killed in all seasons, even by light severity fires. Regeneration of mountain big sagebrush is from on-site or off-site seed. Depending on environmental circumstances, the mountain big sagebrush seedbank tends to survive fire much better than Wyoming big sagebrush and seedlings may be abundant or sparse following fire; these relationships are not well understood. Although data from 1 study suggest that mountain big sagebrush seed germination may be stimulated by fire, most studies suggest that seed germination and survival is low following fire.

Species Response to Mechanical Treatment: Mountain big sagebrush responds to mechanical treatments similarly to Wyoming big sagebrush.

Species Response to Herbicides: This taxa is readily killed by herbicides.

Basin big sagebrush: An important note on basin big sagebrush is that, due to the species' larger vertical structure relative to other big sagebrush subspecies resulting in plant exposure above deep snows, these plants may provide critical winter forage for sage-grouse and other wildlife species in some locations under certain conditions.

Species Response to Fire: Basin big sagebrush is readily killed when aboveground plant parts are charred by fire. Basin big sagebrush does not sprout after fire and reinvades a site primarily by off-site seed or seed from plants that survive in unburned patches. Basin big sagebrush seed is not disseminated for great distances so off-site sources are probably less important than on-site seed; shrubs surviving within the perimeter of a disturbed area provide a more important seed source than those on the perimeter. The vast majority of basin big sagebrush seed produced during fall is gone by spring when few seeds persist. Emergence of basin big sagebrush seed appears to be reduced by exposure to heat

Species Response to Mechanical Treatment: No species specific information.

Species Response to Herbicides: This taxa is readily killed by herbicides.

Little or Low sagebrush:

Species Response to Fire: Low sagebrush is a non-sprouter which is readily killed by fire; the species reestablishes on burned sites through small, light, wind-dispersed seed. Because sites dominated by low sagebrush generally have minimal herbaceous understories (e.g., fine fuels), fires in these communities are comparatively rare except in the Intermountain Region where exotic annuals have invaded some sites sufficiently to provide adequate fine fuels to carry fire.

Species Response to Mechanical Treatment: No species specific information.

Species Response to Herbicides: This taxa is readily killed by herbicides.

Black sagebrush: An important note on black sagebrush is that, due to nutritional values, this species when available above the snow provides preferred winter forage for many wildlife species.

Species response to Fire: Black sagebrush is highly susceptible to fire; plants are readily killed by all fire intensities and do not sprout. Following burning, reestablishment occurs through off-site seed sources and most seeds are dispersed close to the parent plant. Effective soil moisture and patterns of burning have an influence upon the rate of site recovery. Historically fire has had little or no influence in communities dominated by black sagebrush.

Species response to Mechanical Treatment: No species specific information but likely similar to Wyoming and mountain big sagebrush.

Species response to Herbicides: This taxa is readily killed by herbicides.

Plains silver sagebrush:

Species Response to Fire: Silver sagebrush has a strong sprouting response after top-kill by fire. Because it sprouts from the root crown, rhizomes, and roots after top-kill by fire, silver sagebrush is not as susceptible to fire mortality as most woody sagebrush species. Additionally, silver sagebrush requires an open, disturbed seedbed and fire may prepare a favorable site for seedling establishment. Seeds produced from post-fire sprouts are the most likely sources of seedling establishment, but wind, water, and animal transport of seed onto burns may also contribute to post-fire seedling establishment. Studies on plains silver sagebrush indicate that as burn intensity and severity increase, plant mortality also increases and regrowth decreases.

Species Response to Mechanical Treatment: Silver sagebrush can regenerate from root sprouts and can recover quickly following mechanical disturbance.

Species response to Herbicides: This taxa is apparently readily killed by herbicides, but given its re-sprouting ability this may not be an accurate assessment.

SAGE-GROUSE RESPONSE TO TREATMENT:--Conducting shrub manipulating treatments (e.g., prescribed fire, herbicide application, mechanical treatment) across relatively large proportions of available sagebrush habitats appears to result in sage-grouse population declines. Sage-grouse habitat management guidelines (Connelly et al. 2000*b*, Bohne et al. 2007) recommend that no more than 20% of the nesting, early brood-rearing and wintering habitats (combined) in a landscape be treated at any one time, and that subsequent treatments be deferred until initially treated habitats have recovered to ≥12% canopy cover in Wyoming big sagebrush and ≥15% in mountain big sagebrush-dominated areas. Swenson et al. (1987) compared treatment studies conducted in Wyoming, Montana and Colorado and found that eliminating ≥16% of the sagebrush in the landscape closely associated with a group of leks either through plowing or herbicide spraying was correlated with a 50 to 100% reduction in sage-grouse populations; conversely, elimination of ≤11% of the sagebrush was correlated with stable to increasing populations. Data presented in Leonard et al. (2000) suggest that a threshold exists where numbers of males occupying leks are adversely affected by converting between 18 and 28% of a sagebrush-dominated landscape in Idaho. Mechanical or herbicide treatment of >23% of the total suitable sage-grouse habitat within 1.5 km of leks resulted in >50% declines in the number of breeding males on those leks relative to controls in Montana for up to 2 years post-treatment (Wallestad 1975). Connelly et al. (2000*a*) reported that the effects of removing 57% of Wyoming big sagebrush-dominated habitats within a 5,000-ha area on the sage-grouse breeding population (estimated by lek counts) in that area included increased loss of leks and increased decline in average male lek attendance for up to 5 years post-treatment when compared to a control area in Idaho. The conversion of 30% of the wintering sagebrush-dominated habitats in a 202-km^2 area in Montana by plowing resulted in a 73% decline in the number of breeding male sage-grouse on leks in the area relative to controls (Swenson et al. 1987). In Idaho, the removal of approximately 60% of the sagebrush cover in a 5000-ha area resulted in a significant decline in the use of these sites during the winter (34 and 42% of locations pre- versus 6% post-burn; Connelly et al. 1994).

As cited in Connelly et al. (2010), Shepherd (2006) reported that sagebrush patches adjacent to large, abrupt patches of habitats not dominated by sagebrush (e.g., burned areas, crested wheatgrass seedings) may be functionally smaller than their physical size due to avoidance of the periphery of these patches (both treated and untreated portions); more interspersed sagebrush patches may receive higher relative use. Shepherd (2006) recommended that 60 to 70% of the landscape be maintained as sagebrush dominated, and that these sagebrush patches be well interspersed within treated landscapes.

Slater (2003) surmised that the existence of 2 highly polar views on the use of sagebrush manipulating treatments in sage-grouse habitat, despite the failure of previous studies to strongly support either position, "suggests the 2 views may represent the disparate biases of range and sage-grouse managers" (Slater 2003:11-12). However, the Sage and Columbian Sharp-tailed Grouse Technical Committee (2009) suggested that the scientific evidence supporting the use of sagebrush treatment for sage-grouse conservation is scant while considerable information documenting negative effects of treatment on sage-grouse exists.

In general, increasing sage-grouse populations and populations below their potential carrying capacity do not appear to be adversely affected by treatment of sagebrush; however, neither do they show a positive response through an increase in relative abundance (Wallestad 1975, Martin 1990, Fischer et al. 1996). In contrast, Connelly et al. (1994) found that a declining population declined to a much greater extent in treated areas relative to untreated area. Miller and Eddleman (2001) suggested that four factors determine impacts of treatment on sage-grouse habitat: site potential, site condition, functional plant group(s), and the pattern or size of the treatment. The authors suggested that goals for managing sage-grouse habitat at multiple spatial scales are similar to goals for restoring or maintaining form, function, and process in sagebrush-dominated ecosystems. Miller and Eddleman (2001) concluded that there exists no evidence to suggest treatment will enhance sage-grouse habitat in Wyoming big sagebrush-dominated communities where there already exists a balance of native shrubs, perennial grasses and forbs; and they recommend against treatment where sagebrush cover is seasonally limiting (e.g., sage-grouse winter habitat), where the understory lacks perennial forbs and grasses, where introduced annuals are present, or where high amounts of shrub species capable of sprouting following a treatment are present (e.g., rabbitbrush, horsebrush, or snakeweed). Nelle et al. (2000) reported that prescribed fire negatively affected habitat conditions for sage-grouse nesting and brood rearing up to 15 years post-burn. Beck et al. (2009), after investigating the impact to wintering, nesting, and early brood habitat 14 years post-burn, concluded managers should not consider prescribed fire in xeric sagebrush habitats.

Research in southwestern Wyoming suggested that general grouse use of treated habitats was concentrated within 60 m of untreated edges (Slater 2003), Wilson (2000) reported sage-grouse using treated areas (burned or disked / reseeded) were within 60 m of sagebrush (remnant island or treatment edge) in Utah, and sage-grouse use of a treatment area (2,4-D spray strips) was restricted almost exclusively to remnant sagebrush patches in Montana (Martin 1970). Some research has suggested that sage-grouse females restrict their nesting use of manipulated areas to remaining patches of live sagebrush (Connelly et al. 1994, Fischer 1994); however others have found similar nesting densities between treated and untreated areas (Klebenow 1970, Slater 2003). Sage-grouse have been documented nesting under non-sagebrush shrubs and even grass in treated habitats, but these selected areas were structurally similar to untreated habitats in terms of overall shrub cover (Connelly et al. 1991, Slater 2003). Slater (2003) reported no difference in nest success probabilities within and outside burn boundaries (35 vs. 20% respectively), but overall nest success in his study (24%) was very low, suggesting potential impacts to nest success at spatial scales larger than actual treatments. Connelly et al. (1991) reported that average nest success was lower for females using non-sagebrush sites (22%) compared to those using sagebrush (53%).

Theoretically, the inability of sagebrush-removal treatments to consistently increase forbs or insects (Gates 1983, Martin 1990, Connelly et al. 1994, Fischer et al. 1996, Nelle et al. 2000, Slater 2003) will limit their utility as a tool for sage-grouse brood-rearing habitat management. However, many studies have documented that grouse broods do not either select or avoid treated habitats (Martin 1990, Connelly et al. 1994, Nelle 1998). Slater (2003) found that brooding females in treated habitats in southwestern Wyoming moved shorter distances from their nests compared to those brooding in untreated areas, and suggested that suitable brood rearing habitat may have been created by burning. In contrast, Klebenow (1970) reported that broods did not use herbicide-treated areas for 2 years post-treatment. Fischer et al. (1996) reported that the abundance and biomass of ants was reduced the 2nd and 3rd years post-treatment in southeastern Idaho; grasshopper densities were reduced by 60% the first year after a prescribed burn (Bock and Bock 1991); and 6 years after a big sagebrush wildfire, half of the ground dwelling beetle species were less abundant on burned sites (Rickard 1970). The abundance of ants and beetles on the Upper Snake River Plain was significantly greater in a 1-year old burn, but had returned to unburned levels 3 to 5 years post-burn (Nelle et al. 2000). In Oregon, Pyle and Crawford (1996) reported abundance of 2 beetle species were unaffected by fire. In a southwestern Wyoming, Slater (2003) reported no difference in insect abundance and biomass between burned and unburned sites; the author did find a lower abundance of ants and total mass of optimal-sized insects at brood sites from within burns compared to brood sites outside of burns, and lower beetle abundance on 1-year old burns and higher beetle abundance on a 12-year old burn compared to unburned sites.

The Sage and Columbian Sharp-tailed Grouse Technical Committee (2009) recommend the following in regard to restoration treatments where sagebrush stands lack an understory:
(1) Avoid use of prescribed fire in xeric sagebrush habitats;

(2) Conduct mechanical and/or chemical restoration treatments only with an understanding of their impacts on sage-grouse habitats and how these areas are affected by other factors such as habitat conversion and anthropogenic developments (that is, cumulative effects on the landscape);
(3) In areas of large-scale habitat loss, protect all remaining sagebrush habitats from further loss, fragmentation, or treatment that reduces sagebrush canopy cover;
(4) Use an adaptive approach with the intent of minimizing impacts to sage-grouse, sagebrush, and perennial native vegetation; consider impacts on all native organisms and ecosystem processes;
(5) Review past treatments in similar range sites to ascertain vegetation responses; use pilot treatments to refine techniques and study vegetation responses;
(6) Conserve and enhance remnant native vegetation and soils (Allen 1995); and
(7) Where feasible, use carefully managed grazing in place of intensive treatments that involve fire, mechanical or chemical applications.

Other recommendations to reduce the potential of negative impacts of treatments to sage-grouse found in the literature include: Most importantly, the habitat needs of sage-grouse throughout their complete life-cycle must be considered prior to treatment (Miller and Eddleman 2001). If possible, this should include the identification and protection of leks and nesting, brood-rearing, summering, and winter habitats (Benson et al. 1991, Connelly et al. 2000*b*). A minimum 3.2 km radius of sagebrush should be protected around known leks for non-migratory populations; protection buffers may have to increase for migratory populations (Braun et al. 1977, Connelly et al. 2000*b*). Sagebrush treatment techniques should not be used where nesting or winter sagebrush cover is limited or where live sagebrush cover is less than 20% (Klebenow 1972, Braun et al. 1977, Robertson 1991, Miller and Eddleman 2001). The site condition and potential for improved forb production should be evaluated before treatments for sage-grouse (Peterson 1995, Miller and Eddleman 2001), and treatments should be avoided in xeric sage-grouse habitats (Connelly et al. 1994, Fischer 1994). Snow (the predominant supplier of moisture in sagebrush ecosystems) accumulation, distribution and melt should be considered before treating sagebrush (Laycock 1979*b*). Sagebrush along streams, drainages, and meadows should be protected (Braun et al. 1977, Schneegas 1967), as these areas provide important summer habitat. The presence of undesirable invasive species (e.g., cheatgrass) should be taken into consideration prior to treatment implementation (Wright and Bailey 1982, Bunting et al. 1987, Winward 1991). The recommended size of treatment blocks ranges from <16 to <100 ha, and treatments should occur in a small-grained mosaic pattern (Wright 1974, Braun et al. 1977, Benson et al. 1991, Pyle and Crawford 1996). Post-treatment livestock grazing management and monitoring are essential for successful restoration (Braun et al. 1977, Benson et al. 1991).

SAGEBRUSH SEEDING:--Sedgwick (2004) concluded that techniques for reseeding sagebrush have been successfully demonstrated, but that surface sowing followed by compaction of the soil may be necessary for establishment. Establishment of forbs important to sage-grouse has also shown promise, but availability of seed tends to limit their widespread use on rangeland restoration and rehabilitation projects. Natural colonization of native plant communities into a site where sagebrush is desired is generally the preferred management action (seeds deposited by nearby plants are inherently adapted to a particular site; Lambert 2005, Shaw et al. 2005). However, on highly disturbed sites or where competition from weeds is excessive, seeding can be utilized to restore big sagebrush. Extensive site preparation and weed control will be necessary where dense stands of annuals are present pretreatment; a firm seedbed is required along with control of annual grasses (e.g., cheatgrass or medusahead [*Taeniatherum caput-medusae*]). Seeding or interseeding in late fall or early winter is recommended, as this is when big sagebrush naturally disperses and soil surfaces are most likely to be conducive to seedling establishment; spring seeding should be avoided. Big sagebrush can be seeded with other species to increase diversity; however, Schuman et al. (2005) contend that seeding requirements and the relative seedling growth rates of each species must be considered when developing a seed mix; because of difference in maturing time, seeded grasses establishing with big sagebrush have an initial advantage and may suppress big sagebrush seedlings; as a general rule of thumb, Shaw et al. (2005) suggest that grass be seeded at low rates if big sagebrush establishment is one of the treatment objectives. In contrast, research on reclaimed mine lands in northern Wyoming suggested that sagebrush survival 8 years post-planting was highest in areas where grasses were seeded at the higher rate; sagebrush survival was 46, 58, and 72% for the grass seeding rates of 0, 16, and 32 kg pure live seed per ha, respectively (Schuman and Belden 2002). A method to reestablish big sagebrush is to use a rangeland drill at a shallow setting. When big sagebrush is drill seeded with other seed types, it is recommended that it be seeded through a separate drill box to permit very shallow seeding and proper seed placement for plant establishment. Dr. A. Sands (Conservation Ecologist, The Nature Conservancy; personal communication) suggests that drill seeders bury sagebrush seed too deeply, and advises broadcasting seed followed by tamping (e.g., using a cultipacker) to ensure seed contact with the soil. Seedings of native plants, including big sagebrush, should be protected from grazing for at least 3 to 5 years to allow time for the shrubs and forbs to become established (Lambert 2005, Lysne 2005, Shaw et al. 2005). Redente et al. (1984) reported that the addition of shrub, grass and forb seed to sites soil bladed 35 cm deep to simulate anthropogenic disturbance associated with energy development in Colorado did not dramatically alter results; plant species composition showed that all stands were dominated by grasses after 5 years regardless of the composition of the original seed mixture.

HERBACEOUS PLANT INTERSEEDING:--Huber-Sannwald and Pyke (2005) suggested that the use of interseeding techniques may be successfully used for restoring herbaceous species in dense big sagebrush stands and should be considered for restoration where shrub retention is an important consideration. The authors found that neither shading nor root exclusion from big sagebrush plants affected artificially planted seedling survival of bluebunch wheatgrass. Wirth and Pyke (2003) successfully seeded and transplanted 2 hawksbeard spp. into sagebrush habitats; they report that seedling emergence increased with fire (vs. unburned) and in mounds associated with sagebrush (vs. interstitial spaces). Milkvetch was not effectively introduced. Walker and Shaw (2005) suggest that to increase revegetation success, seed sources may be selected from species and ecotypes indigenous to the planting area; management of local stands to improve seed production may be required to insure the availability of adequate quantities of seed. Yeo (2009) reported that seeding of bluebunch wheatgrass following Lawson aerator application resulted in substantial increases in cover within 4 growing seasons in a Wyoming big sagebrush community in east-central Idaho.

FERTILIZATION:--In Canada, Barrett (1979) reported that protein content of silver sagebrush on plots fertilized with nitrogen or nitrogen and phosphorous increased initially, but by late summer nutritional content was essentially similar in fertilized and control areas for at least the first 3 years post-application. The author noted that forage production increased in each of the 3 years; that the application of P in addition to N had little additional impact on forage quality and production; and that pronghorn selectively utilized the fertilized plots more heavily than adjacent control areas. Miller et al. (1991) found that the addition of nitrogen (either nitrate or ammonium) to a Wyoming big sagebrush site in Oregon resulted in increased thurber needlegrass aboveground biomass and tiller density, and increased aboveground biomass, total shoot density and individuals shoot weight in sagebrush. Sneva et al. (1983) reported a tendency for lower total essential oil (primarily monoterpenes) concentrations in fertilized mountain and Wyoming big sagebrush; oil concentrations in low sagebrush were unaffected. The authors mention that anecdotal observations suggested increased sagebrush growth on fertilized plots compared to controls. Pierce et al. (1998) suggested that the addition of biosolids (sanitation department waste) to a sagebrush community in Colorado increased the biomass of perennial grasses and increased plant tissue N concentrations in western wheatgrass, bluebunch wheatgrass, and Indian ricegrass; biosolids application did not alter plant canopy cover nor relative plant species abundance. In contrast, Carpenter and West (1987) reported that mountain big sagebrush in Wyoming did not respond in terms of aboveground biomass and relative growth rate with the addition of ammonium nitrate fertilizer. McLendon and Redente (1991) studied the effects of nitrogen addition on successional responses of sagebrush-dominated sites soil bladed 35 cm deep to simulate anthropogenic disturbance associated with energy development in Colorado. The authors reported that 3 seral groups developed on the non-fertilized plots, the first two dominated by annuals and lasting 3 years (Russian thistle (*Salsola kali*), lambsquarter (*Chenopodium album*), cheatgrass, tumble mustard [*Sisymbrium altissimum*]), the third transitional and dominated by perennials (yellow sweetclover (*Melilotus officinalis*), wheatgrasses, Indian ricegrass, needle-and-thread, grey rabbitbrush [Ericameria nauseosa]); the addition of N altered this successional pattern by allowing annuals to remain as site dominants through year 5.

WATER DEVELOPMENT:--Open water has been suggested as a limiting factor for summering sage-grouse. Autenrieth et al. (1982) inferred that water was important to sage-grouse, and Patterson (1952) suggested that water availability markedly affected the species' summer distribution. However, Connelly and Doughty (1989) suggested that movements to summer range were probably in response to lack of succulent forbs in an area rather then a lack of free water. Research suggests that grouse do not regularly use water developments even during relatively dry years, but obtain required moisture from consuming succulent vegetation (Connelly 1982, Connelly and Doughty 1989). Water developments tend to attract other animals and thus may serve as a predator "sink" for grouse (Connelly and Doughty 1989). Wallestad (1971) suggested that reservoirs may provide islands of succulent vegetation used by sage-grouse during the summer; Connelly and Doughty (1989) further suggest that use of water developments may be enhanced by placing them along migration routes or close to occupied summer range. An interdisciplinary group developing grazing management objectives in Wyoming's sage-grouse habitats [WYIGMG (2009)] suggested that escape ramps should be installed in all water tanks as a standard practice; the authors stress that it is imperative that ramps be installed such that they are encountered by animals swimming along the edge of a tank.

PREDATOR CONTROL:--Due to its effect on bird populations and the difficulty of controlling other factors, predation is often seen as an important source of mortality that can be reduced if necessary (Cote and Sutherland 1997). Predator control is currently conducted in many areas used by sage-grouse to reduce predation on livestock that share these ranges. Predation is generally of greatest concern to sheep and various studies have documented the significant impact of predators on these range animals (Tigner and Larson 1977, McAdoo and Klebenow 1978, Scrivner et al. 1985). In a review of 20 studies on the effectiveness of predator removal in protecting bird populations, it was found that removal can reduce early mortality, but that it may not increase the breeding bird population to any great extent (Cote and Sutherland 1997). The effectiveness of predator control appears to be influenced by the status of the target population; stable and increasing populations appear to respond positively to predator removal, while declining populations are likely to continue declining (Cote and Sutherland 1997). Connelly et al. (1994) and Braun (1998) contend that although predation could play a role in reducing sage-grouse production, the quality of breeding habitat is believed to be an overriding factor controlling the importance of predation. Connelly et al. (2004) concluded that predator control was not a viable long-term management option due to recent scientific evidence suggesting its ineffectiveness in increasing sage-grouse populations combined with recognition of the financial and political costs of removing predators.

Commonly cited mammalian sage-grouse and nest predators, namely red foxes (*Vulpes vulpes*), coyotes (*Canis latrans*), bobcats (*Felis rufus*), and badgers (*Taxidea taxus*), have a great overlap of diets (Patterson 1952, Voigt and Earle 1983, Major and Sherburne 1987, Dibello et al. 1990). As a result, resource competition likely exists and the failure to remove all predator species may simply allow the remaining species to increase in their absence. Using trapping as an index to population, Robinson (1961) found that a decrease in coyote numbers over a 20-year period corresponded to an increase of bobcat, badger, skunk (*Mephitis mephitis*), and other carnivores. Other species interactions must be considered as well. Studies of red fox / coyote interactions have shown that red foxes strongly avoid the territories of coyotes. Because coyotes generally have much larger home ranges, their presence may seriously limit the fox population of an area (Voigt and Earle 1983, Major and Sherburne 1987, Sargeant et al. 1987, Harrison et al. 1989, Mezquida et al. 2006). As coyote control became more effective during the 1930s and 1940s, the number of coyotes in farmland areas was reduced and red fox populations began to expand, with red fox becoming more numerous relative to recorded history beginning in the late 1940's (Sargeant et al. 1987). Predator removal is generally focused on the coyote because it is responsible for the vast majority of sheep predator kills (Tigner and Larson 1977, Taylor et al. 1979). However, it may not be an important sage-grouse nest predator (Patterson 1952). Diet studies of the coyote indicate that birds as a whole contribute <7% of the yearly dry weight consumed (Johnson and Hansen 1979, Reichel 1991). In contrast, the red fox is known to be a significant predator of ground nesting ducks and eggs (Sargeant 1972).

In southwest Wyoming, Slater (2003) compared predator density and species composition and sage-grouse productivity in 2 areas, 1 with extensive coyote control and 1 with limited recreational predator control. The results suggested that the coyote control program decreased coyote abundance, but that badger abundance was increased in the coyote control area (although a direct link between decreased coyote and increased badger abundance was not established).

However, nest success and brood survival did not differ between the 2 areas, suggesting reduced coyote abundance and coyote control did not benefit sage-grouse populations.

FENCES and TRANSMISSION LINES:--Fence lines represent potential movement barriers (especially woven-wire fences), predator perches or travel corridors, and are a cause of direct mortality to sage-grouse (Braun 1998). Although no quantified information has been published establishing the direct mortality impacts to sage-grouse populations, anecdotal information suggests fence-strikes may be a substantial source of mortality, especially during the breeding season. Additionally, fence-related mortalities are an important source of mortality on female lesser prairie-chickens (*Tympanuchus pallidicinctus*; Patten et al. 2005). The WYIGMG (2009) suggested that not every fence represents a strike-hazard; those that tend to cause problems are: (1) constructed with steel t-posts, (2) constructed near leks, (3) bisect winter concentration areas, or (4) border riparian areas. They suggest avoiding these areas when planning fences, and that fence markers may be employed to reduce collisions. An anecdotal report produced by the Wyoming Game and Fish Department suggests that reflective fence markers effectively reduced sage-grouse collision with a fence line in southwestern Wyoming. An interesting theory pertaining to sage-grouse nest depredation probabilities and the presence of potential predator travel corridors (e.g., fence lines, trails, etc.) was investigated in central Wyoming (Kuipers 2004). In terms of trail configuration within 100 m of sage-grouse nests, important predictors of nest success were trail absence within 25 m, and trail presence at 100 m. Kuipers (2004) theorized that if trails represented attractive travel paths for predators, trail presence close to a nest would increase nest detection probabilities, whereas trails farther away would act to draw predators away from a nest and increase hatching probabilities.

Although transmission line construction does not cause direct habitat loss, sage-grouse avoidance of vertical structure, due to altered raptor distributions and raptor species composition within relatively flat landscapes, results in habitat exclusion (≤1 km wide band centered on power lines; USDI BLM 1979, Braun 1998). The construction of transmission line structures located within 200 m of an active sage-grouse lek and between the lek and cock day use areas in northeastern Utah resulted in a 72% decline in the mean number of strutting males and an alteration in daily dispersal patterns during the breeding season within 2 years (Ellis 1985). Sage-grouse strongly avoid vertical structure presumably as a defense mechanism against raptors (Connelly et al. 2004). It is unknown if impacts to sage-grouse populations (e.g., reduced use of habitats closely associated with transmission lines) result from avoidance or increased interactions with predators (Stahlecker 1978), but the installation of raptor perch deterrents may decrease raptor and corvid use of these structures (Lammers and Collopy 2007). However, perch-deterrents may not mitigate effects if sage-grouse population declines result from avoidance of habitats in close proximity and not reduced survival due to changes in predator distributions.

5. If habitat treatments (e.g., prescribed fire, herbicide application, mechanical manipulation) are prescribed:

(A) Establish site potential to respond to treatment in a desirable manner;
(B) Develop a post-treatment management plan that will ensure desired vegetative responses can be achieved and maintained; and

(C) ***Assess the presence of undesirable plant species (e.g., cheatgrass, invasive noxious weeds, conifers) and the risk of these species increasing as a result of the proposed treatment.***

We caution again that tremendous uncertainty exists as to the vegetative and sage-grouse population outcomes of habitat treatments. Although managers should develop management or mitigation plans at landscape spatial scales, this does not suggest that treatments be implemented across these scales. Given the recommendations of the Wyoming sage-grouse habitat management guidelines, which mirror to a large degree recommendations forwarded by Connelly et al. (2000*b*), no more than 20% of the nesting, early brood-rearing and wintering habitats (combined) in a landscape should be in a treated state at any one time; subsequent treatments should be deferred until initially treated habitats have again recovered to at least 12% canopy cover in Wyoming big sagebrush and 15% in mountain big sagebrush dominated areas (Bohne et al. 2007). An interdisciplinary group developing grazing management objectives in Wyoming's sage-grouse habitats (2009) recommends a small scale case by case disturbance regime conducted over the long-term. When herbicides are used to reduce sagebrush canopy, the chemical selected should be researched carefully prior to implementation in relation to site-by-site objectives. Extreme caution and discretion should be employed when proposing a habitat treatment, especially on drier sites, sites where cheatgrass may invade, or sites with limited potential to produce sagebrush (e.g., the interface between the Wyoming Basin and the Great Plains). **This document should not be cited or used as justification for wide-scale treatments as a sage-grouse management tool.**

The following sections provide information on state-and-transition concepts, concepts that have recently become the paradigm for describing vegetation dynamics on rangelands. A developed state-and-transition model for a particular site will inherently establish site potential and the probable response of a site to vegetative management options.

The most critical element of a post-treatment management plan is livestock grazing management; the critical nature of adhering to utilization levels and timing that ensure short and long-term quality sage-grouse habitats cannot be overemphasized. The following sections provide information on the historical ramifications of overgrazing, and livestock management options that should be considered post-treatment.

Finally, the following sections provide information pertinent to undesirable plant species that may influence sage-grouse habitats. The presence of exotic invasive species within or near a proposed project area establishes a substantial risk of these species increasing as a result of habitat treatment. These situations need to be addressed on a site-by-site basis, but extreme caution needs to be employed if evidence of undesirable plant species is established during pre-treatment monitoring.

STATE-AND-TRANSITION THEORY

The following summary of state-and-transition theory is based on Pyke (2010), Briske et al. (2005, 2008), Stringham et al. (2003), and an unpublished report by an interdisciplinary group developing grazing management objectives in Wyoming's sage-grouse habitats [WYIGMG (2009)].

Ecological sites are unique sites with a given set of environmental conditions (e.g., precipitation, soil characteristics, hydrology, etc.) conducive to the production of a given vegetative community. Ecological site descriptions use state (a relatively stable set of plant communities that are resilient to disturbances) and transition (the drivers of change among alternative states) successional concepts to describe the natural range in variation of plant communities that can occur on a given ecological site (Pyke 2010). The purpose of state-and-transition models is to produce a management focused theory hypothesizing potential vegetation dynamics in a non-linear framework. This framework is based on potential alternative vegetation states on a site, potential transitions between vegetation states, and recognition of natural and management actions resulting in transitions. In sagebrush-dominated systems, the primary drivers transitioning a given state to another state are time, vegetation treatments (e.g., prescribed fire, mechanical treatments, herbicide application), and livestock grazing management (e.g., rest, timing and rates of stocking). According to the WYIGMG (2009), state-and-transition models are of keynote importance to managing rangelands for sage-grouse because some states within a given ecological site offer more sage-grouse habitat value than other potential states of that site. A developed and quantified state-and-transition model for a particular ecological site will inherently establish the potential vegetative attributes of that site, the potential habitat deficiencies relative to what a site is capable of producing, and vegetative management options resulting in transitions between states for that site.

A generalized conceptual model for sagebrush ecosystems showing plant dynamics using state-and-transition models is a follows:

A Reference State represented by 2 plant communities where the 2 communities (cool-season bunchgrass dominance and sagebrush / cool-season bunchgrass co-dominance) are identified as a single state. These 2 communities are identified as a single state because the change from the bunchgrass plant community to the sagebrush / bunchgrass plant community does not entail crossing an ecological threshold; sagebrush will advance on the bunchgrass plant community with time alone. The sagebrush / bunchgrass plant community can persist indefinitely, and is regarded as the preferred community for sage-grouse habitat because it provides an optimum mix of sagebrush and herbaceous understory. Cool-season bunchgrass species (e.g., bluebunch wheatgrass) generally provide the tallest herbaceous material a site can produce thus enhanced vertical structure, while the bunched nature of the growth form provides enhanced horizontal structure; both structural components are important as hiding cover especially for nesting sage-grouse and young chicks. Additionally, this is the state that offers the most biological diversity of shrubs, grasses, and forbs; diversity that is important for providing diverse food resources for adults (forbs) and chicks (insects and forbs). The WYIGMG (2009) stresses that the most effective management action in sage-grouse habitat may be to address management of sites in the reference state that are potentially at risk of transitioning to a rhizomatous grass understory (see next 2 paragraphs).

Not all site progression pathways lead to the Reference State; all states presented in the conceptual model are persistent plant communities that are the product of a site's history; the term Reference State is used to identify the state to which all other states are normally compared.

With excessive or ill-timed grazing, a site will transition from the Reference State into a state dominated by herbaceous species more resilient to grazing. This state is generalized as the sagebrush / rhizomatous grass state; the term rhizomatous grass is used here to denote any species that is grazing tolerant, and may include lower stature bunchgrasses (e.g., junegrasses [*Koeleria* spp.]), blue grama (*Bouteloua gracilis*), or upland sedges (*Carex spp.*). Rhizomatous grasses (e.g., thickspike and western wheatgrass) are more resistant to grazing as these species are capable of reproducing from underground roots that can sprout to form new plants; this is a grazing resistant reproduction strategy compared to the bunchgrasses that must reproduce from seed. Although grazing tolerant grasses do not provide the structural complexity of bunchgrasses, because of the large spatial extent of rangelands in the sagebrush / rhizomatous grass state (given season-long grazing, most ranges with access to water have progressed to, and are currently in, the sagebrush / rhizomatous grass state, which is an exceptionally stable state that persists under most management scenarios), this state is exceptionally important to sage-grouse; when management promotes health and vigor of the rhizomatous herbaceous community, this state produces volumes of herbaceous cover that meet the breeding season habitat requirements of sage-grouse. From a management perspective, it is critical that managers do not predicate habitat management strategies on the presumption that a backwards transition from a rhizomatous to a bunchgrass understory is readily achievable through grazing management; this transition generally requires a shrub manipulating disturbance (WYIGMG 2009). The WYIGMG (2009) further cautions, however, that because areas dominated by bunchgrasses (e.g., early stages of Reference State) provide limited sage-grouse habitat, while areas in a sagebrush / rhizomatous state provide adequate habitats if managed correctly, extreme caution needs to be employed if a treatment plan to convert rhizomatous grass-dominated sites to bunchgrass-dominated is undertaken.

Sagebrush / rhizomatous grass states can transition into sagebrush / bare ground; on these sites herbaceous vegetation occurs primarily in locations protected by shrub canopies or cactus (*Opuntia* spp.). According to the WYIGMG (2009), sites in this state are relatively rare except in areas closely associated with livestock water sources or in areas where livestock are confined. Sites in this condition provide sage-grouse with quality winter habitat only.

Pyke (2010) describes two alternative stable states sagebrush-dominated sites can transition into: exotic annual grasslands and tree-dominated areas. These alternative communities differ considerably in structure and function from the Reference State, and neither provides sage-grouse habitat. Pyke (2010) concludes that restoration of these sites is no longer possible, but rehabilitation defined as an alternative to the historic native plant community that provides similar structure and function without allowing further degradation of the site may be the only remaining alternative. If even remote possibility exists that management actions may transition a site into a state dominated by annual grasses, those actions should be reconsidered; annual grasslands provide no habitat value for sage-grouse. Anderson and Inouye (2001), after studying long-term (45 years) landscape-scale vegetation dynamics in natural sagebrush steppe in southwestern Idaho, concluded that the abundance of invasive species (primarily cheatgrass) was negatively correlated with cover of native species and that adequate cover of native species could result in a semiarid community more resistant to invasion.

LIVESTOCK GRAZING

HISTORICAL CONTEXT:--Connelly et al. (2004) summarize from the literature the long-term reaction of western rangelands to the introduction of livestock: A dramatic increase in the numbers of livestock and the area grazed from 1880 to 1905 combined with the drought that followed in the 1920s and 1930s severely altered the condition of western landscapes. Native perennial grasses and forbs that were not adapted to heavy grazing pressure were depleted from the vegetative community and replaced in much of the Great Basin and surrounding region by exotic annual grasses. Loss of protective vegetation cover in some communities resulted in extensive soil disturbance and erosion and shrub density increased (although the total distribution of shrubs across the region likely remained similar). Research estimated that the decline of palatable forage species and increases in plant species of low palatability took only 10 to 15 years at any given site under heavy uncontrolled grazing. Forage production for livestock dropped to an estimated 10% of the site potential following depletion of the vegetation community in some regions. The area estimated to support a cow and calf (estimated as an animal unit month (AUM) which is the amount of forage required to feed one 1,000 lb cow and her calf, one horse, five sheep, or five goats for one month) was estimated to 0.83 AUM/ha prior to European settlement, 0.27 AUM/ha in the 1930's, and 0.31 AUM/ha in the 1970s.

The effect of livestock grazing is one of the most contentious issues underlying the management and use of sagebrush habitats (Wambolt et al. 2002, Crawford et al. 2004). However, livestock grazing is the most widespread land uses across the sagebrush biome. Isolated areas exist that have not been grazed by domestic livestock, however most sagebrush habitats have been grazed in the past century (Saab et al. 1995, West 1996, West and Young 2000, Hockett 2002). Connelly et al. (2004) describe livestock grazing as a diffuse form of biotic disturbance that exerts repeated pressure over many years on a system; unlike point-sources of disturbance (e.g., fire that have acute perturbations from a well-defined origin) the impact of livestock grazing is spread unevenly across the landscape in space and time. Thus, effects of grazing are not likely to be detected as disruptions (except in extreme cases) but rather as differences in the processes and functioning of the sagebrush system.

A commonly asked question relates to the declining numbers of both livestock and sage-grouse: how can sage-grouse habitat loss be attributed to livestock when sage-grouse were more numerous when livestock numbers were also more numerous? The WYIGMG (2009) suggests that while grazing is only a part of the habitat fragmentation issues adversely affecting sage-grouse, at least part of the answer may lie in four core premises. (1) The sagebrush / bunchgrass community (Reference State in sagebrush systems) offers the most sage-grouse habitat value; (2) the sagebrush / bunchgrass community readily transitions to the sagebrush / rhizomatous grass state; (3) many ranges have been converted from sheep to cattle, and cattle are more likely to trigger this transition; and (4) sites that transition from the Reference State to sagebrush / rhizomatous grass states persist even after grazing management is improved. In combination, this suggests that even though livestock numbers are lower since the implementation of the Taylor Grazing Act that was passed in 1934 to prevent overgrazing and damage to public lands, and grazing management across the West has steadily improved, the acreage transitioning from the Reference State to the sagebrush / rhizomatous grass state is still accumulating. Connelly et al. (2004) contend that the productivity of western shrublands has declined due to grazing history and drought. The authors suggest they cannot conclude that the effect of historical grazing has been reduced because even reduced numbers of livestock from the Taylor Grazing Act may still influence those habitats; the absence of information on management coupled with vegetation changes limits the understanding of the effect of livestock grazing on long-term dynamics of sagebrush systems.

Livestock grazing can affect soils, vegetation, and animal communities by consuming or altering vegetation, redistributing nutrients and plant seeds, trampling soils and vegetation, and disrupting microbiotic crusts (Miller et al. 1994, West 1996, Jones 2000, Belnap and Lange 2001). At unsustainable levels of grazing, these impacts can lead to loss of vegetative cover, reduced water infiltration rates, and increased soil erosion past thresholds to which the system can return (Society for Range Management 1995). The current evolution in assessments of habitats and the effect of grazing is based on indicators of soil characteristics and erosion, plant communities, and underlying processes to evaluate the "health" of the ecosystem (National Research Council 1994). The WYIGMG (2009) suggests that the major influence of grazing on sage-grouse habitat is the potential to cause a transition from an ecological state dominated by sagebrush and cool season bunchgrasses to a site dominated by sagebrush and rhizomatous grasses; additionally, unsustainable livestock use may result in conversion of sagebrush steppe to annual grasslands. Anderson and Inouye (2001) studied the long-term changes of sagebrush habitats in southwestern Idaho to the removal of livestock and in the absence of anthropogenic disturbances and reported that the richness of shrubs, perennial grasses and forbs, and vegetative heterogeneity steadily increased through 45 years post-removal of livestock; areas with higher species richness maintained consistently greater cover compared to mean levels, suggesting links between species richness and function.

LIVESTOCK MANAGEMENT:--The WYIGMG (2009) explains that the grazing influence on sage-grouse habitat is a function of both long-term management to promote desirable plant communities and annual management of the standing crop to provide cover for sage-grouse habitat. Managing for the sagebrush / bunchgrass plant community addresses many but not all grazing issues. The potential exists to manage a site for its long-term forage plant health but fail to achieve sage-grouse habitat objectives. Sage-grouse initiate nesting in April, prior to production of the current year's standing crop of herbaceous forage, thus residual grasses left from the previous year represent the initial cover available for nesting sage-grouse. With few exceptions leaving adequate residual forage will provide for both long and short-term objectives, and adherence to light utilization standards is the single best tool to ensure a healthy community.

Grazing management is particularly important the first 2 growing seasons following disturbance. Deferment in this period allows the cool season bunchgrasses to capitalize on the open niches created by treatment. Cool season bunchgrass plants are vulnerable following treatment, and must be protected by grazing management (WYIGMG 2009). Conversely, Bates et al. (2009) contend that timing, use, and duration of grazing of treated rangelands are more important than a specific period of rest after fire. The authors suggest that moderate grazing use after perennial grass dormancy (e.g., late season) within the first two summers after fire should not reduce the recovery ability of herbaceous communities in sagebrush steppe. However, the study was short-term (3 years post-treatment); West and Yorks (2002) reported no differences in herbaceous cover among burn–ungrazed and burn–grazed areas the first 6 yr after fire, but between years 7 and 18 after fire, perennial grass cover in grazed areas decreased compared with ungrazed areas (timing of use was not provided). Treated areas may draw grazing pressure from all herbivores, thus treatment design should consider the possibility of an unplanned escalation of use by wild horses or elk.

The timing and intensity of grazing are the two key factors that will affect plant health. Timing refers to when the plant is grazed. Annual growth of herbaceous vegetation in Wyoming big sagebrush habitat is generally concentrated in a four to six week period in the spring. Repeated grazing in this critical period causes plants to reinitiate growth from root reserves multiple times, without sufficient energy capture from photosynthesis to replenish root reserves. Cool season bunchgrasses require periodic opportunity to photosynthesize without interruption from grazing, thus grazing during the grass growing period has the greatest negative effect on plant survivorship. The key consideration of grazing management in sagebrush habitat is to assure that the cool season bunchgrass growth cycle is not interrupted repeatedly by defoliation. If plants are not allowed to replenish root reserves during a given year, grazing strategies generally must allow for uninterrupted growth in subsequent years. Cool season bunchgrasses that are not provided the opportunity to recover from grazing will become smaller, and eventually yield their space to more grazing resistant species. Intensity refers to the level of utilization the plant receives. Use levels are important because grazing systems seldom compensate for heavy utilization; no more than moderate utilization is recommended. Moderate utilization generally results in <35% use on total herbaceous vegetation and <60% use of key species. In contrast, Dr. A. Sands (Conservation Ecologist, The Nature Conservancy; personal communication) suggests that 35% use on key native cool season bunchgrasses in the growing season should be the maximum use level if maintaining or improving a native cool season bunchgrass community is desirable. Moderate use levels provide a patchy appearance to the observer where utilization is apparent but ungrazed seedstalks and herbaceous production is readily apparent across the landscape (WYIGMG 2009).

Connelly et al. (2004) suggests that changes to season grazed or reductions in numbers of livestock will show improvements in sage-grouse habitat quality only if the community retains both the sagebrush and the tall bunchgrass necessary for quality habitat before grazing changes are implemented. Improvements in habitat quality may be expressed in the next growing season, but might take 3 to 5 years for pre-existing plants to fully express themselves and 10 to 15 years for seed production and new plant recruitment to occur. If the desired vegetative components are not present in a site where grazing management alterations are proposed, additional manipulations may be required including the additions of life forms through revegetation.

There is little scientific data linking grazing practices to sage-grouse population levels (Connelly and Braun 1997). However, comparing sage-grouse seasonal habitat requirements to studies investigating the response of the habitat to livestock grazing can provide suggestions. Short-term rotational grazing patterns (vs. continuous grazing patterns) benefits native grass and forb production (Derner et al. 1994), which are key habitat features associated with hatching success and hen pre-laying nutrition. However, heavy spring and spring-fall grazing is detrimental to upland herbaceous understories essential for sage-grouse nesting success, while fall utilization is neither detrimental nor advantageous (Mueggler 1950, Laycock 1979*a*, Owens and Norton 1990). Insect diversity and density is positively correlated with herbaceous density and diversity (Hull et al. 1996, Jamison et al. 2002), thus spring or spring-fall grazing could also negatively impact young chick survival. Stocking rate appears to be the variable impacting residual grass stubble height (important during the initial stages of nest incubation), with high stocking rates reducing heights (Owens and Norton 1990, Derner et al. 1994). Conversely, spring grazing at high stocking rates is potentially beneficial on sage-grouse winter range, while heavy fall utilization is detrimental (because of differing impacts to sagebrush densities; Wright 1970, Owens and Norton 1990, Angell 1997). The importance of annual and seasonal range monitoring and subsequent removal of livestock as utilization reaches capacity cannot be over-emphasized (Holechek 1996, Thurow and Taylor 1999).

Laycock (1979*a*) reported that changing from fall to spring grazing on sagebrush-grass rangelands between 1950-1964 resulted in: (1) further declines of pastures in poor condition with continued spring grazing; (2) range in good condition declined to poor; (3) a 78% increase in sagebrush canopies and an increase in cheatgrass; and (4) a decline in total grass production of 22% and total forb production of 73%. Additionally, the grazing capacity of sagebrush-grass rangeland in central Idaho used annually in both spring and fall was less than one-third that of the fall-only grazed pastures (range grazed in both the spring and fall contained 173% as much brush, 72% as much grass cover, and 20% as much forb cover as fall only grazed areas; Mueggler 1950). Mueggler (1950) reported that during the following 18 years of comparatively light spring stocking, however, grasses substantially increased and forbs improved slightly. But, if forbs have been reduced or removed by heavy spring grazing, or annuals have replaced native grasses and forbs, range recovery is slow or non-existent (Laycock 1979*a*). Interestingly in extreme cases of widespread range degradation, reduced vegetative cover can change surface reflectivity, which will theoretically inhibit cloud formation and reduce precipitation (Charney et al. 1975). There is some evidence to indicate that fall grazing may cause significant reductions in sagebrush, realized through cumulative effects of even light utilization over many years (Wright 1970). A program of spring deferment combined with heavy fall grazing also offers an alternative pro-active management method of range improvement (vs. burning, herbicide spraying, or mechanical treatments; Laycock 1979*a*).

In southeast Wyoming, cattle selected forbs and native cool-season grasses, (western wheatgrass, needle-and-thread, and sedges) and selected against warm-season grasses (which included about 80% blue grama; Samuel and Howard 1982). Vavra et al. (1977) reported that warm-season grasses became increasingly important as cattle use increased; blue grama consumption increased on heavily stocked pastures vs. lightly stocked pastures. Consequently, the amount of time spent on individual pastures annually was crucial for range sustainability (T. Malmberg, Rancher's Management Company, Inc., personal communication). Due to rapid declines in diet quality and ingestion rate, Olson et al. (1989) recommended that paddocks be grazed for 2 or fewer days during the active growth period. Additionally, a year of rest (no livestock grazing) every third year allowed for the recovery of forbs with enhanced forb production the year following rest (Neel 1980).

The most direct effect of livestock on riparian vegetation is removal of the lower vegetation layers; livestock exclusion from riparian habitats results in increased sedge cover, forb cover, foliage height diversity, water-table depth and an expansion of riparian vegetation laterally from stream channels (Dobkin et al. 1998). Livestock distribution patterns (which are directly linked with water availability) and impacts to riparian habitats primarily influence sage-grouse late brood-rearing and summering habitats. The transition zones or ecotones between types (upland sagebrush and wet meadow) provide food forbs with associated protective cover and are important areas for sage-grouse broods (Klebenow 1982). However, meadows that are heavily invaded by sagebrush and heavy vegetation on ungrazed meadows are not utilized by sage-grouse (Oakleaf 1971, Klebenow 1982). High stocking rates in areas with limited water resource availability is detrimental to forage productivity surrounding water sources (Hall and Bryant 1995, Dobkin et al. 1998). Summer grazing on riparian habitats also appears to concentrate livestock on riparian corridors, resulting in decreased low vegetative growth (typically the forb communities essential in sage-grouse summer diets) and the extent of the hyporheic zone (reducing the lateral extent of succulent vegetation associated with the riparian corridor). However, sage-grouse use grazed over ungrazed meadows where protective cover conditions are otherwise equal (Neel 1980). Grazing increases the quality of the forb resource (by interrupting and delaying maturation) and increases accessibility to low-growing food forbs (by producing patchy small openings) sought by sage-grouse (Neel 1980, Evans 1986). Bryant (1982) suggests that stocking pastures containing riparian zones with cow/calf pairs (vs. yearlings) during the cooler part of the grazing season will decrease adverse livestock impacts to the riparian habitats. Additionally, Neel (1980) maintains that rest-rotation grazing can beneficially impact sage-grouse summering habitat if moderate stocking levels are maintained, and rest is afforded a given meadow every 3 years. Bryant (1982) emphasizes that when fencing riparian areas to exclude livestock, insure that all riparian plant community types are included; and to eliminate livestock concentrations in the riparian zones, place the fence on the first flat area above the stream. However, if herding, turning water on or off, or some other method can control animal distribution and movement, fences are not necessary (Hart et al. 1989). Braun (1998) indicates that: (1) fences with 1-3 strands of wire are normally not negative to sage-grouse (although grouse have been documented flying into fences), (2) woven wire fences are more negative to sage-grouse than single strand fences, (3) fences with maintained trails adjacent to them provide corridors for predators, and (4) fences with wood posts provide perch sites for potential avian predators.

A study conducted in central Wyoming that was focused on the potential effects of livestock grazing management practices to sage-grouse productivity (Kuipers 2004) suggested that reduced forage utilization, extended periods of rest, and reduced spring grazing could provide conditions suitable for sage-grouse nesting and early brooding during periods of extensive drought (precipitation 68% of normal during study). Grazing system (based on rotation period) appeared to be less important relative to stocking rates and season of use. Comparing the grazing systems: herbaceous cover and height estimates were consistently lower in livestock grazed relative to non-grazed pastures; residual and live grass height and cover and forb cover were lower in deferred (essentially season long grazing) compared to rotation systems; and grass and forb cover were lower in spring – fall grazed compared to summer grazed rotation systems. Interestingly, bare ground doubled during the time of the study in pastures grazed season long. Shrub components did not appear to be influenced by grazing system. Kuipers (2004) concluded that pastures grazed during the summer and the non-grazed control pastures best mimicked suitable sage-grouse nesting and early brood-rearing habitat during an extensive drought.

INVASIVE PLANT SPECIES

Miller et al. (2010) contend that cheatgrass and medusahead have become the most problematic of the exotic annual grasses in western North America. The authors continue by indicating that cheatgrass invasion, in particular, can result in a near monoculture in the more arid lower elevation Wyoming big sagebrush communities; these monocultures can be considered as new steady states over much of eastern Washington, eastern Oregon, southern Idaho, Nevada, and Utah. In addition to its displacement of native understory species, because cheatgrass germinates and thus dries early in the growing season, infestation leads to an increased risk of wildfires that eliminate the sagebrush overstory (Klemmedson and Smith 1964). By changing fire-frequency, cheatgrass infestations cause the direct elimination of native shrubs, forbs, and perennial grasses and result in a self-perpetuating stand of cheatgrass. Soil erosion also can be accelerated in systems dominated by cheatgrass because of bare ground left after early season fires and the poor ground cover the species provides during drought conditions. Although cheatgrass tends to be most competitive with native vegetation following disturbance, Connelly et al. (2004) note that cheatgrass is now replacing sagebrush slowly over time without disturbance (e.g., fire), especially on drier Wyoming big sagebrush sites, and also on some salt desert shrub sites (first documented on Anaho Island National Wildlife Refuge, Washoe County, Nevada).

Cheatgrass has been a major factor in the loss of Wyoming big sagebrush communities (Chambers et al. 2007), and is consistently cited as a major challenge to the maintenance of sagebrush steppe habitats (Young and Allen 1997, Knick 1999). Medusahead is filling a similar niche in more mesic communities with heavier clay soils (Dahl and Tisdale 1975).

Sedgwick (2004) concludes from a review of the literature that rehabilitation and restoration techniques to transform lands currently dominated by invasive annual grasses into quality sage-grouse habitat have been largely unproven and experimental. Several components of the process are being investigated with varying degrees of success. The first aspect of the process will be the reduction in the competition that invasive annual grasses provide against native seedlings during the establishment phase. Proposed techniques to reduce cheatgrass densities include herbicides imazapic (Plateau) and glyphosate, defoliation via livestock grazing, and pathogenic bacteria and fungi. Although prescribed fire alone is not recommended, it may be an effective technique worth investigation if applied in combination with a spring glyphosate treatment and conducted either in late spring or autumn. The glyphosate will kill the current-year's plants, thus reducing or eliminating seed production, and will prepare a fuel bed for the fire that will reduce the litter seed bank. In addition to density reduction techniques, applications of carbon in a form readily available for microbial uptake in the soil may increase soil microbial content and cause the microbes to reduce the available soil nitrogen, thus reducing the growth and competitive ability of cheatgrass. Immediate revegetation is required after reduction of invasives; otherwise invasive annual grasses that escape treatments will grow unabated, produce large numbers of seeds, and quickly dominate a site again. Successful revegetation efforts are generally those where introduced forage grasses have been sown. Some evidence from wildfire rehabilitation studies shows that native plants can be sown and eventually coexists with invasive annuals, but these were generally sown in combination with introduced grasses. Theoretical frameworks hypothesize that multiple native species representing a variety of growth and life forms may successfully compete with invasive plants where any one species would be unsuccessful.

In contrast to Sedwick's (2004) overall conclusion that restoration of sites invaded by annual exotic vegetation is unproven, Dr. A. Sands (Conservation Ecologist, The Nature Conservancy; personal communication) argues that this is a situation where practice is ahead of research. The

BLM has been doing operational level rehabilitation seedings using big sagebrush and native perennial grasses with some initially substantial success, especially in southcentral Idaho. However, the use of non-native perennial grasses may be required in the more xeric Wyoming big sagebrush sites. Dr. Sands has had success preventing cheatgrass stand establishment and reducing cheatgrass seedbanks by burning cheatgrass sites to clear the litter which facilitates application of glyphosate to germinating and young cheatgrass plants from the seedbed.

Utah juniper (*Juniperus osteosperma*), western juniper (*J. occidentalis*), single-leaf pinyon (*Pinus monophylla*) and two needle pinyon (*P. edulis*) are the primary conifer species encroaching into large portions of sagebrush-dominated communities at higher elevations; Rocky Mountain juniper (*J. scopulorum*) also is expanding into sagebrush communities only to a lesser extent (Miller et al. 2010). Increases in the distribution and density of conifer woodlands have resulted from the combination of inappropriate livestock grazing, alteration of fire regimes, and climate change (Connelly et al. 2004). Conifer species can eliminate the understory component of a sagebrush community after encroachment (Johnsen 1962, Tausch and Tueller 1990, Miller et al. 2000). Additionally, sage-grouse tend to avoid conifer-dominated habitats (Doherty 2008), thus conifer expansion results in the function declines of available habitat.

INVASIVE SPECIES SPECIFICS:--On semiarid rangelands, cheatgrass reaches its greatest development on degraded big sagebrush / bunchgrass ranges in the intermountain area between the Sierra-Cascade and the Rocky Mountains (Young et al. 1987). Young et al. (1987) report that excessive disturbance across sagebrush-dominated regions due to historical grazing management and the dust bowl (beginning around 1870) resulted in the colonization of Russian thistle which was replaced by mustards, and eventually the successional continuum climaxed in cheatgrass. In more arid environments, germination occurs in the early spring (4 out of 5 years) or the fall (1 out of 5 years), and cheatgrass completes its life cycle before soil moisture is exhausted (Young et al. 1987). Cheatgrass continually adapts to a variety of different range types, but has a low tolerance for soluble salts (Emmerich et al. 1993).

Medusahead is a winter annual (germinates in the fall); due to its relatively early germination, its roots develop early and reach deep in the soil so that it is capable of outcompeting native plants for moisture. As the grass grows it accumulates silica, making it unpalatable to livestock except for early in its life cycle. It creates a dense layer of litter, and because of the silica content, the litter decomposes slower than that of other plants; this litter suppresses native plant growth while encouraging the germination of its own seed, and after a few years it creates an enormous load of fuel that can lead to wildfires. No single control method will eradicate medusahead, and it is often necessary use a form of integrated control that combines two or more methods (/en.wikipedia.org/).

Russian knapweed (*Centaurea repens*), spotted knapweed (*C. maculosa*), and diffuse knapweed (*C. diffusa*) are highly competitive, noxious natives of Eurasia, probably introduced in North America as a contaminant of alfalfa and clover seed in the late 1800's (Whitson et al. 1991). Kurtz (1995) established that the invasion of Russian knapweed into areas of Colorado and Wyoming resulted in the decline of native plant species and small mammal species abundance. Additionally, spotted knapweed ranks as the number one weed problem on rangelands in eastern Montana (Whitson et al. 1991). Knapweeds readily establish as colonies on any disturbed soil (including cultivated fields, orchards, pastures and roadsides), and their early spring growth makes them competitive for soil moisture and nutrients (Whitson et al. 1991). There is also some evidence that knapweeds release chemical substances that inhibit surrounding vegetation (Whitson et al. 1991).

Dalmation Toadflax (*Linaria dalmatica*) is a noxious weed introduced from southeastern Europe (probably as an ornamental) that invades rangelands, overgrazed pastures, roadsides, and disturbed areas (Whitson et al. 1991, Butler and Burrill 1994, Rose 1999). Dalmation toadflax is most common east of the Cascade Mountains and is found locally throughout most of the range country in the western United States (Butler and Burrill 1994). Extensive root systems along with waxy leaves make this an extremely difficult plant to control (Whitson et al. 1991, Rose 1999).

Musk thistle (*Carduus natans*) was introduced to the U.S. in the earlier part of the century from Europe and western Asia, and is now widespread throughout the U.S. and Canada (Whitson et al. 1991). It aggressively invades rangelands, forestlands, roadsides, waste areas, ditch banks and grain fields and spreads rapidly forming extremely dense stands which crowd out desirable forages (Whitson et al. 1991). However, the use of perennial cool season grasses, initially established by chemical control followed by seeding, was shown to provide sustainable control (given proper grazing management which allows the grasses to remain competitive) for Dalmatian toadflax and musk thistle (Rose 1999).

Canada thistle (*Cirsium arvense*) is an aggressive, creeping perennial weed that infests crops, pastures, rangeland, roadsides and noncrop areas. Canada thistle was introduced from Eurasia in the 1600s probably as a seed contaminant. Generally, infestations start on disturbed ground, including ditch banks, overgrazed pastures, tilled fields or abandoned sites. Canada thistle reduces forage consumption in pastures and rangeland because cattle typically will not graze near infestations. The key principle to Canada thistle control is to stress the plant and force it to use stored root nutrients (Beck; www.ext.colostate.edu/).

Whitetop (*Cardaria* spp.) is a member of the mustard family, and is an aggressive plant that is competitive with and eventually will eliminate native plants. Whitetop favors alkaline soil, disturbed soils (soil that has been turned and/or cultivated), and overgrazed areas. Whitetop has spread to cover the entire nation, except for extreme southern U.S.; it is thought to have come over from Eurasia as seed in the soil used as the ballast of sailing ships, and some think that it came as an impurity in mattress material, carried over in 1809. Livestock will eat whitetop if forced by limited food availability (/mtwow.org/).

Halogeton (*Halogeton glomeratus*) is a poisonous weed that occurred on >1 million acres of land in Wyoming in 1955, with the largest infestation of this annual located in the Big Horn Basin (Rauchfuss 1955). Halogeton is a native of Asia that is adapted to the alkaline soils and semi-arid environment of high-desert winter livestock ranges, and is usually most concentrated along roadsides, sheep trails, and near areas where livestock congregate (Whitson et al. 1991). The plant produces toxic oxalates that are especially poisonous to sheep, though cattle may also be affected (Whitson et al. 1991). However, Halogeton cannot compete successfully with stands of vigorous native perennial vegetation (Rauchfuss 1955, Whitson et al. 1991).

Leafy Spurge (*Euphorbia esula*) is native to Eurasia, was brought into the United States as a seed impurity about 1830, and now infests almost 2.5 million acres, mostly in southern Canada and north-central United States (Whitson et al. 1991). It has been reported to cause severe irritation of the mouth and digestive tract of cattle that may result in death (Whitson et al. 1991). An extensive root system containing large nutrient reserves makes leafy spurge extremely difficult to control (Whitson et al. 1991).

Russian thistle entered the United States via contaminated flax seed in 1873, and had infested nearly a million acres of land in southern Idaho by the 1930's (Boerboom 1993). It is well adapted to cultivated dryland agriculture, and is also found on disturbed wastelands, overgrazed rangeland, and some irrigated cropland (Whitson et al. 1991). The weed's drought tolerance and long-distance seed dispersal has made Russian thistle one of the most common weeds in the semi-arid range of the western United States (Whitson et al. 1991, Boerboom 1993).

6. Ensure short-term and long-term post-prescription monitoring of project.

Bohne et al. (2007) succinctly state the importance of this step (paraphrased): "monitoring the post-prescription response of vegetation and sage-grouse populations to habitat manipulations is critical. Monitoring plans require a commitment of manpower and resources; if these resources cannot be committed prior to initiation of management change, then the value of the proposed project should be questioned and the project should be reconsidered or terminated." Post-prescription monitoring is additionally required to ensure adaptive management can be effectively implemented (for example: (1) apply a prescription; (2) monitor the outcome over a period of time; (3) based on success or failure adjust treatments accordingly).

Post-prescription monitoring methodology should follow that established pre-prescription (see steps 1 and 2) to ensure comparable data. A monitoring plan should ensure that reference (e.g., control) sites are monitored in conjunction with sites where management prescriptions are implemented to ensure monitoring data can be analyzed in a before-after control-impact (BACI) design. Pre-prescription data should be collected a minimum of the 2 years proceeding implementation; post-prescription monitoring should be collected a minimum of 5 years post-implementation to account for lags in the response of sage-grouse populations to habitat change (Walker et al. 2007, Holloran et al. 2010). If sagebrush-manipulating habitat treatments (e.g., prescribed fire, herbicide application, mechanical treatment) are implemented, post-prescription vegetation and sage-grouse population monitoring may have to continue on a regular basis (e.g., every 3 to 5 years) for at least 35 years post-implementation.

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