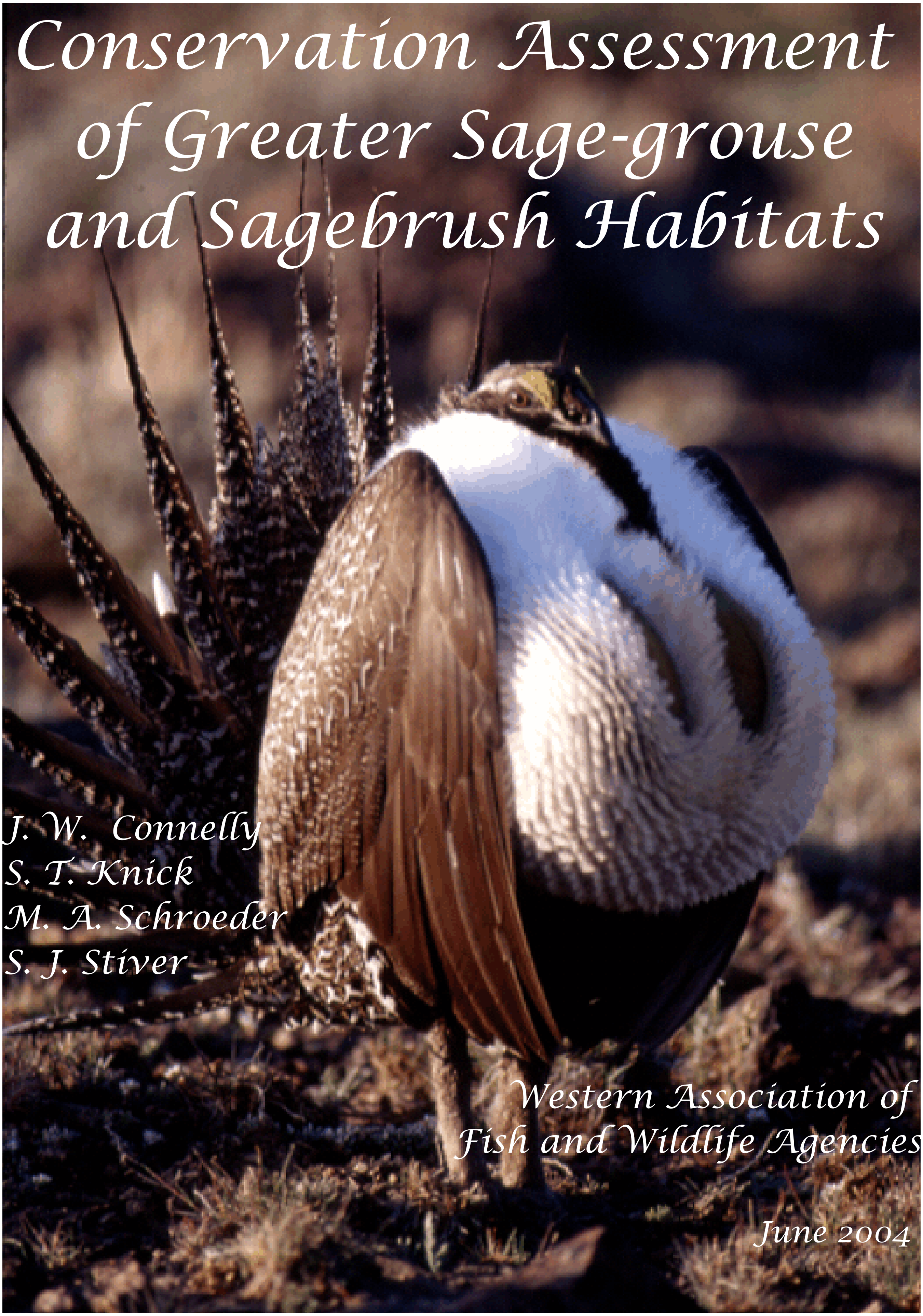
Conservation Assessment of Greater Sage-grouse and Sagebrush Habitats



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Western Association of Fish and Wildlife Agencies

June 2004

CONSERVATION ASSESSMENT OF GREATER SAGE-GROUSE and SAGEBRUSH HABITATS

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Conservation Assessment of Greater Sage-grouse and Sagebrush Habitats

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Acknowledgments

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Contributors to the various chapters of this report are listed in a separate section. However, many other individuals unselfishly shared their knowledge and technical expertise. Sean Finn, Steve Hanser, Cara Meinke, and Linda Schueck conducted the GIS analyses and created many of the graphics used in the Conservation Assessment. Their work to assemble, format, reconcile, and analyze the data is the foundation for this report. Their persistence over the marathon spanning more than a year is remarkable. A finer group of talented and enthusiastic individuals exists nowhere.

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June 2004

Executive Summary

EXECUTIVE SUMMARY

Greater sage-grouse (*Centrocercus urophasianus*) once occupied parts of 12 states within the western United States and 3 Canadian provinces. Populations of greater sage-grouse have undergone long-term population declines. The sagebrush (*Artemisia* spp.) habitats on which sage-grouse depend have experienced extensive alteration and loss. Consequently, concerns raised for the conservation and management of greater sage-grouse and their habitats have resulted in petitions to list greater sage-grouse under the Endangered Species Act. In this report, we assessed the ecological status and potential factors that influenced greater sage-grouse and sagebrush habitats across their entire distribution. We used a large-scale approach to identify regional patterns of habitat, disturbance, land use practices, and population trends. We included literature spanning the last 200 years, landscape information dating back 100 years, and population data collected over the last 60 years.

We described the primary issues that influenced greater sage-grouse and sagebrush habitats for an area that exceeded >2,000,000 km^2 (>770,000 mi^2) in size. To do this, we compiled, integrated, and analyzed data obtained from agencies and organizations within 14 states, >13 federal agencies, and 2 nations. We did not make recommendations or suggest management strategies. Rather, our goal was to present an unbiased and scientific documentation of dominant issues and their effects on greater sage-grouse populations and sagebrush habitats.

We organized the Conservation Assessment into 4 main sections. In the first section, (Chapters 1 and 2), we present background information on greater sage-grouse and sagebrush habitats. We first introduce the factors that have contributed to widespread concern about conservation and management of greater sage-grouse and sagebrush habitats. We also describe the historical and legal administration as well as the current stewardship of sagebrush habitats. We then provide information on the conservation status of the species across its range-wide distribution. The second section (Chapters 3-5) provides information on the basic ecology of greater sage-grouse and sagebrush habitats. Our objectives were to develop the underlying foundation on which to assess information presented in the remainder of the document. In the third section (Chapters 6-12), we describe the current situation and trends in greater sage-grouse populations and the dominant factors that individually and cumulatively influence sagebrush habitats. In the fourth section (Chapter 13), we integrate the habitat and population trend information into a synthesis of the conservation status for greater sage-grouse and sagebrush ecosystems in western North America.

Sagebrush Habitats

Sagebrush ecosystems dominate approximately 480,000 km^2 throughout western North America. Almost all (70%) of the existing sagebrush habitats are publicly owned and managed by a state or federal agency. The U.S. Bureau of Land Management is the primary agency responsible for management of public lands containing sagebrush and has stewardship for 50% of the sagebrush habitats in the United States. Multiple use is the dominant management objective on almost all sagebrush habitats.

Using a landscape perspective, we described the current status of sagebrush ecosystems (Chapter 5), trends within these systems (Chapter 7), and assessed impacts of anthropogenic change with respect to sage-grouse (Chapter 12). In most cases, we quantified the changes, the regional distribution of a factor, or the area influenced by the disturbance.

The sagebrush biome has changed since settlement by Europeans. The current distribution, composition and dynamics, and disturbance regimes of sagebrush ecosystems have been altered by interactions among disturbance, land use, and invasion of exotic plants. The primary areas in which sagebrush habitats currently cover a large regional portion of the landscape were in central Washington; southeastern Oregon, northern Nevada, and southwestern Idaho; and central Wyoming. Landscapes were highly fragmented surrounding these regions.

The number of fires and total area burned have increased across much of the sagebrush biome over the past 20 years (for which records are more reliable). Cheatgrass (*Bromus tectorum*) and other exotic plant species have invaded lower elevation sagebrush habitats across much of the western part of the biome, further exacerbating the role of fire in these systems. At higher elevations, juniper (*Juniperus* spp.) and pinyon (*Pinus* spp.) woodland invasions into sagebrush habitats also have altered disturbance regimes.

Land conversions were significant factors in separating habitat patches and fragmenting landscapes. Sage-grouse populations and sagebrush habitats that once were continuous now are separated by agriculture, urbanization, and development in the Snake River corridor in southern Idaho. Highly productive regions throughout the sagebrush biome that had deeper soils and higher precipitation have been converted to agriculture in contrast to the low elevation, more xeric climates that characterized the larger landscapes still dominated by sagebrush. Agriculture currently influences 56% of the Conservation Assessment Area and 49% of the sagebrush habitats by fragmenting the landscape or facilitating movements of potential predators, such as common ravens (*Corvus corax*) on greater sage-grouse.

Urbanization and increasing human populations throughout much of the sagebrush biome have resulted in an extensive network of roads, powerlines, railroads, and communications towers and an expanding influence on sagebrush habitats. Roads and other corridors promote the invasion of exotic plants, provide travel routes for predators, and facilitate human access into sagebrush habitats. Human-caused fires were closely related to existing roads. Less than 5% of the existing sagebrush habitats were >2.5 km from a mapped road.

We evaluated the influence of livestock grazing primarily by the effect on habitats resulting from management practices and habitat treatments. Numbers used by agencies (e.g., permitted Animal Unit Months) do not provide the information on management regime, habitat condition, or kind of livestock that can be used to assess the direct effects of livestock grazing at large regional scales. Indices of seral stage used to relate current conditions to potential climax vegetation may not correlate with current understanding of the state-and-transition dynamics of sagebrush habitats. Over half of the public lands have not been surveyed relative to standards and guidelines established for those lands. Although large treatments designed to remove sagebrush and increase forage palatable to livestock no longer are conducted, habitat manipulations, water developments, and fencing still are done to manage livestock grazing. Widespread water developments throughout sagebrush habitats increased the amount of area that can be grazed. More than 1,000 km of fences have been constructed each year on public lands from 1996 to 2002; linear density of fences exceeded 2 km/km^2 in some regions of the sagebrush biome. Fences provide perches for raptors, and modify access and movements by humans and livestock, thus exerting a new mosaic of disturbance and use on the landscape.

Energy development for oil and gas influences sagebrush habitats by physical removal of habitat to construct well pads, roads, and pipelines. Indirect effects include habitat fragmentation and soil disturbance along roads, spread of exotic plants, and increased predation from raptors that have access to new perches for nesting and hunting. Noise disturbance from construction activities and vehicles also can disrupt sage-grouse breeding and nesting. Development of oil and gas resources will continue to be a significant influence on sagebrush habitats and sage-grouse because of advanced technological capability to access and develop reserves, high demand for oil and gas resources, and the large number of applications submitted (4,279 in fiscal year 2002) and approved each year.

Some land use factors that we considered, such as military training, may have very intense effects on habitats but are restricted to relatively small regions across the entire sagebrush biome. In contrast, livestock grazing influences sagebrush ecosystems across the entire biome. The cumulative impacts of the disturbances and the interactions among disturbance regime, invasive species, and land use have the most significant influence on the trajectory of sagebrush ecosystems rather than influences attributed to any single source.

Sage-grouse populations depend on relatively large expanses of sagebrush-dominated shrub steppe. However, the appropriate patch size needed for winter and breeding habitats used by greater sage-grouse is uncertain. It is likely that this patch size is not a fixed amount but depends on various factors including migration patterns and productivity of the habitat.

Greater Sage-grouse Populations

We describe the population biology (Chapter 3) and habitat needs (Chapter 4) of sage-grouse. Chapter 6 addresses sage-grouse databases, distribution, and population trends. We also review information on genetics (Chapter 8), hunting (Chapter 9), predation and disease (chapter 10) and current monitoring techniques (Chapter 11).

Sage-grouse are a relatively long-lived species of upland game bird with low reproductive rates. Sage-grouse are entirely dependent on sagebrush habitats for successful reproduction and winter survival. Disease and hunting have generally not been major factors in sage-grouse population change but new information suggests West Nile Virus may pose a significant threat.

All state and provincial fish and wildlife agencies monitor sage-grouse breeding populations annually, but monitoring techniques vary among areas and years both within and among agencies. This variation complicates attempts to understand grouse population trends and make comparisons among areas. However, virtually all states and provinces have increased monitoring efforts, especially over the last 10 years. Range-wide, population monitoring efforts increased by 737% between 1965 and 2003. The largest increases in effort occurred in Montana and Wyoming, two of the key sage-grouse states. Our analysis indicated that 2,637 leks are now censused annually.

We conducted a comprehensive analysis of sage-grouse population changes throughout their range by accumulating and analyzing all available male counts at 5,585 leks identified since agencies began routine monitoring of this species. We applied several different techniques to evaluate greater sage-grouse populations in North America. These techniques included: 1) changes in the average and median number of males per active lek; 2) changes in the average and median number of males per lek (including leks that are inactive); 3) annual changes in the number of males attending leks monitored in consecutive years (rate of change data); 4) evaluation of spatial patterns of lek extirpation; 5) evaluation of patterns of range extirpation; and 6) delineation and evaluation of distinct breeding populations.

The overall distribution of potential pre-settlement habitat was estimated to have been 1,200,483 km^2 and the current distribution to be 668,412 km^2. Approximately 56% of the potential pre-settlement distribution of habitat is currently occupied. The area currently occupied by sage-grouse is clearly smaller than was occupied in pre-settlement times.

With most of the analysis of sage-grouse numbers, we focused on the 1965-2003 period. Although many states and provinces were collecting data prior to 1965, this 39-year range provided an opportunity to analyze data after a sample of leks had been identified and protocols for data collection had been established and implemented. Eleven of 13 (85%) states and provinces showed significant long-term declines in size of active leks. Similarly, eight of 10 states (80%) showed population declines over the same time frame. Two of 10 (20%) appeared to be stable or slightly increasing. Only California had an increase in both the population index and lek size.

When sage-grouse breeding populations were delineated based on separation by distance and unsuitable habitat, trends for populations were similar to those of the states. Our analysis of the entire sage-grouse population indicated that sage-grouse declined dramatically from the 1960s to the mid-1980s and then tended to stabilize. This analysis indicated that these changes were often not density-independent. If trends characteristic of the 1960s through the mid-1980s continued, sage-grouse had a relatively high likelihood of being extirpated. However, those trends have not continued. As a result, data suggest sage-grouse populations in most areas have been relatively stable or slightly declining during the last 15-20 years. In many areas numbers increased between 1995 and 2003. Although there are areas that presently could be considered population strongholds, some populations are still declining rather precipitously in various portions of the species range.

Annual rates of change suggest a long-term decline for greater sage-grouse in western North America and support the trend information obtained from lek attendance (males/lek) data. Sage-grouse populations declined at an overall rate of 2.0% per year from 1965 to 2003. From 1965-85, the population declined at an average rate of 3.5%. From 1986 to 2003, the population declined at a lower rate of 0.4% and fluctuated around a level that was 5% lower than the 2003 population. A total of 50,566 male sage-grouse were counted on leks in 2003 throughout western North America. However, we are not optimistic about the future of sage-grouse because of long-term population declines coupled with continued loss and degradation of habitat and other factors (including West Nile Virus).

Conclusion

This report is the first detailed assessment of range-wide population and habitat data for greater sage-grouse. The information and analysis included in this report can be used to monitor future population changes and responses to management activities. As such, we hope that the information that we have presented now can be the foundation for increasing our understanding of the ecology of sagebrush-dominated landscapes and species that depend upon them.

Chapter 1

Introduction

CHAPTER 1

Introduction

Abstract. Population declines of greater sage-grouse (*Centrocercus urophasianus*) and alterations and loss of sagebrush (*Artemisia* spp.) have prompted petitions to the U.S. Fish and Wildlife Service to list the species under the Endangered Species Act. Our objectives were to present an unbiased assessment from an ecological perspective of the current status and the potential factors that influenced the long-term conservation of greater sage-grouse populations and the sagebrush ecosystems on which they depend. We reviewed the primary literature, and conducted new analyses and presented results on data collected for greater sage-grouse populations and sagebrush habitats. Our approach was large-scale, and was intended to identify regional patterns of disturbances, land use practices, and population trends. A blind review of this document was conducted by the Ecological Society of America. In addition, members of the National Sage-grouse Conservation Planning Framework Team and representatives from each state and province in the current range of greater sage-grouse reviewed the document for completeness and technical accuracy. In this chapter, we present the background, objectives, perspective, and geographical and temporal scope for the Conservation Assessment. Because 70% of the lands dominated by sagebrush cover are managed by public agencies, we summarized the primary legislation directing the historical disposition and governing the use of public lands. We also presented information on the administrative jurisdiction of sagebrush habitats because many of the stressors on sagebrush ecosystems involve land use and management practices. However, we did not provide management recommendations. Rather, this document was intended as an objective scientific presentation of the individual and cumulative influences on greater sage-grouse and sagebrush habitats.

Range-wide Conservation Assessment

Background and Rationale

Historically, greater sage-grouse occurred in parts of 12 states within the western United States and 3 Canadian provinces (Fig. 1.1) (Schroeder et al. 2004). Greater sage-grouse populations have declined throughout much of their former range and have been extirpated from Nebraska, and British Columbia (Connelly and Braun 1997, Schroeder et al. 1999, Schroeder et al. 2004). The historical distribution of greater sage-grouse in Arizona currently is being questioned. Estimates of regional declines ranged from 17 to 47% (Connelly and Braun 1997). Greater sage-grouse currently occupy 670,000 km^2, or 56%, of their potential pre-settlement range, which once covered approximately 1,200,000 km^2 (Schroeder et al. 2004). Current distributions of “fringe” populations are fragmented and increasingly disjunct from core regions of the distribution (e.g., Mono Lake, California; eastern Washington; southern Utah) (Schroeder et al. 1999). Despite widespread concerns regarding the species’ status and declining numbers, there has been no definitive range-wide assessment of sage-grouse populations and habitats.

The greater sage-grouse is entirely dependent on sagebrush ecosystems that dominate much of western North America. The sagebrush biome, comprised primarily of 20 taxa encompassing 11 major *Artemisia* species and subspecies groups (McArthur and Plummer 1978, McArthur and Sanderson 1999), covers approximately 480,000 km^2 (118.6 million acres) and includes 14 states (Washington, Oregon, California, Idaho, Nevada, Utah, Arizona, Montana, Wyoming, Colorado, New Mexico, Nebraska, South Dakota, and North Dakota) (sagebrush habitats in Oklahoma and Kansas were outside of the pre-settlement range of greater sage-grouse were not included in this assessment) and 3 provinces (Alberta, British Columbia, Saskatchewan) (Fig. 1.2). Vegetation and wildlife communities vary greatly across the range covered by sagebrush as a function of differences in underlying soils, climate, elevation, and geographic location (Miller and Eddleman 2001). The relatively simple structure and floristic characteristics of sagebrush landscapes (West 1996, West and Young 2000) mask complex community dynamics, disturbance regimes, and system resiliency.

Three fundamental characteristics of the landscape that early European explorers once described as a vast sea of sagebrush (Fremont 1845) have been altered from pre-settlement conditions. First, the total land area dominated by sagebrush has been reduced in many regions of the sagebrush biome. For example, approximately 75% of the shrubsteppe habitats occurring on deep, loamy soils in the state of Washington and virtually all of the basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*) habitats in southern Idaho have been converted to agricultural croplands (Hironaka et al. 1983, Noss et al. 1995, McDonald and Reese 1998, Vander Haegen et al. 2000). Second, the composition of sagebrush communities has been changed, primarily through alterations in the understory vegetation and soils. Replacement of native perennial bunchgrasses by cheatgrass (*Bromus tectorum*), an exotic annual, has profoundly altered the fire regime and led to extensive loss of large expanses of sagebrush habitats (d'Antonio and Vitousek 1992, West and Young 2000, Brooks and Pyke 2001). Finally, the configuration of sagebrush habitats within the larger context of the landscape has been changed. The increased edge in landscapes fragmented by roads, power-lines, fences, and other linear features promote spread of exotic invasive species (Gelbard and Belnap 2003), facilitates predator movements (Tewksbury et al. 2002), and isolates wildlife populations (Saunders et al. 1991, Trombulak and Frissell 2000). Changes in quantity, composition, and configuration of sagebrush habitats have consequences on the ecological processes within the sagebrush ecosystem and the resources available to support wildlife (Wisdom et al. 2002). Few pristine and intact sagebrush ecosystems remain (Noss and Peters 1995, Noss et al. 1995, West 1996, Mac et al. 1998).

Over 350 species of flora and fauna depend on sagebrush habitats for all or part of their existence; a high proportion of the endemic and imperiled species in the western United States are found within the sagebrush distribution. The Great Basin ecoregion contains the second highest number of imperiled endemic species in the United States (Chaplin et al. 2000:166). The Columbia Basin population of pygmy rabbits (*Brachylagus idahoensis*) and Gunnison sage-grouse (*Centrocercus minimus*) are highly dependent on sagebrush habitats and currently are candidate species for federal listing under the Endangered Species Act (U.S. Fish and Wildlife Service 2000, U.S. Fish and Wildlife Service 2003). The current range occupied by Gunnison sage-grouse has been reduced to 5,000 km^2 from its pre-settlement distribution of 45,000 km^2

primarily because of habitat loss and alteration (Oyler-McCance et al. 2001, Schroeder et al. 2004).

Petitions filed to list the greater sage-grouse under the Endangered Species Act are based on concerns for long-term conservation because of potential threats to the species and the sagebrush habitats on which it depends (Wambolt et al. 2002). Public perception has progressed beyond the prediction that the "much-maligned sagebrush will be regarded with increasing favor by land managers" (McArthur and Plummer 1978) to genuine concern about these ecosystems (Braun et al. 1976, Knick 1999) to requests for legal action (Chapter 2).

A decision to give the greater sage-grouse protected status across its entire range has significant consequences for management and use of a large part of the western United States. Less than 1% of the 668,412 km^2 currently occupied by greater sage-grouse, and very little sagebrush habitat is legally protected (Caicco et al. 1995, Stoms et al. 1998, Scott et al. 2001, Wright et al. 2001, Knick et al. 2003). Multiple-use management dominates approximately 70% of the sagebrush habitats, which are owned publicly (Box 1990, Poling 1991). Consumptive uses that potentially influence sagebrush habitats include livestock grazing, mining, energy development, conversion to agriculture, and urbanization. Non-consumptive uses, such as use of off-road vehicles for recreation, also have the potential to influence habitats and populations of sage-grouse. Greater sage-grouse also are legally hunted in 10 states, and some populations are also subject to subsistence hunting by Native Americans.

Objectives and Perspective of the Conservation Assessment

Our primary objective was to document the current status and the potential factors that influence the long-term conservation of greater sage-grouse populations and the sagebrush ecosystems on which they depend. We based our analysis throughout this document on an ecological perspective of the dynamics inherent in sagebrush ecosystems and the requirements of greater sage-grouse. In contrast, land-use perspectives have goals to maximize a particular function that may have objectives competing with other resource use. For example, evaluation of sagebrush communities primarily based on their ability to provide forage for livestock may result in extensive alterations that are unsuitable for greater sage-grouse and other species dependent on sagebrush habitats (Schneegas 1967, Klebenow 1970, Braun et al. 1976, Reynolds and Trost 1981, Crawford et al. 2004).

An ecological perspective is critical to providing a common denominator within which land uses can be evaluated in relation to disturbance and resiliency of the system. We have used ecological terms to describe population or habitat patterns and processes rather than value-laden terms which may have alternate connotations. Commonly used terms, such as "decadent" or "catastrophic" evoke an attitude that something must be fixed or controlled, or is irreversible. Similarly, "range improvement" may reference multiple objectives but has traditionally connoted a management effort to increase the land's capacity to provide livestock forage and not an alteration that necessarily reflects changes beneficial to sage-grouse (Vale 1974, Crawford et al. 2004). To present an unbiased assessment, we have attempted to use objective descriptors throughout.

We did not evaluate the feasibility of a land use, the need for a resource commodity, or the public perception of land use (e.g., Donahue 1999). Nor did we present strategies for mitigating uses or recommend alternative levels of use. Rather, we attempted to answer the questions about current uses (actions) and the way in which they influence the ecological functions of sagebrush ecosystems (reactions). In that context, we presented the ecological framework, but also recognize that political, economic, and social arenas also are components in the discussions of how we use sagebrush ecosystems (Torell et al. 2002, Wambolt et al. 2002).

Geographic, Temporal, Jurisdictional, and Scientific Scope

We conducted our assessment of the status of greater sage-grouse and sagebrush habitats for the region delineated by the pre-settlement distribution of greater sage-grouse (Schroeder et al. 2004). Choosing the larger historical distribution as our base analysis region permitted us to compare differences relative to the current distribution and detect potential determinants of population changes or extirpation. We buffered the historical distribution by 50-km to include an evaluation of external factors that may have contributed to current trends in populations or habitats. For example, this buffer would include potential spatial processes including spread of invasive plant species, influence of hunting from urban centers, or movement of predators from farmlands. Therefore, the total area bounded by this assessment was approximately 2,063,000 km^2 (Fig. 1.3, Table 1.1).

The large area included in the assessment dictated that we addressed issues at the spatial and temporal scales appropriate to understanding how land use and habitat changes influenced greater sage-grouse populations or altered processes, such as disturbance, within sagebrush ecosystems. Thus, we focused on large-scale, regional patterns and processes in the sagebrush biome to identify the dominant patterns and processes that were expressed over large regions. In doing so, we recognized the hierarchical nature of ecological systems (Allen and Starr 1982, Peterson and Parker 1998) and the way in which local patterns and processes interact to influence dynamics at regional scales. Impacts from land uses or "natural" habitat disturbance can range from loss of a single sagebrush plant to effecting changes across entire landscapes. Local disturbances such as a small water impoundment or the pad on which an oilrig was constructed may directly impact a relatively small (<2 ha) area of sagebrush habitat. We focused on the cumulative contribution of water impoundments or oil rigs to directly remove sagebrush habitat and indirectly to change the dynamics of sagebrush ecosystems. Similarly, we did not assess individual grazing allotments to determine the effects of livestock grazing but rather addressed the collective influence of livestock grazing across the sagebrush biome. Hunting also can be assessed from the perspective of deaths to individual birds or by the impact on populations. We focused on large-scale population changes within and among states. To the extent possible, we agglomerated local information into the larger context of patterns and processes for populations and ecosystems. Therefore, our analysis was designed to identify and evaluate issues at the scale at which greater sage-grouse populations and sagebrush ecosystems were likely to be most influenced and at which management might be most effective.

The temporal period covered by this assessment ranges from settlement by Europeans in western North America and the disposition of western lands (approximately mid-1800's) to the most recent statistics available on the current status of greater sage-grouse and sagebrush habitats (generally 2000-2003). Although we attempted to reconstruct as much of the historical setting as possible to interpret the underlying mechanisms responsible for today's conditions, our primary focus is on the current state of these ecosystems and the implications for the future. Concerns about the ecological status of sagebrush ecosystems have been expressed for a long time (Patterson 1952, Braun et al. 1976). However, we still lack baseline information across much of the sagebrush biome against which to evaluate population and habitat changes. Therefore, most information that we present is recent but perhaps now we can begin the daunting task of providing a baseline database for future efforts.

Many of our summary analyses are presented by administrative units, and primarily are organized by state or province. State statistics were derived for Arizona, California, Colorado, Idaho, Montana, Nebraska, Nevada, New Mexico, North Dakota, Oregon, South Dakota, Utah, Washington, and Wyoming in the United States. Statistics derived from county records, such as the areas in the Conservation Reserve Program, were presented graphically in maps but were presented by state when summarized in tabular form. We also included information for the Canadian provinces of British Columbia, Alberta, and Saskatchewan.

A plethora of ecological regions have been designated by agencies and organizations. Each of these regional delineations, such as the USDA Forest Service Ecological Units (ECOMAP 1993), The Nature Conservancy's Ecoregions (The Nature Conservancy 1999), and Küchler's ecoregions (Küchler 1964) emphasize different characteristics (e.g., hydrological or physiognomic) or serve different purposes (Fig. 1.4). When summarizing information by regions, we used 7 primary geographical subdivisions within the sagebrush range (West 1983) because of the general similarities in climate, elevation, topography, geology, and soils within each division (Miller and Eddleman 2001) (Fig. 1.5).

The spatial and temporal resolution (the unit at which measurements were taken) of data used in this assessment varied greatly. For example, many of the habitat analyses were based on satellite imagery that varied in spatial resolution from 90-m to 1.1-km grid cells and for comparisons across temporal resolution that varied from weeks to decades. We have documented the measurement units in the methods associated with each analysis section or in the metadata record attached to each data set.

We have listed all measurements in metric units through this document. Units of area were reported in square kilometers (km^2); we used hectares (ha) when the area was <10 km^2.

Treatment of Uncertainty

Our analysis presented in this Conservation Assessment is based primarily on correlative information. Controls on disturbances or availability of comparison regions are difficult and often not possible, particularly when comparing large-scale effects across landscapes and among populations. Few studies at regional scales are able to attribute cause and effect in relation to management action. Because of the nature of some land uses (we cannot develop one oil field and have another undeveloped field equal in all other variables), our evaluation suffers from a lack of replication (Johnson 2002). We have dealt with these situations by statistically comparing biological variability and responses (Osenberg et al. 1994, Underwood 1994, Wiens and Parker 1995, Manly 2001).

We present our treatment of the data whenever appropriate with an associated statistical probability of accepting the result rather than a subjective or descriptive appraisal. In statistical tests, we must recognize the problems of Type I (concluding that a perturbation has an effect when it did not) and Type II (concluding that a disturbance has no effect when one actually exists) errors. Each of these statistical errors has consequences for our interpretation and subsequent management action (Eberhardt and Thomas 1991).

We have tried to resolve differences resulting from methods for measuring vegetation, lek counts, resolutions of GIS layers and information, and terminology when compiling information across administrative or research boundaries. Nonetheless, reconciling differences in spatial or temporal resolution, collection or analysis method, and incomplete or incongruent data represented one of the most significant challenges to developing consistent data for the large region covered by this assessment. Whenever possible, we merged individual datasets if data and underlying methods were complimentary. However, we often had only data from part of the region (e.g., the INVADERS plant database covered only Washington, Oregon, Idaho, Montana, and Wyoming; Toney et al. 1998) or disparate data sources on which to conduct our analyses.

We often relied on statistics created for a single management entity in our assessment. The Public Land Statistics compiled by the U.S. Bureau of Land Management represent activities by that agency. Even though the U.S. Bureau of Land Management is the largest federal land management authority for sagebrush lands in the United States (Knick et al. 2003), these statistics cannot be extrapolated to activities on private lands or to other agencies. They represent only part of the entire management and use scenario for sagebrush habitats.

Each of these problems in data management and analysis techniques introduce uncertainty into our assessment. Consequently, we have chosen to limit our projections of habitats and populations through modeling exercises. Although these may provide insights into potential alternatives (e.g., Hemstrom et al. 2002, Wisdom et al. 2002, Pedersen et al. 2003), our primary objective was to present information on the current dynamics and status of sagebrush ecosystems rather than to evaluate management scenarios or project future trends. To do this, we have documented our data and sources, detailed our methods, and presented statistical probabilities to our conclusions.

Review of the Conservation Assessment

Document review was accomplished at several levels. The primary level of data review was the responsibility of the authors of each individual section or chapter. The authors were requested to authenticate their datasets. The next level of review was accomplished by the National Sage-grouse Conservation Planning Framework team. This team reviewed the document for its completeness. Each state and province in the Western Association of Fish and Wildlife Agencies (WAFWA) and within the current range of greater sage-grouse reviewed the document to evaluate the appropriateness of treatment of their data and subsequent analysis. WAFWA was responsible for delivery of this assessment. The final level of review was a scientific peer review. The Ecological Society of America (ESA) conducted a blind peer review of the document using a panel of reviewers recruited from a broad array of natural resource fields.

Scientific Criteria and Documentation of Sources

Criteria for Use of Data and Scientific Information

A broad spectrum of information is available on sage-grouse and their habitats. This information includes newspaper and sporting magazine articles, newsletters, agency reports, technical reports issued by agencies and universities with little or no peer review, peer-reviewed agency and university technical reports, masters and doctorate theses and peer reviewed papers in scientific journals. As a general rule, we have greatest confidence in findings and conclusions developed in scientific papers. Normally, results from graduate theses are also quite reliable, as well as those from technical reports issued by agencies and universities that have received outside peer-review. Although findings in non peer-reviewed publications may be correct, we have less confidence in their validity because they have not undergone detailed review by scientists outside of the immediate issuing agency and thus may be subject to biases, deficiencies in study design, or misinterpretation of data.

We did not intend for this Conservation Assessment to be an extensive bibliography of all available material. Rather, we have considered only sources for information that have a high degree of reliability and stability. Thus, for findings, conclusions, and management concerns incorporated into the conservation assessment, we depended on papers published in scientific journals, peer-reviewed agency reports, and graduate theses. We largely avoided other sources of information that might contain less reliable information. However, if information on a particular aspect was found only in a nonpeer-reviewed source, we presented that information but provided a caveat that indicated the source. We considered state and federal reports (e.g., Pittman-Robertson reports) as well as online documents (e.g., U.S. Public Land Statistics) as reliable sources for data sets on sage-grouse populations (e.g., harvest, lek counts) and habitat attributes (e.g., number of water developments, miles of fencing).

Documentation of Data and Sources

We have archived the data and documented the sources used in this conservation assessment on the U.S. Geological Survey (2001) SAGEMAP Website (http://sagemap.wr.usgs.gov/conservation_assessment.htm). All spatial datasets used in our analysis will be available for download with the exception of proprietary data. Each dataset will have an associated metadata record documenting the original data and GIS procedures.

Management and Stewardship of Sagebrush Habitats

Principal Legislation Governing the Management and Use of Public Lands

Landscapes dominated by sagebrush have been managed primarily for multiple use since European settlement began in the 1800's (Table 1.2). Mining for coal and mineral resources, forage production for livestock, and developing lands for irrigated agriculture dominated uses of sagebrush lands. By the late 1800's, the federal government began a series of legislative actions under a succession of Homestead Acts to dispose of public lands to the private sector. Approximately 1.2 million km^2 of public lands were disposed of under the Homestead Acts (Ross 1984). Large amounts of lands also were granted to build railroads connecting eastern and western United States.

During the same period, a series of legislative acts were passed to regulate grazing on public lands and delegated responsibility to the U.S. Forest Service in the Department of Agriculture and the Grazing Service in the Department of Interior for administrating public land grazing. The Taylor Grazing Act passed in 1934 authorized the Secretary of Interior to establish grazing districts of "vacant, unappropriated and unreserved land from any parts of the public domain, excluding Alaska, which are not national forests, parks and monuments, Indian reservations, railroad grant lands, or revested Coos Bay Wagon Road grant lands, and which are valuable chiefly for grazing and raising forage crops." The Secretary of Interior also was authorized to issue permits to graze livestock upon annual payment of fees, of which a portion was returned to the individual states. Public lands not reserved or withdrawn as refuges were designated as "national resource lands" and placed under the jurisdiction of the federal Grazing Service, which was merged with the General Land Office in 1946 to form the U.S. Bureau of Land Management. The U.S. Bureau of Land Management is the principal federal agency in the United States responsible for management of sagebrush habitats (Table 1.3). More recent legislation (Federal Land Policy and Management Act 1976, Public Rangelands Improvement Act 1978) has reaffirmed administrative policies that public lands are to be managed for multiple use and sustained yield.

Passage of the Energy Policy and Conservation Act (1975) emphasized the need to stabilize the supply of energy and develop fossil fuels located on federal public lands. The reauthorization of the Energy Policy and Conservation Act in 2000 also directed the U.S. Departments of Energy, Agriculture, and Interior to inventory all onshore oil and gas reserves and to identify impediments to the development of those resources. Executive Order 13212, signed in 2001, stated "agencies shall expedite their review of permits or take other actions as necessary to accelerate the completion of such projects, while maintaining safety, public health, and environmental protections" (White House 2001). In response, the U.S. Bureau of Land Management has followed an administrative policy to ensure the timely development of these critical energy resources in an environmentally sound manner and has directed land-use planners to not unduly restrict access to Federal lands, while continuing to protect resources when they review oil and gas lease stipulations, (U.S. Bureau of Land Management 2003a,b,c).

Most of the legislation establishing land use policy contained language directing that conservation of land, resources, and wildlife be considered in implementing management actions. However, the Wilderness Act (1964) specifically recognized the need to protect areas from human encroachment and use and to preserve those places for future generations. The National Environmental Policy Act (1969) and the Endangered Species Act (1973) specifically required an assessment, review, or consultation of the potential for management actions to adversely impact species, their habitats, or the quality of their environments.

A perspective that the primary use of western public shrublands should be for commodities has often conflicted with ecological or botanical perspectives, beginning with the early surveyors and scientists studying these habitats in the 1800's (Box 1990). These different perspectives continue to shape our view and use of public lands (Young et al. 1981, Poling 1991, West 1996, Holechek et al. 1998, Wambolt et al. 2002, Crawford et al. 2004).

Stewardship of Sagebrush Lands

Sagebrush habitats included in this assessment covered approximately 48 million ha and were distributed across 13 states and 3 provinces (Fig. 1.6). Nevada and Wyoming had the largest total area covered by sagebrush. Idaho, Nevada, Oregon, and Wyoming each had >20% of their area dominated by sagebrush (Table 1.1). Approximately 12% of the area in Washington and 17% in Utah was in sagebrush habitat. All other states and provinces had <10% of their total area in sagebrush cover (Table 1.1). We likely have underestimated the area covered by sagebrush in Montana because silver sagebrush (*Artemisia cana*) and Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis*), the dominant sagebrush species in northeastern Montana, are distributed sparsely across much of the region among grassland habitats and are not easily mapped from satellite imagery. We also were unable to obtain current maps of sagebrush for the eastern portions of the sagebrush biome because they had not been completed at the time of this assessment.

Approximately 30% of the sagebrush lands in the United States is privately owned (Table 1.3). Percent of privately owned sagebrush lands within states ranged from 0 to 56% and was greatest in Montana, Colorado, Washington, and South Dakota. Of the states containing the largest total area of sagebrush, the percent in private ownership was 17% in Nevada, 38% in Wyoming, 17% in Idaho, and 27% in Oregon. In Canada, 90% of the sagebrush area in Saskatchewan, 28% in Alberta, and 0% in British Columbia was privately owned.

Federal agencies in the United States were responsible for management of 66% of the sagebrush landscape (Table 1.3). Of these agencies, the U.S. Bureau of Land Management had management authority for one-half of the sagebrush lands in the United States. Within states, the percent of sagebrush habitat managed by the U.S. Bureau of Land Management ranged from <5 (North Dakota, South Dakota, and Washington) to >40% (California, Idaho, Nevada, Oregon, Utah, and Wyoming). The U.S.D.A. Forest Service had stewardship of 8% of the sagebrush habitats. The U.S.D.A. Forest Service managed <10% of the sagebrush habitats within each state except for California, Idaho, North Dakota, and Utah. Other Federal agencies in the U.S. Department of Defense, U.S. Department of Energy, and the U.S. Department of Interior (including the Bureau of Indian Affairs, Fish and Wildlife Service, and National Park Service) were responsible for management of <5% of sagebrush lands within the United States. (Fig 1.7). Almost all sagebrush lands under authority of the US Bureau of Indian Affairs were in Arizona, New Mexico, and South Dakota. (Table 1.3).

State agencies managed 5% of the total landscape dominated by sagebrush in the United States. Only Arizona and Washington had >10% of their sagebrush habitat managed by state agencies.

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Fig. 1.1. Current distribution of sage-grouse and pre-settlement distribution of potential habitat in North America (Schroeder 2004). For reference, Gunnison sage-grouse in southeastern Utah and southwestern Colorado are shown.

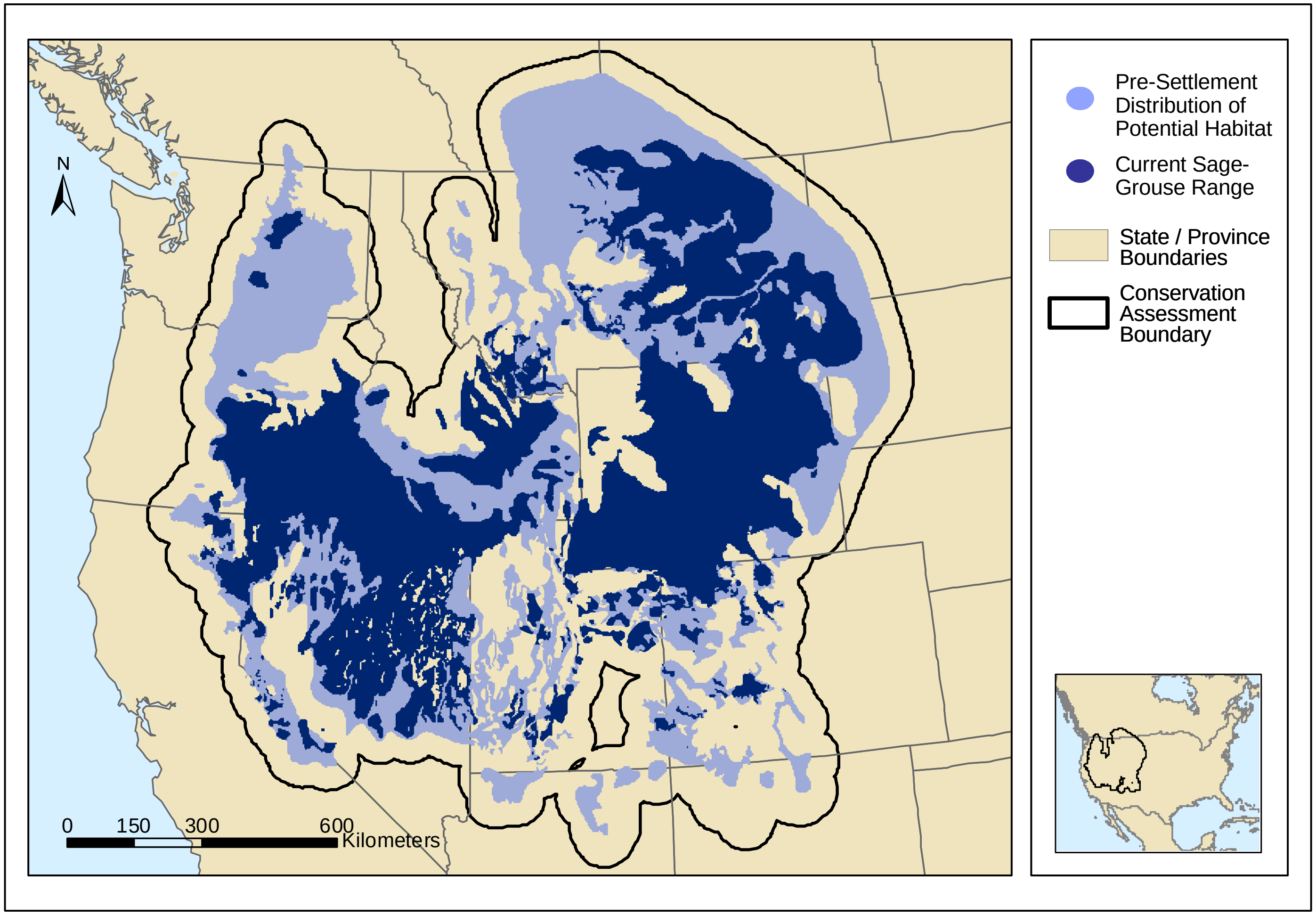

Fig. 1.2. Current distribution of the sagebrush biome in western North America (adapted from West 1983; Miller and Eddleman (2001). Sagebrush distribution in eastern Montana likely is under represented because remote sensing methods from which the map was derived, may not adequately delineate sparsely distributed sagebrush. We could not obtain comparable maps of sagebrush distribution for North Dakota and South Dakota. The map represents the percent of the landscape dominated by sagebrush habitats (Chapter 5), not site-specific values of ground cover. As such, it is intended to be a general representation of sagebrush distribution (more detailed maps of distribution and fragmentation are presented in Chapter 5).

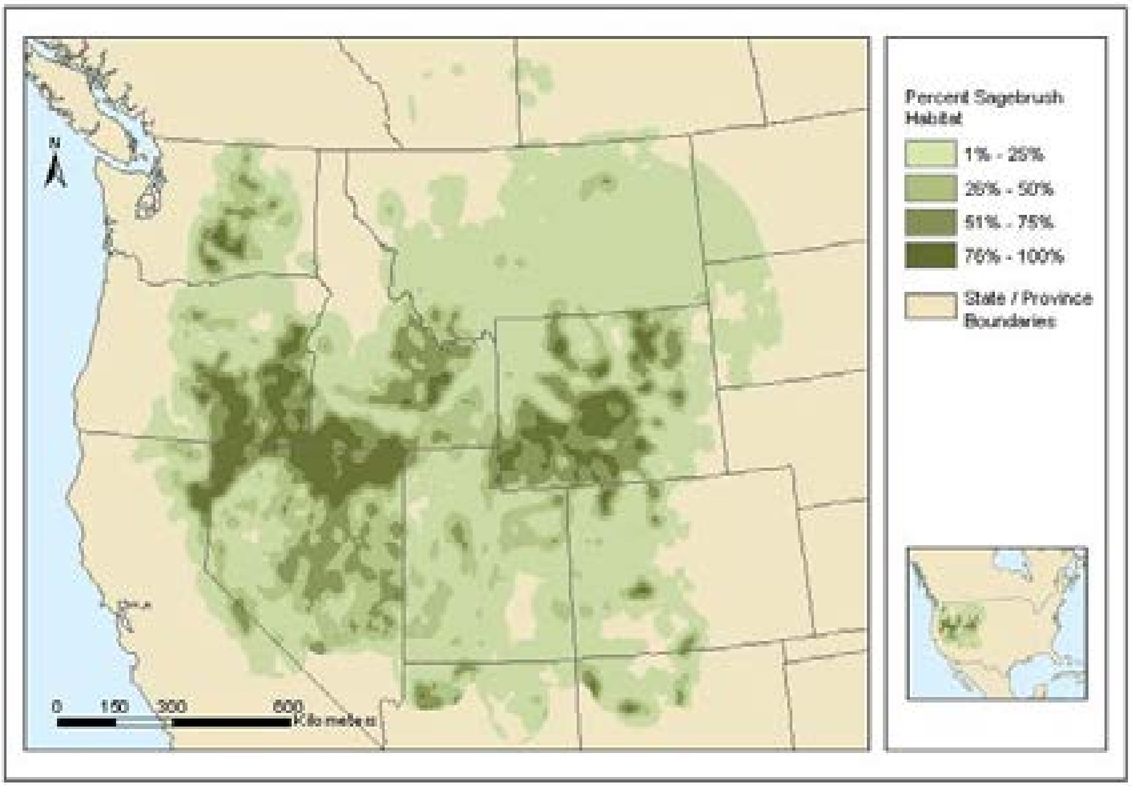

Fig. 1.3. Area included in the range-wide conservation assessment for greater sage-grouse and sagebrush habitats. The study area was delineated by buffering the pre-settlement distribution of sage-grouse by 50-km (including the range of the Gunnison sage-grouse even though that species was not included in the assessment) (Schroeder et al. 2004). Excluded areas or "islands" in southern Utah and southwestern Colorado are a function of the buffering delineation in which those regions are >50 km from the historical or current distribution of sage-grouse.

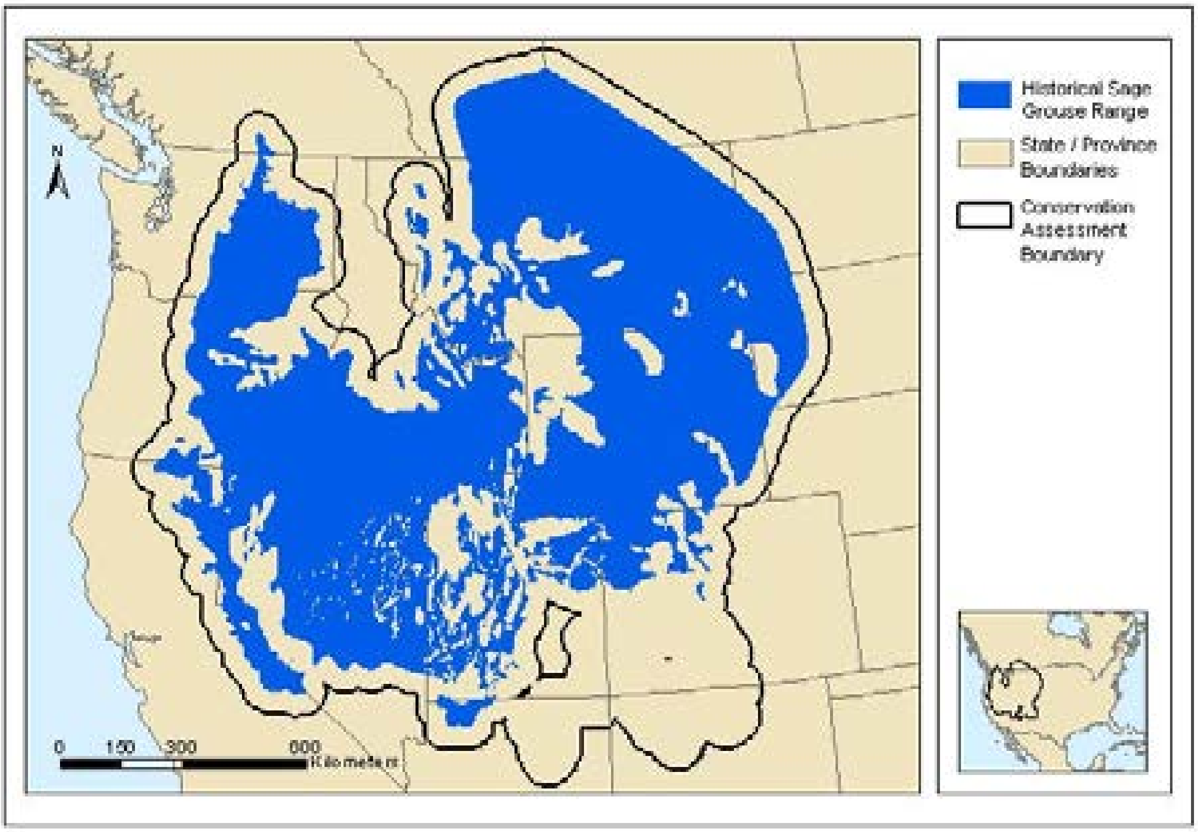

Fig. 1.4. Pre-settlement distribution of sage-grouse with overlays of boundaries for (1) Omernik Regions, (2) The Nature Conservancy ecoregions, (3) Bird Conservation Regions, and (4) Bailey's ecoregions.

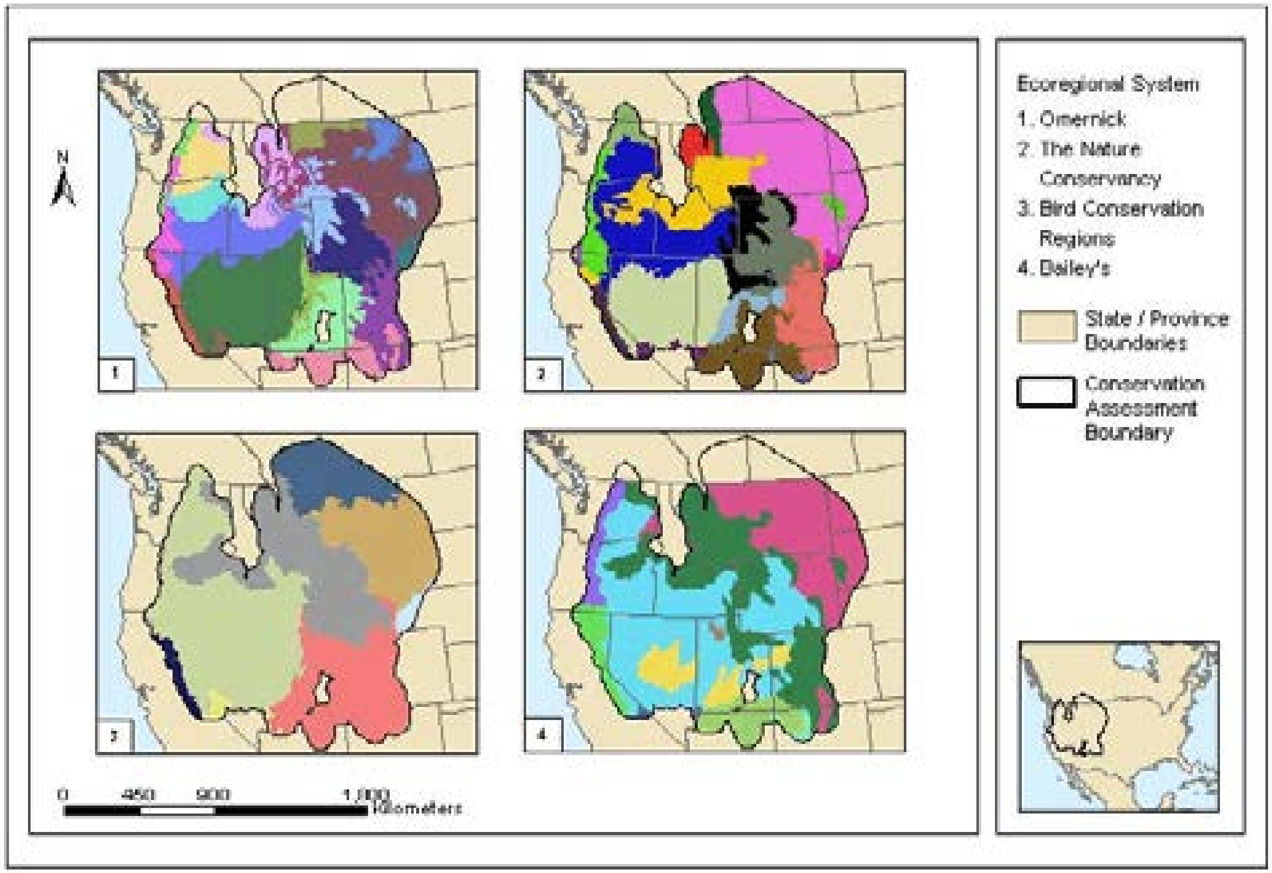

Fig. 1.5. Geographic subdivisions within the sagebrush biome (adapted from West 1983, Miller and Eddleman 2001).

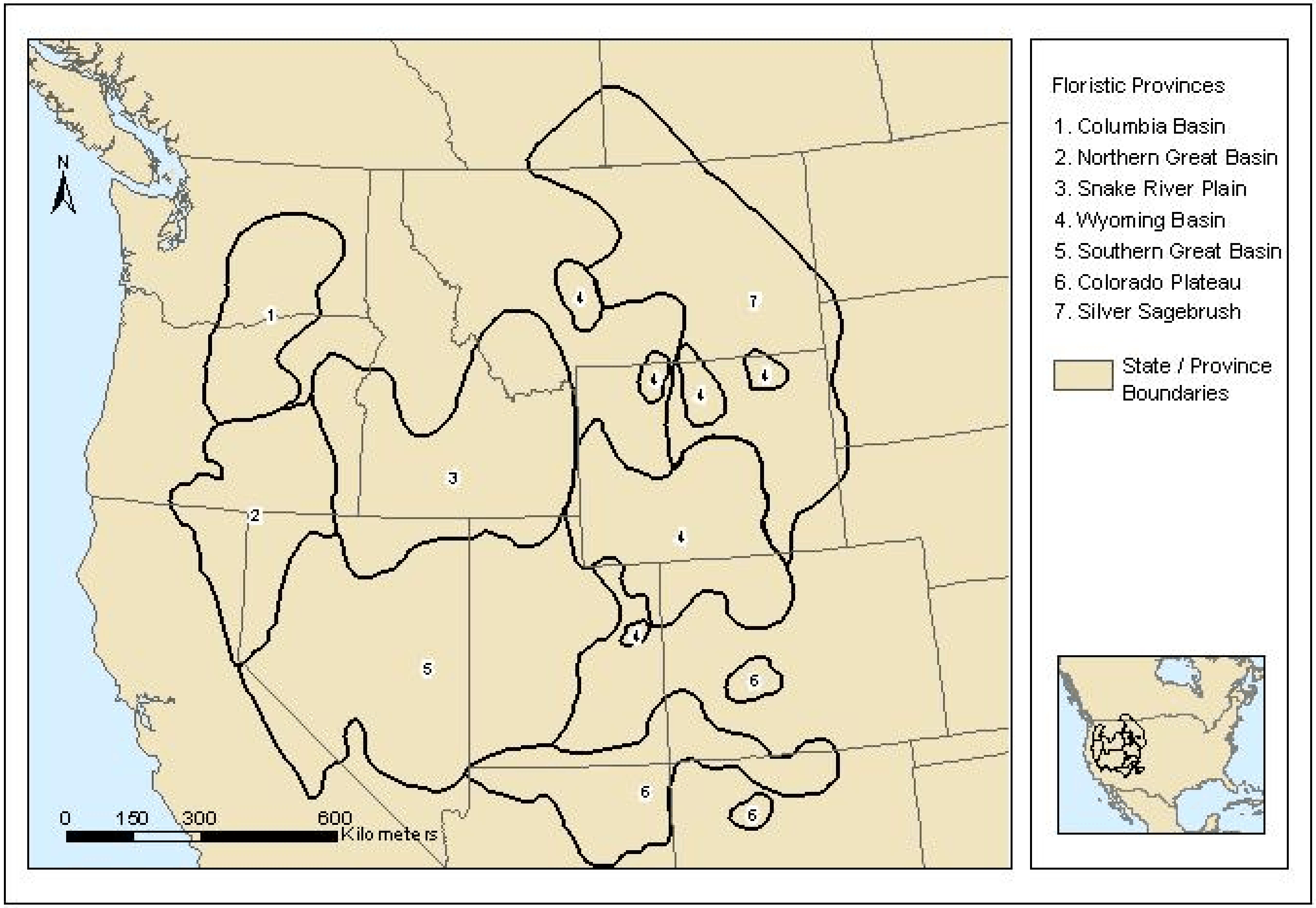

Fig. 1.6. Stewardship of sagebrush lands in the United States and Canada. Location of private lands is shown in Chapter 5 (Fig. 5.19).

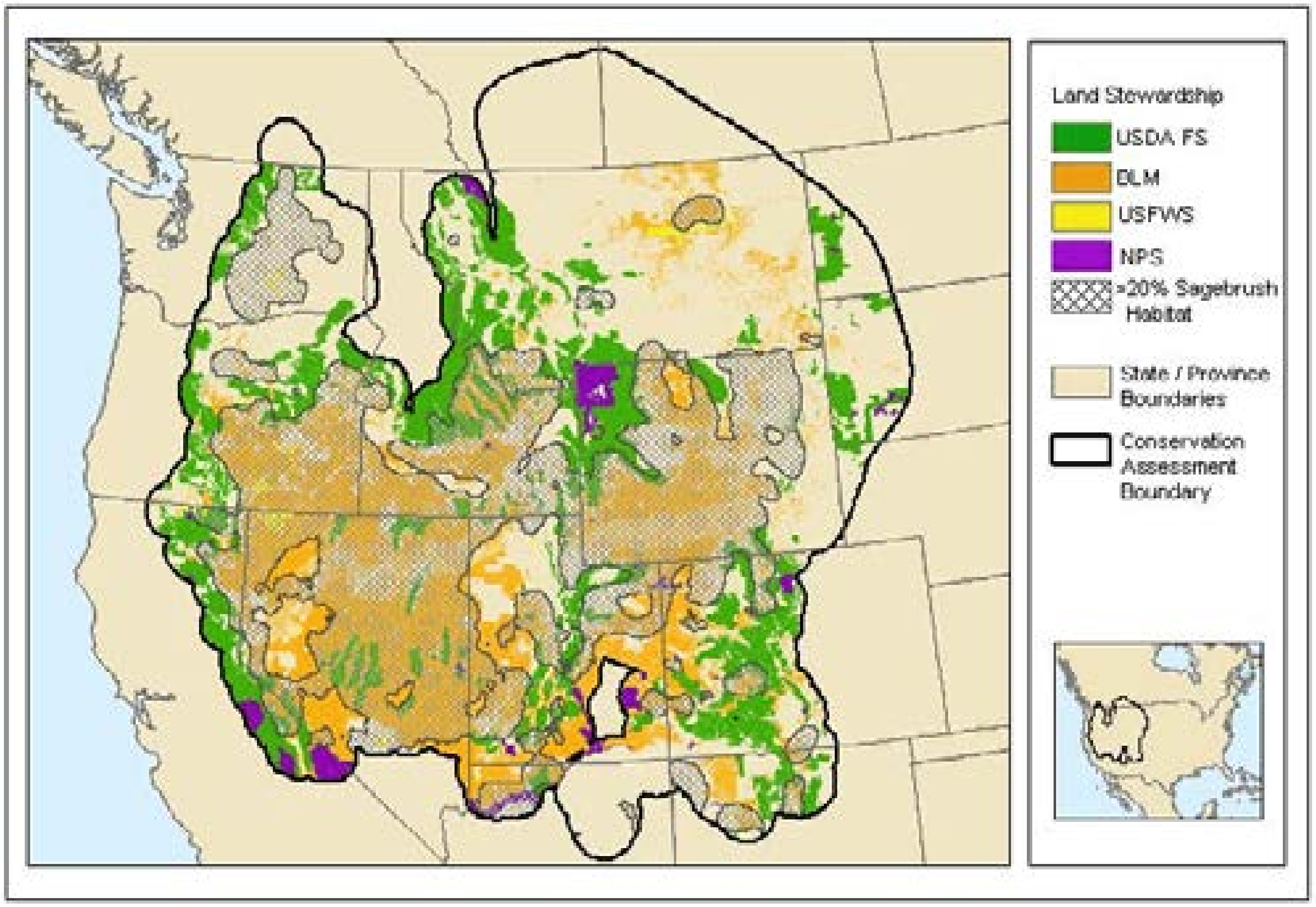

Fig. 1.7. Distribution of national parks and lands managed by the U.S. Department of Defense and the U.S. Department of Energy on which access or land use is restricted.

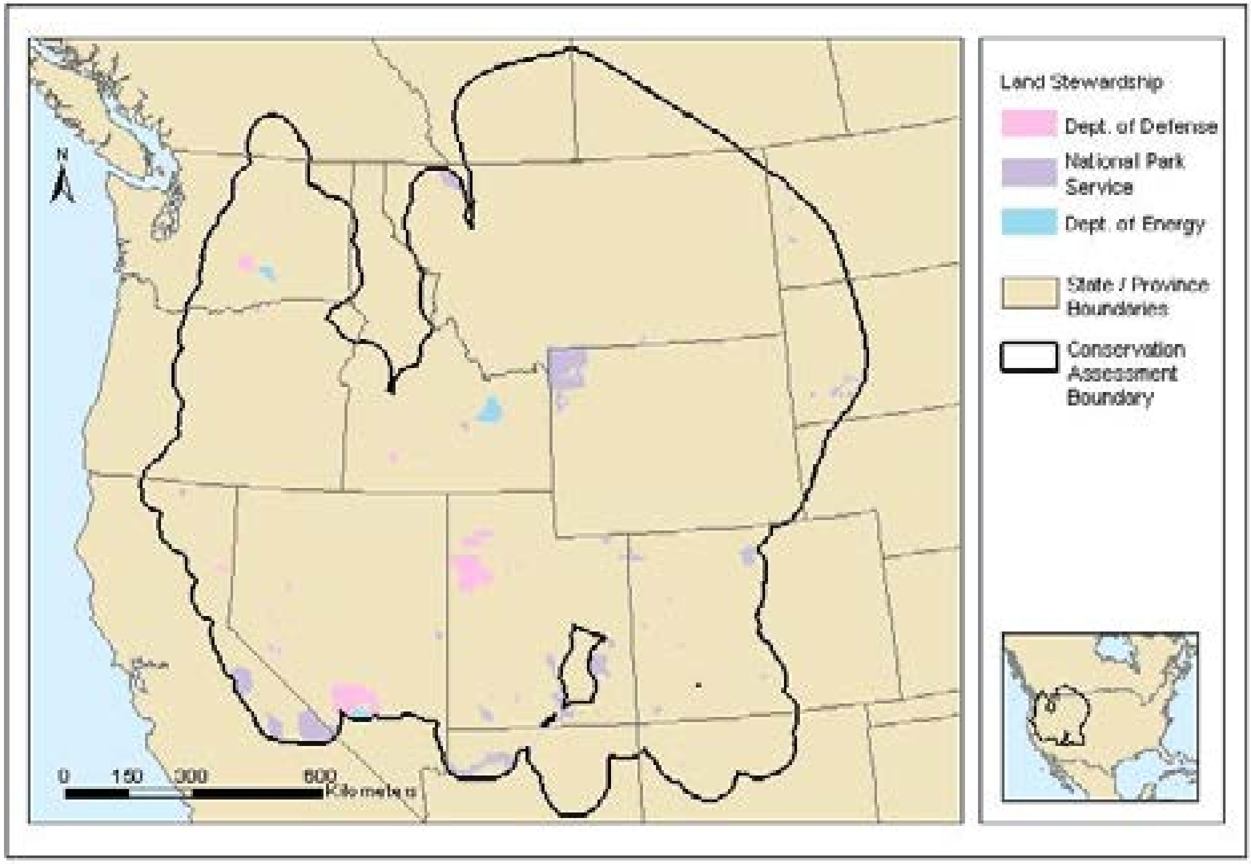

Table 1.2 State and provincial summaries of total area, area included in the Conservation Assessment, and area dominated by sagebrush[a]. All areas are in km^2.

State/Province	Total Area	Area included in Conservation Assessment	Sagebrush area	% of total area
Arizona	294,505	58,734	15,142	5
California	408,638	77,474	12,993	3
Colorado	269,616	146,823	18,993	7
Idaho	215,850	161,195	56,566	26
Montana[b]	381,344	365,187	24,255	6
Nevada	286,626	252,863	108,725	38
New Mexico	315,349	53,518	10,592	3
North Dakota[b]	183,398	30,920	4,283	2
Oregon	251,411	170,017	56,715	23
South Dakota[b]	199,933	55,940	479	0
Utah	219,814	208,475	37,379	17
Washington	174,277	100,435	20,131	12
Wyoming	253,301	252,724	95,699	38
United States			461,954	
Alberta	666,034	61,824	10,620	2
British Columbia	944,510	11,196	1,591	0
Saskatchewan	652,023	55,548	6,969	1
Canada			19,180	
Totals		2,062,872	481,134	

[a]Sagebrush communities included Wyoming and Basin big sagebrush, black sagebrush, low sagebrush, low sagebrush–mountain big sagebrush, low sagebrush–Wyoming big sagebrush, mountain big sagebrush, scabland sagebrush, threetip sagebrush, Wyoming big sagebrush, and Wyoming big sagebrush–squaw apple.

[b]Total area of sagebrush in the eastern portion of the sagebrush biome likely is underestimated because current maps of equivalent spatial and thematic resolutions were not available.

Table 1.2 Principal federal legislation governing the management and use of public sagebrush lands.

Year	Legislative Act		Land Use
1862	Homestead Act	37th Congress, Chapter 75, 12, Stat. 392	Permitted entry on 160 acres provided the settler built a home and lived on the land, and made improvements and farmed it for 5 years.
1872	General Mining Act	30 USC 21-54	Declared that all valuable mineral deposits on lands belonging to the United States were free and open for purchase. Anyone could stake a claim at no cost.
1877	Desert Land Act	43 USC 321-339	Permitted entry on 640 acres at $0.25/acre provided the lands could be irrigated.
1897	USDA Forest Service Organic Act	16 USC 473	Established grazing management on forest reserves
1909	Enlarged Homestead Act	43 USC 218-221	Permitted entry to 320 acres for dry-land farming
1912	Three-year Homestead Act	37 Stat. 123-125	Reduced the occupancy period to 3 years
1916	Stock Raising Homestead Act	Statutes at Large, vol. 39, p.864	Permitted entry to 640 acres that had been designated for grazing. Federal government retained subsurface rights to minerals and coal. The area was still too small for many arid sections.
1920	Mineral Leasing Act	30 USC 181-287	Directed management of the energy resources on Federal lands to be developed by leasing exploration and development rights.
1934	Taylor Grazing Act	43 USC 515-315r	Established grazing fees and districts, lands were classified as to their best use, federal government has to care for the land and take into account the people who use it.
1946	Creation of the U.S. Bureau of Land Management	28 USC 403	Merged the Grazing Service with the General Land Office to form the Bureau of Land Management within the U.S. Dept. of Interior

1954	Recreation and Public Purposes Act	43 USC 869	Authorized the sale or lease of public lands to states, state agencies, other political subdivisions, or nonprofit organizations for recreational or public uses (campgrounds, parks, fairgrounds, landfills, historic monuments).
1964	Wilderness Act	16 USC 11131-1136	Recognized the need for protection and preservation of lands in their natural condition. A wilderness was defined as an area, generally >2000 ha, of underdeveloped Federal land retaining its primeval condition without permanent improvements, such as roads, or human habitation. These lands were to be protected and managed to preserve the natural character for future generations.
1964	Classification and Multiple Use Act	43 USC 2420	Directed that natural resource lands be managed under the principles of multiple use consistent with the Taylor Grazing Act.
1969	National Environmental Policy Act	42 USC 4321-4347	Federal agencies must consider the impact of their actions on the quality of the environment.
1971	Wild Free-Roaming Horses and Burros Act	26 USC 1331-1334	Stated that wild horses and burros were a symbol of the western landscape. Gave the Secretary of Interior the authority to control the proliferation of wild horses and burros.
1973	Endangered Species Act	16 USC 1531-1543	Section 7 required that the U.S. Fish and Wildlife Service must be consulted by Federal agencies to insure that any action authorized, funded or carried out by them is not likely to jeopardize the continued existence of listed species or modify their critical habitat.
1975	Energy Policy and Conservation Act	42 USC 6201-6202	Developed the provisions to stabilize the energy supply through the creation of the Strategic Petroleum Reserve, establish energy conservation programs and regulatory mechanisms, increase the supply of fossil fuels in the United States through price incentives and production requirements, reduce the demand for petroleum products and natural gas by making coal a more

			feasible alternative, assure the reliability of energy data, and conserve water by improving the water efficiency of certain plumbing products and appliances.
1976	Federal Land Policy and Management Act	43 USC 1701-1782	Public lands must be managed for multiple use and sustained yield and maintain quality of land. Directed that a portion of grazing fees should be returned for range improvements. The United States must receive fair market values for the use of public lands and resources unless otherwise provided for by statute.
1977	Surface Mining and Reclamation Act	30 USC 1201-1202	Recognized the need for reclamation of coal and other surface mining areas
1978	Public Rangelands Improvement Act	43 USC 1901-1908	Provided for restoration of damaged rangelands, and recognized the need for a policy of inventory and monitoring. Established a formula for calculating grazing fees.
2000	Energy Policy and Conservation Act (reauthorization)	P.L. 106-469	Called for an inventory of all onshore Federal lands to identify and estimate oil and gas reserves and the extent or nature of any restrictions or impediments to the development of such resources.

Table 1.3 State and provincial summaries of area (km^2 and % of sagebrush area) by management authority and stewardship of sagebrush lands. Specific agencies for which data were presented included the U.S. Bureau of Land Management (BLM), U.S.D.A. Forest Service (USDA FS), Bureau of Indian Affairs (BIA), U.S. Fish and Wildlife Service (US FWS), and U.S. National Park Service (US NPS).

	Sagebrush Management and Stewardship[a]															
	Private		BLM		USDA FS		BIA		US FWS		US NPS		Federal[b]		State	
State/Province	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%
Arizona	2,812	19	3,323	22	872	6	4,637	31	0	0	1,652	0	267	2	1,578	10
California	2,405	19	55,768	43	3,902	30	6	0	70	1	252	0	556	4	158	1
Colorado	9,126	48	6,809	36	1,684	9	213	1	62	0	116	0	51	0	929	5
Idaho	9,852	17	30,065	53	9,996	18	1,053	2	63	0	23	0	2,139	4	3,330	6
Montana[c]	13,642	56	5,574	23	1,471	6	779	3	480	2	79	0	56	0	2,094	9
Nevada	13,800	13	77,654	71	10,261	9	967	1	2,384	2	135	0	3,441	3	21	0
New Mexico	2,087	20	1,956	18	470	4	5,573	53	41	0	8	0	3	0	455	4
North Dakota[c]	2	0	16	0	989	23	316	7	14	0	61	0	42	1	169	4
Oregon	15,363	27	37,138	65	418	1	230	0	999	2	9	0	418	1	2,051	4
South Dakota[c]	222	46	12	3	22	5	218	46	0	0	0	0	4	1	0	0
Utah	10,825	29	16,721	45	4,402	12	1,179	3	0	0	499	0	376	1	3,351	9
Washington	10,590	53	1,011	5	177	1	2,915	14	770	4	15	0	2,160	11	2,407	12
Wyoming	36,004	38	44,952	47	3,633	4	3,524	4	127	0	658	0	301	0	6,376	7
United States	126,730	27	230,807	50	38,297	8	21,610	5	5,010	1	3,506	0	9,814	2	22,918	5
Alberta	2,927	28											7,400	70		
British Columbia	5	0											9	1		
Saskatchewan	6,272	90											283	4		

Canada	9,204	48	7,692	40

[a]GIS coverages of land ownership and management authority were developed from individual state coverages.
[b]For the United States, federal includes the U.S. Bureau of Reclamation, U.S. Department of Energy, and U.S. Department of Defense
[c]Total area of sagebrush in the eastern portion of the sagebrush biome likely is underestimated because current maps of equivalent spatial and thematic resolutions were not available.

Chapter 2

Conservation Status of Greater Sage-grouse Populations

CHAPTER 2

Conservation Status of Greater Sage-grouse Populations

Abstract. Greater sage-grouse (*Centrocercus urophasianus)* conservation efforts began in the mid 1990s in response to perceived declines in numbers, by state wildlife and federal land management agencies. Conservation actions are planned, coordinated, funded and accomplished by a partnership of state and federal agencies, landowners, industry, non-governmental organizations and the public. Six of 11 western states and both Canadian provinces have completed state or provincial strategic plans to manage greater sage-grouse. The remaining five states are working on strategic plans. All plans are expected to be completed by July 2005. Conservation planning and conservation actions have been accomplished by local sage-grouse working groups. These groups are locally based, sage-grouse and sagebrush ecosystem advocates. Forty-one local working groups are active in the western United States and > 70 groups are scheduled. Canada is completing greater sage-grouse conservation efforts through local partnerships. Seven total petitions have been filed with the U.S. Fish and Wildlife Service to protect greater sage-grouse under provisions of the Endangered Species Act (1973). The finding, to protect greater sage-grouse in the state of Washington, was warranted but precluded. The 90-day finding for the petitions to protect greater sage-grouse in Mono Basin, western subspecies and eastern subspecies of sage-grouse were all negative. The remaining three petitions requesting protection for greater sage-grouse across their range have a positive 90-day finding.

Introduction

The Western Association of Fish and Wildlife Agencies (WAFWA) initiated formation of the Western States Sage-grouse Technical Committee in 1954 to develop strategies to monitor and manage sage-grouse. This committee had its first official meeting in 1959 and eventually evolved to include Columbian Sharp-tailed grouse. Contemporary sage-grouse conservation efforts began to focus in 1995 when the Western States Sage and Columbian Sharp-tailed Grouse Technical Committee determined, through harvest estimates and lek counts, that sage-grouse across the west were showing a sustained downward trend. The Technical Committee evaluated trends in number and distribution and recommended that WAFWA begin proactive conservation measures to protect sage-grouse and sagebrush habitats. In 1996, the Western Association of Fish and Wildlife Agencies developed a Memorandum of Understanding between their members regarding sage-grouse conservation. An element of that MOU suggested that each state begin local area conservation planning groups to address sage-grouse conservation issues at population levels. A reiteration of this MOU along with a MOU between WAFWA, U.S. Bureau of Land Management, U.S. Forest Service and U.S. Fish and Wildlife Service was executed in 1999 and 2000 (Appendix 1). This chapter outlines the status of wildlife laws, petitions filed to protect sage-grouse and the status of conservation planning efforts in the various states and provinces. Many of the reports, plans and petitions are not peer reviewed, are the output from *ad hoc* committees, and have not been widely published or are works in progress.

Greater sage-grouse are classified as resident wildlife by all states and provinces. State and provincial laws have classified the species as either protected or a game bird dating back to the 1800's or early 1900's. This classification generally allowed the direct human takings of the bird during sanctioned hunting seasons. Hunting seasons were relatively liberal with high daily limits during the late 1800's and generally short with small daily limits during the last quarter century. Hunting seasons and harvest are addressed in Chapter 9.

United States and Canadian Federal Laws

Greater sage-grouse in the United States are managed by the various states as resident native game birds. No federal laws provide greater sage-grouse extraordinary status.

Greater sage-grouse are cooperatively managed by provincial and federal governments in Canada. The greater sage-grouse is afforded legal protection under schedule 1, of the Species at Risk Act (Canada Gazette, Part III, Chapter 29, Vol. 25, No. 3, 2002). The Species at Risk Act is similar to the United States Endangered Species Act of 1973. The purpose of the Act is to prevent the extinction or extirpation of any indigenous Canadian wildlife species, subspecies or distinct population segment. The Act also provides for the recovery of endangered or threatened wildlife and encourages the management of other species to prevent them from becoming species at risk.

State and Provincial Laws

Alberta

Alberta manages greater sage-grouse under the statutory authority of its Wildlife Act Chapter/Regulation: W-10 RSA 2000. Greater sage-grouse are classified as endangered under Schedule 6, Part 1, sub-part 12, of Alberta Wildlife Regulation 143/97. These laws and regulations provide greater sage-grouse with protection of birds or nests and provide for the development of recovery strategies and plans.

California

California Department of Fish and Game manages greater sage-grouse under various Fish and Game Codes (Title 14). Specifically, Part 1, Chapter 8, Article 1, section 1801 provides state policy.

> *"1801. It is hereby declared to be the policy of the state to encourage the preservation, conservation, and maintenance of wildlife resources under the jurisdiction and influence of the state. This policy shall include the following objectives:*
>
> > *(a) To maintain sufficient populations of all species of wildlife and the habitat necessary to achieve the objectives stated in subdivisions (b), (c), and (d).*
> >
> > *(b) To provide for the beneficial use and enjoyment of wildlife by all citizens of the state.*

> *(c) To perpetuate all species of wildlife for their intrinsic and ecological values, as well as for their direct benefits to all persons.*
> *(d) To provide for aesthetic, educational, and nonappropriative uses of the various wildlife species."*

Section 1802 provides jurisdiction of California Department of Fish and Game:

> *"1802. The department has jurisdiction over the conservation, protection, and management of fish, wildlife, native plants, and habitat necessary for biologically sustainable populations of those species. The department, as trustee for fish and wildlife resources, shall consult with lead and responsible agencies and shall provide, as available, the requisite biological expertise to review and comment upon environmental documents and impacts arising from project activities, as those terms are used in the California Environmental Protection Act (Division 13 commencing with Section 21000) of the Public Resources Code)."*

Sage-grouse are classified as resident upland game birds under Part 2, Chapter 1, Section 3500.

> *"3500. Resident game birds are: Chinese spotted doves, ringed turtle doves of the family Columbidae, California quail and varieties thereof, Gambel or desert quail, mountain quail and varieties thereof, sooty or blue grouse and varieties thereof, ruffed grouse, sage hens and sage grouse, Hungarian partridges, red-legged partridges including the chukar and other varieties, ring-necked pheasants and varieties, and wild turkeys of the order Galliformes.*
> *Migratory game birds are: ducks and geese, coots and gallinules, jacksnipe, western mourning doves, white-winged doves and band-tailed pigeons.*
> *References in this code to 'game birds' means both resident game birds and migratory game birds."*

Colorado

Colorado thorough their Division of Wildlife (CDOW) has responsibility for the management and conservation of wildlife resources, including the conservation and management of threatened and endangered species, within their borders as defined and directed by state laws (i.e. Colorado Revised Statutes, Title 33 Article 1). Title 33 Article 1-101, Legislative declaration states:

> *"It is the policy of the State of Colorado that the wildlife and their environment are to be protected, preserved, enhanced and managed for the use, benefit, and enjoyment of the people of this state and its visitors. It is further declared to be the policy of this state that there shall be provided a comprehensive program designed to offer the greatest possible variety of wildlife-related recreational opportunity to*

> *the people of this state and its visitors and that, to carry out such program and policy, there shall be a continuous operation of planning, acquisition, and development of wildlife habitats and facilities for wildlife-related opportunities."*

Idaho

Title 36 of Idaho state code defines the authority for the management and protection of wildlife in the state of Idaho. Specifically, Title 36, Chapter 1, 36-102 defines the role of the Idaho Fish and Game Commission. This section follows:

> *"36-102 Idaho Fish and Game Commission. (a) Creation. There is hereby created the Idaho fish and game commission. The department of fish and game of the state of Idaho is hereby placed under the supervision, management and control of said Idaho fish and game commission, hereinafter referred to as the commission or as said commission."*

The department of fish and game is given broad authority for wildlife protection under Title 36, Chapter 1, 36-103 follows:

> *"36-103. Wildlife Property of State Preservation.*
>
> > *(a) Wildlife Policy. All wildlife, including all wild animals, wild birds, and fish, within the state of Idaho, is hereby declared to be the property of the state of Idaho. It shall be preserved, protected, perpetuated, and managed. It shall be only captured or taken at such times or places, under such conditions, or by such means, or in such manner, as will preserve, protect, and perpetuate such wildlife, and provide for the citizens of this state and, as by law permitted to others, continued supplies of such wildlife for hunting, fishing and trapping.*
> >
> > *(b) Commission to Administer Policy. Because conditions are changing and in changing affect the preservation, protection, and perpetuation of Idaho wildlife, the methods and means of administering and carrying out the state's policy must be flexible and dependent on the ascertainment of facts which from time to time exist and fix the needs for regulation and control of fishing, hunting, trapping, and other activity relating to wildlife, and because it is inconvenient and impractical for the legislature of the state of Idaho to administer such policy, it shall be the authority, power and duty of the fish and game commission to administer and carry out the policy of the state in accordance with the provisions of the Idaho fish and game code. The commission is not authorized to change such policy but only to administer it."*

Montana

Montana manages and protects greater sage-grouse under the statutory authority of Title 87 of Montana Code Annotated 2003. Montana Department of Fish Wildlife and Parks' authority is described in part by the following:

> *"MCA 87-1-201. (Temporary, until March 2006) Powers and duties.*
>
> *(1) The department shall supervise all the wildlife, fish, game, game and nongame birds, waterfowl, and the game and fur-bearing animals of the state and may implement voluntary programs that encourage hunting access on private lands and that promote harmonious relations between landowners and the hunting public. It possesses all powers necessary to fulfill the duties prescribed by law and to bring actions in the proper courts of this state for the enforcement of the fish and game laws and the rules adopted by the department.*
>
> *(2) The department shall enforce all the laws of the state respecting the protection, preservation, management, and propagation of fish, game, fur-bearing animals, and game and nongame birds within the state."*

The Montana Fish, Wildlife and Parks Commission provide policy for the Department in the matters of wildlife management as set forth in the following MCA:

> *"MCA 87-1-301. Powers of commission. (1) The commission:*
>
> *(a) shall set the policies for the protection, preservation, management, and propagation of the wildlife, fish, game, furbearers, waterfowl, nongame species, and endangered species of the state and for the fulfillment of all other responsibilities of the department as provided by law;*
>
> *87-2-101. Sage-grouse are classified as upland game birds by statute.*
>
> *87-1-102, chapter 3, and this chapter, unless the context clearly indicates otherwise, the following definitions apply:*
>
> *(15) 'Upland game birds' means sharptailed grouse, blue grouse, spruce (Franklin) grouse, prairie chicken, sage hen or sage grouse, ruffed grouse, ring-necked pheasant, Hungarian partridge, ptarmigan, wild turkey, and chukar partridge."*

Nevada

Nevada manages greater sage-grouse under statutory authority of the Nevada Revised Statutes (NRS) and Nevada Administrative Code (NAC). Statute and code identify the Nevada State

Board of Wildlife Commissioners with the establishment of broad policy for the management and protection of the State's wildlife. Nevada Department of Wildlife is the agency charged with the execution of State law, Commission regulation and policy. Significant law and regulations are:

> *"NRS 501.110 Classification of wildlife.*
>
> *1. For the purposes of this title, wildlife must be classified as follows:*
>
> *. . . Wild birds, which must be further classified as either game birds, protected birds or unprotected birds. Game birds must be further classified as upland game birds or migratory game birds.*
>
> *NRS 501.181 Duties; regulations. The Commission shall:*
>
> *1. Establish broad policies for:*
>
> *(a) The protection, propagation, restoration, transplanting, introduction and management of wildlife in this state*
>
> *Nevada Administrative Code 503.040 Wild birds. Wild birds include all species classified as game, protected and unprotected birds.*
>
> *1. Upland game birds, which include: Centrocercus urophasianus."*

North Dakota

North Dakota manages and protects greater sage-grouse through Title 21.1 of state statutes. The Game and Fish Department is authorized under the following laws:

> *"20.1-01-03. Ownership and control of wildlife is in the state - Damages Schedule of monetary values - Civil penalty. The ownership of and title to all wildlife within this state is in the state for the purpose of regulating the enjoyment, use, possession, disposition, and conservation thereof, and for maintaining action for damages as herein provided. Any person catching, killing, taking, trapping, or possessing any wildlife protected by law at any time or in any manner is deemed to have consented that the title thereto remains in this state for the purpose of regulating the taking, use, possession, and disposition thereof. The state, through the office of attorney general, may institute and maintain any action for damages against any person who unlawfully causes, or has caused within this state, the death, destruction, or injury of wildlife, except as may be authorized by law. The state has a property interest in all protected wildlife.*
>
> *This interest supports a civil action for damages for the unlawful destruction of wildlife by willful or grossly negligent act or omission. The director shall adopt by rule a schedule of monetary values of various species of wildlife, the values to represent the replacement costs of the wildlife and the value lost to the state due to the destruction or injury of the species, together with other material elements of value. In any action brought under this section, the schedule constitutes the measure*

> *of recovery for the wildlife killed or destroyed. Notwithstanding the director's schedule of monetary values, an individual who unlawfully takes a bighorn sheep, elk, or moose is subject to a civil penalty for the replacement value of the animal of five thousand dollars for a bighorn sheep, three thousand dollars for an elk, and two thousand dollars for a moose. For a male bighorn sheep, elk, or moose over two and one-half years of age, the civil penalty for the replacement value of the animal is an additional fifty percent of the penalty. The funds recovered must be deposited in the general fund, and devoted to the propagation and protection of desirable species of wildlife.*
>
> *20.1-01-02. Definitions. In this title, unless the context otherwise requires:*
>
> > *15. 'Game birds' includes all varieties of geese, brant, swans, ducks, plovers, snipes, woodcocks, grouse, sagehens, pheasants, Hungarian partridges, quails, partridges, cranes, rails, coots, wild turkeys, mourning doves, and crows.*
>
> *20.1-04-02. Game birds protected. No person may hunt, take, kill, possess, convey, ship, or cause to be shipped, by common or private carrier, sell, or barter any game bird or any part thereof taken in this state, except as provided in this title."*

Oregon

Oregon manages greater sage-grouse through policy developed by the State Fish and Wildlife Commission and executed by the Department of Fish and Wildlife. Oregon Revised Statutes (ORS) Chapter 496 delineates the laws that relate to fish and wildlife management and protection. ORS 496.138 establishes the State Fish and Wildlife Commission a budget, policy and program body. ORS 496.012, Wildlife policy directs the main mission of both the Commission and the Department in matters relating to Oregon's wildlife. ORS 496.118 establishes that the director of the Department carry out the policies of the Commission and the wildlife laws of the State of Oregon. ORS 496.007 establishes sage-grouse as a game bird in Oregon. Annotated subject statutes follow below:

> *"Chapter 496 Application, Administration and Enforcement of Wildlife Laws*
>
> *496.138 General duties and powers; rulemaking authority; hearing prior to budget request to Governor. (1) Consistent with the policy of ORS 496.012, the State Fish and Wildlife Commission shall implement the policies and programs of this state for the management of wildlife. These policies and programs shall consider the uses of public and private lands and utilize voluntary partnerships with private and public landowners to protect and enhance wildlife habitat and effectively manage wildlife. In addition, the commission shall perform any other duty vested in it by law.*

496.012 Wildlife policy. It is the policy of the State of Oregon that wildlife shall be managed to prevent serious depletion of any indigenous species and to provide the optimum recreational and aesthetic benefits for present and future generations of the citizens of this state. In furtherance of this policy, the State Fish and Wildlife Commission shall represent the public interest of the State of Oregon and implement the following coequal goals of wildlife management:

> *(1) To maintain all species of wildlife at optimum levels.*
> *(2) To develop and manage the lands and waters of this state in a manner that will enhance the production and public enjoyment of wildlife.*
> *(3) To permit an orderly and equitable utilization of available wildlife.*
> *(4) To develop and maintain public access to the lands and waters of the state and the wildlife resources thereon.*
> *(5) To regulate wildlife populations and the public enjoyment of wildlife in a manner that is compatible with primary uses of the lands and waters of the state.*
> *(6) To provide optimum recreational benefits.*
> *(7) To make decisions that affect wildlife resources of the state for the benefit of the wildlife resources and to make decisions that allow for the best social, economic and recreational utilization of wildlife resources by all user groups. [1973 c.723 6; 1993 c.659 2; 2001 c.762]*

496.118 Duties and powers of director. (1) Subject to policy direction by the State Fish and Wildlife Commission, the State Fish and Wildlife Director shall:

> *(a) Be the administrative head of the State Department of Fish and Wildlife;*
> *(b) Have power, within applicable budgetary limitations, and in accordance with ORS chapter 240, to hire, assign, reassign and coordinate personnel of the department;*
> *(c) Administer and enforce the wildlife laws of the state*

496.007 "Game bird" defined. As used in the wildlife laws, unless the context requires otherwise, "game bird" means:

> *(1) Those members of the family Anatidae, commonly known as swans, geese, brant and river and sea ducks.*
> *(2) Those members of the family Columbidae, commonly known as mourning doves and bandtailed pigeons.*
> *(3) Those members of the family Tetranidae, commonly known as grouse, ptarmigan and prairie chickens"*

South Dakota

South Dakota Game, Fish and Parks Department defines greater sage-grouse under the authority of the following statute:

> *"41-1-1. Definition of terms. Terms used in this title mean:*
>
> > *(24) 'Small game,' anatidae, commonly known as swans, geese, brants, merganser, and river and sea ducks; the rallidae, commonly known as rails, coots, and gallinue; the limicolae, referring specifically to shore birds, plover, snipe, and woodcock; the gruidae, commonly known as sandhill crane; the columbidae, commonly known as the mourning dove; the gallinae, commonly known as grouse, prairie chickens, pheasants, partridges, and quail but does not include wild turkeys; cottontail rabbit; and fox, grey and red squirrel. The term includes facsimiles of small game used for law enforcement purposes*
>
> *41-1-2. Game birds, animals, and fish as property of state. No person shall at any time or in any manner acquire any property in, or subject to his dominion or control, any game bird, game animal, or game fish, or any part thereof, but they shall always and under all circumstances be and remain the property of the state, except as provided by 41-1-3.*
>
> *41-2-18. Rules for implementation of game, fish and conservation laws. The Game, Fish and Parks Commission may adopt such rules as may be necessary to implement the provisions of chapters 41-1 to 41-15, inclusive. The rules may be adopted to regulate:*
>
> > *(1) The conservation, protection, importation, and propagation of wild animals and fish except for any nondomestic animal which is regulated pursuant to 40-3-26;*
>
> *41-3-1. Department in charge of propagation and preservation of game and fish. The Department of Game, Fish and Parks shall have charge of the propagation and preservation of such varieties of game and fish as it shall deem to be of public value.*
>
> *41-3-2. Collection and publication of conservation information. The Department of Game, Fish and Parks shall have charge of the collection and diffusion of such statistics and information as shall be germane to the purpose of conservation.*
>
> *41-2-23. Improvement of wildlife habitat -- Access lands -- State title not required. The Department of Game, Fish and Parks shall have the power and duty, when directed by the Game, Fish and Parks Commission, to expend funds for the*

> *improvement of wildlife habitat, access to hunting, and access to fishing or recreation areas on any land, public or private within the state, notwithstanding the provisions of 5-14-10, provided, however, that any land so improved shall be open to reasonable use by the public."*

Utah

Title 23 of the Utah Code is the Wildlife Resources Code of Utah and provides the Utah Division of Wildlife Resources (UDWR) the powers, duties, rights, and responsibilities to protect, propagate, manage, conserve, and distribute wildlife throughout the state. Section 23-13-3 declares that wildlife existing within the state, not held by private ownership and legally acquired, is property of the state. Sections 23-14-18 and 23-14-19 authorize the Utah Wildlife Board to prescribe rules and regulations for the taking and/or possession of protected wildlife.

Washington

The Washington State Department of Fish and Wildlife is directed by the Fish and Wildlife Code of the State of Washington:

> *"RCW 77.04.012 Mandate of department and commission. Wildlife, fish, and shellfish are the property of the state. The commission, director, and the department shall preserve, protect, perpetuate, and manage the wildlife and food fish, game fish, and shellfish in state waters and offshore waters.*
> *The department shall conserve the wildlife and food fish, game fish, and shellfish resources in a manner that does not impair the resource."*

Wyoming

Wyoming maintains statutory authority to manage and protect sage-grouse through Title 23 of Wyoming state law and policies and direction from the Wyoming Game and Fish Commission. The major enabling Wyoming statutes include the duties and authority of the Commission delineated in 23-1-302:

> *"23-1-302. Powers and duties.*
> > *(a) The commission is directed and empowered:*
> > > *(i) To fix season and bag limits, open, shorten or close seasons on any species or sex of wildlife for any type of legal weapon, except predatory animals, predacious birds, protected animals, and protected birds, in any specified locality of Wyoming, and to give notice thereof;*
> > > *(iii) To acquire lands and waters in the name of Wyoming by purchase, lease, agreement, gift or devise, not including powers of*

> *eminent domain, and to develop, improve, operate, and maintain the same for the following purposes:*
>
> > *(B) Management of game animals, protected animals and birds, furbearing animals, game birds, fish, and their restoration, propagation, or protection"*

Sage-grouse Petitions

Greater sage-grouse have been petitioned for protection under the Endangered Species Act of 1973 in a total of seven petitions. The first petition, dated May 14, 1999 was directed at the Washington state population and asserted that population was a distinct population segment. The last petition on record was dated December 22, 2003 and was directed at all greater sage-grouse (Table 2.1).

The Washington population of the Western Sage-grouse petition was submitted to the U.S. Fish and Wildlife Service on May 19, 1999. The petition was submitted by Northwest Ecosystem Alliance and Biodiversity Legal Foundation. The petitioners sought a listing as threatened or endangered for the Columbia Basin distinct population segment of greater sage-grouse. The U.S. Fish and Wildlife Service made a positive 90-day finding on August 24, 2000 (U.S. Fish and Wildlife Service, 2000) and a 12-month finding on May 7, 2001 (U.S. Fish and Wildlife Service, 2001). The outcome of the 12-month finding was that the petition presented substantial information and that listing was warranted but precluded because of higher priority listing actions. The U.S. Fish and Wildlife Service classified the Columbia Basin Distinct Population Segment as a candidate species priority number 9 under U.S. Fish and Wildlife Service policies. No legal actions are pending on this petition.

A petition entitled Mono Basin population of the greater sage-grouse was submitted to the U.S. Fish and Wildlife Service on December 28, 2001. The petitioner was the Institute for Wildlife Protection. This petition sought an emergency listing of this population as endangered. An initial evaluation of greater sage-grouse in the Mono Basin indicated that an emergency listing was not warranted. The U.S. Fish and Wildlife Service published a 90-day finding on December 26, 2002 (U.S. Fish and Wildlife Service 2002). The U.S. Fish and Wildlife Service determined that information provided in the petition was not substantial. A complaint was filed on July 3, 2002 by the Institute for Wildlife Protection. The U.S. District Court found in favor of the U.S. Fish and Wildlife Service on December 1, 2003 and dismissed the plaintiff's case. Institute for Wildlife Protection filed another complaint on January 9, 2003 seeking relief from the U.S. Fish and Wildlife Service's 90-day finding. This action is still pending.

A petition, Western subspecies of the greater sage-grouse, was filed on January 24, 2002 by the Institute for Wildlife Protection. The petition requested that the subspecies be listed. The U.S. Fish and Wildlife Service produced a 90-day finding on February 7, 2003 (U.S. Fish and Wildlife

Service 2003) and found that information contained in the petition was not substantial. The Institute for Wildlife Protection filed a Notice of Intent on February 7, 2003 regarding the 90-day finding. A complaint was filed on June 6, 2003 by the Institute for Wildlife Protection seeking Court relief for the findings of the 90-day determination. These actions are still pending.

A petition to list greater sage-grouse was filed with the U.S. Fish and Wildlife Service on June 18, 2002 by Craig Dremann. This petition requested listing greater sage-grouse as endangered. The USFWS made a positive 90-day finding on April 5, 2004 (U.S. Fish and Wildlife Service 2004a). No legal actions have been taken on this petition.

A petition, Eastern subspecies of the greater sage-grouse, was filed on July 3, 2002 by Institute for Wildlife Protection. The petition requested that the Eastern subspecies of greater sage-grouse be listed as endangered. The U.S. Fish and Wildlife Service was ordered to make a 90-day finding October 3, 2003. That finding, issued on January 7, 2004 (U.S. Fish and Wildlife Service 2004) was that the information presented in the petition was not substantial.

The Institute for Wildlife Protection combined the previously submitted Western subspecies of the greater sage-grouse and the Eastern subspecies of the greater sage-grouse on March 19, 2003. This combined petition requests that the U.S. Fish and Wildlife Service list greater sage-grouse as endangered. The USFWS made a positive 90-day finding on April 5, 2004 (U.S. Fish and Wildlife Service 2004a). No legal actions have been taken on this petition.

On December 22, 2003, a petition was filed with the U.S. Fish and Wildlife Service requesting greater sage-grouse be listed as threatened or endangered. This petition was submitted by a coalition including, American Lands Alliance, Biodiversity Conservation Alliance, Center for Biological Diversity, Forest Guardians, The Fund for Animals, Gallatin Wildlife Association, Great Old Broads for Wilderness, Hells Canyon Preservation Council, The Larch Company, Northwest Ecosystem Alliance, Northwest Council for Alternatives to Pesticides, Oregon Natural Desert Association, Oregon Natural Resources Council, Predator Defense Institute, Sierra Club, Sinapu, Western Fire Ecology Center, Western Watersheds Project, Wild Utah Project and Wildlands CPR. The USFWS made a positive 90-day finding on April 5, 2004 (U.S. Fish and Wildlife Service 2004a). No legal actions have been taken on this petition.

Conservation Plans

In 1996, the Western Association of Fish and Wildlife Agencies developed their first Memorandum of Understanding between their members regarding sage-grouse conservation. An element of that MOU suggested that each state begin local area conservation planning groups to address sage-grouse conservation issues at population levels. Local working groups (LWG) were organized on the San Juan in Utah in 1997 and Parker Mountain in Utah, Curlew, Upper Snake and

Owyhee in Idaho in 1998. By 2000, the number of Local Working Groups focusing on greater sage-grouse numbered 12 with 44 groups organized by 2004.

Coordination and Standards

Greater sage-grouse conservation efforts have been driven by the formation of partnerships between wildlife agencies and Indian tribes, charged with managing the species, private landowners, Indian tribes and public landowners, who manage the habitat and interested stakeholders. These conservation efforts have taken place at the range-wide scale, country scale, state and provincial scale, local area and project scales. Conservation efforts are directed at evaluating populations and their habitat, determining risks or vulnerabilities to the species or habitat, plotting a course of action to meet objectives and carrying out those actions. Planning efforts have monitoring and adaptive management components for evaluating the effectiveness of the conservation actions.

Many greater sage-grouse populations have distributions that span one or more jurisdictional boundaries (Chapter 6). Effective management of these populations requires coordination between the various landowner, wildlife managers and the public. The WAFWA, U.S. Bureau of Land Management, USFS and USFWS memorandum of understanding (Appendix 1) directs these agencies to form the National Sage-grouse Conservation Planning Framework Team (Framework Team). The Framework Team is charged with the facilitation and coordination of conservation efforts between the various jurisdictional units across the range of the species. This coordination insures that appropriate management strategies are applied to a population that shares multiple management authorities.

WAFWA recognizes that conservation plans and conservation actions should meet standards for evaluation. The USFWS and National Oceanic and Atmospheric Administration (NOAA) developed the Policy for the Evaluation of Conservation Efforts (PECE) (Appendix 2) in 2003. Each planning group is aware of the evaluation criteria and many are developing plans that have components of PECE.

U. S. Bureau of Land Management

In July 2003, the U.S. Bureau of Land Management (BLM) Director issued a memorandum regarding the development of a BLM Sage-grouse Habitat Conservation Strategy (U.S.D.I. Bureau of Land Management 2003) and interim program guidance on sage-grouse habitat conservation (Director's Office Instruction Memorandum No. 2003-003). The memorandum directs that each Washington Office Group *"immediately review and evaluate program policies that potentially impact or threaten long-term health of sage-grouse populations and their habitat on BLM land."* The memorandum also states that the BLM would issue interim guidance *"focused on actions that can be taken immediately to minimize or eliminate threats to sage-grouse and their habitat and that do not require NEPA (National Environmental Policy Act) review before implementation."*

Also in July 2003, the BLM released for public comment its draft National Sage-grouse Habitat Conservation Strategy. The strategy was developed by an interdisciplinary team comprised of senior agency staff and managers representing all affected program areas and administrative levels. The stated intent of the BLM strategy is to *"serve as a framework to address the conservation of sage-grouse habitats on BLM-managed land,"* with a vision *"to manage public land in a manner that will maintain, enhance, and restore sage-grouse habitats while providing for multiple uses of BLM-administered public land."* The draft presents five goals, with a total of forty-one (41) action items. The goals are to:

1. Develop a consistent and effective management framework for addressing conservation needs of sage-grouse on public lands.

2. Increase our understanding of resource conditions and priorities for maintaining restoring habitat.

3. Expand available research and information that supports effective management of sage-grouse habitat.

4. Develop partnerships to enhance effective management of sage-grouse habitats.

5. Ensure leadership and resources are adequate to implement national and state-level sage-grouse habitat conservation strategies.

Based on comments received, the BLM Director decided to postpone finalizing the BLM strategy until after this greater sage-grouse conservation assessment had been completed.

In addition to the foregoing documents, the BLM has issued formal directives related to sage-grouse habitat mapping, (and) participation in State-led sage-grouse conservation planning (IM 2004-136), and gathering data about BLM *"activities and management requirements that provide benefits or offer protection to greater sage-grouse and its habitat"* (IM 2004-180).

Alberta

The greater sage-grouse is federally and provincially listed as endangered in Canada. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) listed sage-grouse in Canada as threatened in 1997 and, after further review, changed the status to endangered in 1998. In Alberta, sage-grouse were included on the 'blue list' of species that may be at risk. The Alberta Endangered Species Conservation Committee recommended that sage-grouse be included on the list of endangered animals under the Alberta Wildlife Act, and that listing occurred early in 2000. The provinces of Saskatchewan and Alberta jointly formed a Sage-grouse Recovery Team outside of the

Recovery of Nationally Endangered Wildlife Committee (RENEW) process in November 1997. The Sage-grouse Recovery Team is composed of representatives from government (provincial and federal), land managers, landowners, conservation organizations and industry from Saskatchewan and Alberta. The team developed the Canadian Sage-grouse Recovery Strategy (Canadian Sage-grouse Recovery Team 2001) and published the strategy in June 2001. The strategy calls for the formation of Local Working Groups to implement strategies directed in the plan. In practice sage-grouse conservation efforts have been undertaken by partnerships.

California

California Department of Fish and Game, BLM, U.S. Forest Service, other agencies and key private stakeholders have begun work on a statewide conservation plan. This plan is expected to be completed in December 2004. California Department of Fish and Game, U.S. Bureau of Land Management and US Forest Service have participated with Nevada Department of Wildlife and Nevada's Governor's sage-grouse team in developing local planning efforts for populations that the two states share.

California has a total of four Local Working Groups. Two of the groups border Nevada and one group is within California. The Local Working Groups that border Nevada are effectively directed by the frameworks outlined in the Nevada plan. These groups are writing plans that can be evaluated by Policy for the Evaluation of Conservation Efforts PECE (U.S. Fish and Wildlife Service and National Oceanic and Atmospheric Administration, 2003) Appendix 2.

Local Working Groups are open to participation from any agency, non-governmental organization or citizen. The planning groups are staffed by agency personnel and the sessions are facilitated by the University of Nevada, Cooperative Extension or other professional facilitators.

Some planning conservation implementation has begun ahead of the actual completion of the plans. These implementation actions are generally related to statewide objectives such as hunting season conformance to the sage-grouse guidelines (Connelly et al. 2000) and fire suppression strategies to protect sage-grouse habitats.

Colorado

Colorado began conservation planning at the local level with a focus on Gunnison sage-grouse in 1994. Because of the conservation challenges facing the Gunnison sage-grouse, Colorado has expended much of its efforts for that species. Conservation planning for greater sage-grouse began in 1996 with the establishment of the Moffat county Local Working Group. This group has been renamed the Northwestern Colorado Working Group. Three additional groups have formed and are scheduled to complete conservation plans in the summer 2004. All groups are scheduled to complete conservation plans in the summer 2004.

Colorado has developed a schedule for the development of a statewide greater sage-grouse conservation strategy and plan that incorporates LWG efforts. The planning process is expected to begin in July 2004 and finish one year later.

Idaho

Idaho developed a statewide conservation plan for sage-grouse in 1997. This plan is now in revision and is expected to be completed September 2004 and approved in December 2004. The revised plan provides strategic guidance for greater sage-grouse management in Idaho and includes the incorporation of six local working group conservation plans. The six planning units cover approximately 57% of the distribution of greater sage-grouse in Idaho. Idaho has identified 13 total greater sage-grouse management units. All management units will have Local Working Groups over time. The comprehensive Idaho plan should have components of PECE for evaluation.

Idaho's first local working group, Shoshone Basin, formed in 1994 and has a finished conservation plan. The Owyhee LWG formed in 1998 and has also completed its plan. Three of the other groups are in final draft and one group is in an early draft stage. Idaho Local Working Groups are self-directed and staffed with agency personnel.

Idaho has begun a number of local conservation actions as suggested by the Local Working Groups. Further, on a statewide basis, Idaho is implementing conservation actions. These statewide actions include following the sage-grouse guidelines for hunting and protection of habitats from wildfire.

Montana

Montana completed its draft statewide conservation plan, *Management and Conservation Strategies for Sage Grouse in Montana*, (Montana Sage-grouse Work Group 2003) in the summer of 2003. The plan has been reviewed and was approved in the spring 2004. The effort is the result of more than two years of research and deliberation by the Montana Sage-grouse Work Group, which included a wide and diverse spectrum of Montanans.

The goal is described, "To provide for the long-term conservation and enhancement of the sagebrush steppe/mixed-grass prairie complex within Montana in a manner that supports sage-grouse and a healthy diversity and abundance of wildlife species and human uses." The Montana plan directs conservation efforts be implemented on both the statewide and local level.

The statewide working group used the best available information, to develop a plan that describes the current status of Montana's sage-grouse population and sagebrush habitat, describes the desired conditions for habitat, and identifies risks confronting habitat and sage-grouse populations. Specifically the plan addresses the following:

"It responds to concerns about the loss of sagebrush habitat and declines in sage grouse numbers in the state.

It includes conservation objectives for both sagebrush habitat and sage grouse populations.
It provides guidelines and tools for assessing different habitats to obtain standardized results which will be useful in protecting, improving, and restoring habitat.
It explains the roles of the federal, state, and tribal agencies involved in sagebrush and sage grouse management.
It provides a framework for establishing local groups of diverse stakeholders to adapt the plan to their respective geographical areas."

Local Working Groups and agencies responsible for sage-grouse conservation are the primary elements for carrying out provisions of the statewide sage-grouse conservation plan. The statewide plan identified a total of eleven planning group distributed throughout sage-grouse distribution in the state. The first three groups began deliberations in the winter of 2004. The groups are located in Dillon, Glasgow and Miles City. Within a year of the start of the first groups, an additional three or four groups will begin, and within a year of that, another three or four groups will commence. Due to the long-term nature of the plan, we anticipate that local groups will be active for 10-20 years. Within a year of a group's formation the group will implement some conservation actions. Within two years, local groups are expected to:

"Coordinate issue development with appropriate agencies.
Develop action steps to implement the plan.
Seek creative solutions.
Identify priority areas through issue development.
Have at least one project funded ...
Provide a list of measurable results with a timeline.
Provide a plan for monitoring results."

The Local Working Groups are self-directed within the frameworks developed by the statewide plan. The membership of the groups is open to any interested parties, but should include a balanced selection of local stakeholders. The sessions will be initially hosted by a professional facilitator.

Conservation actions are being implemented ahead of the formal adoption of the statewide plan and local area plans. The state is expending considerable effort to protect sagebrush habitats with long-term conservation easements. Montana's sage-grouse hunting seasons are the longest at 62 days. Montana Department of Fish, Wildlife and Parks, estimates that they are harvesting < 10% of their standing population and are within the sage-grouse management guidelines (Connelly et al. 2000).

Nevada

Nevada completed its statewide conservation strategy, entitled *The Nevada Sage-grouse Conservation Strategy in Nevada* in November 2001 (Nevada Governor's Team 2001). The planning effort was sponsored by the Governor's office through the Governor's Sage-grouse conservation team. This team was composed of a wide variety of stakeholders including members from industry, government, tribal interests, non-governmental organizations, academia and citizens. The Nevada Department of Wildlife was the lead agency. California Department of Fish and Game personnel participated for populations of grouse that ranged across the border of the two states. Group membership generally ranged from 25 - 30 individuals. The Governor's team was empaneled in August 2000.

The strategy as described by the plan is:

> *"The foundation of Nevada's plan of action lies in the creation of local planning groups charged with designing workable solutions to specific on-the-ground challenges in their respective areas. LWGs will consider alternatives and develop and implement strategies for natural resource management actions that will enhance and benefit Sage Grouse. Through this process, local planning groups will have a unique opportunity to create conservation strategies before regulatory actions limit options and flexibility."*

Goals that are included for the strategy include the following:

> *1. Create healthy, self-sustaining Sage Grouse populations well distributed throughout the species historic range by maintaining and restoring ecologically diverse, sustainable, and contiguous sagebrush ecosystems and by implementing scientifically-sound management practices.*
> *2. Throughout the Sage Grouse's range in Nevada, have locally functional, well informed groups empowered to actively contribute to Sage Grouse conservation while balancing habitat, bird, and economic considerations.*

In order to assist the Team in evaluating whether or not it is meeting these goals, the Team has further defined the following set of Desired Outcomes:

> *"1. Over the next 20 years, apply active management techniques designed to improve Sage Grouse habitat quality from non-suitable to suitable to an average of 250,000 acres per year statewide.*
> *2. Over the next 20 years, maintain or increase Sage Grouse management unit populations' statewide (currently delineated into 64 units).*

> *3. Over the next 20 years, maintain or increase Sage Grouse numbers statewide as indicated by assessment of 20-year population trend data.*
> *4. Nevada's local conservation plans will include positive, incentive-driven solutions. Such solutions will minimize adverse economic impacts while maximizing the likelihood of plan success.*
> *5. Nevada's local conservation plans will be models for collaborative planning that yield balanced solutions meeting the needs of Sage Grouse, Sage Grouse habitat and people."*

Nevada's strategy plan identified six Local Working Groups to plan for approximately 60 greater sage-grouse population management units. With the exception of the Elko Stewardship group that formed in 1999, these groups began their deliberations in January 2002. The University of Nevada, Cooperative Extension service was contracted to facilitate meetings. The groups were given a framework for plans that included templates for Conservation agreements, and PECE criteria. Staffing for the plan was supplied by various agencies.

The Local Working Groups were given a charge by the Governor's team to complete their Local Area Conservation plans by 2004. Population Management Unit (PMU) plans have been completed for 20 of the 64 PMU's delineated within the state and the remaining plan will continue to formulate in the future by the Local Working Groups. The first edition of the Sage-grouse Conservation Plan for Nevada and Portions of Eastern California, formulated from these plans, is scheduled to be completed by June 2004.

Some planning conservation implementation has begun ahead of the actual completion of the plans. These implementation actions are generally related to statewide objectives such as hunting season conformance to the sage-grouse guidelines (Connelly et al. 2000) and fire suppression strategies to protect sage-grouse habitats.

The Department of Wildlife has expended approximately $828,000 on conservation planning at the state and local level since the inception of this project in 2001.

North Dakota

North Dakota's greater sage-grouse are confined to a single population in the southwestern corner of the state. The state has not developed a statewide strategy document to guide conservation planning. North Dakota Game and Fish Department in cooperation with South Dakota Game, Fish and Parks Department and the National Sage-grouse Conservation Planning Framework team have joined together to develop a conservation agreement for greater sage-grouse in southwestern North Dakota. This plan is being written by an independent contractor with input from agencies, private landowners and other stakeholders. The plan is designed to be evaluated by PECE criteria. This plan is scheduled to be completed by late summer 2004.

Oregon

Oregon has not developed a statewide conservation plan for greater sage-grouse. The state formed a working group in 2000; however, little was accomplished until the winter of 2004. Oregon Department of Fish and Wildlife is now directing significant effort toward the completion of a statewide plan. The agency has committed approximately $231,000 during the current biennium for sage-grouse conservation planning. Additionally, the agency has a sage-grouse conservation planner on staff. The sage-grouse working group has been reconvened and a plan is expected by summer 2005.

Greater sage-grouse conservation actions are taking place ahead of completion of either a strategic or tactical plan. Oregon has maintained a very conservative hunting season for a number of years. They prescribe harvest levels < 5% of the standing population and hunt populations that fall within the parameters outlined in the sage-grouse guidelines (Connelly et al. 2000).

Saskatchewan

The greater sage-grouse is federally and provincially listed as endangered in Canada. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) listed sage-grouse in Canada as threatened in 1997 and, after further review, changed the status to endangered in 1998. In Saskatchewan, the species was listed as threatened in 1987 and in 1999, was listed as endangered under Saskatchewan's Species at Risk revisions of The Wildlife Act. The provinces of Saskatchewan and Alberta jointly formed a Sage-grouse Recovery Team outside of the Recovery of Nationally Endangered Wildlife Committee (RENEW) process in November 1997. The Sage-grouse Recovery Team is composed of representatives from government (provincial and federal), land managers, landowners, conservation organizations and industry from Saskatchewan and Alberta. The team developed the Canadian Sage-grouse Recovery Strategy and published the strategy in June 2001. The strategy calls for the formation of Local Working Groups to implement strategies directed in the plan. In practice sage-grouse conservation efforts have been undertaken by partnerships (S. McAdams, Personal Communication).

South Dakota

South Dakota's greater sage-grouse are confined to two populations, one in northwestern South Dakota and one in the southwestern corner of the state. The state has not developed a statewide strategy document to guide conservation planning. South Dakota Game, Fish and Parks Department and North Dakota Game and Fish Department in cooperation with the National Sage-grouse Conservation Planning Framework team have joined together to develop a conservation agreement for greater sage-grouse in South Dakota. This plan is being written by an independent contractor with input from agencies, private landowners and other stakeholders. The plan is designed to be evaluated by PECE criteria. This plan is scheduled to be completed by late summer 2004.

Utah

Utah completed its statewide conservation strategy, entitled *Strategic Management Plan for Sage-grouse* (Utah Division of Wildlife Resources 2002) in June 2002. The plan is designed to address management plans for both species of sage-grouse. The plan was completed by the Utah Sage-grouse Working Group comprised of representatives from state and federal natural resource agencies concerned with the health and proper management of sage-grouse and the sagebrush-steppe ecosystem in Utah.

Utah's sage-grouse management plan identifies its objective as follows: "This plan is designed as a framework for Local Working Groups to develop area-specific management programs to maintain, improve and restore local sage-grouse populations and their habitat. Management areas, key local issues, conservation strategies and population information are provided as a starting point for Local Working Groups."

The goal of the plan is stated to: "Protect, enhance, and conserve sage-grouse populations and sagebrush-steppe ecosystems. Establish populations of sage-grouse in areas where they were historically found and the current sagebrush-steppe habitat is capable of maintaining a viable population of sage-grouse."

The state strategy plan identified thirteen planning units. Four Local Working Groups were formed before the development of the state strategy plan. One group, the San Juan was formed to deal with Gunnison sage-grouse and the others, Parker Mountain/John's Valley, Box Elder and Color Country were formed to address conservation issues for greater sage-grouse. San Juan and Parker Mountain/John's Valley were formed in 1997 and 1998 respectively. Box Elder and Color Country Local Working Groups were established in 2001. The Rich/Summit LWG was established in 2002, the Western Desert, Southwest Desert and Strawberry Local Working Groups were established in the spring of 2003 and three units, the Book Cliff/Unitah Basin, North Slope/Daggett and South Slope/Unitah Basin are addressed by a single local working group established in the fall of 2003. The two remaining planning units have not formed Local Working Groups.

Utah's Local Working Groups are self-directed within the framework established by the strategy and organized and facilitated by Utah State University Community-based Cooperative Extension Specialists. The goal of the groups is: "To assist in the development of sage-grouse management efforts that achieves local population and community goals." The membership of the groups includes a leadership group comprised of at least one representative from an agricultural group, one from a federal or state land management group, one from the Wildlife department and one from a wildlife conservation group. General membership is open to anyone that has an interest in sage-grouse management.

The *Utah Sage-grouse Strategic Management Plan* directs the Local Working Groups to use the strategic plan to develop their own local strategic plan within one year of the group formation. The local strategic plan is reviewed by the State Working group and incorporated into the state strategic plan. The local working group then meets twice a year to review progress, address new issues and modify actions as necessary. To date, only the San Juan plan for Gunnison sage-grouse has been completed. All other groups that are meeting are in various stages of plan drafting.

Utah is implementing a number of sage-grouse conservation efforts throughout the state ahead of the completion of the local plans and incorporation in the state strategic plan. Conservation efforts in Utah include hunting seasons that follow the sage-grouse management guidelines, sagebrush protection measures and rangeland improvements and population augmentation efforts.

Washington

Conservation efforts for the greater sage-grouse in Washington have taken a significantly different path than other western states. Greater sage-grouse disappeared from some parts of their range in Washington as early as 1860 (Tirhi 1995). Washington Department of Fish and Wildlife developed a species management plan in 1995 entitled *Washington State Management Plan for Sage-grouse* (Stinson et al. 2004). The sage-grouse was listed by the state of Washington as a threatened species in 1998. In May 2001, the Washington population of the sage-grouse also became a Candidate for listing under the federal Endangered Species Act when the U.S. Fish and Wildlife Service found that listing as Threatened was warranted but precluded by higher priority listing activities (U.S. Fish and Wildlife Service, 2001). Washington developed a Recovery Plan that summarizes the status of sage-grouse in Washington and outlines strategies to increase their population size and distribution to ensure the existence of a viable population of the species in the state.

The Recovery Plan identifies a goal statement as follows: "The goal of the sage-grouse recovery program is to establish a viable population of sage-grouse in a substantial portion of the species' historic range in Washington." The Plan further identifies population objectives related to down-listing or up-listing the species from threatened status to unlisted or from threatened status to endangered status. The performance criteria for down-listing the species includes a sustained (>10 year) breeding population of >3,200 birds and active breeding populations in six or more management units. The up-listing criteria include a population level dropping to <650 birds and a downward population trajectory (Stinson et al. 2003).

Washington has a primary partnership with fifteen agencies and Indian tribes for plan implementation. The plan has identified eleven primary recovery task categories. Within those categories the plan identified 44 species recovery tasks. The implementation schedule provides an annual cost estimate, priority rating and responsibility for each task. Washington is implementing many of the recovery tasks ahead of the final approval of the Recovery Plan.

Wyoming

Wyoming began their statewide sage-grouse conservation plan in July 2000 with the formation of the Wyoming Sage-grouse Working Group. The working group was comprised of eighteen individuals from Wyoming that represented a diverse array of backgrounds, interests and geographical residence. Although the group membership was restricted, the public was invited to attend meetings and provide input. A plan draft was submitted to the Wyoming Game and Fish Commission in July 2002. Public and agency review was directed back to the working group and the final plan *Wyoming Greater-Sage Grouse Conservation Plan* (Wyoming Sage-grouse Working Group 2002) was submitted to the Commission in May 2003 and the plan was approved by the Game and Fish Commission in June 2003.

The statewide plan identified a series of fourteen guiding goals and principles. These goals are listed below in no priority order:

- *"increase the present abundance and distribution of sage-grouse in Wyoming*
- *halt sage-grouse population declines in Wyoming*
- *determine the primary causes of sage-grouse declines*
- *provide Recommended Management Practices aimed at productive and healthy sage-grouse populations*
- *promote management that results in diverse, productive, and healthy sagebrush habitats while recognizing that sagebrush habitats provide values for species other than sage-grouse*
- *promote public involvement in planning and decision-making*
- *provide a framework for the development and implementation of local sage-grouse conservation plans to address and rectify potential impacts*
- *maintain an atmosphere of cooperation, participation, and commitment among wildlife managers, landowners, land managers, other stakeholders and interested public in development and implementation of conservation actions*
- *respect individual views and values, and implement conservation actions in a cooperative manner that generates broad community support*
- *implement conservation actions in a manner that meets the needs of sage-grouse, and are least disruptive to a stable and diverse economic base in Wyoming*
- *recognize the need to continually update data and apply them to local situations*
- *monitoring and evaluation are an important part of this plan, and adjustments to the goals, objectives, and conservation actions will be made considering the best available data*
- *identify research needs where knowledge is lacking*

- *encourage long-term funding for collecting and analyzing data over a period of time adequate to make appropriate resource management decisions"*

The statewide plan stated its purpose in a series of seven bullets (presented in no priority order):

- *"establish the framework for local working groups to guide management efforts directed at halting long-term population declines*
- *maintain and improve sage-grouse habitats in Wyoming*
- *provide for coordinated management across jurisdictional or ownership boundaries*
- *develop the statewide support necessary to assure the survival of Wyoming's sage-grouse populations*
- *be dynamic and flexible enough to include new information and issues as well as results from current and future conservation efforts*
- *provide Wyoming-based management solutions to sage-grouse problems using Wyoming-based data and research to the extent practicable*
- *address the five listing factors as defined by the Endangered Species Act of 1973, as amended"*

The Wyoming statewide plan provides the framework for greater sage-grouse management in Wyoming. The plan identified eleven local planning units across the state that covers nearly every location where sage-grouse are distributed. The plan directs the Local Working Groups to develop site-specific implementation actions within the geographical boundary of the Local Working Group and the framework of the statewide plan.

The Local Working Groups are provided structure within the statewide plan. The membership of the group is representative with "equal number of knowledgeable individuals from four areas - agriculture, conservation, industry, and agencies, with single representatives from local government, tribes, public at large, etc." The framework suggests a limit of no more than 12 individuals in a group. Membership in the group is by appointment from the Wyoming Game and Fish Commission and members should live or work in the geographical area of the plan. The Local Working Groups are self-directed.

The statewide plan provides a schedule for group formation and milestones for the group. The plan calls for the establishment of three working groups in the winter of 2004, four groups in the winter of 2005 and the final four groups to form in the winter of 2006. Each group is expected to provide the following products within two years of group formation:

- *"identify and prioritize issues affecting sage-grouse in their area*
- *identify solutions to problems affecting sage-grouse in their area*
- *develop an action plan geared toward addressing these problems*
- *identify priority areas for implementation of conservation actions*
- *identify funding sources to implement conservation actions*

- *recommend to private, State or Federal land managers at least one project*
- *provide annual updates of progress to the Wyoming Game and Fish Commission and other affected agencies."*

Summary

Greater sage-grouse conservation planning efforts are being conducted on two primary scales. The strategy level planning has been scheduled in two of the 13 states or provinces, been completed in eight of 13 states or provinces, is being worked on in one state and will be included in local planning efforts in two states. All state and provincial sage-grouse strategy documents are scheduled to be completed by the summer 2005.

Western states and provinces are expected to have a total of more than 70 Local Working Groups in various phases of planning through implementation by winter of 2006. A total of 44 Local Working Groups are planning conservation actions in the spring 2004. Twenty-three Local Working Groups are scheduled to have completed conservation plans by the summer of 2004 (Figure 2.1). Montana local planning groups have not mapped their respective planning boundaries. Oregon has not established its frame of tactical planning. Range-wide coverage of conservation plans are expected by the winter 2008 (Table 2.2).

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Table 2.1. Summary of Sage-Grouse Petitions Submitted to the U.S. Fish and Wildlife Service (USFWS)[a]
(as of May 27, 2004)

Petition Date: May 14, 1999 (74 pages)	**Petition Date:** January 25, 2000 (254 pages)	**Petition Date:** December 28, 2001 (493 pages)
Species: Washington population of the Western Sage Grouse *Centrocercus urophasianus phaios*	**Species:** Gunnison Sage Grouse *Centrocercus minimus*	**Species:** Mono Basin population of the Greater Sage Grouse *Centrocercus urophasianus phaios*
Petition Request: List as threatened or Endangered	**Petition Request:** List as endangered or threatened, emergency listing, and designation of critical habitat	**Petition Request:** Emergency list as endangered
Petitioners: Northwest Ecosystem Alliance and Biodiversity Legal Foundation	**Petitioners:** Mark Salvo, American Lands Alliance, Dr. Randy Webb, Net Work Associates, Andy Kerr, The Larch Company, Jasper Carlton, Biodiversity Legal Foundation, Susan Ash, Wild Utah Forest Campaign, Rob Edwards, Sinapu	**Petitioners:** Donald Randy Webb, Institute for Wildlife Protection
USFWS Determination: Both a 90-day finding (August 24, 2000) and a 12-month finding (May 7, 2001) published in the Federal Register. Outcome was that the petition presents substantial information and listing is warranted but precluded for the Columbia Basin Distinct Population Segment (occurs in WA and northern OR); became a candidate by default under USFWS policy.	**USFWS Determination:** The species was designated as a candidate by USFWS prior to receipt of the petition. The Listing priority number was elevated in a May 4, 2004 Federal Register Notice of Review to a 2. However the Service does not believe that emergency listing is warranted at this time.	**USFWS Determination:** Initial review indicated that the situation does not warrant an emergency listing. A 90-day finding was initiated August 1, 2002. The 90-day finding was published in the Federal Register December 26, 2002 with an outcome that the information presented in the petition is not substantial.
Legal Action: No Notice Of Intent (NOI**) to date	**Legal Action:** Court complaint dated September 29, 2000 from the American Lands Alliance et al. In summer 2003 the Court rules in the USFWS's favor. The ruling is that USFWS candidate process and the determination by USFWS that a species should be on the candidate list is equivalent to a 12-month finding. On March 16, 2004 the plaintiffs file a lawsuit in Washington D.C. District Court asking that Court to order the USFWS to emergency list the species as endangered.	**Legal Action:** A court complaint dated July 3, 2002 was received from the Institute for Wildlife Protection. On December 1, 2003 U.S. District Court judge issued an order in favor of the USFWS and dismissing the plaintiff's case. Plaintiffs have filed a notice that they intend to appeal the Courts decision. Another NOI, dated January 9, 2003, was filed by the plaintiffs regarding the merits of the USFWS's 90-day finding itself.
Lead USFWS Office: Upper Columbia Fish and Wildlife Office, Spokane, Washington (509) 891-6839	**Lead USFWS Office:** Western Colorado Field Office, Grand Junction, Colorado (970) 243-2778	**Lead USFWS Office:** Nevada Fish and Wildlife Office, Reno, Nevada (775) 861-6325
USFWS Contact: Chris Warren	**USFWS Contact:** Terry Ireland	**USFWS Contact:** Kevin Kritz

Table 2.1 cont.

Petition Date: January 24, 2002 (468 pages)	**Petition Date:** June 18, 2002 (7 pages)	**Petition Date:** July 3, 2002 (524 pages)
Species: Western subspecies of the Greater Sage Grouse *Centrocercus urophasianus phaios*	**Species:** Greater Sage Grouse *Centrocercus urophasianus*	**Species:** Eastern subspecies of the Greater Sage Grouse *Centrocercus urophasianus urophasianus*
Petition Request: List the subspecies	**Petition Request:** List as endangered	**Petition Request:** List as endangered
Petitioners: Donald Randy Webb, Institute for Wildlife Protection	**Petitioners:** Craig Dremann	**Petitioners:** Donald Randy Webb, Institute for Wildlife Protection
USFWS Determination: A 90-day finding was initiated October 30, 2002. The 90-day finding was published in the Federal Register on February 7, 2003 with an outcome that the information presented in the petition is not substantial.	**USFWS Determination:** 90-day finding initiated December, 2003. USFWS published the 90-day finding in Federal Register on April 21, 2004. Outcome was a positive 90-day finding; information presented, and in USFWS files, was substantial. USFWS initiates a status review. Public input on status or threats to species should be submitted by June 21, 2004.	**USFWS Determination:** 90-day finding initiated on October 3, 2003 as per court order. The 90-day finding was published in the Federal Register on January 7, 2004 with an outcome that the information presented in the petition is not substantial.
Legal Action: NOI dated February 7, 2003 from the Institute for Wildlife Protection regarding the 90-day finding. Court complaint dated June 6, 2003 from the Institute for Wildlife Protection challenging the merits of the 90-day finding. Both parties waiting on outcome of the Courts decision on this case.	**Legal Action:** No legal action to date	**Legal Action:** Court complaint dated January 10, 2003 filed in the Western District Court of Washington by the Institute for Wildlife Protection for failure to do a 90-day finding. On October 3, 2003 the District Court judge ordered the USFWS to make a 90-day finding which is due by January 3, 2004.
Lead USFWS Office: Oregon Fish and Wildlife Office, Portland, Oregon (503) 231-6179	**Lead USFWS Office:** Wyoming Ecological Services Field Office, Cheyenne, Wyoming (307) 772-2374	**Lead USFWS Office:** Wyoming Ecological Services Field Office, Cheyenne, Wyoming (307) 772-2374
USFWS Contact: Jeff Dillon	**USFWS Contact:** Pat Deibert	**USFWS Contact:** Pat Deibert

Table 2.1 cont.

Petition Date: March 19, 2003 (992 pages; combination of previous petitions for Western and Eastern subspecies)	**Petition Date:** December 22, 2003 (218 pages)
Species: Greater Sage Grouse *Centrocercus urophasianus*	**Species:** Greater Sage Grouse *Centrocercus urophasianus*
Petition Request: List as endangered	**Petition Request:** List as threatened or endangered
Petitioners: Donald Randy Webb, Institute for Wildlife Protection	**Petitioners:** Mark Salvo American Lands Alliance, Biodiversity Conservation Alliance, Center for Biological Diversity, Forest Guardians, The Fund for Animals, Gallatin Wildlife Association, Great Old Broads for Wilderness, Hells Canyon Preservation Council, The Larch Company, Northwest Ecosystem Alliance, Northwest Council for Alternatives to Pesticides, Oregon Natural Desert Association, Oregon Natural Resources Council, Predator Defense Institute, Sierra Club, Sinapu, Western Fire Ecology Center, Western Watersheds Project, Wild Utah Project, Wildlands CPR, and Center for Native Ecosystems
USFWS Determination: 90-day finding initiated December, 2003. USFWS published the 90-day finding in Federal Register on April 21, 2004. Outcome was a positive 90-day finding; the information presented, and in USFWS files, was substantial. USFWS initiates a status review. Public input on status or threats to species should be submitted by June 21, 2004	**USFWS Determination:** 90-day finding initiated December, 2003. USFWS published the 90-day finding in Federal Register on April 21, 2004. Outcome was a positive 90-day finding; the information presented, and in USFWS files, was substantial. USFWS initiates a status review. Public input on status or threats to species should be submitted by June 21, 2004
Legal Action: No legal action to date	**Legal Action:** No legal action to date
Lead USFWS Office: Wyoming Ecological Services Field Office, Cheyenne, Wyoming (307) 772-2374	**Lead USFWS Office:** Wyoming Ecological Services Field Office, Cheyenne, Wyoming (307) 772-2374
USFWS Contact: Pat Deibert	**USFWS Contact:** Pat Deibert

[a]Table compiled by Kevin Kritz, U.S. Fish and Wildlife Service, Nevada Fish and Wildlife Office, 1340 Financial Blvd. Suite #234 , Reno, NV 89502-7147 (775) 861-6300

**** 60-day Notice of Intent to Sue (NOI)**

Table 2.2. Greater Sage-grouse Local Working Groups.

State	Local Working Group	Start Date	Projected Ending Date	Stage of Plan	Lead Agency	Participants	Charter
California	Washoe/Modoc	01-Jan-2002	01-Jul-2004	Final Draft	CFG	All	Yes
California	Western Modoc	15-Feb-2004	01-Jul-2006	Meeting	CFG	All	Yes
California	Southwest Bistate	01-Jan-2002	01-Jul-2004	Final Draft	CFG	All	Yes
California	Surprise-Vya	01-Jan-2002	01-Jul-2004	Final Draft	CFG	All	Yes
Colorado	Middle Park	01-Jan-1999	01-Jan-2001	Complete			
Colorado	Northwest Colorado Working Group	01-Jan-1996	Summer 2004	Formation			
Colorado	North Park	01-Jan-1999	01-Dec-2001	Complete			
Colorado	Eagle/South Routte	01-Mar-1999	Summer 2004	Formation			
Idaho	Challis	01-Dec-2002	01-Sep-2004	Formation	IFG	All	Yes
Idaho	Curlew	01-Apr-1998	01-Sep-2004	Final Draft	IFG	All	Yes
Idaho	Jarbidge	01-May-1999	01-Sep-2004	Final Draft	IFG	All	Yes
Idaho	Owyhee	01-Apr-1998	01-Sep-2004	Complete	IFG	All	
Idaho	Shoshone Basin	01-Jan-1994	01-Jul-2003	Complete	BLM	All	Yes
Idaho	Upper Snake	01-Dec-1998	01-Sep-2004	Final Draft	IFG	All	Yes
Idaho		Not Started		Groups not formed			
Idaho		Not Started		Groups not formed			
Idaho		Not Started		Groups not formed			
Idaho		Not Started		Groups not formed			
Idaho		Not Started		Groups not formed			
Idaho		Not Started		Groups not formed			
Idaho		Not Started		Groups not formed			
Montana	Broadus	01-Jan-2004	01-Jan-2006	Meeting	MFWP	All	Yes
Montana	Dillon	01-Jan-2004	01-Jan-2006	Meeting	MFWP	All	Yes
Montana	Glasgow	01-Jan-2004	01-Jan-2006	Meeting	MFWP	All	Yes
Montana	Harlem/ Chinook, Malta	01-Jan-2006	01-Jan-2008		MFWP	All	Yes
Montana	Jordan	01-Jan-2006	01-Jan-2008		MFWP	All	Yes
Montana	Miles City/ Forsyth	01-Jan-2005	01-Jan-2007		MFWP	All	Yes
Montana	Red Lodge	01-Jan-2005	01-Jan-2007		MFWP	All	Yes
Montana	Roundup/ Ryegate	01-Jan-2006	01-Jan-2008		MFWP	All	Yes
Montana	Terry	01-Jan-2006	01-Jan-2008		MFWP	All	Yes
Montana	White Sulpur Springs	01-Jan-2005	01-Jan-2007		MFWP	All	Yes
Montana	Winnett/Grass Range/Winifred	01-Jan-2005	01-Jan-2007		MFWP	All	Yes
Nevada	Central	01-Jan-2002	01-Jul-2004	Final Draft	NDOW	All	Yes
Nevada	Eastern	01-Jan-2002	01-Jul-2004	Final Draft	NDOW	All	Yes

State	Local Working Group	Start Date	Projected Ending Date	Stage of Plan	Lead Agency	Participants	Charter
Nevada	Elko Stewardship	01-Jun-1999	01-Jul-2004	Final Draft	Elko Stewardship	All	Yes
Nevada	North-Central	01-Jan-2002	01-Jul-2004	Final Draft	NDOW	All	Yes
Nevada	Southwest Bistate	01-Jan-2002	01-Jul-2004	Final Draft	NDOW/CFG	All	Yes
Nevada	Washoe/Modoc	01-Jan-2002	01-Jul-2004	Final Draft	NDOW/CFG	All	Yes
North Dakota	Bowman/Slope	01-Mar-2004	01-Aug-2004	Formation	NDGF	Representative	No
Oregon		Not Started		Groups not formed			
Oregon		Not Started		Groups not formed			
Oregon		Not Started		Groups not formed			
Oregon		Not Started		Groups not formed			
Oregon		Not Started		Groups not formed			
Oregon		Not Started		Groups not formed			
South Dakota	Butte/Harding/Fall River	01-Mar-2004	01-Aug-2004	Formation	SDGFP	Representative	No
Utah	Book Cliff/Unitah Basin	08-Nov-2003	01-Jul-2005	Meeting	UDNR	All	Yes
Utah	Box Elder	30-Aug-2001	01-Jul-2005	Mid Draft	UDNR	All	Yes
Utah	Color Country	15-Nov-2001	01-Jul-2005	Mid Draft	UDNR	All	Yes
Utah	East Manti/Carbon	Not Started	01-Jul-2006	Groups not formed	UDNR	All	Yes
Utah	North Central Valleys	Not Started	01-Jul-2006	Groups not formed	UDNR	All	Yes
Utah	North Slope/Daggett	08-Nov-2003	01-Jul-2005	Meeting	UDNR	All	Yes
Utah	Parker Mountain/Johns Valley	11-Feb-1998	01-Jul-2005	Final Draft	UDNR	All	
Utah	Rich/Summit	07-Dec-2002	01-Jul-2005	Mid Draft	UDNR	All	Yes
Utah	San Juan	13-May-1997	01-Jul-2005	Complete	UDNR	All	Yes
Utah	South Slope/Unitah Basin	08-Nov-2003	01-Jul-2005	Meeting	UDNR	All	Yes
Utah	Southwest Desert	08-May-2003	01-Jul-2005	Meeting	UDNR	All	Yes
Utah	Strawberry	15-May-2003	01-Jul-2005	Meeting	UDNR	All	Yes
Utah	West Desert	12-Nov-2003	01-Jul-2005	Meeting	UDNR	All	Yes
Washington	Douglas/Grant	01-Jan-1998	01-Apr-2004	Complete	WDFW	Representative	
Washington	Yakima	01-Jan-1998	01-Apr-2004	Complete	WDFW	Representative	
Wyoming	Bates Hole/Shirley Basin	01-Jan-2004	01-Jan-2006	Meeting	WGFD	Representative	Yes
Wyoming	Bighorn Basin	01-Jan-2006	01-Jan-2008		WGFD	Representative	Yes
Wyoming	Cheyenne River Basin	01-Jan-2006	01-Jan-2008		WGFD	Representative	Yes
Wyoming	Great Divide Basin	01-Jan-2005	01-Jan-2007		WGFD	Representative	Yes
Wyoming	Jackson Hole	01-Jan-2006	01-Jan-2008		WGFD	Representative	Yes
Wyoming	Lower Green River	01-Jan-2005	01-Jan-2007		WGFD	Representative	Yes
Wyoming	Powder River Basin	01-Jan-2004	01-Jan-2006	Meeting	WGFD	Representative	Yes
Wyoming	Southwest	01-Jan-2005	01-Jan-2007		WGFD	Representative	Yes

State	Local Working Group	Start Date	Projected Ending Date	Stage of Plan	Lead Agency	Participants	Charter
Wyoming	Upper Green River	01-Jan-2004	01-Jan-2006	Meeting	WGFD	Representative	Yes
Wyoming	Upper North Platte	01-Jan-2006	01-Jan-2008		WGFD	Representative	Yes
Wyoming	Wind River/ Sweetwater River	01-Jan-2005	01-Jan-07		WGFD	Representative	Yes

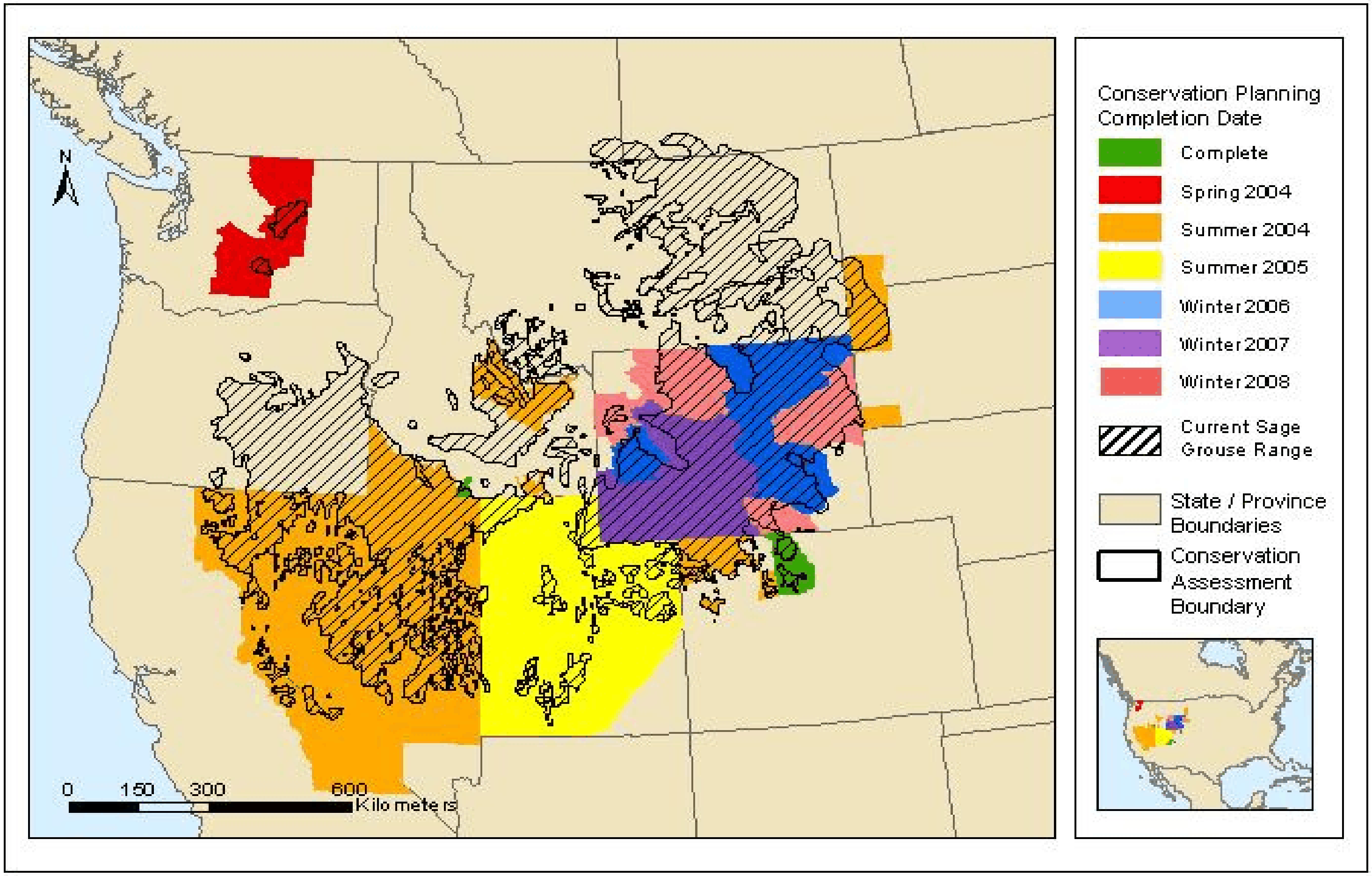

Figure 2.1 Local working group planning completion schedules

Chapter 3

Population Ecology and Characteristics

CHAPTER 3

Population Ecology and Characteristics

Abstract. The greater sage-grouse (*Centrocercus urophasianus*) is the largest species of grouse in North America. It is appropriately named due to its year-round dependence on sagebrush (*Artemisia* spp.) for both food and cover. Insects and forbs also play an important role in their food habits, but primarily during the breeding season. In general, the sage-grouse is a mobile species, capable of movements greater than 50 km between seasonal ranges. Despite this mobility, sage-grouse appear to display substantial amounts of fidelity to seasonal ranges. Sage-grouse populations are characterized by relatively low productivity and high survival. Hence, they do not fit the paradigm of an r-selected bird.

Taxonomy, Systematics, & General Description

The greater sage-grouse is the largest grouse found in North America, followed closely in size by the congeneric Gunnison sage-grouse (*Centrocercus minimus*). Although the 2 species were formerly recognized as a single 'sage grouse' species, they are now considered 2 distinct species based on genetic, morphological, and behavioral research by Young et al. (2000). The greater sage-grouse was first described in print by Meriwether Lewis near the confluence of the Marias and Missouri rivers in Montana on 5 June 1805 (Zwickel and Schroeder 2003). The sage-grouse (greater and Gunnison, combined) was originally named *Tetrao urophasianus* (Bonaparte 1827) and subsequently renamed *Centrocercus urophasianus* (Swainson and Richardson 1831). Although the greater sage-grouse was divided into a western (*C. u. phaios*) and eastern (*C. u. urophasianus*) subspecies based on research by Aldrich (1946), subsequent genetic analysis has not supported the subspecific delineation (Benedict et al. 2003). Nevertheless, the same research showed that sage-grouse along the California-Nevada border near Mono Lake appeared to display numerous unique genetic characteristics.

The greater sage-grouse is characterized by substantial dimorphism in size and appearance of males and females. The average mass is 2.5 – 3.2 kg for adult males (at least 1.5 years old) and 1.3 – 1.7 kg for adult females, with some variation by region and season (Dalke et al. 1963, Eng 1963, Beck and Braun 1978, Hupp and Braun 1991, Schroeder et al. 1999). Male sage-grouse tend to be heaviest in early spring (start of seasonal display cycle) and females tend to be heaviest in late spring (start of egg laying period); both sexes tend to be lightest in autumn. Yearlings (about 0.5 – 1.5 years old) average 0.1 – 0.2 kg lighter than adults among females and 0.3 – 0.4 kg lighter among males (Schroeder et al. 1999).

Both sexes have long pointed tails, with the tails on males significantly longer (Schroeder et al. 1999). The overall length of a male greater sage-grouse can be about 75 cm, with the female about 15 cm shorter. Both sexes have a general fuscous appearance overall, with small marks of drab-gray and white; their primaries are solid hair brown and their abdomen is sepia-colored. Although the female tends to be cryptically colored, the breast and neck feathers of males offers more contrast. The breast feathers in males are prominently white and composed of short, stiff feathers (Brooks 1930, Hjorth 1970). Males also have long filoplumes that arise from the back of the neck, a sepia throat, and 2 yellowish cervical apteria that are prominently visible on the lower neck/upper breast during the breeding display.

Food Habits

General

Greater sage-grouse are sagebrush obligates and the importance of sagebrush as a source of food and cover has been well established (Patterson 1952, Braun et al. 1977, Connelly et al. 2000*b*). Sagebrush is a primary food item for adults throughout the year (Wallestad et al. 1975). However, sage-grouse food habits are complex and forbs and insects are consumed at certain times of year (Knowlton and Thornley 1942, Leach and Hensley 1954, Pyle 1993, Drut et al. 1994*a*). Diet composition may have an influence on reproductive success of females (Barnett and Crawford 1994) and is crucial for growth and survival of chicks (Johnson and Boyce 1990). In addition, seasonal variation in sage-grouse diets particularly during spring, summer, and fall/winter may directly influence habitat use (Klebenow and Gray 1968, Peterson 1970, Hupp and Braun 1989, Barnett and Crawford 1994).

Spring

Information on diets of sage-grouse during spring is limited. Sagebrush composed 97 to 100% of the diet during February and March in Montana (Wallestad et al. 1975). Sage-grouse diets during April contained 100% sagebrush in Colorado (Rogers 1964) and 89% sagebrush in Wyoming (Patterson 1952). Diets during spring may be important for reproduction in tetronids (Moss et al. 1975, Beckerton and Middleton 1982). However, these studies did not differentiate between diets of males and females. Barnett and Crawford (1994) investigated the pre-laying diet and nutritional content (crude protein, phosphorus and calcium) of sage-grouse hens in Oregon. Hens were collected from 2 areas in low and big sagebrush cover types during the 5-week period preceding incubation to determine food availability and nutrient content of plant parts (leaves, flowers, or buds) consumed by hens (Barnett 1992). Sagebrush leaves composed the bulk of the diet (ranged from 50 to 82%), with the remainder composed primarily of forbs (Barnett and Crawford 1994). However, sagebrush was the least selected food for dietary items that composed >1% of the dry matter of the diet.

Important forbs in the diet of pre-laying hens included desert-parsley (*Lomatium spp*.), hawksbeard (*Crepis spp*.), phlox (*Phlox spp*.), everlasting (*Antennaria spp*.), mountain-dandelion (*Agoseris spp.),* clover (*Trifolium spp.),* Pursh's milk-vetch (*A. purshii*), obscure milk-vetch (*A. obscurus*), and buckwheat (*Eriogonum spp*.) (Barnett 1992). Although sagebrush leaves composed a substantial proportion of the dry mass in the diets of pre-laying hens, the nutrient contribution of forbs exceeded that of sagebrush (Barnett and Crawford 1994). Consumption and availability of forbs differed between years, which coincided with substantial differences in sage-grouse productivity and suggested that diet of sage-grouse hens during the pre-laying period may influence reproductive success (Barnett and Crawford 1994).

Insects and forbs are critically important foods for juvenile sage-grouse and may influence survival and growth rates. Newly hatched chicks in captivity that were deprived of insects died within 10 days and older chicks (> 3 weeks) had reduced growth rates (Johnson and Boyce 1990). A comparison between 2 areas in Oregon found that sage-grouse production was greater in the area where forbs and insects composed >80% of the diet of chicks (Drut et al. 1994*a*). In the less productive area, sagebrush accounted for 65% of the diet (Drut et al. 1994a). The availability of fewer forbs in the less productive site may have increased home range size for hens with broods and resulted in increased exposure of chicks to predation, accident, and other mortality factors (Drut 1993). The protein-rich diet of forbs and insects of chicks at the more productive area likely enhanced the nutritional status of chicks and increased survival (Drut 1993).

Summer

Insects and forbs composed the bulk of the diet of juvenile sage-grouse in Idaho, Montana and Oregon (Klebenow and Gray 1968, Peterson 1970, Drut et al. 1994*a*). As chicks aged, consumption of sagebrush increased (Klebenow and Gray 1968, Peterson 1970). Insects were the greatest component of the diet for chicks until the second week post-hatch when forbs were consumed in greater quantities (Klebenow and Gray 1968, Peterson 1970). Diet of juvenile sage-grouse consisted of 75% forbs in Montana (Peterson 1970), > 77% forbs in Idaho (Klebenow and Gray 1968), and 66% in Oregon (Pyle 1993). Juvenile sage-grouse consumed a high diversity of forbs (Table 3.1). In an experiment with human-imprinted juveniles in Colorado, chicks gained more weight when they fed in a relatively forb-rich habitat than when they fed in a forb-poor habitat (Huwer 2004). In Oregon, diets of chicks contained items from 122 foods, including 41 families of invertebrates and 34 genera of forbs (Pyle 1993, Drut et al. 1994*a*). Among the most common plant foods were hawksbeard (*Crepis spp*), clover (*Trifolium spp*.), milk-vetch (*Astragalus spp*.), false dandelion (*Agoseris glauca*), and microsteris (*Phlox gracilis*) (Pyle 1993, Drut et al. 1994*a*). Although chicks consumed a variety of insects, the most common in chick diets included beetles (*Scarabeidae and Tenebrionidae*), thatch ants (*Formicidae*), and grasshoppers (*Orthoptera)* (Klebenow and Gray 1968, Peterson 1970, Pyle 1993, Drut et al. 1994*a*).

Hens with broods are typically found in areas with the greatest forb availability (Klebenow 1969, Drut et al. 1994*b*, Apa 1998, Sveum et al. 1998*b*). As summer progresses, the availability of forbs varies depending on habitat type, moisture, and elevation. Hens with broods altered habitat use in response to changes in forb availability (Klebenow 1969, Peterson 1970, Wallestad 1971, Dunn and Braun 1986*b*, Connelly et al. 1988, Drut et al. 1994*a*, 1993, Fischer et al. 1997). Hens with broods used greasewood (*Sarcobatus vermiculatus*) bottoms and grassland cover types when desiccation reduced forb availability in sagebrush uplands in Montana (Peterson 1970, Wallestad 1971) and hens with broods responded to changes in forb availability by following elevation gradients of succulent vegetation in Idaho (Klebenow 1969).

During summer, diets of adult sage-grouse were diverse and consisted of sagebrush, forbs, and insects (Rasmussen and Griner 1938, Wallestad et al. 1975). Sagebrush composed

<60% of the summer diet of adult birds in Montana (Wallestad et al.1975) and Oregon (Hanf et al. 1994). Although adult sage-grouse also consumed insects, forbs made up the majority of the non-sagebrush component of the diet during summer (Wallestad et al. 1975). The availability of forbs may influence both habitat use and movements of broodless hens (Gregg et al. 1993), but unlike juvenile birds, sagebrush may remain a large component of the diet of adult sage-grouse during summer (Rasmussen and Griner 1938, Wallestad et al. 1975).

Fall and Winter

During fall and winter, diets of juvenile and adult sage-grouse were similar and, in Wyoming consisted primarily of sagebrush leaves (Patterson 1952). Similarly, sagebrush was the only food item found in crops of sage-grouse during winter in Montana (Wallestad et al. 1975). Sage-grouse consume leaves from a variety of sagebrush species including big sagebrush (*A. tridentata*), low sagebrush (*A. arbuscula*), alkali sagebrush (*A. longiloba*), and black sagebrush (*A. nova*) (Wallestad et al. 1975, Remington and Braun 1985, Welch et al. 1988). In Colorado, sage-grouse selected for Wyoming big sagebrush (*A. t. wyomingensis*) during winter presumably because of higher protein content compared to mountain big sagebrush (*A. t. vaseyana*) and alkali sagebrush (*A. longiloba*) (Remington and Braun 1985). Unless snow completely covers sagebrush, severe weather conditions apparently do not seriously impact sage-grouse populations and sage-grouse gain weight during winter (Beck and Braun 1978).

Seasonal Movement and Fidelity

A general pattern of movement (or migration) between seasonal home ranges has been noted in many populations of sage-grouse (Dalke et al. 1960, Gill and Glover 1965, Berry and Eng 1985, Connelly et al. 1988, Bradbury et al. 1989). Considerable variation exists with respect to these patterns, including characteristics that are specific to populations (Connelly et al. 1988) and in relation to the distribution of habitat and/or corridors (see Chapter 4). Some populations have been suggested to be resident, while others have been recorded traveling distances as far as 161 km (Patterson 1952). In addition, migratory populations can migrate between winter/breeding and summer areas (2 stages), between winter and breeding/summer areas (2 stages), and between breeding, summer, and winter areas (3 stages, Connelly et al. 1988).

In a 2-stage migrant population in the Lemhi Valley - Birch Creek area of Idaho (winter and breeding area the same and summer area different), Connelly et al. (1988) found an average distance of 13.1 km (range 1-64 km, n = 65) between summer and winter range. No difference in age or sex was noted. The average distance between winter/breeding areas and summer areas was 35.2 km (up to 82 km) for 47 males and 11.1 km (up to 72 km) for 27 females. Hausleitner (2003) found an average distance of 9.9 km for 76 females between their winter range and their leks. In addition to the significant difference associated with gender, migration distance tended to be related to the configuration of seasonal habitat (Connelly et al. 1988). The observations were in contrast to those of many other species of grouse in which females typically moved farther and more frequently than males (e.g., Schroeder and Braun 1993).

Numerous explanations have been proposed to explain patterns of movement observed for greater sage-grouse including differences in seasonal habitat selection, desiccation of succulent forbs during summer, harsh weather during winter, and seasonal site fidelity (Dalke et al. 1960; Gill and Glover 1965; Wallestad 1971; Berry and Eng 1985; Connelly et al. 1988; Fischer et al. 1996, 1997). The close configuration of winter and breeding habitat in some areas may result in relatively short or non-existent movements between winter and breeding areas, whereas the long distance between breeding and summer habitat results in long movements (Connelly et al. 1988). In other areas, breeding habitat may be positioned between winter and summer range (e.g., California; Bradbury et al. 1989). In a Wyoming study, the breeding and summer ranges tended to be relatively close together and winter range was more distant (Berry and Eng 1985).

The peak of autumn migration is mid-October through late November, spring migration is mid-February through mid-March, and summer migration is late May through early August (Schroeder et al. 1999). Autumn movements of greater sage-grouse in Idaho have been described as "slow and meandering" with a travel rate of 0.3 km/day and summer movements as "more direct and rapid" with a rate of 0.9 km/day (Connelly et al. 1988:119-120). Connelly et al. (1988) also noted that males tend to move faster than females. Berry and Eng (1985) also noted that the onset of migration could be associated with weather. Although it is clear that weather and habitat distribution influence patterns of migration, they are not always sufficient to explain the relatively large migration distances in relation to the short distances between seasonal habitat types. One possible explanation for this 'discrepancy' is that birds may display fidelity to their first winter and breeding areas (Berry and Eng 1985, Connelly et al. 1988, Schroeder and Robb 2004). Hence, migration may reflect the original dispersal tendencies of juveniles rather than being a simple reflection of habitat distribution.

Dispersal may be extremely important for integrating populations, recolonization of habitats, and for maintaining genetic flow (Greenwood and Harvey 1982, Linberg et al. 1998, Barrowclough et al. 2004). Unfortunately, very little is known about dispersal in greater sage-grouse. In one Colorado study, 12 females dispersed a median of 8.8 km and 12 males a median of 7.4 km between their approximate places of hatch to their approximate place of breeding or attempted breeding (Dunn and Braun 1985). Dispersal appears to be discrete from brood breakup (Browers and Flake 1985) and the movements are relatively gradual and sporadic (Dunn and Braun 1986*a*).

An understanding of seasonal movements requires an understanding of site fidelity. If sage-grouse did not display fidelity to their breeding, summering, and/or wintering areas, then their migratory movements would be better described as nomadic movements. Fidelity to display sites (leks) has been well documented in greater sage-grouse (Dalke et al. 1963, Wallestad and Schladweiler 1974, Emmons and Braun 1984, Dunn and Braun 1985, Schroeder and Robb 2004), a trait they share with other species of grouse that breed on leks (Schroeder and Robb 2004). In addition, visits to multiple leks tend to be less frequent for adult males than yearlings, suggesting an age-related period of establishment (Emmons and Braun 1984, Schroeder and Robb 2004).

The fidelity of females to their nesting areas also has been examined. The distance between an individual female's nests in consecutive years was a median of 0.7 km (range 0.0 – 2.6 km) in an Idaho study (Fischer et al. 1993) and an average of 3.0 km (SD = 6.8 km) in a Washington study (Schroeder and Robb 2004). In Washington and Colorado, unsuccessful females tended to move farther between consecutive nests (Hausleitner 2003, Schroeder and Robb 2004). However, there was no statistical indication that these relatively long movements increased their subsequent likelihood of nesting success. The average distance between first nests and renests was 2.6 km (SD = 4.5 km) in Washington; these consecutive nests were closer together when the female was an adult than when she was a yearling (Schroeder and Robb 2004). As with yearling males, this behavior suggests a period of establishment. One published explanation for the relatively large distances in Washington was the substantial amount of habitat fragmentation; one exceptional female had consecutive nests 32 km apart (Schroeder and Robb 2004). In contrast to the information on breeding site fidelity, there is little published information on site fidelity during other seasons.

The relatively large seasonal movements have made the estimation of home range size difficult to measure and extremely variable in greater sage-grouse. Home ranges can vary from 0.1 – 28.6 km^2 during the breeding season, 0.1 – 25.9 km^2 during summer, 22.5 – 44.2 km^2 during autumn, and 0.6 – 18.2 km^2 during winter (Schroeder et al. 1999). Some of the variation is associated with seasonal behavior, habitat requirements, and the juxtaposition of seasonal habitats (Connelly and Markham 1983, Holloran 1999, Hausleitner 2003). When considered on an annual basis, a migratory individual may occupy an areas 6 - 615 km^2 (Connelly et al. 2000*b*, Hausleitner 2003). Some of the variation is associated with behavior and habitat requirements.

Breeding Biology

Mating System

The greater sage-grouse, as well as most grouse species, is polygynous. Polygyny can be defined as 1 male mating with multiple females and each female selectively chooses the male she mates with (Bergrud 1988). Sage-grouse exhibit "clumped polygyny" where multiple males display on the same arena for females (Bergerud 1988). The evolution of polygyny in grouse species has been discussed by Wiley (1974), Wittenberger (1978) and Bergerud and Gratson (1988). Wiley (1974) suggested that polygyny evolved because there is a sexual difference in age when breeding first occurs. However, young males are physiologically capable of breeding as yearlings (Eng 1963, Bergerud 1988). Bergerud (1988) suggested that instead, yearlings may choose not to breed during their first year because they cannot compete with adults. If yearlings remain inconspicuous they have a higher probability of surviving and thus, a better chance of becoming a successful breeder later in life (Wittenberger 1978, Bergerud 1988). Estimates of survival support this (Zablan 2003). Wittenberger (1978) suggested that female choice determines the evolution of grouse mating systems. He also suggested that polygyny evolved in steppe grouse occupying open habitats because several individuals can detect predators better than one individual. Bergerud (1988) hypothesized that male's benefit by displaying and traveling in flocks for predator vigilance. The possibility that males may congregate as a response to female movement and sociality has also been considered (Schroeder and White 1993).

The greater sage-grouse performs an elaborate display for females on communal breeding grounds called leks. The unique quality of a sage-grouse lek compared to other grouse species is that only a few males mate with most of the females in a particular area (Gibson et al 1991). Gibson and Bradbury (1986) defined a lek system by four criteria: males do not care for young; displaying males occur in groups; leks occur away from nesting areas; and females have freedom to choose a mate. Leks can range from one to at least 16 hectares in size (Scott 1942, Patterson 1952).

Sage-grouse use a variety of locations for leks, specifically open areas. Leks can occur on wind swept ridges and rocky knolls, low sagebrush, bare openings created by roads and fire, stock ponds, air strips, natural meadows, dry lake beds, alkaline flats, and ant hills (Patterson 1952, Giezentanner and Clark 1974, Connelly et al. 1981). Scott (1942) and Patterson (1952) noted that leks occur in the same location each year. Gibson and Bradbury (1987) observed a shift in lek location following a severe winter when the traditional lek sites were covered in snow until May. Gibson (1996) hypothesized that leks are located in areas of high female traffic. He defined this as "hotspot" settlement. Gibson (1996) also noted that the size of a lek corresponded positively with number of hens traveling near that lek. Male numbers increased when females arrived and remained stable when females were present (Gibson 1996).

Territoriality

Gibson (1992) defined territoriality on a lek as any male that consistently used the same area and excluded other males. Territories can range from small, exclusive areas to larger overlapping ranges (Gibson and Bradbury 1987). Adult males usually establish 5-100 m^2 territories on leks, often maintaining them throughout the breeding season, and occasionally between years (Simon 1940, Patterson 1952, Dalke et al. 1960, Wiley 1973, Gibson and Bradbury 1987, Hartzler and Jenni 1988, Gibson 1992, Gibson and Bradbury 1986, Bradbury et al. 1989). Yearling males rarely defend territories, although they are physiologically capable of breeding (Eng 1963). Male sage-grouse do not exhibit territorial behavior during any other time of year (Schroeder et al. 1999). The possibility of female territoriality, perhaps associated with nest sites, has not been explored.

Scott (1942) suggested that territories are established and dominance determined early in the breeding season as there is much fighting, challenging, and display occurring at that time. Wiley (1973) suggested that territory size relates to the males success in mating. Territories near the mating center are smaller than periphery territories. Patterson (1952) and Wiley (1973) observed dominant males returning to the same territories on a lek each year. Gibson (1992) suggested that successful males exhibit fidelity to the lek and tend to return to the same location the following year. He also noted that a territory abandoned by a successful male becomes the focal point for subsequent male-male competition. Wiley (1973) noted that territories on small leks and leks that are disturbed are less consistent. Gibson and Bradbury (1987) observed successful and unsuccessful males shifting territories when females visited the lek, and that leks appeared to be more unstable than stable regarding establishment of territories. Eng (1963) noted that yearling males do not typically establish territories or breed. He stated that yearling males may be physiologically unfit to compete with older males. Territorial males within the mating center perform the majority of copulations (Wiley 1973, Patterson 1952, Scott 1942). Wiley reported 30 copulations in 3 hours by a single male. Patterson (1952) observed 4 males performing 18 copulations. One of these males performed 7 of the 18 copulations. Scott (1942) suggested that guard cocks kept intruders away from the master cocks and sub-cocks. He observed 114 matings by master cocks, 20 by sub-cocks, and 5 by guard cocks. Scott (1942) also observed 15 additional matings away from the mating center. Hartzler and Jenni (1988) reported 169 copulations by 1 male during a single breeding season. Wiley (1978) observed interruptions in mating activities when a male mounted a female too close to another male's territorial boundary. Males typically do not lose their territory as a result of mating interruptions or encounters with other males (Wiley 1973).

Physical Interactions

Physical interactions between males often occur on leks. During intense encounters, male sage-grouse stand side-by-side, a few centimeters apart and often erupt in wing fights (Wiley 1973, Hjorth 1970). Each opponent attempts to force the other to retreat, causing movements ½ m forwards and backwards (Wiley 1973). Patterson (1952) and Wiley (1973) observed more wing fights at the beginning of the breeding season before territories are fully established, and at the end of the breeding period directed at un-established males and yearlings. During wing fights, males stretch their necks upwards, close their tail and hold it horizontally above the ground (Wiley 1973). They may slap each other vigorously with wings and jump a few centimeters into the air, slamming their wings down on their opponent (Schroeder et al. 1999). Females may also chase, mount, and peck other females on the lek, although they are generally relatively tolerant of each other (Scott 1942, Hjorth 1970, Wiley 1973).

Courtship

Males attending leks each spring perform an elaborate display for females called the 'strutting display' (Scott 1942, Eng 1963, Hjorth 1970). A detailed description of the entire strutting display is described by Hjorth (1970). Most behavior studies on sexual selection by female greater sage-grouse were conducted in California (Bradbury et al. 1989, Gibson 1989, Gibson 1992, Gibson 1996*a*, Gibson and Bachman 1992, Gibson and Bradbury 1986, Gibson et al. 1990, Gibson et al. 1991). Successful males display 6-10 times/minute, 3-4 hours each morning (Gibson et al. 1991). Most of the activity occurs around sunrise when many of the matings occur (Scott 1942, Patterson, 1952, Hjorth 1970). Hens typically arrive on a lek in clusters and are attracted to specific groups of displaying males (Hartzler and Jenni 1988, Wiley 1978). Single hens walking onto leks often join an existing cluster (Hartzler and Jenni 1988).

Wiley (1978) and Gibson et al. (1991) hypothesized that these hens may lack experience and thus 'copy' the mate selection of other hens.

Timing of Breeding Behavior

Male sage-grouse attend leks for up to 3 months each spring (Vehrencamp et al 1989). Males appear on leks just prior to sunrise during the early part of the display season and depart shortly after sunrise (Jenni and Hartzler 1978). As the season progresses, males arrive on the leks earlier and remain later, especially when hens are present (Jenni and Hartzler 1978). During peak attendance, males may display for up to 3-4 hours each morning and often during the late evening and night (Scott 1942, Paterson 1952, Hjorth 1970, Walsh 2002).

Depending on snow depth, elevation, weather, and region, male sage-grouse begin displaying around the end of February to early April and end displaying in late May or early June (Eng 1963, Schroeder et al. 1999, Aldridge 2000, Hausleitner 2003). Adult males arrive at the strutting grounds earliest in the season followed by females and subadult males (Dalke et al.1960, Eng 1963, Jenni and Hartzler 1978, Emmons and Braun 1984).

Female greater sage-grouse begin moving from winter-use areas to breeding areas in late-February to early-March (Patterson 1952, Eng and Schladweiler 1972, Braun and Beck 1976, Petersen 1980, Schoenberg 1982, Bradbury, et. al 1989*a*). Females typically start visiting leks in early to mid-March in Washington (Sveum 1995, Schroeder 1997), Oregon (Hanf et al. 1994), and lowland populations of Idaho (Idaho Department of Fish and Game, unpublished data), mid- to late March in Montana (Jenni and Hartzler 1978), California (Bradbury et al. 1989*a*), and mountain valley populations of Idaho (Idaho Department of Fish and Game, unpublished data), and late March to early April in Alberta (Aldridge 2000), Colorado (Petersen 1980, Hausleitner 2003), and Wyoming (Patterson 1952). Peak hen attendance is typically mid- to late March in Washington (Schroeder 1997), late March to early April in California (Bradbury et al. 1989*a*), Oregon (Hanf et al. 1994), and lowland populations of Idaho (Idaho Department of Fish and Game, unpublished data), early April in Alberta (Aldridge and Brigham 2001), Moffat County Colorado (Hausleitner 2003), and Montana (Jenni and Hartzler 1978), and early to mid-April in Wyoming (Patterson 1952), North and Middle Park Colorado (Petersen 1980, Walsh 2002), and mountain valley populations of Idaho (Idaho Department of Fish and Game, unpublished data). Weather variation of about 2 weeks was noted in Washington (Schroeder 1997) and Montana (Jenni and Hartzler 1978). Once peak hen attendance occurs, numbers drop relatively quickly with only a small number of hens attending a lek each day (Jenni and Hartzler 1978).

Eng (1963) observed a gradual increase in lek attendance by males as females arrived on the leks. Following peak hen attendance, more subadult (yearling) males tend to appear with peak male attendance occurring approximately 3 weeks after peak hen attendance (Patterson 1952, Eng 1963, Jenni and Hartzler 1978, Emmons and Braun 1984, Walsh 2002, Walsh et al. 2004). Emmons and Braun (1984) observed that only 64% of radio-marked yearling sage-grouse attended leks during peak female attendance while 100% of adults attended during that period. Conversely, 100% of both radio-marked yearling and adult sage-grouse attended leks during peak male attendance (Emmons and Braun 1984). In contrast Walsh et al. (2004) found that on 58% of days in which 7 radio-marked adult males were observed, they did not apparently attend a lek.

Nesting

General Characteristics. Females lay eggs that vary from olive-buff to greenish-white with small spots and fine dots of brown and brownish olive (Patterson 1952, Short 1967, Schroeder et al. 1999). The overall color, especially the spots, fades in sunlight. Clutch size of greater sage-grouse hens varied between 6.3 and 9.1 eggs (average = 7.3 eggs for 11 studies; Table 3.2). Clutch size was higher for adults than yearlings in Colorado (Petersen 1980, Haustleitner 2003) and Montana (Wallestad and Pyrah 1974), and for first nests than renests in north-central Washington (Schroeder 1997). Clutch size also varied significantly by year in Washington (Schroeder 1997).

The nest is a bowl on the ground that is sparsely lined with vegetation and feathers from the brood patch (Schroeder et al. 1999). The peak of egg-laying and incubation varies from late March through mid-June, with renesting stretching into early-July (Schroeder et al. 1999). The typical date for initiation of incubation appears to be about 3-4 weeks following the peak of female attendance on leks (Schroeder 1997, Aldridge and Brigham 2003, Hausleitner 2003). Adults initiated incubation on average 9 days earlier than yearlings in north-central Washington (Schroeder 1997). Hatched chicks are precocial and leave the nest soon after hatch; they are capable of weak flight by 10-days of age and strong flight by 5-weeks of age (Schroeder et al. 1999).

Nest Placement. The average distance between a female's nest and the lek where she was captured varied dramatically by region and study: 2.7 km (n = 22) in Montana (Wallestad and Pyrah 1974), 3.4 km (n = 94) in Idaho (Fischer 1994), 4.0 km (n = 101) in Colorado (Hausleitner 2003), 4.6 km (n = 36) in Idaho (Wakkinen et al. 1992), 7.8 km (n = 138) in Washington (Schroeder et al. 1999). Other studies have illustrated similar variation (Berry and Eng 1985, Hanf et al. 1994, Holloran 1999, Lyon and Anderson 2003, Slater 2003). Braun et al. (1977) indicated that most hens nest within 3.2 km of a lek, but most recent literature suggests many hens nest further from their lek of capture than previously documented. Average distance moved by hens from undisturbed leks to nests in western Wyoming was 2.1 km, while average distance traveled from disturbed leks to nests was 4.1 km (Lyon and Anderson 2003).

Nest Likelihood. Observed nest initiation rates may be somewhat dependent on research methodologies, but also may vary by region (Table 3.2). The average likelihood of a female nesting in a given year was 79.9% for 11 different studies with a range of 63 to 100%. Nest initiation rate tended to be higher for adults (78-92% than yearlings (55-79%) in two separate studies in Idaho (Connelly et al. 1993; N. Burkepile, unpublished data). Nest initiation rate was also higher for hens captured on undisturbed leks in western Wyoming than for hens captured on disturbed leks (Lyon and Anderson 2003).

The frequent presence of > 15 ovulated follicles (Dalke et al. 1963) and the secondary peak of female attendance at leks (Eng 1963) provided indirect evidence that females may renest following failure of their first nest. Direct evidence from radio-telemetry studies has illustrated dramatic variation in renesting likelihood by study, and perhaps region (Table 3.2). The average likelihood of renesting was 28.9 for 9 different studies with a range of 9 to 87%. Females were observed nesting 3 times in 2 studies in Washington (Sveum 1995, Schroeder 1997). Adults appeared more likely to renest than yearlings in Washington (Schroeder 1997) and Idaho (N. Burkepile, unpublished data). The lower likelihood of renesting by yearlings in Washington was attributed, in part, to later initiation of their first nests and hence, their shorter nesting season (Schroeder 1997).

Nest success. Nest success of greater sage-grouse varied between 14.5% and 86.1%, depending on area, study, and methodology (Table 3.2). The average nest success for 16 studies using radio telemetry was 47.7%. Nest success did not differ between adults and yearlings except in central Montana where adults had greater nest success than yearlings (Wallestad and Pyrah 1974). Interestingly, nest success varied from one study to another in Clark County, Idaho. A study conducted without radio telemetry in the early 1940’s indicated nest success was 76.6% (Bean 1941). A current study with radio telemetry indicates that nest success in the same area is 52% (Burkepile, unpublished data). Further, a small-scale study conducted in Lassen County, California suggested that hens that traveled further from their lek of capture were more successful than those remaining closer to the lek; successful hens traveled 4.1 km on average while unsuccessful hens traveled 2.3 km (Hall 2001). Renests were more successful (56%) than first nests (29%) in a California study (Hall 2001). At the Hart Mountain National Wildlife Refuge, nest success was 19.6% (*n*=63) in the early 1990’s (Gregg et al. 1994) and 36.5% (n=76) during the late 1990’s (Coggins 1998).

Annual reproductive success (probability of a female hatching ≥ 1 egg in a season) was higher than rate of nest success because of renesting attempts. The high rate of renesting in north-central Washington resulted in a 61.3% annual reproductive success for 111 females (some females monitored > 1 year) compared with a 36.7% rate of nest success for 188 nests (Schroeder 1997). There have been numerous explanations for these low rates of nest success and/or low rates of annual reproductive success. These include the lack of adequate forbs and low residual herbaceous cover (Barnett and Crawford 1994, Gregg et al. 1994, Hanf et al 1994, DeLong et al. 1995, Coggins 1998).

Survival and Population Dynamics

Survival in a sage-grouse population can be partitioned into two basic stages; 1) survival of juveniles from hatch to their first potential breeding season and 2) annual survival of breeding-aged males and females. Because there are numerous methods used for evaluating survival (bands, radio transmitters, poncho-tags, brood observations), it is difficult to obtain estimates of survival that are comparable between studies. Crawford et al. (2004) averaged several studies to obtain estimates, including an estimate of 10% survival for juveniles from hatch to the age of breeding or attempted breeding; the estimate was based in part on an estimates of early juvenile survival including 33% for Washington (Schroeder 1997), 60% for Wyoming (Holloran 1999), 7% for Utah (Bunnell 2000), and 19% for Alberta (Aldridge and Brigham 2001). Although food availability, habitat quality, harvest, and weather may impact juvenile survival (Rich 1985, Pyle and Crawford 1996, Sveum et al. 1998*b*, Holloran 1999, Aldridge 2000, Huwer 2004), the lack of adequate survival estimates has made these potential relationships difficult to test.

Zablan (2003) estimated survival for 6,021 banded sage-grouse in Colorado using bands recovered from hunters. They estimated survival to be 59.2% (95% CI, 57.1 – 61.3%) for adult females, 77.7% (95% CI, 71.8 – 75.3%) for yearling females, 36.8% (95% CI, 35.4 – 44.8%) for adult males, and 63.5% (95% CI, 56.9 – 64.6%) for yearling males. They recovered 1 female ≥ 9 years old, 3 females ≥ 8 years old, and 3 males ≥ 7 years old. Females had higher survival than males and adults had lower survival than yearlings. Wittenberger (1978) and Bergerud (1988) suggested that yearling males remain inconspicuous during their first year and thus, have a better chance of surviving to adulthood. Male survival was estimated to be 59% in Wyoming (June 1963), 58-60% in Idaho (Connelly et al. 1994, Wik 2002), and 29.6% in Utah (Bunnell 2000). In contrast, female survival was estimated to be 67-78% in Wyoming (June 1963, Holloran 1999), 48-75% in Idaho (Connelly et al. 1994, Wik 2002), 57% in Alberta (Aldridge and Brigham 2001), 60.6% in Colorado (Hausleitner 2003) and 36.8% in Utah (Bunnell 2000). The lower survival rate of males (in 4 of 5 previously listed studies with both sexes) is the primary reason why the male:female sex ratio appears to decline as birds in the population age (Patterson 1952, Braun 1984, Swenson 1986). Swenson (1986) suggested that lower male survival was due to the adverse effects of sexual dimorphism.

Zablan (2003) also estimated recovery rates of 14.0-18.7%. Although they argued that these recovery rates were similar to other studies, there are few populations of sage-grouse with published survival and recovery estimates. They also were unable to detect any relationships between survival and weather, despite the large sample of banded birds.

There has been little range-wide effort to examine the seasonal patterns of mortality. Nevertheless, most research suggests that over-winter mortality is low (Robertson 1991, Wik 2002, Hausleitner 2003, Zablan 2003). Although most mortality of sage-grouse is due to predation (see Chapter 10 for detailed assessment), a substantial amount of mortality in some areas may be associated with harvest (Connelly et al. 2003, Zablan 2003; see Chapter 9 for detailed assessment). The reduced winter mortality may have implications on the appropriate level of harvest within populations of sage-grouse (Connelly et al. 2000*a*).

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Table 3.1. Common forbs identified from juvenile sage-grouse diets (from Klebenow and Gray 1968, Peterson 1970, Pyle 1993, Drut et al. 1994*a*).

Common Name	Scientific Name
Nevada desert-parsley	*Lomatium nevadense*
Hawksbeard	*Crepis spp.*
Mountain dandelion	*Agoseris spp.*
Milk-vetch	*Astragalus spp.*
Broomrape	*Orbanche spp.*
Clover	*Trifolium spp.*
Microsteris	*Phlox gracilis*
Fleabane	*Erigeron spp.*
Common dandelion	*Taraxacum officinale*
Yellow salsify	*Tragopogon dubius*
Curlcup gumweed	*Grindelia squarrosa*
Prickly lettuce	*Lactuca serriola*
Prairie pepperweed	*Lepidium densiflorum*
Fringed sagewort	*Artemisia frigida*
Sego lily	*Calochortus macrocarpus*
Harkness gilia	*Linanthus harknessii*
Common yarrow	*Achillea millifolium*
Aster	*Aster occidentalis*
Monkey flower	*Mimulus nanus*
Ground smoke	*Gayophytum spp.*
Everlasting	*Antennaria microphylla*
Blepharipappus	*Blepharipappus scaber*

Table 3.2. Average data for female greater sage-grouse during the breeding season (after Schroeder et al. 1999).

State or Province	Clutch Size (n)	% Nest Likelihood (n)	% Renest Likelihood (n)	% Nest Success (n)	% Breeding Success (n)	Chick Survival to 50 days (n)	Source
Alberta	7.6 (28)	100.0 (28)	35.7 (14)	46.2 (26)	54.5 (22)	18.5 (88)	Aldridge and Brigham 2001
Colorado	7.0 (29)		37.5 (16)	45.2 (31)	46.7 (30)		Petersen 1980
	6.7 (81)	84.9 (119)	11.8 (34)	55.1 (107)	56.9 (130)		Hausleitner 2003
Idaho	6.3 (25)	92.1 (38)	16.7 (18)	44.7 (38)			Wik 2002
	7.6 (47)			76.6 (47)[a]			Bean 1941
	6.7 (30)	68.6 (242)	15.2 (79)	52.0 (166)	48.1 (52)		Connelly et al. 1993
				51.2 (41)			Apa 1998
Montana	8.2 (22)	71.0 (31)		70.0 (20)	70.0 (20)		Wallestad and Pyrah 1974
							Jenni and Hartzler 1978
Oregon		78.2 (119)	9.3 (75)	14.5 (124)	15.1 (119)		Gregg 1991, Gregg et al. 1994
			22.0 (19)	36.5 (76)		37.0 (35)	Coggins 1998
Utah		63.2 (19)		66.0 (19)		7.0 (42)	Bunnell 2000
	6.8 (147)			60.2 (161)[a]			Rasmussman and Griner 1938
				86.1 (36)[a]			Trueblood 1954
		63.1 (103)		70.6 (84)			Chi 2004
Washington	9.1 (55)	100.0 (129)	87.0 (69)	36.7 (188)	61.3 (111)	33.4 (515)	Schroeder 1997
	6.6 (38)	80.0 (95)	25.0 (44)	40.9 (93)	40.0 (95)		Sveum et al. 1998*a*
Wyoming	7.4 (154)			38.4 (216)[a]			Patterson 1952
				23.7 (76)			Slater 2003
		77.1 (70)		50.0 (54)			Lyon and Anderson 2003
				61.0 (82)		74.8 (131)	Holloran 1999

[a]Four studies of nest success that were conducted without radio telemetry produced an apparent rate of nest success of 65.3%. These were not considered in the overall average for the 18 studies conducted with telemetry which produced an average estimate of 45.1%.

Chapter 4

Sage-Grouse Habitat Characteristics

CHAPTER 4

Greater Sage-Grouse Habitat Characteristics

Abstract. Greater sage-grouse (*Centrocercus urophasianus*) depend on sagebrush (*Artemisia* spp.) for much of their annual food and cover. This close relationship is reflected in the North American distribution of sage-grouse, which is closely aligned with sagebrush, and in particular big sagebrush (*A. tridentata*) and silver sagebrush (*A. cana*). This relationship is perhaps tightest in the late autumn, winter, and early spring when sage-grouse are dependent on sagebrush for both food and cover. However, sage-grouse also depend on sagebrush at other times of year, primarily for protective cover, such as for nests during the breeding season. Other habitat characteristics may be less overtly important than sagebrush, but may be nearly as important. For example, herbaceous cover may provide both food and cover during the nesting and early brood-rearing seasons, thus playing a major role in the population dynamics of sage-grouse.

Introduction

Sage-grouse are closely allied with the large, woody sagebrushes of western North America and depend on these for food and cover during all periods of the year (Patterson 1952, Connelly et al. 2000*a*). Due to sage-grouse dependence on sagebrush habitats they are considered a sagebrush obligate (Braun et al. 1976). Sagebrush habitats across the range of sage-grouse may vary considerably (Tisdale and Hironaka 1981, West and Young 2000), and the specific habitat components used by the species can vary due to biotic and abiotic factors. Large, woody species of sagebrush including big sagebrush, silver sagebrush, and threetip sagebrush (*A. tripartita*) are used by sage-grouse throughout the year in all seasonal habitats (Griner 1939, Patterson 1952, Dalke et al. 1963). Other species of sagebrush such as low sagebrush (*A. arbuscula*) and black sagebrush (*A. nova*) provide important seasonal habitat components during spring and winter (Griner 1939, Patterson 1952, Dalke et al. 1963). Other shrub species such as rabbitbrush (*Chrysothamnus spp*.), antelope bitterbrush (*Purshia tridentata*), and horsebrush (*Tetradymia canescans*) have also been used for nesting and hiding cover by sage-grouse (Patterson 1952, Dalke et al. 1963, Connelly et al. 1991).

Summer habitats used by sage-grouse include riparian and upland meadows and sagebrush grasslands (Griner 1939, Patterson 1952, Dalke et al. 1963). Sage-grouse have also been documented using a variety of human-modified habitats, such as irrigated and non-irrigated croplands and pasturelands (Patterson 1952, Sime 1991). Disturbed areas such as roads, plowed fields, gravel pits, and stock ponds have been used as lekking sites (Patterson 1952, Connelly et al. 1981, Gates 1985). The value of these modified habitats to sage-grouse depends upon the usefulness of the habitat and the juxtaposition of the modified habitat in relationship to adjacent sagebrush habitats (Patterson 1952, Sime 1991). Although we attempt to provide comparable measures of seasonal habitats in the following examination, it should be noted that habitat values can depend on the techniques used to examine them (Connelly et al. 2003). Similar, greater sage-grouse have not been studied in detail in all portions of their range (e.g. North and South

Dakota). Consequently, care should be taken when extrapolating observations for range-wide considerations.

Breeding Habitats

Sage-grouse breeding habitats are defined as those where lek attendance, nesting, and early brood-rearing occur (Connelly et al. 2000*a*, Connelly et al. 2003). These habitats are sagebrush-dominated rangelands, typically consisting of large, relatively contiguous sagebrush stands, and are critical for survival of sage-grouse populations (Connelly et al. 2000*a*, Leonard et al. 2000). The following discussion includes information on habitat selection and relevant functions of the three components of breeding habitats (lekking, nesting, and early brood-rearing).

Leks

General Description. Leks are a traditional courtship display and mating areas attended by sage-grouse in or adjacent to sagebrush dominated nesting habitat (Patterson 1952, Wakkinen et al. 1992). Sage-grouse are polygamous and exhibit consistent breeding behavior each year on ancestral strutting grounds (leks; Patterson 1952, Wiley 1978). Scott (1942) reported a lek as active in 1940; this lek was still active 28 years later (Wiley 1973). Leks are situated in relatively open areas with less herbaceous and shrub cover than surrounding areas (Dingman 1980, Klott and Lindzey 1990). Wiley (1973) reported that selection of leks by sage-grouse occurs at course and fine resolutions. At course resolutions, leks appear to be located in sparser shrubby vegetation (Wiley 1973). At fine resolutions, male sage-grouse choose sod-forming grasses or bare ground for display. Lek selection at both resolutions increased the displaying males' conspicuousness and freedom of movement (Wiley 1973:103). Lek habitat is not considered limiting to sage-grouse populations (Schroeder et al. 1999).

Leks may be natural openings within sagebrush communities or openings created by human disturbances, including dry stream channels, edges of stock ponds, ridges, grassy meadows, burned areas, gravel pits, sheep bedding grounds, plowed fields, and roads (Patterson 1952, Dalke et al. 1963, Rogers 1964, Connelly et al. 1981, Hofmann 1991). Leks are typically surrounded by potential nesting habitat, and are adjacent to relatively dense sagebrush stands (Wakkinen et al. 1992). These sagebrush stands are used for escape, thermal, and feeding cover (Patterson 1952, Gill 1965).

Gentle terrain is a common characteristic of leks (Rogers 1964), as is their location in valley bottoms or draws (Patterson 1952, Rogers 1964). Nisbet et al. (1983) developed a lek preference model that included slope (<10%), precipitation (>25 cm), distance to nearest water source (<2,000 m), and predicted encroachment by pinyon (*Pinus spp*.)-juniper (*Juniperus spp*.) woodlands. Rogers (1964) reviewed characteristics of 120 leks throughout Colorado during 1953-1961 and found that, on average, 50% were in sagebrush; 54% were on gentle slopes; 55% were in bottoms; only 5% were within 200 m of a building; and that although 42% were >1.6 km from an improved road, 26% were within 100 m of a county or state highway. Nisbet el al. (1983) reported 41 leks in Nevada and Utah were preferentially located in black sagebrush habitats (based on use versus availability). Smith (2003) reported that sagebrush within 1.5 km of active leks in North and South Dakota was taller than sagebrush around inactive leks. In addition, he found that sagebrush density, forb cover, and bare ground were greater around (i.e., within 1.5 km of) active leks in North Dakota than around inactive leks. Petersen (1980) reported mating areas (arenas) within leks in North Park, Colorado, had an average canopy cover of only 7.3% and a mean vegetation height of 5.3 cm; sagebrush species present included big sagebrush, alkali sagebrush (*A. longiloba*), and black sagebrush.

Specific Description. Dalke et al. (1963) and Klebenow (1973) documented leks on open or cleared areas 0.04 to 4 ha in size in southeastern Idaho. Scott (1942) observed leks that generally ranged from 0.25-16 ha, but recorded one lek that was 20 ha and with 400 strutting males. Hofmann (1991) reported a mean size of 36 ha for the four largest leks in a study in central Washington.

In non-migratory populations, leks may occur near the center of seasonal ranges (Eng and Schladweiler 1972, Wallestad and Pyrah 1974, Wallestad and Schladweiler 1974). Migratory populations typically do not exhibit this pattern (Dalke et al. 1963, Wakkinen et al. 1992). Travel by females dispersing between wintering and nesting areas, rather than vegetation type, may influence lek locations in some instances (Bradbury et al. 1989, Gibson 1996).

Leks often occur in complexes, composed of the primary lek and one or more satellites. Satellite lek attendance fluctuates depending upon population size, and satellite leks may not be used in years when populations are low (Dalke et al. 1963). In a study of 31 leks in Idaho, mean interlek distance (i.e., distance between nearest-neighbor leks) was about 1.6 km (Wakkinen et al. 1992). Of 13 leks examined in the Upper Snake River Plain in Idaho, 10 were in threetip sagebrush (Klebenow 1969). For two of these leks, interlek distance was 0.8 km, and for eight others, 2.4 km. In Wyoming, lek density averaged 6.8 leks per 100 km^2 ($n = 29$) within a water-reclamation project area, and 8.4 leks per 100 km^2 ($n = 18$) in nearby, undeveloped sagebrush habitats (Patterson 1952). In a non-peer reviewed report, Willis et al. (1993) reported similar lek densities in Oregon and found 4.3 leks per 100 km^2 at Hart Mountain National Antelope Refuge and 4.7 leks per 100 km^2 at Jackass Creek.

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 0.5-0.8 km from the lek (Ellis et el. 1989), 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). 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).

Ellis et al. (1989) found in daytime, male sage-grouse in northeastern Utah used areas near leks that had comparatively greater canopy cover (mean = 31%) and taller shrubs (mean = 53 cm) than did nearby non-use areas. Minimum core day-use areas of males were 0.25 km^2 in size, and the birds often walked to such sites from leks for feeding and loafing (Ellis et al. 1989).

Relevant Features. Leks are typically adjacent to sagebrush with adequate cover for nesting hens as well as protection for both sexes from predators (Patterson 1952). Leks are characterized by low, sparse vegetation and higher amounts of bare ground than adjacent sites (Scott 1942, Petersen 1980, Klebenow 1985, Bradbury et al. 1989). The most important characteristic for leks may be their proximity and configuration with nesting habitat for females, consistent with theories of lek evolution and mating behavior (Gibson 1996).

Nesting

General Description. Sage-grouse nesting habitat is often a broad area within or adjacent to winter range or between winter and summer range (Klebenow 1969, Wakkinen 1990, Fischer 1994). Productive nesting habitat includes sagebrush with horizontal and vertical structural diversity (Wakkinen 1990, Gregg 1991, Schroeder et al 1999, Connelly et al 2000*a*). The understory of productive nesting habitat should be composed of native grasses and forbs that provide a food source of insects, concealment of the nest and hen, and herbaceous forage for pre-laying and nesting hens (Gregg 1991, Schroeder et al 1999, Connelly et al 2000*a*).

Sage-grouse females move into the vicinity of their nest location within a few days of being bred, and remain relatively sedentary until they nest (Patterson 1952). Spring is a period when birds are changing diets from sagebrush to forbs as forbs become available (Barnett and Crawford 1994). Forbs provide increased levels of calcium, phosphorus, and protein that may affect nest initiation rate, clutch size and reproductive rates (Barnett and Crawford 1994, Coggins 1998). Little information is available documenting pre-nesting habitat selection.

Sage-grouse nest in many different sagebrush-dominated cover types and most nests are located under sagebrush plants (Patterson 1952, Gill 1965, Wallestad and Pyrah 1974, Petersen 1980, Drut at al. 1994*a*, Gregg et al. 1994, Sveum at al. 1998*b*). Throughout Wyoming, between 92% (Patterson 1952) and 100% (Rothenmaier 1979, Holloran 1999) of nests were under sagebrush, 90% of nests were located under silver sagebrush plants in southern Canada (Aldridge and Brigham 2002), and 94% of nests were under big sagebrush plants in northern Colorado (Petersen 1980). In southeastern Idaho, Connelly et al. (1991) reported that 21% of sage-grouse hens nested under shrub species other than sagebrush. Popham and Gutiérrez (2003) reported that 41% of sage-grouse nests in California were located under shrubs other than big sagebrush. Other plants that sage-grouse nest under include greasewood (*Sarcobatus vermiculatus*), bitterbrush, rabbitbrush, horsebrush, snowberry (*Symphoricarpos spp.*), shadscale (*Atriplex confertifolia*), mountain mahogany (*Cercocarpus spp.*), and basin wildrye (*Leymus cinereus*; Patterson 1952, Klebenow 1969, Wakkinen 1990, Connelly et al. 1991, Popham and Gutiérrez 2003). Patterson (1952) also located nests on bare ground with no sagebrush overstory.

Throughout the range of studied sage-grouse populations, nests were consistently located under larger bushes (Wakkinen 1990, Gregg 1991, Fischer 1994, Delong et al. 1995, Holloran 1999) with more obstructing cover (Wakkinen 1990, Fischer 1994, Popham and Gutiérrez 2003) within shrub patches. In addition, selected nesting habitat had more sagebrush canopy cover (Klebenow 1969, Fischer 1994, Sveum et al. 1998*b*, Holloran 1999, Aldridge and Brigham 2002) and taller sagebrush (Wallestad and Pyrah 1974, Sveum et al. 1998*b*, Holloran 1999, Lyon 2000, Slater 2003) compared to available habitats. Other relatively consistent differences between selected nesting sites and randomly selected sites included: higher sagebrush density (Klebenow 1969, Holloran 1999, Aldridge and Brigham 2002), taller live and residual grasses (Wakkinen 1990, Holloran 1999), more live and residual grass cover (Klebenow 1969, Lyon 2000), and less bare ground (Sveum et al. 1998*b*, Lyon 2000, Slater 2003).

Additionally, increased spring forb cover (14.5–18.2% vs. 6.8–12.8%), food forb cover (3.1–5.6% vs. 0.5–1.9%), and tall (>18 cm) grass cover (4.7–17.2% vs. 0.3–4.7%) was correlated with increased overall nest initiation rates (99 vs. 65%), renesting rates (30 vs. 14%), and nesting success rates (37 vs. 22%) in southeastern Oregon (Coggins 1998). Mean distance from lek-of-capture to selected nest sites was 4.6 km in southeastern Idaho (Wakkinen et al. 1992) and 8.6 km (range 0.4 – 63.8 km) in west-central Wyoming (Lyon 2000). In southwestern Wyoming, between 75% and 87% of nests were located within 5 km of the lek-of-capture (Slater 2003). In central Montana, 68% of hens nested within 2.5 km of the lek-of-capture, but 2 hens nested >4.8 km (Wallestad and Pyrah 1974). No differences were found between nest-to-lek versus random point-to-lek distances in southeastern Idaho, suggesting nests in this study were placed without regard to lek location (Wakkinen et al. 1992).

Measurement of distances between consecutive-year nests (females followed through consecutive nesting seasons) suggests female fidelity to nesting areas. Mean distance between consecutive-year nests averaged 552 m in southwestern Wyoming (Berry and Eng 1985), 710 m in central Wyoming (Holloran 1999), and 683 m in west-central Wyoming (Lyon 2000). Median distance between consecutive-year nests was 740 m in southeastern Idaho (Fischer et al. 1993), and 67% of consecutive-year nests were <600 m apart in southwestern Wyoming (Slater 2003).

Specific Description. Sage-grouse hens in southeastern Idaho nested under taller bushes with a larger area and greater lateral obstructing cover compared to random sites within the same shrub patch (Wakkinen 1990). Fischer (1994) continued Wakkinen's study for an additional 3 years and indicated that nests were located in areas with increased nest bush total area and height, ground obstructing cover, lateral obstructing cover, sagebrush density of shrubs ≥40 cm tall, and total shrub canopy cover compared to dependent random sites (i.e., random locations between 40 and 200 m from the nest; Fischer 1994). Mean height of nest bush (46.4 cm) was greater than the mean height of shrubs in the surrounding area (Holloran 1999). Additionally, although presented in non-peer reviewed reports, nest locations in western (Heath et al. 1997) and south-central (Heath et al 1998) Wyoming had taller live and residual grasses, more residual grass cover, and less bare ground within 2.5 m compared to plots between 50 and 200 m from the nest but located within the same sagebrush stand.

Selection of specific habitat features, such as sagebrush height and canopy cover within a landscape by nesting sage-grouse has been extensively documented. Connelly et al. (2000*a*) suggested that nesting habitat within sagebrush stands should contain between 15 and 25% canopy cover. Females preferentially selected areas with sagebrush 36 to 63.5 cm tall and with canopies 15 to 50% for nesting in Utah (Rasmussen and Griner 1938. Rothenmaier (1979) reported that mean sagebrush canopy cover was 21.6% (12–29%) and average sagebrush height was 30.6 cm (22.6–38.1 cm) within a 37.2-m^2 plot surrounding nests in southeastern Wyoming. In western Wyoming, 83% of nests were under bushes between 25 and 51 cm tall (average nest bush height 35.6 cm; Patterson 1952). In central Montana, all nests were located in areas with >15% sagebrush canopy cover (Wallestad and Pyrah 1974). Petersen (1980) reported sagebrush height and canopy cover was 32 cm and 24% within 15 m of nests in northern Colorado.

In southeastern Idaho, nests within a threetip sagebrush type were found in areas with higher big sagebrush density (13.3 vs. 1.6 plants/122-m2), basal area of grasses (3.7 vs. 2.9%), and threetip sagebrush canopy cover (14.1 vs. 12.5%) compared to random plots within the same habitat type (Klebenow 1969). Overall, total shrub canopy cover was greater at nests compared to random locations (18.4 vs. 14.4%; Klebenow 1969). Fischer (1994) reported that nests had higher nest bush total area, ground obstructing cover, lateral obstructing cover, and total shrub canopy cover compared to random sites. In Wyoming, higher total shrub canopy cover and taller live sagebrush occurred in the nest area than at random sites (Holloran 1999, Lyon 2000, Slater 2003).

In south-central Washington, nests were consistently located in areas with more shrub cover at or within 5 m of the nest compared to randomly-selected sites (Sveum at al. 1998*b*). Taller average sagebrush heights (40.4 vs. 23.4 cm) occurred near nests compared to random locations in central Montana (Wallestad and Pyrah 1974). Aldridge and Brigham (2002) reported sagebrush canopy cover was dominant at nest sites (31.9 ± 4.07%) in southern Canada and greater than that at random sites (15.7 ± 2.44%). Sagebrush canopy cover was the only variable that discriminated between nests and random sites (Aldridge and Brigham 2002). Herbaceous differences within 2.5 m of nests compared to random plots in Wyoming included taller residual grasses (Holloran 1999), more live (Lyon 2000) and residual grass cover (Lyon 2000), more total herbaceous (Lyon 2000), non-food forb (Holloran 1999), and total forb (Lyon 2000) covers, and less bare ground (Lyon 2000, Slater 2003). In Idaho, Wakkinen (1990) reported taller grasses occurred near nests compared to random locations. The covers of short grasses (<18 cm) and bare ground were consistently lower, and vertical cover height were consistently greater near nests compared to available sites (Sveum et al. 1998*b*).

Relevant Features. Studies have reported somewhat conflicting results regarding nest success in relation to vegetation at nest sites. Connelly et al. (1991) reported that sage-grouse nesting under sagebrush experience greater nest success (53%) than those nesting under non-sagebrush (22%). Wallestad and Pyrah (1974) reported a significant relationship between nest success and vegetation characteristics in Montana. Successful nests (n = 31) were in areas of higher sagebrush density than unsuccessful nests (n = 10), and canopy cover of sagebrush was greater (27%) in stands with successful nests compared to unsuccessful nests (20%). In southeastern Oregon, the canopy cover of medium height shrubs (40-80 cm) and tall grasses (>18 cm) was higher at successful nests than unsuccessful nests or random sites (Gregg et al. 1994).

Contrasting to Connelly et al. (1991) nesting success for non-sagebrush nests (42%) was higher than sagebrush nests (31%) in California (Popham and Gutiérrez 2003). In southeastern Idaho, Wakkinen (1990) found vegetation characteristics had no relationship to nesting success. Sveum et al. (1998*b*) found no differences in nesting success between nests placed beneath big sagebrush shrubs than under other shrub species in Washington.

Herbaceous vegetation characteristics that were consistently higher at successful versus unsuccessful sage-grouse nests throughout the range of studied populations included live and residual grass height (Wakkinen 1990, Sveum et al. 1998*b*, Aldridge and Brigham 2002, Hausleitner 2003), residual vegetative cover (Gregg et al. 1994, Sveum et al. 1998*b*), forb cover (Holloran 1999, Hausleitner 2003) and visual obstruction (Wakkinen 1990, Popham 2000, Slater 2003). Successful nests in southern Canada had taller grasses and palatable forbs, and less grass cover compared to unsuccessful nests (Aldridge and Brigham 2002). In California, percent rock cover (rocks >10 cm in diameter; 27.7 vs. 14.5%), total shrub height (65.5 cm vs. 49.2 cm), and visual obstruction (40.2 vs. 32.5 cm) were greater at successful than unsuccessful nest sites (Popham and Gutiérrez 2000). Nests destroyed by avian predators in southwestern Wyoming consistently had less overhead cover (live sagebrush and total shrub canopy cover) within 15 m of the nest and increased lateral cover (herbaceous cover and height) within 2.5 m of the nest compared to nests in general and mammalian-destroyed nests (Slater 2003). Hausleitner (2003) reported that successful nests in northwestern Colorado had higher average forb (9.3 vs. 7.2%) and grass cover (4.8 vs. 3.9%) within 10 m of the nest, and taller grasses at the nest (15.4 vs. 11.7 cm) and at 1 m from the nest (18.2 vs. 13.5 cm) compared to unsuccessful nests. Additionally, in southeastern Idaho, successful nests tended to have taller grass and more lateral obstructing cover within 2.5 m of the nest compared to unsuccessful nests (Wakkinen 1990). In central Wyoming, food-forb cover within 2.5 m tended to be higher at successful nests relative to unsuccessful nests (2.1 vs. 1.3%; Holloran 1999).

Successful artificial nests placed between 800 and 1440 m from active and inactive sage-grouse leks in southern Canada consistently had more forb and total sagebrush canopy cover, taller grasses, and lower numbers of sagebrush plants within 0.5 m compared to unsuccessful artificial nests (Watters et al. 2002). DeLong et al. (1995) reported that a combination of greater amounts of tall (>18 cm) grass and medium height (40-80 cm) shrub cover within 1 m of artificial sage-grouse nests in southeastern Oregon increased the probability of success. However, in northwestern Utah, the proportion of artificial nests along 1.6-km transects radiating perpendicularly from active sage-grouse leks that were destroyed increased with more horizontal cover (density board centered at artificial nest and read from 10 m), and more herbaceous cover and taller sagebrush within 20 m of artificial nests (Ritchie et al. 1994). Although no relationship between lek-of-capture to nest distances and nesting success were reported in southeastern Idaho (Wakkinen et al. 1992), Popham and Gutiérrez (2003) found that mean distance from lek to nest site was greater for successful than unsuccessful nests in California (3.6 versus 2.0 km, respectively). High nest densities surrounding a lek may result in increased predation through destruction of multiple nests in a given area by a single predator (Neimuth 1992, Popham 2000).

Early Brood-rearing

General Description. Early brood-rearing habitat is defined as sagebrush habitat within the vicinity of the nest used by sage-grouse hens with chicks up to 3 weeks following hatch (Connelly et al. 2000*a*). Compared to selected nesting habitat, early brood-rearing locations in central Wyoming had less live sagebrush (15.8 vs. 25.4%) and total shrub (19.3 vs. 30.5%) canopy cover, shorter average sagebrush heights (25.5 vs. 31.4 cm), and more total herbaceous (37.3 vs. 29.6%) cover (Holloran 1999). Additionally, total forb (9.3 vs. 7.3%), food-forb (3.6 vs. 1.8%) and bare ground (7.3 vs. 5.0%) cover tended to be higher at selected early brood-rearing than nesting habitat (Holloran 1999).

Specific Description. Hens rear their broods for the first 2-3 weeks in the vicinity of their nest (Berry and Eng 1985). Early brood-rearing areas were located 0.2-5.0 km (mean = 1.1 km) from the nest in west-central Wyoming (Lyon 2000). Slater (2003) found 80% of early brood locations were within 1.5 km of the nest in southwest Wyoming. Movements from nest to early brooding areas in northern Colorado were between 0.3 and 2.3 km (mean = 0.8 km; Petersen 1980). 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 habitats (Peterson 1970). Between 75-80% of brood locations from June 1 through June 15 were in areas with 1-25% sagebrush canopy cover (Wallestad 1971). In south-central Wyoming, 68% of sage-grouse brood locations were in sagebrush-grass or sagebrush-bitterbrush habitats (Klott and Lindzey 1990). Brooding females during the early brood-rearing stages in south-central Washington preferentially selected for big sagebrush-bunchgrass habitats and against grassland habitats (areas devoid of sagebrush); 70% of locations were within big sagebrush-bunchgrass habitats (Sveum et al. 1998*a*).

Brood-use sites within big sagebrush-dominated habitats in southeastern Idaho had lower big sagebrush density (64 vs. 104 plants) and canopy cover (8.5 vs. 14.3%), and higher percent frequency of yarrow (*Achillea millefolium*; 23.5 vs. 9.4%), lupine (*Lupinus caudatus*; 18.3 vs. 7.5%), dandelion (*Taraxacum officinale*; 12.0 vs. 3.1%) and common salsify (*Tragopogon dubius*; 2.2 vs. 0.3%) compared to random locations within the same vegetation type (Klebenow 1969). A combination of more residual grass and total forb cover, and shorter effective vegetation height were the best predictors of early brood-rearing use compared to available habitats in central Wyoming (Holloran 1999). Early brood-rearing locations had less live sagebrush (15.8 vs. 20.2%) and total shrub (19.3 vs. 24.1%) canopy cover, more residual grass (2.9 vs. 2.0%), total forb (9.3 vs. 6.6%), and total herbaceous (37.3 vs. 29.4%) cover, relative to available habitats (Holloran 1999). In west-central Wyoming, early brood-rearing locations had less live sagebrush density (1.9 vs. 2.3 plants/m^2), live sagebrush (21.5 vs. 27.0%), total shrub canopy cover (30.0 vs. 35.0%), and bare ground (23.5 vs. 39.6%) compared to available habitat (Lyon 2000). Lyon (2000) also found more total herbaceous (24.8 vs. 9.1%) cover compared to available habitat. Early brood-rearing locations had more sagebrush cover compared to random locations (8.7 vs. 4.5%) in southern Canada (Aldridge and Brigham 2002). Total forb (25 vs. 8%) and food forb (8 vs. 2%) cover were higher, and residual herbaceous cover (1 vs. 3%) and height (1 vs. 3 cm) were lower within 10 m of early brooding areas compared to random locations in south-central Washington (Sveum et al. 1998*a*).

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 spp*.), yarrow, and common salsify were more abundant at sage-grouse brooding sites than at random sites (Klott and Lindzey 1990). In southeastern Oregon, key forbs (those occurring in the crops of at least 10% of collected chicks or having aggregate mass ≥1%) cover (4 vs. 1%) were higher in habitats preferentially selected by broods relative to habitats selected in less than available proportions (Drut et al. 1994*b*). In southeastern Idaho, Fischer (1994) reported higher Hymenoptera (ants, bees, wasps) abundance and higher Orthoptera (grasshoppers, crickets) frequency (no difference in abundance), but no difference in abundance of Coleoptera (beetles) at brood use vs. random sites. However, Slater (2003) reported no vegetative or insect quality differences between selected early brood-rearing and random locations in Wyoming. During the early stages of life, sage-grouse broods consistently selected areas with more forb (Klebenow 1969, Klott and Lindzey 1990, Sveum et al. 1998*a*, Holloran 1999) and total herbaceous (Holloran 1999, Lyon 2000) cover, and less shrub canopy cover (Klebenow 1969, Holloran 1999, Lyon 2000) than at randomly-selected areas.

Relevant Features. The availability of forb-rich habitats in close proximity to adequate protective cover appears to be an important consideration in brood habitat (Kebenow 1969, Sveum et al. 1998*a*, Klott and Lindzey 1990, Holloran 1999, Lyon 2000). These habitat features appear to be related to the selection of food items by chicks, particularly forbs (Drut et al. 1994*b*) and insects (Fischer 1994). The literature is somewhat ambiguous concerning the management of these specific habitat types.

Summer and Late Brood-Rearing Habitats

General Description

Late brood-rearing habitats are those habitats used by sage-grouse following desiccation of herbaceous vegetation in sagebrush uplands (Klebenow and Gray 1968, Savage 1969, Fischer et al. 1996*b*). Gates (1983) and Connelly et al. (1988) observed sage-grouse associated with agricultural lands and irrigated lawns during the summer period. Sage-grouse often use sagebrush habitats for late brood-rearing throughout the summer but select habitats based on availability of forbs. This is often accomplished by moving up in elevation or selecting sites where moisture collects and maintains forbs throughout the summer (Martin 1970, Wallestad 1971, Fischer et al. 1996*b*, Hausleitner 2003). Fischer et al. (1996*b*) found that sage-grouse moved to late brood-rearing habitats when vegetal moisture was ≤60%. The beginning of late brood-rearing also coincides with the change in diets of sage-grouse chicks from predominantly insects to forbs (Patterson 1952, Klebenow and Gray 1968, Klebenow 1969, Peterson 1970, Drut et al. 1994). These habitats are generally used from July to early September but vary annually due to annual weather conditions (Patterson 1952, Dalke et al. 1963, Gill and Glover 1965, Savage 1969, Wallestad 1971, Connelly et al. 1988).

Sage-grouse use a variety of sagebrush habitats and other habitats (e.g., riparian, wet meadows and alfalfa [*Medicago spp*.] fields) during summer. These sites typically provide an abundance of forbs and insects for hens and chicks (Schroeder et al. 1999, Connelly et al. 2000*a*). As vegetation in upland sagebrush habitats desiccate, hens move to more mesic sites to summer and rear broods (Klebenow 1969, Gates 1983, Connelly et al. 1988, Fischer et al 1996b). These movements vary in response to such factors as plant moisture, vegetal cover, and elevation (Dalke et al. 1960, Wallestad et al. 1975, Connelly 1982). Sage-grouse in southeastern Idaho moved as far as 82 km from breeding and nesting to summer ranges (Connelly et al. 1988). Klebenow and Gray (1968) observed grouse migrating as far as 8-24 km to summer ranges at higher elevations ranging from 1,600 m to over 2,150 m. Fisher et al. (1997) recorded movements up to 62 km from nesting to summer habitats. Wallestad (1971) reported that some broods only traveled short distances to summer habitats, whereas others moved as much as 5 km.

Sage-grouse movements to breeding, nesting and summer ranges may also be influenced by tradition. For instance, Fisher et al. (1997) reported significantly more sage-grouse than expected moved to traditional summer grounds, rather than to closer (15-20 km) irrigated agricultural fields. Wallestad (1971) also observed hens moving 5 km to summer habitat, bypassing a comparable area that was 3.2 km closer.

Unsuccessful hens and cocks move from sagebrush habitat as forbs desiccate and will occupy a variety of habitats during the summer, including irrigated hayfields and wet meadows that are adjacent to sagebrush habitats (Gates 1983, Connelly et al. 1988). Connelly et al. (1988) and Gregg et al. (1993) reported that movements of broodless hens to mesic areas in the summer preceded that of brooding hens' arrival and that flocks were generally segregated by sex. Segregated flocks were also observed by Dalke et al. (1963) during the summer.

Seasonal Differences

Sage-grouse use many different habitats during the late brood-rearing period, such as sagebrush, wet meadows, and irrigated farmland adjacent to sagebrush habitats (Gates 1983, Connelly et al. 1988). Klebenow (1969) reported broods in Idaho typically move up in elevation, following the gradient of food availability. Wallestad (1971) observed that some broods remained in the sagebrush by seeking out microhabitats such as small swales or ditches where forbs were still available. A lack of shift in habitat selection between early and late brood-rearing may also suggest there is no difference in the availability of forbs in the area (Aldridge and Brigham 2002). Aldridge (2000) suggested that broods do not move from sagebrush uplands to more mesic sites during wet years, and that wetland complexes may be limiting in dry years because of low food availability, and ultimately, low recruitment.

Specific Description

Adult and juvenile sage-grouse tend to use sagebrush adjacent to mesic areas during summer as loafing sites and for cover (Savage 1969). Midday locations had greater shrub cover and height compared to morning and afternoon loafing locations (Sveum et al. 1998*a*). Dunn and Braun (1986) reported grouse select feeding habitat near edges of cover types with more horizontal and vertical cover and less variation in shrub densities and size compared to random sites. Hens with broods also used sites with more horizontal cover and greater variation in sagebrush canopy cover than random sites to roost, but fed in open homogeneous areas during the morning and afternoon periods (Braun 1986). In Colorado, Hausleitner (2003) found that female night-roost sites had less bare ground and visual obstruction, but greater forb cover than at random sites. These areas may increase the opportunity of foraging by hens with broods with high energic demands and provide open cover types with greater escape potential from predators (Hausleitner 2003). Sveum et al. (1998*a*) found that morning and afternoon locations differed from midday and random locations by having taller (≥18 cm) grass and less shrub cover and height. Klott and Lindzey (1990) reported broods used large openings and meadows, foraged on the edges and avoided the centers.

Peterson (1970) found that forb canopy cover averaged 33% at brooding sites in Montana over 2 distinctly different summers (in terms of precipitation). In Colorado, young broods used areas with low forb canopy cover (mean = 6.9%) after hatching and then quickly moved to wet meadows with far greater (mean = 41.3%) forb canopy cover (Schoenberg 1982). Hausleitner (2003) reported females selected brood-rearing sites with higher average forb canopy cover (8% vs. 4%) and less bare ground than random sites. Sveum et al. (1998*a*), reported that hens selected areas with 19-27% forb canopy cover for late brood-rearing in Washington. In Idaho, Apa (1998) found sites used by sage-grouse broods had twice as much forb cover as did independent sites. However, researchers in southeast Alberta recorded an average forb cover in late brood-rearing habitat of 12.6% and suggested that 12-14% forb canopy cover might represent the minimum needed for brood habitat (Aldridge and Brigham 2002).

Sage-grouse chicks consume a wide variety of forbs and insects depending on availability. Drut et al. (1994*a*) found that chicks consumed 122 different foods, which included 34 genera of forbs, 2 genera of shrubs and 1 genus of grass, and 41 families of invertebrates.

Food availability appears to be a strong determinant of which vegetation types are selected by broods during different periods of the summer (Wallestad 1971).

Relevant Features

The availability and use of forbs in summer habitats by sage-grouse has been reported by many investigators (Patterson 1952, Peterson 1970, Gregg et al. 1993, Apa 1998, Sveum et al. 1998*a*, Aldridge and Brigham 2002, Hausleitner 2003). Juvenile sage-grouse rely heavily on animal matter (insects) and forbs for food during the first few months after hatching (Patterson 1952, Klebenow and Gray 1968, Braun et al. 1977). Succulent forbs dominate the diet of chicks from about 2 weeks of age (Nelson 1955, Klebenow and Gray 1968) until 3 months, when sagebrush then becomes the primary food component (Peterson 1970). Coggins (1998) reported an increase in forb availability may allow hens to remain in upland brood-rearing habitats longer which could contribute to increased chick survival due to decreased brood movements.

Patterson (1952) implied that water was important to sage-grouse and suggested that its availability could affect summer distributions. However, although theorized in a non-peer reviewed report, movements to agricultural lands or high elevation summer ranges are probably in response to lack of succulent forbs in an area rather than a lack of free water (Connelly and Doughty 1989). It has further been suggested that grouse do not commonly use water developments even during relatively dry years, but instead obtain moisture from consuming succulent vegetation (Connelly 1982). Water developments tend to attract other animals and thus may serve as a predator "sink" for sage-grouse (Connelly and Doughty 1989). Free water reservoirs can, however, provide islands of succulent vegetation (Wallestad 1971).

Autumn Habitats

Autumn is a transitional period for sage-grouse (Wambolt et al. 2002), when their 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). Autumn 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).

During early autumn, in addition to sagebrush, habitats may include upland meadows, riparian areas, greasewood bottoms, alfalfa fields, and irrigated native hay pastures (Patterson 1952, Gill 1965, Savage 1969, Wallestad 1971, Connelly 1982). As vegetation in these habitats desiccates or is killed by frost, sage-grouse begin using sagebrush habitats more often and form larger flocks (Patterson 1952, Savage 1969). During early autumn 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). During a seven-year study in eastern Idaho, sage-grouse gathered in large flocks near water during the autumn migration, watering from 10 to 30 minutes daily (Dalke et al. 1963).

Autumn habitats used by sage-grouse in northeastern Wyoming supported higher densities of sagebrush (3.1-7.4 plants/m^2) than the study area as a whole (Postovit 1981). Wallestad (1971) found that sage-grouse used habitats with greater sagebrush cover in the autumn than during the late brood-rearing period. This shift coincided with the transition to a diet of sagebrush, as sage-grouse broods that had occupied bottomland vegetation types (greasewood and alfalfa fields) shifted back into sagebrush in late August and September (Wallestad 1971). Connelly and Markham (1983) reported that some sage-grouse did not return to sagebrush habitats until October or November. During the autumn in Colorado, sage-grouse used the same upland sagebrush habitats used for breeding; however, their use in the autumn appeared random, and not tied to lek location, as it was during the breeding season (Gill 1965).

In Idaho, movements from autumn 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 autumn may return to sites adjoining late brood-rearing habitat (Patterson 1952). Sage-grouse in Utah typically moved to winter range around mid-November; this movement appeared to be independent of snow depth (Welch et al. 1990).

Winter Habitats

General Description

Winter habitats of sage-grouse are dominated by sagebrush that provides shelter and food during this time of the year (Rassmussen and Griner 1938, Patterson 1952, Remington and Braun 1985, Robertson 1991). Variation in topography and availability of sagebrush above the snow under various conditions determine the location of these habitats (Beck 1977, Connelly 1982, Robertson 1991).

Sage-grouse habitat selection during winter is influenced by several factors, including snow depth and hardness, topography (e.g., elevation, slope, and aspect), and vegetation height and cover (Gill 1965, Schoenberg 1982, Robertson 1991, Schroeder et al. 1999). In North Park, Colorado sage-grouse selected either relatively exposed, windswept ridges or draws and swales (Beck 1977, Schoenberg 1982). Both windswept ridges and draws provided access to sagebrush above snow for food and cover (Beck 1977, Schoenberg 1982).

Fidelity to winter areas has not been well studied, although some evidence of fidelity to winter areas among years has been demonstrated in Washington (Schroeder et al. 1999) and Wyoming (Berry and Eng 1985). In Utah, Welch et al. (1990) found that sage-grouse showed less fidelity to winter range than to other seasonal ranges.

Winter habitats of sage-grouse generally are dominated by big sagebrush; however, low sagebrush and silver sagebrush communities also are used during winter (Schroeder et al. 1999, Crawford et al. 2004). Sage-grouse in Idaho and Nevada often use low sagebrush habitats while other sagebrush-dominated habitats are used in proportion to their availability (Connelly 1982, Klebenow 1985). However, in Oregon, 98% of winter observations were in mountain big sagebrush (Hanf et al. 1994).

Specific Description

During winter, sage-grouse rely almost exclusively on sagebrush exposed above snow for forage, (Patterson 1952, Schroeder et al. 1999, Connelly et al. 2000*a*, Crawford et al. 2004). In central Montana, sage-grouse foraged during winter in big sagebrush with a mean canopy cover of 28%, and observations in dense (>20%) cover were more common than those in less dense sagebrush (Eng and Schladweiler 1972).

During winter, sage-grouse may roost in snow burrows or snow forms, apparently for energy conservation (Beck 1977, Back et al. 1987). In Montana, winter roost sites were in sagebrush with a mean canopy cover of 26%, and usually on flat terrain (Eng and Schladweiler 1972). In Colorado, characteristics (e.g., shrub height, percent slope) of winter feeding-loafing sites (n = 173) did not differ from roosting sites (n = 26) (Beck 1977).

Relevant Features

The spatial distribution of sage-grouse in winter often is related to snow depth (Patterson 1952; Dalke et al. 1963; Gill 1965; Klebenow 1973, 1985; Beck 1975, 1977; Welch et al. 1990). At the onset of winter, sage-grouse typically move to lower elevations with greater exposure of sagebrush above snow (Patterson 1952) and taller sagebrush; in migratory populations, this movement may extend up to 160 km (Patterson 1952). During more severe winters, a large proportion of the sagebrush may be beneath snow and thus unavailable for roosting or foraging.

Shrub density and structure, including height and canopy cover, also influence habitat selection by sage-grouse during winter. Connelly et al. (2000*a*) recommended that canopy cover of sagebrush in both arid and mesic sites should be maintained at 10 to 30% in wintering habitat and further reported that grouse use shrub heights of 25-35 cm above the snow. In Colorado, female sage-grouse used more dense (68 plants/0.004 ha) stands of mountain big sagebrush (*primarily A. t. vaseyana*) than did males (46 plants/0.004 ha; Beck 1977). Height of sagebrush on winter ranges is typically 25-80 cm (Crawford et al. 2004). Schoenberg (1982) found that sage-grouse selected wintering areas having greater sagebrush cover than at random sites and sagebrush heights were 2-3 times greater at use verses random sites. Connelly (1982) reported total height of sagebrush at winter use sites by sage-grouse was greater than at random sites, and provided evidence suggesting that sage-grouse moved to taller sagebrush as snow depth increased.

Shrub canopy cover on winter ranges generally varies from 6-43% (Schroeder et al. 1999). Studies in central Oregon found that sagebrush canopy cover was typically >20% at winter-use sites (Hanf et al. 1994). Within these sites, however, grouse tended to use patches with lower canopy cover (12-16%); in this study, most of the winter use sites were in mountain big sagebrush (Hanf et al. 1994). In central Montana, sage-grouse selected dense (>20% canopy cover) stands of big sagebrush during winter (Eng and Schladweiler 1972), whereas in central Idaho they preferred black sagebrush when these shrubs were available above the snow (Dalke et al. 1963). Robertson (1991) reported Wyoming big sagebrush canopy cover and height were consistently greater at use sites when compared to random sites.

In Utah, Homer et al. (1993) used satellite imagery to classify winter habitat of sage-grouse into seven shrub categories. Wintering grouse preferred shrub habitats with medium to tall (40-60 cm) shrubs and moderate shrub canopy cover (20-30%; Homer et al. 1993). Sage-grouse avoided winter habitats characterized by medium (40-49 cm) shrub height with sparse (<14%) sagebrush canopy cover. Cover of grasses and forbs for wintering habitats generally is irrelevant, because of the nearly complete reliance of sage-grouse upon sagebrush during this period (Homer et al. 1993).

Topography also influences use of winter habitats by sage-grouse. Flocks are typically found on south- or southwest-facing aspects (Beck 1977, Crawford et al. 2004) and on gentle slopes (<5%; Beck 1977). Microsites ameliorate effects of wind, especially at low temperatures (Sherfy and Pekins 1995), and contribute to maintaining energy balance. In eastern Idaho, the mean distance moved between summer and winter ranges was 48.2 km for 28 hens; this movement involved a decrease in mean elevation of 446 m (Hulet 1983).

Landscape Context Issues

General Description

Sage-grouse populations typically inhabit large, interconnect expanses of sagebrush and thus have been characterized as a landscape-scale species (Patterson 1952, Wakkinen 1990). Historically, the distribution of sage-grouse was closely tied to the distribution of the sagebrush ecosystem (Wambolt et al. 2002, Schroeder et al. 2004). However, populations of sage-grouse have been extirpated at places throughout their former range (Schroeder et al. 1999, Wambolt et al. 2002), concomitant with habitat loss and degradation, so that the species' current distribution is less closely aligned with that of sagebrush.

Causes for habitat loss, fragmentation, and degradation in sagebrush are many and varied, and include brush control and other means to remove sagebrush (Klebenow 1970, Martin 1970, Wallestad 1975), inappropriate livestock management, energy development, urbanization, and the infrastructure necessary to maintain these activities (Hulet 1983, Evans 1986, Beck and Mitchell 2000, Bunting et al. 2002, Braun et al. 2002, Lyon and Anderson 2003). Increased fire frequency in lower elevation sagebrush habitats, often closely tied to invasion of annual grasses such as cheatgrass, has resulted in losses of sagebrush over large expanses in the Intermountain West and Great Basin (Mack 1981, Miller et al. 1994, Crawford et al. 2004). In addition, decreased fire frequency in higher elevation sagebrush habitats and impacts from inappropriate livestock grazing and other factors have resulted in conifer encroachment and subsequent reduction of the herbaceous understory and sagebrush canopy cover over large areas (Miller and Rose 1995, Miller and Eddleman 2001, Crawford et al. 2004).

Prescribed fire has also been an issue. Pyle and Crawford (1996) found that prescribed fire in Oregon decreased sagebrush cover, but increased total forb cover and diversity, hypothesizing that prescribed fire may increase forbs in montane sagebrush habitats used for brood-rearing. However, Fischer et al. (1996*a*) found that forb cover was similar in burned and unburned areas of Wyoming big sagebrush in Idaho, but the abundance of hymenoptera was lower in burned habitats. Sage-grouse abundance was not different between burned and unburned areas, and the authors indicated that fire did not enhance sage-grouse brood-rearing habitats (Fischer et al. 1996*a*). Slater (2003) found that sage-grouse were willing to used areas impacted by both prescribed burns and wildfires in Wyoming, but that the area's suitability appeared to be related to age of the burn and the availability of alternate shrubs. Additionally, Nelle et al. (2000) reported that fire had long-term negative impacts on sage-grouse nesting and brood-rearing habitats in mountain big sagebrush stands (*A. t. vaseyana*). Connelly et al. (2000*b*) indicated that prescribed burning during a drought resulted in a large decline of the sage-grouse breeding population. Byrne (2002) documented avoidance of burned habitats by nesting and brood rearing sage-grouse hens in Oregon and concluded that fire provided no apparent value in low sagebrush and Wyoming sagebrush cover types. Finally, in a modeling exercise, Pedersen et al. (2003) warned that although small fires may benefit sage-grouse, large fires occurring at high frequencies may lead to the extinction of sage-grouse populations. They defined a large fire as one that burns >10% of the spring use area and they defined high frequencies as 17 years between fires.

The use of herbicides and insecticides also has the potential to directly and indirectly impact sage-grouse. The impacts can be through direct contact (Ward et al. 1942, Post 1951, Blus et al. 1989) or through the indirect alteration of components of the habitat. These alterations can include the removal of sagebrush (Carr and Gover 1970, Klebenow 1970) and the reduction of forbs or insects (Eng 1952).

Few studies have been conducted to examine landscape-level issues regarding sage-grouse populations and habitats. Leonard et al. (2000) found a negative relationship between mean numbers of males/lek and agricultural development during a 17-year period in the Upper Snake River Plain in Idaho; nearly 30,000 ha of sagebrush in the study area were converted to cropland from 1975 to 1992. In North Park, Colorado, Braun and Beck (1996) examined lek counts in relation to habitat loss from both plowing and spraying with 2,4-D of >28% of the study area, a site known as "one of the best sage-grouse habitats in Colorado." Initial spraying of >1,600 ha occurred in 1965, with an additional 500 ha sprayed and 1,460 ha plowed and seeded during the following 5 years (Braun and Beck 1996). The 5-year mean of males on active leks declined from 765 (1961-1965) to 575 (1971-1975; Braun and Beck 1996). Numbers rebounded by 1976-1980, however, and even exceeded the pre-treatment levels (five-year mean = 1,109 males).

A recent study comparing the percentage of tilled versus non-tilled land surrounding sage-grouse leks in North Dakota revealed that abandoned leks had a higher percentage of tilled lands within a 4-km buffer of leks than did active leks (Smith 2003). However, there was no increase in the percentage of tilled land from the 1970s to the late 1990s, suggesting that if the amount of tilled land was a factor in lek abandonment, this effect had occurred prior to the 1970s (Smith 2003).

Conserving large landscapes with suitable winter habitat may be important for conservation of sage-grouse (Eng and Schladweiler 1972). Sage-grouse in North Park, Colorado concentrated during winter in 7 small areas that totaled only 85 km^2; these areas comprised only 7% of the sagebrush in the entire study area (Beck 1977). Swenson et al. (1987) found marked declines in sage-grouse abundance in Montana when a large (30%) percentage of the winter habitat was plowed, primarily for grain production. Eng and Schladweiler (1972) suggested that sagebrush removal in winter habitats may be especially detrimental because of the relatively long periods that winter habitat may be occupied by sage-grouse annually. Maintaining intact winter habitat for sage-grouse may also be an issue in areas of energy development (e.g., natural gas fields, coal-bed methane), especially if several populations converge in a common wintering area (Lyon 2000).

Fragmenting sagebrush habitats may also change course-resolution distribution patterns of sage-grouse. During a study in Colorado, in which >120 flocks (>3,000 birds total) were observed during 2 winters, only 4 flocks were found in altered (by spraying with 2,4-D, plowing, burning, or seeding) sagebrush habitats, although >30% of the study area had been treated (Beck 1977).

Although sage-grouse are considered a landscape species, conclusive data are unavailable on minimum patch sizes of sagebrush necessary to support viable populations of sage-grouse. In Wyoming, Patterson (1952) found that sage-grouse "packs" could range as widely as several thousand square kilometers. Migratory populations of sage-grouse may use areas exceeding 2,700 km^2 (Connelly et al. 2000*a*, Leonard et al. 2000). Sagebrush patches used by broods averaged 86 ha in early summer (June and July) in central Montana but diminished to 52 ha later in summer (August and September; Wallestad 1971).

Mosaics, Juxtaposition, and Diversity

Sagebrush habitats are generally characterized by the sagebrush overstory, which is, both spatially and temporally diverse due to the large area occupied by the sagebrush ecosystem (Miller and Eddleman 2001, Schroeder et al. 1999). Sage-grouse use of different heights and canopy cover of sagebrush is seasonally, ranging between 29-80 cm in height and 19-38% canopy cover during the nesting season (Gregg 1991, Heath et al. 1997, Apa 1998, Aldridge and Brigham 2002). During other periods of the year (summer, autumn, winter) sagebrush heights range between 25-46 cm, with canopy cover from 12-43% (Eng and Schladweiler 1972, Robertson 1981, Schoenberg 1982, Hanf et al. 1994, Holloran 1999). Within the sagebrush ecosystem, a wide range of understory vegetation is used by sage-grouse during the breeding and brood-rearing periods (Wakkinen 1990, Gregg 1991, Fischer 1994, Holloran 1999, Aldridge and Brigham 2002). Wakkinen (1990) found sage-grouse using habitats for nesting with grass heights averaging 18 cm and grass canopy cover of 3-10% while Aldridge and Brigham (2002) had grass heights averaging 16 cm and canopy coverage averaging 32%. Early brood-rearing habitats reported by Holloran (1999) averaged 19 cm in height and 5% canopy cover for grasses, while Aldridge and Brigham (2002) reported findings of 45 cm and 34% respectively.

Although sage-grouse typically occupy large expanses of sagebrush habitats composed of a diversity of species and subspecies of sagebrush, they may also use a variety of other habitats such as riparian meadows, agricultural lands, steppe dominated by native grasses and forbs, scrub willow (*Salix spp.*), and sagebrush habitats with some conifer or quaking aspen (*Populus tremuloides*) (Patterson 1952, Dalke et al. 1963). These habitats are almost always intermixed in a sagebrush-dominated landscape (Griner 1939, Patterson 1952, Dalke et al. 1963, Savage 1969). Sage-grouse have been observed using habitats altered by man throughout their range. However, the ability of sage-grouse to use these habitats, and their value to sage-grouse in meeting their seasonal habitat requirements, are dependent on the juxtaposition of these habitats in relation to sagebrush. Altered habitats used by sage-grouse include alfalfa, wheat (*Triticum spp*.), crested wheatgrass (*Agropyron cristatum*), potatoes (*Solanum tuberosum*), and other crops (Patterson 1952, Gates 1983, Connelly et al. 1988, Blus et al. 1989, Sime 1991).

Migratory Corridors

Migratory corridors are determined by the relationship between habitat configuration and seasonal movements and habitat requirements of sage-grouse (Dalke et al. 1963, Connelly et al. 1988). These seasonal movements are generally traditional in nature and may occur between 2 or 3 seasonal ranges (Dalke et al. 1963, Beck 1977, Schoenberg 1982, Connelly et al. 1988, Wakkinen 1990, Robertson 1991, Fischer 1994, Connelly et al. 2000*a*). Wakkinen (1990) reported that sage-grouse did not readily change traditional movements in southeastern Idaho.

Differences in techniques used to measure movements of sage-grouse make comparisons among studies, or reporting of average seasonal ranges and migratory movements, difficult (Schroeder et al. 1999). In North Park, Colorado, adult sage-grouse hens moved on average 5.4 km from leks to nest sites, whereas yearling females traveled only 2.3 km (Petersen 1980). Distances moved by female sage-grouse from leks to nests in central Montana were similar between age classes, with adults moving 2.5 km, and yearlings 2.8 km (Wallestad and Pyrah 1974). Among male sage-grouse in Montana, the majority (76%) were within 1 km of their associated leks during the breeding season; however, movements up to 1.3 km from the lek were common (Wallestad and Schladweiler 1974). Daily movements of males from leks to day-use areas in Utah were 0.5 to 0.8 km on average, and core day-use areas were a minimum of 0.25 km^2 (Ellis et al. 1985).

Sage-grouse may move much longer distances between seasonal ranges. Connelly (1982) reported movements of three male sage-grouse from leks to summer habitats with distances ranging from 42 to 50 km. Males (n = 14) in Washington on the Yakima Training Center dispersed an average maximum distance of 15.5 km from the lek (Hofmann 1991). In eastern Idaho, the mean distance moved between summer and winter ranges was 48.2 km for 28 hens; this movement involved a decrease in mean elevation of 446 m (Hulet 1983). Travel of 35 km from a lek to a winter area was recorded in southwestern Wyoming (Berry and Eng 1985). In Idaho, Dalke et al (1963) reported movements of sage-grouse along established routes of 80-160 km, depending upon the severity of winter weather, from winter habitats to leks. Unfortunately, the distribution, configuration, and characteristics of these migration corridors is largely unknown in most portions of the sage-grouse distribution.

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Chapter 5

Sagebrush Ecosystems: Delineation and Dynamics of Primary Sagebrush Habitats

CHAPTER 5

Sagebrush Ecosystems: Dynamics of Primary Sagebrush Habitats

Abstract. We described the general characteristics and dynamics of sagebrush (*Artemisia* spp.) habitats. Our objective was to provide background information on the basic patterns and processes against which the changes in disturbance, influence of invasive species, and effects of land uses presented in Chapter 7 can be assessed. We included all regions of the sagebrush biome in our assessment, including areas not currently occupied by greater sage-grouse (*Centrocercus urophasianus*). Disturbance regimes in the sagebrush biome were spatially and temporally variable across the biome. Sagebrush species were distributed along elevation, precipitation, slope, and salinity gradients in a multivariate ordination conducted from field samples collected throughout the biome. Elevation and precipitation differences generally held within regions among the primary subspecies of big sagebrush: basin big sagebrush and Wyoming big sagebrush were found in lower, drier sites compared to mountain big sagebrush. Similarly, cheatgrass (*Bromus tectorum*) and exotic grasses were in lower, more xeric elevations. Cheatgrass was distributed widely across the western and central portion Conservation Assessment area. The primary regions in which the sagebrush habitats dominated the landscape were in central Washington; southeastern Oregon, northern Nevada; southwestern Idaho, and central Wyoming. Fifty-five percent of the area that potentially supported sagebrush habitats based on Küchler's map (1970) of potential vegetation for Great Basin Sagebrush, Sagebrush Steppe, and Wheatgrass-needlegrass (*Agropyron-Stipa-Artemisia*) Shrubsteppe currently existed in sagebrush; the remaining area was in agriculture (13%), urban (1%), and non-sagebrush habitats (31%). Fragmentation of sagebrush habitats was the dominant feature of the landscape in regions surrounding these strongholds. Productive lands characterized by deeper soils and higher precipitation have been converted to agriculture croplands compared with more xeric climates, shallower soils, and lower elevation in the large landscapes dominated by sagebrush. Lands managed by the U.S. Bureau of Land Management had higher proportions of sagebrush, received lower amounts of precipitation, and had shallower soils compared with lands in private ownership. Sagebrush habitats managed by the U.S.D.A Forest Service were at the highest elevations and received the most precipitation.

DELINEATION AND DESCRIPTION OF SAGEBRUSH HABITATS

Introduction

We described the general dynamics inherent in sagebrush ecosystems to better understand how recent changes have influenced these patterns of communities across the land and processes that drive these changes. Our purpose was to present basic information about the fundamental dynamics of these systems that might operate in the absence of land use or disrupted disturbance regimes. In Chapter 7, we describe the current stressors on sagebrush ecosystems and mechanisms by which those influences have altered the dynamics of these systems.

We included all portions of the sagebrush biome in our analysis, including those habitats not currently occupied or used by sage-grouse (*Centrocercus urophasianus*, and *C. minimus*) but that continues to contain sagebrush within the Conservation Assessment area. In many regions of the Assessment area, islands of sagebrush habitats remain embedded within larger expanses of highly altered landscapes. Other wildlife, such as sage sparrows (*Amphispiza belli*), Brewer's sparrows

(*Spizella breweri*), and sage thrashers (*Oreoscoptes montanus*) continue to occupy sagebrush regions not currently used by sage-grouse. As such, these regions may retain critical plant and wildlife components that might be used in restoration of sagebrush ecosystems (West 1996, West and Young 2000, Longland and Bateman 2002). These sagebrush regions also continue to interact with other habitats in the landscape matrix by providing seed sources and habitat for dispersing wildlife.

Intermountain Region: Sagebrush Taxa

The Intermountain Flora (Cronquist et al. 1994) recognized 14 shrubs and half shrubs and 13 subspecies in the genus *Artemisia*. Taxa that account for the largest area dominated by sagebrush in the Intermountain Region are the *Artemisia tridentata*[1] group represented by subspecies *tridentata, wyomingensis,* and *vaseyana,* and two low sagebrush species *A. arbuscula,* and *A. nova.* The dominant sagebrush taxa characterizing potential natural vegetation is determined primarily by soils and climate (West 1983) (Fig. 5.1). The level of sagebrush dominance or cover on a site is primarily determined by soils, climate, topography, and disturbance history. The big sagebrush subspecies (*Artemisia tridentata* ssp.) are usually found on well drained moderately deep sandy to clay loam soils. Wyoming big sagebrush (*A. tridentata* subsp. *wyomingensis*) is usually found on the warmer drier sites on elevations ranging between 150 to 1,200 m. Mountain big sagebrush (*A. tridentata* subsp. *vaseyana*) is found on relatively cooler sites varying from 1,200 to 3,200 m. (Although Wyoming big sagebrush, basin big sagebrush, and mountain big sagebrush technically are subspecies in the tridentata group, we treated them separately as species in our analysis because of their differences along the primary environmental gradients). The low forms of sagebrush including low sagebrush (*A. arbuscula*), black sagebrush (*A. nova*), and rigid (stiff or scabland) sagebrush (*A. rigida*) are generally found on shallow or poorly drained soils Eckert 1957, Fosberg and Hironaka 1964). A strong argillic horizon, duripan or bedrock are generally less than 33 cm from the surface or 50 cm in wet areas. When black or low sagebrush are found on deeper soils, depth to the wetting horizon is usually limited, and soils are coarse textured (Fosberg and Hironaka 1964, Sabinske and Kight 1978, Tisdale 1994).

Northern Great Plains: Sagebrush Taxa

The primary *Artemisia* species occupying the northeastern range of greater sage-grouse, which includes northeastern Wyoming, eastern Montana, southeastern Alberta and southwestern Saskatchewan are big sagebrush (*Artemisia tridentata*), prairie sagewort (*A. frigida*), silver sagebrush (*A. cana* ssp. *cana),* and sand sagebrush (*A. filifolia*). Wyoming big sagebrush is the most common subspecies of the *tridentata* group in this region, and typically is found in marine shale's and upland soils (Morris et al. 1976, Wambolt and Frisina 2002). Prairie sagewort is a very widely

[1] Scientific nomenclature from: Cronquist, A. A.H. Holmgren, N.H. Holmgren, and J.L. Reveal. 1972-1996. Intermountain-flora: vascular plants of the Intermountain West, USA.

distributed species, characteristic of the high plains of Central North America extending west into south central Idaho, eastern Washington and central Utah. It is widely spread throughout eastern Montana (Morris et al. 1976, Wambolt and Frisina 2002). It grows in dry open places from the plains and foothills to middle of sometimes upper elevation in the mountains up to 3,400 m). Silver sagebrush is widespread throughout the northern Great Plains 1200-2100 m occupying well drained alluvial flats, terraces, valley bottoms, and drainage ways. In Montana, it is primarily distributed throughout central and eastern portion of the state and is the most common shrubby *Artemisia* species in the north and northeastern plains (Morris et al. 1976). Silver sagebrush/western wheatgrass (*Pascopyrum smithii*) is a major habitat type throughout this region. Sand sagebrush**,** commonly growing in dunes and sandy soils, has a more limited distribution across the western Great Plains; primarily eastern Wyoming.

Classification of Alliances and Plant Associations

International Classification of Ecological Communities has separated sagebrush communities based on floristics into Alliances and Plant Associations. Alliances are delineated by a species of sagebrush and sometimes a second diagnostic shrub, which include antelope bitterbrush (*Purshia tridentata*), snowberry (*Symphoricarpos* spp.), and shadscale saltbush (*Atriplex confertifolia*). Rabbitbrush (*Chrysothamnus* spp.), which often increases with disturbance is typically not used as a diagnostic species distinguishing plant alliances. Plant associations, which further separate sagebrush communities, are usually delineated by perennial grass species. Common examples in the Intermountain Region include Idaho fescue (*Festuca idahoensis*), bluebunch wheatgrass (*Pseudoroegneria spicata*), Thurber's needlegrass (*Stipa thurberiana*), needle and thread (*S. comata*), Columbia needlegrass (*S. columbiana*), western needlegrass (*S. occidentalis*), California brome (*Bromus carinatus*), squirreltail (*Sitanion hystrix*), and Sandberg's bluegrass (*Poa sandbergii*). In the northern Great Plains common species delineating plant associations are blue grama (*Bouteloua gracilis*), sideoats grama (*B. curtipendula*), prairie sandreed (*Calamovilfa longifolia*), western wheatgrass (*Agropyron smithii*), mesa dropseed (*Sporobolus flexuosus*), sand dropseed (*S. cryptandrus*), and sand bluestem (*Andropogon hallii*).

The structure of sagebrush plant communities are typically characterized by four layers: (1) the shrub layer 3-10 dm tall, (2) forbs and caespitose grasses 2-6 dm, (3) low growing grasses and forbs <2 dm, and (4) the cryptogramic soil crust. Plant cover is usually not continuous with considerable bare ground exposed, but can approach 100% in very wet sagebrush communities. The floristic diversity in sagebrush communities is usually considered as moderate (West 1983). On sites with minimal disturbance, species numbers ranged from 20 in the Columbia Basin (Daubenmire 1975), 13-24 in the Snake river Plain (Tisdale et al. 1965), 54 species across several sagebrush communities in Nevada (Zamora and Tueller 1973), and 24 to 56 species in mountain big sagebrush communities in the northern Great Basin (Miller et al. 2000). Herbaceous biomass and cover are usually dominated by perennial grasses. There are usually a greater number of forb species than grass species, but forbs constitute a smaller portion of the biomass and ground cover.

Sagebrush Types

The sagebrush biome extends across much of the Western United States from the east slopes of the Sierra Nevada and Cascade mountain ranges east to the western edge of South Dakota. The biome stretches north from Cache Creek British Columbia and Saskatchewan south to northern Arizona and New Mexico. The sagebrush biome has been separated into several ecosystem types and subdivisions based on general differences in climate, elevation, topography, geology, and soils (Fig. 5.2, Miller and Eddleman 2001). However, differences between ecosystem types and subdivisions are not always clear due to modifying effects of variable elevation and topography. Generally a greater separation occurs along elevation gradients and aspect, which involves changes in soils and climate (West 1983). Küchler (1964, 1970) (Fig. 5.3) separated the sagebrush biome into two potential natural vegetation types; the sagebrush steppe where sagebrush is frequently a codominant with perennial bunchgrasses under potential natural conditions and the Great Basin sagebrush type where sagebrush can often be the dominant plant layer. These two vegetation types, which exclude the Great Plains region, occupy over 600,000 km^2 (Table 5.1, Küchler 1970, West 1983). The majority of the Great Basin sagebrush type lies south of the polar front gradient (Miller and Eddleman 2001) where temperatures are warmer, summer precipitation increases, and winter precipitation decreases. A third vegetation type, the northern Great Plains, also is important to greater sage-grouse and is distinctly different than the above two vegetation types.

Several geographic subdivisions in the sagebrush biome have been defined (West 1983, Miller and Eddleman 2001) that differ generally in climate, elevation, topography, geology, and soils (Fig. 5.1). For example, Wyoming big sagebrush occupies relatively warm low elevation sites highly susceptible to cheatgrass (*Bromus tectorum*) invasion compared to relatively cold high elevation sites where risk of cheatgrass invasion is considerably lower. Biotic and abiotic differences among subdivisions has a large impact on response of these communities to management and disturbance. Sagebrush habitats in the Columbia Basin (geographic subdivision 1), Northern Great Basin (2), Snake River Plain (3), Wyoming Basin (4), Southern Great Basin (5), and Silver Sagebrush (7) (Fig. 5.2) are of primary importance to greater sage-grouse.

Environmental Characteristics and Gradients of Sagebrush Ecosystems

Sagebrush species and subspecies are adapted to occupy different environments across the sagebrush biome (McArthur and Plummer 1978, Shumar and Anderson 1986, Jensen 1990, Miller and Eddleman 2001). We determined the primary environmental characteristics for each subdivision in the sagebrush biome (Fig. 5.2) and the primary gradients along which sagebrush species were distributed from field sites sampled on mapping projects in Arizona, Colorado, Idaho, Montana, Nevada, Oregon, Utah, Washington, and Wyoming. In all, our analysis included data collected from 64,454 field sites (Fig. 5.5). We caution that samples included in this analysis were collected for the purposes of generating habitat maps from satellite imagery. Therefore, the sampling design was based on obtaining adequate numbers of points in each habitat rather than on a randomly-based design and methods to estimate percent cover varied among the different data sources. Regional gaps in sampling locations resulted from lack of sampling efforts, our inability to locate or obtain comparable data sets, or incomplete information in data that we obtained from different sources.

Geographic coordinates for each site were determined from a Global Positioning System. We did not use percent cover of shrub and grass species when determining summary statistics for each species because of differences in sampling methods. We grouped individual grass species into native or exotic categories. Elevation (m), slope (%), and aspect (degree) were determined from digital elevation models (Fig. 5.4). Annual precipitation (cm) was determined from PRISM models (Daly et al. 1994) (Fig. 5.4). Each site was merged in a Geographical Information System (GIS) with the STATSGO soils database to determine depth to rock, soil pH, salinity, and available water capacity (U.S.D.A. Natural Resources Conservation Service 1995) (Fig. 5.4). Available water capacity was the total depth (cm) of available water in the soil profile. Soil pH represented the maximum value for soil reaction of the surface soil layer. Salinity (mmhos/cm) was measured as electrical conductivity of the soil in a saturated paste. We deleted sites that contained incomplete information or lacked identification of sagebrush species.

The primary sagebrush species and subspecies sampled within each subdivision of the sagebrush biome (except for the Silver Sagebrush region) are presented in Tables 5.2-7. A general relationship among the three primary subspecies of big sagebrush held consistent although absolute values differed among regions: Wyoming and Basin big sagebrush occupied lower, more xeric sites compared to mountain big sagebrush. Similarly cheatgrass and the combined category of exotic grasses (which included cheatgrasses) were found at lower elevations and generally in drier climates compared to native grasses (Tables 5.8-13). Other variables were inconsistent among the major grass categories.

We used a detrended canonical correspondence analysis (Hill and Gauch 1980, ter Braak 1986) (CANOCO; ter Braak and Smilauer 1997) to identify the relationships among sagebrush species relative to the environmental gradients. We used percent cover estimated from field surveys in this analysis. Canonical correspondence is a statistical method that emphasizes the pattern of species distribution along primary gradients (ordination axes) and is relatively robust to absolute abundances of species (ter Braak 1987, Peet et al. 1988, ter Braak 1995). Detrending is a mathematical rescaling to create consistent relationships among distances along the individual axes (Hill and Gauch 1980). We included the biome-wide sample in a single ordination to describe the dominant environmental gradients. We conducted the ordination on 24,608 sites for which we had complete sagebrush taxonomy and environmental information. We included only the 8 most common sagebrush species (low sagebrush, silver sagebrush, black sagebrush, threetip sagebrush, basin big sagebrush, mountain big sagebrush, Wyoming big sagebrush, and bud sagebrush) in addition to cheatgrass and native grasses as species variables. Environmental variables included elevation, slope, aspect, precipitation, depth to rock, soil pH, salinity, and available water capacity.

The first axis in the detrended canonical correspondence analysis extracted 42% of the variation in the relationship between sagebrush and environmental variables. The second axis explained an additional 22% of the variation. Dispersion of species along the individual axes was small (eigenvalues, λ, for the first 2 axes were: $\lambda 1 = 0.23$; $\lambda 2 = 0.09$) indicating a large overlap among sagebrush species along the environmental gradients.

Elevation, slope, precipitation, and salinity were most strongly correlated with the first axis along which sagebrush species were distributed (Table 5.14, Fig. 5.6). The second axis was a function primarily of elevation and available water capacity in the soil. The dispersion of cheatgrass, exotic grasses, and native grasses was driven largely by an elevation gradient.

Long-Term Dynamics of Sagebrush Ecosystems

Long-term dynamics of sagebrush ecosystems extend over centuries or millenniums. pre-settlement shifts in Potential Natural Vegetation were primarily caused by long term changes in climate and catastrophic disturbances (e.g. volcanic eruptions) resulting in a change in plant associations, alliances, and disturbance regimes.

Since the end of the Pleistocene 10,000 years BP (before present) climate has fluctuated with periods of cooler and wetter, warmer and drier, and warmer and more humid patterns (Antevs 1938, 1948, Davis 1982). The duration of these periods extended from centuries to several millenniums and resulted in changes in abundance between sagebrush and graminoids, and the distribution of pinyon, juniper, sagebrush, grassland, and salt desert communities (Mehringer 1985).

During the late Holocene (2,500 to 140 years BP) severe drought and major fires followed the Neoglacial (Wigand 1987) resulting in rapid regional declines in juniper and perennial grasses and expansion of sagebrush at the upper elevations and salt-desert shrub at lower elevations. Examination of charcoal layers, pollen cores, and sediments indicated that frequent large fires in combination with climate impacted pinyon and juniper abundance and distribution (Miller et al. 2001). Just prior to settlement by people of European descent, the Little Ice Age (700 to 150 years BP) was the wettest and coolest period during the last half of the Holocene. Since the end of the Little Ice Age (ca. 1850) there has been a general warming trend similar to post-Neoglacial conditions (Ghil and Vautgard 1991). However, major fires which immediately followed the post-Neoglacial period are in contrast to region-wide declines in fire events in the late 1800s and early 1900s resulting in conifer expansion that exceed anything that has occurred during a similar length of time (Miller and Wigand 1994, Miller and Tausch 2001).

Short-term Dynamics of Sagebrush Ecosystems

Short-term changes, (usually calculated in years or decades) are a function of weather and disturbance (e.g. fire, diseases, molds, changes in herbivory, etc) resulting in the fluctuation or permanent change in relative abundance of species and thus structure of plant communities. Short-term climatic cycles measured in years can greatly affect plant community dynamics, particularly in combination with disturbance through influencing plant succession, annual abundance and diversity of plant species, and length of the growing season. Two potential outcomes resulting from disturbance or weather are: 1) plant communities shift within their range of natural variability or within a steady state (succession) and potential natural vegetation remains the same, or 2) shift to a new steady state. Shifts between multiple stable states represent a transition across a threshold that requires large inputs to return to the site back to the previous state (Bestelmeyer et al. 2003, Briske et al. 2003, Westoby et al. 1989). Vegetation composition and structure that is the most persistent through time is typically determined by the intensity and frequency of disturbance events.

Continued changes in plant composition and structure resulting from a disturbance event may occur over a decade or decades. If disturbances become chronic, new seed pools become available, and/or disturbance regimes are altered, the potential natural community can shift outside of the range of historic variation to a new steady state. There is a high degree of variation in the resistance and resilience to change outside the range of natural variation across sagebrush plant associations. Resistance and resilience of change generally increases with increasing moisture and decreasing temperatures. Soil characteristics are another important characteristic that influences the stability of a site.

Changes in Distribution and Composition of Sagebrush Habitats

Post-Settlement Long-Term Dynamics (New Steady States)

Shifts in climate will continue to drive changes in potential natural plant communities. However, the addition of new disturbance factors since Eurasian settlement has created new steady states; some that have never existed in the past (Fig 5.7). The general structure of current major communities are reported in Table 5.15. New factors include domestic herbivory, introduction of exotic plants, changes in disturbance regimes (e.g. fire), and atmospheric CO_2. This has resulted in a significant portion of potential natural communities across the sagebrush biome to shift to new steady states. Several of these steady states are at risk of shifting a new steady state following fire (Laycock 1991).

Current and Potential Distribution of Sagebrush Habitats

We determined the difference between the area that would be dominated by sagebrush in Küchler's (1970) potential vegetation map for Great Basin Sagebrush (type 38), Sagebrush Steppe (type 55), and Wheatgrass-Needlegrass Shrubsteppe (*Agropyron-Stipa-Artemisia*) (type 56) (Fig. 5.3) to the current distribution of sagebrush habitats (Fig. 5.2). Our analysis determined the difference between the potential vegetation type and what currently was present (Fig. 5.8). We used only the regions included in Great Basin Sagebrush, Sagebrush Steppe, and Wheatgrass-needlegrass

Shrubsteppe types because of increased uncertainty in current vegetation maps to depict sagebrush habitats in eastern regions (such as eastern Montana) of the Conservation Assessment area. We recognize that some of the difference is a function of the difference between the coarse resolution in the Küchler's map compared to the finer resolution in the sagebrush map. To partially correct for differences in thematic and spatial resolutions, we subtracted forested, water, marsh, and wetland habitats present in the map of current habitats (Comer et al. 2002) from the total area for each sagebrush type in Küchler's (1970) map. We emphasize the analysis was for differences between current and potential distribution in sagebrush habitats and was not specific to sage-grouse habitats.

Fifty-five percent of the area that potentially could be dominated by sagebrush cover delineated by the three primary Küchler's types describing sagebrush habitats across Washington, Montana, (sagebrush habitats in eastern Montana were not included in this analysis), Wyoming, Idaho, Oregon, Nevada, Utah, California, and Colorado currently was in sagebrush habitats (Table 5.16). Wyoming (67%), California (63%), and Oregon (62%) had the highest percentage of potential vegetation that currently was in sagebrush habitats. Utah (36%) and Montana (42%) had the lowest percentage of potential area for these Küchler vegetation types that currently existed in sagebrush habitat. Agriculture was the largest single category of landcover (13%) in areas not currently mapped in sagebrush habitats. Washington (49%), Idaho (25%), and Colorado (20%) had the most potential sagebrush area currently in agriculture. Urban areas covered 1% of the potential sagebrush areas. The remaining 31% in other habitats consisted of categories that included barren, grassland, burn, exotic grassland, shrubland, and juniper woodland as the dominant landcover.

Previous estimates of potential sagebrush habitat now in urban or agriculture or were in landcover types that no longer could support the dominant vegetation were 3% for Great Basin Sagebrush, 5% for Wheatgrass-needlegrass-shrubsteppe, and 15% for Sagebrush Steppe (Klopatek et al. 1979). Based on updated maps of urban and agriculture areas (and corrected for other non-sagebrush habitats), we estimated that 6% (7,398 km^2) of the Great Basin Sagebrush region now was in agriculture, urban, or industrial areas, 46% (58,874 km^2) was in sagebrush habitats, and 48% (60,473 km^2) in other habitats. For the Wheatgrass-needlegrass-Shrubsteppe region, 7% (1,775 km^2) was in agriculture, urban, or industrial habitats; 51% (12,286 km^2) was in sagebrush habitats, and 42% (10,243 km^2) in other habitats. For the Sagebrush Steppe region, 17% (59,161 km^2) was in agriculture, urban, or industrial categories; 56% (197,379 km^2) was in sagebrush habitat; and 27% (95,731 km^2) in other habitats. We emphasize that these analysis are based only on dominant landcover across large regions; no information about understory, soil, or other characteristics not mapped in satellite imagery or captured in coarse resolution maps is implied.

Annual Grasses

Eurasian annual grasses were introduced in the assessment area in 1890 and have continued to expand their spread (Mack 1981). Cheatgrass and medusahead (*Taeniatherum caput-medusae*) have become the most problematic species within the sagebrush biome. Both are winter annuals that rely on winter precipitation for their ability to invade and dominate lands. They tend to be more of a problem in the Intermountain West (Washington, Oregon, Idaho, Nevada and Utah) than in the Rocky Mountain states that receive more summer precipitation (Montana, Wyoming, Colorado) and tend to become infestations or monocultures in the more arid Wyoming big sagebrush communities. Within the Intermountain West, Mack (1981) provides evidence that cheatgrass was first introduced via contaminated imports of grains and expanded along transportation routes and in locations of documented severe livestock grazing and reductions in native perennial grasses. For cheatgrass, it appeared to reach its current range expansion during the 1930's. From the late 1800's to the mid 1900's, livestock grazing intensities and seasons reduced the density and size of native perennial grasses and forbs and kept residual litter, the typical fuel for fires, at a minimum thus allowing sagebrush, a less palatable species for livestock, to increase during this period. Adjustments in livestock utilization and grazing seasons during the later portion of the 1900's continue today.

Although cheatgrass has been a major factor in the loss of Wyoming big sagebrush communities, medusahead is filling a similar niche in more mesic communities with heavier clay soils. Initial occurrences of this invader began in the late 1800's and early 1900's with it continuing to spread and occupy new locations (Miller et al. 1999). These communities include low sagebrush and mountain big sagebrush communities as well as some Wyoming big sagebrush communities. One common feature among these communities is the presence of a clay horizon in the soil (Dahl and Tisdale 1975).

During pre-settlement, the amount of sagebrush versus herbaceous vegetation on any particular site in the sagebrush biome has been widely debated, but heavy livestock grazing during the growing season of the herbaceous vegetation is thought to lead to greater shrub dominance (see citations in West 1983). West (1983) states that the herbaceous component of communities are more prominent in the northern portion (western intermountain sagebrush steppe) of the biome, than the southern (Great Basin-Colorado Plateau sagebrush semi-desert). The Wyoming big sagebrush communities in the Intermountain West consisted of widely spaced sagebrush and perennial grasses that generally did not carry fires except under extreme conditions. Annual grasses invaded and filled the interspaces between the shrubs and grasses and provided a continuous fuel source for fires. Pickford (1932) and Piemeisel (1951) were among the first to report the increased occurrence of fires with the invasions of annual grasses. Fires removed the sagebrush that is intolerant of fire, while cheatgrass recovers and increases in density within two years after a fire (Young and Evans 1978). Dense stands of cheatgrass reduce the probability of reestablishment of perennial grasses and shrubs (Harris 1967, Francis and Pyke 1996).

Whisenant (1990) in a non-peer-reviewed article, has shown a significant increase in fire fuels and frequency with an increase in cheatgrass relative frequency. All estimations of fire return intervals in drier Wyoming big sagebrush communities are based on expert opinion rather than clear experimental data. Wright and Bailey (1982) estimated a minimum fire return interval of 100 years whereas Whisenant (1990) reports fire return intervals were as low as 3 to 5 years in portions of the Snake River Plains where cheatgrass now dominates, but does not document thoroughly how he reached this conclusion.

Estimated area lost. The best estimate of the land area in the sagebrush ecosystems dominated by introduced annual grasses comes from a qualitative survey conducted by the U.S. Bureau of Land Management in 1991. This survey covered 400,000 km^2 of BLM-managed lands in Washington, Oregon, Idaho, Nevada and Utah. Introduced annual grasses from Eurasia, cheatgrass and medusahead, now either dominate or have a significant presence (estimated > 10% composition based on biomass) on 70,000 km^2 of public land within these 5 states. Over much of this area, annual-dominated communities can be considered a new steady state (Laycock 1991). Although Whisenant (1990) cites that cheatgrass has become a major herbaceous species in the West dominating over 400,000 km^2, his estimate is actually a major overestimate and a misinterpretation of the original citation that states that cheatgrass now dominates on many rangelands within the 41 million ha of potential steppe vegetation in the intermountain west (Mack 1981). This incorrect figure has been repeated in other prominent review papers on the topic (e.g., d'Antonio and Vitousek 1992).

We mapped the locations at which cheatgrass was recorded from our sample of field data points (Fig. 5.9). We did not estimate total area covered by cheatgrass because of uneven sampling distribution throughout its distribution. Nonetheless, the conclusion remains that cheatgrass is a dominant factor in the plant community and potentially influences fire dynamics across almost half of the sagebrush distribution.

Post settlement Woodland Expansion

Utah juniper (*Juniperus osteosperma*), western juniper (*J. occidentalis*), single-leaf pinyon (*Pinus monophylla*) and two needle pinyon (*P. edulis*) are the primary conifer species occurring in the sagebrush biome. Rocky mountain juniper to a lesser extent is expanding into sagebrush communities in portions of its range. Post settlement expansion, which began in the late 1800s, is at rates exceeding that of any expansions during the Holocene (Miller and Wigand 1994). The accelerated expansion of pinyon and juniper is synchronous with the introduction of livestock, changes in mean fire return intervals, and optimal climatic conditions during this time period (Tausch et al. 1981, Miller and Rose 1999, Miller and Tausch 2001).

As woodlands encroach and increase in dominance sagebrush steppe communities shrubs rapidly decline (Fig. 5.10). Juniper and pinyon woodlands occupy approximately 189,000 km^2 in the Intermountain Region (Miller and Tausch 2001). Estimates of woodland expansion vary regionally throughout the Intermountain West, ranging 60 to 90%. Expansion has most impacted the big sagebrush group but has also occurred in habitats dominated by low sagebrush and black sagebrush. Although we have limited documentation, other conifer species such as Douglas-fir

(*Pseudotsuga menziesii*) have been actively expanding into mountain big sagebrush. These conifers currently occupy far less land than they are capable of under current climatic conditions (West and Van Pelt 1986, Miller et al. 2001). In addition, many of these woodlands are in a transitional state where tree densities and cover are continuing to increase, resulting in a loss of sagebrush steppe communities.

Shrub Die-off

Areas of shrub die-offs are locally common in the Great Basin and other arid shrublands. In addition to sagebrush, shadscale saltbush and other species of salt desert shrubs are affected (Pyke and Dobrowski 1989). Possible causes for the die-offs include single factors or interactions among drought, excessive moisture, increased soil salinity, parasites, disease, insects, and grazing pressure (McArthur et al. 1990). In Utah, approximately 2,544 km^2 of shrublands have experienced shrub die-off (Fig. 5.11). Additional die-offs have occurred in northwestern Utah and other parts of the Intermountain West but we were unable to find maps to delineate the regions and estimate the total area.

Sagebrush Ecosystems: Landscape Characteristics

Primary components of habitats that influence underlying processes are the quantity, composition, and configuration in the landscape (Wilcox and Murphy 1985, Turner 1989, Turner et al. 2001). The quantity, or amount of habitat can contribute to spatial and temporal stability of habitats and animal populations living within those regions (Saunders et al. 1991, Shugart 1998). Many species of wildlife are sensitive to the amount of available habitat; loss or degradation of suitable habitat is a significant cause in declines of many species (Simberloff 1988, Opdam 1991). In addition, larger habitat patches often contain higher total numbers and diversity of species (Rosenzweig 1995).

Composition of sagebrush habitats strongly influences inter-related processes, such as cheatgrass-fire cycle (Young and Evans 1973, West 1979, d'Antonio and Vitousek 1992). Disturbance regimes of sagebrush habitats and management actions in sagebrush communities dominated by a cheatgrass understory are very different from sagebrush communities largely composed of native bunchgrasses (Brooks and Pyke 2001, Hemstrom et al. 2002, Wisdom et al. 2002).

Configuration or the pattern of habitats within the landscape influences processes such as the spread of disturbance, predation, and dispersal of wildlife (Wiens et al. 1986). Increased fragmentation in the landscape was correlated with increased invasion by cheatgrass into sagebrush patches, which subsequently facilitates fire spread, loss of sagebrush, and conversion to annual grasslands (Knick and Rotenberry 1997). Persistence of animal populations and success of dispersing individuals in locating suitable habitat was highly dependent on the spatial arrangement of the landscape (Doak et al. 1992, Fahrig and Merriam 1994, Flather and Bevers 2002). In addition, many wildlife species are less numerous at habitat edges because of sensitivity to edge characteristics or because of increased predation along habitat boundaries (Andren 1994, Paton 1994, Wiens 1989).

Sagebrush habitats always have contained temporal and spatial variation because of past disturbance history (Young et al. 1979, West and Young 2000). Although changes in landscape characteristics can explain trajectories in habitats and wildlife and guide restoration efforts (Whisenant 1999, Morrison 2002), we often lack the information necessary to estimate those dimensions of pre-settlement landscapes for comparison to current conditions and form (Knick et al. 2003). For example, shrubsteppe habitats interspersed within an agricultural landscape in Washington State were characterized by smaller but more numerous patches in 1990 than in 1900 (McDonald and Reese 1998). Recently, we have developed an understanding of the influence of fragmentation on bird population dynamics: habitat fragmentation was strongly correlated with distribution of shrubland birds and predation on nests in these regions (Vander Haegen et al. 2000, 2002). Because comparable information does not exist on bird populations in 1900, we infer that predation was greater in 1990 because of increased fragmentation.

State and transition models of dynamics of sagebrush habitats (Westoby 1981, Laycock 1991, Allen-Diaz and Bartolome 1998) also neglect spatial aspects that influence the inertia and scale of state changes. The spatial characteristics of a disturbance can range from loss of a single plant to conversion of a sagebrush-dominated landscape to annual grassland (Fig. 5.12). Across this wide variation, the spatial extent of disturbance and the configuration of the resulting landscape influence internal functions, future disturbance regimes, and temporal dynamics of recovery. Spatial and temporal characteristics also are significant components necessary to measure the effect of disturbance and habitat change on sage-grouse and other dependent animals (Knick and Rotenberry 2000, 2002) (Fig. 5.13). Finally, unique patterns (signatures) result from different disturbance regimes in shrubsteppe landscapes and can be important in management decisions on land use (Knick and Rotenberry 1997).

Analysis of the landscape components of sagebrush ecosystems is a critical step to understanding the interaction of disturbance, habitat change, and distribution and population dynamics of dependent wildlife (Anderson and Inouye 2001, Crawford et al. 2004). Therefore, we described quantity, composition, and configuration of sagebrush landscapes to interpret the current and future dynamics of habitats and distribution and population trends of sage-grouse.

Methods

Sagebrush Distribution Map. We created the base layer of sagebrush distribution used in our analysis of landscape characteristics by merging individual state and provincial coverages. An initial coverage included Washington, Idaho, Montana, Oregon, Wyoming, Nevada, Utah, Colorado, and California (Comer et al. 2002). Habitats in the coverage were classified from Landsat Thematic Mapper satellite imagery taken in 1990 (± 2 years). More recent vegetation data was incorporated into the coverage when possible. However, many changes in sagebrush habitats over the past 10 years caused by extensive fires and continued conversion to agriculture are not reflected in this map. New maps of sagebrush distribution across the Intermountain Region currently are in progress but will not be available until 2005 and will not include Wyoming and Montana.

The coverage contained 58 habitat classes, which were reclassed into urban, agriculture, water, forested habitats, and sagebrush. The sagebrush class represented 10 communities: (1) Wyoming and Basin big sagebrush, (2) black sagebrush, (3) low sagebrush, (4) low sagebrush–mountain big sagebrush, (5) low sagebrush–Wyoming big sagebrush, (6) mountain big sagebrush, (7) scabland (stiff) sagebrush, (8) threetip sagebrush, (9) Wyoming big sagebrush, and (10) Wyoming big sagebrush–squaw apple. Although silver sagebrush is a dominant sagebrush species in eastern Montana and an important sage-grouse habitat, existing vegetation maps from which we developed the range-wide map of sagebrush distribution did not identify silver sagebrush as separate category in the map legend.

Differences in thematic classes among states were reconciled by reclassifying habitats according to the National Vegetation Classification System (NatureServe 2002, Reid et al. 2002). Differences in spatial resolution also were rectified and standardized at 90-m resolution for the extent of the grid coverage. We then added vegetation coverages for Arizona, New Mexico, South Dakota, and the provinces of Alberta, Saskatchewan. We were unable to obtain sagebrush information for North Dakota. Accuracy was not assessed for the entire coverage. However, accuracy of the original vegetation maps was approximately 75-80% for individual regions in which accuracy was assessed (Caicco et al. 1995, Edwards et al. 1998). Higher rates of accuracy are difficult to achieve in semi-arid regions because of the low amounts of vegetative material that contribute to the spectral signature (Wilson and Tueller 1987, Pilon et al. 1988, Knick et al. 1997).

GIS procedures and landscape analyses. We resampled the coverage of sagebrush vegetation into a 250-m and 500-m grid-cell resolution (actual cell size was 270 and 540 m), which permitted us to detect relatively fine patterns in habitat configurations. Even though the resolution of our base coverage was 90 m, the ecological patterns and processes (e.g., fire, spread of exotic plant species, predation by raptors) that we studied likely operate at much larger spatial scales; increasing the coarseness of our minimum cell size does not influence detection of these larger-scale disturbances and land cover changes. Similarly, little management occurs on such small areas but our broad-scale approach is appropriate for the larger scale at which land use plans are developed.

We derived landscape metrics (Turner et al. 2001) from the base vegetation coverage to describe patterns of sagebrush distribution and fragmentation at different scales using a moving window analysis in a GIS. We performed all spatial analysis in the ARC (ESRI 1998) and GRASS

GIS (USA-CERL 1993) programs. Output maps of landscape metrics were derived either in ARC/GRID or the r.le module of GRASS (Baker and Cai 1992).

Multiscale Patterns of Distribution of Sagebrush Habitats

We produced a series of maps to illustrate patterns of sagebrush distribution at different spatial resolutions. Our objective was to depict the landscape from multiple perspectives that represent levels of management actions (regional to site-based treatments and assessments), from the large scales at which disturbances such as fire operate, and from an organism's perspective in selecting habitats (Johnson 1980, Rotenberry and Knick 1999, Wiens 2002). Although we often introduce bias in the scale that we choose to study or depict in a system, we have tried to select scales that are appropriate to the fundamental processes in the system (Levin 1992). In addition, the appreciation of the different levels of sagebrush cover may help in understanding the hierarchical organization of these ecosystems (Urban et al. 1987).

Sage-grouse likely select habitats based on characteristics present at multiple scales. Large-scale patterns of sagebrush and other habitats may influence general movements for populations that may move >200 km between seasonal ranges and whose annual ranges may encompass >2,700 km (Schroeder et al. 1999, Leonard et al. 2000). Smaller scale features of sagebrush habitats may influence within season movements or choice of nesting locations (Connelly et al. 2000). Therefore, our multi-scale analyses emphasize the local to regional patterns of sagebrush habitats. Using 0.5-km grid cells and a moving window analysis, we determined the percent of the total area dominated by sagebrush habitats within a 5-, 18-, 50-, and 100-km radius of each cell.

Primary regions containing the highest percentage of sagebrush habitats over large areas (50- and 100-km radius; Fig. 5.14) were located in central Washington; southeastern Oregon, northern Nevada; southwestern Idaho, and central Wyoming. Local patterns in the sagebrush distribution emerged more strongly when we mapped the percent of sagebrush cover within 18- and 5-km radii (Fig. 5.14).

We examined the underlying environmental characteristics related to the average patch size of sagebrush (represented by the proportion of sagebrush within 5-km radii; Fig. 5.15) in the landscape. To determine the characteristics related to patch size of sagebrush, we estimated the elevation, precipitation, rock depth, and average water capacity in relation to increases in the patch size (represented by the proportion of landscape covered by sagebrush habitats in the 5-km radius from 0 to 100%).

Landscapes having a low cover of sagebrush were in regions characterized by deep soils, lower elevations, high precipitation and average water capacity in the soils. Agriculture regions were primarily represented at lower elevations. In contrast, landscapes containing a larger proportion of sagebrush cover were drier and in shallower soils. Mean elevation increased rapidly for landscapes containing little or no sagebrush to those containing 20% cover. Mean elevation then fluctuated up to 90% cover in the landscape after which those regions in which patch sizes of sagebrush covering a minimum diameter of $\geq$10 km were found at lower elevations. The larger landscapes dominated by Wyoming big sagebrush in more xeric conditions were likely represented at this extreme. More productive lands have been converted to agriculture croplands on which remaining sagebrush habitat is highly fragmented (e.g., Vander Haegen et al. 2000).

Multiscale Fragmentation Patterns of Sagebrush Habitats

We examined fragmentation at 2 spatial scales based on management guidelines for habitats surrounding sage-grouse leks. We determined the number of edges in the map of sagebrush distribution (Fig. 5.2) between sagebrush and other habitats within 5- and 18-km radii of each map (Fig 5.16) (number of edges is determined from the total possible number of edges in a gridded map within which each square grid cell has 4 edges). To benefit nesting and early brood-rearing sage-grouse in non-migratory populations, protection of sagebrush within 5 km of leks was recommended in landscapes containing uneven habitat distribution. For migratory populations, protection of 18 km of habitat surrounding leks was recommended (Connelly et al. 2000).

Fragmentation begins to have significant effects when suitable habitat becomes less than 30-50% of the landscape (Andrén 1994, Flather and Bevers 2002). At low levels of suitable habitat, inter-patch distances increase exponentially and spatial arrangement becomes the critical factor determining success of dispersers finding and using potential habitat. The shift in the dominant landscape characteristic between habitat quantity to spatial arrangement of suitable habitats likely is represented by a threshold rather than a gradual change in dominant characteristics (With and King 1999).

We represented the dominance of sagebrush habitat relative to landscape configuration by combining maps of 18-km metrics for percent cover of sagebrush (Fig. 5.14) and habitat fragmentation (Fig. 5.16). We used a threshold of 30% cover of sagebrush in the landscape at which fragmentation becomes the primary characteristic. The final map (Fig. 5.17) depicts those regions containing >30% of the landscape in sagebrush cover within 18-km of each cell, which then grade from areas containing low amounts of sagebrush cover into landscapes characterized by fragmentation of sagebrush habitats (Knick et al. 2003).

We measured proportion of non-sagebrush habitat for 5-, 18-, and 54-km moving window sizes across the sagebrush area. Our objectives were to depict the relative amount of habitats other than sagebrush at the perspective of a sage-grouse selecting habitats at local scales near lek or nest sites (5- and 18-km) as well as at a scale that might influence larger seasonal patterns of use. We considered urban, agriculture, water, and forested habitats to be unsuitable habitats for greater sage-grouse. Grasslands were included in potential habitat only if located <5 km from a sage-grouse lek that was active within the past 10 years. Otherwise grasslands were classed as nonhabitat.

Nonhabitat regions (Fig. 5.18) represented an inverse relationship to those regions in which sagebrush was the dominant feature in the landscape. Decreasing the resolution from 54- to 5-km emphasized more of the fragmentation patterns within the sagebrush strongholds.

Characteristics of Lands under Private and Public Ownership

Approximately 70% of the sagebrush habitat is managed by federal or state agencies; the remaining 30% is owned privately (Table 5.17). Of the federal agencies, the U.S. Bureau of Land Management has responsibility for 50% of the sagebrush habitat in the United States. Sagebrush habitats managed by the USDA Forest Service were on the boundaries of the dominant sagebrush regions (Fig. 5.19). State agencies managed <5% of the sagebrush habitats nationally and >10% of the sagebrush in Arizona and Washington. Privately owned lands were distributed throughout the sagebrush biome, but were a major (>35%) constituent of sagebrush landscapes in eastern Montana, eastern Wyoming, Washington, and Colorado (Fig. 5.19).

We described characteristics from the perspective of primary management authority or ownership. In this analysis, we determined the dominant habitat, environmental, and soil characteristics of all areas within the Conservation Assessment study area grouped into primary stewardship categories.

Lands in private ownership contained relatively small sizes of sagebrush patches (represented by the proportion of sagebrush within a 5- and 18-km radius) at low elevations having high precipitation and deep soils (Table 5.17). In comparison, lands managed by the U.S. Bureau of Land Management had higher proportions of sagebrush receiving lower amounts of precipitation and more xeric soils compared to lands in private ownership. Lands managed by the USDA Forest Service were at the highest elevations and received the most precipitation. Private lands were the least fragmented because croplands impose continuous blocks of non-sagebrush habitat on the landscape (Knick and Rotenberry 1997).

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Zamora, B., and P. T. Tueller. 1973. *Artemisia arbuscula, A. longiloba, and A. nova* habitat types in northern Nevada. Great Basin Naturalist 33:225-242.

Fig 5.1. Ordination of major sagebrush taxa in the Intermountain Region against gradients of soil temperature and soil moisture (adapted from West and Young 2002, with additions from Roberston et al. 1966; McArthur 1983, and modifications by the authors). Sagebrush species not shown in the graph were prairie sagewort (*Artemisia frigida*), Owyhee sage (*A. papposa*), birdfoot sagebrush (*A. pedatifida*), and bud sagebrush (*Picrothamnus desertorum*).

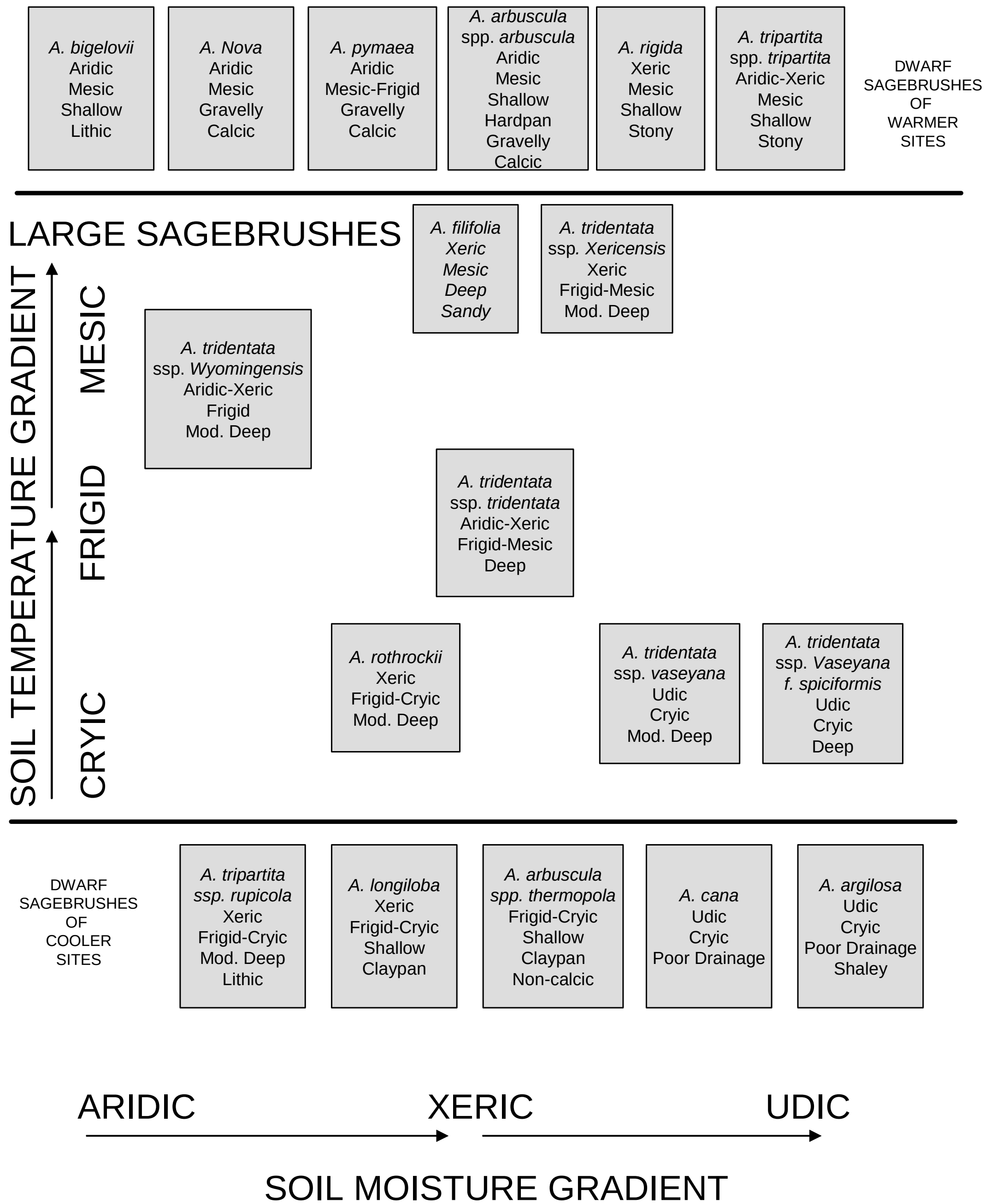


Fig. 5.2. Geographic subdivisions within the *Sagebrush Steppe* are (1) Columbia Basin, (2) Northern Great Basin, (3) Snake River Plain, and (4) Wyoming Basin. The *Great Basin* includes (5) Southern Great Basin, and (6) Colorado Plateau. The *Great Plains* overlaps the Silver Sagebrush (7) subdivision (derived from West 1983, Kuchler 1970 with additions from Miller and Eddleman 2001, and authors). Percent sagebrush habitat is the general landscape distribution of sagebrush.

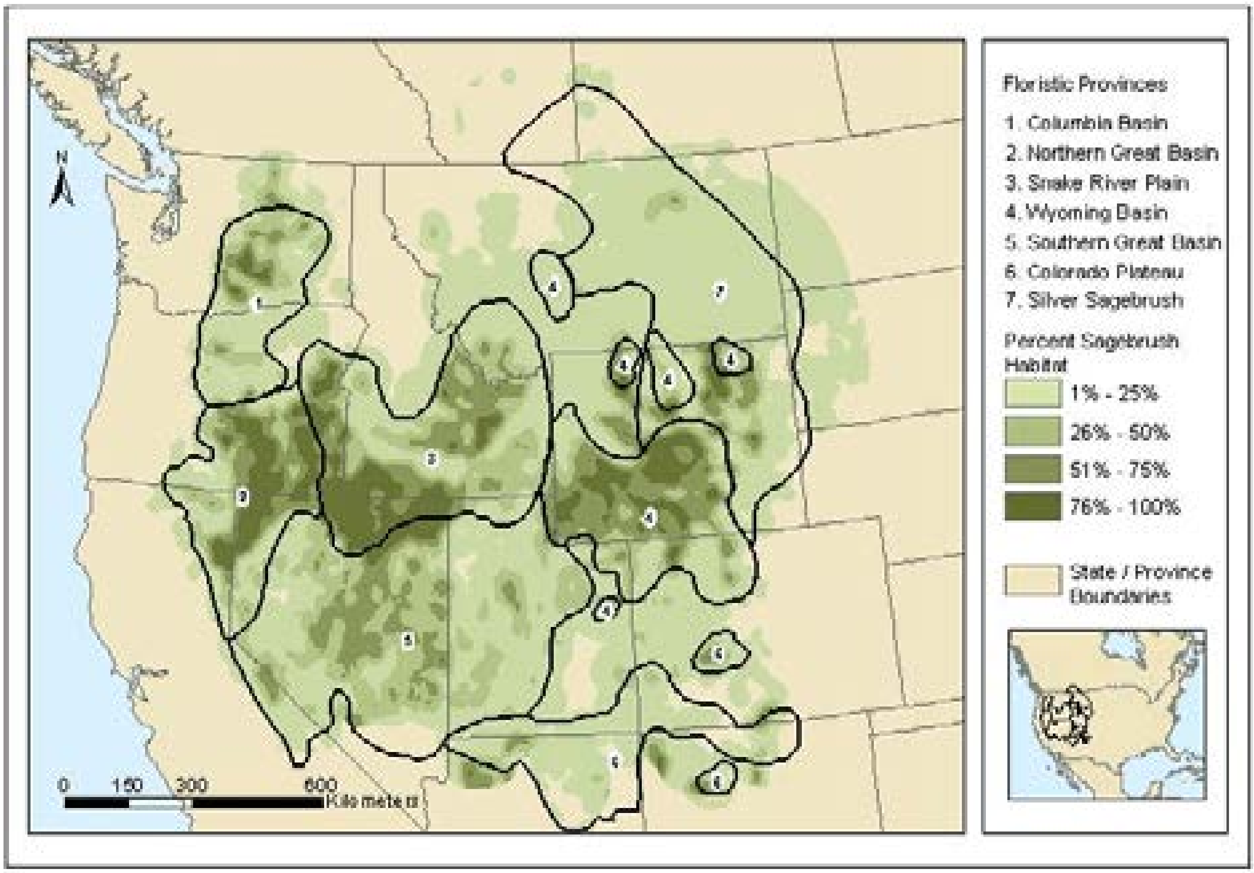


Fig. 5.3. Küchler map (1970) of potential sagebrush distribution. Küchler's map represents "the vegetation that would exist today if man were removed from the scene and if the resulting plant succession were telescoped into a single moment." The map illustrates the potential distribution of Great Basin Sagebrush, Sagebrush Steppe, and Wheatgrass-needlegrass Shrubsteppe vegetation within the Conservation Assessment area. The vegetation classes depicted in Küchler's map represent the distribution of sagebrush habitats, and not the distribution of sage-grouse habitat. Extensive areas of sage-grouse habitats exist in other portions of the sagebrush biome (eastern Montana) that are not included in these vegetation classes (e.g., compare sagebrush distribution in Fig. 5.2 and sage-grouse distribution in Fig. 1.1).

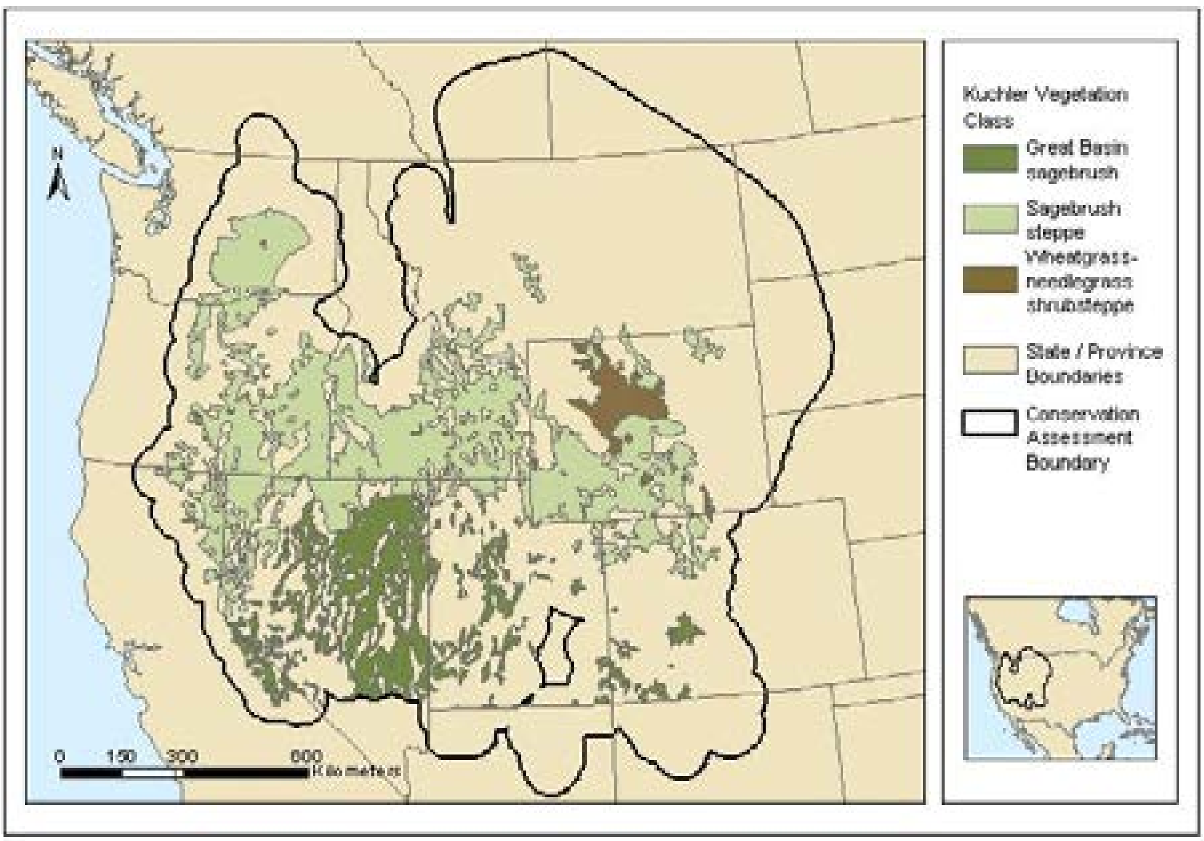


Fig. 5.4. Spatial distribution of environmental variables used to determine gradients separating sagebrush communities. Elevation (m), was determined from digital elevation models. Annual precipitation (cm) was determined from PRISM models (Daly et al. 1994). We merged from individual STATSGO coverages (U.S.D.A. Natural Resources Conservation Service 1995) to obtain soil characteristics. Available water capacity was the total depth (cm) of available water in the soil profile. Salinity (mmhos/cm) was measured as electrical conductivity of the soil in a saturated paste. Soil pH represented the maximum value for soil reaction of the surface soil layer.

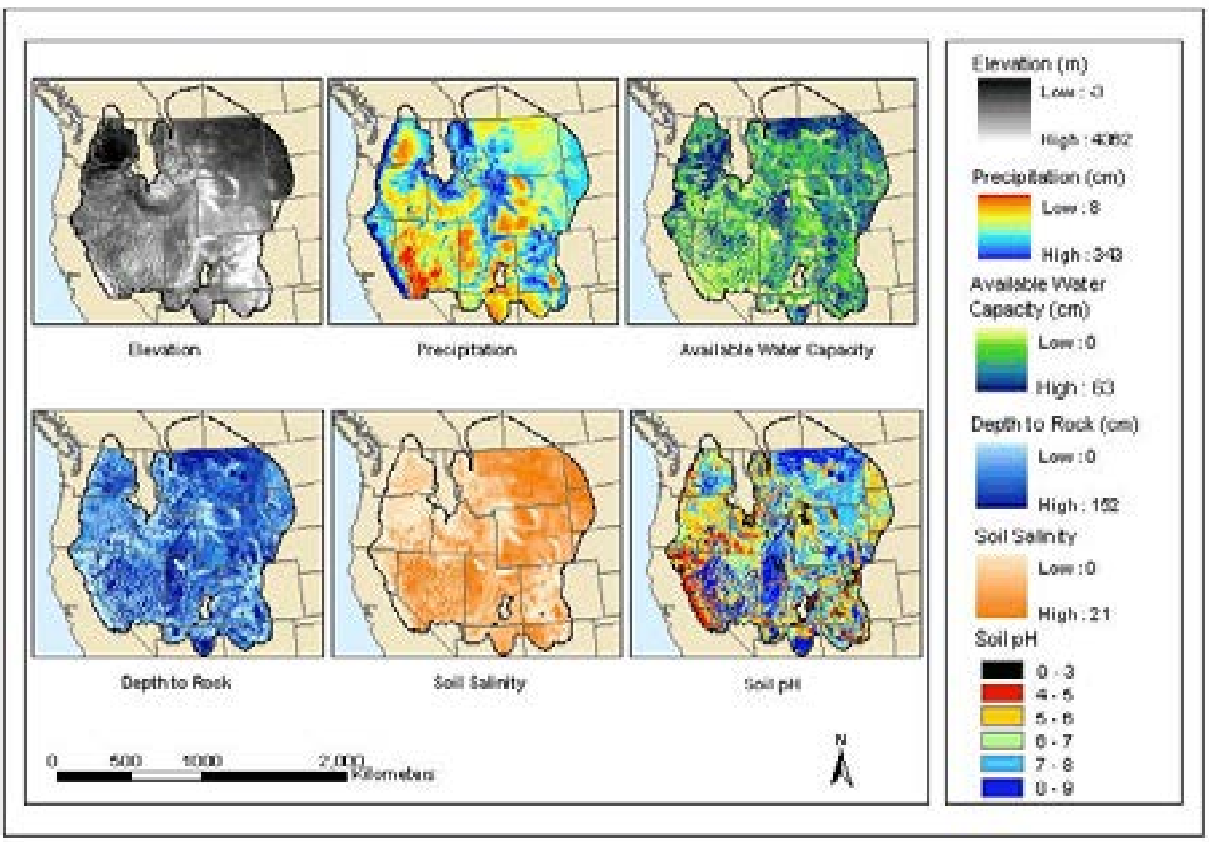


Fig. 5.5. Locations at which vegetation data used in this study were collected. Analysis of vegetation characteristics from the combined data set were limited because the primary objective of the data were to develop models to map habitats from satellite imagery. Differences in sampling methods also precluded analysis of cover statistics. Gaps in sampling distribution represented lack of sampling efforts, our inability to locate or obtain comparable data sets, or incomplete information in data that we obtained from different sources.

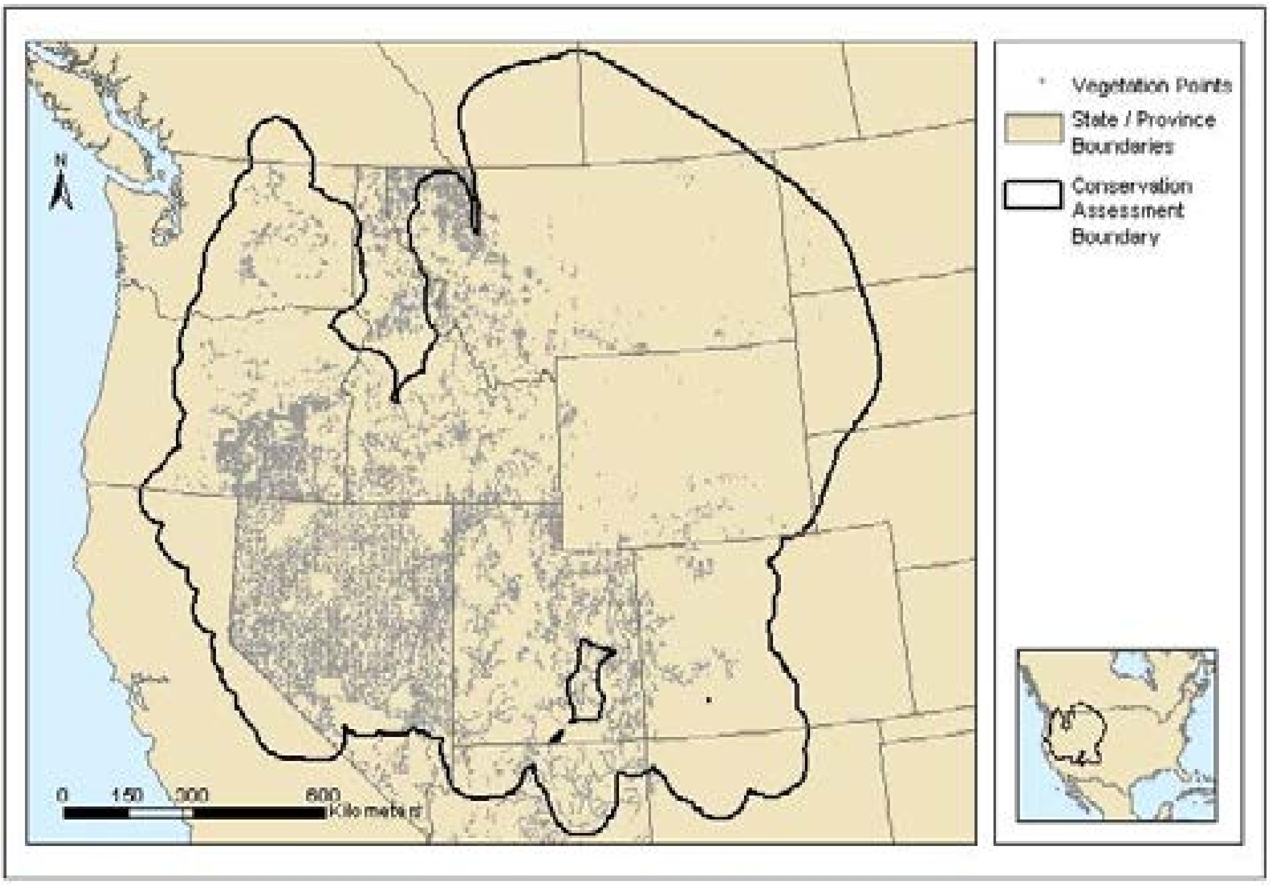


Fig. 5.6. Distribution of sagebrush communities along environmental gradients. The ordination diagram, produced by detrended canonical correspondence analysis, was based on 24,608 field sites sampled across Washington, Oregon, Idaho, Utah, Nevada, Arizona, Wyoming, and Colorado. Sagebrush taxa represented in the ordination were: low sagebrush, silver sagebrush, black sagebrush, stiff sagebrush, threetip sagebrush, basin big sagebrush, mountain big sagebrush, and Wyoming big sagebrush. Cheatgrass, exotic grasses (including cheatgrass), and native grasses also were included in the ordination. Relationship of environmental variables with ordination axes are shown above the species distributions. Correlations of environmental variables with the species axes are given in Table 5.14.

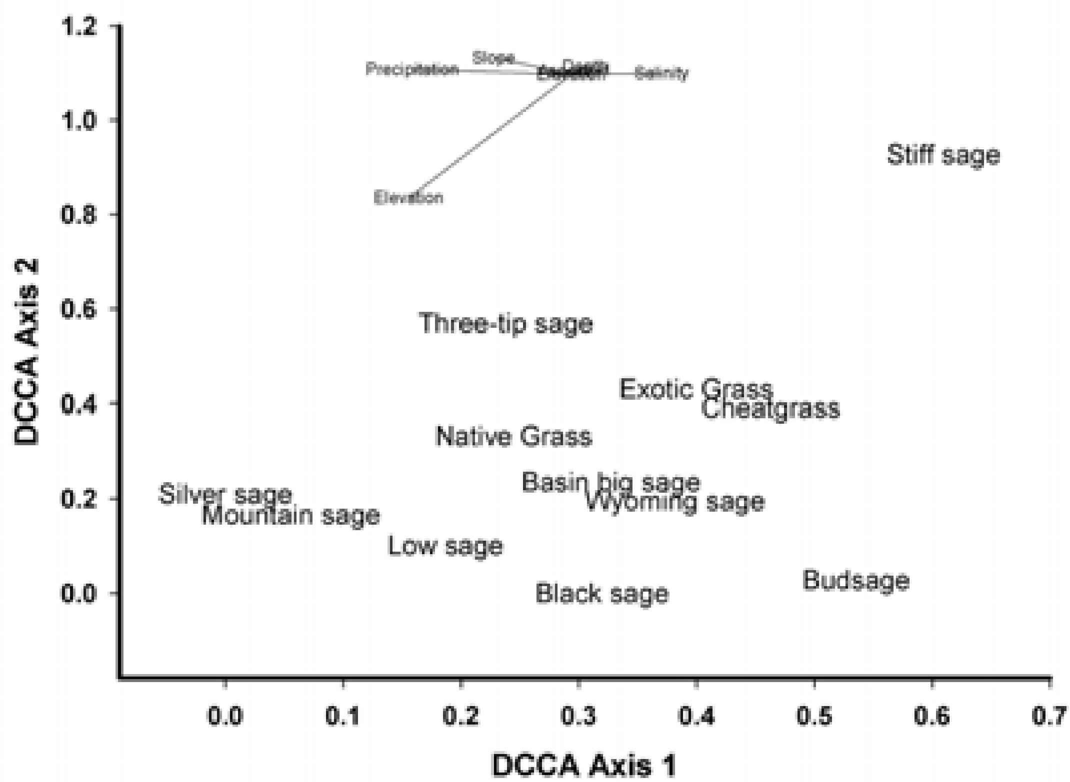


Figure 5.7. Pre- and post-settlement dynamics in the sagebrush biome. Box and arrow size is an estimate of the relative proportion of shift from one steady state (community) to another.

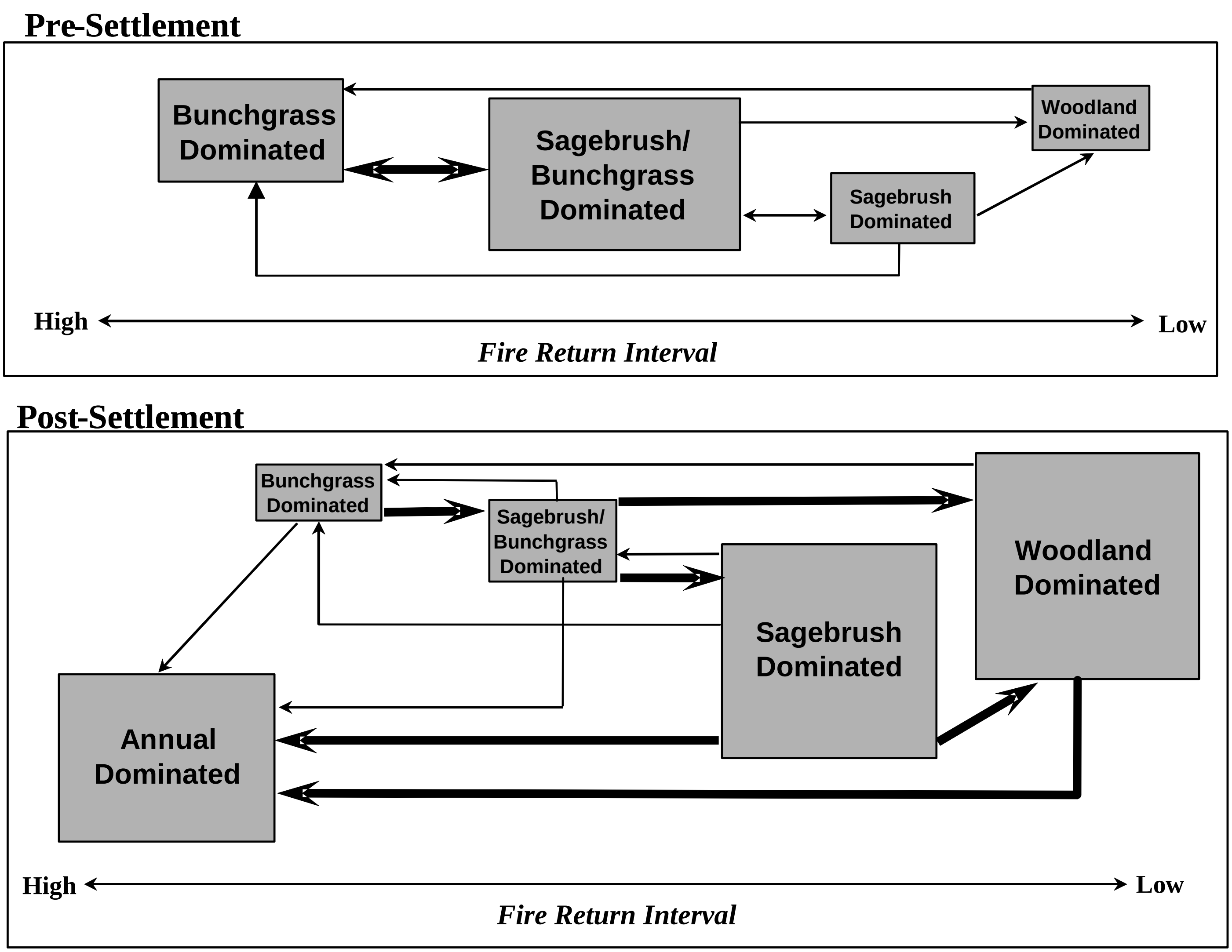


Fig. 5.8. Difference between Küchler map (1970) of potential sagebrush distribution (Fig. 5.3) and current distribution of sagebrush (Fig. 5.2). Only the distribution of Küchler's categories for Great Basin Sagebrush, Sagebrush Steppe, and Wheatgrass-needlegrass Shrubsteppe are used in this analysis. Sagebrush habitats also exist outside of the distribution of these habitat types.

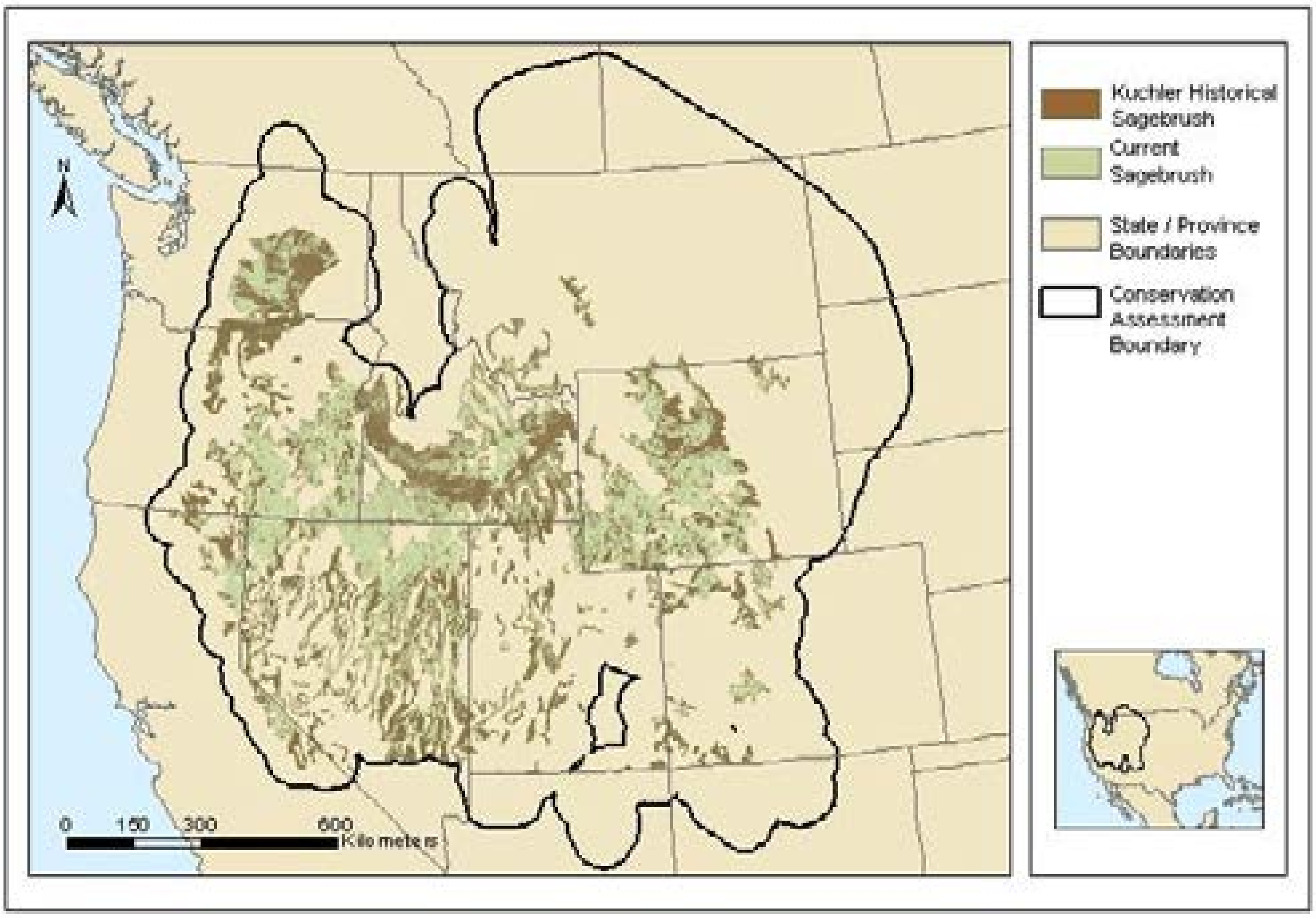


Fig. 5.9. Spatial distribution of cheatgrass. We depicted the boundary enclosing all field sampling points included in our analysis, the sampling points at which cheatgrass was recorded, and a 95% kernel distribution derived from those points. In this analyses, cheatgrass distribution was estimated (modeled) from actual sampling points. The cheatgrass risk model (Chapter 7, Fig. 7.10) was an assessment of locations where cheatgrass was likely to invade or increase in abundance based on a set of predictive variables.

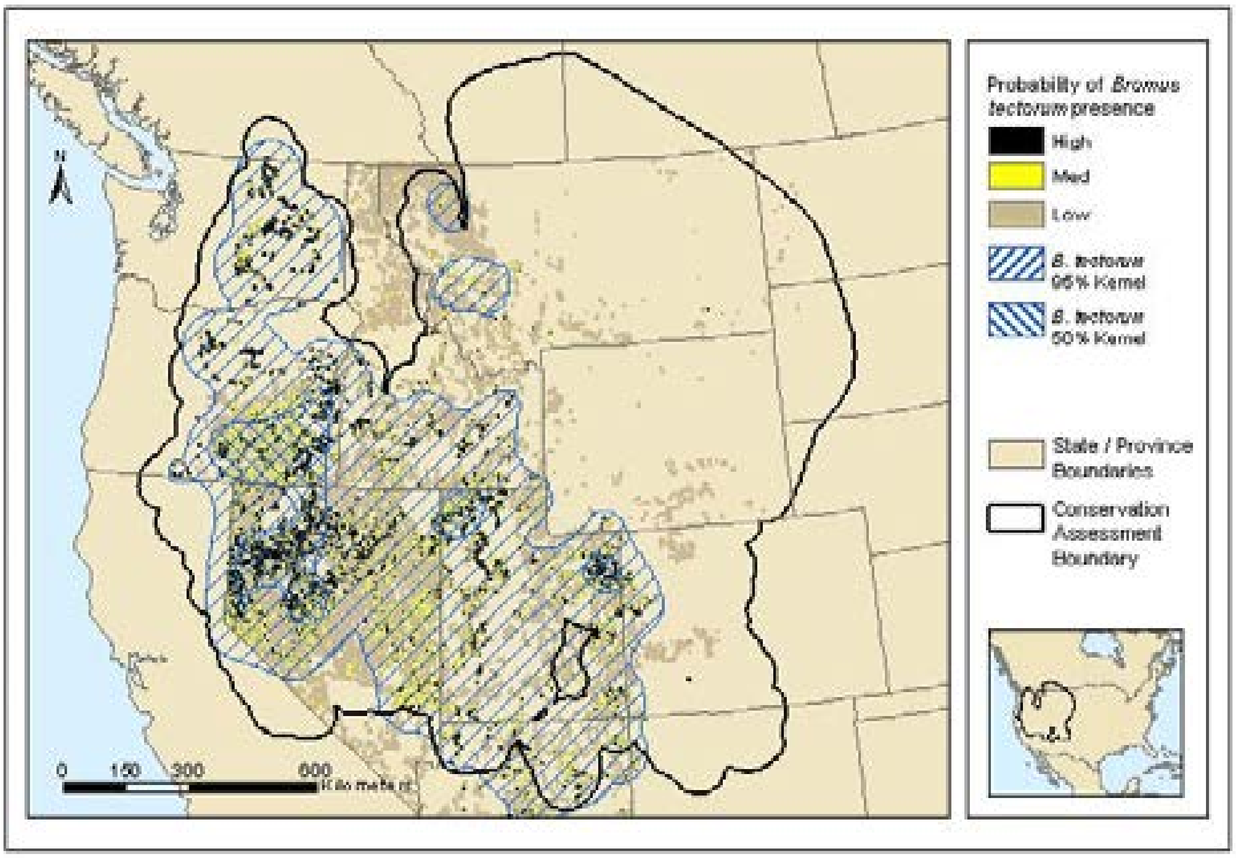


Figure 5.10. Relationship between juniper and mountain big sagebrush canopy cover in three plant associations; *Artemisia tridentata* subsp. *vaseyana-Symphoricarpos oreophilus/Stipa columbiana* (ARTRV-SYOR/STCO), *A. tridentata* subsp. *vaseyana/Festuca idahoensis* (ARTRV/FEID), and *A. tridentata* subsp. *vaseyana*/*Stipa thurberiana* (ARTRV/STTH) (Miller et al. 2000).

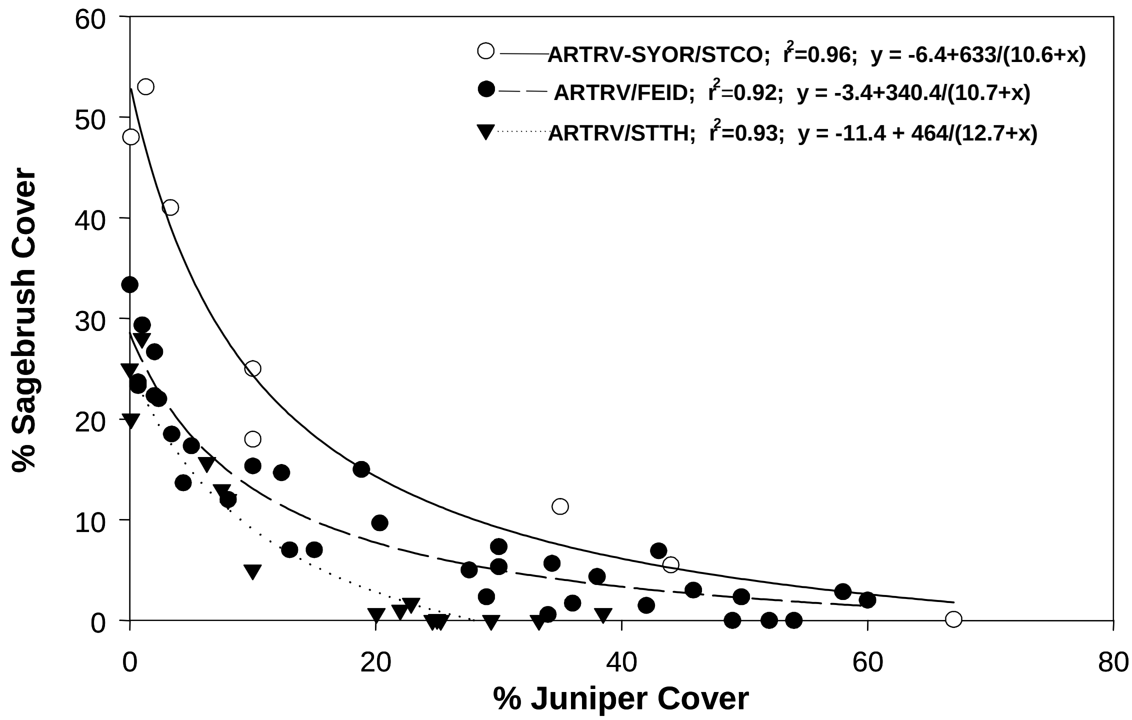


Fig. 5.11 Regions of shrub die-off in Utah.

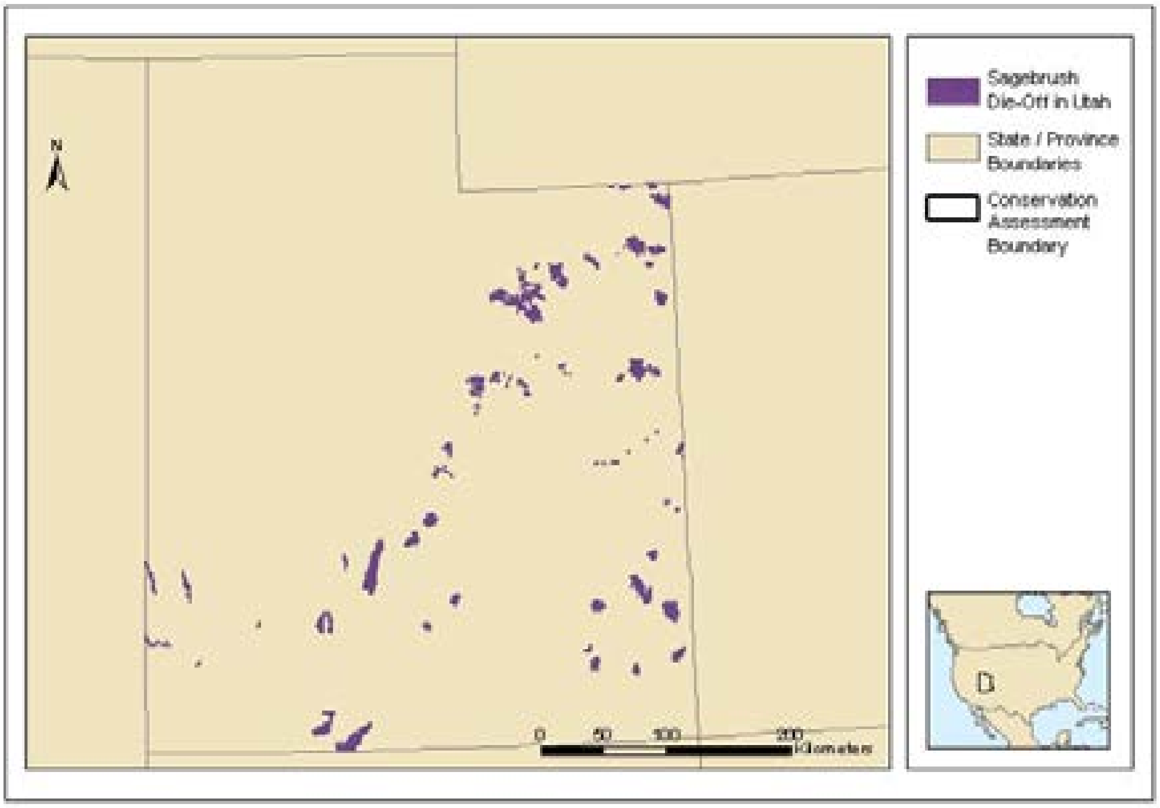


Fig. 5.12. The contribution of spatial dimensions to state and transition models for a sagebrush system with 2 alternate endpoints. A sagebrush community containing a natural grass understory represents one stable community. Similarly, a cheatgrass-dominated grassland without sagebrush is a stable endpoint because recurrent fires prevent recolonization by sagebrush. The intermediate mosaic represents an unstable habitat from which slight increases in either disturbance space (a function of frequency and intensity) or the spatial extent dominated by cheatgrass flips the system from returning to sagebrush and sends the dominant habitat into cheatgrass from which a return to sagebrush is unlikely. (Conceptual development of spatial components to state and transition models [Fig. 5.7, 7.32] by authors).

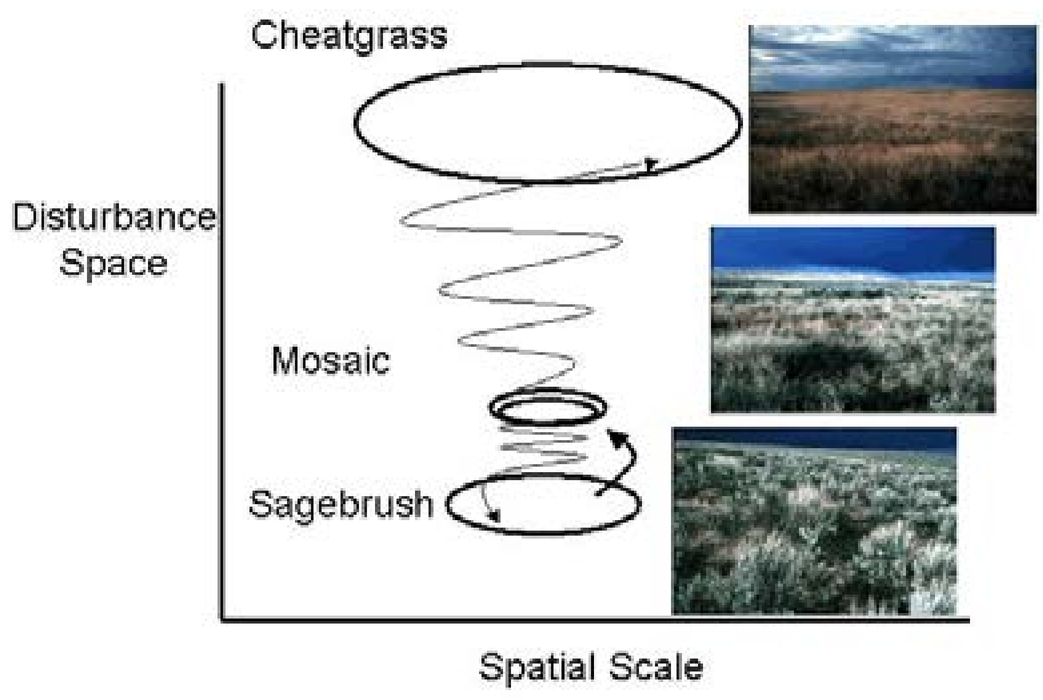


Fig. 5.13. The relationship between temporal and spatial scale of habitat dynamics and effect on sage-grouse. Disturbances that influence small spatial extents may be absorbed within the relatively large ranges used by sage-grouse. Conversely, species such as Brewer's sparrows may be more affected by disturbances that alter sagebrush habitats at smaller spatial extents. (Conceptual development of spatial components to state and transition models [Fig. 5.7, 7.32] by authors).

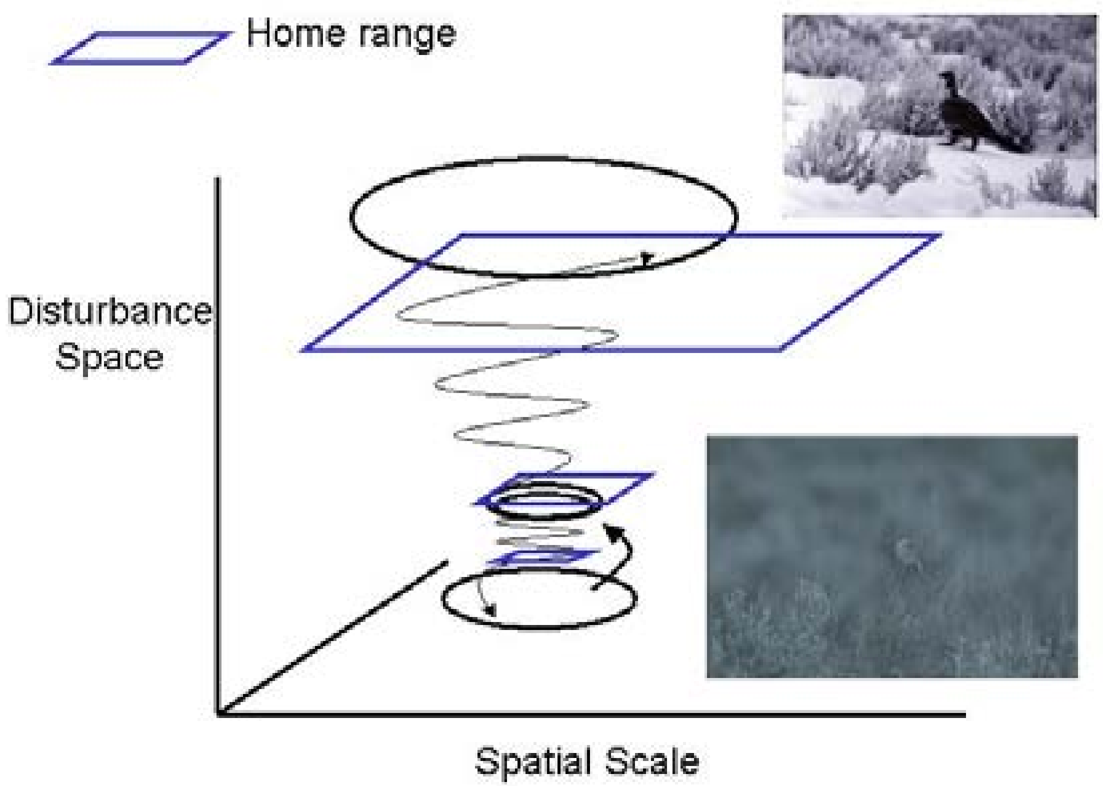


Fig. 5.14. Percent area in sagebrush habitat within a 5-, 18-, 50- and 100-km radius of each 0.5 km grid cell.

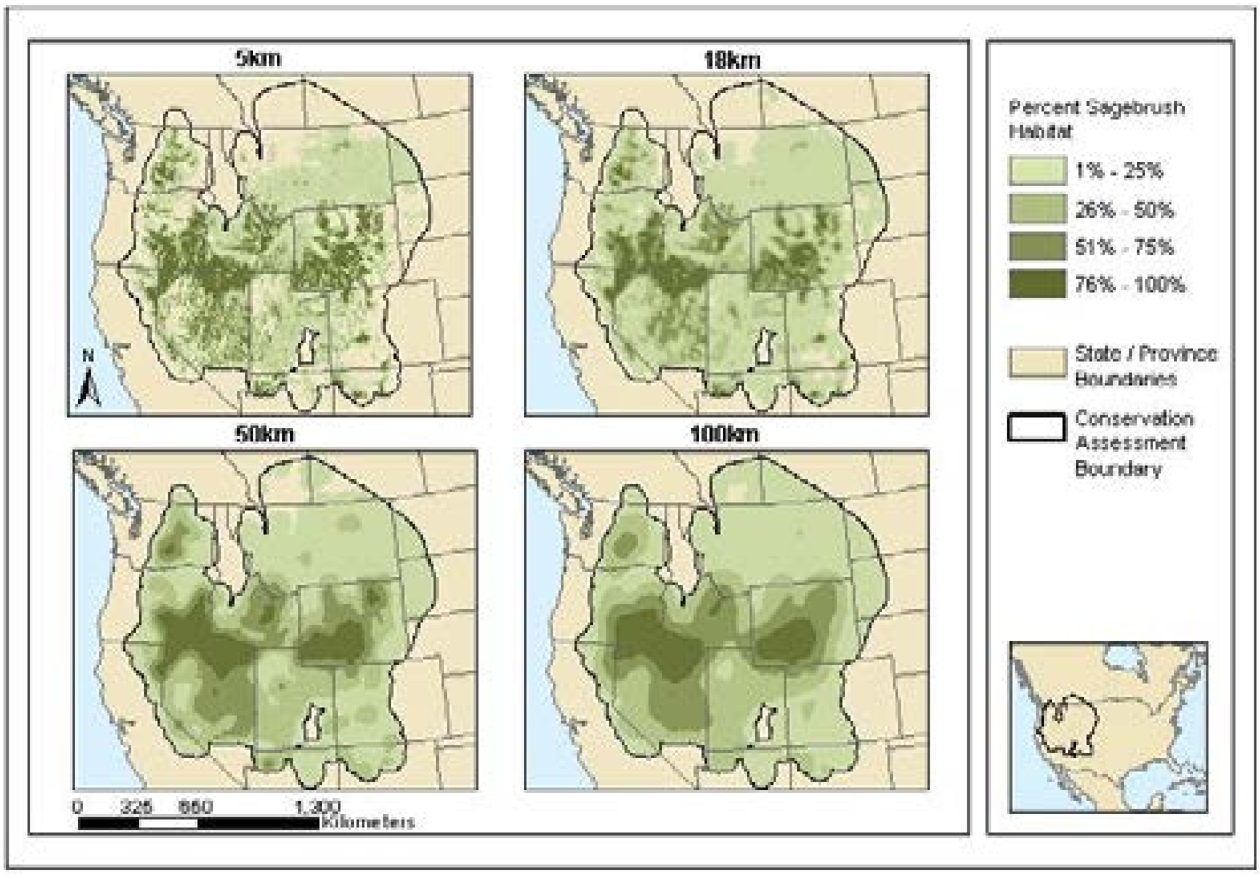


Fig. 5.15. Relationship between % of the landscape dominated by sagebrush cover (5-km radius) and elevation, precipitation, available water capacity (cm), and depth to rock (cm). Percent cover is the landscape component and does not represent site-specific estimates of percent ground cover by sagebrush.

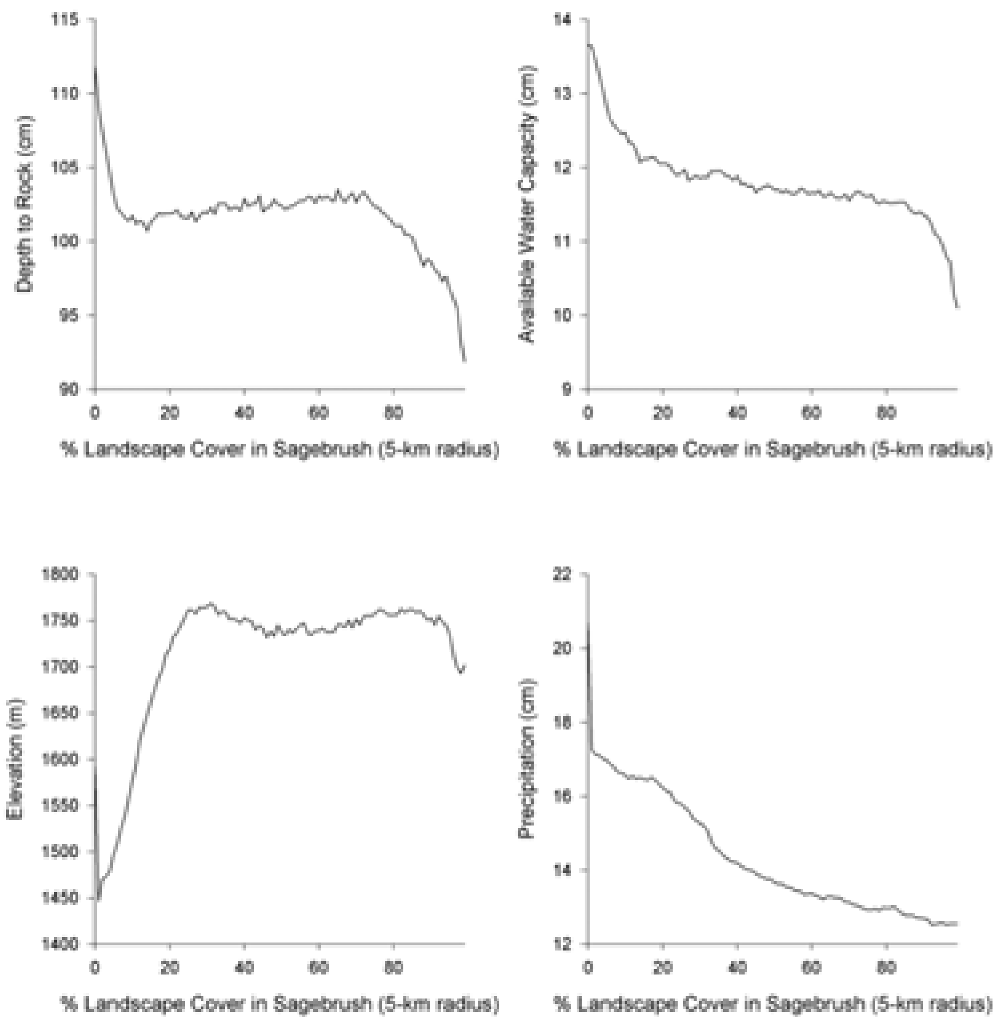


Fig. 5.16. Small-scale and large-scale fragmentation of sagebrush habitats, represented by the total numbers of edge between sagebrush and other habitats within a 5-, and 18-km radius of each 0.5 km grid cell. Total number of edge was determined by summing the number of edges in the sagebrush habitat map (Fig. 5.2) that differed between sagebrush and other habitats.

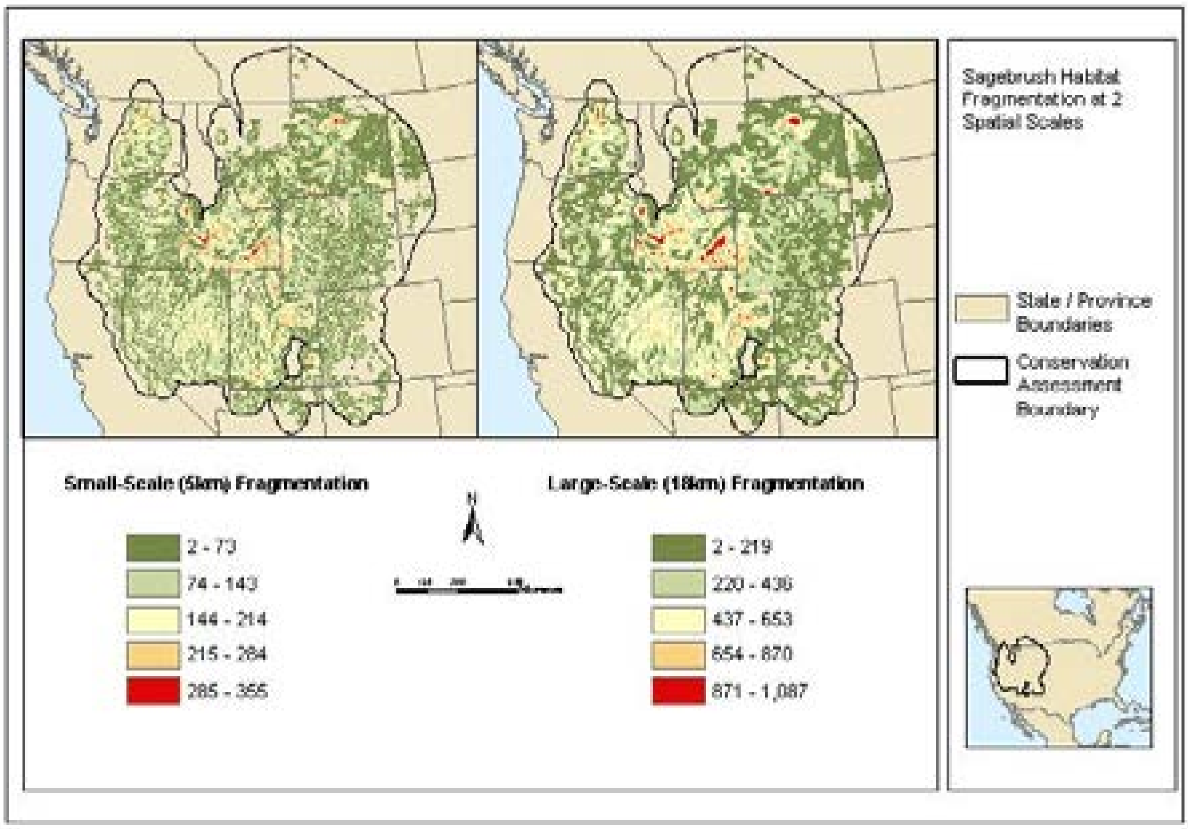


Fig. 5.17. Changing dominance of sagebrush cover and fragmentation across the sagebrush biome. The percent cover of sagebrush (Fig. 5.14) is represented in those regions containing >30% cover within the landscape. The number of habitat edges (Fig. 5.16) is shown in regions having <30% of the landscape in sagebrush cover and in which the fragmentation of the landscape is the dominant feature.

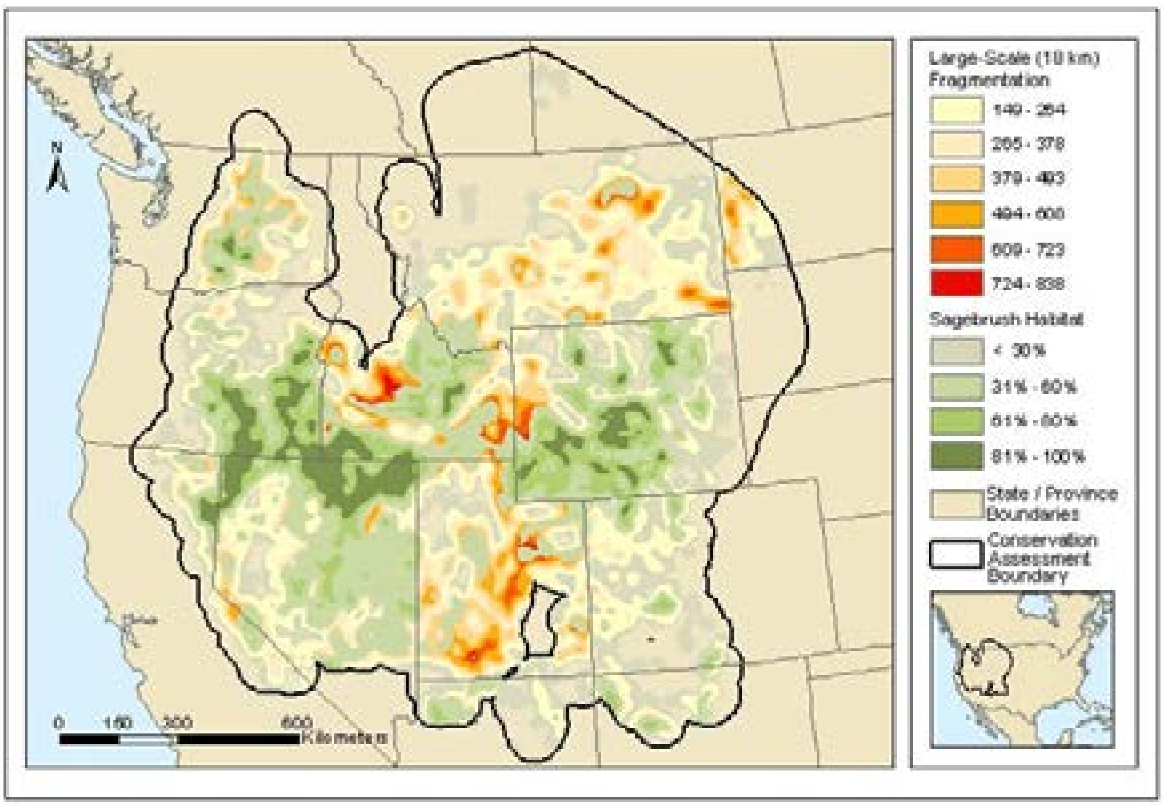


Fig. 5.18. Proportion of nonhabitat within a 5-, 18-, and 54-km moving window across the sagebrush biome. Nonhabitat included habitats other than sagebrush and grasslands >5 km from active leks.

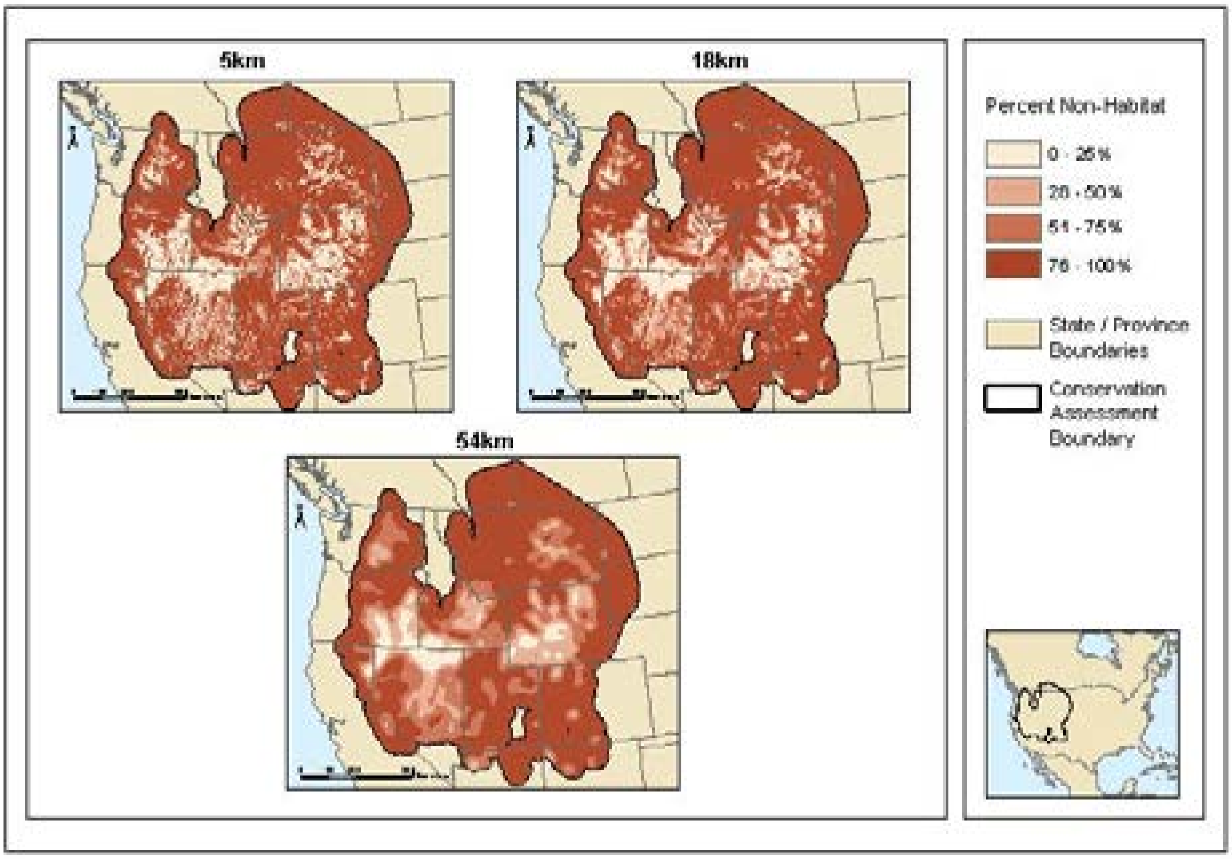
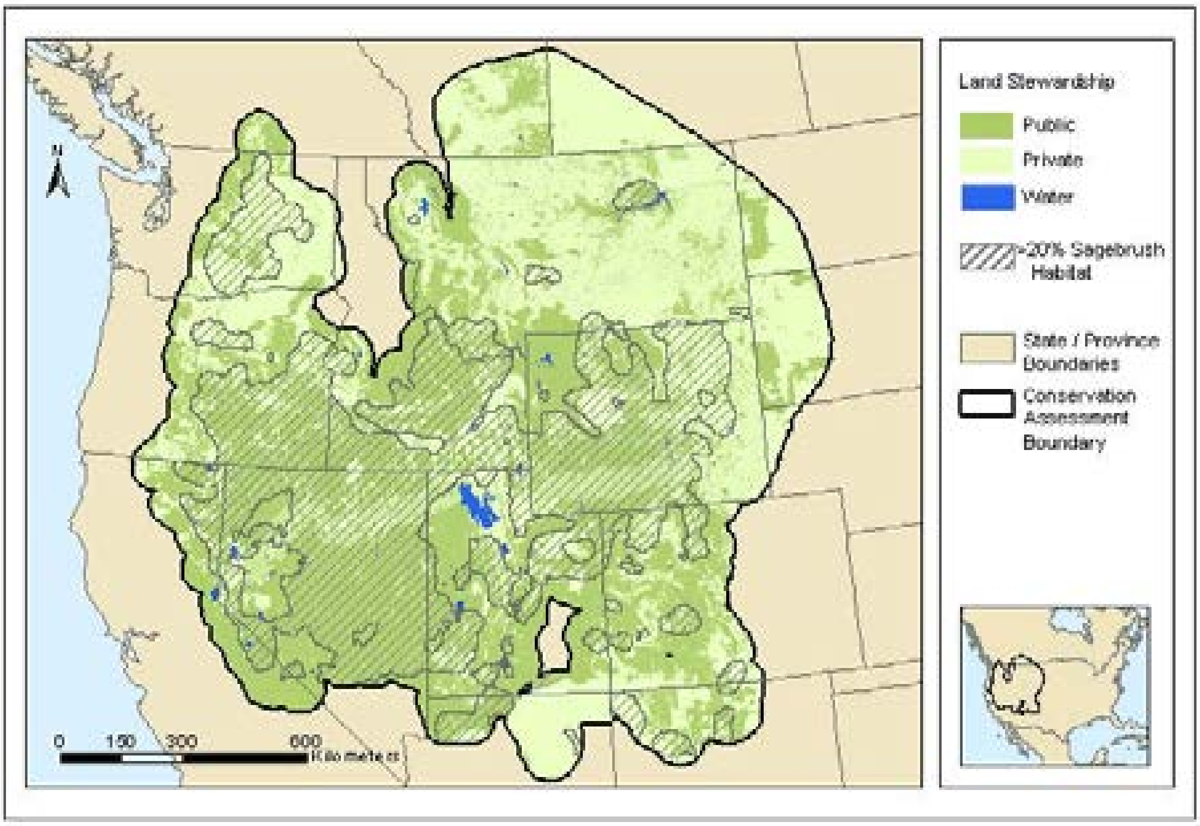


Fig. 5.19. Distribution of public and private lands within the sagebrush biome. Land ownership information was compiled from state GAP analysis programs, the U.S.G.S. National Land Cover Database, U.S. Bureau of Land Management, and individual state sources.



Tables

Table 5.1. Area occupied by sagebrush in the Intermountain Region estimated from Küchler's map of potential vegetation (Küchler 1970 and West 1983). The regional boundaries used for these area estimates were delineated in Küchler's map (1970) of potential vegetation (Fig. 5.3) and did not include eastern parts of sagebrush biome within the Conservation Assessment area.

State	Area (km^2)	Percentage of total area
Sagebrush Steppe		
Wyoming	109,000	24.3
Idaho	103,000	22.9
Oregon	92,000	20.5
Nevada	47,000	10.6
Washington	38,000	8.5
California	18,000	4.1
Colorado	17,000	3.8
Montana	13,000	2.9
Utah	11,000	2.4
Total	448,000	100.0
Great Basin Sagebrush		
Nevada	106,000	59.2
Utah	27,000	15.1
Arizona	24,000	13.5
Colorado	9,000	5.1
California	8,000	4.4
New Mexico	5,000	2.7
Total	179,000	100.0

Table 5.2. Summary statistics for environmental variables associated with sagebrush species sampled in the Colorado Plateau division (Fig. 5.2) of the sagebrush biome. Only sagebrush species recorded in ≥5 samples were included in the summary. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Basin big sagebrush	70	1,670[c]	3	29[b]	105	6.8	0.86[b]	14
Mountain big sagebrush	7	2,173[a]	5	46[a]	106	6.9	2.16[a]	13
Wyoming big sagebrush	15	1,754[bc]	9	35[b]	107	6.7	0.86[b]	16

Table 5.3. Summary statistics for environmental variables associated with sagebrush species sampled in the Columbia Basin division (Fig. 5.2) of the sagebrush biome. Only sagebrush species recorded in ≥5 samples were included in the summary. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Low sagebrush	54	1,273	8	46	78	5.6	0.11	10
Alkali sagebrush	10	1,388	5	30	97	5.8	0.41	8
Stiff sagebrush	186	664	9	30	79	6.1	0.27	10
Basin big sagebrush	33	841	16	35	85	5.8	0.23	12
Threetip sagebrush	21	848	19	31	72	6.2	0.20	8
Mountain big sagebrush	22	1,452a	2	24	134	5.8	0.26	11
Wyoming big sagebrush	467	682	13	30	93	6.2	0.32	13

Table 5.4. Summary statistics for environmental variables associated with sagebrush species sampled in the Northern Great Basin division (Fig. 5.2) of the sagebrush biome. Only sagebrush species recorded in ≥5 samples were included in the summary. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Low sagebrush	148	1,607	5	36	79	5.4	0.22	9
Alkali sagebrush	40	1,513	8	29	84	5.5	0.36	8
Silver sagebrush	24	1,472	1	36	122	6.5	1.84	14
Black sagebrush	102	1,693	10	38	96	5.5	1.50	10
Stiff sagebrush	31	1,459	6	34	73	5.3	0.32	11
Alpine sagebrush	6	1,911	4	35	105	5.9	2.06	12
Basin big sagebrush	151	1,490	4	29	114	6.4	2.36	12
Threetip sagebrush	14	1,571	3	31	69	5.7	0.35	7
Mountain big sagebrush	183	1,844	19	50	98	5.6	0.73	10
Wyoming big sagebrush	603	1,479	7	30	103	6.0	1.81	10
Bud sagebrush	62	1,354	3	23	141	7.6	4.01	13

Table 5.5. Summary statistics for environmental variables associated with sagebrush species sampled in the Snake River Plain subdivision (Fig. 5.2) of the sagebrush biome. Only sagebrush species recorded in ≥5 samples were included in the summary. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Low sagebrush	314	1,903	11	43	108	6.4	0.95	10
Alkali sagebrush	20	1,822	3	45	80	5.8	0.68	8
Silver sagebrush	39	1,782	5	41	100	6.1	0.55	12
Prairie sandwort	32	2,034	20	43	124	6.5	1.04	10
Black sagebrush	161	1,799	9	37	124	7.1	2.01	10
Stiff sagebrush	25	1,145	12	57	86	5.9	0.01	12
Threetip sagebrush	178	1,711	8	42	121	6.6	0.72	18
Basin big sagebrush	534	1,468	7	37	111	6.5	1.33	15
Mountain big sagebrush	831	1,992	22	55	109	6.3	0.74	12
Wyoming big sagebrush	1,140	1,533	8	33	112	6.4	1.55	12
Bud sagebrush	11	1,217	1	24	130	7.2	3.76	16

Table 5.6. Summary statistics for environmental variables associated with sagebrush species sampled in the Southern Great Basin division (Fig. 5.2) of the sagebrush biome. Only sagebrush species recorded in ≥5 samples were included in the summary. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Low sagebrush	378	2,165	16	41	92	6.1	0.98	8
Alkalai sagebrush	68	1,929	12	32	103	6.0	1.31	8
Bigelow sagebrush	7	1,782	8	22	78	6.0	3.96	9
Silver sagebrush	46	2,484	11	64	120	6.4	1.07	14
Black sagebrush	1,726	1,924	11	29	101	6.3	1.81	8
Pygmy sagebrush	10	1,819	2	25	124	7.2	2.26	11
Alpine sagebrush	7	3,126	28	85	89	5.1	--	7
Threetip sagebrush	13	2,025	14	42	98	6.0	1.35	10
Subalpine sagebrush	129	1,784	15	24	89	6.2	1.65	7
Basin big sagebrush	769	1,845	8	30	103	6.5	2.09	10
Mountain big sagebrush	1,518	2,274	24	50	88	6.0	0.71	9
Wyoming big sagebrush	2,521	1,832	7	29	120	6.7	2.33	11
Bud sagebrush	790	1,567	3	20	134	7.4	4.03	11

Table 5.7. Summary statistics for environmental variables associated with sagebrush species sampled in the Wyoming Basin division (Fig. 5.2) of the sagebrush biome. Only sagebrush species recorded in ≥5 samples were included in the summary. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Low sagebrush	97	2,059	10	36	111	7.4	1.34	14
Silver sagebrush	28	2,144	8	45	125	7.1	0.98	15
Prairie sandwort	11	2,310	18	35	67	6.5	1.58	9
Black sagebrush	147	1,976	13	30	75	6.7	1.42	10
Basin big sagebrush	160	1,969	8	33	102	7.2	1.48	13
Mountain big sagebrush	256	2,163	13	45	105	7.1	0.98	13
Wyoming big sagebrush	419	1,976	7	32	104	7.1	1.54	14

Table 5.8. Summary statistics for environmental variables associated with cheatgrass, and combined species of exotic grasses (including cheatgrass) and native grasses sampled in the Colorado Plateau division (Fig. 5.2) of the sagebrush biome. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Cheatgrass	78	1551	4	28	88	6.3	1.12	11
Exotic grasses	92	1645	5	31	88	6.3	1.12	11
Native grasses	198	1660	11	29	87	6.3	1.21	10

Table 5.9. Summary statistics for environmental variables associated with cheatgrass, and combined species of exotic grasses (including cheatgrass) and native grasses sampled in the Columbia Basin division (Fig. 5.2) of the sagebrush biome. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Cheatgrass	495	655	13	29	93	6.2	0.34	13
Exotic grasses	577	667	13	30	95	6.2	0.32	13
Native grasses	744	761	13	31	90	6.1	0.28	12

Table 5.10. Summary statistics for environmental variables associated with cheatgrass, and combined species of exotic grasses (including cheatgrass) and native grasses sampled in the Northern Great Basin division (Fig. 5.2) of the sagebrush biome. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Cheatgrass	705	1,456	10	30	111	6.4	2.52	11
Exotic grasses	777	1,462	9	30	110	6.3	2.39	11
Native grasses	923	1,591	8	35	99	5.8	1.45	10

Table 5.11. Summary statistics for environmental variables associated with cheatgrass, and combined species of exotic grasses (including cheatgrass) and native grasses sampled in the Snake River division (Fig. 5.2) of the sagebrush biome. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Cheatgrass	877	1379	10	34	106	6.4	1.55	13
Exotic grasses	1,502	1504	10	38	111	6.5	1.46	13
Native grasses	3,795	1787	13	45	111	6.3	1.15	12

Table 5.12. Summary statistics for environmental variables associated with cheatgrass, and combined species of exotic grasses (including cheatgrass) and native grasses sampled in the Southern Great Basin division (Fig. 5.2) of the sagebrush biome. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Cheatgrass	2,383	1,655	10	28	113	6.8	2.86	10
Exotic grasses	3,057	1,744	11	32	113	6.7	2.59	11
Native grasses	6,560	1,956	13	37	110	6.6	2.12	11

Table 5.13. Summary statistics for environmental variables associated with cheatgrass, and combined species of exotic grasses (including cheatgrass) and native grasses sampled in the Wyoming Basin division (Fig. 5.2) of the sagebrush biome. Percent cover was not reported because of differences in sampling methods.

	n	Elevation (m)	Slope	Precipitation (cm)	Depth to Rock (cm)	Soil pH	Soil Salinity	Available water capacity (cm)
Cheatgrass	172	1,843	9	27	88	6.6	1.89	11
Exotic grasses	298	1,940	10	39	100	6.7	1.43	12
Native grasses	1,099	2,093	12	44	103	6.9	1.23	13

Table 5.14. Correlations of environmental variables with ordination axes (Fig. 5.6) using a detrended canonical correlation analysis of sagebrush communities and environmental variables.

	Correlations of Environmental Variables with Axes	
Variable	Axis 1	Axis 2
Elevation (m)	-0.47	-0.37
Slope	-0.22	0.05
Aspect	-0.02	0.01
Precipitation (cm)	-0.46	0.02
Depth to rock (cm_	0.04	0.03
Soil pH	0.08	0.01
Soil Salinity	0.26	0.00
Available water capacity (cm)	0.01	0.15

Table 5.15. General structure and dominant species groups representing major existing post-settlement steady states, which occur across *Artemisia* plant associations.

Community Structure	Conifer sp.	Artemisia sp.	Native Perennials	Introduced Perennials	Exotics[a]
Shrublands					
Shrub-steppe		√	√		c
Shrubland[b]		√			
Shrub-exotic herbland[b]		√			√
Herblands					√
Exotic herbland					
Perennial grassland				√	
Woodlands	√		√		
Woodland native understory					
Woodland	√				
Woodland exotic understory[b]	√				√

[a]Exotics represent a range of annual, biennial, and perennial Eurasian and Mediterranean weedy species including cheatgrass, medusahead, knapweed sp., leafy spurge, and others.
[b]Steady states that will probably shift to annual-biennial-perennial herbland following fire.
[c]Exotics often present but in low abundance

Table 5.16. Percent of sagebrush in Küchler's (1970) map of potential vegetation (Fig. 5.3) currently in sagebrush habitat. Only comparisons for Küchler's categories of Great Basin sagebrush, sagebrush steppe, and wheatgrass-needlegrass shrubsteppe were used in this analysis. Remaining percentage consisted of habitat categories describing barren, burn, grassland, exotic grassland, burn, nonsagebrush shrublands, and juniper woodland habitats.

		Current Condition			
	Potential Area	Sagebrush		Agriculture	
State	(km^2)	km^2	(%)	km^2	(%)
Washington	32,645	13,945	(42.7)	15,880	(48.6)
Montana[a]	9,321	3,953	(42.4)	585	(6.3)
Wyoming	91,783	61,151	(66.6)	5,525	(6.0)
Idaho	88,084	43,438	(49.3)	21,966	(24.9)
Oregon	71,873	44,345	(61.7)	9,044	(12.6)
Nevada	130,542	73,079	(56.0)	2,910	(2.2)
Utah	30,976	11,253	(36.3)	4,105	(13.3)
California	13,463	8,515	(63.2)	1,444	(10.7)
Colorado	18,409	8,860	(48.1)	3,670	(19.9)
Total	487,095	268,539	(55.1)	65,129	(13.3)

[a]Sagebrush lands in eastern Montana outside the boundaries of the Küchler's vegetation types used in this analysis.

Table 5.17. Characteristics of lands within the sagebrush biome under different management authority and ownership status. We determined characteristics for lands managed by the U.S. Bureau of Land Management (BLM), U.S.D.A. Forest Service (USFS), and combined totals for individual state agencies. Fragmentation was the number of edges in the gridded map (Fig. 5.16) between sagebrush and other habitats in the landscape within a 5- and 18-km radius.

Stewardship	Sagebrush Cover (% 5-km)[a]	Sagebrush Cover (% 18-km)[a]	Precipitation (cm)	Elevation (m)	Salinity	Soil pH	Available Water Capacity (cm)	Soil Depth (cm)	Fragmentation (5-km)	Fragmentation (18-km)
Private	16	16	39	1,356	1.92	6.7	15	114	87	263
BLM	41	39	30	1,694	2.24	6.4	10	99	101	321
USFS	10	12	75	2,169	0.46	5.3	11	104	100	290
State Agency	18	17	39	1,551	1.93	6.6	13	109	104	315
BLM / USFS	7	10	64	2,462	0.48	4.8	8	84	54	163
Other	15	15	38	1,478	2.10	5.9	11	94	79	230

[a]Determined from GIS coverage of % cover of sagebrush within 5-km radius (Fig. 5.14)

Chapter 6

Greater Sage-Grouse Populations

CHAPTER 6

Greater Sage-Grouse Populations

Abstract. This chapter presents information on three different but related subjects. 1) State and province sage-grouse population databases. To obtain information on the scope and extent of population databases we mailed a detailed questionnaire to 11 western states and 2 Canadian provinces. We requested information on methods used for monitoring sage-grouse (*Centrocercus urophasianus and C. minimus)* populations, production, and harvest, as well as information on data storage and data retrieval. Results from our questionnaire indicated monitoring techniques vary among areas and years both within and among agencies. This variation complicates attempts to understand grouse population trends and make comparisons among areas. Moreover, there were discrepancies between information reported by agencies in the questionnaire and that obtained from the agencies' lek databases. 2) Sage-grouse distribution. We presented information on the range-wide distribution of sage grouse. The overall distribution of potential pre-settlement habitat was estimated to have been 1,200,483 km^2 and the current distribution to be 668,412 km^2. Approximately 56% of the potential pre-settlement distribution of habitat is currently occupied. Future examinations of regional habitat and habitat change should provide more insight into long-term changes in the distribution of sage-grouse. The area currently occupied by sage-grouse is clearly smaller than was occupied in pre-settlement times. Declines in distribution have been noted throughout the twentieth century. 3) Sage-grouse population trends. We conducted a comprehensive analysis of sage-grouse population changes throughout their range by accumulating and analyzing all available male counts at 5,585 leks identified since agencies began routine monitoring of this species. A substantial number of lek routes and leks are censused each year throughout North America and many of these databases have > 30 years of information. We discuss problems associated with the collection and analysis of these data sets but they represent the only long-term database available for sage-grouse. Virtually all states and provinces have increased monitoring efforts, especially over the last 10 years. Our analysis indicated that a total of 2,637 leks are now censused annually. We used three different but related methods to assess population trend. Eleven of 13 (85%) states and provinces showed significant long-term declines in size of active leks. Similarly, eight of 10 states (80%) showed population declines over that time frame. Two of 10 (20%) appeared to be stable or slightly increasing. Only California had an increase in both the population index and lek size. Annual rates of change suggest a long-term decline for sage-grouse in western North America and generally support the trend information obtained from lek attendance (males/lek). Range-wide sage-grouse populations declined at an overall rate of 2.0% per year from 1965 to 2003. This annual rate of decline was much higher during the first two decades of our analysis period (1965-86) compared to the last two decades (1986-2003). Although a total of 50,566 male sage-grouse were counted on leks in 2003 throughout western North America, long-term population changes coupled with continued loss and degradation of habitat and other factors (including West Nile Virus) do not provide causes for optimism.

POPULATION DATABASES

Introduction

Greater sage-grouse populations have been declining for at least 25 years (Braun 1995, Connelly and Braun 1997, Aldridge and Brigham 2003, Beck et al. 2003, Schroeder et al. 2004). Because of concerns about this species and its habitats, appropriate monitoring efforts have become more important. Connelly et al. (2000) indicated that monitoring was a major component of a sage-grouse management program. Additionally, representatives to the Western Association of Fish and Wildlife Agencies (WAFWA) signed memorandum of agreements among themselves (1999) and with federal agencies (2001) to collect data in a manner recommended by the Western States Sage and Columbian Sharp-tailed grouse Technical Committee.

As part of the Conservation Assessment for greater sage-grouse, we attempted to obtain information on current sage-grouse monitoring programs by mailing a questionnaire to all state and provincial agencies (n = 13) that manage this species. Here we summarize the responses to this questionnaire and identify strengths and weaknesses of current data sets.

Methods

We mailed a detailed questionnaire to 11 western states and 2 Canadian provinces. This questionnaire requested information on methods used for monitoring sage-grouse breeding populations, production, and harvest. We also requested information on data storage and retrieval. To aid biologists responding to this questionnaire, we provided the following definitions:

Lek—a traditional display area where > 2 male grouse have attended in > 2 of the previous five years.

Lek count—a tally of male sage-grouse on a lek or group of leks with no assumption that the leks represent all or part of a single breeding population.

Lek route—A count of male sage-grouse on a group of leks that are relatively close and represent all or part of a single breeding population.

Lek survey—A classification of leks as active or inactive, often done from an aircraft.

All states and provinces returned completed questionnaires. Because no respondents indicated those questions were difficult to understand or ambiguous, we did not follow up with additional questions or phone calls. Data reported in this section refer to only information obtained from the questionnaires and not to databases subsequently obtained from states and provinces.

Results

Population Data

All state and provincial fish and wildlife agencies monitor sage-grouse breeding populations annually, but different approaches are employed (Table 6.1). Five agencies use only lek counts. Eight use a combination of techniques: three agencies use lek counts and lek routes; one agency uses lek counts and lek surveys; four agencies use lek counts, lek routes, and lek surveys.

Responses from agencies indicated that a total of 237 lek routes and 2,046 leks are censused throughout North America (Table 6.1). Within the same area, 2,304 leks were surveyed. Montana, Wyoming, and Idaho count the greatest number of leks (498, 375, 352, respectively). Actual values from state databases are presented later in this chapter.

Twelve agencies indicated that their monitoring data were widely distributed across the range of the species within their state or province. Ten agencies (77%) reported starting monitoring programs in the 1940s or 1950s. Two (15%) started programs in the 1960s or 1970s, and one agency started monitoring in the 1990s (Table 6.1). Of the 12 agencies responding, 11 indicated that population data were obtained over multiple administrative units (range = 2-35, x = 10). These administrative units were counties, agency-delineated regions, or hunting units.

Table 6.1. Year monitoring started, number of lek routes run, and leks counted or surveyed in western states and provinces, 2002.[1]

State/Province	Year Started	Number of Routes	Leks Counted	Leks Surveyed
Alberta	1968	0	32	27
California	1953	3	64	100
Colorado	1959	0	278	0
Idaho	1951	51	352	273
Montana	1950s	4	498	0
North Dakota	1951	0	17	25
Nevada	1950s	4	110	994
Oregon	1947	39	124	46
Saskatchewan	1994	0	35	0
South Dakota	1972	0	20	0
Utah	1959	0	170	0
Washington	1954	0	20	0
Wyoming	1949	139	375	831
Totals		240	2,095	2,296

[1]These data were reported by the agencies and do not necessarily reflect information obtained from the agencies' databases for the conservation assessment.

Although most agencies indicated that they attempted to replicate counts of leks over several weeks (i.e., counting individual leks or lek routes >3 times), at least 2 agencies attempt to complete all counts within a 1-week period and one only counts leks once during this time. In addition, some states provided data indicating leks were censused at inappropriate times (late February, early to mid-March, mid-May). Eight (62%) agencies indicated gaps in their databases since initiating monitoring efforts and five (38%) reported relatively continuous databases. Eleven of 13 (85%) agencies reported changing inventory methods over the years.

Harvest Data

Ten of 11 states have hunting seasons. Sage-grouse are not hunted in Washington, Alberta and Saskatchewan. All states with a hunting season collected wings (Table 6.2). Three states collected >1000 wings each year, 5 states collected 200-600 wings each year and 2 states collected < 50 wings per year. The number of administrative units that each state used for data analysis varied greatly among states (Table 6.2) and seven states analyzed data by administrative unit (usually counties or game management units). Only four states averaged >150 wings per administrative unit (range = 200-397 wings), while six states averaged <100 wings per administrative unit (range = 8-83).

Table 6.2. A summary of sage-grouse wing and harvest surveys from western states as of 2002.

State/Province	Number wings[1]	Administrative Units	Years with current survey[4]
California	150	4	16
Colorado	250	3	4
Idaho	1986	34[2]	3
Montana	200	4[2]	20
North Dakota	30	1	25
Nevada	2500	10[3]	38
Oregon	550	6	9
South Dakota	8	1	3
Utah	325	4	2
Wyoming	1440	18[2]	46

[1]Approximate number collected annually.
[2]Does not analyze data by administrative unit.
[3]Range of 7-10 units given.
[4]Indicates years of current technique for estimating sage-grouse harvest and other data.

Nine of ten agencies reported providing personnel with training for classifying wings. However, of the nine agencies with training, two indicated that wings were only read by a certain group or an individual (apparently indicating quality control). Only five of nine agencies indicated they conducted some kind of annual training. All states (n = 10) obtained age and gender information from wings. Additionally, nine of 10 states obtained production data (juvenile to adult ratios), six recorded the proportion of successful hens, and five assessed hatching distribution.

All states with a hunting season conducted harvest surveys, but the states employed seven different techniques for obtaining harvest information. Five states indicated that they contacted 75-100% of grouse hunters, two states indicated that they contacted 10-30% of grouse hunters, and three states reported that they did not know what percent of grouse hunters they contacted. Information obtained from harvest surveys included number of grouse harvested, number of hunter days, hunter success, number of hunters, county of harvest, hours/hunter/day, number hunters/party, hunting trips/county, birds/trip, hunter satisfaction, use of dogs, location of hunting area, wounding rates, weapons used, and number of grouse seen. However, the information recorded varied substantially from state to state. Harvest survey techniques have been in place nine or more years for six states (Table 6.2) and the average of all states was 16.6 years (range = 2-46 years, sd =15.7).

Production Data

Only Oregon and California reported conducting routine production surveys in addition to collecting wings. California indicated that they conducted production surveys in some areas to monitor for "abnormalities" and track key brood rearing areas. None of the agencies that did not have hunting seasons routinely monitored production. Thus, 10 of 11 agencies assessed production with wing collections and in two cases also with brood counts.

Data Storage and Retrieval

Nine of 13 (69%) agencies stored their data in two or more formats. Five of 13 (38%) indicated that at least some of their data were stored on paper (i.e., information was not in a database or spreadsheet program). Only four of 13 (31%) agencies reported that their data were stored in a single format (e.g., Excel).

Seven of nine agencies reported that they could transmit data to the Conservation Assessment Team in two weeks or less. Unfortunately, four agencies indicated it would take 2-6 months and one of these said that it could take up to a year. Montana did not provide an estimate.

Eight of 13 agencies (62%) rated their monitoring data as good; an additional state indicated that some of their data were good. Four of 13 (31%) indicated that their monitoring data was fair and one state reported that about 60% of their data was fair or poor. Of these five states, four (ID, OR, CO and NV) have long histories of sage-grouse work and all four started breeding population data collection in the 1940s or 1950s.

Discussion

Sage-grouse populations in parts of western North America have been monitored in some fashion for over 50 years (Patterson 1952, Dalke et al. 1963). In a non peer-reviewed report, Autenrieth et al. (1982) recognized the variation in monitoring efforts among agencies. In an attempt to improve data collection, the Western States Sage Grouse Committee (now the Western States Sage and Columbian Sharp-tailed Grouse Technical Committee) issued a bulletin on sage-grouse management practices (Autenrieth et al. 1982). This report, in part, attempted to standardize population data collection techniques and describe methods available for documenting sage-grouse population characteristics. However, results from our questionnaire indicated monitoring techniques continue to vary among areas and years both within and among agencies. This variation complicates attempts to understand grouse population trends and make comparisons among areas. Moreover, there were many discrepancies between information reported by agencies in the questionnaire and that obtained from the agencies' lek databases. This suggests that until very recently many agencies did not have a clear understanding of their data or methods for collecting it. Clearly, this assessment has resulted in most agencies closely examining their sage-grouse monitoring program. The sage-grouse guidelines (Connelly et al. 2000) stressed the importance of population monitoring and collecting quality data in sage-grouse management programs. Information from the questionnaire suggests some recent effort among agencies to improve or update data collection efforts.

Population Data

Although lek counts are widely used, a non-peer reviewed report (Beck and Braun 1980) for the Western Association of Fish and Wildlife Agencies questioned their usefulness. However, techniques for correctly conducting lek counts have been described (Jenni and Hartzler 1978, Emmons and Braun 1984) and problems generally seem to be related to disregarding accepted techniques. We did not question agencies about lek counting procedures because lek data submitted by states and provinces provided insight into methods used. It was clear that some agencies attempted to follow accepted techniques while others developed their own approach. An evaluation of lek data indicated that some leks were counted incorrectly, because observers collected data too early or late in the breeding season, in poor weather and/or later in the morning.

Because sage-grouse gather on traditional display areas (leks) each spring, biologists are afforded relatively easy methods for tracking breeding populations (Connelly et al. 2003). These methods have included lek censuses (annually counting the number of male sage-grouse attending leks in a given area), lek routes (annually counting the number of male sage-grouse on a group of leks that are relatively close and represent part or all of a single breeding population or deme), and lek surveys (annually counting the number of active leks in a given area). All lek monitoring procedures are supposed to be conducted during early morning (1/2 hour before to 1 hour after sunrise), with reasonably good weather (light or no wind, partly cloudy to clear) from early March to early May (Connelly et al. 2003). Timing of lek monitoring is dependent on elevation of leks and persistence of winter conditions. Sage-grouse will begin displaying in late February at lower elevations with milder climates and in years with mild winter weather (e.g., southern Washington). Lek attendance will persist into early or mid-May at higher elevations.

A substantial number of lek routes and leks are censused each year throughout North America and many of these databases have > 30 years of information. Virtually all agencies report monitoring sage-grouse throughout much of the species' range within their respective state or province. However, methods for gathering these data vary widely among agencies and sometimes within agencies among years.

Responses to the questionnaire suggested that the same leks or leks within the same area, have been counted for long periods of time. These leks were likely originally selected because of size (leks with many males), access, or both reasons. In any case, leks that are censused in most states and provinces are probably not a random sample of available leks and thus data obtained from these leks may be biased (but see Appendix 3). However, some states and provinces attempt to monitor all of their leks.

Lek counts focus on attendance of males. Male sage-grouse sometimes do not attend a lek or may attend two or more different leks (Jenni and Hartzler 1978, Walsh 2002). Lek data used to track population trends have an implied assumption that the probability of detection of birds does not change among years (i.e., the proportion missed because of non-attendance or attendance at a lek that is not counted remains about the same). Even if the detection probability is unknown (almost always the case), the problem can be somewhat minimized, and thus provide more precise counts, if leks counted on a single morning are relatively close and represent all or a significant part of a given breeding population. Thus lek routes are preferable to lek counts because they should increase the probability of detection of male grouse (i.e., if a male grouse normally is counted on lek "A" but spends 20% of his time on lek "B" and both leks are counted on the same morning, the probability of detecting that bird is greater than if just lek "A" were counted).

Harvest Data

Although numerous wings are collected in many states and the wings subsequently classified in "wing-bees", numbers may be insufficient to characterize populations, depending on the number of administrative units used for analysis. In a non-peer-reviewed report, the Western States Sage and Columbian Sharp-tailed Grouse Technical Committee (Autenrieth et al. 1982) suggested that at least 100 wings from adult and yearling hens are needed from each population to obtain meaningful data. Questionnaire responses suggest six of 10 states have less than 100 wings per administrative unit. Thus, at least 60% of states likely do not obtain adequate samples of wings to assess production. Moreover, these wings represent both genders and age groups, further indicating that sample size is quite low. Additionally, because grouse are often hunted on summer ranges and birds from two or more breeding populations may use the same summer ranges (Connelly et al. 1988, Wakkinen 1990, Fischer 1994, Wik 2002), production data may be difficult to interpret. Collection techniques currently do not appear designed to reflect known breeding populations. Because of low sample sizes, lack of wings in some states and provinces, and an inability to relate wing surveys to breeding populations we did not use wing data to help assess sage-grouse population trends.

Training or other quality control practices are used in most states but only five of nine states indicated annual training for wing-bee participants. One state indicated that each participant was tested before beginning to read wings. Where participants remain the same each year, frequent training may not be necessary.

All states with hunting seasons monitor sage-grouse harvest. However there seems to be a great deal of uncertainty associated with harvest estimates. Seven different techniques are used among the 10 agencies that administer hunting seasons and a variety of information is obtained from these techniques. This information varies tremendously among states.

Production Data

Brood routes are not a commonly used technique for monitoring production. Oregon uses these data in conjunction with breeding population data to develop hunting seasons (W. Van Dyke, Oregon Department of Fish and Wildlife, personal communication) and apparently is the only state to use these data for this purpose. The state and provinces without hunting seasons do not use brood routes and thus have no means of assessing production, except by using studies involving radio telemetry.

Data Storage and Retrieval

Agencies not only vary greatly in how they collect data but also how they manage databases. However, most agencies indicated that they could compile and transmit their data rather quickly. Interestingly, 62% of the agencies had a similar view of their data by rating it as good. Agencies that took a more critical view of their data sets were involved with long-term monitoring and research programs on sage-grouse. This emphasis may have provided these agencies with a better understanding of the quality and quantity of their data compared to agencies whose sage-grouse management program is only comprised of monitoring.

DISTRIBUTION

The general distribution of sage-grouse is clearly associated with distribution of sagebrush (*Artemisia* spp.), in particular, big sagebrush (*A. tridentata*), but also silver sagebrush (*A. cana*). This relationship has been illustrated in numerous descriptions or maps of the sage-grouse distribution (summarized in Schroeder et al. 2004). Schroeder et al. (2004) revised the current and pre-settlement (past, original, or historical) sage-grouse map with numerous sources of information including: 1) past maps showing distribution of sage-grouse (e.g. Aldrich and Duvall 1955); 2) maps and research on the distribution of habitat types in western North America (e.g. Kuchler 1985); 3) recovery locations for 1,167 sage-grouse museum specimens (Schroeder et al. 2004); 4) published sage-grouse observations, many from the 1800s (Schroeder et al. 2004); 5) known information on current and past occurrence of sage-grouse, such as lek locations; and 6) research elucidating the movement (Connelly et al. 1988) and habitat use (Schroeder et al. 1999, Connelly et al. 2000) of sage-grouse. 'Pre-settlement' was used to define the period prior to 1800 before rapid settlement by people of European descent, particularly in Nevada, Oregon, and Utah (Miller and Eddleman 2001).

Seven core habitats from Kuchler's (1985) map were used to help define the pre-settlement distribution (see Chapter 5, Fig. 5.3). The seven core habitats that supported the most sage-grouse included: 1) sagebrush steppe, 2) Great Basin sagebrush, 3) wheatgrass (*Agropyron spicatum*)-needlegrass (*Stipa* spp.) shrubsteppe, 4) grama (*Bouteloua* spp.)-needlegrass-wheatgrass, 5) wheatgrass-needlegrass, 6) wheatgrass-bluegrass (*Poa* spp.), and 7) fescue (*Festuca* spp.)-wheatgrass. Although Kuchler's habitat map indicates some of these core habitats are not dominated by sagebrush, a key component of sage-grouse habitat, data from portions of these regions (e.g., Daubenmire 1970, Brown and Lowe 1980, Jacobson and Snyder 2000) indicates that sagebrush may be locally abundant within definable portions of an otherwise grass-dominated habitat type. Consequently, only sagebrush-dominated portions of three core habitat types (wheatgrass-needlegrass, wheatgrass-bluegrass, fescue-wheatgrass) were mapped as part of the pre-settlement distribution of potential sage-grouse habitat. Comparison of Kuchler's (1985) map with known information on sage-grouse abundance and habitat use illustrated the existence of several 'secondary' habitats. Secondary habitats are characterized by variation in suitability due to tree abundance, sagebrush type and density, and connectivity and proximity to core habitats (Connelly et al. 2000, Miller and Eddleman 2001). Secondary habitats include: 1) foothills prairie, 2) saltbush (*Atriplex* spp.)-greasewood (*Sarcobatus vermiculatus*), 3) juniper (*Juniperus* spp.)-pinyon (*Pinus edulis*) woodland, 4) grama-buffalo grass (*Buchloe dactyloides*), and 5) desert. Secondary habitats were mapped locally in specific situations where the habitats: 1) are currently occupied, 2) were clearly occupied in the past, and 3) are generally within 10 km of core habitats. Habitats without known use by sage-grouse were excluded from the pre-settlement distribution of potential habitat, even if there were scattered observations and/or recoveries of sage-grouse.

The map revisions were designed to result in the following improvements: 1) mapping at larger scale to improve precision with respect to topography and habitat types (1:2,000,000); 2) placement in Geographical Information System (Arc/INFO 1998) to permit additional analysis (Wisdom et al. 2002*a, b*); 3) elimination of habitat that was clearly unsuitable (e.g., forest); 4) inclusion of suitable, or potentially suitable, habitat that had been excluded from earlier maps; and 5) integration of a map with at least 30 years of lek surveys throughout the range.

The descriptions of the sage-grouse distribution have been divided into general regions similar to those described by Miller and Eddleman (2001). These include the Wyoming Basin (W two-thirds of Wyoming, NE Utah, SE tip of Idaho, NW Colorado, SW Montana), Snake River Plain (S Idaho, NW Utah, NE Nevada, E Oregon), Columbia Basin (north-central Oregon, E Washington, south-central British Columbia), northern Great Basin (south-central Oregon, NE California, NW Nevada), southern Great Basin (east-central California, S two-thirds of Nevada, W and central Utah), and Colorado Plateau (only the NW Arizona and southern Utah). The Great Plains or silver sagebrush region (E and S Montana, W North Dakota, W South Dakota, NW Nebraska, E Wyoming, SW Saskatchewan, SE Alberta) has been added to include areas outside the scope of Miller and Eddleman's (2001) map (Figure 1.5). Although the sage-grouse maps originally included the Gunnison (*C. minimus*) and greater species (Schroeder et al. 2004), only the greater sage-grouse is described in detail here.

General

The overall distribution of potential pre-settlement habitat was estimated to have been 1,200,483 km^2 and the current distribution to be 668,412 km^2 (Chapter 1, Fig. 1.1) (Schroeder et al. 2004). Approximately 56% of the potential pre-settlement distribution of habitat is currently occupied. Specimens from Washington and Oregon were believed to be the western subspecies, specimens from California were believed to be an intermediate form, and all other specimens were believed to be the eastern subspecies (Aldrich and Duvall 1955). However, we made no attempt to quantify the respective distributions of the eastern and western subspecies (*C. u. urophasianus* and *C. u. phaios*, Aldrich 1946) because of the lack of a clear dividing line (Aldrich and Duvall 1955) and the lack of genetic differentiation (Chapter 8, Benedict et al. 2003).

Great Plains

Members of the Lewis and Clark expedition observed sage-grouse in western Montana, but not in central and eastern Montana (Zwickel and Schroeder 2003). Swainson and Richardson (1831:359) reported that sage-grouse "do not exist on the banks of the river Missouri; nor have they been seen in any place east of the Rocky Mountains." Audubon mentioned that sage-grouse were observed by a member of his expedition prior to 1843 at some point on the Yellowstone River, however Audubon himself did not observe sage-grouse along the Missouri (Audubon 1960). Coues (1874) was the first to mention sage-grouse in central Montana. In contrast to earlier accounts, Bendire (1892) suggested the area of sage-grouse occupation included most of Montana and western North Dakota, stretching about 50 km north of the U.S.-Canadian border along the upper tributaries of the Missouri River. An examination of museum specimens and published observations supports the past occurrence of sage-grouse up to 240 km north of the U.S.-Canadian border, however specimens and observations more than 100 km north of the border are all more recent than 1945 (Schroeder et al. 2004).

In his "List of Birds observed at Grand River Agency, Dakota Territory, from October 7, 1872 to June 7, 1873, W.J. Hoffman M.D., the Acting Assistant Surgeon for the U.S. Army stationed near the confluence of the Missouri and Grand Rivers in Dakota Territory reported that the "Sage Cock" *Centrocercus urophasianus* was not often found near the Agency but were often brought into the agency by the Indians "who shot them on the plains where *artemisia* occurs." A 1914 report by Stephen S. Visher, published in Bulletin Number 6 of the South Dakota Geological Survey details the Biology of Harding County Northwestern South Dakota at that time. Visher indicated that sage-grouse "were formerly found in many sections of Western South Dakota and westward." Visher further stated in his report that the sage hen was an:

> "*abundant resident in the areas covered with scrub-bush (*Artemisia tridentata*), where water is far distant; therefore mainly found on the terraces in the stream valleys. The last ones recorded from this state, except in the northwestern corner, were found in Sage Creek in the Badlands in 1907. By 1910 all were gone except those in Harding and Butte Counties. Now, (1913) after three more years of homesteading Sage Grouse are restricted in this state to the Little Missouri Valley in Harding County and to the headwaters of Indian Creek in Butte. In a very few years they will occur in South Dakota only as a rare winter straggler from Montana."* This information is contained in the South Dakota Geological Survey; Bulletin Number Six; Report of State Geologist; State Publishing Co., Pierre, SD and was provided to us by John Wrede (South Dakota Game, Fish and Parks).

The dates and locations of observations between 1805 (Meriwether Lewis in Moulton 1987) and the mid-1900s support the possibility of a northward transition in distribution. However, there are limited data regarding the pre-settlement distribution of sagebrush throughout the region and regional variation in the density of ungulates (Martin and Szuter 1999, Lyman and Wolverton 2002). Additionally, sage-grouse are known to use alternate species of sagebrush such as silver sagebrush in Alberta, Saskatchewan, North Dakota, and South Dakota (Sealy 1963; Aldridge and Brigham 2002, 2003; Smith 2003). The lack of solid data on the presence of sage-grouse in most areas precluded attempts to divide the distribution into 'originally occupied' and 'acquired' portions. As a result, most observations and specimens within Alberta and Saskatchewan are included in the potential pre-settlement distribution of habitat (Schroeder et al. 2004).

Most museum specimens and early observations of sage-grouse in North and South Dakota are from an area that is either currently occupied or close to an area that is occupied (Schroeder et al. 2004). The absence of sage-grouse in most of North and South Dakota is further supported in a review by Johnson and Knue (1989) where they report the absence of sage-grouse remains at 27 of 29 Indian villages where sharp-tailed grouse (*Tympanuchus phasianellus*) remains were found. Consequently, the potential pre-settlement distribution of habitat was delineated to be consistent with most specimens and observations. The revised distribution of habitat includes southwestern North Dakota, western South Dakota (except for forested portions of the Black Hills), and northwestern Nebraska, and does not extend into central North Dakota as shown by Aldrich and Duvall (1955).

Wyoming Basin

Most early observations in this region were in south-central or southwestern Wyoming, due in part to the location of a prominent travel corridor over South Pass (Stansbury 1852, Frémont 1887, Thwaites 1978). Field (1857) stated that sage-grouse were supported by vast expanses of sagebrush, particularly in southwestern Wyoming. Although there are no published early observations of sage-grouse in eastern portions of their current distribution in this region, it is not clear if this represents a shift in distribution or a bias related to their abundance in different areas and habitats. Aldrich and Duvall (1955) illustrated both current and past distribution of sage-grouse as being relatively continuous in most of the Wyoming Basin, with a general absence in the Grand Tetons, Yellowstone, and Wind River Mountain areas of Wyoming. However, sage-grouse currently occur around Jackson, Wyoming. The distribution of forested and alpine habitats also indicates that greater sage-grouse were likely absent from portions of the Big Horn Mountains and Black Hills in Wyoming, Uinta and Wasatch mountains in Utah, and Uncompahgre Plateau, Gore Range, and Flat Top Mountains in Colorado (Rogers 1964, Braun 1995). More than 6,000 birds were captured in Wyoming between 1940 and 1951 and moved to other areas in Wyoming with existing populations, but some were released in New Mexico within the former range of the Gunnison sage-grouse (Patterson 1952).

Snake River Plain

The area on both sides of the continental divide near Lemhi Pass has been occupied by sage-grouse for at least the last 200 years (Zwickel and Schroeder 2003). Aldrich and Duvall (1955) also showed an area of historical occupation by greater sage-grouse along the Bitterroot Valley in northwestern Montana; Aldrich's revised map in 1963 did not include this area. One specimen apparently was collected near Missoula in 1900 (Schroeder et al. 2004). This area generally corresponds with the foothills prairie habitat (Kuchler 1985) that is occasionally occupied by sage-grouse in other portions of Montana. In 1942, 242 birds were captured in Montana and released in several locations including the Bitterroot Valley, an effort that was ultimately unsuccessful in establishing (or re-establishing) a population (Reese and Connelly 1997).

Current populations in the region occupy substantial areas of habitat that appear continuous across multi-state borders. The largest of these occupies the tri-state area of Idaho, Nevada, and Oregon, but smaller populations appear to cross other borders. Most of the fragmentation in populations is associated with the extirpations along the Snake River and its tributaries. In 1986, 196 sage-grouse were captured in Idaho and translocated to the Sawtooth Valley in an attempt to augment an isolated population that had declined to one displaying male. Although the ultimate fate of this translocation effort is unclear (Musil et al. 1993), population data suggests that it might be extirpated (Appendix 4, Figure A4.26).

Most areas close to the Snake River, which are almost completely unoccupied at present, likely supported sage-grouse in the past (Bean 1941). Observations of habitat along the Snake River Valley during the mid-1800s indicated that many of the well-traveled areas close to the river were dominated by sagebrush and little grass (Vale 1975). In addition, these areas are the lowest elevation and driest, and the most likely to be developed and/or converted (Bunting et al. 2002). It is possible that areas along the Snake River may have supported wintering sage-grouse when higher elevation habitats were covered with deep snow. Nevertheless, most museum specimens were collected at least 25 km from the Snake River, except for 3 specimens west of American Falls and another near Wilder, all collected in 1933 or earlier (Schroeder et al. 2004). This suggests that sage-grouse were extirpated close to portions of the Snake River relatively early, perhaps prior to 1900 in most cases. In addition, populations of sage-grouse are apparently continuing to recede from the Snake River and its tributaries, indicating this may be a long-term trend.

Columbia Basin

Available evidence indicates that potential sage-grouse habitat once covered much of north-central Oregon, central and eastern Washington, and extended northward along the Okanogan River into southern British Columbia. Lewis and Clark noted sage-grouse on the plains between the confluence of the Columbia and Snake rivers (near present city of Kennewick, Washington) and the confluence of the Columbia and Deschutes rivers (Zwickel and Schroeder et al. 2003). David Douglas (Royal Historical Society 1914) observed sage-grouse near Wallula in 1826, and extremely large numbers of birds along the Columbia River near Priest Rapids, Washington (now under Priest Rapids Lake). The first record for sage-grouse in British Columbia was in 1864 near Osoyoos Lake; most subsequent records were in the same area (Campbell et al. 1989).

Jewett et al. (1953) and Aldrich and Duvall (1955) showed the past distribution of sage-grouse straddling the border of Idaho and Washington on the eastern edge of the Palouse Prairie. However, because the potential pre-settlement habitat of the Palouse Prairie was likely dominated by perennial grasses with little sagebrush (Daubenmire 1970, Kuchler 1985) and because there are no observations or museum specimens in these portions of Idaho and Washington (Yocom 1956, Schroeder et al. 2000), the revised map of potential pre-settlement habitat does not include this area. Potential pre-settlement habitat in Washington was probably continuous with north-central Oregon (Schroeder et al. 2004). The continuity of these populations is one reason why the U.S. Fish and Wildlife Service considered the Oregon-Washington area to be a ‘distinct population segment’ (Warren 2001). Despite this apparent continuity, most of the early observations in the region are from the area near the confluence of the Snake and Columbia rivers (Peale 1848, Cassin 1978) with few observations in north-central Oregon (Schroeder et al. 2004).

Reductions in the distribution of sage-grouse in the region were noted as early as 1920 (Jewett et al. 1953). These reductions have been monitored to the present, with the declines continuing (Schroeder et al. 2000). Sage-grouse are almost completely extirpated from Washington, British Columbia, and northern Oregon. The last observation (prior to translocation efforts) in British Columbia was in 1918 near Oliver (Campbell et al. 1989). Although 57 birds were translocated from the Malheur County, Oregon area to the Richter Lake area of British Columbia in 1958, the translocation was unsuccessful, with the last observation of birds in 1966 (Campbell et al. 1989). Between 1941 and 1951 there were efforts to re-establish or augment populations in Sherman and Umatilla counties in northern Oregon with birds captured in other areas of Oregon, however these translocations were also unsuccessful (Reese and Connelly 1997).

Northern Great Basin

The potential pre-settlement distribution of habitat in this region is believed to encompass substantial portions of south-central Oregon, northeastern California, and northwestern Nevada (Schroeder et al. 2004). The distribution of museum specimens and published observations appears to approximate the distribution of potential pre-settlement habitat for this area. Although early populations in this region appeared to be connected with the Columbia Basin through relatively narrow corridors near Bend, Prineville, and Dayville, Oregon, there is no direct evidence. In contrast, the connections with the Snake River Plain and southern Great Basin regions were, and still are, substantial. Much of the potential pre-settlement habitat within this region is relatively continuous and inter-connected. The distribution west of Goose Lake in northeastern California and the Klamath Basin area of Oregon is an exception due to separation from other potential sage-grouse habitats by substantial areas of forest.

Southern Great Basin

The pre-settlement distribution of potential habitat in this region includes substantial portions of east-central California, central Nevada, and western and central Utah (Schroeder et al. 2004). The distribution appears to be separated geographically into three general areas. The first is along the eastern edge of central California and the western edge of central Nevada. The second is a relatively continuous area in central and eastern Nevada and the western edge of Utah. The third is a fragmented area associated with the north-south oriented mountain ranges in central Utah. Most of the changes in distribution appear to be along the southern perimeter and/or in the relatively arid habitats. Some of these changes have been attributed to long-term changes in habitat (Christensen and Johnson 1964, Brown and Davis 1995, Miller and Eddleman 2001) and these range retractions are continuing at the present time (Connelly and Braun 1997).

Previous distribution maps for Utah often show continuous occupation by sage-grouse, essentially border to border (Griner 1939, Lords 1951, Aldrich and Duvall 1955). However, an assessment of historic habitat (Kuchler 1985) suggests many areas were likely unoccupied by sage-grouse (Beck et al. 2003. For example, it is unlikely the barren lands around, and to the west of, the Great Salt Lake were ever occupied by sage-grouse. In addition, forested and alpine habitats in mountainous areas were also likely unoccupied including portions of the San Pitch Mountains, Markagunt Plateau, Paunsaugunt Plateau, Tushar Mountains, Aquarius Plateau, and Escalante Mountains. Nevertheless, current populations in Utah appear to be more isolated than they likely were in pre-settlement times (Schroeder et al. 2004). Declines in abundance in the developed areas around Salt Lake City may have occurred relatively early (Hayward et al. 1976). Between 1987 and 1990, 43 birds captured in Uintah and Carbon counties were released in Sevier County, Utah. The translocation appears to have been successful (Reese and Connelly

1997). There is also an ongoing effort to translocate birds from Parker Mountain in Wayne County to the Strawberry Valley in Wasatch County, Utah (38 in 2003).

Colorado Plateau

The potential pre-settlement distribution of habitat is somewhat continuous with the adjacent southern Great Basin region, but it is naturally fragmented by topography and alternate habitat types. Museum specimens and published observations are rare for the region, with no museum specimens for Arizona (Schroeder et al. 2004). The southernmost observation of a sage-grouse in North America is from an area west of Mt. Trumbull, Arizona in 1937, approximately 65 km south of the Utah-Arizona border (Huey 1939). However, Phillips et al. (1964) considered the range of sage-grouse in Arizona to be hypothetical.

Current populations are restricted to relatively small and isolated portions of southwestern Utah (Behle 1943). The southernmost active lek is northeast of Kanab; about 43 km north of the Arizona-Utah border (Schroeder et al. 2004). The history of sage-grouse on the Colorado Plateau is poorly documented, due in part to the small number of travelers and early changes in the region associated with settlement (Brown and Lowe 1980, Miller and Eddleman 2001). Nevertheless, recent history has shown populations receding northward. For example, two recently extirpated leks in southern Utah were only 30 km north of the Arizona-Utah border (Schroeder et al. 2004). In addition, sage-grouse have been extirpated from formerly occupied areas in the southwestern corner of Utah.

Other regions

Sage-grouse were observed in southwestern Kansas during the 1870s (Goss 1883, 1886), west of Wilburton, Kansas in the early 1930s, near Waynoka, Oklahoma in 1902 (Tate 1923), and north of Beaver Creek in Cimarron County, Oklahoma in 1910-1920 (Tate 1923). The past presence of sage-grouse in southwestern Kansas and western Oklahoma has been considered hypothetical (Thompson and Ely 1989) and observations have been attributed to erratic wanderings, mistaken identities (Applegate 2001), or Gunnison sage-grouse (Young et al. 2000). The number of distinct observations (at least 5) in a relatively small area supports the possibility that sage-grouse may have been resident (Schroeder et al. 2004). However, their relationship with specific habitat types in the region is not clear. Sand sagebrush (*A. filifolia*) is the dominant shrub species in the region, but has an extensive distribution that includes many areas where sage-grouse have not been observed; in particular, the adjacent areas of eastern Colorado and the Panhandle Region of Texas. Because of these contradictions, these areas are not included in pre-settlement distribution maps (Schroeder et al. 2004).

Summary

Although the distribution maps represent the pre-settlement distribution of potential habitat and the current distribution (Chapter 1, Fig. 1.1), a distribution is dynamic due to factors such as habitat conversion or degradation, alteration of fire frequency, and climate change (Miller and Eddleman 2001). Some of these factors may explain changes in distribution (Brown and Davis 1995). Potential deficiencies with mapping are exacerbated by inaccuracies in habitat data and differences in the timing of landscape alteration. For example, changes associated with settlement began in the southwestern United States as early as the 1600s, while widespread settlement in northern areas did not commence until the mid-1800s. Another challenge with mapping is that habitat types can be difficult to define consistently over large regions. Although Patterson (1952) argued that the past distribution of sage-grouse was defined by the presence of sagebrush-dominated habitats, the quantity of sagebrush in a given habitat type is not always known and/or consistent. For example, some grassland habitats (fescue-wheatgrass, wheatgrass-needlegrass, wheatgrass-bluegrass, grama-needlegrass-wheatgrass; Kuchler 1985) may have a large component of sagebrush in some regions and virtually none in others. In addition, sagebrush-dominated habitat types may lack sagebrush in some areas, perhaps due to recent fires. Similar factors may influence the suitability of habitats with regard to conifer encroachment (Connelly et al. 2000, Miller and Eddleman 2001). Habitats characterized by an open tree canopy may support sage-grouse when the canopy is reduced, whereas habitats dominated by sagebrush may cease to support sage-grouse when the density and height of trees is increased; changes in the frequency of fire may have a fundamental influence in these processes (Miller and Eddleman 2001).

A lack of data may make it difficult to know whether there is an absence of birds or whether there is inadequate documentation of existing birds. It is possible that Lewis and Clark failed to observe sage-grouse along the Missouri River, even though they were present. If sage-grouse did occupy the Missouri watershed in eastern Montana and North Dakota, their densities may have been low in areas visited by early explorers. A similar issue applies to southern portions of the distribution. The 1912 extirpation of Gunnison sage-grouse in New Mexico (Ligon 1961) suggests that changes in distribution were occurring too early for adequate documentation. Future examinations of regional habitat and habitat change should provide more insight into long-term changes in the distribution of sage-grouse. The area currently occupied by greater and Gunnison sage-grouse is clearly smaller than was occupied in pre-settlement times (Schroeder et al. 2004). Declines in distribution have been noted throughout the twentieth century (Judd 1905, Hornaday 1916, Locke 1932, McClanahan 1940, Aldrich and Duvall 1955, Connelly and Braun 1997).

POPULATION TRENDS

Introduction

Almost 20 years ago, Crawford and Lutz (1985) reported significant declines of greater sage-grouse in Oregon and warned that further declines could lead to extirpation of the species from that state. However, little more work was done on population trends in other parts of the species' range until the mid-1990s. Since that time, numerous investigators have assessed sage-grouse population trends in various states and provinces. Braun (1995) reported that both greater and Gunnison sage-grouse populations in Colorado had decreased markedly. Schroeder et al. (2000) indicated that greater sage-grouse in Washington declined by at least 77% from 1960 to 1999. Beck et al. (2003) reported that lek size declined in greater and Gunnison sage-grouse populations in Utah, but also provided some evidence suggesting populations in parts of the state have been stable or increasing. Aldridge and Brigham (2003) and McAdam (2003) indicated that sage-grouse in Alberta and Saskatchewan had declined by 66% to 92%. Smith (2003) stated that sage-grouse surveys provided evidence of a steady decline in both North Dakota and South Dakota.

Unfortunately, recent population assessments have not been published for some of the key sage-grouse states (e.g., Idaho, Montana, Nevada, Oregon, Wyoming). However, Connelly and Braun (1997) synthesized available data for 9 western states and 1 province. They compared long-term averages to data obtained from 1985-94 and concluded that sage-grouse breeding populations have declined by 17% to 47%. They also examined sage-grouse production data for 6 states (CO, ID, MT, OR, UT, WY) and reported that production declined by an overall rate of 25%, comparing long-term averages to 1985-94 data. Of the six states with production data, only Utah reported a slight increase (1%) in production. Finally, Connelly and Braun (1997) classified sage-grouse populations in 5 states as "secure" and indicated that populations in 6 states and 2 provinces were "at risk".

All states and provinces monitor sage-grouse breeding populations by counting males attending leks during the spring breeding season. Standard techniques for censusing leks have been available for a number of years (Patterson 1952, Eng 1963, Jenni and Hartzler 1978, Emmons and Braun 1984) and were recently summarized (Connelly et al. 2003). Connelly et al. (2003) differentiated between lek survey, lek count and lek route (see Population Database section in this chapter) and recommended the use of lek routes whenever possible. Despite available information, censusing methods may differ markedly among some agencies and even among years within agencies (Connelly et al. 2003). Rather than using multiple counts over several weeks, some agencies have used single counts, or multiple counts in a 1-week period. In other cases, lek counts appeared to have very low priority and were not done at all in some years. These inconsistencies confound attempts to make comparisons of population trends among states and provinces. Nevertheless, long-term lek counts comprise the largest range-wide database available on sage-grouse populations and generally appear to provide reliable data on population trends at a relatively broad scale.

In addition to lek counts, states that have hunting seasons monitor production with wing surveys that allow classification of juveniles and adults from hunter-harvested birds. However, in many cases sample sizes of wings are quite low and the migratory nature of some populations (Connelly et al. 2000) does not allow inferences to be made with regard to a single breeding population. Only Oregon and California conduct routine brood routes in addition to lek counts and wing surveys.

Information on lek distribution, activity, and attendance provides the only long-term data on sage-grouse breeding population trends. Thus, those are the only data we use in this chapter to assess sage-grouse population trends. The purpose of this section is to assess changes in long-term sage-grouse populations using data on numbers and distribution of leks as well as information obtained from lek counts. We make no attempt to estimate populations using lek counts because of the erratic nature of some data collection efforts and problems associated with detection rates.

Methods

In a lek mating system males congregate on relatively small sites to display and breed. Because females exhibit relative unanimity in mate choice (Gibson et al. 1991), few males do most of the mating. Older males are more likely than yearlings to attend leks and copulate; yearling attendance at leks appears to increase throughout the breeding season (Emmons and Braun 1984, Hartzler and Jenni 1988). Many males may concentrate around successful males in the hopes of a 'spillover effect' (Gibson et al. 1991). Because these lek sites tend to be traditional, they offer the best opportunity for monitoring populations (Jenni and Hartzler 1978; Beck and Braun 1980; Connelly et al. 2000, 2003).

For the purposes of this chapter, we generally define a lek as a traditional display site with 2 or more males that have been recorded during the assessment period or within 5-years of that period. However, despite the traditional nature of lek sites, they are difficult to precisely define. For example, leks are spatially defined as a single point (center of the lek), even though the males on a lek occupy an area rather than a point. When more males attend the lek, the area occupied is larger due to the territoriality of the males (Gibson and Bradbury 1987). Males (and hence leks) also may sometimes shift locations between years. The gradual shifting of a lek's location during a period of many years can influence a lek count. A lek count may be further complicated by the formation of satellite leks that may develop near a large lek during years with relatively high populations. When leks are well attended by males and expansive in area, males may concentrate in multiple locations that are relatively close. Although this phenomenon has been poorly studied, the reason for this behavior may be related to localized variation in topography and habitat. At times observers have considered these multiple locations as separate leks rather than separate activity centers on a single lek.

There may be substantial variation among states and populations with regard to the definition of a lek. A biologist in one area might define an expansive group of 100 males as a single lek, while a biologist from a different area might interpret the same group as 2 leks based on their separate concentrations on 2 adjacent activity centers. Therefore, we carefully examined each state's database and removed questionable data, leks for which no count data could be provided, and replicate locations. Many states had records for leks that had no count data associated with them (and thus no way of confirming that they actually were leks). Because there is little published research documenting the fluidity of lek establishment, formation, and extinction, we established a set of rules for defining lek locations throughout the range of sage-grouse in North America. First, the center of the largest and most regularly attended location was considered to be the center of a lek complex. Males from other locations within 2.5 km of the complex's center were added to the annual totals for the lek complex. This distance was based on an interpretation of data throughout the species' range showing that these adjacent sites appeared to be inter-related. For example, in many state databases it was common for one lek to 'disappear' or 'fade away' concurrent to the appearance of an adjacent lek. This effect was also noted when a lek was incorrectly identified because males were displaying in an atypical location. Display behavior in atypical locations can be a result of males being flushed off the regular site by a predator or being observed at an atypical time of year or day. If all the individual lek locations were considered separately, without regard to their inter-dependence with adjacent sites, the overall count of males would not be affected. However, the continuity of data between years would be dramatically influenced. For example, many more leks would be considered to have become 'vacant' even though the males merely changed locations. The use of the 2.5 km distance also allowed for a consistent definition throughout the species' range. Even though data presented here will reflect these lek complexes, they will simply be referred to as leks hereafter.

All information relating to population trends refers to changes in breeding populations. Connelly et al. (1988) suggested that sage-grouse populations be defined on a temporal and geographic basis. Connelly et al. (2003) defined a breeding population as a group of sage-grouse associated with 1 or more occupied leks in the same geographic area separated from other leks by >20 km. We followed these definitions for this analysis, and further defined greater sage-grouse populations throughout their North American distribution based on the known locations of leks. Concentrated areas of leks were considered breeding populations if they were separated from the nearest adjacent concentration of leks by at least 30 km and/or separated by unsuitable habitat such as mountain ranges, desert, or large areas of cropland. Thus, we modified Connelly et al.'s (2003) definition somewhat by expanding spatial separation and including physical barriers.

Because of the massive size of five of the defined populations, the discontinuous nature of their distribution across the landscape, and regional differences in habitat within each population's perimeter, these some populations were further delineated into subpopulations. These subpopulations tended to be separated by both distance and topography, but the separation was quantifiably less than was required for delineation of populations. Populations and subpopulations also were grouped into floristic regions including the Great Plains (or silver sagebrush), Wyoming Basin, Snake River Plain, Columbia Basin, Northern Great Basin, Southern Great Basin, and Colorado Plateau (Figure 1.5, Miller and Eddleman 2001).

Monitoring effort

We assessed monitoring effort by individual states and provinces by examining the average number of leks and number of active leks censused over 5-year periods. This allowed us to assess overall monitoring effort (number of leks counted) and effective monitoring (number of active leks counted). We then calculated the change in number of leks censused to better understand whether monitoring effort was changing over time.

Population trends

Lek attendance data were obtained by counting the number of males attending leks during late March and April. Normally, multiple counts of individual leks were made and these counts occurred from 0.5 hour before sunrise to 1.5 hours after sunrise (Braun 1995, Schroeder et al. 2000, Beck et al. 2003). In some cases, counts were made over a relatively short time frame or not made in consecutive years (Aldridge and Brigham 2003). For instance, Alberta conducted lek counts every other year for many years while North Dakota conducted lek counts only during the third week of April (but has used this approach for well over 30 years).

The use of lek counts to evaluate populations has been controversial with greater sage-grouse (Walsh 2002, Walsh et al. 2004) and with other species of prairie grouse (Applegate

2000). A central issue involves the use of lek counts to estimate populations, even without supporting data on the population's actual parameters (Anderson 2001, Walsh et al. 2004). Some of this concern is based on attendance rates for males on leks. Walsh et al. (2004) also expressed concerns that estimation of long-term trends might have similar problems but did not examine the relationship between lek counts and trends. To better understand the reliability of lek data for assessing population trends in this conservation assessment, we tested the lek count procedure using simulated populations.

The average annual rate of population change for 10,000 simulated populations deviated from the observed rate (using simulated surveys of each population) by an average of 0.0384 (SD = 0.0308); this number was not correlated with the actual rate of population change (r^2 = 0.0001). An evaluation of accuracy suggested that accuracy increased with the observed rate of population change. Accuracy was generally greater than 80% for populations with an observed annual rate of change of at least 0.03 and greater than 95% with rates of a least 0.07. This is the first indication that the significance of trends using lek counts can be supported by data other than with the finality of localized extirpations (Schroeder et al. 2004). A complete description of this analysis is provided in Appendix 3.

Changes in sage-grouse breeding populations can be related to changes in the number of leks, changes in lek size, or both. Moreover, ability to detect changes will be dependent upon monitoring effort. In all states and provinces, different numbers of leks were often sampled annually so total counts of males provide almost meaningless information. Therefore, we used three related but different methods to assess population trend: changes in males per lek, changes in lek class size, and changes in a population rate index. We provide population trend data in the form of descriptive statistics for all states and provinces. In most cases, we used 1965 as a baseline for assessing population change but some states and provinces began routine data collection much later. In those cases we used the earliest date available. In assessing population trends, we assumed that detection rates did not vary among years.

We calculated mean and median lek size for all leks to assess population trends. However, because inadequate sampling may result in a lek being incorrectly recorded as inactive (0 males) if an observer arrived at a lek just after a disturbance (e.g., predator, livestock operations, etc.), or if a lek moved and was not detected, we also calculated mean and median lek size for active leks. We averaged these values over 5-year intervals to provide a broader perspective of change and presented these data in tables. Missing data were estimated by averaging values for the year before and year after the missing value. If the value came at the beginning or end of the assessment period, we averaged the two years following or preceding the value, respectively. We then used simple linear regression to assess changes in lek size over all years of the assessment period for each state and province. We considered changes significant if $P \leq 0.05$.

Because lek counts may be influenced by local disturbance levels, observer interest, observer training, and weather, we also assessed population change by examining the change in lek size classes over time. This technique is less sensitive to accurate counts of grouse attending leks and simply categorizes leks into size classes. We classified leks as small (1-19 males), medium (20-49 males) and large (>49 males) and qualitatively examined changes in these size classes over years.

Finally, we calculated annual rates of change for each state, province, floristic region, and population with sufficient data. Only leks counted in consecutive years were used in the analysis. For example if 10 leks were counted in both 1998 and 1999, the instantaneous rate of change between 1998 and 1999 was estimated as the natural log of total number of males counted on the 10 leks in 1999 divided by the total number of males counted on the same 10 leks in 1998 ($\ln [Y_N / Y_{N-1}]$, where Y = count of males and N = year). Any leks counted in only 1 of the 2 years were not used in the analysis. This technique helped to reduce the biases associated with sampling selectivity. Long-term rates of change for each area were estimated by averaging the instantaneous rates of change for each 2-year interval. We then examined the long-term patterns of change using a base population of 100% in the most recent survey year (2003) and presented this information in graphic form for each area we analyzed. For example, the population in 2002 was estimated relative to the 2003 population based on the estimated instantaneous rate of population change between 2002 and 2003. This approach provided a population index value that allowed an assessment of change over time and allowed an assessment of how previous population changes compared with the current situation. We then used these data to evaluate trends, variation, and density dependence in rates of population change over the assessment period and during early (mid-1960s to mid-1980s; a time of active sagebrush eradication programs) and late (mid-1980s to 2003; generally a time of reduced sagebrush control programs) portions of the assessment period. We also treated the overall population with a density independent model to provide an unbiased assessment of trend over the entire assessment period and, for the range-wide population, allowed an estimate of probability of persistence. The density independent approach assumes normally distributed variation in the annual instantaneous growth rate (Dennis et al. 1991). Additionally, we applied a density dependent model (Dennis and Taper 1994) to each time series (overall, early, and late) and assessed the likelihood of density dependence and approximate equilibrium population size as a proportion of the 2003 population. We used linear regression to estimate the parameters of the density dependent model.

We used three separate but related approaches to comprehensively assess changes in sage-grouse populations at regional (sub-population, population, state) and range-wide scales. Because of the problems with lek count data previously discussed, no method currently available is free from biases or thought to give a highly accurate assessment of trends. Thus, it is necessary to examine results from all three analyses for each state or province and combine that with information provided in the distribution section of this chapter and habitat information (Chapter 7) to arrive at what should be a reasonably clear picture of the situation.

Range-wide population assessment

Lek distribution and numbers**.** We obtained UTM locations for all leks recorded within all states and provinces since the agency began routine lek censuses. In some cases, UTM locations indicated 2 or more leks were within 2.5 km of each other. In those cases, we grouped all into a single lek using the same approach taken for state and provincial populations. We mapped these locations and compared the distribution to that of all leks reported as active since

2000. We then used these maps to define discrete populations of greater sage-grouse throughout western North America without regard to political boundaries. These populations were identified based on groups of leks and their relative spatial isolation from other nearby groups due to distance and topography.

Population status and change. We analyzed discrete populations throughout the species' range, without regard to political boundaries in the same manner we used for state and provincial populations. Thus we provide information on lek size and changes in a population index developed from rate of change data for a population representing all sage-grouse in North America. For this range-wide population we used population index data to evaluate trends, variation, and density dependence in rates of population change over the assessment period and during early (mid-1960s to mid-1980s) and late (mid-1980s to 2003) portions of the assessment period. We again treated the overall population with a density independent model that provided an unbiased assessment of trend over the entire assessment period. The density independent approach assumes normally distributed variation in the annual instantaneous growth rate (Dennis et al. 1991). Additionally, we applied a density dependent model (Dennis and Taper 1994) to each time series (overall, early and late) and assessed the likelihood of density dependence and approximate equilibrium population size as a proportion of the 2003 population. We used linear regression to estimate the parameters of the density dependent model. Values for all populations, subpopulations, and regions are summarized in appendices 4, 5, and 6, respectively.

Results

States/Provinces

Alberta

Monitoring effort. Alberta identified 30 sage-grouse leks and has routinely monitored these leks since 1975. Thus, we used 1975-2003 as our assessment period. Monitoring efforts increased from 1975 to 2003 and $\geq$ 28 leks have been monitored annually since 1997. In 22 of 29 (76%) years, $\geq$ 10 leks were censused. From 1975 to 2003, in 5-year periods, an average of 6 to 29 leks were monitored (Table 6.3). Over these same 5-year periods, the number of active leks monitored declined from 13-16 leks (1975-89) to 9 active leks (2000-2003) (Table 6.3). Alberta did not employ a standard monitoring scheme of multiple counts spread over a 4-6 week period. Instead, lek counts were made once each spring in about the third week of April, generally thought to be the peak of male attendance. However, not all leks were counted each year and in some years, lek counts were not made at all. From about the late 1980s onward, efforts appeared to be made to count all leks. In 1996 and 1997 intensive efforts resulted in leks being censused weekly from about the first week of April until the third week in May (Aldridge and Brigham 2003).

Population Changes. The proportion of active leks decreased over the assessment period, averaging 100% from the mid-70s to the mid-80s but decreasing to 29% by 2000-2003 (Table 6.3). Similarly, average and median males per lek also decreased over the assessment period by 78% and 100%, respectively (Table 6.3). Average and median males per active lek showed a similar trend, decreasing by about 28% and 31%, respectively (Table 6.3). Monitoring data (males/lek) indicated a significant decline (r^2 =0.48, P = 0.00) in lek size from 1975 to 2003 (Fig. 6.1). This coupled with the decrease in active leks suggests that the number of breeding birds associated with some leks have declined while other subpopulations have disappeared. Monitoring data suggest that current populations may be at their lowest point of the assessment period, but they appear to have been relatively stable at this level since 1997.

The majority of leks in Alberta are relatively small but the proportion of small leks has increased somewhat since the mid-80s (Fig. 6.2). The proportion of leks with 20-49 males did not change appreciably and was relatively low (Fig. 6.2), except for 1985-89 when overall populations in Alberta apparently increased. Large leks ($\geq$50 males) were seldom detected throughout the assessment period. The relatively high proportion of small leks was likely the result of the Alberta sage-grouse population being on the northern edge of the species' range.

Because of relatively small samples of leks and inconsistent monitoring, we were unable to calculate long-term annual rates of change.

Summary. Connelly and Braun (1997) reported that the Alberta population had declined by 38% over the long-term. Aldridge and Brigham (2003) estimated that the total breeding population for Alberta was 1,839-2,724 birds in the late 1960s but by 1999 they reported that this population had declined to 420-622 total grouse. They further indicated that sage-grouse in Alberta declined by 66% to 92% over this period and that 62% of the leks in the Province have been abandoned. There was an average of 29.2 males/lek during 1968 but average lek size decreased to 5.8 males per lek by 1994 (Aldridge and Brigham 2003). Our analysis suggests that the estimate by Connelly and Braun was too conservative (i.e., not high enough) but supports Aldridge and Brigham's (2003) findings and indicates an overall decrease in the breeding population of about 80% from the mid-1970s to present.

Table 6.3. Sage-grouse monitoring and population trends in Alberta, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94[2]	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	29	24	6	8	10	14	0	0
Number of active leks[1]	9	9	11	13	16	14	0	0
Percent active leks	29	45	69	100	100	100		
Average males/lek	4	6	13	25	29	18		
Median males/lek	0	0	13	23	25	16		
Average males/active lek	13	13	15	25	29	18		
Median males/active lek	11	9	17	23	25	16		

[1] Averaged over each year for each period.
[2] Only 2 of years of data available for this 5-year period.

Fig. 6.1. Change in sage-grouse lek size in Alberta, 1975-03.

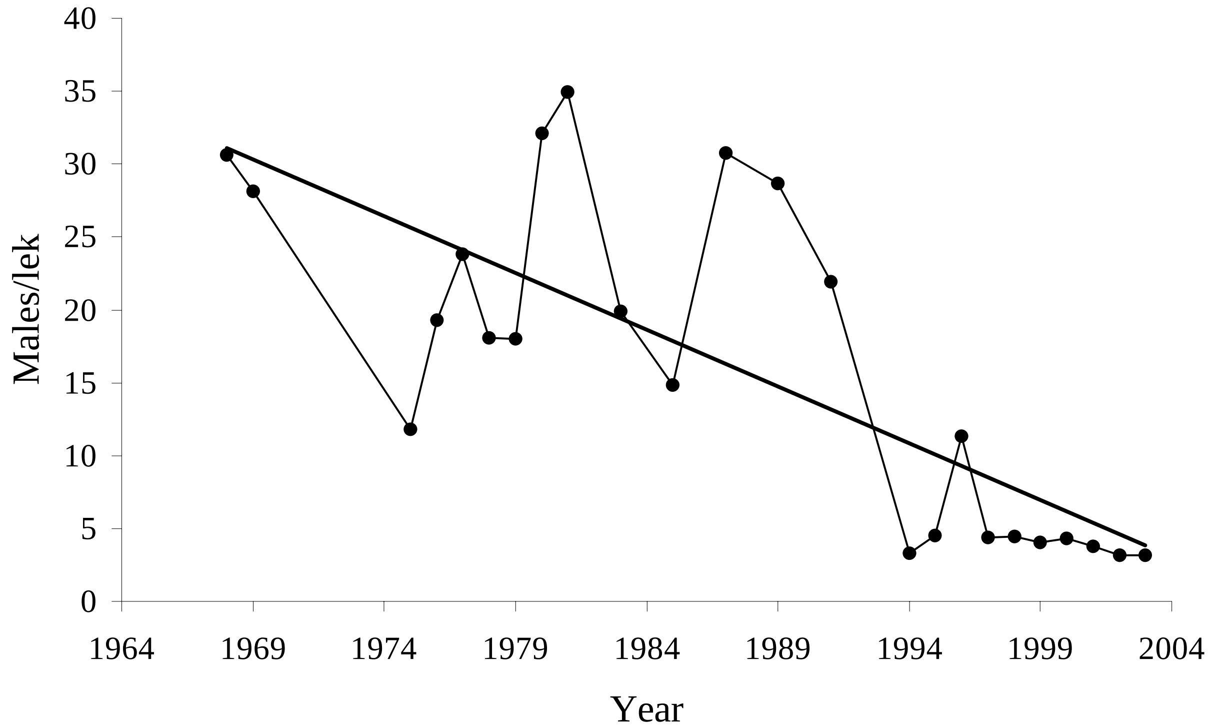
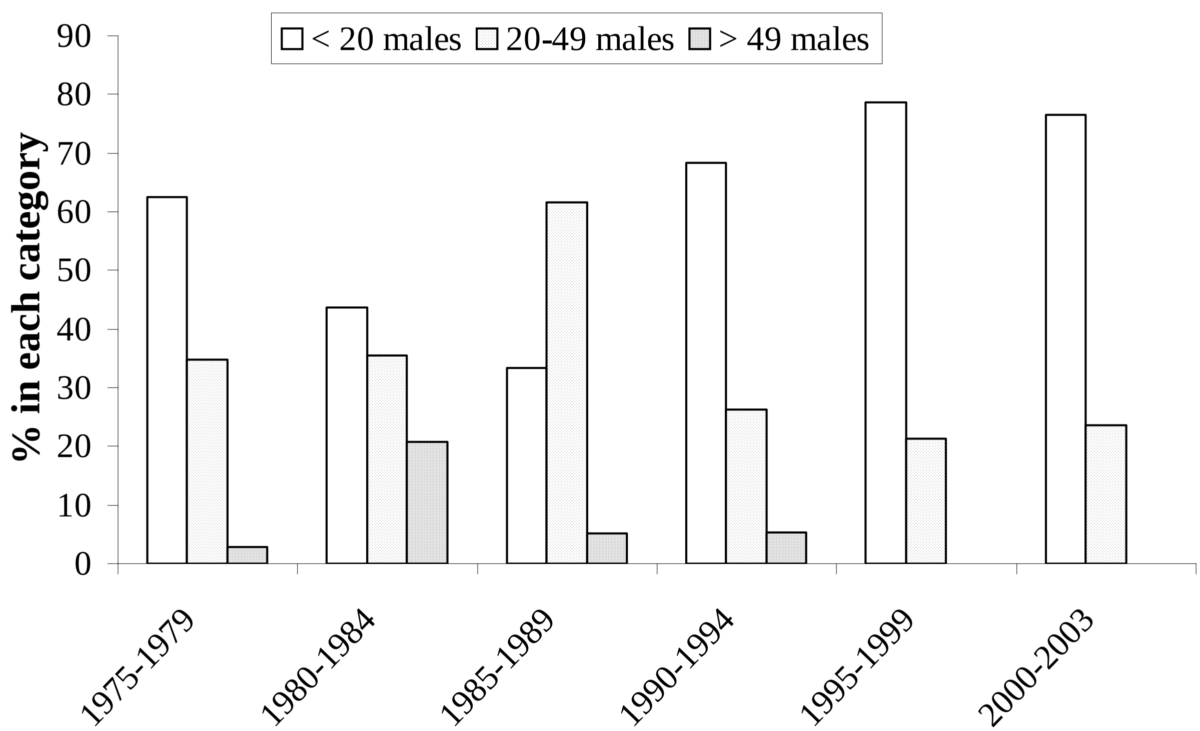


Fig. 6.2. Change in lek size class for Alberta, summarized over 5-year periods, 1975-2003.



California

Monitoring effort. California has identified 71 sage-grouse leks and has routinely monitored many of these leks since 1965. Thus, we used 1965-2003 as our assessment period. Monitoring efforts increased substantially (400%) from 1965 to 2003 and $\geq$ 20 leks have been monitored annually since the early 1980s (Table 6.4). The number of active leks (i.e., effective monitoring) censused increased by 388% over the same period suggesting that few leks became inactive over the assessment period.

Population change. The proportion of active leks remained relatively stable and high throughout the assessment period, with 5-year averages varying from 77% to 90% between 1965 and 2003 (Table 6.4). The average and median males per lek fluctuated over the assessment period but showed a gradual increase over time (Table 6.4). Numbers apparently peaked from 1985 to 1994 and decreased somewhat since then (Table 6.4). The average number of males per active lek followed a similar pattern. Monitoring data (males/lek) indicated an increasing lek size (r^2 =0.14, P = 0.02) from 1965 to 2003 (Fig. 6.3). Although lek size class varied over the assessment period no obvious patterns could be documented, further suggesting a relatively stable population (Fig. 6.4).

Annual rates of change standardized on 2003 populations indicated a relatively stable to increasing population trend (Fig. 6.5). Sage grouse populations increased at an overall rate of 0.7% per year from 1965 to 2003. Our analysis suggested a high likelihood of density dependence for the overall assessment period (likelihood = 1.00) and for both the early (likelihood = 0.98) and late (likelihood = 0.88) periods. From 1965-1985, the population increased at an average rate of 2.82% and fluctuated around a level that was approximately 1.1 times higher than the 2003 population. From 1986 to 2003, the population declined at an average rate of 1.9% and fluctuated around a level that was approximately 1.2% above the 2003 population. Populations in the late 1960s and early 1970s fluctuated from 65% to 114% of the current populations (Fig. 6.4). Although breeding populations have varied considerably over time, the available data suggest there is little evidence for long-term population change.

Summary. Little published information is available on California sage-grouse population trends. These populations are on the western fringe of the species' range and one of these populations is genetically unique and relatively isolated (Benedict et al. 2003). However, available data do not provide any evidence of a long-term population decline but instead suggest widely fluctuating but perhaps relatively stable to increasing populations.

Monitoring data indicated an increase in lek size while rate of change data suggested a stable population. This difference may be due to a sampling bias towards populations in the southern portion of the species range in California (i.e., Mono Lake area). In this area from 1965 to 1986, an average of 13 leks per year were counted with an average of 23 males per lek. In the northern portion of the species' range (Lassen and Modoc counties) during the same period only 3 leks per year were censused with an average of 22 males per lek. From 1987 to 2003, an average of 18 leks per year were counted in the southern area with an average of 24 males per lek. However, in the northern area there was a marked increase in the number of leks counted, including some relatively large leks that were not monitored during the early period. In this area from 1987 to 2003, an average of 17 leks per year were counted (an increase of 467% over the 1965-86 effort) with an average of 36 males per lek. Thus it appears that later censusing efforts in the northern area included some relatively large leks inflating overall estimates of lek size and resulting in an apparent significant positive change in lek size. Rate of change data would not have been affected by this change in sampling effort and thus probably more closely reflects the status of California populations.

Table 6.4. Sage-grouse monitoring and population trends in California, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	45	35	33	23	16	15	24	9
Number of active leks[1]	39	29	30	20	12	12	20	8
Percent active leks	88	84	90	88	77	82	83	87
Average males/lek	25	27	37	32	19	21	26	21
Median males/lek	14	17	20	20	14	14	18	14
Average males/active lek	28	32	41	36	25	26	31	24
Median males/active lek	18	22	22	23	19	12	23	16

[1] Averaged over each year for each period.

Fig. 6.3. Change in sage-grouse lek size in California, 1965 - 2003.

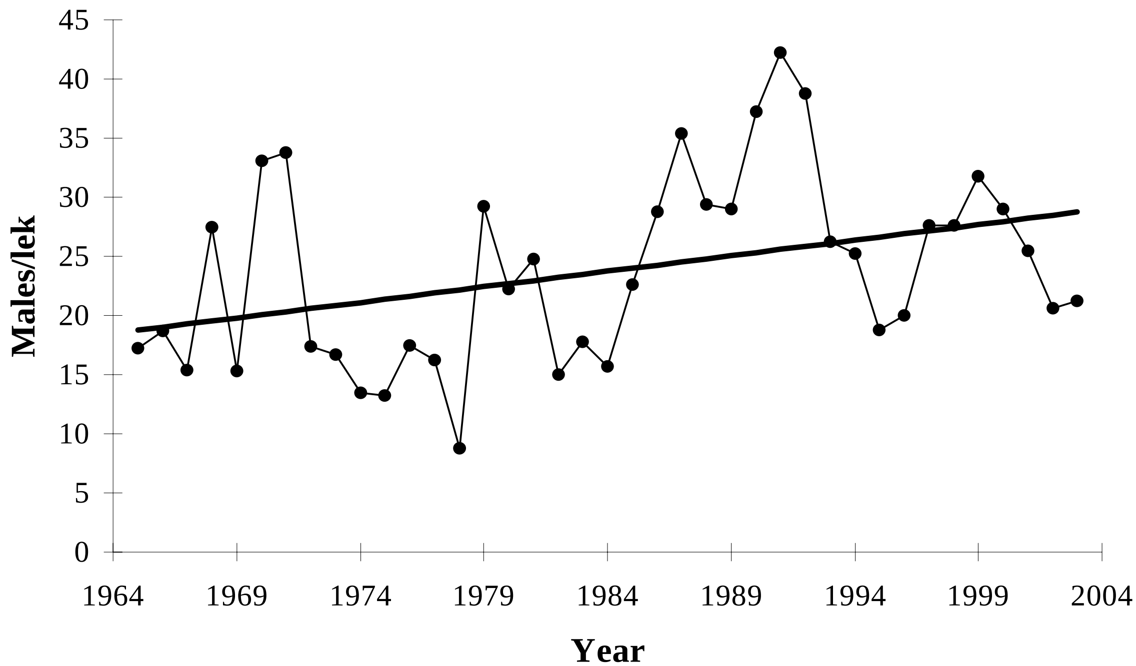


Fig. 6.4. Change in lek size class for California, summarized over 5-year periods, 1965 - 2003.

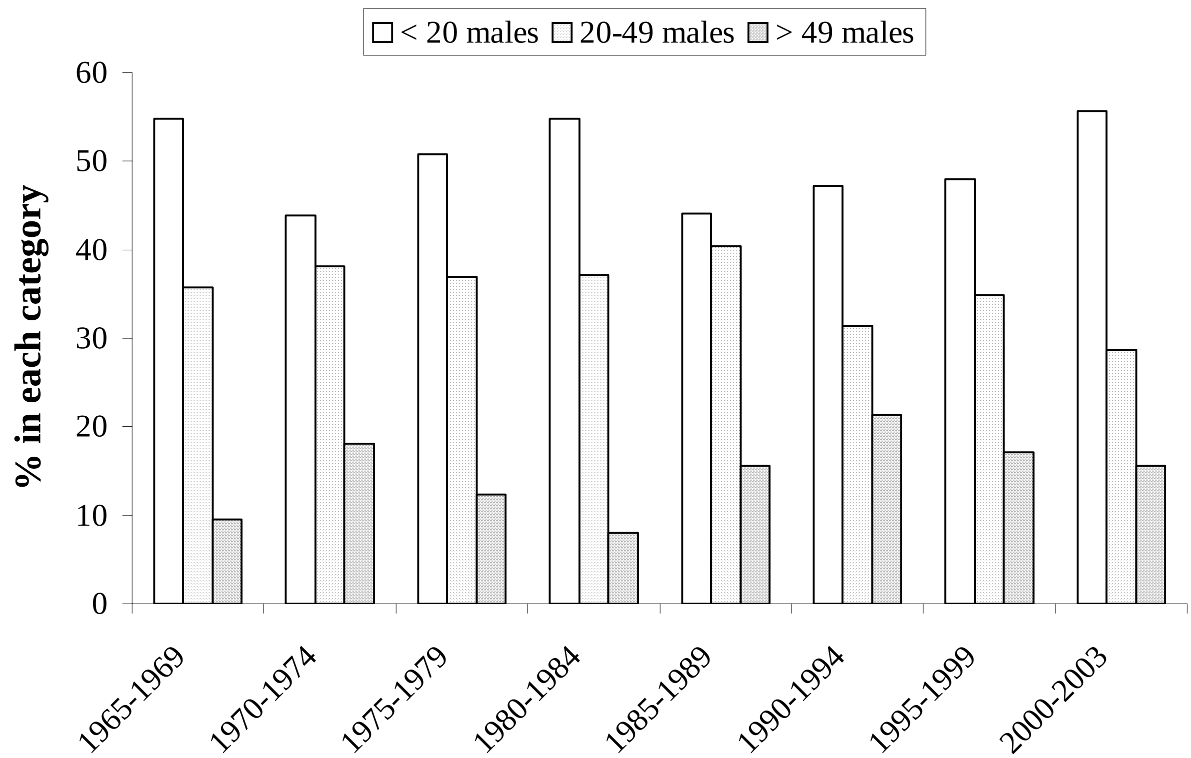
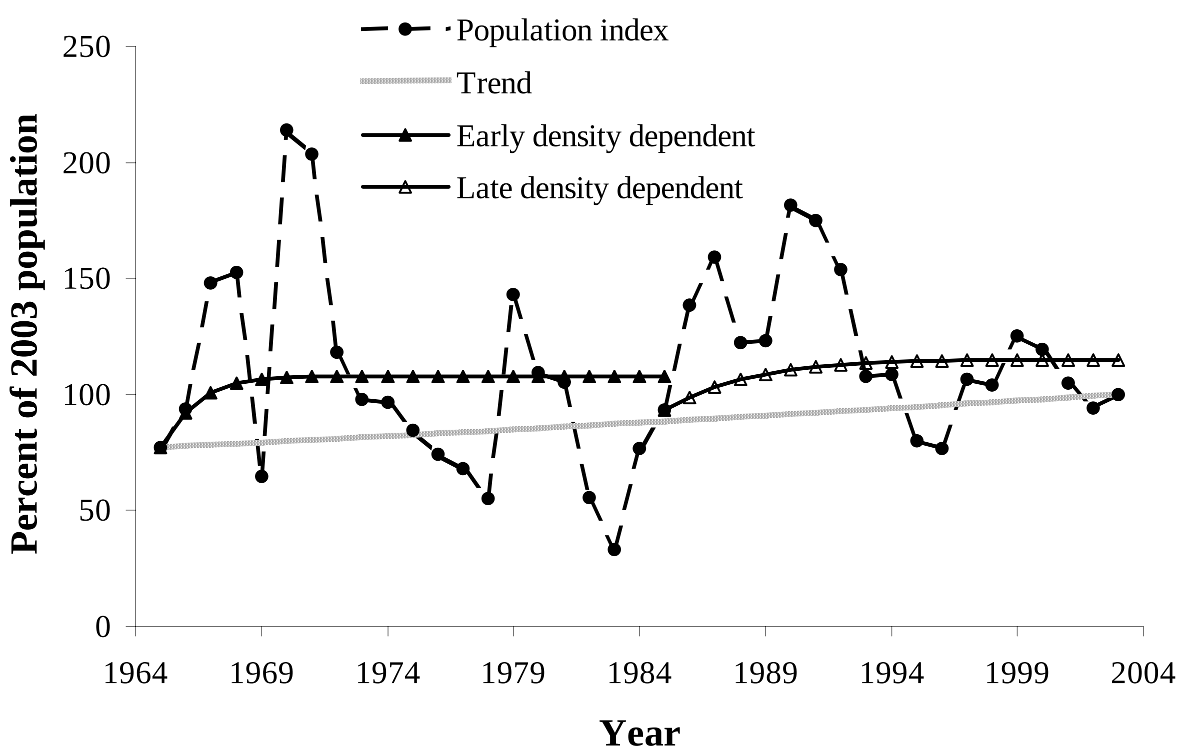


Fig. 6.5. Change in the population index for greater sage-grouse in California, 1965 - 2003.



Colorado

Monitoring effort. Colorado has had a long-term extensive monitoring program for sage-grouse and has identified 275 greater sage-grouse leks in the state. We used 1965-2003 as our assessment period. An average of 44 to 171 leks were censused in 5-year periods from 1965-69 through 2000-03. The average number of leks censused per 5-year period increased by 159% from 1965 to 2003. The number of active leks censused was similarly high, ranging from 35 to 114 and increasing by 124% over these same periods. For some parts of the state, observers in Colorado generally followed established procedures for lek census and attempted to count leks ≥3 times over an approximately 5-week period from late March to early May (Emmons and Braun 1984, Braun 1995). However, we detected a span of approximately 10 years where observers only recorded total numbers of sage-grouse on leks (males and females) in Moffat County and thus were unable to use these data in our analysis.

Population Changes. The proportion of active leks fluctuated considerably over the assessment period, ranging from 41% to 96% between 1965 and 2003 (Table 6.5). The highest proportion of active leks was recorded in 1975-79 while the lowest occurred in 1990-94. Population trends indicated by average and median males per lek decreased over the assessment period by 31% and 57%, respectively (Table 6.5). Average and median males per active lek also declined by 20% and 29%, respectively, over the assessment period (Table 6.5). Monitoring data (males/lek) indicated that average lek size significantly decreased ($r^2 = 0.40$, $P = 0.00$) from 1965 to 2003 (Fig. 6.6).

Information on lek size classes also suggests an overall decline in lek size. From 1965 through 1979, medium and large leks each comprised >30% of the leks sampled. Over the remainder of the assessment period, medium leks varied from 30% to 40% of the leks sampled during 5-year periods. However, large leks declined from 1985 through 2003, while the number of small leks increased from 27% in 1975-79 to 57% in 1995-99 (Fig. 6.7). From the late 1960s to 1979, 32% to 39% of the leks censused contained ≥50 males. From 1984 to 2003, ≤21% of the leks censused contained 50 or more males (Fig. 6.7).

Annual rates of change standardized on 2003 populations indicated a relatively stable to increasing population trend (Fig. 6.8). Sage-grouse populations increased at an overall rate of 1.0% per year from 1965 to 2003. Our analysis suggested a high likelihood of density dependence for the overall assessment period (likelihood = 0.96) and for the early period (likelihood = 0.97). However, the likelihood of density dependence was relatively low (likelihood = 0.54) for the late period. From 1965-85, the population increased at an average rate of 2.21% and fluctuated around a level that was approximately the same as the 2003 population. From 1986 to 2003, the population increased at an average rate of 4.3 % and fluctuated around a level that was again approximately the same as the 2003 population. Populations in the late 1960s and early 1970s were approximately 0.7-1.6 times the current populations (Fig. 6.8) with relatively large population fluctuations. These data do not support the trend information obtained from lek attendance (males/lek) and lek class size. Lows were reached in the mid-1980s and there has been a considerable increase in numbers since that time.

Summary. Braun (1995) reported a long-term decline in sage-grouse distribution and abundance and suggested it was largely related to habitat loss, alteration, and degradation. Similarly, Connelly and Braun (1997) indicated that sage-grouse breeding populations declined by 31% and production declined by 10% when they compared the long-term average of males/lek to the average obtained from 1985-94 data. The results of our analysis are somewhat ambiguous. Our data indicate that lek size has decreased but populations have increased. In part, this may be due to a change in how data were recorded in parts of Colorado from the mid-1980s to the mid-1990s. During this period total birds were recorded and gender not specified for many leks over a relatively large area. Thus we were not able to use these data sets when analyzing changes in lek size. Regardless, greater sage-grouse in Colorado have been generally increasing for about the last 17 years and available information does not suggest a dramatic overall decline in breeding populations over the last 39 years.

Table 6.5. Sage-grouse monitoring and population trends in Colorado, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	171	116	84	76	66	59	44	66
Number of active leks[1]	114	80	35	35	52	57	36	51
Percent active leks	67	69	41	46	79	96	81	78
Average males/lek	22	17	11	11	24	41	35	32
Median males/lek	9	10	0	0	14	34	27	21
Average males/active lek	33	25	26	25	30	43	44	41
Median males/active lek	22	16	21	19	22	34	36	31

[1] Averaged over each year for each period.

Fig. 6.6. Change in lek size for sage-grouse in Colorado, 1965-2003.

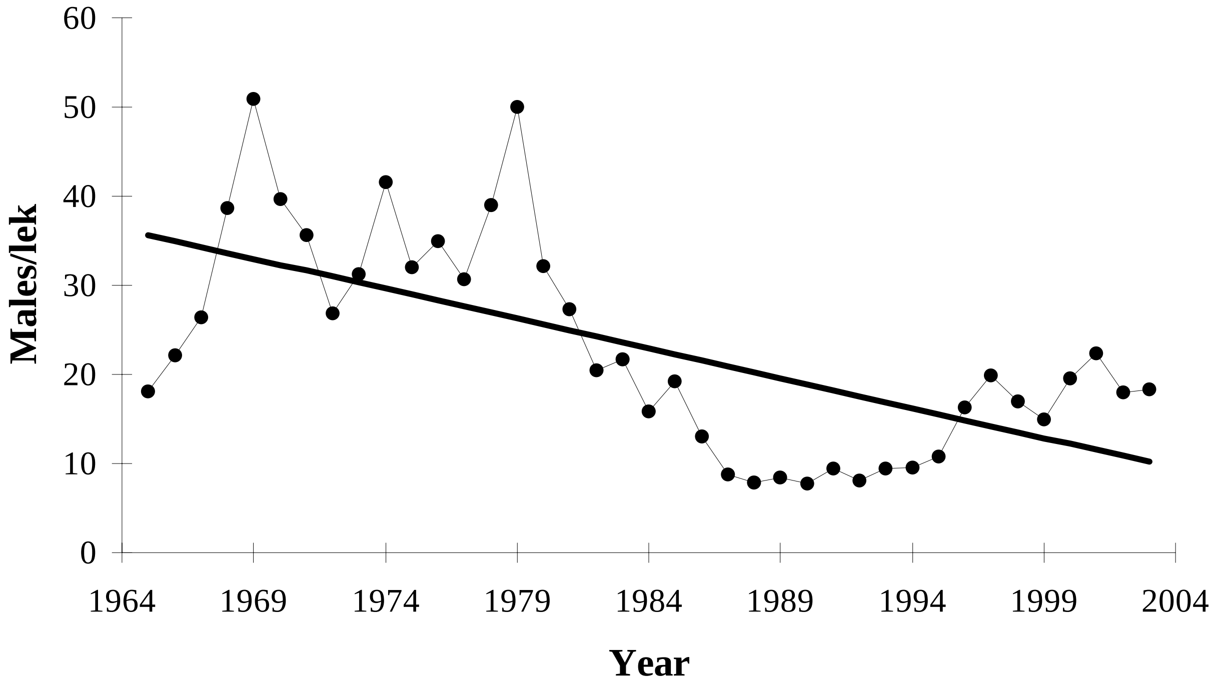


Fig. 6.7. Change in lek size class for Colorado, summarized over 5-year periods, 1965 - 2003.

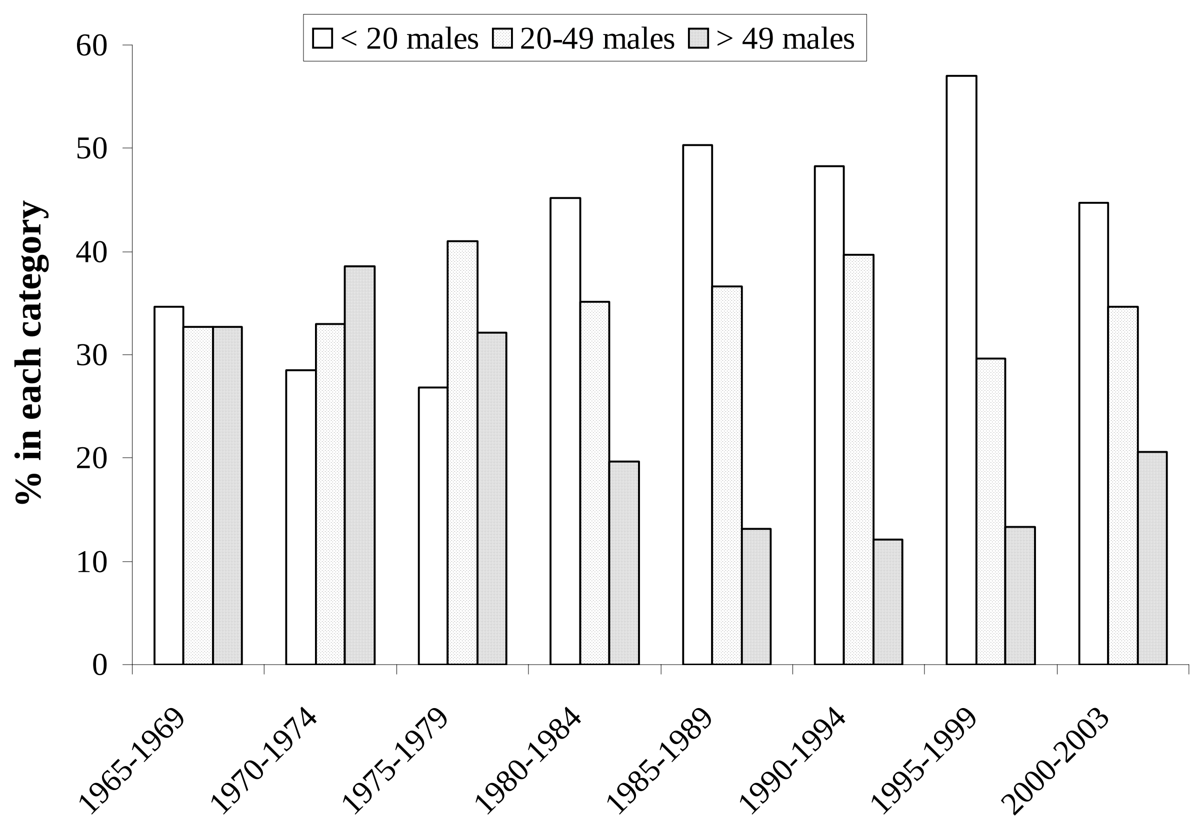
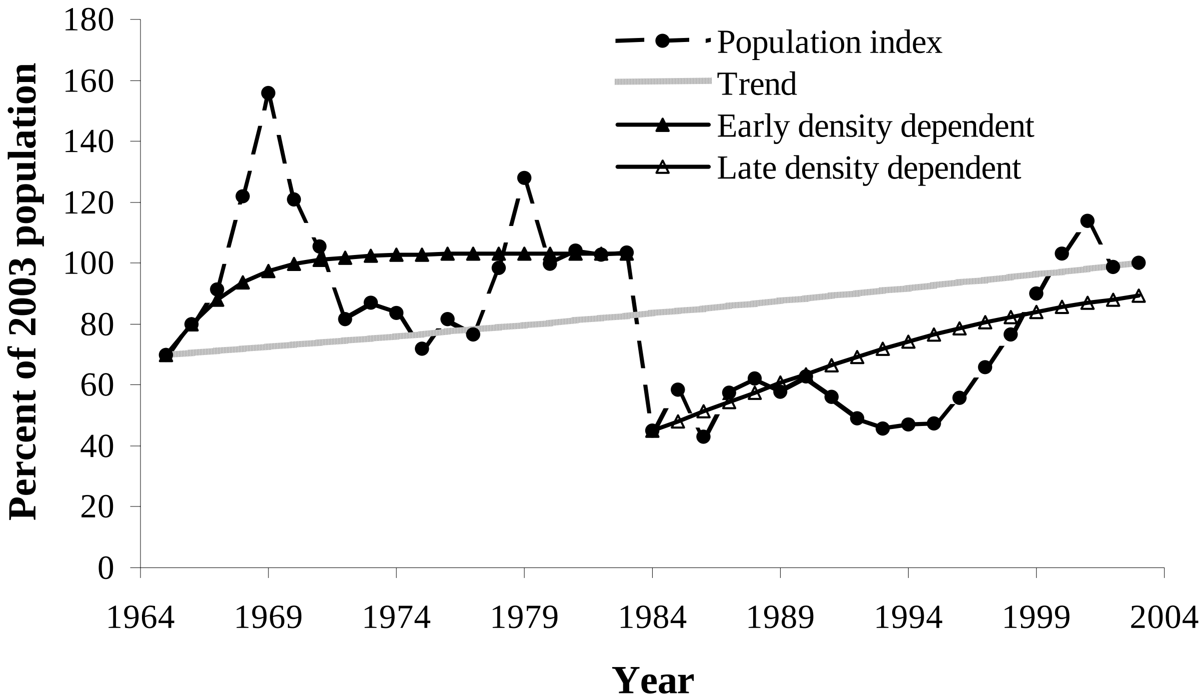


Fig. 6.8. Change in the population index for greater sage-grouse in Colorado, 1965 -2003.



Idaho

Monitoring effort. Idaho has had a long-term extensive monitoring program for sage-grouse and has identified 790 leks in the state. We used 1965-2003 as our assessment period. An average of 74 to 319 leks were censused in 5-year periods from 1965-69 through 2000-03. From 1965 to 2003, the average number of leks censused in 5-year periods increased by 331%. The number of active leks censused was similarly high, ranging from 69 to 245 and increasing by 255% over these same periods. Observers in Idaho generally followed established procedures for lek census and attempted to count leks $\geq$3 times over an approximately 5-week period from late March to early May.

Population Changes. The proportion of active leks decreased over the assessment period, averaging between 90% and 94% from 1965 to 1975 but decreasing to 73% to 77% from 1990 to 2003 (Table 6.6). Similarly, population trends indicated by average and median males per lek also decreased over the assessment period by 53% and 59%, respectively (Table 6.6). Average and median males per active lek also declined by 41% over the assessment period (Table 6.6). Monitoring data (males/lek) indicated that lek size significantly decreased (r^2 =0.50, P = 0.00) from 1965 to 2003 (Fig. 6.9).

There was an overall decline in lek size. Beginning in the early 1980s, the proportion of small leks increased. At the same time, the proportion of large leks decreased (Fig. 6.10). From the late 1960s to the late 1970s, approximately 25% to 35% of the leks censused contained $\geq$50 males. From 1995 to 2003, <15% of the leks censused contained 50 or more males. The proportion of medium leks has remained relatively stable, varying between 30% and 40% over the assessment period (Fig. 6.10).

Annual rates of change suggest a long-term decline for sage-grouse in Idaho (Fig. 6.11) and support the trend information obtained from lek attendance (males/lek) and lek class size. Sage-grouse populations declined at an overall rate of 1.47% per year from 1965 to 2003. Our analysis suggested a reasonably high likelihood of density dependence for the overall assessment period (likelihood = 0.84) and late period (likelihood = 0.88). However, we did not find substantial evidence for density dependence in the early period (likelihood = 0.47). From 1965-84, the population declined at an average rate of 3.04% and fluctuated around a level that was approximately 110% of the 2003 population. From 1985 to 2003, the population fluctuated around a level that was approximately 7% below the 2003 population and had an average change of 0.12% per year. Populations in the late 1960s and early 1970s were approximately 2 to 3 times higher than current populations (Fig. 6.11). The population reached a low in the mid-1990s and has increased since that time. However, previous population recoveries did not reach levels attained in the late 1960s and early 1970s.

Summary. There has been no published assessment of sage-grouse trends in Idaho but in a non-peer reviewed report, Autenrieth (1981:71) provided data that suggested Idaho had a relatively stable sage-grouse population from 1960 through 1979. In a more recent study, Connelly and Braun (1997) indicated that sage-grouse breeding populations had declined by 40% when they compared the long-term average of males/lek to the average obtained from 1985-94 data. Our analysis generally supports the findings of previous research efforts. However, the estimated decline provided by Connelly and Braun (1997) was lower than that indicated by our current data. This may be due to our use of a larger, more complete data set as well as the addition of 9 more years of data.

Table 6.6. Sage-grouse monitoring and population trends in Idaho, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	319	250	152	128	165	139	114	74
Number of active leks[1]	245	181	117	106	137	126	102	69
Percent active leks	77	73	77	83	83	91	90	94
Average males/lek	20	15	20	29	19	32	34	43
Median males/lek	13	8	12	18	12	21	25	32
Average males/active lek	27	20	26	36	23	35	38	46
Median males/active lek	20	14	18	25	16	23	30	34

[1] Averaged over each year for each period.

Fig. 6.9. Change in lek size for sage-grouse in Idaho, 1965-2003.

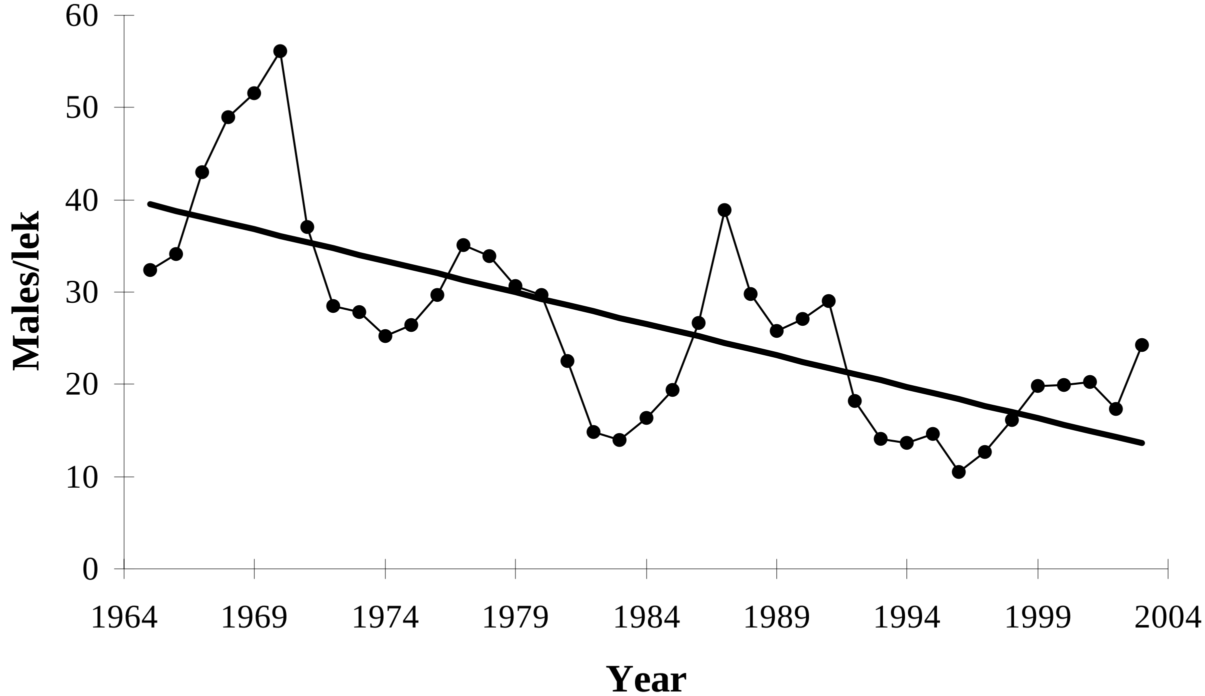


Fig. 6.10. Change in lek size class for Idaho, summarized over 5-year periods, 1965 - 2003.

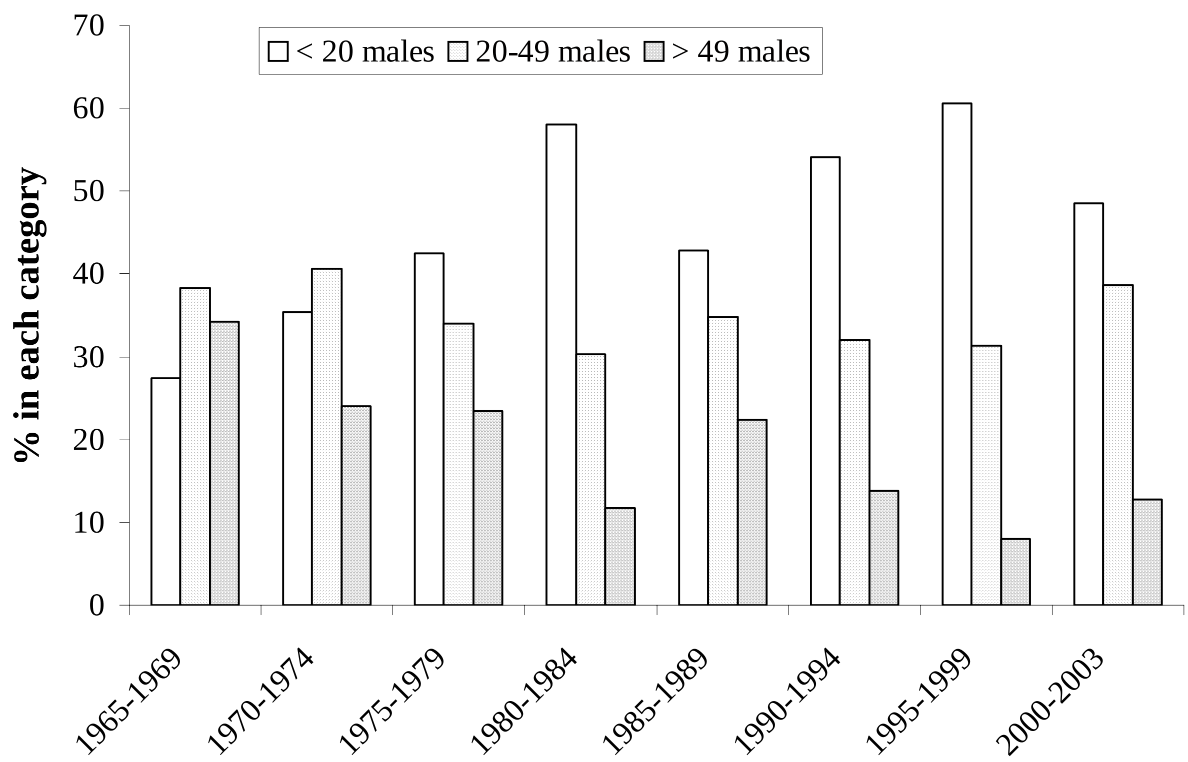
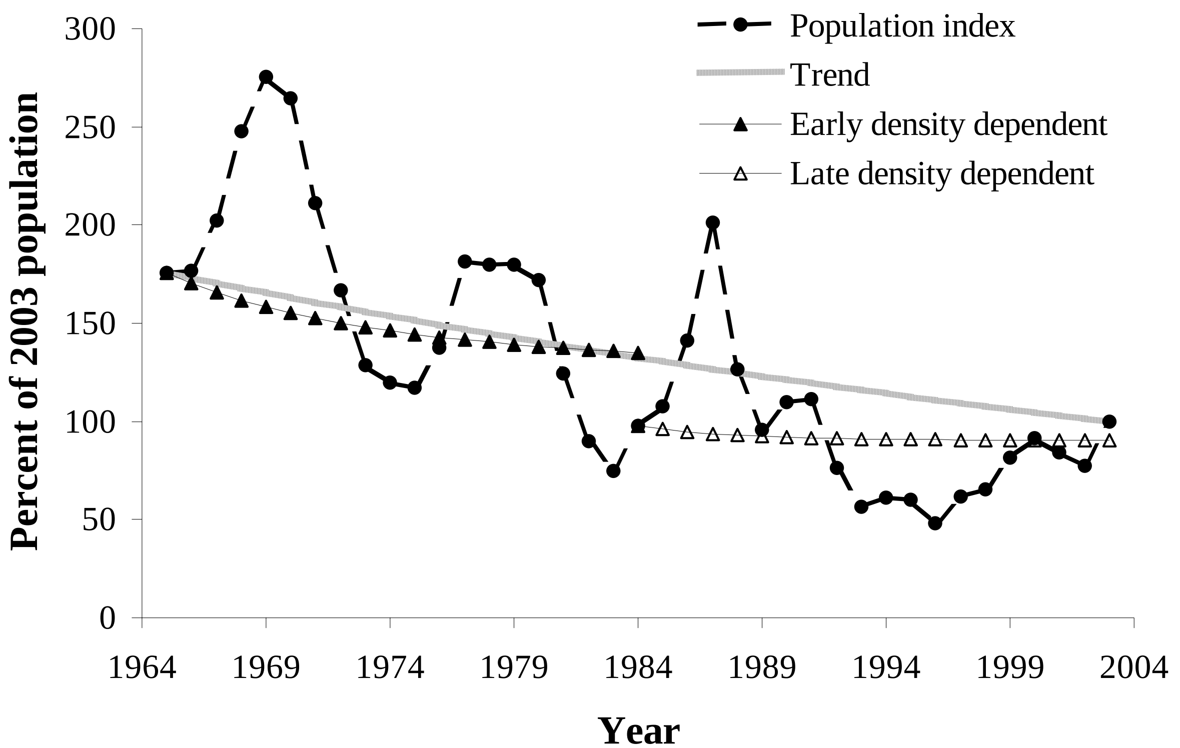


Fig. 6.11. Change in the population index for greater sage-grouse in Idaho, 1965-2003.



Montana

Monitoring effort. Montana has identified 1,094 leks within the state but their monitoring efforts have been inconsistent. During the late 1960s and early 1970s, relatively few leks were censused. However, the number of leks counted increased somewhat and then remained relatively stable from the early 1980s until the late 1990s (Table 6.7). By 2000, monitoring efforts increased substantially when the average number of leks counted during 2000-03 increased by >200% over the average number of leks counted in 1995-99 (Table 6.7).

From 1965 to 1979, observers apparently counted few inactive leks. As the number of leks censused increased, observers apparently increased efforts to monitor both active and inactive leks. Overall, the number of active leks monitored followed the same increasing pattern as total number of leks (Table 6.7).

Population Changes. The proportion of active leks decreased over the assessment period, averaging between 92% and 98% from 1965 to 1984 but decreasing to 73% to 78% from 1990-2003 (Table 6.7). Similarly, population trends indicated by average and median males per lek decreased over the assessment period by 41% and 45%, respectively (Table 6.7). Average and median males per active lek also declined by 21% and 10%, respectively, over the assessment period (Table 6.7). Monitoring data (males/lek) indicated that lek size decreased significantly (r^2 =0.52, P = 0.00) from 1965 to 2003 (Fig. 6.12).

Beginning in the mid-1980s, the proportion of small leks increased. At the same time, the proportion of large leks decreased (Fig. 6.13). From 1965 to 1989, 40% to 50% of the leks censused contained <20 males. From 1990 to 2003, this proportion increased by about 10%. The proportion of leks with 20-49 males has remained relatively stable, varying between 30% and 40% over the assessment period (Fig. 6.13). From 1965 to 1984, 10 to 20% of the leks censused had >49 males. This range decreased to ≤10% from 1985 to 2003.

Annual rates of change suggest a long-term decline for sage-grouse in Montana (Fig. 6.14) and support the trend information obtained from lek attendance (males/lek) and lek class size. Sage-grouse populations declined at an overall rate of 1.6% per year from 1965 to 2003. Our analysis suggested a high likelihood of density dependence for the overall assessment period (likelihood = 0.98) and for both the early (likelihood = 0.97) and late (likelihood = 0.91) periods. From 1965-87, the population declined at an average rate of 2.69% and fluctuated around a level that was approximately 1.4 times higher than the 2003 population. From 1987 to 2003, the population fluctuated around a level that was approximately 9% below the 2003 population and had an average change of -0.07% per year. Populations in the late 1960s and early 1970s were approximately 2 times higher than current populations (Fig. 6.14). The population reached a low in the mid-1990s and has increased since that time. However, previous population recoveries (early and mid-1980s) did not reach levels attained in the late 1960s and early 1970s.

Summary. Connelly and Braun (1997) reported that populations in southeastern and southwestern Montana declined by about 30% when they compared average lek sizes from 1985-94 to long-term averages. Additionally, production declined by 17% (Connelly and Braun 1997). We increased the database used by Connelly and Braun (1997) and had an additional 9 years of lek count data. Our results suggested that population declines were somewhat greater than those reported by Connelly and Braun (1997) but supported their overall conclusion of a long-term population decline.

Table 6.7. Sage-grouse monitoring and population trends in Montana, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	546	180	167	175	160	106	61	13
Number of active leks[1]	404	131	130	142	147	98	56	13
Percent active leks	74	73	78	81	92	92	92	98
Average males/lek	17	14	15	17	25	23	24	29
Median males/lek	11	10	11	12	20	18	19	20
Average males/active lek	23	19	20	21	27	24	26	29
Median males/active lek	18	15	15	16	23	20	21	20

[1] Averaged over each year for each period.

Fig. 6.12. Change in lek size for sage-grouse in Montana, 1965 - 2003.

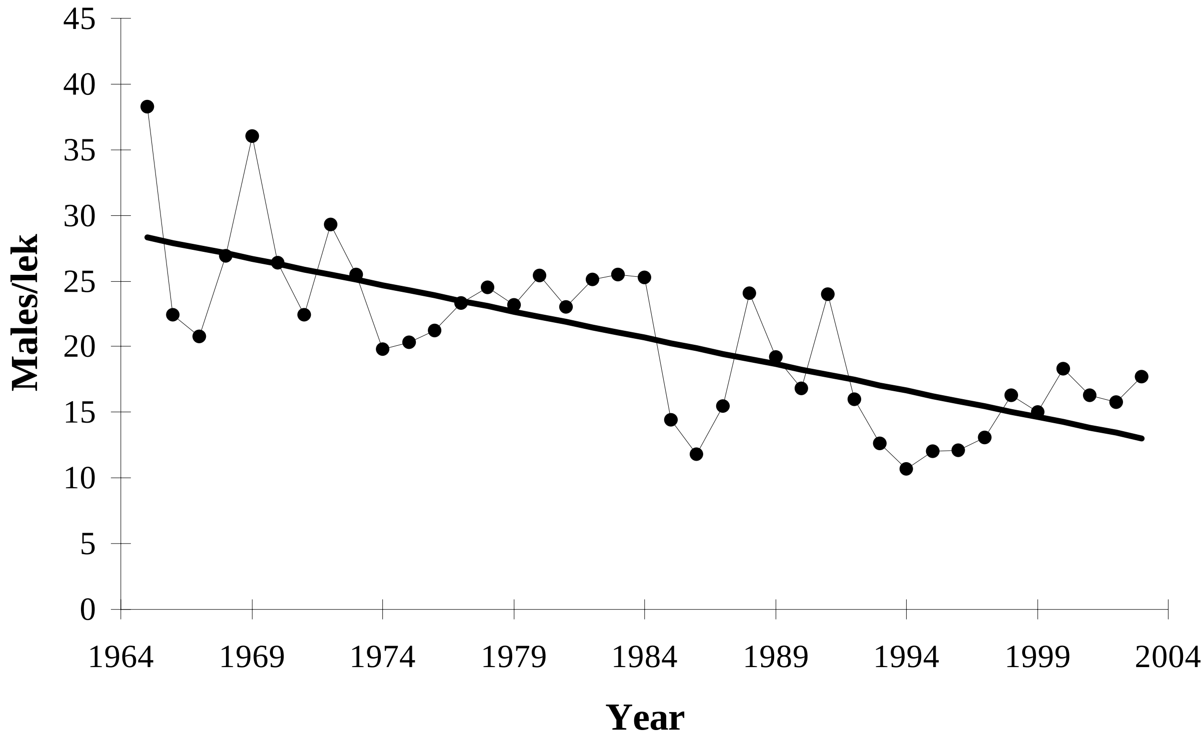


Fig. 6.13. Change in lek size class for Montana, summarized over 5-year periods, 1965 - 2003.

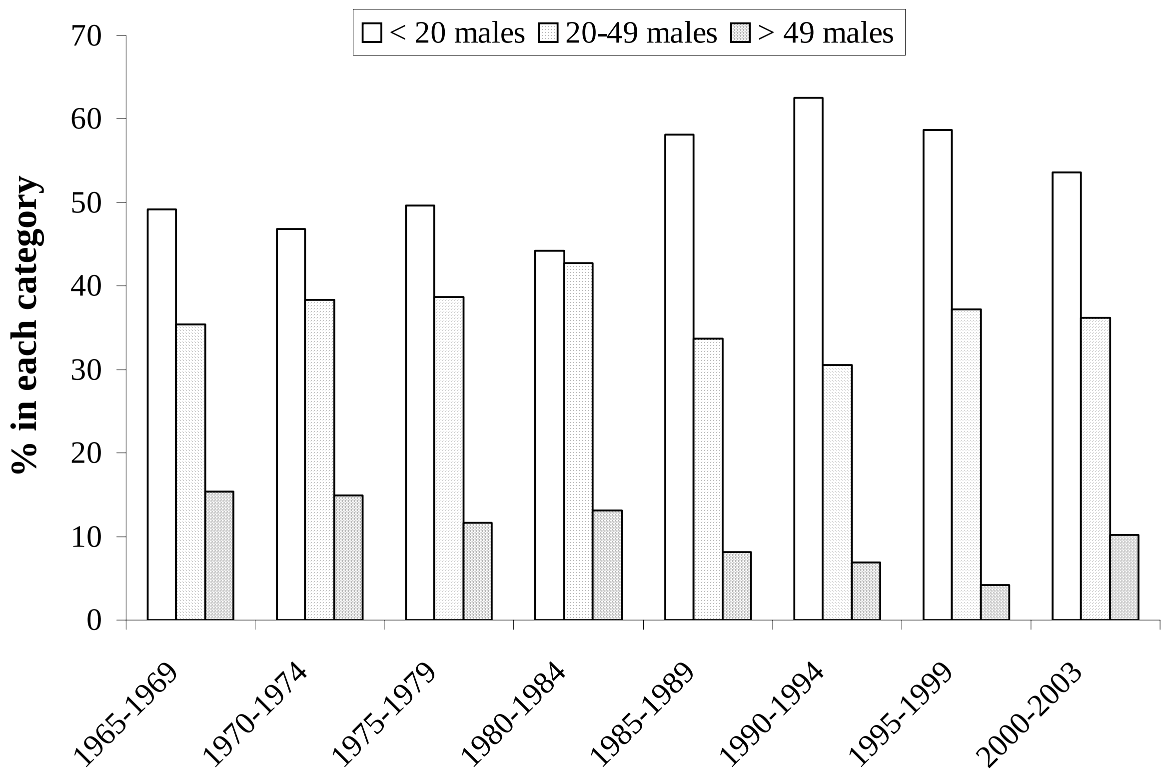
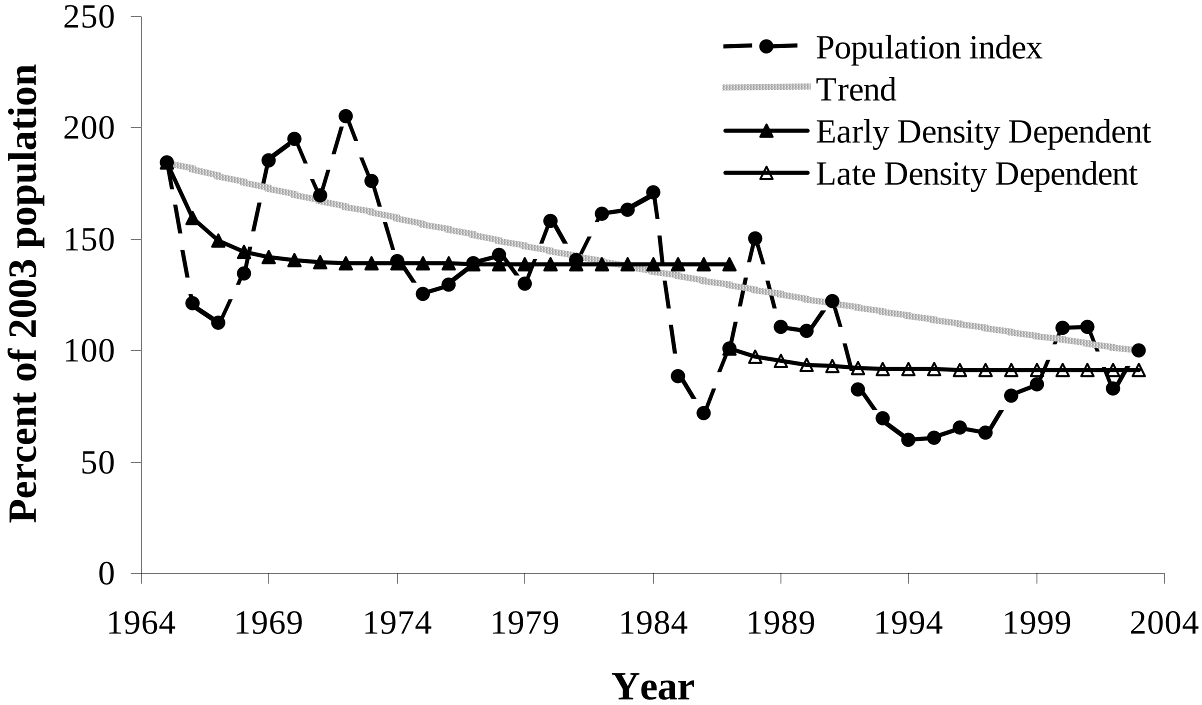


Fig. 6.14. Change in the population index for greater sage-grouse in Montana, 1965 -2003.



Nevada

Monitoring effort. Nevada has identified 1,077 sage-grouse leks within the state but monitoring efforts have been erratic. Because of inconsistent census efforts, we were able to assess change in lek size from 1965 to 2003 but could only examine changes in populations from 1974 to 2003. During the late 1960s and 1970s, relatively few leks were censused. However, the number of leks counted increased and then remained relatively stable until the late 1990s (Table 6.8). By 2000, monitoring efforts increased substantially when the average number of leks counted during 2000-03 increased by 146% over the average number of leks counted in 1995-99 (Table 6.8). Overall, the number of active leks monitored followed the same increasing pattern as total number of leks (Table 6.8).

Population Changes. The proportion of active leks remained relatively high over much of the assessment period. However, population trends indicated by average and median males per lek decreased over the assessment period by 48% and 57%, respectively (Table 6.8). Both average and median males per active lek declined by 37% and 42%, respectively over the assessment period (Table 6.8). Monitoring data (males/lek) indicated that lek size decreased significantly (r^2 =0.23, P = 0.00) from 1965 to 2003 (Fig. 6.15).

Beginning in the mid-1970s, the proportion of small leks increased. At the same time, the proportion of medium leks and large leks began to decrease (Fig. 6.16). From 1965 to 1979, 39% to 58% of the leks censused contained <20 males. From 1990 to 2003, this proportion increased to about 65%. The proportion of large leks decreased from 20% from 1965 to 1979 to 7% in 2000-03 (Fig. 6.16).

Annual rates of change suggest a long-term decline for sage-grouse in Nevada (Fig. 6.17) and support the trend information obtained from lek attendance (males/lek) and lek class size. Sage-grouse populations declined at an overall rate of 2.1% per year from 1974 to 2003. Our analysis provided some evidence of density dependence for the overall assessment period (likelihood = 0.74) and for the early (likelihood = 0.67) and late (likelihood = 0.53) periods. From 1974-85, the population declined at an average rate of 1.41% and fluctuated around a level that was approximately 2.1 times higher than the 2003 population. From 1986 to 2003, the population fluctuated around a level that was approximately 1.1% above the 2003 population and had an average change of -2.53% per year. Populations in the mid to late 1970s were approximately 1.2 to 3.5 times higher than 2003 populations (Fig. 6.17). Populations in the late 1960s and late 1970s fluctuated widely (Fig. 6.17) and there is no way of assessing whether these were actual changes in the populations or artifacts of sampling effort. The population reached a low in the mid-1990s and has not changed substantially since that time.

Summary. There is little published information on sage-grouse population trends in Nevada. The current data sets are somewhat ambiguous and likely reflect erratic monitoring efforts. Therefore, results from this analysis should be viewed cautiously.

Table 6.8. Sage-grouse monitoring and population trends in Nevada, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	182	74	64	52	68	33	22	22
Number of active leks[1]	145	66	62	51	65	24	17	21
Percent active leks	79	89	96	98	96	72	77	94
Average males/lek	15	16	19	22	27	25	22	29
Median males/lek	10	11	12	15	19	16	11	23
Average males/active lek	19	18	20	22	28	35	29	30
Median males/active lek	14	13	13	16	19	24	17	24

[1] Averaged over each year for each period.

Fig. 6.15. Changes in lek size for sage-grouse in Nevada, 1965-2003.

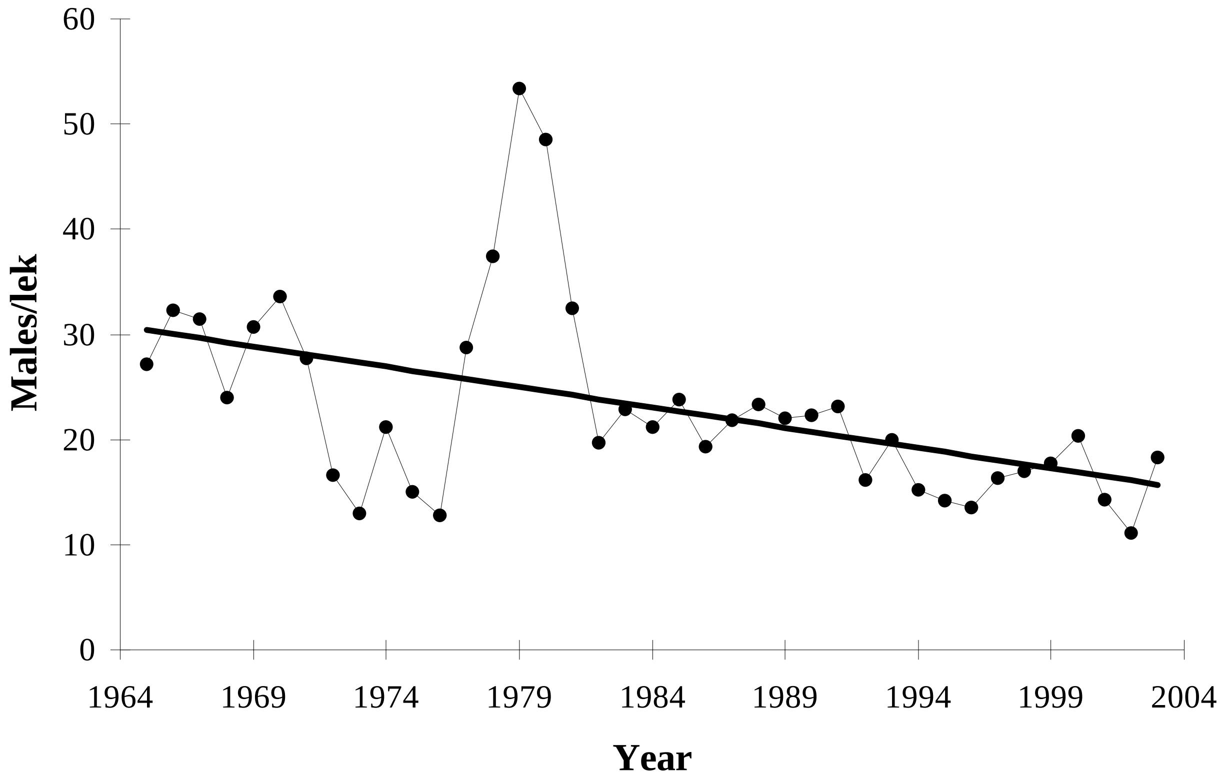


Fig. 7.16. Change in lek size class for Nevada, summarized over 5-year periods, 1965 - 2003.

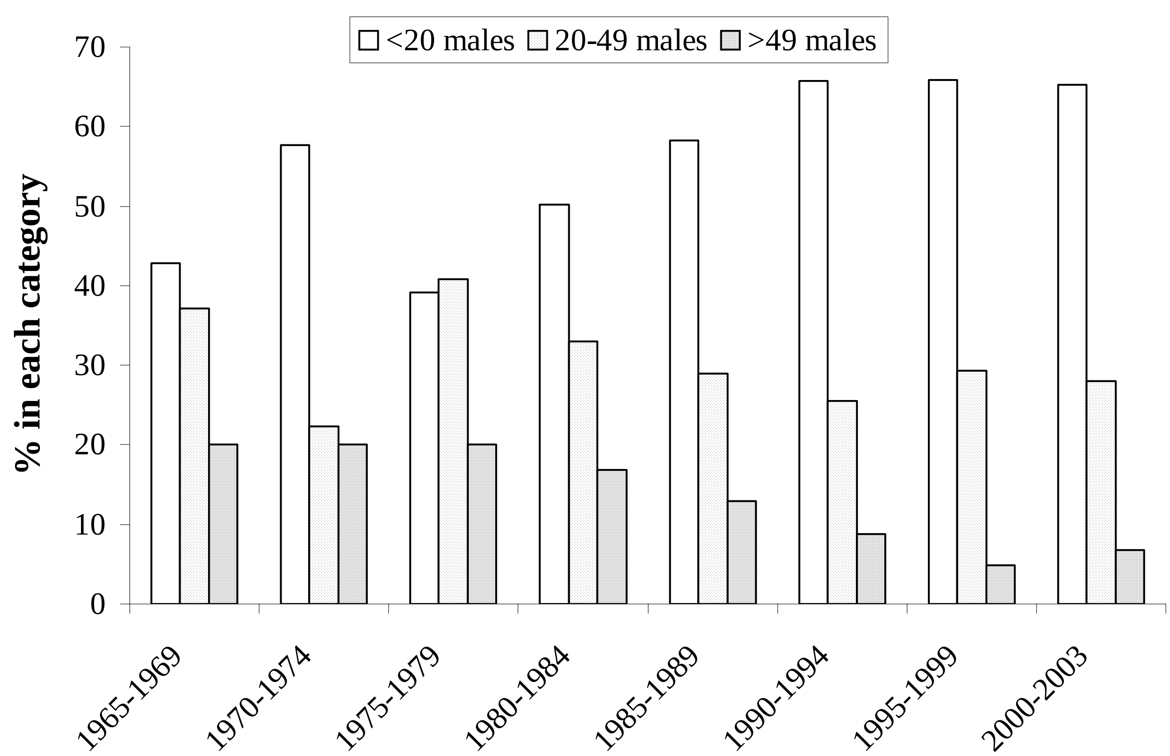
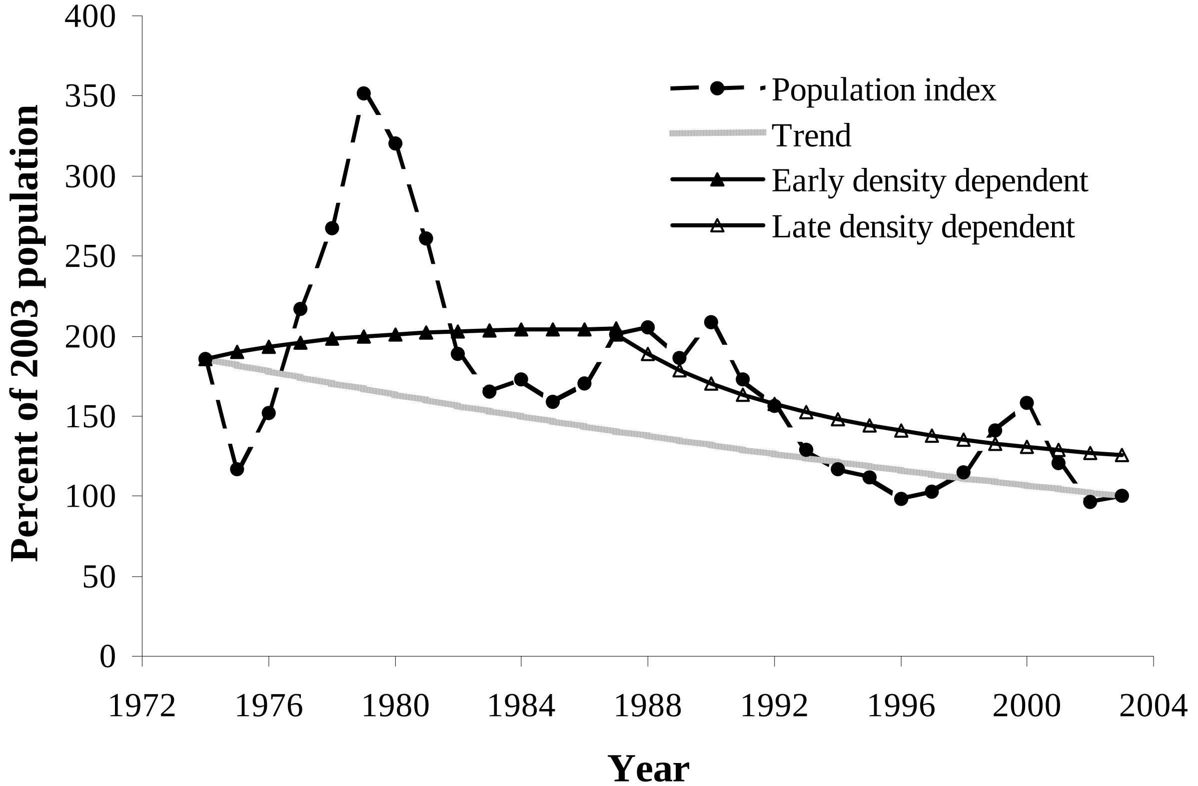


Fig. 6.17. Change in the population index for greater sage-grouse in Nevada, 1974-2003.



North Dakota

Monitoring effort. North Dakota identified 42 sage-grouse leks in the state and has rather consistently monitored their sage-grouse breeding population for over 40 years. Therefore, we used 1965 to 2003 as our assessment period. From 1965 to 2003, in 5-year periods, an average of 17 to 27 leks were monitored (Table 6.9). In 26 of 39 (67%) years, $\geq$ 20 leks were censused. The average number of leks counted per 5-year period increased by 42% from 1965 to 2003. Over these same 5-year periods, effective monitoring was relatively stable with an average of 14 to 21 active leks censused (Table 6.9). North Dakota did not employ a standard monitoring scheme of multiple counts spread over a 4-6 week period. Instead, all counts were conducted in about a one-week period during mid-April and observers attempted to count all leks $\geq$ 2 times (Smith 2003). However, this approach was consistently applied over the last 40 years.

Population Changes. The proportion of active leks decreased over the assessment period, averaging between 87% and 93% from the mid-60s to the mid-80s but decreasing to 58% by 2000-2003 (Table 6.9). Similarly, population trends indicated by average and median males per lek decreased over the assessment period by 38% and 80%, respectively (Table 6.9). Average and median males per active lek also indicated a decline over the assessment period, but this decline was not as great as that for males/lek (Table 6.9). Monitoring data (males/lek) indicated decreasing lek size (r^2 =0.35, P = 0.00) from 1965 to 2003 (Fig. 6.18). It appears that some subpopulations have disappeared while the number of breeding birds associated with some leks has declined.

The proportion of small leks did not change appreciably and was relatively high (Fig. 6.19), ranging from 59% to 90% averaged over 5-year periods beginning in 1965. Large leks (≥50 males) were seldom detected throughout the assessment period. This was likely the result of North Dakota sage-grouse population being on the easternmost edge of the species' range.

Annual rates of change suggest a long-term decline for sage-grouse in North Dakota (Fig. 6.20) and support the trend information obtained from lek attendance (males/lek) and lek class size. Sage-grouse populations declined at an overall rate of 2.79% per year from 1965 to 2003. Our analysis indicated a reasonably high likelihood of density dependence for the overall assessment period (likelihood = 0.86) and for both the early (likelihood = 0.76) and late (likelihood = 0.83) periods. From 1965-85, the population declined at an average rate of 4.48% and fluctuated around a level that was approximately 2.5 times higher than the 2003 population. From 1986 to 2003, the population fluctuated around a level that was approximately 1.4% above the 2003 population and had an average change of –0.66% per year. Populations in the late 1960s and early 1970s were approximately 3-6 times higher than current populations (Fig. 6.20).

Summary. Since the late 1960s, sage-grouse in North Dakota have declined by well over 50%. Connelly and Braun (1997) indicated that breeding populations declined by 27% when long-term averages were compared to data from 1985-94. Similarly, Smith (2003) reported a steady long-term decline in the number of male sage-grouse counted on leks. Our analysis suggests that the decline was much greater than that indicated by Connelly and Braun but supports Smith's (2003) conclusion. However, Smith (2003) also reported that grouse fluctuations roughly followed a 10-year cycle but our results do not show a cyclic pattern (Fig. 6.20).

Table 6.9. Sage-grouse monitoring and population trends in North Dakota, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	27	23	22	21	23	17	18	19
Number of active leks[1]	16	16	17	18	21	14	16.	17
Percent active leks	58	68	80	87	93	87	91	90
Average males/lek	8	6	11	11	14	11	16	13
Median males/lek	2	3	6	7	10	9	17	10
Average males/active lek	14	9	14	12	15	13	17	15
Median males/active lek	10	7	11	9	11	11	18	12

[1] Averaged over each year for each period.

Fig. 6.18. Changes in lek size for sage-grouse in North Dakota, 1965-2003.

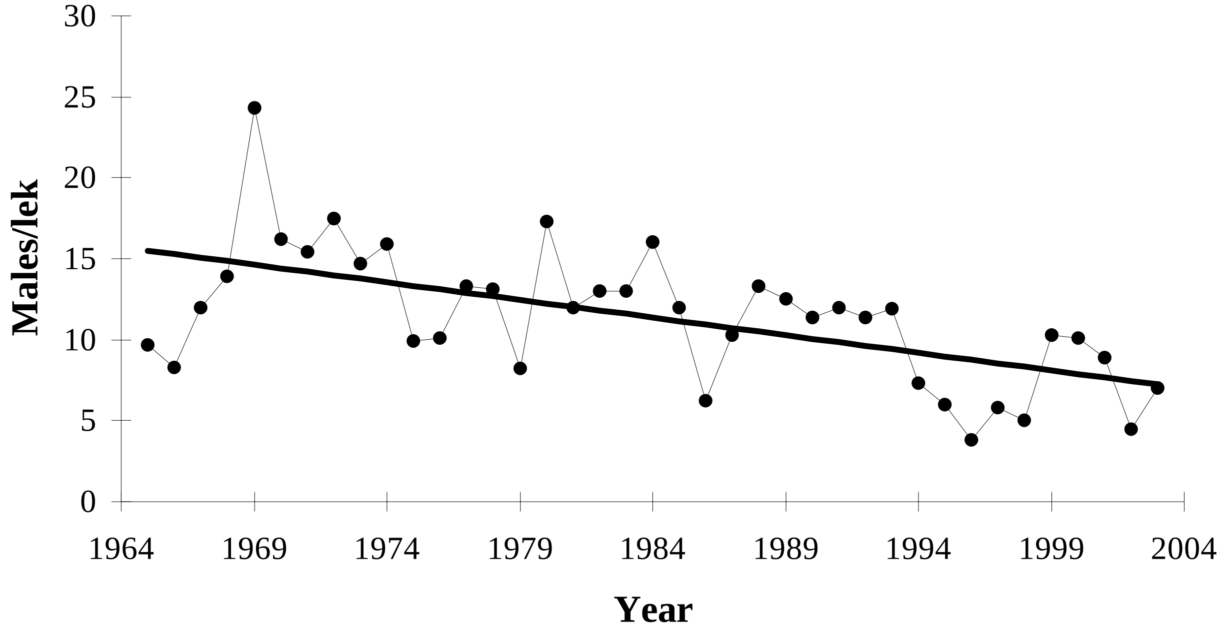


Fig. 6.19. Change in lek size class for North Dakota, summarized over 5-year periods, 1965 - 2003.

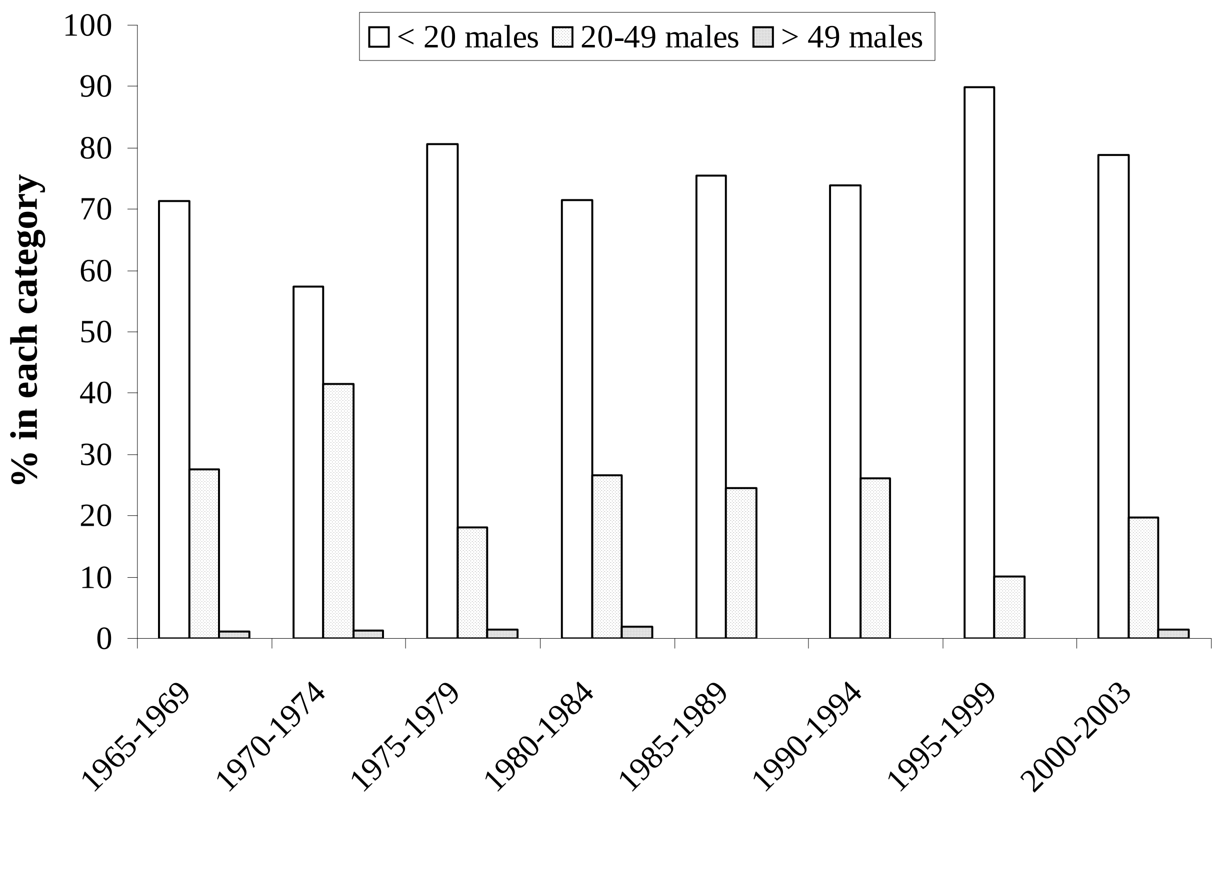
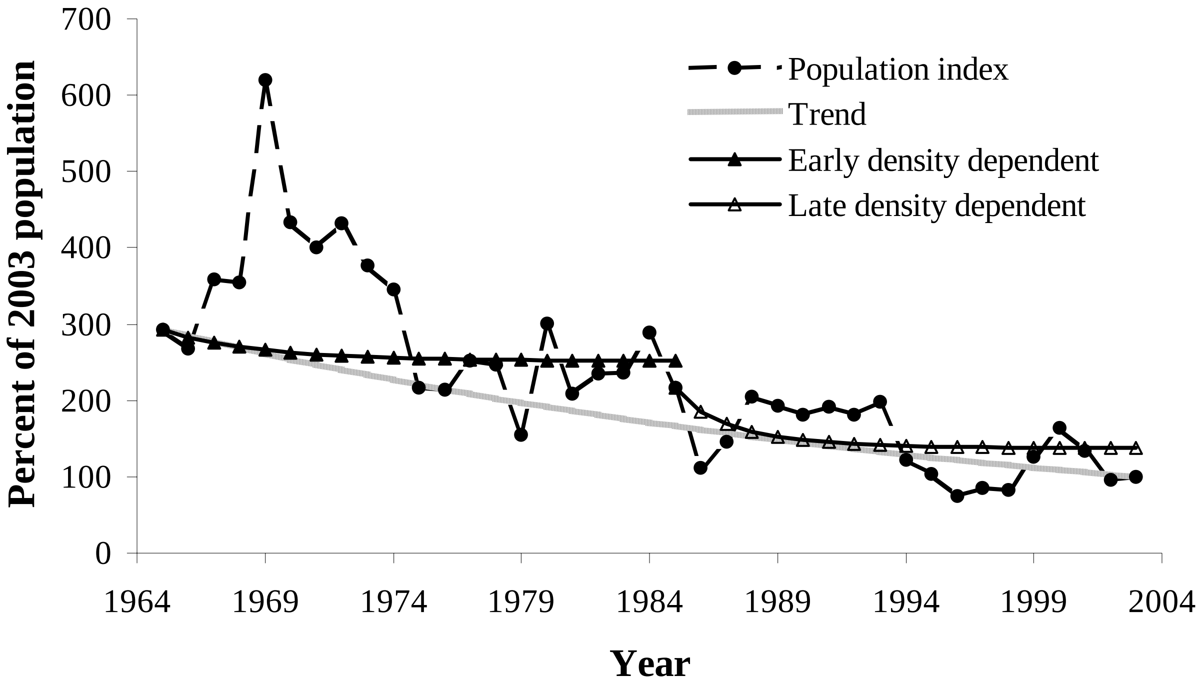


Fig. 6.20. Change in the population index for greater sage-grouse in North Dakota, 1965-2003



Oregon

***Monitoring effort*.** Oregon has had a long-term extensive monitoring program for sage-grouse and has identified 377 leks in the state. We used 1965-2003 as our assessment period. The average number of leks counted per 5-year period increased by 750% from 1965 to 2003 (Table 6.10). For the periods 1995-99 and 2000-03, ≥132 leks were censused in the state. The number of active leks monitored followed the same increasing pattern as total number of leks (Table 6.10).

Population Changes. The proportion of active leks remained relatively stable over the assessment period, ranging from 68% to 92% over the 5-year periods from 1965 to 2003 (Table 6.10). Population trends indicated by average and median males per lek fluctuated somewhat but largely remained unchanged over the assessment period (Table 6.10). Average and median males per active lek also changed little (Table 6.10). Monitoring data (males/lek) indicated that lek size did not change significantly (r^2 =0.05, P = 0.19) from 1965 to 2003 (Fig. 6.21).

Lek size classes fluctuated from 1965 to 2003 but there was no discernible trend (Fig. 6.22). Small leks were relatively common throughout the assessment period and averages ranged from 45% to 68% over each of the 5-year periods. Medium leks comprised ≥25% of the leks censused in each 5-year period. Large leks comprised a relatively small proportion of the leks sampled over the 5-year periods.

Annual rates of change suggest a long-term decline for sage-grouse in Oregon (Fig. 6.23) and thus do not support the trend information obtained from lek attendance (males/lek) and lek class size. Sage-grouse populations declined at an overall rate of 3.50% per year from 1965 to 2003. Our analysis indicated a reasonably high likelihood of density dependence for the overall assessment period (likelihood = 0.95) and for both the early (likelihood = 0.78) and late (likelihood = 0.74) periods. From 1965-85, the population declined at an average rate of 7.33% and fluctuated around a level that was approximately 10% greater than the 2003 population. From 1986 to 2003, the population fluctuated around a level that was approximately 13% above the 2003 population and had an average change of 0.95% per year. Populations in the late 1960s and early 1970s were approximately 2 to 4 times higher than current populations (Fig. 6.23). The population reached lows in the mid 1970s and mid 1990s and has increased somewhat since that time. However, a previous population recovery (late 1970s) did not reach levels attained in the late 1960s and early 1970s (Fig. 6.23).

Summary. Crawford and Lutz (1985) reported significant declines in Oregon sage-grouse populations through the early 1980s. More recently, Connelly and Braun (1997) reported that breeding populations in Oregon declined by 30% when they compared average lek sizes from 1985-94 to long-term averages. Production also decreased by 51% when 1985-94 data were compared to long-term averages (Connelly and Braun 1997). However, recent brood survey data from Oregon indicates that average production from 1985 to 2003 has steadily increased (average = 1.55 chicks per hen), and indicates a 37% reduction in production from the long-term average (Oregon Department of Fish and Wildlife, personal communication). We increased the database used by Connelly and Braun (1997) and had an additional 9 years of lek count data. Because lek size showed no apparent change over the assessment period, but overall populations declined, these losses are likely due to the disappearance of subpopulations associated with individual leks or groups of leks. Further, our results suggested that population declines were considerably greater than those reported by Connelly and Braun (1997) but supported their overall conclusion of a long-term population decline.

Table 6.10. Sage-grouse monitoring and population trends in Oregon, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	153	132	76	50	34	16	21	18
Number of active leks[1]	133	119	68	45	26	11	17	17
Percent active leks	87	90	89	89	76	68	80	92
Average males/lek	22	16	23	27	19	13	20	22
Median males/lek	14	12	16	19	11	9	11	14
Average males/active lek	25	17	26	30	25	19	25	24
Median males/active lek	18	13	17	21	18	14	14	15

[1] Averaged over each year for each period.

Fig. 6.21. Changes in lek size for sage-grouse in Oregon, 1965-2003.

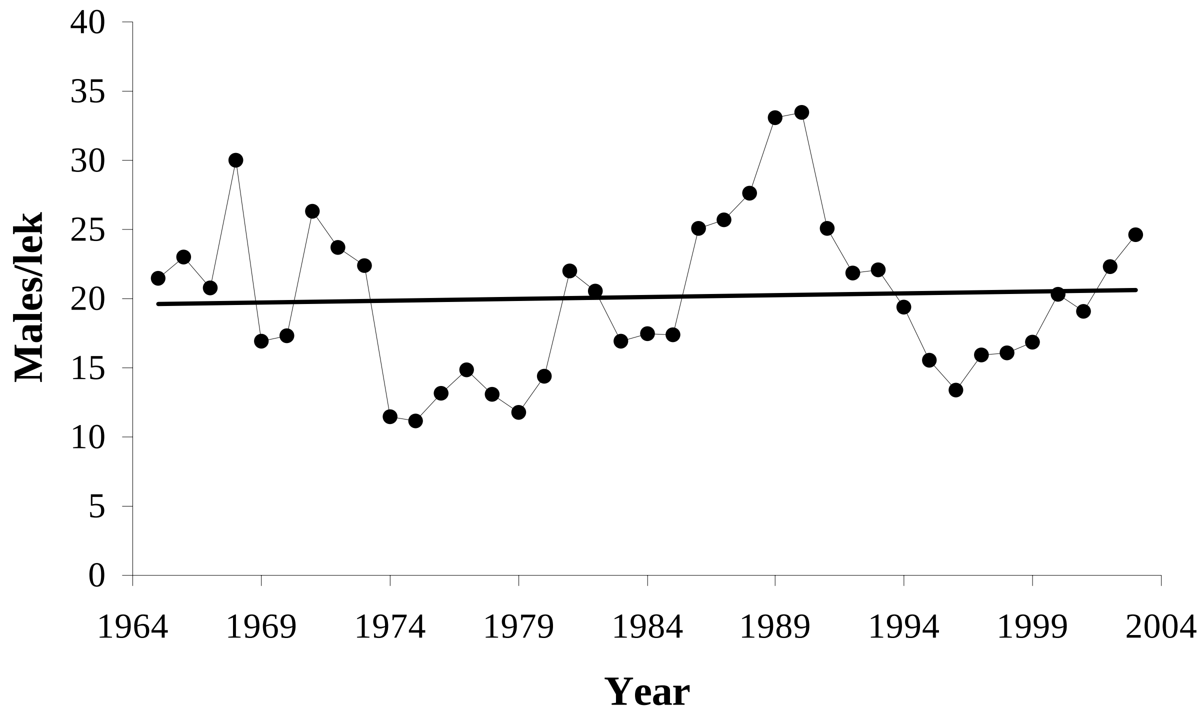


Fig. 6.22. Change in lek size class for Oregon, summarized over 5-year periods, 1965 - 2003.

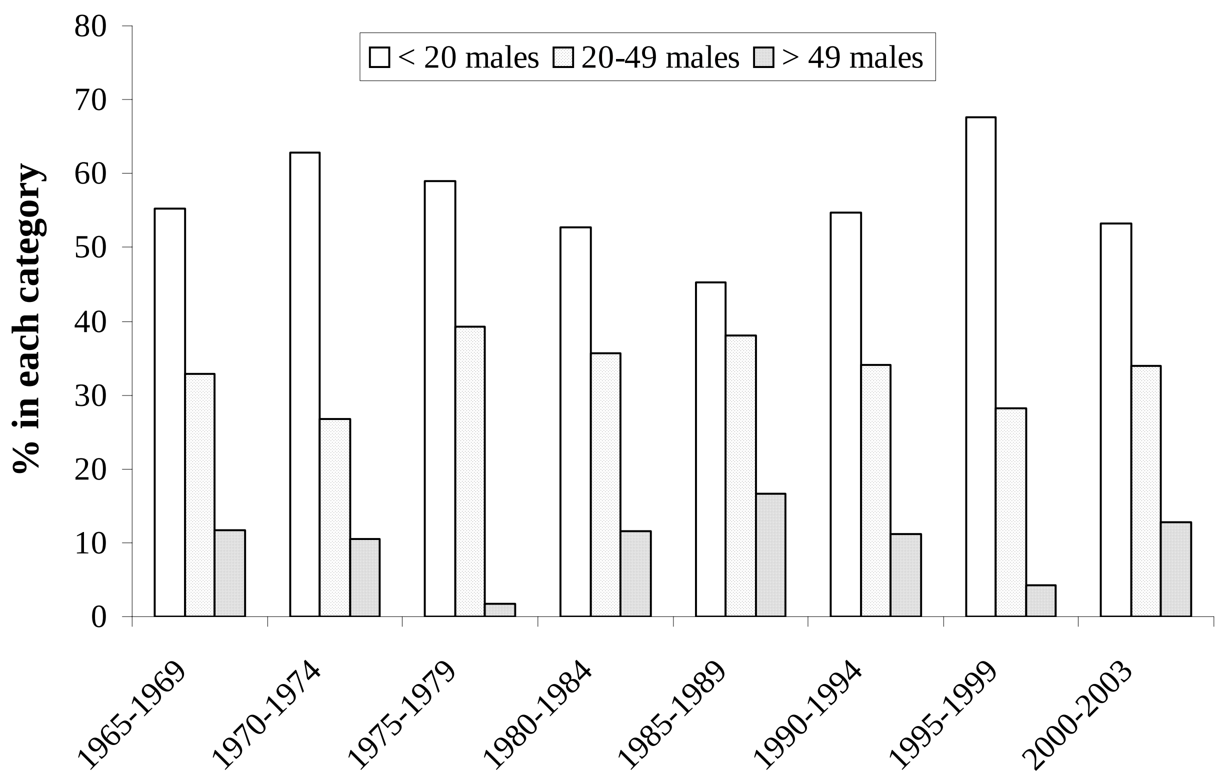
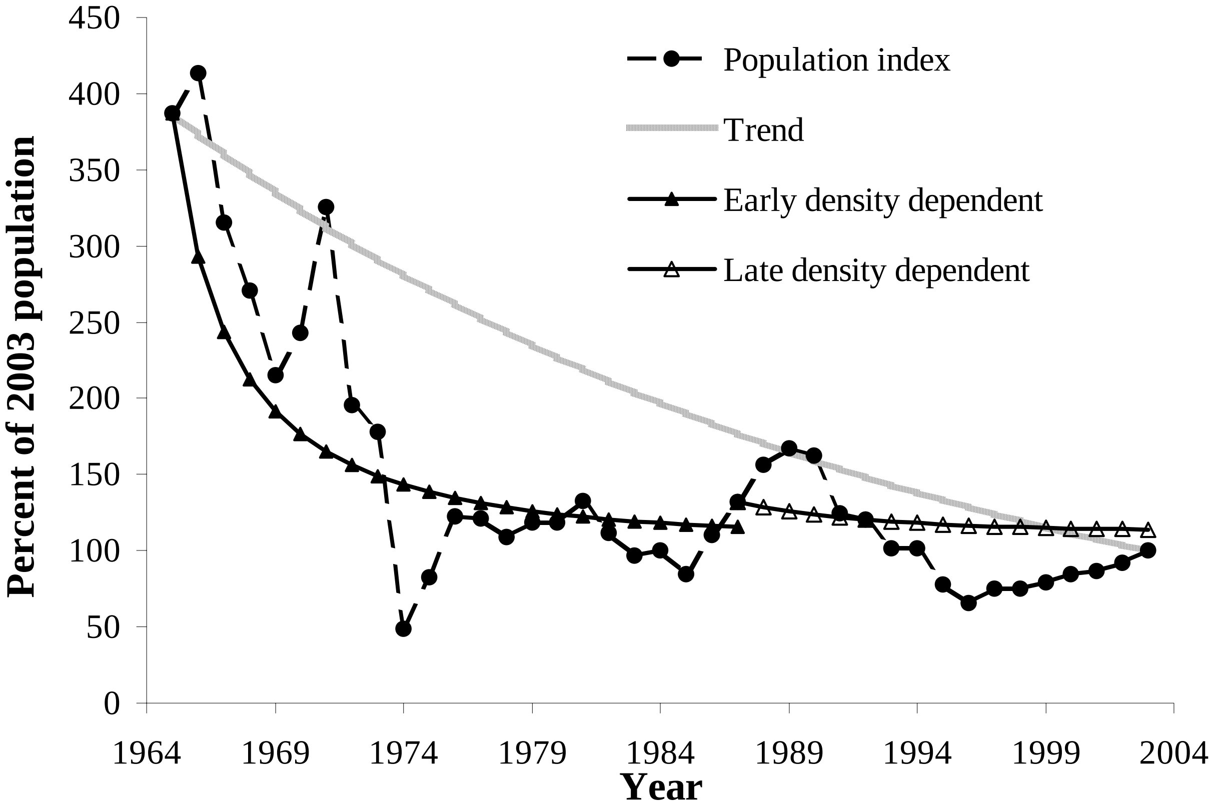


Fig. 6.23. Change in the population index for greater sage-grouse in Oregon, 1965 - 2003.



Saskatchewan

***Monitoring effort**.* Saskatchewan has identified 35 leks since the mid-1980s and routine censusing began in 1994 (Aldridge and Brigham 2003). Consequently, we present some data from the late 1980s but consider 1994-2003 as our assessment period. From 1985 to 2003, in 5-year periods, an average of 7 to 20 leks have been counted (Table 6.11). Over these same periods, the average number of active leks declined 58%, from 19 in 1985-89 to 8 in 2000-03.

Changes in populations. The proportion of active leks decreased over the assessment period, averaging 87% in the 1985-89 period and decreasing to 51% in 2000-03 (Table 6.11). Similarly, population trends indicated by average and median males per lek also decreased from 1985-89 to 2000-03 but increased somewhat from 1994 to 2003. Average and median males per active lek also showed similar trends (Table 6.11). Monitoring data were only sufficient to examine change in lek size from 1994 to 2003. Over that period, the population did not change significantly (r^2 =0.30 P= 0.10) (Fig. 6.24). Thus, it appears sage-grouse populations may have been stable over the last 10 years but it is not possible to compare these trends to long-term data.

During 1985-89, 42% of the leks had >20 males and about 14% of the leks contained >50 males (Fig. 6.25). Currently, over 90% of the leks have <20 males, suggesting a decline in lek size and conflicting with data on males per lek.

Because of relatively small samples of leks and inconsistent monitoring, we were unable to calculate long-term annual rates of change.

Summary. McAdam (2003) concluded that sage-grouse in Saskatchewan have undergone a profound population and range reduction since the late 1980s, and further indicated that an estimated population of 2,500 in 1987 decreased to 240 birds in 1997. McAdam (2003) also reported that lek abandonment occurred after 1987 and the remaining occupied leks decreased in size. Aldridge and Brigham (2003) also reported a decline in the Saskatchewan population and indicated the mean number of males per lek was 21.8 in 1988 but this declined to 6.8 males per lek by 1994, a 64% loss. Our analysis agrees with that of both McAdam (2003) and Aldridge and Brigham (2003) although we provide some evidence of a stable population over the last 10 years. However, sage-grouse in Saskatchewan apparently declined by 60-90% prior to 1994.

Table 6.11. Sage-grouse monitoring and population trends in Saskatchewan, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94[2]	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	16	20	7	13	0	0	0	0
Number of active leks[1]	8	10	132	19	0	0	0	0
Percent active leks	51	50	412	87				
Average males/lek	6	5	32	21				
Median males/lek	1	1	02	16				
Average males/active lek	12	10	72	24				
Median males/active lek	11	8	42	18				

[1] Averaged over each year for each period.
[2]Only data for 1994 available.

Fig. 6.24. Changes in lek size for sage-grouse in Saskatchewan, 1994-2003.

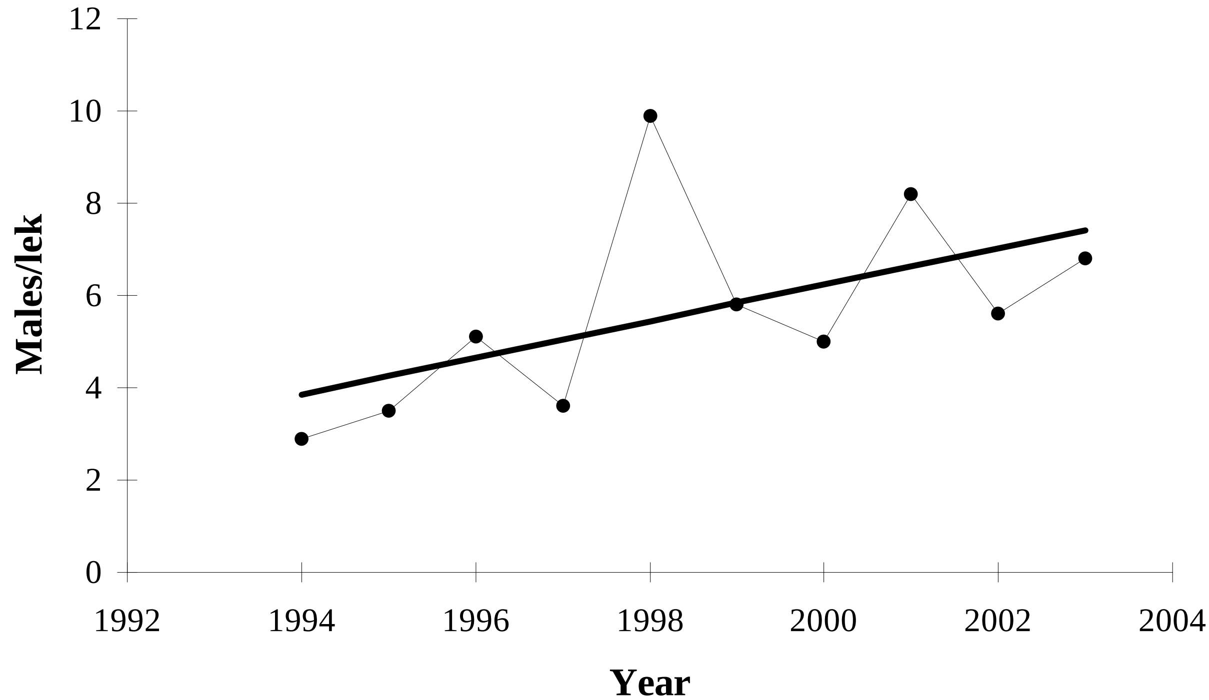
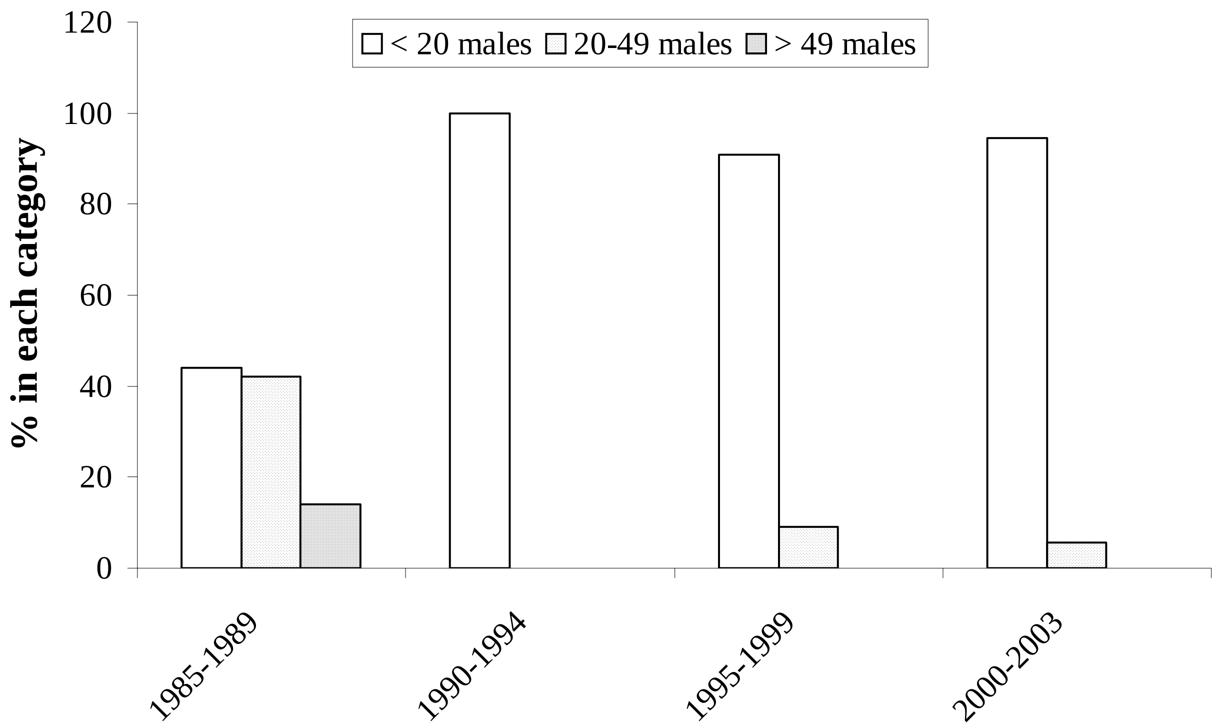


Fig. 6.25. Change in lek size class for Saskatchewan, summarized over 5-year periods, 1987 - 2003.



South Dakota

Monitoring effort. South Dakota has identified 24 leks. Population monitoring efforts were sporadic and data were often collected late in the breeding season (e.g., 64% of the counts made in Butte County in 1973 [n = 11] occurred after 1 May, no lek was counted >1 time and data for 2 counts failed to differentiate between genders). Smith (2003) also cautioned against using data collected prior to 1989. Consequently, we only used lek count data obtained since 1990 to assess population trends for this state. From 1990 through 2003, the number of leks censused remained relatively stable but the number of active leks decreased by 46% (Table 6.12).

Population change. The average percent of active leks censused decreased from 87% during 1990-94 to 45% in 2000-03. The average and median males per lek also decreased by 50% and 100%, respectively (Table 6.12). The average and median males per active lek remained largely unchanged over this period. Moreover, analysis of monitoring data indicated no significant change ($P = 0.40$) in lek size from 1990-2003 (Fig. 6.26). This may suggest that population declines were largely due to loss of subpopulations associated with certain leks or groups of leks, likely as a result of habitat change. However, an assessment of change in proportion of lek size classes suggests a decrease of medium leks and an increase of small leks over the assessment period. In 1990-94, about 65% of the leks contained <20 males and about 35% had 20-49 males. By 200-03, about 80% of the leks contained <20 males while 20% had 20-49 males (Fig. 6.27). Because the data are somewhat ambiguous, they do not allow a meaningful assessment of population status and trends.

Summary. Connelly and Braun (1997) indicated that South Dakota breeding populations declined by 45% when they compared the long-term average of males/lek to the average for 1985-94. Although some of their analysis for South Dakota was based on questionable data, our present assessment also suggested a decline of similar magnitude. Smith (2003) reported that 12 of 25 (48%) known leks in South Dakota are currently active and that one of these leks represents a small, isolated population of sage-grouse in Fall River County. Smith (2003) also concluded that South Dakota sage-grouse populations underwent a steady decline from 1973 to 1997, with recovery from 1997 to 2002. However, he cautioned that survey efforts in South Dakota have been very inconsistent. Available data suggest that a large part of the decline was the result of loss of entire subpopulations but there is also some evidence suggesting some decline in numbers of birds associated with existing leks.

Table 6.12. Sage-grouse monitoring and population trends in South Dakota, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	16	8	15	0	0	0	0	0
Number of active leks[1]	7	7	13	0	0	0	0	0
Percent active leks	45	80	87					
Average males/lek	6	6	12					
Median males/lek	0	6	8					
Average males/active lek	13	8	13					
Median males/active lek	12	7	12					

[1] Averaged over each year for each period.

Fig. 6.26. Change in lek size for sage-grouse in South Dakota, 1990-2003.

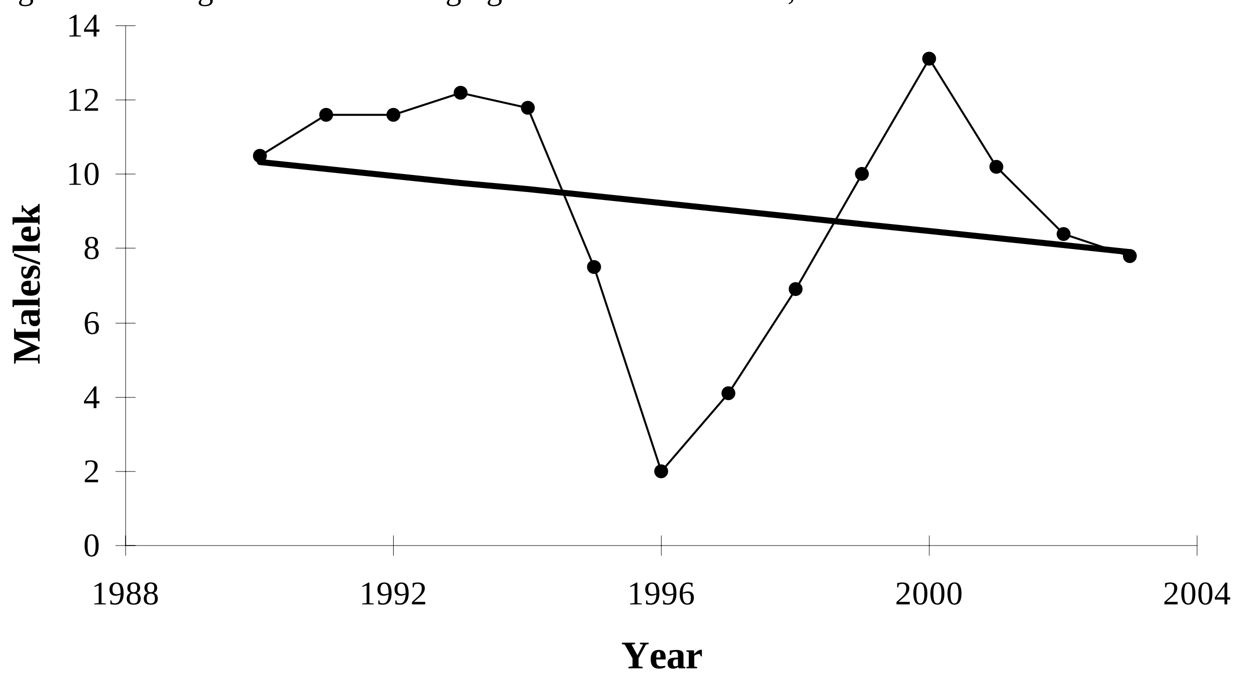
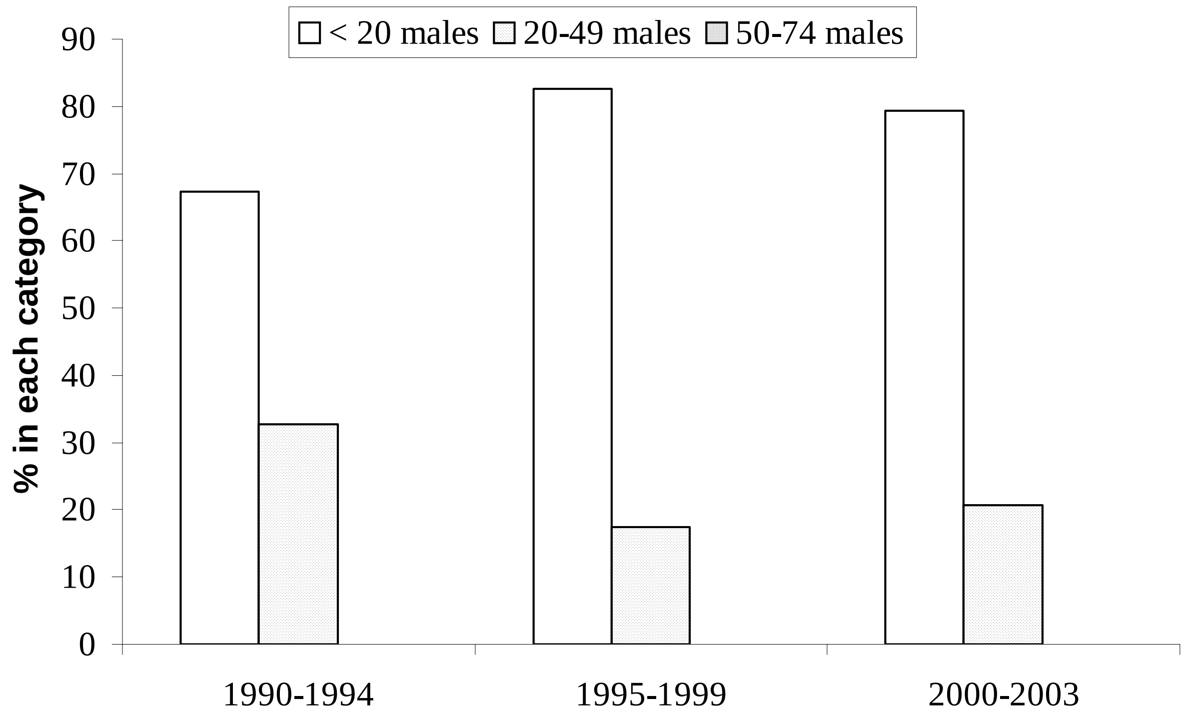


Fig. 6.27. Change in lek size class for South Dakota, summarized over 5-year periods, 1990 - 2003.



Utah

Monitoring effort. Utah has had a long-term extensive monitoring program for sage-grouse and has identified 254 leks in the state. Although the average number of leks monitored in the 1970-75 period increased by >160% over the average number censused in 1965-70, we were still able to use 1965-2003 as our assessment period. The average number of leks counted per 5-year period increased by 289% from 1965-70 to 2000-03 (Table 6.13). The number of active leks monitored followed the same increasing pattern as total number of leks (Table 6.13).

Population Changes. The proportion of active leks decreased somewhat over the assessment period, ranging from 89% to 72% over the 5-year periods from 1965 to 2003 (Table 6.13). Population trends indicated by average and median males per lek decreased over the assessment period by 19% and 42%, respectively (Table 6.13). However, average and median males per active lek changed little over the assessment period (Table 6.13). Monitoring data (males/lek) indicated that lek size changed significantly (r^2 =0.16, P = 0.01) from 1965 to 2003 (Fig. 6.28) but these data varied greatly and should be viewed with caution.

The proportion of small leks increased somewhat over the assessment period. In general, lek size classes fluctuated from 1965 to 2003 but there were no major changes, especially in the proportion of medium leks (Fig. 6.29). Large leks comprised <20% of the leks sampled over the 5-year periods throughout the assessment period.

Annual rates of change suggest a long-term decline for sage-grouse in Utah (Fig. 6.30) and support the trend information obtained from lek attendance (males/lek) and lek class size. Sage-grouse populations declined at an overall rate of 0.35% per year from 1965 to 2003. Our analysis indicated a reasonably high likelihood of density dependence for the overall assessment period (likelihood = 0.90) and for both the early (likelihood = 0.87) and late (likelihood = 0.71) periods. From 1965-85, the population declined at an average rate of 0.83% and fluctuated around a level that was approximately 1.4 times higher than the 2003 population. From 1986 to 2003, the population fluctuated around a level that was approximately 5% below the 2003 population and increased at an average rate of 0.18% per year. Populations in the early 1970s were approximately 2 times higher than current populations (Fig. 6.30). The population reached a low in the mid-1990s and has increased considerably since that time. However, previous population recoveries (late 1970s and late 1980s) did not reach levels attained in the late 1960s and early 1970s.

Summary. Connelly and Braun (1997) reported that breeding populations in Utah declined by 30% when they compared average lek size from 1985-94 to long-term averages. However, production increased by 1% when 1985-94 data were compared to long-term averages (Connelly and Braun 1997). Beck et al. (2003) reported that males per lek declined for all populations in Utah from 1971 to 2000 but they also provided some evidence indicating populations in 3 counties may be stable or increasing. Our results suggested that population declines may have been greater than those reported by Connelly and Braun (1997) and Beck et al. (2003) but supported their overall conclusion of a long-term population decline.

Table 6.13. Sage-grouse monitoring and population trends in Utah, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	144	118	99	109	94	108	97	37
Number of active leks[1]	111	85	82	84	74	92	83	33
Percent active leks	77	72	83	77	79	86	85	89
Average males/lek	21	15	19	19	19	24	23	26
Median males/lek	11	8	10	11	12	13	16	19
Average males/active lek	28	21	22	24	24	28	27	30
Median males/active lek	19	14	15	16	17	17	20	21

[1] Averaged over each year for each period.

Fig. 6.28. Changes in lek size for sage-grouse in Utah, 1965 - 2003.

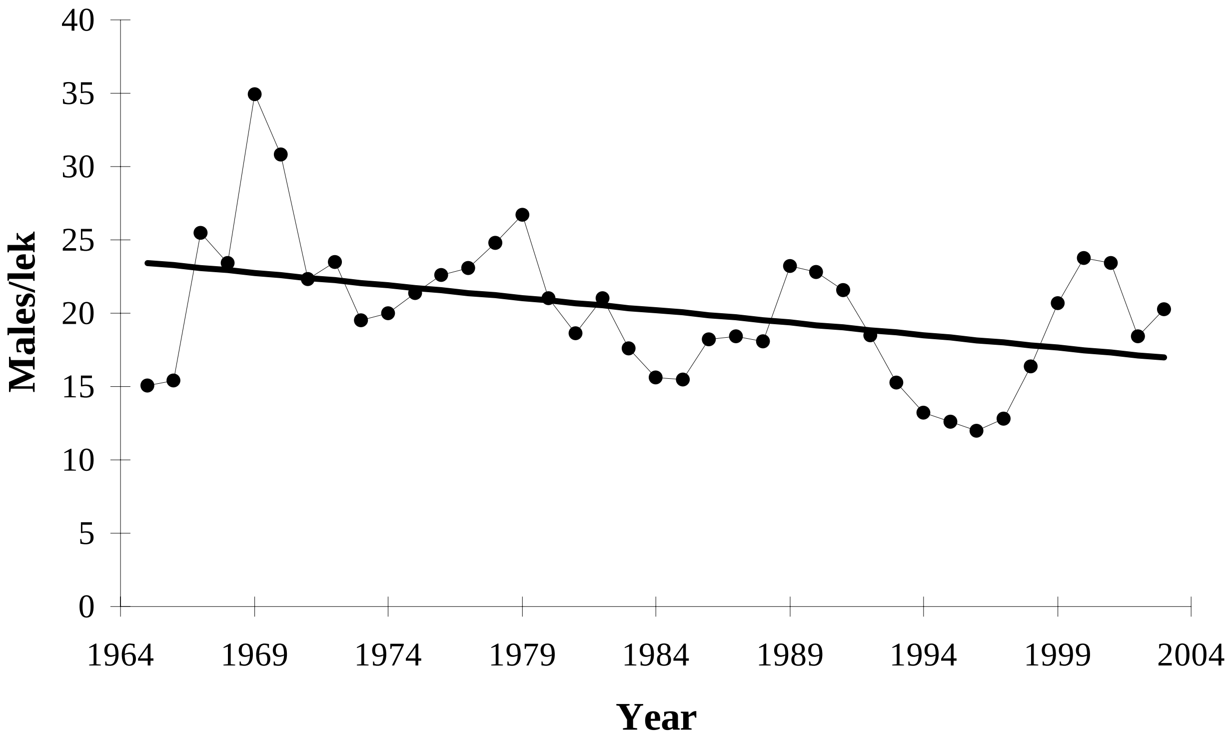


Fig. 6.29. Change in lek size class for Utah, summarized over 5-year periods, 1965 - 2003.

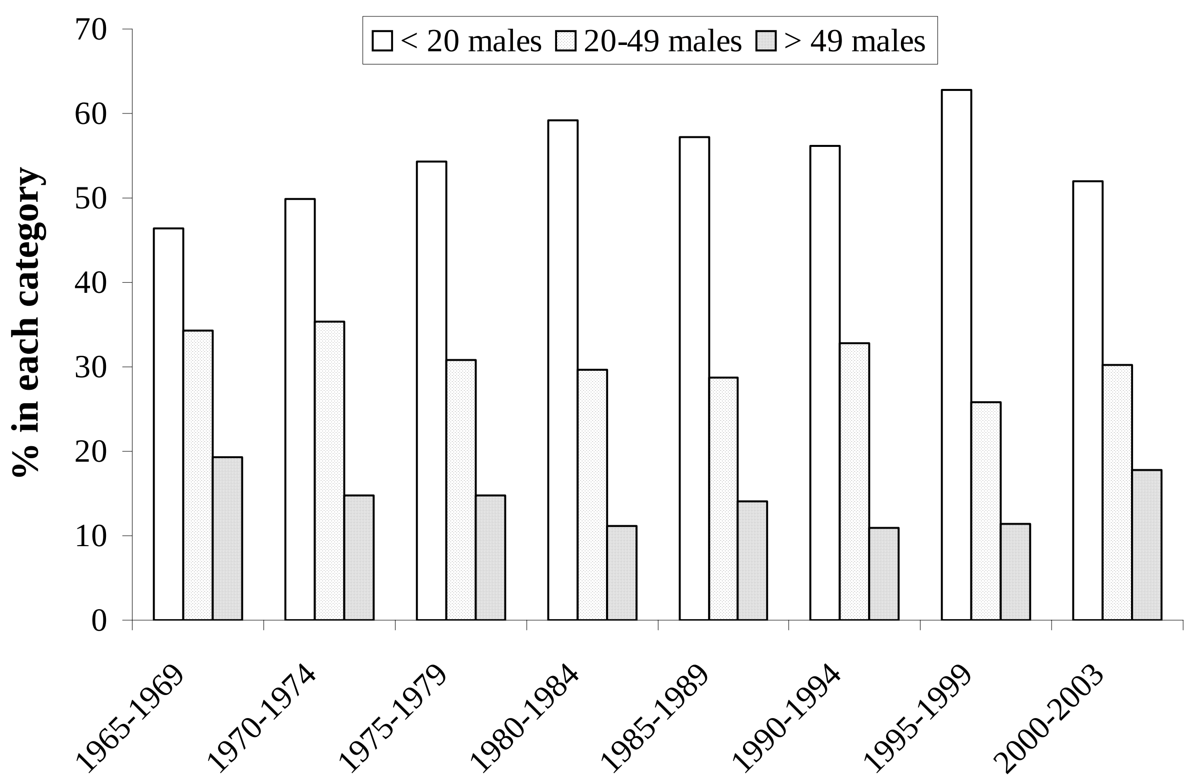
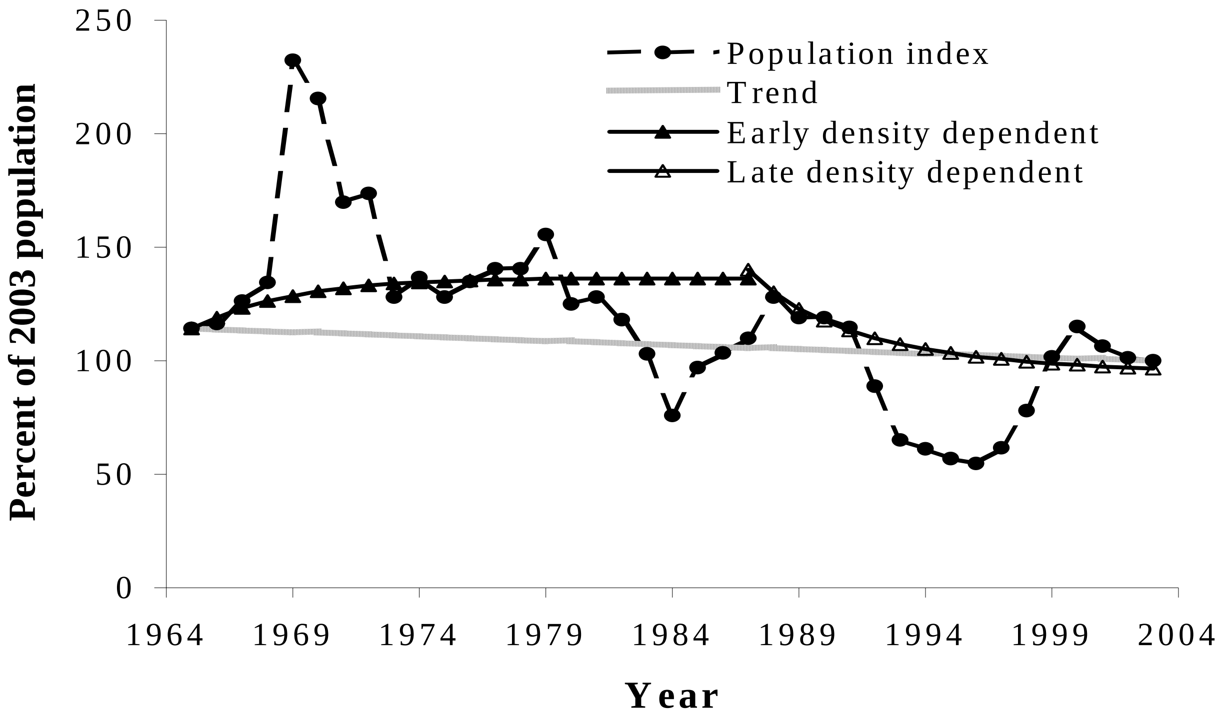


Fig. 6.30. Change in the population index for greater sage-grouse in Utah, 1965 - 2003.



Washington

Monitoring effort. Washington has identified 62 leks and has had a long-term monitoring program in place. Thus, we used 1965-2003 as the assessment period. The average number of leks counted per 5-year period increased substantially over the assessment period (Table 6.14). In 1965-69, an average of 3 leks per year were censused but by 2000-03, an average of 47 leks per year were counted, an increase of >1400%. The average number of active leks counted per 5-year period also increased by >500%. Observers in Washington generally followed established procedures for lek census and attempted to count leks $\geq$3 times over an approximately 5-week period from mid-March to early May.

Population Change. The proportion of active leks decreased over the assessment period, averaging between 92% and 100% from 1965 to 1984 but decreased to 41% by 2000-2003 (Table 6.14). Population trends indicated by average and median males per lek also decreased over the assessment period by >75% (Table 6.14). Average and median males per active lek also indicated a decline over the assessment period, and both decreased by about 45% (Table 6.14). Monitoring data indicated that lek size decreased significantly (r^2 =0.69, P = 0.00) from 1965 to 2003 (Fig. 6.31). Although some subpopulations have disappeared (Schroeder et al. 2000), it also appears that the number of breeding birds associated with some leks has declined substantially.

Relatively few leks were censused from 1965 to 1984 (Table 6.14) so information on changes in lek size classes during these years should be viewed with caution. From 1985 to

2003, the proportion of small leks remained relatively steady while the proportion of medium leks increased. Over the same period, large leks decreased from about 18% to 3% (Fig. 6.32).

Annual rates of change suggest a long-term decline for sage-grouse in Washington (Fig. 6.33) and support the trend information obtained from lek attendance (males/lek) and lek class size. Sage-grouse populations declined at an overall rate of 4.79% per year from 1965 to 2003. Our analysis indicated a high likelihood of density dependence for the overall assessment period (likelihood = 0.96) and for both the early (likelihood = 0.83) and late (likelihood = 0.84) periods. From 1965-85, the population declined at an average rate of 8.73% and fluctuated around a level that was approximately 1.4 times higher than the 2003 population. From 1986 to 2003, the population fluctuated around a level that was approximately 1.2% above the 2003 population and had an average change of –0.20% per year. Populations in the late 1960s and early 1970s were approximately 4-6 times higher than 2003 populations (Fig. 6.33).

Summary. The relatively large decrease in active leks over the assessment period may be due to inconsistent data collection (Schroeder et al. 2000), a tendency of early biologists to only census active leks or an actual decrease in the number of active leks. Most likely it was a combination of these factors. Connelly and Braun (1997) reported that sage-grouse populations had declined by 47% in Washington. Schroeder et al. (2000) reported that sage-grouse in Washington declined by at least 77% from 1960 to 1999 and that the current populations only occupy about 8% of historic range. Our current analysis suggests that Connelly and Braun's (1997) estimate was likely too conservative and generally supported Schroeder et al.'s (2000) estimate.

Table 6.14. Sage-grouse monitoring and population trends in Washington, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	47	31	25	20	18	15	14	3
Number of active leks[1]	20	18	16	14	16	14	13	3
Percent active leks	41	56	62	71	91	92	99	100
Average males/lek	8	11	13	17	26	15	24	33
Median males/lek	0	5	4	6	23	13	20	31
Average males/active lek	18	20	20	23	28	16	25	33
Median males/active lek	17	18	14	14	24	14	20	31

[1] Averaged over each year for each period.

Fig. 6.31. Change in lek size for sage-grouse in Washington, 1965 - 2003.

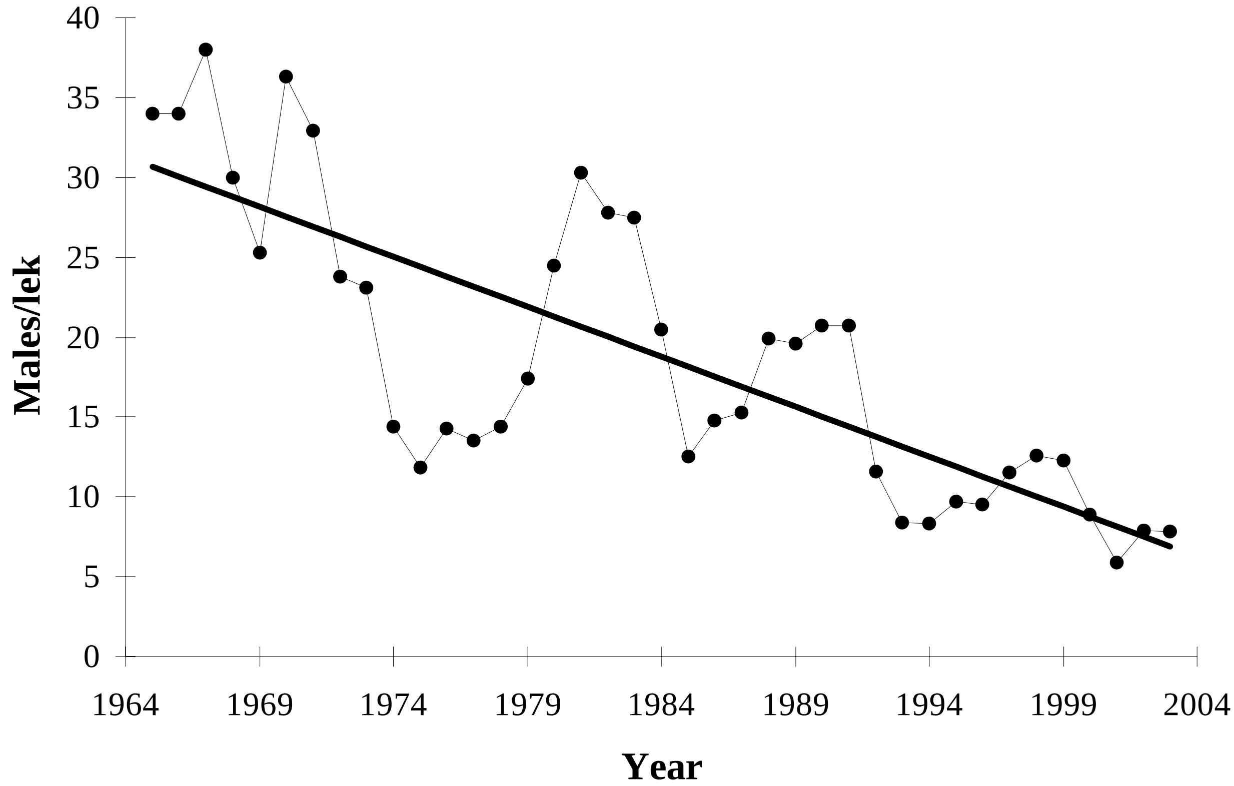


Fig. 6.32. Change in lek size class for Washington, summarized over 5-year periods, 1965 - 2003.

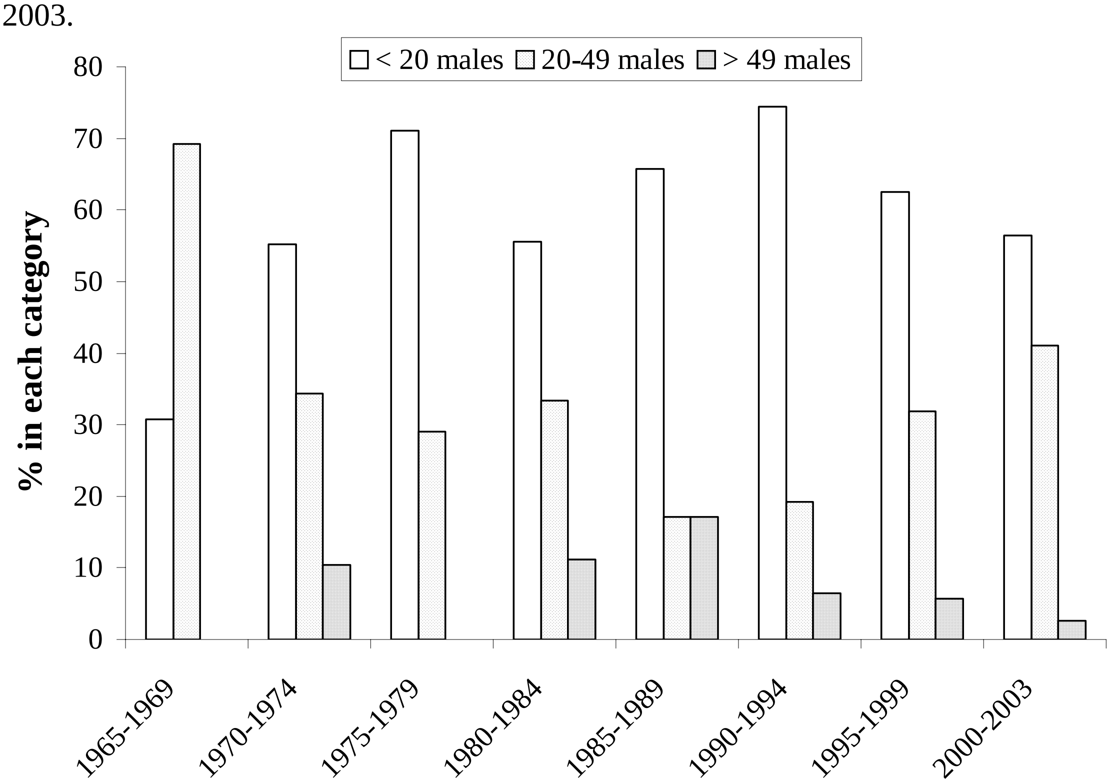
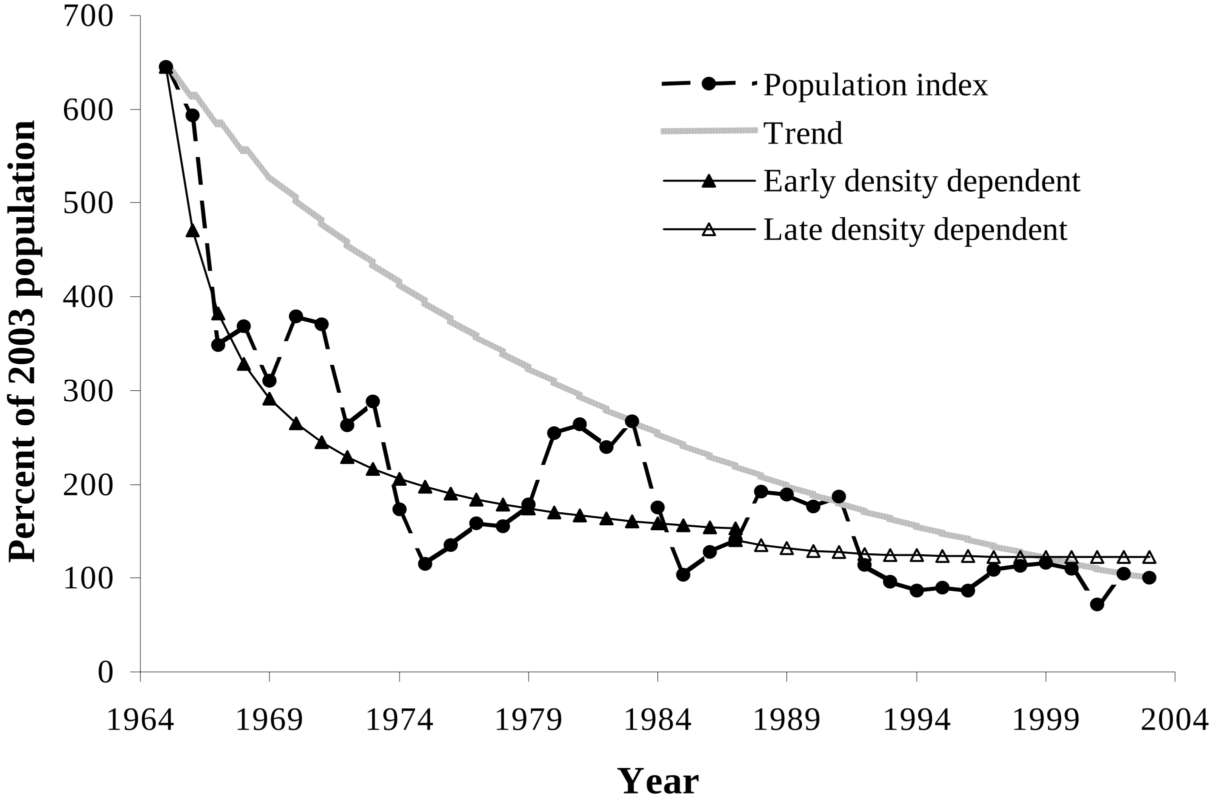


Fig. 6.33. Change in the population index for greater sage-grouse in Washington, 1965 -2003.



Wyoming

Monitoring effort. Wyoming has had a long-term extensive monitoring program for sage-grouse and has identified 1,454 leks in the state. We used 1965-2003 as our assessment period. The average number of leks censused per 5-year period increased by > 1000% from 1965 to 2003 and ranged from 80 to 945 over the assessment period (Table 6.15). The average number of active leks censused per 5-year period was similarly high, ranging from 58 to 672. Observers in Wyoming generally followed established procedures for lek census and attempted to count leks $\geq$3 times over an approximately 5-week period from late March to early May.

Population Changes. The proportion of active leks remained relatively stable over the assessment period, ranging from 63% to 78% from 1965 to 2003 (Table 6.15). Population trends indicated by average and median males per lek decreased over the assessment period by 49% and 48%, respectively (Table 6.15). Average and median males per active lek also declined by 48% and 43%, respectively, over the assessment period (Table 6.15). Monitoring data (males/lek) indicated that lek size significantly decreased (r^2 =0.49, P = 0.00) from 1965 to 2003 (Fig. 6.34).

There appeared to be an overall decline in lek size. Beginning in the early 1980s, the proportion of small leks increased. There also appeared to be an decrease in large leks from the late 1960s to the late 1990s (Fig. 6.35). However, the proportion of medium leks remained relatively stable, varying between about 30% and 40% over the assessment period (Fig. 6.35).

Annual rates of change suggest a long-term decline for sage-grouse in Wyoming (Fig. 6.36) and support the trend information obtained from lek attendance (males/lek) and lek class size. Sage-grouse populations declined at an overall rate of 5.22% per year from 1968 to 2003. Our analysis indicated a high likelihood of density dependence for the overall assessment period (likelihood = 0.99) and for both the early (likelihood = 0.93) and late (likelihood = 0.84) periods. From 1968-86, the population declined at an average rate of 9.66% and fluctuated around a level that was approximately 19% below the 2003 population. From 1987 to 2003, the population fluctuated around a level that was approximately 2% below the 2003 population and had an average change of 0.33% per year. Lows were reached in the mid-1990s and there has been some gradual increase in numbers since that time.

Summary. Connelly and Braun (1997) indicated that sage-grouse breeding populations had declined by 17% when they compared the long-term average of males/lek to the average obtained from 1985-94 data. A similar analysis suggested that production declined by 33% (Connelly and Braun 1997). Our analysis suggests that the decline was considerably greater than that reported by Connelly and Braun (1997). However, this may be due to our use of a larger and more complete data set as well as the addition of 9 more years of data. Our analysis generally supports the previous finding of a declining sage-grouse population. Additionally, lek surveys and counts were recently completed in an area that was intensively studied in the late 1940s and early 1950s (Patterson 1952). In 1949, 3,118 males were counted on 42 leks (74 males/lek). In 2003, 40 of the 42 leks were unoccupied but seven other leks were documented that were either not present in 1949 or not known to Patterson (1952). Thus, there were 9 active leks in this area in 2003 (a 79% decline) and 318 males (35 males/lek) were counted on these leks, indicating a 90% decline in the population (Wyoming Department of Game and Fish, personal correspondence, 2004).

Table 6.15. Sage-grouse monitoring and population trends in Wyoming, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	945	674	568	482	355	144	89	80
Number of active leks[1]	672	428	384	360	264	96	69	58
Percent active leks	71	63	68	75	75	66	78	73
Average males/lek	19	13	15	17	21	23	24	37
Median males/lek	11	6	8	12	14	12	16	21
Average males/active lek	26	20	22	23	29	35	31	50
Median males/active lek	20	14	16	18	21	25	23	35

[1] Averaged over each year for each period.

Fig. 6.34. Change in lek size for sage-grouse in Wyoming, 1965 - 2003.

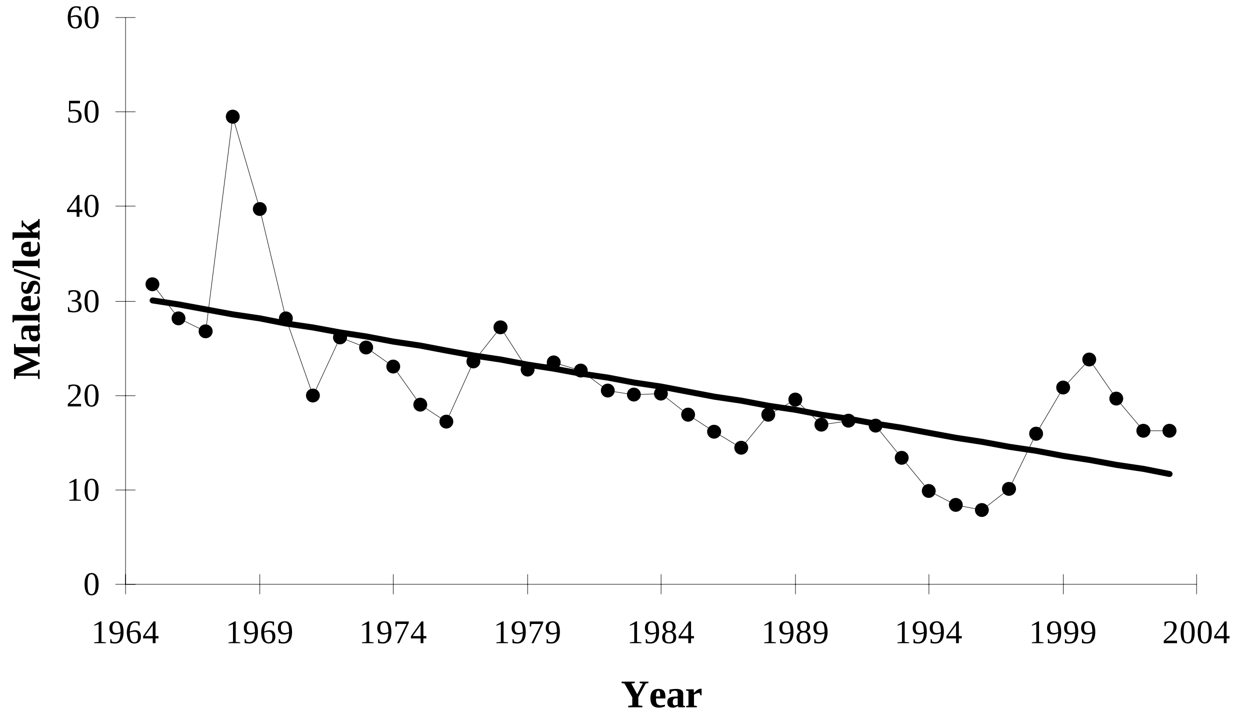


Fig. 6.35. Change in lek size class for Wyoming, summarized over 5-year periods, 1965 – 2003.

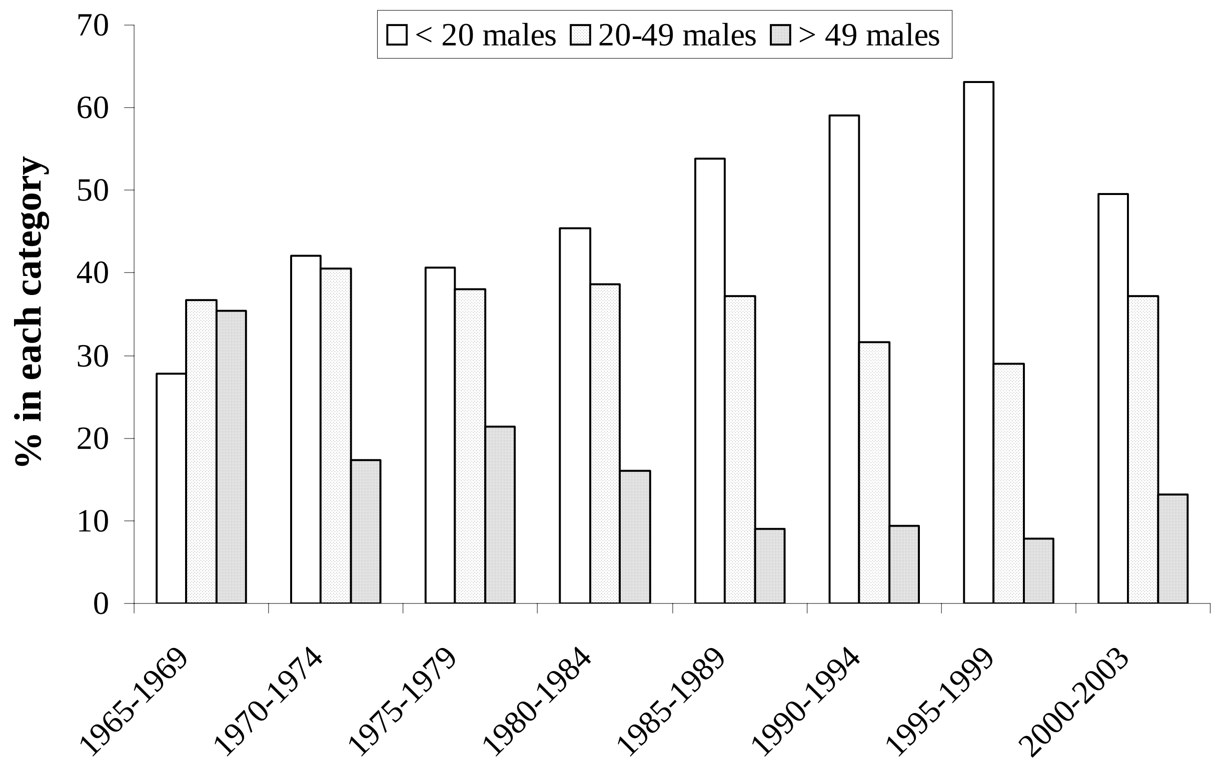
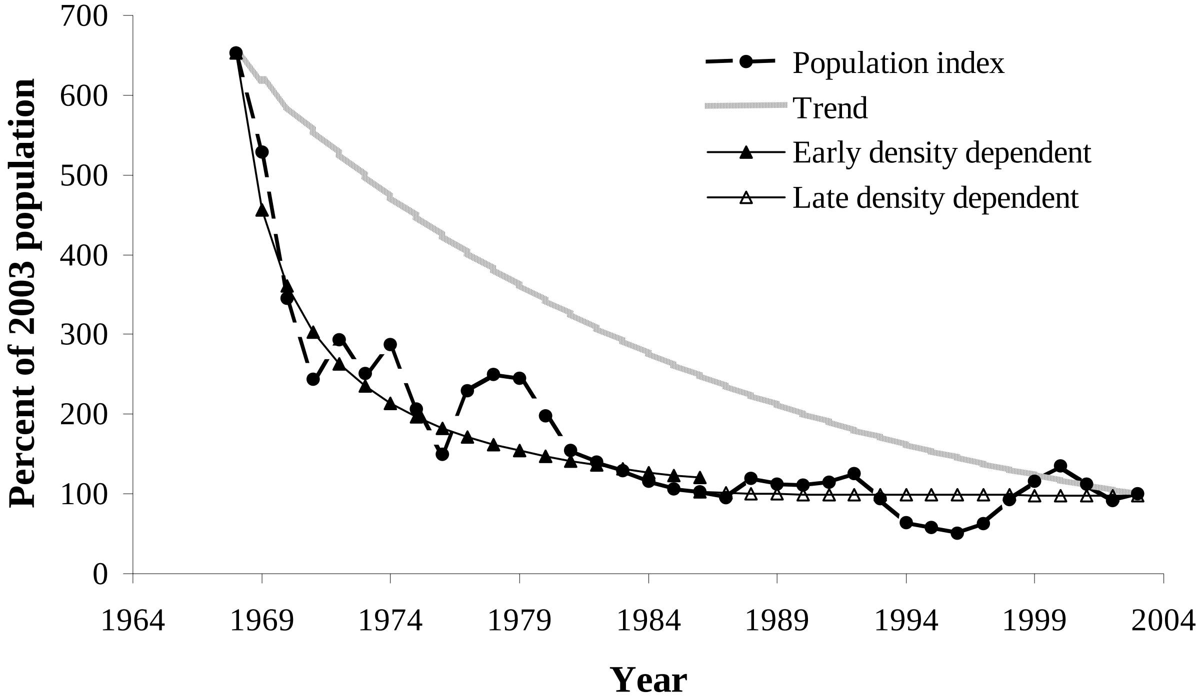


Fig. 6.36. Change in the population index for greater sage-grouse in Wyoming, 1968-2003.



Range-Wide

Populations. Previous sections described sage-grouse populations within each state and province supporting this species. However, many populations overlap state boundaries (Schroeder et al. 2004) and may be subject to different environmental and management variables. Thus, we also provide an overall review of sage-grouse population data and, where possible, examine population changes without regard to political boundaries.

We based our analysis on 41 relatively discrete populations and 24 subpopulations of greater sage-grouse in western North America (Table 6.16). Many of these appear to be spatially isolated while narrow corridors connect other populations (Fig. 6.37). The most isolated populations occur in parts of Colorado, Utah, Nevada, California, and Washington.

Table 6.16. General description and justification for delineation of greater sage-grouse breeding populations and subpopulations in North America (also see Figure 6.37).

Population Subpopulation	Separation from adjacent populations[a]	Fragmentation within population[b]	Brief description of population and justification for its delineation
Baker OR	~30 km	N.A.	Small population in Baker County OR. It appears separated by cropland from the nearest population, E-Central OR.
Bannack MT	~30–50 km and Continental Divide	N.A.	Small population E of Lemhi Pass near Bannack MT. It appears separated from 4 adjacent populations by distance, narrow corridors, and the continental divide.
Belt Mountains MT	~70 km along narrow corridor	~20-30 km	Small population or populations near Belt Mountains MT. In addition to being separated from the adjacent Central MT population, it also appears characterized by internal fragmentation.
Central OR	~30 km	~20-30 km	Population in central OR is separated by distance and topography from Lake Area OR/NE CA/NW NV and E-Central OR populations. Fragmentation within population is substantial.
Eagle/S Routt CO	~20-30 km and mountains	Population linear	Small population along Colorado River in Eagle and S Routt counties CO. It appears isolated from 3 adjacent populations by both distance and topography.
E Tavaputs Plateau UT	~50 km	N.A.	Small population on E Tavaputs Plateau UT. It appears separated from adjacent populations by > 50 km.
E-Central ID	~30-50 km	~10-20 km on periphery	Population E of Snake River in E-central ID. Population appears isolated by distance, topography, and habitat.
Garfield CO	~40 km	N.A.	Small population in Garfield County CO appears isolated from both Eagle/S Routt CO and Piceance CO populations.
Great Basin Core	~20-60 km and topography	~10-30 km	Large population in NV, SE OR, NE CA, SW ID, and NW UT. Natural fragmentation within population is common. Seven subpopulations have been delineated.
Central NV	N.A.	~20-30 km throughout	Large population in central NV is loosely connected with SE NV/SW UT and NE NV/S-Central ID/NW UT. Fragmentation within population and among adjacent populations is substantial.
E-Central OR	~10-30 km	~10-20 km	Population in E-central OR is loosely connected with Lake Area OR/NE CA/NW NV and N-Central NV/SE OR/SW ID populations. Some internal fragmentation is also apparent.
Lake Area OR/NE CA/NW NV	~20-50 km	~20-30 km	Large population in NE CA, NW NV, and S-central OR. Population appears loosely connected with S-Central OR/N-Central NV and E-Central OR. Some peripheral areas in NE CA may be partially fragmented.
N-Central NV/SE OR/SW ID	~10-20 km	N.A.	Loosely connected with NE NV/S-Central ID/NW UT, E-Central OR, and S-Central OR/N-Central NV populations.
NE NV/S-Central ID/NW UT	~10-20 km	~10-20	Large population in NE NV, S-central ID, and NW UT. Population appears loosely connected with N-Central NV/SE OR/SW ID and Central NV populations.
S-Central OR/N-Central NV	~20-30 km	~10-20 km	Population straddling the border of NV and OR. Appears loosely connected with 4 adjacent populations.
SE NV/SW UT	N.A.	~10-20 km	Large naturally fragmented population in SE NV that appears loosely interconnected with Central NV population.
Gunnison Range UT	~200 km	N.A.	Small translocated population of greater sage-grouse in SE UT within population of Gunnison sage-grouse. It is also isolated from nearest Gunnison sage-grouse populations by > 70 km.
Jackson Hole WY	~50 km	~20-30 km	Small isolated population in Jackson Hole WY area. Population also appears internally fragmented.
Klamath OR/CA	~50 km	~20-30 km	Small population on E side of Klamath Basin OR and CA. Population also appears internally fragmented.
Laramie WY	~30 km and mountains	N.A.	Small population SW of Laramie WY. Appears isolated by both distance and topography from adjacent populations.

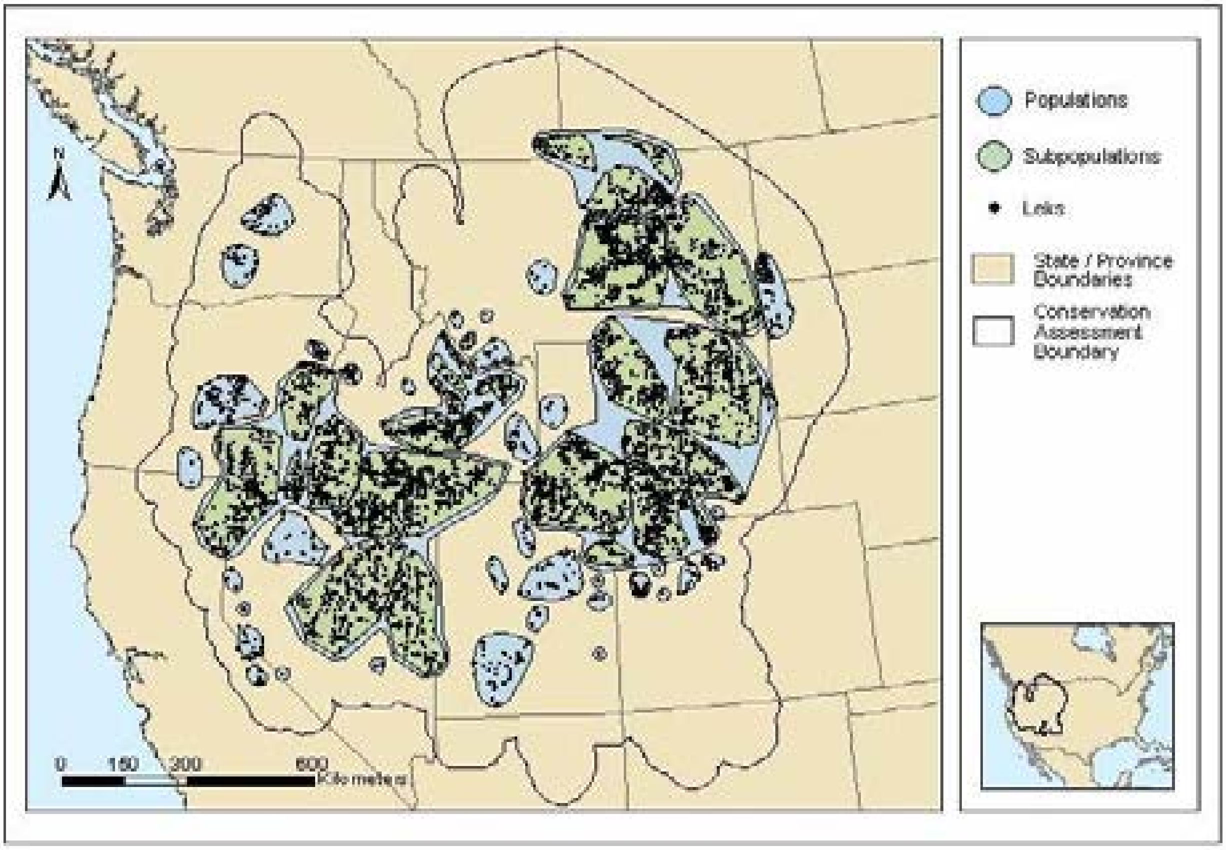
Middle Park CO	~20-30 km and mountains	~20 km	Population in Middle Park CO appears isolated from North Park CO and Garfield CO populations.
Moses Coulee WA	~50 km and Columbia River	~10-20 km	Population along Moses Coulee in N-central WA is isolated by distance and topography from Yakima WA population. Peripheral parts of population are extirpated.
MT/ND/NW SD	~30-40 km	~10-20 km	Population centered in SW ND and NW SD is largely isolated by distance and habitat from E-Interior MT/NE Tip WY population. Internal fragmentation is also apparent.
N Mono Lake CA/NV	~20-40 km and mountains	~10-20 km	Population on N side of Mono Lake area in CA and NV is relatively isolated from adjacent populations by both distance and topography. There is some natural internal fragmentation also.
NE-Interior UT	~30-50 km	~10-30	Population in NW-interior portion of Utah appears isolated by both distance and topography from adjacent populations. Natural fragmentation within population is also a factor.
Northern Montana	~20 km and Missouri River	~20-40 km	Large population N of Missouri River in N-central MT, SE AB, and SW SK. Divided into 3 subpopulations.
AB/SW SK/MT	~20 km along a narrow corridor	N.A.	Population in SE AB, SW SK and N edge of MT. It is separated from other birds in Saskatchewan by ~50 km
N-Central MT	~20 km	~20-30 km	Large population N of Missouri River in N-central MT. Loosely connected with populations to N, but separated from populations to S by Missouri River.
S-Central SK/MT	~20-40 km	~20-30 km	Population straddling the border of SK and MT. Population appears to be fragmented within by distance and habitat.
NW-Interior NV	~20-30 km	~20-30 km	Topographically dispersed population in interior NV. It appears largely isolated from 5 adjacent populations.
Piceance CO	~30-40 km	N.A.	Small population in the Piceance Basin CO. Adjacent populations appear isolated by both distance and topography.
Pine Nut NV	~50-60 km and valleys	N.A.	Small population in Pine Nut Mountains NV. Appears relatively isolated from adjacent populations by both distance and topography.
Quinn Canyon Range NV	~50-80 km and valleys	N.A.	Small population in the Quinn Canyon Range NV. Appears isolated from adjacent populations by both distance and topography.
Red Rock MT	~20-40 km and mountains	~10-20 km	Small, naturally fragmented population in SW Montana on N side of Monida Pass. Population appears isolated by distance and topography from adjacent populations.
S Mono Lake CA	~20-50 km and mountains	~10-20 km	Small population on S side of Mono Lake area in CA appears relatively isolated from adjacent populations by both distance and topography. There is some natural internal fragmentation also.
S White River UT	~40-50 km	N.A.	Small population S of White River UT. It is separated from adjacent populations by > 40 km.
Sanpete/Emery UT	~50-60 km	~20 km	Small population in central UT that is isolated by both distance and topography.
Sawtooth ID	~70-80 km	~10-20 km	Small isolated population near Stanley, ID in Sawtooth Mountains.
S-Central UT	~50-70 km and mountains	~20-40 km	Clearly isolated population in S-central UT. Population appears to have a high degree of natural fragmentation within it.
Snake, Salmon, and Beaverhead	~20-40 km	~10-30 km	Large population along upper Snake, Salmon, and Beaverhead watersheds. Six subpopulations appear loosely connected through mountain valleys and passes.
Big Lost ID	~10 km along narrow corridors	Population linear	Population follows Big Lost and Willow Creek Valleys in ID. Connected with Little Lost ID and N-Side Snake ID populations along narrow corridors.
Lemhi-Birch ID	~20 km and topography	Population linear	Population along Lemhi and Birch Creek Valleys ID. Appears largely isolated by both distance and topography.
Little Lost ID	~20 km and narrow corridors	Population linear	Population along Little Lost and Pahsimeroi valleys ID. Population may be loosely connected with Big Lost ID and Lemhi-Birch ID populations along narrow corridors.
N Side Snake ID	~10-30 km	~10 km	Large population on N side of Snake River ID. Loosely connect along narrow corridor with Big Lost ID population.

Upper Snake ID	~20-40 km and mountains	N.A.	Population along upper Snake River ID. Population appears mostly isolated from adjacent populations.
Summit/Morgan UT	~20-40 km and mountains	N.A.	Small population in NE UT appears separated from SW WY/NW CO/NE UT/SE ID by both distance and topography.
Tooele/Juab UT	~40 km	~10-20 km	Small isolated population in central UT. Population also appears naturally fragmented.
Twin Bridges MT	~60 km	N.A.	Small isolated population in SW MT.
Warm Springs Valley NV	~30-60 km and valleys	~10-20 km	Small, fragmented, and isolated population along the W edge of NV.
Weiser ID	~20 km	N.A.	Small and mostly isolated population in Weiser area ID.
White Mountains NV/CA	~50 km and topography	N.A.	Small and isolated population in White Mountains straddling border of CA and NV.
White River CO	~30-40 km and mountains	N.A.	Small isolated population along White River CO.
Wisdom MT	~4-60 km	N.A.	Small isolated population in SW MT.
Wyoming Basin	~20-30 km and topography	~10-20 km	Massive population centered in WY. Seven subpopulations have been delineated.
Dinosaur UT/CO	~10-20 km along narrow corridors	~10-20 km	Population largely between Yampa and White rivers along UT-CO border. Appears mostly separated from 6 adjacent populations by both distance and topography.
Fall River SD/E-Edge WY	~10-20 km	10-20 km	Small population in SW SD and E-edge WY may be loosely connected along narrow corridor to E-Interior MT/NE Tip WY population. Fragmentation within population is apparent.
NE WY/SE MT	~10-20 km	~10-20	Large population primarily in NE WY. Population appears continuous within and loosely connected with adjacent populations.
North Park CO/WY	~10 km along narrow corridor	N.A.	Population in North Park CO appears connected with S-Central WY/N-Central CO population along narrow North Platte River corridor. Other adjacent populations separated by topography.
S-Central MT/N-Central WY	~10-40 km	~20-30 km	Large population primarily in N-central WY. Appears isolated from Central MT population, but loosely interconnected with NE WY/SE MT and S-Central WY/N-Central CO populations. There may be isolated pockets within the overall population.
S-Central WY/N-Central CO	N.A.	~10-20 km	Large population primarily in S-central WY. It appears loosely connected with all adjacent populations.
SW WY/NW CO/NE UT/SE ID	N.A.	N.A.	Large population primarily in SW WY. Population appears loosely connected with S-Central WY/N-Central CO and Dinosaur UT/CO.
Yakima WA	~50 km and Columbia River	N.A.	Population near Yakima in S-central WA is isolated by distance and topography from Moses Coulee WA population. Peripheral parts of population are extirpated.
Yellowstone watershed	~20-30 km	~20-30 km	Large population in central and SE MT. Mostly separated from adjacent populations by distance and topography.
Central MT	N.A.	~20-30 km on periphery	Large population in central MT is loosely connected with E-Interior MT/NE Tip WY population. Population is separated from N-Central MT population by ~20-30 km and Missouri River.
E-Interior MT/NE Tip WY	N.A.	~20-30 km on periphery	Large population primarily in SE Montana. It appears loosely connected with Central MT population, but separated from N-Central MT, MT/ND/NW SD, and NE WY/SE MT populations by distances of 20-30 km.

[a]N.A. (not apparent) was used to describe several populations where separation and/or isolation from adjacent populations was not apparent. These populations were delineated on the basis of their large size and broader differences associated with region.

[b]N.A. (not apparent) was used to describe populations where fragmentation within the population was not apparent.

Fig. 6.37. Discrete populations and subpopulations of sage-grouse in western North America.



Of the 7 floristic regions that support sage-grouse, the Snake River Plain and southern Great Basin regions contain the greatest number of populations, 16 and 13, respectively (Table 6.17). The Columbia Basin only has 2 populations.

Table 6.17. Categorization of greater sage-grouse populations and subpopulations by floristic regions in North America (partly following Miller and Eddleman 2001).

Great Plains	Wyoming Basin	Snake River Plain	Columbia Basin	Northern Great Basin	Southern Great Basin	Colorado Plateau
MT/ND/NW SD	Belt Mountains MT	Baker OR	Moses Coulee WA	Central OR	Summit/Morgan UT	E Tavaputs Plateau UT
AB/SW SK/MT	Eagle/S Routt CO	Bannack MT	Yakima WA	Lake Area OR/NE CA/NW NV	Quinn Canyon Range NV	Garfield CO
Fall River SD/E-Edge WY	SW WY/NW CO/NE UT/SE ID	E-Central ID		S-Central OR/N-Central NV	N Mono Lake CA/NV	Gunnison Range UT
S-Central SK/MT	S-Central MT/N-Central WY	NE NV/S-Central ID/NW UT		Klamath OR/CA	Sanpete/Emery UT	White River CO
E-Interior MT/NE Tip WY	S-Central WY/N-Central CO	N-Central NV/SE OR/SW ID		Warm Springs Valley NV	White Mountains NV/CA	S White River UT
NE WY/SE MT	Jackson Hole WY	E-Central OR			Central NV	Piceance CO
N-Central MT	North Park CO/WY	Red Rock MT			SE NV/SW UT	
Central MT	Laramie WY	Sawtooth ID			S Mono Lake CA	
	Middle Park CO	Big Lost ID			NE-Interior UT	
	Dinosaur UT/CO	Lemhi-Birch ID			S-Central UT	
		Little Lost ID			Pine Nut NV	
		N-Side Snake ID			Tooele/Juab UT	
		Twin Bridges MT			NW-Interior NV	
		Upper Snake ID				
		Weiser ID				
		Wisdom MT				

Monitoring. Virtually all states and provinces have increased monitoring efforts, especially over the last 10 years (Table 6.18; also Appendix 3, Fig. A3.2). Our analysis indicated that a total of 2,637 leks are now censused annually. This is about 30% more than the number reported by the agencies in their response to the questionnaire (see previous section on Population Databases in this chapter). Range-wide, population monitoring efforts increased by 729% over the long-term. The largest increases in effort have occurred Montana and Wyoming, 2 of the key sage-grouse states. Montana had 2 periods when efforts increased substantially; 1965-69 to 1975-79 when the average number of leks monitored increased by >700% and 2000-03 when the average number of leks censused tripled over the number censused in 1995-99. Increasing census efforts may be related to concerns over sage-grouse numbers during the late 1960s and early to mid 1970s. During this period, data from most states indicated declining sage-grouse populations (see state accounts in this Chapter). However, this seems an unlikely explanation for increased censusing because these declines did not appear to be widely recognized at that time. From the 1950s through the 1980s, the Western States Sage and Columbian Sharp-tailed Grouse Technical Committee compiled unpublished qualitative reports from western states and provinces on the status of sage-grouse populations (Sage Grouse Questionnaire, Numbers I-XXIII, Western States Sage and Columbian Sharp-tailed Grouse Technical Committee, unpublished reports). Qualitative information obtained from these questionnaires for 1965 to 1979 does not show a clear recognition of population trends throughout western North America (Table 6.19). From 1965 through 1979, most states and provinces indicated populations were stable to increasing, in general disagreement with population data we obtained from these states and provinces. Moreover, New Mexico reported that sage-grouse populations in that state were stable or increasing from 1956 through 1975 even though Aldrich (1963) indicated the species had been extirpated from that state. Additionally, Reese and Connelly (1997) reported introductions of greater sage-grouse into the former range of Gunnison sage-grouse in New Mexico in the 1930s, 1940s, and 1960s, further suggesting that sage-grouse populations in New Mexico were likely declining for a relatively long period of time. Currently, neither species of sage-grouse occurs in New Mexico (Schroeder et al. 2004). Similarly, Montana reported stable sage-grouse populations for each year from 1968 through 1975, but lek size and population trend data indicated increasing and decreasing populations over this period (see Figures 6.12 and 6.14). Thus, it is more likely that increases in monitoring effort were the result of changes in program priorities or interests of individual agency biologists.

Although monitoring efforts have increased, there still appears to be a reluctance by some states/provinces to use established and accepted monitoring techniques (Jenni and Hartzler 1978, Emmons and Braun 1984, Connelly et al. 2003). Although data collected within these states or provinces may indicate population trends over time, these different methods confound attempts to make comparisons with other states.

Table 6.18. Changes in long-term monitoring efforts for states and provinces supporting sage-grouse populations.

State/Province	Leks Censused 1965-69	Leks Censused 2000-2003	% Change
Alberta	14 (starting 1975-1979)	29	107
California	9	45	400
Colorado	66	171	159
Idaho	74	319	331
Montana	13	546	4,100
Nevada	22	182	727
North Dakota	19	27	42
Oregon	18	153	750
Saskatchewan	13 (starting 1985-1989)	16	23
South Dakota	15 (starting 1990-1994)	16	7
Utah	37	144	289
Washington	3	47	1,467
Wyoming	80	945	1,081
Average			729

Table 6.19. Qualitative assessment of population trends by states and provinces for 5-year periods, 1965-69 to 1975-79.[1]

State/Province	1965-69	1970-74	1975-79	Overall
Alberta	I/S[2]	S/D	D	D
California	S	S	S	S
Colorado	S	S/D	I	S/I
Idaho	S/I	D	I	D
Montana	S	S	S	S
Nevada	I	S/D	S/I	S/I
North Dakota	S	S	D	S
Oregon	D	D	S	D
Saskatchewan	S/I	S	S	S
South Dakota	S	S	D	D
Utah	I	S	I	I
Washington	S	S/D	D	S
Wyoming	S/I	S/D	S	S
% Decreasing	8	15	31	31

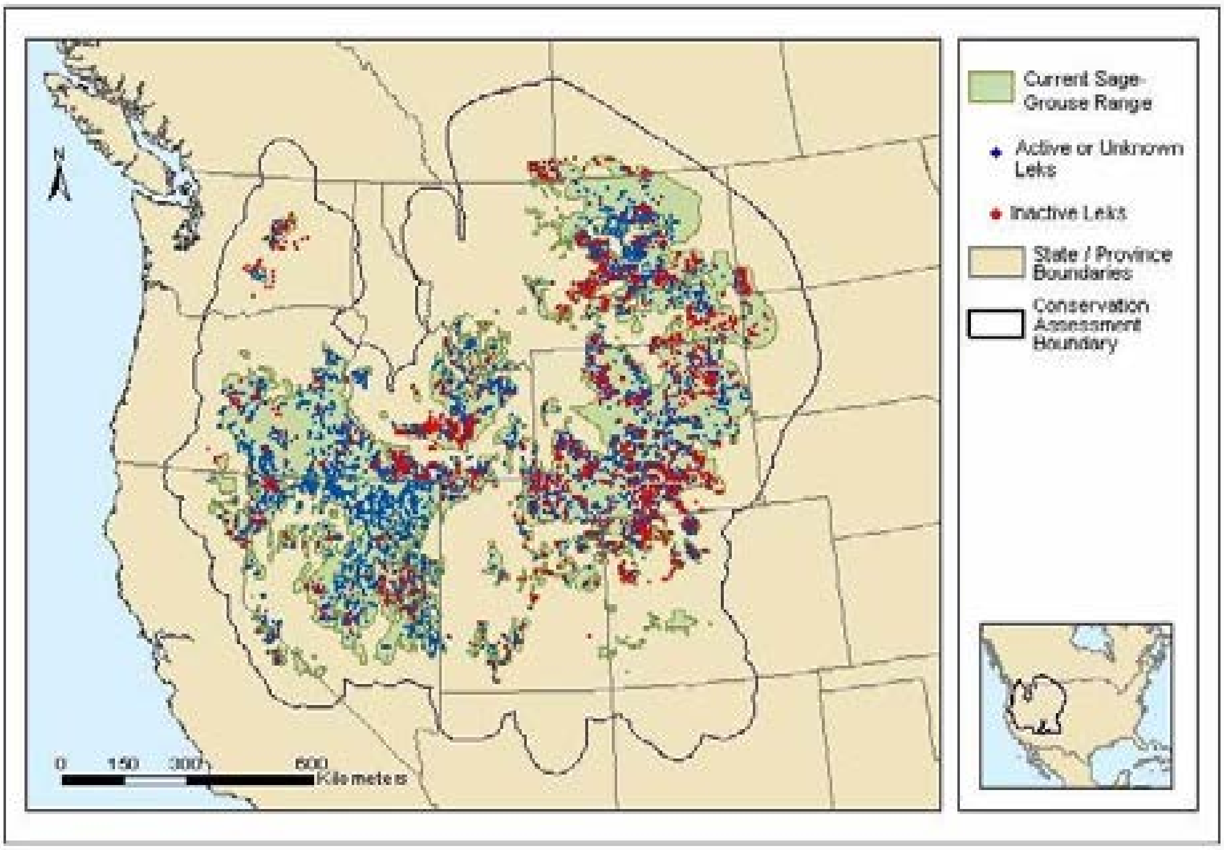
[1] Information obtained from Western States Sage and Columbian Sharp-tailed Grouse Technical Committee Sage Grouse Questionnaire, Numbers I-XXIII.
[2] D = decreasing; I = increasing; S = stable.

Lek Distribution and Numbers. Throughout western North America 5,585 sage-grouse leks have been identified since agencies first began routine monitoring of this species. The number of leks counted has increased dramatically from 1965 to 2003 and an average of 2,637 leks/year and 48,378 males were counted from 2000 to 2003 (Table 6.20). In 2003, a total of 50,566 male sage-grouse were counted on leks throughout western North America. Not all leks are currently active and many have been inactive for years. The total number of leks that has been extirpated is unknown and hinders our attempts to fully understand the magnitude of change in sage-grouse populations. Although leks have become inactive throughout the species' range, the distribution of inactive leks appears clustered rather than widespread (Fig. 6.40). Proportionally, the largest number of inactive leks appears to occur in Colorado, Utah and Washington.

Table 6.20. Average number of leks and male sage-grouse counted by 5-year period, 1965-2003.

Parameter	1965-69	1970-74	1975-79	1980-84	1985-89	1990-94	1995-99	2000-03
Leks	351	510	676	1,016	1,162	1,318	1,689	2,637
Active leks	299	434	565	828	828	961	1,182	1,925
Males counted	11,402	13,573	17,239	21,985	22,278	21,783	23,687	48,378

Fig. 6.40. Distribution of active and inactive sage-grouse leks in western North America, spring 2000. The range of the Gunnison sage-grouse in SW Colorado and SE Utah was not included in this analysis although the distribution is shown.



Population Status and Change. Generally, the proportion of small leks has increased over the assessment periods for most states and provinces while the proportion of large leks has decreased. Although some states have monitored sage-grouse populations since the 1940s, little is known about the protocols used or consistency of monitoring efforts during those early years. There was much more variation associated with estimates of lek size in the 1940s and 1950s and variation was noticeably reduced by the mid-1960s (Fig. 6.41). An examination of all trend data from the mid 1940s to 2003 suggests a substantial decline in the overall sage-grouse population in North America. However, because data collected in the 1940s and 1950s is highly variable (Fig. 6.41) and may have been collected in a somewhat haphazard fashion, there is no means of assessing the true magnitude of the population change.

Eleven of 13 (85%) states and provinces showed significant long-term declines in size of active leks (Table 6.21). Similarly, eight of 10 states (80%) showed population declines over that time frame (Table 6.21). Two of 10 (20%) appeared to be stable or slightly increasing. Only California had an increase in both the population index and lek size.

Fig. 6.41. Average active lek size (with 95% C.I.) for greater sage-grouse in North America by 5-year intervals, 1945-2003.

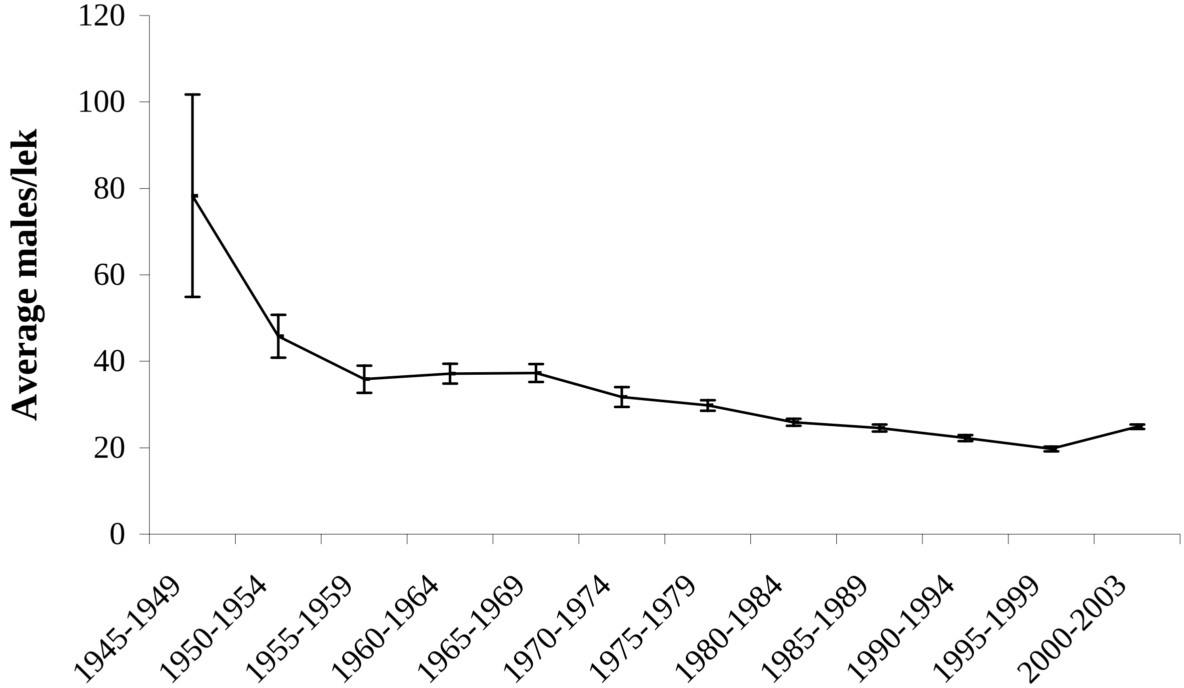


Table 6.21. Summary table for regression analysis of maximum counts on active leks and average instantaneous rates of population change by state or province. Data for states and provinces other than Alberta, Saskatchewan, and South Dakota are for 1965-2003.

State/province	Average lek size in relation to year					Average instantaneous rate of population change
	Intercept	Slope	r^2	**F**	P	
Alberta	923.21	-0.455	0.083	26.648	< 0.001	ID[1]
California	-183.29	0.108	0.001	0.885	0.347	+0.7
Colorado	841.69	-0.407	0.025	57.342	< 0.001	+1.0
Idaho	1037.07	-0.507	0.035	186.369	< 0.001	-1.5
Montana	330.51	-0.155	0.006	33.090	< 0.001	-1.6
Nevada	920.43	-0.451	0.039	86.553	< 0.001	-2.1
North Dakota	243.50	-0.116	0.013	8.596	0.004	-2.8
Oregon	223.61	-0.101	0.002	3.025	0.082	-3.5
Saskatchewan	2513.14	-1.252	0.135	24.406	< 0.001	ID[1]
South Dakota	558.68	-0.274	0.069	18.817	< 0.001	ID[1]
Utah	245.11	-0.111	0.002	6.396	0.012	-0.35
Washington	421.31	-0.201	0.012	6.404	0.012	-4.79
Wyoming	755.92	-0.367	0.017	194.033	< 0.001	-5.2

[1] Insufficient data to calculate population index.

Trends for distinct populations were similar to those of states. We had sufficient data to assess trends in maximum lek counts for 34 of the 41 (83%) identified populations (Table 6.22). Twenty-one of 22 significant correlations were negative (Table 6.22). Within the five largest populations, 16 of 17 subpopulations with significant correlations were also negative. We also examined the trend in maximum lek counts by region (Table 6.23). Lek counts in 5 of 7 (71%) floristic regions declined significantly over the long-term and two remained unchanged (also see appendix 6). Characteristics of sage-grouse populations and subpopulations are summarized in appendices 4 and 5.

Table 6.22. Summary table for regression analysis of maximum counts on active leks by populations and subpopulations between 1965-2003 (not all years available for each population or subpopulation). Significant slopes are in bold.

Population/subpopulation	Intercept	Slope	r^2	**F**	P
Baker OR	624.71	-0.300	0.008	0.520	0.473
Bannack MT	1019.95	**-0.502**	0.134	21.879	< 0.001
Belt Mountains MT	2087.17	**-1.036**	0.328	36.173	< 0.001
Central OR	415.01	**-0.200**	0.027	23.811	< 0.001
E Garfield CO	427.43	-0.214	0.107	0.120	0.788
Eagle/S Routt CO	908.65	**-0.449**	0.149	17.353	< 0.001
East Tavaputs Plateau UT	577.54	**-0.288**	0.145	4.250	0.050
E-Central ID	654.92	**-0.322**	0.103	14.365	< 0.001
Great Basin Core	627.10	**-0.303**	0.014	80.572	< 0.001
Central NV	536.42	**-0.257**	0.017	11.885	< 0.001
E-Central OR	677.28	**-0.329**	0.020	5.144	0.024
Lake Area OR/NE CA/NW NV	13.99	0.011	0.000	0.008	0.928
N-Central NV/SE OR/SW ID	1154.59	**-0.568**	0.057	32.042	< 0.001
NE NV/S-Central ID/NW UT	960.11	**-0.471**	0.041	99.280	< 0.001
S-Central OR/N-Central NV	3706.12	**-1.852**	0.264	7.877	0.010
SE NV/SW UT	553.73	**-0.270**	0.021	21.453	< 0.001
Gunnison Range UT	1199.43	**-0.599**	0.361	6.201	0.030
Jackson Hole WY	2119.60	**-1.052**	0.061	5.845	0.018
Klamath OR/CA	-609.59	**0.310**	0.303	6.963	0.018
Laramie WY	3270.96	**-1.643**	0.375	16.782	< 0.001
Middle Park CO	443.86	**-0.214**	0.024	6.183	0.014
Moses Coulee WA	414.93	**-0.198**	0.013	5.384	0.021
MT/ND/NW SD	170.86	**-0.079**	0.006	5.469	0.020
N Mono Lake CA/NV	1083.41	**-0.531**	0.053	15.571	< 0.001
NE Interior UT	670.867	**-0.328**	0.023	8.328	0.004
Northern Montana	-4.98	0.014	0.000	0.064	0.800
AB/SK/MT	1041.34	**-0.515**	0.108	43.980	< 0.001
N-Central MT	-652.42	**0.340**	0.024	14.342	< 0.001

S-Central SK/MT	2631.30	**-1.310**	0.171	26.616	< 0.001
NW-Interior NV	-677.96	0.346	0.012	0.684	0.411
Piceance CO	51.65	-0.022	0.001	0.067	0.796
Pine Nut NV[a]	-	-	-	-	-
Quinn Canyon Range NV[a]	-	-	-	-	-
Red Rock MT	4514.27	**-2.248**	0.785	262.576	< 0.001
S Mono Lake CA	-616.60	0.324	0.011	2.845	0.093
S White River UT	90.26	-0.034	0.000	0.006	0.938
Sanpete/Emery UT	-178.57	0.094	0.027	1.160	0.288
Sawtooth ID	1092.00	-0.547	0.170	2.654	0.127
S-Central UT	-9.29	0.019	0.000	0.041	0.839
Snake, Salmon, and Beaverhead	854.10	**-0.413**	0.019	63.454	< 0.001
N Side Snake ID	662.37	**-0.317**	0.012	20.335	< 0.001
Big Lost ID	2065.49	**-1.025**	0.251	39.482	< 0.001
Upper Snake ID	1092.22	**-0.529**	0.026	24.782	< 0.001
Little Lost ID	192.00	-0.079	0.001	0.117	0.733
Lemhi-Birch ID	414.51	-0.197	0.011	3.198	0.075
Summit/Morgan UT	510.44	-0.246	0.016	2.517	0.115
Tooele/Juab UT	-121.92	0.072	0.002	0.178	0.674
Twin Bridges MT[a]	-	-	-	-	-
Warm Springs Valley NV[a]	-	-	-	-	-
Weiser ID	-428.86	0.228	0.012	1.324	0.252
White Mountains NV/CA[a]	-	-	-	-	-
White River CO[a]	-	-	-	-	-
Wisdom MT	17374.00	**-8.667**	0.354	5.488	0.041
Wyoming Basin	861.20	**-0.419**	0.023	332.364	< 0.001
Fall River SD/E Edge WY	533.35	-0.260	0.025	1.137	0.292
North Park CO	825.93	**-0.396**	0.015	12.544	< 0.001
S-Central MT/N-Central WY	253.50	**-0.118**	0.003	7.083	0.008
S-Central WY/N-Central CO	1185.87	**-0.581**	0.043	258.257	< 0.001
NE WY/SE MT	855.42	**-0.420**	0.042	86.352	< 0.001
SW WY/NW CO/NE UT/SE ID	276.80	**-0.124**	0.002	4.469	0.035
Dinosaur UT/CO	313.304	-0.142	0.002	0.787	0.376
Yakima WA	899.17	**-0.439**	0.031	4.196	0.043
Yellowstone watershed	322.58	**-0.151**	0.007	24.254	< 0.001
Central MT	284.72	**-0.131**	0.005	13.920	< 0.001
E Interior MT/NE tip WY	119.97	-0.053	0.002	1.092	0.297

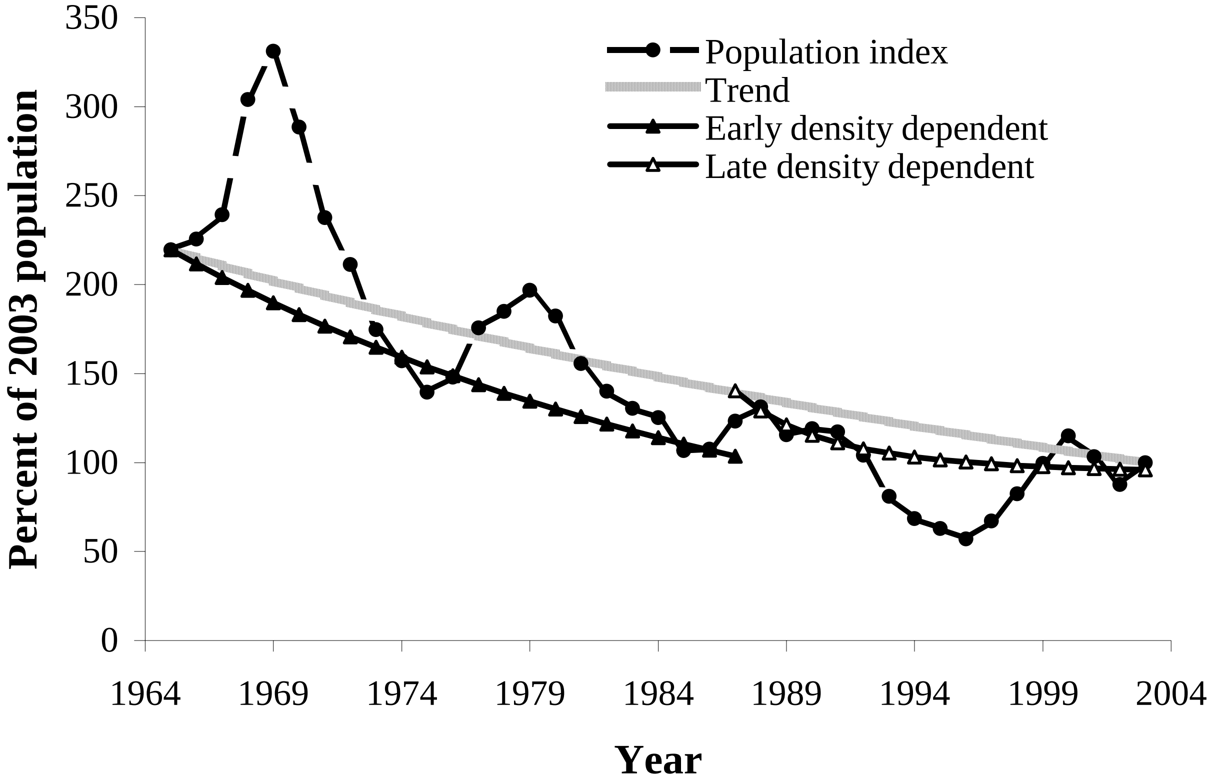
[a]Insufficient data for analysis.

Table 6.23. Summary table for regression analysis of maximum counts for active leks between 1965 and 2003 by floristic region. Significant slopes are in bold.

Floristic Region	Intercept	Slope	r^2	**F**	P
Great Plains	284.68	**-0.133**	0.006	43.174	< 0.001
Wyoming Basin	823.28	**-0.400**	0.021	267.520	< 0.001
Snake River Plain	1042.85	**-0.510**	0.038	275.509	< 0.001
Columbia Basin	421.31	**-0.201**	0.012	6.404	0.018
Northern Great Basin	35.62	-0.004	0.000	0.004	0.950
Southern Great Basin	509.30	**-0.245**	0.013	46.438	< 0.001
Colorado Plateau	-239.63	0.126	0.014	1.904	0.170

Annual rates of change suggest a long-term decline for sage-grouse in western North America (Fig. 6.42) and support the trend information obtained from lek attendance (males/lek, Tables 6.21, 6.22, 6.23). Sage-grouse populations declined at an overall rate of 2.0% per year from 1965 to 2003. From 1965-85, the population declined at an average rate of 3.5%. From 1986 to 2003, the population declined at a lower rate of 0.37% and fluctuated around a level that was 5% lower than the 2003 population. Our analysis indicated a reasonably high likelihood of density dependence for the overall assessment period (likelihood = 0.63) and late period (likelihood = 0.76). However evidence for density dependence for the early period may have been compromised by major reductions in quality and quantity of sagebrush habitats. The density independent model indicated a 68% probability of persistence given trends over the entire assessment period. Because of relatively large declines in sage-grouse populations from 1965 to 1985, the density independent model indicated that over this period the probability of persistence was 25%. However, sage-grouse populations stabilized and some increased from 1986 to 2003, and the density independent model indicated that over this period the probability of persistence increased to 97%, but this was largely driven by population increases in the mid to late 1990s. Continued loss and degradation of habitat and other factors (including West Nile Virus) do not provide causes for optimism. Populations in the late 1960s and early 1970s were approximately 2-3 times higher than current populations (Fig. 6.42).

Fig. 6.42. Range-wide change in the population index for greater sage-grouse in North America, 1965-2003.



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Chapter 7

Sagebrush Ecosystems: Current Status and Trends

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Abstract. The sagebrush (*Artemisia* spp.) biome has changed since settlement by Europeans. The current distribution, composition and dynamics, and disturbance regimes of sagebrush ecosystems have been altered by interactions among disturbance, land use, and invasion of exotic plants. In this chapter, we present the dominant factors that have influenced habitats across the sagebrush biome. Using a large-scale analysis, we identified regional changes and patterns in "natural disturbance", invasive exotic species, and influences of land use in sagebrush systems. Number of fires and total area burned has increased since 1980 across much of the sagebrush biome. Juniper (*Juniperus* spp.) and pinyon (*Pinus* spp.) woodlands have expanded into sagebrush habitats at higher elevations. Cheatgrass (*Bromus tectorum*), an exotic annual grass, has invaded much of lower elevation, more xeric sagebrush landscapes across the western portion of the biome. Consequently, synergistic feedbacks between habitats and disturbance (natural and human-caused) have altered disturbance regimes, plant community dynamics and contributed to loss of sagebrush habitats and change in plant communities. Habitat conversion to agriculture has occurred in the highly productive regions of the sagebrush biome and influenced up to 56% of the Conservation Assessment area. Similarly, urban areas, and road, railroad, and powerline networks fragment habitats, facilitate predator movements, and provide corridors for spread of exotic species across the entire sagebrush biome. Livestock grazing has altered sagebrush habitats; the effects of overgrazing combined with drought on plant communities in the late 1880s and early 1900s still influences current habitats. Management of livestock grazing has influenced sagebrush ecosystems by habitat treatments to increase forage and reduce sagebrush and other plant species unpalatable to livestock. Fences, roads, and water developments to manage livestock movements have further influenced the landscape and increased access into sagebrush habitats. Energy development also influenced sagebrush landscapes by construction of wells, access roads, and pipelines. Treatments to restore sagebrush are becoming a major emphasis of land management agencies. However, revegetation and rehabilitation treatments are limited by the financial, biological, and technological resources needed to restore sagebrush landscapes that function at the spatial and temporal scales used by sage-grouse (*Centrocercus urophasianus*).

STATUS AND TRENDS OF SAGEBRUSH ECOSYSTEMS

Objectives and Approach

The sagebrush biome has been subjected to a broad variety of land uses over the past century (West and Young 2000, Crawford et al. 2004) and very little, if any, remains in a state similar to that encountered by European settlers (West 1999). Interactions among land use, disturbance and vegetation response, and climate have altered patterns and processes within sagebrush communities or caused extensive loss of sagebrush habitats from some regions. Despite these changes, sagebrush habitats in some form still occupy almost 500,000 km^2 distributed across 13 states and 3 provinces. Our review of issues included the entire distribution of sagebrush and was not restricted only to those regions currently or historically occupied by greater sage-grouse.

Our objective was to identify the dominant factors that influenced sagebrush ecosystems across the biome. We presented the issues and factors as separate entities but emphasize the cumulative effects of these stressors on the sagebrush biome and their combined influence on the trajectory of these systems. The broader context of cumulative effects is presented in the analysis of the “human footprint” on sagebrush ecosystems (Chapter 12) and the synthesis chapter (Chapter 13).

We used a top-down landscape approach in our assessment based on an assumption that impacts at local scales can be aggregated into a broader perspective (Allen and Starr 1982, Wiens 1989, Peterson and Parker 1998). Our objective in using this coarse-scale approach was to determine the primary disturbances and resulting patterns that were expressed over the regional landscapes used by greater sage-grouse. Identification of dominant coarse-scale patterns, such as the relative fragmentation of sagebrush habitats, then could be placed into context across the sagebrush biome and form the basis for large-scale management actions. To do this, we conducted extensive analyses on spatial coverages in a Geographic Information System (GIS). Because these coverages describe the spatial arrangement of habitats in the landscape, we then used the patterns to guide our assessment of the underlying processes (Turner 1989, Urban et al. 1987). This approach is appropriate for acute disturbances, such as fire, that have distinct impacts in space and time and result in altered landscape patterns (Pickett and White 1985, Turner and Bratton 1987). However, press forms of disturbance (Bender et al. 1984), such as livestock grazing, which have a diffuse effect over large areas, may be more difficult or not possible to quantify.

We have structured each major section to provide a brief background or historical setting for each factor. We then presented a synopsis of the potential influence of the disturbance or land use on the processes within the system. Finally, we mapped the distribution of the disturbance and quantified (when possible) the potential area altered or influenced by the disturbance or land use. We also attempted to provide temporal information on changes in frequency, intensity, or location of the effect.

We addressed “natural” disturbance and change factors, invasive species and control, and human-related land use influences on sagebrush ecosystems. Although we grouped factors into these general categories, we also considered that the interaction among elements was a significant component in the influence on sagebrush systems.

Methods

Data Sources

We derived the base data used in the habitat portion of the Conservation Assessment from three primary sources of existing data. Management directions and objectives were obtained from published memos and literature, or from online sources. Tabular data and summaries (nonspatial data) were created from online sources, microfilm archives, or published literature. Spatial data for the United States were contributed from state, federal, university, and nongovernmental sources. We documented the dataset and source; the information will be made available on the SAGEMAP website (http://sagemap.wr.usgs.gov) (U.S. Geological Survey 2001) with the exception of certain proprietary or security-sensitive information.

Spatial Data

We attempted to depict as many environmental features as possible in georeferenced maps. These spatial analyses were critical to understanding the context in which the disturbance or habitat feature was assessed. For example, 1,258 km of fences were constructed in year 2000 on lands managed by the U.S. Bureau of Land Management (U.S. Bureau of Land Management, Public Lands Statistics 2001). Fences control livestock, modify habitats, and influence movement of predators, invasive plants, or vehicular travel. In the absence of spatial information, such as linear distance/unit area, and other regional characteristics, our ability to quantify the area over which fences might influence habitats and increase mortality rates is limited. For example, fences can increase mortality directly when sage-grouse fly into fences or indirectly because of increased predation by raptors. Similarly, statistics for number of livestock registered within a county or allotment database provide no information on location or season of grazing. Consequently, we could not develop meaningful correlations with habitat information from livestock statistics obtained from these sources.

We converted linear features to a unit length per unit area to map the relative density of the feature to create a more meaningful measure of the distribution across the landscape. We also converted high densities of point data to a contoured distribution for mapping the general patterns. Specific methods used for individual coverages are given in figure legends, in the accompanying text, or will be included in the metadata record.

We used an ecological rationale when possible to determine the distances around points or lines from which a source had an influence of habitats or sage-grouse. For example, the majority of nesting and early brood-rearing habitat for sage-grouse was within 5 km of lek sites for nonmigratory populations and <18 km for migratory populations (Connelly et al. 2000*a*). Similarly, common ravens (*Corvus corax*) and American crows (*C. americanus*), which are nest predators of sage-grouse and other shrub-nesting birds (Schroeder et al. 1999, Vander Haegen et al. 2002), often forage >10 km from nests or perches. Although we primarily considered avian predators, such as corvids or raptors, we also recognize that other species and taxa also may be important; we lacked similar information for many of these species to effectively model their influence.

Differences in reconciling resolutions among spatial data layers presented a particular problem in our assessment. Resolution among individual layers varied from 90-m to 2.5-km and for some data (e.g., area in the Conservation Reserve Program) at the level of state counties. We reconciled these differences in spatial and thematic resolution when possible. Our inability to reconcile spatial or thematic resolution dictated that we use the coarsest scale or category, which limited or added uncertainty to our conclusions. For example, the broadest thematic category of "sagebrush" had to be used in for some states when developing the map of sagebrush distribution and limited our ability to develop fully the spatial and regional differences among sagebrush species. When comparing maps differing in resolution, such as Küchler's (1964, 1970) map of potential vegetation (Fig. 5.3) to current vegetation mapped by satellite imagery (Fig. 5.2), some of the differences were due to true changes in vegetation and some due to a finer resolution in our ability to eliminate areas of nonhabitat or to classify more habitat categories in the satellite imagery (Chapter 5).

We conducted regional analyses when complete coverages for the Conservation Assessment study area were not possible. Similarly, influences such as oil and gas development were significant in only part of the sagebrush biome and were analyzed on a regional basis. However, our final objective was to present the extent of single and cumulative disturbance factors on sagebrush ecosystems (see also Chapter 13).

We used the Public Land Statistics presented in annual reports by the U.S. Bureau of Land Management to assess management issues, habitat characteristics, and treatments conducted on public lands. We recognize that Public Land Statistics often represent gross numbers that may not capture many of the complexities inherent in management of sagebrush ecosystems. For example, numbers of permitted AUMs carry no information about management regime, type of livestock, and actual densities of livestock per unit area. Nonetheless, these reports are used by the agency to document activities on public lands and we have attempted to use that information to characterize use and management of sagebrush lands. Although agency personnel reviewed our use and interpretation of the Public Land Statistics, the final interpretation and conclusions remain our own.

NATURAL HABITAT DISTURBANCE AND CHANGE

Wildfire

Background

Fire was a primary disturbance process in the sagebrush biome but its role was highly heterogeneous in both space and time. The characteristics of a fire are usually described by mean interval[1], severity[2], intensity[3], season, extent or size, and complexity or patchiness. These variables characterize a fire regime for a specific area or plant association. Fire regimes are determined by climate, source and seasonal patterns of ignition, fuel characteristics, topography, and landscape vegetation patterns. Very broad estimates of mean fire return intervals have been reported in the literature for the sagebrush biome. Frost (1998) estimated fire return intervals of 13 to 25 years while Brown's (2000) estimates were 35 to 100 years. Both estimates capture fire return intervals for different parts of the biome. However, these broad estimates hide the true complexity of this disturbance process operating at individual sites. Each scale is important for understanding sagebrush ecosystems: broad estimates present regional differences in dominant disturbances (e.g., fire is dominant and frequent disturbance in the Snake River Plain) in contrast to site-specific

[1] Mean fire interval (also reported ad fire free interval and fire return interval) is the arithmetic average of all fire intervals determined in a designated area during a designated time period; the size of the area and the time period must be specified.
[2] Severity – is the level of biological and physical impact a fire had on plants, soils, and animals.
[3] Intensity – is a measure of the amount of energy released during a fire.

variation (e.g., fires do not occur on an annual or regular basis at each site over time or fire intervals in mountain big sagebrush types are lower than in more xeric Wyoming big sagebrush stands).

The temporal dynamics of regional fire occurrences over a long time period (5,000 years) have been presented based on the abundance of charcoal and ash collected in pond and lake sediment cores (Mehringer 1985). These data suggest fire occurrence increased during relatively wet periods in the semi-arid Intermountain Region, which increased fuel abundance and resulted in distinct long-term patterns in fire occurrence. However, these studies describe only a very coarse scale picture of fire in the sagebrush biome. Finer scale studies that reconstruct pre settlement fire regimes based on hard data across this region are few and spatially limited. Mean fire return estimates as high as 10 to 20 years have been reported in the mountain big sagebrush/Idaho fescue plant association for communities adjacent to the ponderosa pine alliance (Miller and Rose 1999, Miller et al. 2001). It could be argued these communities were predominately grasslands with open scattered stands of shrubs. However, the proportion of the mountain big sagebrush/Idaho fescue plant association characterized by a 10- to 20-year fire return interval is unknown. At the other extreme, a site that supported a patchy and open distribution of old-growth western juniper and mountain big sagebrush was characterized by a fire return interval of 150 years (Miller et al. 2003). It is unlikely we will have a clear picture of the complex patterns of fire regimes that characterize the sagebrush biome within the near future. At best we can only estimate the potential of these different sites to burn based on proxy data, which include the variables that determine a fire regime.

There is very strong evidence, however, that fire regimes have changed across portions of the sagebrush biome resulting in significant changes in plant composition and structure. The increase in conifer encroachment is at least partially fire driven, suggesting a decline in fire occurrence across these areas since Eurasian settlement. The introduction of livestock in the late 1800s would have greatly reduced the fine fuel component reducing the potential for fires to occur. Several studies have reported a decline in fire in the late 1800s, which coincides with the introduction of livestock and the expansion of conifers into shrub steppe and grassland communities (Miller and Rose 1999, Miller and Tausch 2001). In the relatively arid Wyoming big sagebrush type, the invasion by exotic annuals has resulted in dramatic increases in number and frequency of fire (Young and Evans 1973, West 2000). For example, Whisenant (1990) in a non-peer reviewed article estimated mean fire return intervals in Wyoming big sagebrush communities have been reduced from 50-100 years to <10 years in some areas. Repeat fires that eliminated or reduced shrubs, disturbed soils and biotic crusts, and released nutrients that have allowed cheatgrass and other introduced annuals to replace the native shrub and herb layers. The end results are that herbaceous cover is more susceptible to annual weather patterns and varies greatly from year to year depending on moisture availability. Shrub cover also is absent from these landscapes, the season of available green plant material is shortened, high quality perennial forbs are scarce, forage is absent in late summer through winter, and the fire season lengthened (Miller and Eddleman 2001). Both scenarios of altered fire regimes resulting in conifer and exotic weed encroachment have caused a significant loss in sage-grouse habitat.

Current Status

We developed a database of fire statistics from records assembled across the sagebrush biome. Although the fire records included some forested regions, we attempted to eliminate those portions of fire polygons that did not include sagebrush habitats by masking out forested areas identified in the 1992 U.S.G.S. National Land Cover Database. We plotted the frequency of fires for all years since 1900 for which fires were documented (Fig 7.1-5); records of fires in some regions were present from 1870. We recognized, however, that analyses of these data are confounded by an increase in reporting effort by the agencies, reporting effort varied across regions, and records in some districts were not kept until the 1980's, and GIS-based polygons of fire data until the 1990's. Therefore, we mapped fires only from 1960 through 2003 (Fig. 7.6) and conducted statistical analysis on fire size, number of fires, total area burned, and within-year variation in fire sizes recorded from 1980 through 2003 (Table 7.1).

Number of fires and total area burned across the sagebrush biome increased in each of the geographic divisions from 1980 through 2003 (Table 7.1). Average fire size increased during this period only in the Southern Great Basin and Wyoming Basins. Although within-year variation in fire sizes decreased in all geographic regions, the changes were significant only in the Southern Great Basins and the Silver Sagebrush Region (Table 7.1). The decrease in variation within-years is likely due to greater suppression capabilities. Nonetheless, increased number of fires and (in 2 regions) increased size of fires has resulted in increased total area burned since 1980.

Location of fires mapped since 1960 was related to the distribution of cheatgrass within the intermountain region (compare Fig. 5.9, 7.6). Cheatgrass was established throughout the intermountain west by the 1920s and 1930s (Klemmedson and Smith 1964, Mack 1981, Billings 1990). Much of the cheatgrass region in the Snake River Plain in southern Idaho was well-defined by fires burned since 1960. Fires in northern Nevada and eastern Oregon, also within the cheatgrass region, were more pronounced since 1980. Fires in the eastern regions of the sagebrush biome have been recorded only in more recent years.

Annual areas burned on or adjacent to lands managed by the U.S. Bureau of Land Management were highly variable from 1997 through 2002 and illustrate the difficulty in planning for an "average year." Areas burned per year varied almost 6-fold from 1,383 km^2 (sum of force and contract accounts, BLM and nonBLM lands) in 1997 to 8,142 km^2 in 1999 (Table 7.2-3). Idaho, Nevada, and Oregon had the highest total area burned. Human-caused fires within the Conservation Assessment study area were related to the network of roads within the Conservation Assessment Study Area (Fig. 7.7). In addition to road influences on habitat fragmentation and spread of exotic plant species (Trombulak and Frissell 2000), fire ignitions are an additional consequence of roads and access by humans.

Fires are an increasingly significant disturbance throughout much of the sagebrush biome. Part of our recorded increases may be a function of differences in reporting fires and better technology to map fire polygons. Nonetheless, the increased areas burned each year coupled with decreased total area of sagebrush habitats can further accelerate the trajectory of habitat loss for sage-grouse.

Cheatgrass Invasion and Expansion by Juniper and Woodlands

Authors' Note: The models and results presented in the section on cheatgrass invasion and expansion by juniper and woodlands were developed prior to work on the Conservation Assessment. We included the models because of their relevance to sagebrush ecosystems and because the emphasis on large-scale modeling of risk was consistent with the goals of the Assessment. We note, however, that the results and area estimates extend only to the Great Basin ecoregion. We were unable to model the results for the entire Conservation Assessment area because comparable GIS layers were not available. Area estimates and percentages in this section are due to modeling factors to identify regions at risk and may not be directly comparable to area estimates based on actual presence (Chapter 5).

Background

A variety of land uses and ecological processes poses major threats to the persistence of sagebrush and other native shrublands in the Great Basin Ecoregion (Great Basin) and the entire sagebrush ecosystem (Wisdom et al. 2003). One of the most notable threats is that of invasive plants. Effects of invasive species on ecosystem function (e.g., altered fire regimes, nutrient loss, altered local microclimate, prevention of succession) are significant on the local and regional scales, and becoming increasingly more important on a global scale (D'Antonio and Vitousek 1992). Invasion by exotic species, particularly cheatgrass, is consistently cited as a major challenge to maintenance of sagebrush communities (Young and Allen 1997, Knick 1999). Cheatgrass was introduced to the United States in the 1800s and has become a pervasive problem throughout much of the arid West (Mack 1981, Billings 1990). In addition to its displacement of native understory species, cheatgrass autecology (i.e., early germination and drying) leads to an increased risk of wildfires that eliminate the sagebrush overstory (Klemmedson and Smith 1964). Soil erosion also can be accelerated in systems dominated by cheatgrass because of bare ground left after early season fires.

The increase in the distribution and density of pinyon-juniper woodlands has been identified as an additional threat to the sagebrush ecosystem (Miller and Wigand 1994, Miller and Tausch 2001). These woodlands have expanded greatly in the Great Basin when compared to their distribution >150 yrs ago. Trees in established woodlands have also increased in density. These ecological changes have been linked to a decrease in fire frequencies, changes in the climatic regime, historical patterns of livestock grazing, and increases in atmospheric CO_2 (Miller and Rose 1999, Miller and Tausch 2001).

Wisdom et al. (2003) described an approach to assess the status of sagebrush ecosystems that focused on development of processes and models to evaluate the degree and extent of potential threats to native communities. In this chapter, we build on this approach to describe methods for predicting the intensity, distribution, and area of threats posed to plant communities in the Great Basin by displacement from cheatgrass and pinyon-juniper woodlands. We illustrate these methods with example applications and results. Our specific objectives were to (1) describe and document rules and models used to predict displacement of sagebrush by pinyon-juniper woodlands, and displacement of sagebrush and other native vegetation by cheatgrass; (2) apply these risk models to landscapes in the Great Basin; (3) summarize the results in terms of potential losses of native vegetation over time; and (4) discuss implications of results for management.

We focused on the Great Basin for our assessment of risk of pinyon-juniper and cheatgrass displacement because of the expansive and continuing displacement of sagebrush to these plant species in the Basin. Other parts of the range of greater sage-grouse could be evaluated in the future with the methods presented here, as adapted to those sites and conditions. Although other parts of greater sage-grouse range do not appear as susceptible to these threats, a formal analysis of the risks posed by cheatgrass and pinyon-juniper to quantify the spatial extent and magnitude is needed to make such inferences. Such analyses were not possible with the currently available cover types of sagebrush and pinyon-juniper woodlands.

Modeling Risk of Pinyon Pine and Juniper Displacement of Sagebrush

One of the most evident changes in vegetation of the Great Basin during the past 130 yrs has been the expansion of pinyon and juniper woodlands into the sagebrush ecosystem (Miller and Tausch 2001). Pinyon and juniper species are successionally aggressive across their range and can eliminate the understory component of the community after invasion (Johnsen 1962, Tausch and Tueller 1990, Miller et al. 2000). Increases in the distribution and changes in the structure and composition of juniper and pinyon-juniper woodlands have resulted from the combination of inappropriate livestock grazing, alteration of fire regimes, and climate change (Miller and Rose 1995, 1999). Conversion of sagebrush communities to pinyon-juniper woodlands places additional stress on an ecosystem that has been severely reduced in area and degraded in habitat quality.

The area of pinyon-juniper woodlands has increased approximately 10-fold since the late 1800's in the Great Basin (Miller and Tausch 2001). Moreover, these woodlands are capable of expanding over a far greater area (Betancourt 1987, West and Van Pelt 1987). To assess the potential for changes in the distribution and composition of sagebrush habitats associated with pinyon and juniper displacement, we developed a model to estimate the risk that pinyon-juniper woodlands will displace sagebrush habitats in the Great Basin. Nisbet et al. (1983) developed a model of pinyon-juniper woodland encroachment into sagebrush habitats in Utah, as part an evaluation of habitat quality for sage-grouse that used precipitation, elevation, and a radiation load index to predict the potential for encroachment of pinyon-juniper woodlands into sagebrush habitats.

Methods. We identified the environmental variables most important for estimating the risk that sagebrush will be displaced by pinyon pine or juniper by reviewing the literature and using knowledge from experts on the ecological relationships and invasive traits of pinyon pine and juniper. Variables selected for the risk model included vegetation, elevation, potential for dispersal, precipitation, and landform, with each variable parameterized differently for each ecological province (ecological provinces were a subset of the geographic divisions used in the other analyses in this chapter), as described below. Variables that addressed cold-air sinks were considered but not used because we were not able to model them in a GIS environment. The following sections describe how we used these environmental variables to construct and apply our model to evaluate the risk that pinyon-juniper would displace existing sagebrush in the Great Basin.

We used the ecological provinces from West et al. (1998) and Miller et al. (1999) as the geographic basis for developing our risk model (Fig. 7.8). These provinces discriminate well among a variety of environmental gradients, combined with the basin and range topography, that contribute to extensive environmental variation across the Great Basin and adjacent ecoregions (West et al. 1998). The provinces are based on floristic regions (Cronquist et al. 1972), soil-plant relationships (Anderson 1956), Bailey's ecoregions (Bailey 1980), and climate to define this environmental variation (West et al. 1998, Miller et al. 1999*a*). These ecological provinces are large areas (i.e., thousands of km^2), each of which is defined by similarity in climate, topography, geology, and soils (Table 7.4). The ecological characterization of landscape conditions within each of these provinces provided a useful and important ecological context for describing pinyon pine and juniper relationships with environmental factors.

Although we developed rules for modeling the risk of pinyon-juniper displacement across all ecological provinces (Fig. 7.8), our application of the model was limited to the Central High, High Calcareous, and Bonneville Provinces of eastern Nevada and western Utah. Our review of currently available land cover types for juniper and pinyon pine for other Provinces of the Great Basin suggested that accuracy of maps for these woodland types was not sufficient to apply our risk model.

Some authors have noted an association of specific soil characteristics with the distribution of pinyon-juniper woodlands (e.g., shallow, rocky, low fertility [Pieper 1977, Everett 1985]). Pinyon-juniper woodlands, however, are often restricted to such areas by reoccurring fires, and would readily establish on more productive sites without fire (Thatcher and Hart 1974, Miller and Tausch 2001). Current information is not specific enough to associate the productivity of pinyon-juniper woodlands with soil descriptions (West et al. 1978*a*). As a result, it may be more fruitful to associate the likelihood of site establishment by pinyon-juniper with existing patterns of vegetation.

A number of sagebrush taxa are significant components of the understory of pinyon-juniper woodlands throughout their range. The distribution of specific sagebrush taxa, however, varies in association with environmental factors (Chapter 5). As a result, the presence of a particular sagebrush taxon within, or adjacent to, pinyon-juniper woodlands may be used to provide relative comparisons of the favorableness of the site for maintenance or establishment of pinyon and juniper trees (West et al. 1978*b*, Jensen 1990).

The primary relations of sagebrush taxa with ecological conditions associated with pinyon-juniper sites were summarized by West et al. (1978*b*): (1) Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) occurs in the warmest and driest conditions and in soils of medium depth, (2) Black sagebrush (*A. nova*) occurs in drier conditions where temperatures are intermediate and in dry, stony, relatively shallow soils with limited upper horizon, (3) Low sagebrush (*A. arbuscula*) is restricted to the coldest, driest woodland sites in shallow, alkaline clay soils, (4) Basin big sagebrush (*A. tridentata* ssp. *tridentata*) occurs predominately on the wetter, but relatively warm woodland sites in the deepest, most fertile soils, and (5) Mountain big sagebrush (*A. tridentata* ssp. *vaseyana*) dominates the wettest, coolest sites with moderately deep soils in pinyon-juniper woodlands.

Although there are interrelationships with the other variables used in our risk model, pinyon-juniper establishment on sagebrush sites is most likely on wet, cool sites with moderately deep soil. Less vigorous establishment of pinyon-juniper is likely on dry sites with shallower soils. Wet, warm sites with deep soils and dry, cold sites with limited soil development generally are not as susceptible to establishment of pinyon pine and juniper seedlings. These conditions correspond to the distribution of mountain and Wyoming big sagebrush (West et al. 1979*a*, 1979*b*). Burkhardt and Tisdale (1969) found mountain big sagebrush sites most vulnerable to displacement by western juniper, and black sagebrush sites to be less vulnerable.

One of the most important predictors of the distribution of pinyon-juniper woodlands was elevation. Pinyon-juniper woodlands are generally located between elevations of 1,400 – 2,130 m (Springfield 1976), but are most productive between 1,520 – 2,130 m within the Great Basin (Woodbury 1947). The upper elevation limit is restricted by temperature and the lower limit by precipitation (Wright et al. 1979). The upper elevation limit of western juniper distribution in Oregon and Idaho is approximately 2,130 m (Miller et al. 2003). Pinyon pine and juniper dominated the vegetation community at mid elevations (i.e., approximately 2,000 – 2,300 m) in Nevada and declined in dominance above and below this elevation range (West et al. 1978*a*, Tausch et al. 1981). Tausch et al. (1981) also found that downslope expansion of the pinyon-juniper woodland was more extensive than upslope expansion. Contributing to this pattern of less vigorous upslope expansion are shorter growing seasons, more adverse winter climatic conditions, and greater competition from understory species at higher elevations (based on St. Andre et al. 1965). In addition, human-caused disturbances (e.g., livestock grazing, tree harvest, fire suppression) at lower elevations have facilitated displacement of sagebrush by pinyon-juniper.

Proximity of sagebrush to pinyon-juniper, pinyon, or juniper stands was a critical component in our model of risk for sagebrush displacement by these species. The berries and nuts of pinyon pine and juniper are dispersed into sagebrush communities via gravity, water, or animals (Balda and Bateman 1972, Burkhardt and Tisdale 1976, Schupp et al. 1997). Dispersal via gravity or water was reported to be limited to <2 m downslope and <1 m upslope (Burkhardt and Tisdale 1976). As a result, long-distance (i.e., >100 m, Cain et al. 2000) movement of berries and nuts, which would facilitate displacement of sagebrush stands by pinyon pine or juniper, is primarily accomplished by movement of these materials by birds and mammals. These long-distance dispersals approximate the mechanisms associated with biological invasions (Higgins and Richardson 1999).

Juniper berries and pinyon pine nuts are commonly distributed ≤1.6 km from juniper and pinyon-juniper stands by bird and mammal dispersal agents (Schupp et al. 1999). Thus all stands of sagebrush ≤1.6 km from a pinyon-juniper, pinyon, or juniper stand were considered adjacent to pinyon-juniper and at risk to displacement in this analysis. Birds have been reported to disperse seeds up to 5 km from seed sources (Vander Wall and Balda 1977, 1981). Consequently, stands of sagebrush >1.6 but <5 km from a pinyon-juniper, pinyon, or juniper stand may be at a lesser degree of risk to displacement. Christensen and Whitham (1991) observed a threshold of availability of the cones of pinyon pine, below which birds will not forage in a stand. Santos and Telleria (1994) quantified this relationship in juniper woodlands and reported higher seed predation by small mammals and lower seed dispersal by birds in smaller stands of juniper (i.e., 0.2 – 16 ha) when compared to larger stands (i.e., 150 – 270 ha). For our analysis, we assumed that stands of pinyon-juniper, pinyon pine, or juniper <10 ha were below that threshold. We further assumed that movement of seeds or nuts outside of these stands was minimal. Consequently, a “stand” of juniper or pinyon-juniper woodland was defined as ≥10 ha for our analysis.

Effective moisture is probably the main factor in determining the potential of a site for juniper growth and production. Tree densities in pinyon-juniper woodlands were greater on sites with increasing average annual precipitation (Koniak 1986). Annual precipitation varied from 25 – 40 cm in open stands to >40 cm in more dense stands of pinyon-juniper (Woodbury 1947, Springfield 1976). High-density stands of pinyon pine and juniper usually received between 35 – 40 cm of precipitation (Tueller and Clark 1975). In Oregon >50% of western juniper stands occurred in areas where annual precipitation was between 25 – 40 cm (Gedney et al. 1999).

Exclusion of pinyon-juniper woodlands from valley floors was reported by Woodbury (1947), who observed that distribution was restricted from these sites by fine soils and low precipitation. This finding was verified by Springfield (1976), who reported that pinyon-juniper growth was especially favored on coarse-textured soils. West et al. (1978*a*) also reported that the lower boundaries of pinyon-juniper woodlands appeared related to valley floor topography. Burkhardt and Tisdale (1976) found that invading western junipers favored upper slopes. Tausch and Tueller (1990) reported that foliage biomass of invading pinyon and juniper was 1/3 greater on slopes than on alluvial fans (i.e., < 5% slope). For our model, the valley floor landform was defined as having <5% slope (Meeuwig and Cooper 1981) and ≥40 ha in extent.

Classes of risk that sagebrush would be displaced by pinyon pine, juniper, or pinyon-juniper woodlands in sagebrush cover types were defined and described as follows:

(1) Low – the probability that pinyon/juniper will displace existing sagebrush cover types within 30 years is minimal; little or no pinyon/juniper is likely to be present in the overstory of these sagebrush stands at the current time.

(2) Moderate – the probability that pinyon/juniper will displace sagebrush within 30 years is likely, but less so than sagebrush at high risk; pinyon/juniper is likely to be a minor to

common component of the overstory of these stands at the current time. This class represents a transition phase in the conversion of sagebrush cover types to pinyon/juniper woodlands (Miller et al. 1999*b*). Sagebrush stands are expected to cross the threshold from low risk to high risk relatively quickly. Therefore, the total area in this class is expected to be small when compared to the other classes.

(3) High – the probability that pinyon/juniper will displace sagebrush within 30 years is very likely; pinyon/juniper is likely to be a common to dominant component of the overstory of these stands at the current time.

These definitions of risk were based on a 30-year projection, owing to uncertainties associated with future climate change and other long-term changes in environmental conditions (e.g., stochastic changes in fire regimes or interactions with other plant invasion dynamics). That is, our rules for modeling risk of pinyon-juniper displacement of sagebrush were based on current knowledge of how these woodland types have invaded and displaced sagebrush sites to date. Whether these processes and patterns will continue beyond the next 30 years was not accounted for in our model because of high uncertainties associated with longer-term changes brought about by climate change and other stochastic events that are difficult to project.

A rule-based model was developed to integrate the species of sagebrush, elevation, and proximity of sagebrush lands to existing pinyon-juniper woodlands (Wisdom et al. 2003). Digital maps representing the variables included in the rule-based model were acquired or created. A program was written in Arc Macro Language and used in ArcInfo GIS to access the digital maps, apply the rules, and create spatial representations of the resulting estimates of risk. Due to map accuracy, our application of the risk model was restricted to the Central High, High Calcereous, and Bonneville Ecological Provinces in the eastern Great Basin (Fig. 7.8). Additional applications of the risk model to the other Provinces will be possible with the improvement in mapping accuracy of juniper and pinyon-pine land cover types in the future.

The following spatial databases were used to apply this model in a GIS environment: Ecological provinces, State boundaries, Digital elevation model, Precipitation, and Land cover class.

Results. Nearly 60% of the current area occupied by sagebrush cover types in the Central High, High Calcereous, and Bonneville Ecological Provinces in the eastern Great Basin was estimated to be at low risk to displacement by pinyon-juniper woodlands. Six percent of all sagebrush cover types was estimated to be at moderate risk and 35% at high risk. The Wyoming-basin big sagebrush cover type was found on nearly 60% of the area covered by sagebrush in the Central High, High Calcereous, and Bonneville Ecological Provinces. Black sagebrush was found on 19% of the area and mountain big sagebrush on 15%. These cover types also made up similar percentages in the low and high classes of risk. The moderate risk class was dominated by the mountain big sagebrush cover type (50%) and had a large component of the Wyoming-basin big sagebrush cover type (25%).

The percentage of sagebrush cover types by risk class was fairly consistent among the ecological provinces (Fig. 7.9). The percentage of total sagebrush at low risk to pinyon-juniper ranged from 59% - 61% across the 3 provinces. Percentage of sagebrush at moderate and high risk also varied little across the 3 provinces (5% - 8% for moderate risk; 34% - 36% for high risk).

Large percentages of the low sagebrush-mountain big sagebrush (57% - 86%) and the mountain big sagebrush (30% - 54%) cover types were at high risk to displacement by pinyon-juniper woodlands across the 3 provinces. Conversely, large percentages of black sagebrush (56% - 76%) and Wyoming-basin big sagebrush (61% -72%) cover types were at low risk to displacement by pinyon-juniper woodlands.

Discussion. Although a large percentage of the sagebrush community in the eastern Great Basin was estimated to be at low risk to displacement by pinyon-juniper woodlands, >17,000 km^2 (35%) were considered to be at high risk. Sagebrush cover types identified as being at high risk to displacement by pinyon-juniper encompass vast areas of the three Provinces we evaluated with our model. These areas at high risk appear to be most amenable to the use of prescribed fire to control or eliminate further encroachment by pinyon-juniper. In general, these high-risk sites are at higher elevations, have deeper soils, and receive higher amounts of precipitation than sagebrush types at lower elevations, all of which contribute to higher resiliency and positive responses of these high-risk sagebrush sites to burning (see discussion on burning other parts of this Chapter). Use of prescribed burning to control pinyon-juniper encroachment on low- and moderate-risk sites would likely not be as effective, owing to the lack of resiliency of sagebrush cover types to fire in these areas that are generally lower in elevation, associated with shallow soils, and receive lower amounts of precipitation.

The pinyon-juniper risk model requires extensive evaluation with new field research to assess its performance. Without such evaluation of model performance, management use of the model predictions may result in inappropriate action, due to the high uncertainty associated with the costs and effectiveness of management actions in relation to our results. Consequently, new research to evaluate the performance of our risk model is a critical and compelling need for managers of sagebrush and pinyon-juniper plant communities in the Great Basin.

Although the parameters used in this model have a robust empirical basis, this model should be considered to be a series of hypotheses regarding the individual and combined effect of the parameters on the probability of dominance by pinyon and juniper in sagebrush communities. There has been limited work on integrating the variables used in this effort to predict the risk of dominance by pinyon and juniper in sagebrush communities.

Key Findings. Almost 60% of sagebrush (>28,000 km^2) in the eastern Great Basin is at low risk to displacement by pinyon-juniper woodlands, based on estimates developed from our predictive model.

(1) 35% of sagebrush (>17,000 km^2) in the eastern Great Basin is at high risk to displacement by pinyon-juniper woodlands.

(2) Mountain big sagebrush appears to be the sagebrush cover type most susceptible to displacement by pinyon-juniper woodlands.

(3) Mitigating the threat posed by pinyon-juniper to mountain big sagebrush may be effective with an aggressive program of prescribed burning. Other sagebrush cover types at high risk to pinyon-juniper may not respond as well to burning; in these situations, mechanical control of pinyon-juniper is needed to mitigate threat of sagebrush loss.
(4) Extensive field research is needed to validate our estimates of risk that pinyon-juniper woodlands will displace existing sagebrush in the Great Basin.

Modeling Risk of Cheatgrass Displacement of Sagebrush and other Native Vegetation

Cheatgrass is an exotic annual grass native to Eurasia and the Mediterranean that was probably introduced to western North America in impure grain seed (Mack and Pyke 1983, Novak and Mack 2001). It had spread throughout most of the Intermountain West by 1900 (Klemmedson and Smith 1964, Young 1991) and was well established by the late 1920's (Mack 1981). This rapid and aggressive spread of cheatgrass was facilitated by ecological features such as early germination following late season precipitation, seeds that do not go dormant, rapid fall and spring growth, highly competitive, large numbers of seeds per plant, and resistance to grazing pressure (Hulbert 1955, Hinds 1975, Mack and Pyke 1983).

Cheatgrass readily out-competes native plant species for water and nutrients (Harris and Wilson 1970; Inouye 1980, 1991). Cheatgrass responds dramatically to the availability of nitrogen, to the detriment of perennial plants, since it directly depletes nitrogen from the soil and interferes with N_2-fixation by the biological soil crust (Kay and Evans 1965, Wilson et al. 1966, McLendon and Redente 1991, Evans et al. 2001). Although germination and root growth characteristics make cheatgrass a very aggressive plant, it tends to be most competitive with native vegetation after disturbance (Harris 1967). However, that tendency may be changing. Cheatgrass is now replacing sagebrush slowly over time without disturbance, such as fire, especially on drier Wyoming sagebrush sites, and also on some salt desert shrub sites (first documented on Anaho Island National Wildlife Refuge, Washoe County, Nevada).

The density and structure of standing dead cheatgrass results in increased flammability when compared to native species and leads to increased fire intensity and frequency (Stewart and Hull 1949, Brooks 1999). These factors can change the fire recurrence interval from 20 – 100 yrs for sagebrush ecosystems to much shorter intervals for cheatgrass-dominated sites (Young and Evans 1978, West and Hassan 1985). This increase in fire frequency may eliminate native plant species from a site through increased competition for water and decreased productivity of native species following fire (Melgoza et al. 1990). The frequent cycle of large fires also directly eliminates native shrubs, forbs, and perennial grasses and results in a self-perpetuating stand of cheatgrass. The rate of spread and size of fires also increase with increasing density of cheatgrass. Extensive cheatgrass invasion also modifies the temporal distribution of fires by increasing the occurrence of fire earlier in the growing season, which negatively affects native herbaceous species. Frequent fires may also remove protective plant and litter cover, increasing flooding and susceptibility of soil to wind and water erosion (Klemmedson and Smith 1964).

Methods. The environmental variables most important to estimating the risk of displacement of native vegetation by cheatgrass were determined through review of the literature and personal knowledge of the autecology and ecological relationships of this species. Variables selected in our model included aspect, slope, elevation, and landform by ecological province.

As done for modeling pinyon-juniper risk, we used the ecological provinces as the geographic basis for developing our cheatgrass risk model (Fig. 7.8). The ecological characterization of landscape conditions within each of these provinces provided a useful and important ecological context for describing cheatgrass relationships with environmental factors for modeling the risk that existing native vegetation would be displaced by this invasive species.

South-facing slopes are the most susceptible to displacement by cheatgrass (Mosley et al. 1999). These aspects are energy rich (Hinds 1975). Uptake of minerals (i.e., nitrogen, phosphorus, potassium, and calcium) was approximately 20% greater for cheatgrass on southern exposures than on northern exposures (Hinds 1975). Greater cheatgrass root and seed production also occurred on southern exposures compared to northern exposures (Hinds 1975).

The slope of the ground influences local sun angle. Slopes tipped into the sun have higher sun angles and hence more intense insolation than horizontal or slopes tipped away from the sun (Nikolov and Zeller 1992). Cheatgrass responds positively to increased insolation, especially in the spring (Stewart and Hull 1949, Hulbert 1955, Klemmedson and Smith 1964).

In the northern ecological provinces, cheatgrass is most abundant at lower elevations from 600 – 1,830 m (Hull and Pechanec 1947, Stewart and Hull 1949). However, cheatgrass occurred at higher elevations on south-facing slopes in Idaho than on north-facing slopes (Stewart and Hull 1949). In the southern ecological provinces, cheatgrass was commonly found only at high elevations (e.g., >1,675 m) in 1966 (Beatley 1966) but has since become more common at lower elevations (e.g., <1,220 m) (Hunter 1991).

Lack of a continuous snow cover at low elevations and winter precipitation in the form of rain rather than snow greatly enhanced winter emergence of cheatgrass seedlings (Mack and Pyke 1983). Germination of cheatgrass was substantially enhanced at moderate temperatures (i.e.,~20 C) with very limited germination at <10 C (Harris 1967), indicating that lower elevations with higher temperatures are conducive to cheatgrass establishment. Germination was also greatly reduced for seeds that were frozen while wet as compared to those frozen while dry (Warg 1938). This suggests that wet, cold environments associated with higher elevations are not conducive to cheatgrass survival. However, cheatgrass can maintain significant root growth at lower winter temperatures than can native species (Harris and Wilson 1970). Emergence, survivorship, and fecundity in cheatgrass populations generally decreased with increased elevation primarily as a result of decreasing temperatures and decreasing length of growing season (Pierson and Mack 1990).

Valley bottoms are susceptible to cheatgrass invasion (Monsen 1994), especially in the southern ecological provinces. Sparks et al. (1990) also noted pervasive cheatgrass invasion on flat, mid-elevation (i.e.,~1,675 m) landforms.

Classes of risk of displacement of native vegetation by cheatgrass were defined as follows:

Low – The probability that cheatgrass will displace existing sagebrush or other susceptible cover types within 30 yrs is minimal; native plants are likely to dominate the understory of these stands at the current time.

Moderate – The probability that cheatgrass will displace sagebrush or other susceptible cover types within 30 yrs is moderate, but lower than for types at high risk; either cheatgrass or native plants can dominate the understory at the current time.

High – The probability that cheatgrass will displace sagebrush or other susceptible types within 30 years is very likely; cheatgrass is likely to dominate the understory (vs. native plants) at the current time.

These definitions of risk were based on a 30-year projection, owing to uncertainties associated with future climate change and other long-term changes in environmental conditions (e.g., stochastic changes in fire regimes or interactions with other plant invasion dynamics). That is, our rules for modeling risk of cheatgrass displacement of sagebrush and other shrublands were based on current knowledge of how cheatgrass has invaded and displaced sagebrush sites to date. Whether these processes and patterns will continue beyond the next 30 years was not accounted for in our model because of high uncertainties associated with longer-term changes brought about by climate change and other stochastic events that are difficult to project. Moreover, cheatgrass has continued to evolve and adapt to many sites previously considered not susceptible to invasion by the species (Meyer et al. 2001), and we could not account for the potential future adaptations of the species beyond the next 30 years.

A rule-based model was developed to integrate the parameters (Wisdom et al. 2003). Digital maps representing the variables included in the rule-based model were acquired or created. A program was written in Arc Macro Language and applied in ArcInfo GIS to access the digital maps, apply the rules, and create spatial representations of the resulting estimates of risk. Cover types considered to be susceptible to displacement by cheatgrass included native grasslands, salt desert shrubs, sagebrush, mesic shrubs, and pinyon and juniper woodlands (Hull and Pechanec 1947, Sparks et al. 1990, Mosley et al. 1999, Meyer et al. 2001).

The following spatial databases were used to apply this model in a GIS environment: ecological provinces, and digital elevation model. The digital map of the estimated risk of displacement by pinyon-juniper in the Central High, High Calcereous, and Bonneville ecological provinces in the eastern Great Basin was combined with the digital map of the estimated risk of displacement by cheatgrass in the same area through GIS processes. This allowed examination of the potential risk to sagebrush communities by both threats simultaneously. Risk classes were developed as follows:

Low – low risk from both cheatgrass and pinyon-juniper;

Low-moderate – moderate risk from both cheatgrass and pinyon-juniper, combination of low and moderate risk from either cheatgrass or pinyon-juniper;
High cheatgrass – high risk from cheatgrass and low or moderate risk from pinyon-juniper;
High pinyon-juniper – high risk from pinyon-juniper and low or moderate risk from cheatgrass; High – high risk from both cheatgrass and pinyon-juniper.

Results. Nearly 80% of the land area in the Great Basin was estimated to be susceptible to displacement by cheatgrass (Fig. 7.10). Of that area, >65% was estimated to be at moderate or high risk. The salt desert scrub cover type covers the largest area (i.e., >25%) in the Great Basin, and nearly 80% of this cover type was estimated to be at high risk to displacement by cheatgrass. Sagebrush cover types occupy >28% of the Great Basin. Nearly 38% of the combined area of these sagebrush cover types was at moderate risk and nearly 20% was at high risk.

Discussion. Specific plant communities and environmental conditions vary in their capacity to recover from cheatgrass displacement. Stewart and Hull (1949) suggested that communities at low elevations and sites receiving <23 cm of precipitation annually were less likely to benefit from protection or management practices. Billings (1990) went further and indicated that it is not possible to remove or control cheatgrass once it dominates a sagebrush community. Therefore, areas estimated to be at high risk already may have passed the threshold of recovery.

The cheatgrass risk model requires extensive field evaluation to assess its performance. Although the parameters used in this model have a robust empirical basis, this model should be considered to be a series of hypotheses regarding the individual and combined effect of the parameters on the probability of dominance by cheatgrass in arid shrub communities. There has been limited work on integrating the variables used in this effort to predict the risk of dominance by cheatgrass in arid shrub communities.

Cover types at risk from cheatgrass may be under- or over-estimated because of uncertainties about the changing adaptability of cheatgrass. We assumed that the entire salt desert scrub cover type was susceptible to cheatgrass invasion; however, this assumption may lead to overestimation of the area at risk. Portions of other cover types associated with highly saline or other soil types that inhibit cheatgrass establishment may also have lower risk than we estimated.

Our cheatgrass risk model was not intended to identify areas where cheatgrass has already displaced sagebrush and other susceptible cover types. Rather, the model was designed and applied to predict the risk of future displacement of existing native vegetation by cheatgrass within 30 years.

Key Findings. Approximately 80% of the land area in the Great Basin is susceptible to displacement by cheatgrass. Wyoming-basin big sagebrush and salt desert scrub cover types occupy >40% of the Great Basin and are the cover types most at risk to displacement by cheatgrass. Mountain big sagebrush is generally at lower risk of invasion. Cover types estimated to be at high risk may have already crossed the threshold to conversion to cheatgrass, and this level of risk may be difficult to mitigate.

A small percentage of sagebrush cover types in the eastern Great Basin was estimated to be at high risk to both pinyon-juniper and cheatgrass displacement. However, almost 90% of this area (i.e., the eastern Great Basin) was estimated to be at moderate or high risk from at least 1 of these threats. Ninety-five percent of the Wyoming-basin big sagebrush cover type and lesser amounts of black sagebrush and mountain big sagebrush cover types in the Great Basin were at risk to 1 or both of the 2 threats. The area of the Wyoming-basin big sagebrush cover type at risk was somewhat evenly divided among low-moderate risk versus high risk from either threat.

Weather and Global Climate Change

Background

Two temporal scales of climate scenarios influence sagebrush ecosystems in addition to seasonal variation. First, amount and timing of precipitation has varied annually and sagebrush regions are subjected to periodic drought (Patterson 1952, Thurow and Taylor 1999). An operational definition of drought is based on degree of departure from an average rate of precipitation or other climate variable that has been derived from an historical (usually 30 years) average (U.S. National Drought Mitigation Center 2004). Therefore, drought defined relative to an average set of conditions has occurred periodically, but not regularly in sagebrush habitats. Drought affected the sagebrush biome during the periods approximated by the late 1890's to 1905; mid-1920's to 1940, early 1950's to mid-1960's, mid 1970's, from mid-1980's to mid-1990's and from 1999-2004 (Fig. 7.11).

Sagebrush systems maximize productivity in late spring and early summer when precipitation and warm temperatures coincide (West 1983); most of the water available to plants in the surface soil layers is absent by midsummer (Anderson et al. 1987). Semi-arid shrublands are subject to soil erosion during drought because precipitation is insufficient to maintain vegetative cover (Thurow and Taylor 1999). Consequently, timing and abundance of water availability is the major factor that determines the relative abundance of individual plant species (Toft et al. 1989, West 1996, Anderson and Inouye 2001).

Long-term changes in global climate and atmospheric conditions (particularly in increased levels of CO_2) will shift competitive advantage among individual plant species. Global climate change models predict more variable and severe weather events, higher temperatures, drier summer soil conditions, and wetter winter seasons will dominate future weather patterns at mid-latitude, semi-arid regions (Schlesinger et al. 1990, Schneider 1993). The ability of very broad-scale models of climate changes to predict regional scenarios is limited. Therefore, projecting potential changes in vegetation across arid and semi-arid landscapes or even to functional response of individual species is limited (Reynolds et al. 1997). However, long-term changes in climate that facilitate invasion and establishment by invasive species (Mooney and Hobbs 2000) or exacerbate the fire regime could accelerate the loss of sagebrush habitats.

Ecological Influences and Pathways

Drought is an episodic event in all systems (Thurow and Taylor 1999). Plants in sagebrush communities have evolved mechanisms, such as deep rooting systems or shedding of leaves to survive periods of low water stresses (West 1996). The primary effect of drought is reduced vegetation cover and potential to increase soil erosion, which then leads to reduced soil depth, decreased water infiltration, and reduced water storage capacity (Milton et al. 1994, Thurow and Taylor 1999). Soil erosion is considered to be the greatest threat to long-term sustainability of shrublands (Society for Range Management 1995).

Water stress and reduced vegetation cover contribute to low resistance by plants to disturbance during drought periods. High levels of livestock grazing during severe droughts in the late 1800's and early 1900's resulted in significant changes in soils, loss of perennial grasses and forbs, and increased shrub density (West and Young 2000). The ability to reduce livestock grazing in response to drought is critical to avoid loss of vegetation cover and increased soil erosion. However, stocking rates now are based on an average set of forage conditions and to maximize production and the potential to reduce grazing is influenced more strongly by economic incentives or factors external to the sagebrush community (Holechek 1996). Trends towards "improved" shrubland conditions (Society for Range Management 1989) may be reversed during the recent drought periods if response in grazing levels is absent or delayed (Box 1990).

Atmospheric CO_2 has increased from pre-industrial levels of 280 ppm to current levels of 360 ppm (Bazzaz et al. 1996). Over the last century, atmospheric CO_2 has increased by 20% (Polley 1997). The ecosystem response to enhanced CO_2 is a complex interaction of biogeochemical cycles, water and energy fluxes, and vegetation dynamics dependent on the temporal scale over which CO_2 increases (Walker and Steffen 1996, Körner 1996). Anthropogenic changes on the landscape that influence ecosystem processes may either ameliorate or intensify those effects (Noble 1996).

Future effects of global climate change for shrubsteppe systems must be considered in the context of the current short-term large-scale habitat changes. Exotic annuals, especially cheatgrass, have increased the frequency and intensity of wildfires from the historical disturbance regime and facilitated the large-scale conversion of shrublands into exotic annual grassland. As a result, effects of current disturbances are amplified by greater susceptibility for habitats to burn as well as the decreased likelihood for recovery shrublands. Cheatgrass had positive response to elevated CO_2 when compared to native C_3 grasses (Smith et al. 1987). Cheatgrass already successfully competes against native grasses because of earlier maturation, shallow root systems to collect water in soils, seed production and response to fire disturbance (Klemmedson and Smith 1964). Therefore, further increases in the ability of cheatgrass to compete in sagebrush ecosystems created by enhanced CO_2 or changes in annual precipitation, temperature, or severe storms will facilitate spread and exacerbate the cycle of fire and cheatgrass dominance (d'Antonio and Vitousek 1992, d'Antonio 2000).

INVASIVE SPECIES

Invasive Plant Species

Background

An invasive species is defined as a species that is non-native to the ecosystem under consideration and whose introduction causes or is likely to cause economic or environmental harm or harm to human health (Executive Order 13112, signed by President Clinton 1999). Invasive plants are impacting a wide range of habitats used by sage-grouse and these species are not restricted to just invasive annual grasses such as cheatgrass or medusahead (Table 7.5). Sheley and Petroff (1999) list 29 species of rangeland weeds. With the exception of the snakeweeds (Broom snakeweed [*Gutierrezia sarothrae*] and threadleaf snakeweed [*G. microcephala*]), which are natives to the western United States, the remainder are defined as invasive in one or more of the sagebrush communities within the assessment area. Broom snakeweed is native to many sagebrush communities in the assessment area and may increase in dominance with drought and heavy grazing by livestock (McDaniel et al. 1982), but would not be listed as an invasive plant.

The Interior Columbia Basin Ecosystem Management Project compiled a similar list of 25 species (U.S.D.A. Forest Service and U.S.D.I Bureau of Land Management 1997*a*,*b*). No scientific reports, models or maps currently exist to provide a list of the susceptibility of habitats within the assessment area to invasion by these weeds. Estimates of susceptibility (Table 7.5) are based upon the knowledge of experts and written descriptions of the types of vegetation communities where infestations or colonization populations currently exist. In the case of both Sheley and Petroff (1999) and U.S.D.A. Forest Service and U.S.D.I. Bureau of Land Management (1997*a*,*b*), they both relied heavily on distribution maps of counties in the five-state area (Idaho, Montana, Oregon, Washington, Wyoming) covered by the Invaders Database (Rice 2004) where counties are occupied by the plant if at least one occurrence of a species has been recorded and verified through herbarium collections or reports. Sheley and Petroff (1999) extended these maps into surrounding states, but we were unsuccessful in obtaining and verifying their data.

Estimates of the size of infestations of any of these species are subjective because of the lack of a definition of what constitutes and infestation. Roché and Roché (1999) illustrate the problem with estimates of diffuse (white) knapweed (*Centaurea diffusa*) infestation in Idaho that ranged from 5,670 to 410 km^2. Thus, it is extremely difficult to ascertain a reasonable estimate of the area of lands currently occupied by invasive plants within the assessment area.

Many of the species listed in Table 7.5 have created infestations in some locations in the assessment area. For example, diffuse knapweed is estimated to reach its greatest competitiveness within those shrub grassland communities where bitterbrush (*Purshia tridentata*) may dominate or codominate along the eastern side of the Cascade Range (Roché and Roché 1999). In some locations along the Snake River Plains and the Boise Front Range, areas once dominated by cheatgrass are now dominated by medusahead and are being invaded by rush skeletonweed

(*Chondrilla juncea*) (Pers. Obs. D.A. Pyke) which tends to invade disturbed areas in the drier sagebrush grassland communities (Sheley et al. 1999).

Since an invasive species, by definition, must harm the environment, the economics of the area or human health, what are the typical environmental harms that they may cause to an ecosystem? The impacts of invasive species may be evaluated in how they relate to community structure of the organisms (the type, number and relative abundance), and to functional relationships among organisms in a community.

The structure of plant communities is altered when invasive species are able to replace other plant species within a community. Invasive annual grasses are an excellent example of how structural changes in a plant community may occur. The invasion of cheatgrass into a sagebrush grassland site may provide a continuous fuel source for fires. Most sagebrush species are intolerant of fire and are killed, with the exception of threetip (*Artemisia tripartita*) and silver sagebrush (*A. cana*) which are sprouters. Re-establishment of sagebrush on a site after fire requires available seed, appropriate conditions for germination and survival of a seedling. However, the presence of cheatgrass in the community provides a highly competitive plant making it difficult for other species to establish (Harris 1967, Francis and Pyke 1996). Thus as native plants die at the site they are not replaced and may eventually be eliminated from the site.

A second structural change that may occur with infestations of invasive plants is a change in life forms represented in the community. Once again using the cheatgrass example, the elimination of sagebrush may eliminate the woody plant component from the community for a longer time than it would if the herbaceous understory consisted of native perennial bunchgrasses and forbs rather than cheatgrass. This type of structural alteration becomes extremely important for sagebrush-obligate animals.

A third form of structural change can occur below ground with changes in the forms or amounts of nutrients. Shrub grassland communities that are dominated by perennial plants have a mixture of shallow and deep roots of differing forms of carbon that decomposed at different rates. As a native shrub grassland community is converted to a community dominated by cheatgrass, the roots contain less structural cells and decompose more quickly than woody plant roots changing the distribution of organic matter in the soil to being more concentrated near the surface (Norton, *in press*).

Functional relationships among organisms may also change with the conversion of a community from a diverse native plant system to dominance by an invasive plant. Shifts from fibrous to tap roots can result in reduced water infiltration in some soils (Tisdall and Oades 1982). Within the shrub grassland systems in the assessment area, the native plant communities are characterized by a discontinuous spatial arrangement of perennial plants within the community. The interspaces are generally filled with biological soil crusts (lichens, mosses, and cyanobacteria) (Johansen 1993). These crusts are more prominent on finer textured soils and more arid environments where perennial plants tend to be widely spaced. Communities with wide spatial arrangements of perennial plants or with woody plants tend to concentrate soil nutrients around these plants. This creates a heterogeneous distribution of nutrients with resource-rich patches surrounding perennial plants and resource-poor interspaces (Charley and West 1977, Doescher et al. 1984, Bolton et al. 1993, Jackson and Caldwell 1993, Halvorson et al. 1995, Ryel et al. 1996).

The shift from a native shrub grassland community to a near monoculture of annual invasive grasses such as cheatgrass changes the temporal availability of water and may impact nutrients as well. Stands of cheatgrass reduced growth on native perennial plants, which was a function of significant reduction in water availability and the reduced native plant water content (Melgoza et al. 1990, Booth et al. 2003).

There is considerable speculation regarding the impact that cheatgrass has on nitrogen (N) cycles and how nitrogen levels relate to native plant maintenance and establishment in communities. Evans et al. (2001) speculate that cheatgrass monocultures will lead to reduced N availability at a site, but others have found no evidence for this relationship (Boulton et al. 1990, Svejcar and Sheley 2001). Evidence for increased native perennial establishment with reduced N availability has been found in sagebrush ecosystems in northwestern Colorado and northwestern Nevada (McLendon and Redente 1990, 1992, Young et al. 1997), but whether this is the primary functional driver of succession or a secondary driver associated with water uptake (Booth et al. 2003) is unclear.

LAND USE

Agriculture

Background

The United States government encouraged conversion and development of sagebrush and other arid lands under the Homestead Act of 1862 and Desert Lands Entry Act of 1877 (Chapter 1, Table 1.2). Subsequent editions of Homestead Acts (1909, 1912, 1916) recognized the relatively low productivity of these lands and increased the size of area that could be claimed.

The prime areas for growing crops were claimed early during settlement. An estimated 420,000 km^2 of shrubsteppe existed in Washington State prior to settlement in the 19th century (Dobler et al. 1996). Summer fallowing had started by 1879 and by 1920, 80% of the Palouse region in southeastern Washington State was under cultivation (Buss and Dziedic 1955). Grasslands in eastern Washington, which once covered 25% of the region were reduced to 2% (McDonald and Reese 1998). Only 170,000 km^2 of shrubsteppe across the state was present in 1986 (Dobler et al. 1996). Our estimate of 200,000 km^2 (Table 1.1) was obtained from satellite imagery taken in the early 1990's.

Lands converted to agriculture were concentrated in those regions having deep, fertile soils and water for irrigation (Scott et al. 2001). For example, almost all of the basin big sagebrush region in the Snake River Plains of Idaho have been converted to cropland (Hironaka et al. 1983). In

Washington, agriculture has replaced 75% of the shrubsteppe in deep soils but only 15% in shallow soils (Vander Haegen et al. 2000). An estimated 10% of sagebrush steppe has been converted to agriculture; irrigation is not feasible or the topography and soils are limiting on the remaining 90% (West 1996). However, technological increases in irrigation methods now permit agriculture development on steeper terrains and in regions further from river floodplains.

Ecological Influences and Pathways

Landscapes converted to agriculture directly influence sagebrush systems by habitat loss and fragmentation (Knick and Rotenberry 1997, Wisdom et al. 2002). In the upper Snake River Plains of southeastern Idaho, declines in sage-grouse populations were correlated with amount of area used for agriculture, which increased from 403 km^2 in 1975 to 635 km^2 in 1985, and to 701 km^2 in 1992 (Leonard et al. 2000). Another 45% of the shrublands remaining in the eastern Idaho study area were privately owned and potentially at risk of conversion. Sage-grouse populations declined by 73% on a study area in southcentral Montana concurrent with an increase on sagebrush habitats that were ploughed and converted to croplands in south central Montana (Swenson et al. 1987). Habitat loss and fragmentation were substantial between 1958 and 1993 within the range of the Gunnison sage-grouse (*Centrocercus minimus*) in southwestern Colorado (Oyler-McCance et al. 2001); the Gunnison sage-grouse now occupies approximately 12% of its historical range (Schroeder et al. 2004).

Habitat loss and fragmentation occur at multiple scales. Sagebrush habitats were highly fragmented at smaller scales in landscapes dominated by agriculture in Washington and the northeastern portion of the Conservation Assessment area (compare Fig. 7.12, Fig. 5.17). Isolation of shrubsteppe habitats increased, mean patch size decreased, and number of patches increased with habitat conversion to agriculture in Washington (McDonald and Reese 1998). Additional habitat fragmentation at small scales results from water developments and irrigation channels associated with irrigated croplands. Broad-scale fragmentation resulting from intensive agricultural development has separated sagebrush-dominated landscapes in the Snake River Plains of southern Idaho (Chapter 5) (Fig. 5.17) (Knick et al. 2003); sage-grouse populations once were continuous across this region but now are disconnected.

Agriculture development indirectly influences wildlife in sagebrush habitats by providing access for predators such as domestic cats and red fox (*Vulpes vulpes*) as well as to corvids and cowbirds (*Molothrus ater*) (Vander Haegen and Walker 1999, Vander Haegen et al. 2002). Because of the foraging distance of these predators and nest parasites, the actual influence can extend up to 6.9 km from the agricultural development.

Chemicals applied to crops also can influence sage-grouse and other wildlife that are attracted to irrigated fields. Sage-grouse foraging in alfalfa and potato fields died following application of organophosphate insecticides to those fields (Blus et al. 1989). Because sage-grouse often move long distances between seasonal ranges, the total area influenced may be large.

Current Status

The primary agricultural regions in the sagebrush biome included central Washington and northern Oregon, the Snake River Plains of southern Idaho, northern Utah, northern Montana, southern Alberta, southern Saskatchewan, and western North Dakota (Fig. 7.12). Agricultural lands comprised 248,975 km^2 of the Conservation Assessment area. The total area influenced by agriculture (including a 6.9-km buffer) was 1,152,157 km^2 or 56% of the Assessment Study Area (Fig. 7.12). Irrigation canals covered an additional 6,916 km^2 of the Conservation Assessment Study Area.

The Conservation Reserve Program (CRP) is a voluntary program authorized in 1985 in which land owners receive annual payments in return for establishing permanent vegetation on idle or erodible lands that had previously used for growing crops. The purpose of the program is to control soil erosion, improve water retention, and provide wildlife habitat. Lands placed into the program cannot be grazed except under emergency drought conditions. Beginning in 1987, the amount of lands placed in the Conservation Reserve Program has markedly increased (Fig 7.13). The increase was primarily in the agricultural regions of eastern Washington, Montana, and plains regions of eastern Colorado. The value of lands placed in the Conservation Reserve Program to sage-grouse has yet to be demonstrated.

Urbanization

Background

Indigenous people always had been present in the sagebrush biome. Their low densities (1 person/6-90 km^2 in the Great Basin) limited their impact on the biophysical landscape although their activities for hunting, gathering, and burning may have been significant locally (Griffin 2002). The availability of resources often limited areas that could be settled by early inhabitants of arid regions (West 1999). Early settlements by European settlers were located along transportation corridors, such as rivers or the railroad lines, or in regions where minerals had been discovered (Young and Sparks 2002). Ultimately, settlement by Europeans in sagebrush habitats had a much greater effect on transforming or converting habitat, altering disturbance regimes and animal communities, and facilitating spread of invasive species than exerted by the low densities of indigenous people (West and Young 2000, Griffin 2002).

Populations have grown and expanded over the past century, primarily in the western portion of the sagebrush biome (Fig. 7.14). In 1900, 51% of the 325 counties in the Assessment area had <1 person/ km^2 and 4% of the counties had densities of >10 persons/ km^2 (Fig. 7.14). By 1950, 39% of the counties had <1 person/km^2 and 9% had >10 persons/ km^2. In the most recent census in 2000, 31% of the counties had <1 person/km^2 and 22% had >10 persons/km^2. In addition to increases in total density (Table 7.6), the general movement of populations has been from counties in Midwestern plains to western States (Fig. 7.14).

Ecological Influences and Pathways

People living in cities require resources to sustain their lives. Those resources either need to come from the surrounding region or to be transported into the cities from elsewhere. Increases in technological capabilities have reduced the limitations to moving resources and increased our connectivity. In addition, increased affluence has resulted in additional uses of lands surrounding cities for development of homes on larger acreages (ranchettes) or for motorized recreation using All-Terrain Vehicles, motorcycles, or 4-wheel drive vehicles.

Urban areas by themselves remove habitat and present inhospitable environments for sage-grouse. However, the connecting roads and railways, power line and communications corridors, and use of surrounding regions for recreation exert a greater influence on sagebrush habitats (Chapter 12).

The ecological impact of roads only recently has been recognized and quantified (Forman and Alexander 1998). Roads impact an estimated 15% of the United States (Forman 2000). The effects of roads include (1) increased mortality of wildlife from collisions with vehicles on roads, (2) modification of animal behavior because of habitat or noise disturbance, (3) alteration of physical environment, (4) alteration of chemical environment, (5) spread of exotic species, and (6) increased habitat alteration and use by humans (Trombulak and Frissell 2000). Unpaved roads fragment sagebrush landscapes as well as provide disturbed surfaces that facilitate spread of invasive plant species (Gelbard and Belnap 2003). Roads also may facilitate access for fire suppression and habitat treatments.

Railways were largely responsible for initial spread of cheatgrass in the intermountain region (Young and Sparks 2002). Cheatgrass readily invaded the disturbed soils adjacent to the railroads. Fires caused by trains also promoted expansion of cheatgrass. Finally, cattle transported along the rail system facilitated further spread of this species (Young and Longland 1996).

Powerlines and communications towers provide perches for corvids raptors (Steenhof et al. 1993, Knight and Kawashima 1993). In addition, the corridors and accompanying roads facilitate predator movements and spread of invasive plant species (Gelbard and Belnap 2003).

Current Status

The dominant urban areas in the Assessment study area were located in the Columbia River valley of Washington, the Snake River valley of southern Idaho, and the Bear River Valley of northern Utah. With the exception of cities in Nevada, most urban development was on the edge of regions dominated by sagebrush. Landfills associated with urban areas potentially facilitated corvid and predator movements across 22,650 km^2 (Fig. 7.15).

Interstates and major paved roads covered an estimated 14,272 km^2 throughout the Assessment study area. Interstates and major paved roads tend to follow river valleys at lower elevations. When a buffer of 10 km was included to account for an influence from predation and noise disturbance, the total area influenced by interstates and highways was 1,137,038 km^2 (Fig. 7.16). Secondary paved roads exist in almost all areas of the sagebrush biome (Fig. 7.17); density of secondary paved roads ranged to >2 km/km^2. In addition to area influenced by road corridors, rest areas associated with highways (Fig. 7.18) provide food supplies and perches for corvids and raptors and facilitate their movements into surrounding region (Chapter 12).

A minimum of 15,296 km^2 of lands contained large powerlines in the Assessment Area (Fig. 7.19). We were unable to map or estimate the density of smaller distribution lines in rural areas. Similar to roads, powerlines also tend follow major river valleys at lower elevations. The area influenced by additional perches for corvids and raptors was 672,344 to 837,390 km^2 (buffer size range from 5-6.9 km), or 32-40% of the sagebrush habitats. Railroads covered 137 km^2 of the landscape but influenced 183,915 km^2 (buffer 3 km) (Chapter 13). The railroad corridor of private land is evident in northern Nevada, northern Utah, and southern Wyoming (Fig. 5.20, Fig. 7.20).

We combined 3 categories of communications towers based on height and location relative to glide paths around airports for our assessment. A minimum of 9,510 communications towers >62 m in height were present in the Assessment study area (Fig. 7.21); the area potentially influenced was 99,135 km^2 when we buffered towers by the foraging radius of golden eagles (*Aquila chrysaetos*) (summary in Chapter 13).

Rural areas also have been developed throughout the sagebrush region because of economic factors combined with opportunities for recreation and other natural amenities (Riebsame et al. 1996). In addition, many "exurbanites" have migrated from cities into "ranchettes" created by subdividing larger ranches. Although ranchettes continue to provide some sagebrush habitat in contrast to total urban conversion, road fragmentation and disturbance from human dwellings and activities (Mitchell et al. 2002) probably render much of the area inhospitable to sage-grouse and other wildlife dependent on sagebrush habitats.

Recreation on lands managed by the U.S. Bureau of Land Management was a significant land use (Table 7.7). In addition, campgrounds used by hunters, anglers, and other recreation participants on or near shrublands may provide disturbance from human presence, roads, and facilitate predator movements.

Livestock Grazing

Background

Livestock grazing in western shrubland became a significant factor in the ecological and political landscape in the late 1800s (Young and Sparks 2002). Completion of the first transcontinental railroad in 1869 greatly expanded the livestock industry because products could be shipped to markets on east and west coasts. Early grazing was largely unregulated either by fences or a legal system and competition was intense among ranchers, homesteaders, and free-rangers as well as between cattle grazers and sheepherders (Donahue 1999).

The major period of expansion from 1880 to 1905 in numbers of livestock and areas grazed severely altered the condition of western landscapes (Mitchell and Hart 1987, Box 1990). The primary drought that followed the period of heavy grazing occurred in the 1920s and became severe the 1930's. The native perennial caespitose grasses in the Intermountain West lacked the seed production and morphological characteristics to sustain anything greater than a low level of grazing disturbance (Mack and Thompson 1982, Miller et al. 1994). Native grasses and forbs were depleted from the vegetation community and replaced in much of the Great Basin and surrounding region by exotic annual grasses (Robertson 1954, Young et al. 1972, Vale 1975, Yensen 1981, Miller et al. 1994). Loss of protective vegetation cover in some communities resulted in extensive soil disturbance and erosion (Cottam and Stewart 1940). Shrub density also increased although the total distribution of shrubs across the region likely remained similar (Vale 1975).

Large declines of native grasses and winterfat (*Krascheninnikovia lanata*) occurred between 1870 and 1900 (Cottam and Stewart 1940, Christensen and Johnson 1964). Hull (1976) 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 10% of the site potential following depletion of the vegetation community in some regions (Young et al. 1981). The area estimated to support a cow and calf (or equivalent) was estimated to 0.83 AUM/ha prior to settlement and unrestricted grazing, 0.27 AUM /ha in the 1930's, and 0.31 AUM/ha in the 1970s (West 1983).

Regulations enacted to restrict grazing began in the late 1800's to control the use of public lands but also to ensure that policies of multiple use were mandated. The Organic Act of 1897 (16 USC 473) gave the Forest Service the right to manage grazing on forest reserves. Subsequently, permits to graze a limited number of animals exclusively on tracts within the forest system for a fee were established. In 1934, the Taylor Grazing Act (43 USC 515) was passed to prevent overgrazing and damage to public lands (Chapter 1). The Taylor Grazing Act terminated open rangeland policies, established grazing districts, and required permits for use. The remaining vacant lands were closed to homesteading or withdrawals. More recently, the Classification and Multiple Use Act (43 USC 2420) and the Federal Land Policy and Management Act (43 USC 1701-1782) in 1976 directed that public lands were to be used for multiple use consistent with maintaining environmental standards (Chapter 1). The 1978 Public Rangelands Improvement Act (43 USC 1901-1908) provided for restoration of damaged lands, established a policy of inventory and monitoring, and required periodic reports on the conditions and trends of lands to the Secretaries of Interior and Agriculture. The Act also required that permit holders must graze the lands with 90% of the permitted animals or risk losing their permits. Under the proposed changes to grazing regulations, the U.S. Bureau of Land Management could approve temporary nonuse for no more than one year at a time but would remove restrictions on the number of consecutive years that nonuse could be approved for resource conservation or business/personal needs of the permittee (U.S. Bureau of Land Management 2003*a*).

Permits for grazing livestock use are based on an animal unit month (AUM) and is the amount of forage required to feed one 454 kg (1,000 lb) cow and her calf, one horse, five sheep, or five goats for one month. The current fee applied by the U.S. Bureau of Land Management and USDA Forest

Service for grazing on public lands in 16 western states is $1.43/AUM (effective 1 March 2004) and was $1.35 in 2003.

Treatments to restore the herbaceous understory, maximize forage production for livestock, reduce plants poisonous to livestock, stabilize soils, and reduce shrub cover were implemented following unrestricted grazing in the late 1800s and early 1900s. Treatments were designed to decrease unpalatable plants, such as sagebrush, to livestock even though those plants may be palatable to elk (*Cervus elaphus*) or sage-grouse and other species. Chemicals, ploughing, burning, and other methods were used to remove the competitive woody overstory, particularly sagebrush, and maximize forage production for livestock over large areas of the western states (Pechanec et al. 1965). Estimates of the total area treated vary and range from 10-12% of sagebrush range covering 400,000 km^2 (400,000 to 480,000 km^2) by the 1970s (Vale 1974, Pechanec et al. 1965), and 200,000-240,000 km^2 treated over a 30-year period (Schneegas 1967). Total area of sagebrush controlled on lands managed by the U.S. Bureau of Land Management was 180,000 km^2 between 1940-1994 (Miller and Eddleman 2001); during the peak period of sagebrush removal in the 1960s, 11,000 km^2 were treated.

The amount of sagebrush that could be removed from the landscape, soil characteristics, and the presence of a residual understory that could respond were primary factors in the success of a treatment for increasing forage (Vale 1974, Young et al. 1981, Cluff et al. 1983). Livestock control still was needed after treatment by herbicides to let perennial grasses recover (Young et al. 1981). Replanting areas after sagebrush was removed with nonnative grasses, such as crested wheatgrass (*Agropyron cristatum*), reduced the necessity of a pre-existing understory and was particularly successful at increasing in forage that could be grazed by livestock (Shane et al. 1983). The first reseedings in southern Idaho were conducted in 1932 (Hull 1974). The herbicide 2,4-D (used as a defoliant during World War II) was successful in killing sagebrush but was ineffective against root-sprouting shrubs such as rabbitbrush (*Chrysothamnus* spp.). Different combinations of herbicides and seasons of applications then were developed to remove sagebrush, other unwanted woody shrubs and weedy annuals (Tueller and Evans 1969, Evans and Young 1975, Evans and Young 1977).

More recently, thinning of sagebrush density by Tebuthrion rather than sagebrush removal from large areas, has been the focus of some treatments (Emmerich 1985, Olson and Whitson 2002). Replanting native plants (Richards et al. 1998) also has been emphasized although the use of nonnative species will continue because of seed availability, desirable growth response to achieve short-term objectives such as soil stabilization, and economics (Asay et al. 2001). Nonnatives may be necessary in some closed communities to gain an advantage over cheatgrass (Robertson and Pearce 1945) after which native plants can become established (Cox and Anderson 2004).

Large numbers of treatments continue on public lands across the sagebrush biome (Table 7.8). Treatments to manipulate sagebrush habitats are applied for different purposes and are funded under different programs. The Environmental Impact Statement on Vegetation Treatments in the Western 13 States addressed concerns about habitat treatments and concluded that "Treating vegetation is necessary to develop or restore a desired plant community, create biological diversity, increase forage or cover for animals, protect buildings and other facilities, manage fuels to reduce wildfire hazard, manage vegetation community structure, rejuvenate decadent vegetation, enhance forage//browse quality, or remove noxious weeds or poisonous plants" (U.S. Bureau of Land Management 1991). The preferred alternative was to treat >9,000 km^2 annually in the 13 western states, including 3,500 km^2 with chemical treatments and 2,420 km^2 by prescribed burning. Thus, the majority of lands covered by sagebrush will be used and managed for multiple purposes, including livestock grazing, and will be subjected to many forms of habitat manipulations. A new Environmental Impact Statement on vegetation treatments is being prepared by the U.S. Bureau of Land Management and scheduled for release in Fall 2004.

Ecological Influences and Pathways

The effect of livestock grazing is one of the most contentious issues underlying the management and use of sagebrush habitats (Brussard et al.1994, Noss 1994, Wambolt et al. 2002, Crawford et al. 2004). Livestock grazing is the most widespread (and perhaps most argued) land use across the sagebrush biome. Isolated areas exist, such as inaccessible tops of some buttes, kipukas, regions and in which natural or human-developed water sources are not available in which livestock have not grazed. On other lands now managed by the Department of Defense, Department of Energy, National Park Services, or National Wildlife Refuges, livestock may have grazed earlier but now have been restricted (e.g., Anderson and Holte 1981). However, most sagebrush habitats have been grazed in the past century (Saab et al. 1995, West 1996, West and Young 2000, Hockett 2002).

Opinions and supporting evidence on the effects of livestock grazing are separated across a chasm from a viewpoint of completely compensatory or beneficial influence on sagebrush habitats on one side to a total destructive force that should be removed immediately on the other. We will not resolve that controversy in this assessment. These arguments ignore the complexity of sagebrush systems across the sagebrush biome and the resulting differences in ability of sagebrush habitats to sustain and respond to disturbance. Rather, we will address the available information and impediments to answering the questions of the effects of livestock grazing on sagebrush habitats and populations of sage-grouse. Our assessment is based on the interaction of the form of grazing disturbance and its effects, management actions, perceptions and assumptions about the dynamics of sagebrush ecosystems, and the spatial and temporal scale underlying the conclusions.

Livestock grazing is a "press" or diffuse form of biotic disturbance that exerts repeated pressure over many years on a system (Bender et al. 1984, Pickett and White 1985, Turner and Bratton 1987). Unlike point-sources of disturbance, such as 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. Therefore, effects are not likely to be detected as disruptions (except in extreme disturbance) but rather as differences in the processes and functioning of the sagebrush system.

Dominant trends emerged along a complex gradient of shrubland disturbance regimes following intensive grazing pressure and drought (West and Young 2000). In northern, eastern, and more mesic regions of the sagebrush biome, fire was not an important disturbance necessary to maintain perennial grasses and forbs. Rather, grazing by buffalo was the primary agent of grazing disturbance. In those regions, introduction of domestic livestock increased the site-specific frequency of grazing (Mack and Thompson 1982). Improper grazing that depleted the grass and forb understory facilitated invasions by exotic plants species because of loss of understory, altered soils, or loss of microbiotic crusts in these systems (Mack and Thompson 1982). In more xeric and southern sagebrush regions, possible increases in shrub cover that resulted from grazing further reduced the low cover of native perennials and created a system largely resistant to recolonization by the native flora but increasingly vulnerable to invasion by exotic plants (Young and Sparks 2002).

Sagebrush communities in which fire was a dominant disturbance also became relatively fire-proofed, because the understory biomass to spread fire was lacking (although fire can spread in these habitats characterized by a highly dense shrub cover under dry, hot, and high wind conditions). Fire kills most species of sagebrush plants and creates an opening in the landscape and releases nutrients after a burn. An uneven distribution of soils, micro- and macrotopography, environmental and moisture conditions, and biomass resulted in a mosaic of openings and uneven-aged stand of sagebrush in a landscape. The presence of perennial grass understory and intact microbiotic crusts protecting the soils also were primary components of the community response following burns. The intensity and season of grazing in sagebrush habitats that resulted in depleted understories and disrupted microbiotic crusts reduced the resiliency of these systems. In a synergistic feedback mechanism, the interaction of livestock grazing, loss of understory and altered soil characteristics, and reduced fire frequency resulted increased shrub cover and dominance.

Vegetation communities invaded by cheatgrass represent the another extreme in disturbance. Cheatgrass had invaded much of the Intermountain West by the 1930s after its initial establishment in the 1890s (Mack 1981). Cheatgrass was well adapted to invade areas depleted of native herbaceous vegetation and that lacked the protective cover of biological soil crusts, lichens, and mosses between bunchgrasses as a result of soil trampling (Mack 1981, Young and Allen 1997). Because of ecological and morphological characteristics, cheatgrass can out compete native perennial plants and promote fire (Klemmedson and Smith 1964). The positive feedback cycle of fire, sagebrush loss, and cheatgrass dominance has resulted in entire landscapes now converted to annual grasslands (d'Antonio and Vitousek 1992).

The pattern and influence of livestock grazing in sagebrush habitats is different from the system in which the plants evolved over the past 10-12,000 years before present (BP). Much of the western sagebrush biome (the *Agropyron* region) has had a long period in which large hoofed grazers were rare. Large herbivores became extinct at end of Pleistocene (10,000 – 12,000 years BP) and the American bison (*Bison bison*) largely withdrew its distribution (Mack and Thompson 1982, Billings 1990) but small numbers still ranged in some parts of the Great Basin region and western Montana and were relatively common in eastern Idaho prior to European settlement. In the eastern sagebrush steppe region (the *Bouteloua* region), grazing by bison was locally intense but highly variable in space and time. In each general system, the plant communities developed to optimize the water and growing periods. Even though management plans attempt to use rest-rotation or other forms of variable grazing intensities to mimic the previous natural grazing regimes and vegetation response, the plant communities still are not given the rest from grazing and recycling of resources is dissimilar (Bock et al. 1993, Freilich et al. 2003).

Livestock grazing can affect soils, vegetation, and animal communities (Jones 2000). Livestock consume or alter vegetation, redistribute nutrients and plant seeds, trample soils and sagebrush plants, and can disrupt microbiotic crusts (Miller et al. 1994, West 1996, Belnap and Lange 2001). The extent to which these mechanisms influence habitats depends on the relationship between level of grazing disturbance and the resiliency of the habitat. At unsustainable levels of grazing, these changes can lead to loss of vegetative cover, reduced water infiltration rates, and increased soil erosion (Society for Range Management 1995). Indirect effects of livestock grazing can amplify or facilitate other disturbance. For example, landscapes in southwestern Idaho in which grazing combined with other disturbances experienced the greatest rate of shrub loss and increase in cheatgrass compared to landscapes having a single source of disturbance (Knick and Rotenberry 1997). In shrublands dominated by cheatgrass, grazing can reduce further the remaining perennial grasses or permit excessive growth of cheatgrass if left ungrazed (Young and Allen 1997).

Management of livestock grazing influences ecosystems by actions designed to control or protect livestock or to increase forage availability or foraging conditions (West 1996). Livestock movements are managed by development of water sources, and building fences and roads. Predators are controlled by lethal means or by physical barriers to their movements. Habitat manipulations to increase forage include chemical and mechanical treatments to remove sagebrush followed by reseeding with nonnative plants. These habitat manipulations alter the natural food web, influence fire and other disturbance regimes, change the nutrient dynamics, and affect the vegetation structure used by wildlife (Freilich et al. 2003).

Stocking rates for livestock are based on assumptions of an average set of conditions, estimates about the size of the area necessary to produce enough food to sustain livestock, and the relationship between the current vegetation community and an ideal condition, seral stage, or climax community (Box 1990, Society for Range Management 1995, Holechek et al. 1998). Actual number of livestock that may be grazed is determined by season of use, the distribution, utilization, and actual use of available forage, and the class of livestock. Average conditions rarely exist and western landscapes have experienced extensive drought periods (Fig. 7.11). Stocking rates are based on livestock production and financial livelihood of grazing permittees in addition to environmental considerations (Holechek et al. 1999). In the natural world, the number of herbivores is constrained by food supply. Because the relationship between food supply and numbers of livestock is buffered either by administrative, management, or economic factors, the time lag in changing the numbers of livestock in response to changes in habitat conditions can increase the effects of grazing (Thurow and Taylor 1999).

The definition of “range condition” is widely debated (Friedel 1991, Joyce 1993, Schacht 1993, Scarnecchia 1995, Society for Range Management 1995). The set of environmental qualities that form the conceptual foundation of “range condition” influence not only our understanding of the effects of grazing but also our management actions to manipulate sagebrush habitats (West 2003).

Grazing and management objectives are based on an assumed predictable or linear response to disturbance (Allen-Diaz and Bartolome 1998). Grazing often has been assumed to represent a disturbance that sets a sagebrush community back from a climax stage; release from grazing then will allow the community to return to a climax stage. Under this Clementsian viewpoint of successional stages in vegetation communities, potential natural communities were considered to be in excellent condition, late seral to be good condition, mid-seral to be fair, and early seral in poor range condition. Yet, the dynamics of sagebrush communities are complex and plant species response to grazing may not be correlated closely (Milchunas and Lauenroth 1993, West and Young 2000). A wide range of shrub-grass compositions can be stable, length of time within seral stages may be nonlinear, and transitions among different states may be unpredictable (West et al. 1984, Westoby et al. 1989, Laycock 1991). More important, transitions among vegetation communities may be irreversible and return to a previous state is not possible once it is crossed. Consequently, livestock herbivory can alter vegetation communities, water and nutrient availability, and soils past thresholds to which the system can return.

Stocking rates and estimates of the amount of vegetation that can be removed based on differences between a current vegetation community and an ideal or climax seral stage may not be sustainable (Holecheck et al. 1998). Conversely, release from grazing may have no or unpredictable results (Anderson and Holte 1981, West et al. 1984, Stohlgren et al. 1999, West and Yorks 2002), may not return a site to a previous state (Holechek and Stephenson 1983) or, exacerbate the influence of exotic plants such as cheatgrass (Young and Allen 1997). Therefore, 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).

Early habitat treatments were directed primarily to "improve" forage conditions said to benefit livestock (Pechanec et al. 1965). Increasingly, treatments are conducted to restore hydrologic processes, wildlife habitat, stabilize and rehabilitate soils and vegetation, or reduce biomass to control fires and protect urban interface areas. More recently, objectives for land management have been set by a society interested in preserving wildland, wildlife, and aesthetic components of sagebrush habitats (Young et al. 1981, Box 1990, West 1996, West 2003). Each choice and habitat treatment has consequences for future habitat dynamics and wildlife use of sagebrush habitats because of changes in the quantity of available habitat, its composition, and configuration within the larger landscape. Treatment of large areas, use of herbicides, mechanical treatments, or planting nonnative plant species may be appropriate management tools to control exotic plants, reduce fire hazards, or rehabilitate burned areas (see Restoration and Rehabilitation below). Each potentially decreases the suitability of sagebrush habitats for wildlife that depend on large, unfragmented sagebrush habitats (Knick et al. 2003). With few exceptions, monitoring vegetation and wildlife response to habitat treatments across appropriate spatial and temporal scales is lacking (Crawford et al. 2004).

Current Status

We do not have reliable numbers of livestock, spatial and temporal information on grazing and intensity, and corresponding habitat characteristics to assess grazing effects for a large-scale analysis across the sagebrush biome. Management of livestock grazing and assessment of habitat condition is site-specific rather than at large-scales or landscapes (Mitchell 2000), yet many processes that govern sagebrush habitats are large-scale (Anderson and Inouye 2001). Almost all research of livestock grazing is short-term or small-scale; synthesis of results in a meta-analysis have emphasized the difficulty of drawing broad generalizations about the effects of grazing from site-based information (Milchunas and Lauenroth 1993). Scaling those results or data to larger spatial or temporal regions may not be possible because of differences in the underlying processes (Allen and Starr 1982).

The Society for Range Management (1995) stated, "Current range condition assessment methods do not provide answers to the questions that Congress and the public wants answered about the status of our rangelands." Current efforts to evaluate rangeland status and to recognize that multiple conditions may be present on sites, the nonlinearity of habitat response, and development of site condition thresholds have yet to be implemented on a spatial and temporal scale that will permit answers to (1) how different are vegetation communities now than they would be in the absence of grazing, and (2) would the expected change in the vegetation community in the absence of grazing benefit sage-grouse or other species dependent on sagebrush habitats?

Approximately 12,700,000 permitted AUMs/year were distributed on public lands in the western states from 2000 to 2002 (Table 7.9). Number of permitted AUMs declined slowly from 14,211,000 in 1965, but has remained relatively stable since 1975 (12,504,000 permitted AUMs) (Mitchell 2000; U.S. Bureau of Land Management Public Land Statistics). AUMs represent permitted use but actual use varies because of economics, non-use due to forage or drought conditions, and unreported trespass. AUMs reflect only a general area in which permitted livestock were grazed and not season of use. In addition, AUMs do not distinguish between forage use by cattle and sheep, which have different effects on vegetation.

Changes in permitted use (AUMs) does not necessarily correlate to a changes in effect of livestock grazing on a particular area within an allotment due to the uneven distribution of use. Pastures can be understocked but select portions still can be overgrazed. The management regime of livestock grazing may have as great an effect on vegetation communities as numbers of livestock estimated from annual statistics on AUMs.

Productivity of western shrublands has declined due to previous grazing history or drought (West 1983, Holechek and Stephenson 1998). The distribution of livestock also has changed because water developments have increased the area that could be grazed. We cannot conclude that the effect of grazing has been reduced because even reduced numbers of livestock may still exert a larger influence on those habitats. Therefore, the absence of information on management coupled with vegetation changes (e.g., Yorks et al. 1992) limits our understanding of the effect of livestock grazing on long-term dynamics of sagebrush systems.

The percent of public lands that have an improved "range condition" has increased since the 1900's (Society for Range Management 1989, Box 1990). However, changes in defining and measuring "range condition", and the percent of lands that remain unsurveyed (Table 7.10), limit our ability to quantitatively assess the relationship between livestock numbers and habitat. In the past, range condition was primarily focused on forage production rather than ecological condition of the site (West 2003). In addition, surveys are not be updated every year for which statistics are reported (which has significant implications for proposed grazing regulations and the monitoring information and feedback that would be required when before taking action [U.S. Bureau of Land Management 2003*a*]). Qualifiers, such as "moderate", or "heavy" utilization rates are based on how much vegetation has been removed, and often are subjective and variable across sites (Frost et al. 1994). For sage-grouse, "improved" condition may not necessarily provide required components of sagebrush habitats (Connelly et al. 2000*a*); the areas in which habitats changed may not be used by sage-grouse; and many of the relationships among livestock, habitat, and sage-grouse are not known (Crawford et al. 2004).

The status of lands (rangeland health) administered by the U.S. Bureau of Land Management is defined as "the degree to which the integrity of the soil and ecological processes of rangeland ecosystems are sustained" (National Research Council 1994). Indicators of soil characteristics, erosion rates, plant composition and biomass, and underlying community processes are integrated into a collective assessment of the functioning of grazing lands rather than an assessment of the degree of departure of the vegetation community relative to a potential or seral stage. Standards and guidelines for management of public grazing lands are established by local resource advisory councils and must address habitats and conservation measures for "endangered, threatened, proposed, candidate, or other at-risk or special status species" (U.S. Bureau of Land Management 2003*b*). Under this set of criteria for rangeland health, 58% of lands that have been assessed (25% of all lands under management by the U.S. Bureau of Land Management) (including nonsagebrush habitats) met the standards or were making progress towards meeting those standards (Table 7.11). Livestock were a factor in 36% of the assessed lands not meeting standards (15% of the all lands). Another 6% of the assessed lands were not meeting standards for causes other than livestock grazing. Fifty-seven percent (>37 million ha) of the public lands managed by the U.S. Bureau of Land Management have not been assessed.

The primary habitat treatments on lands managed by the U.S. Bureau of Land Management include construction of fences, development or control of water, and habitat modifications (Table 7.8). More than 1,000 km of fences were constructed each year from 1996 through 2002; most fences were constructed in Montana, Nevada, Oregon, and Wyoming. Linear density of fences exceeds 2 km/km^2 in some regions of the Conservation Assessment study area (Fig. 7.22; Chapter 13 presents additional information on fences and sage-grouse). In addition to influencing livestock and predator movements, facilitating spread of exotic plants, and providing additional travel and access for human activities, fences potentially increase mortality of sage-grouse due to direct collisions or indirectly by increasing predation rates by increasing the number of perches for raptors. Fences used to control grazing management among allotments further modify the landscape by creating an artificial mosaic among separate pastures (Freilich et al. 2003).

Water developments were widespread throughout public lands (Fig. 7.23). Water developments and distribution of water sources substantially influence movements and distribution of livestock in arid western habitats (Valentine 1947, Freilich et al. 2003). Consequently, grazing pressure can be unevenly distributed and influence the composition and relative abundance of the plant relative to water sources.

Areas seeded to improve wildlife habitat ranged from 1,673 km^2 in 2000 to 55 km^2 in 1998 (Table 7.12). Most seeding was done in Nevada in response to the large areas burned. From 1997 through 2002, 376 km^2 were treated by mechanical means to improve wildlife habitat (Table 7.13).

Prescribed Fire

Use of prescribed fire is one of the most common yet most contentious issues in management of sagebrush habitats (Miller and Eddleman 2001, Wambolt et al. 2002). Prescribed fire is used to control annual grasses, reduce density of sagebrush, facilitate growth of grasses and forbs, and control juniper and pinyon woodland expansion into sagebrush habitats. Recovery of burned shrubland is a function of size of the fire, fire frequency, and availability of seed sources. Site specific variables including soil characteristics, climate, previous disturbance history, and presence of livestock grazing also influence the trajectory of vegetation recovery following burns (West and Yorks 2002). Because of this complexity, the dynamics of recovery following burns in sagebrush-dominated landscapes may differ between natural and prescribed fires because of differences in season and intensity of burns.

Recovery from burns, particularly in more xeric sagebrush ecosystems, is long-term and may require centuries or longer (Hemstrom et al. 2002). From 3 to >30 years may be required for seeds to grow seed to mature plants depending on the timing and amount of precipitation. The majority of seed dispersal occurs within 9-12 m of the parent plant (Blaisdell 1953, Mueggler 1956, Johnson and Payne 1968, Daubenmire 1975, Frischnecht 1978). In addition, the short-lived viability of sagebrush seed in the soil further imposes limits on regeneration if unfavorable weather conditions follow a burn.

Prescribed fires were conducted on 2,878 km^2 of public lands from 1997 through 2002 (Table 7.14). Most areas treated by prescribed fires were in Oregon and Idaho. In addition to prescribed fires, the area on which nonfire fuels treatments were conducted increased from 66 km^2 in 1999 to 832 km^2 in 2002 (Table 7.15). Emergency fire rehabilitation was a major activity that varied in expenditures from $2.6 million in 1996 to $78.1 million in 1999; area treated also varied from 281 km^2 in 1997 to 16,135 km^2 in 2002 (Table 7.16). The majority of the areas treated were reseeded with shrubs and forbs following extensive fires in Idaho, Nevada, and Oregon.

Sage-grouse and Fire

Several investigators have suggested that fire may benefit sage-grouse by enhancing nesting and brood-rearing habits (Klebenow 1973, Sime 1991, Pyle and Crawford 1996). Researchers have claimed that fire can be used to maintain a balance of shrubs, forbs, and grasses at various scales throughout the landscape. Additionally, burns may be conducted to limit the expansion of juniper and pinyon and reduce perches-sites for raptors and corvids.

Despite claims to the contrary, there is virtually no empirical evidence that prescribed fires have benefitted sage-grouse. Short-term increases in forb production have been documented following fires (Harniss and Murray 1973; Martin 1990; Pyle and Crawford 1996) but these findings were not related to sage-grouse population characteristics. In contrast, several studies have documented negative effects of fire on sage-grouse populations (Connelly et al. 2000*b*, Byrne 2002, Pedersen et al. 2003) and habitats (Fischer et al. 1996; Nelle et al. 2000). Connelly et al. (2000*b*) documented a significant decline for a sage-grouse breeding population following a prescribed fire and Byrne (2002) reported that greater sage-grouse avoided burns that were <20 years old.

Forb response following a fire is a function of pre-burn site condition and precipitation patterns. The overall effects of fire on sage-grouse habitat will generally be dependent on site potential, site condition, functional plant groups, and burn pattern and size (Miller and Eddleman 2001). Because recovery of sagebrush canopy cover to pre-burn levels may require 20 years or longer, short-term benefits such as increased forb production may not balance the loss of sagebrush canopy required by sage-grouse during the nesting season and winter (Fischer et al. 1996, Connelly et al. 2000*a*, Nelle et al. 2000). Thus, fire may have limited usefulness as a routine tool for managing for sage-grouse habitats in sagebrush communities and decisions to use fire for managing sage-grouse habitat must be made cautiously and on a site-by-site basis.

Wild Ungulate Browsing

Wild Horses and Burros

Free-roaming horses (*Equus caballus*) and asses ('burros'; *E. asinus*) have been a component in the dynamics of sagebrush and other semiarid communities since they were brought to North America at the end of the 16th century (McKnight 1958, Ryden 1978, Wagner 1983, Beever 2003). In addition to occupying public lands administered by the U.S. Bureau of Land Management, free-roaming horses and burros occupy lands under diverse federal and state jurisdictions (e.g., U.S. National Park Service, U.S. Forest Service, U.S. Fish and Wildlife Service, and lands managed by state agencies), although they are being removed from some U.S. National Park Service units. We summarize: (1) their number and distribution in the assessment area, (2) their relationship with AUMs of domestic livestock; and (3) the ways in which they can influence the structure, composition, and function of sagebrush ecosystems. Because of disparity in abundance between horses and burros, much of our discussion focused on horses.

In contrast to their more extensive distribution and greater abundance in recent decades (McKnight 1958), an estimated 5,041 burros occurred in five states (Arizona, Nevada, California, Utah, and Oregon) as of 2003 (Table 7.17). Of this total, only an estimated 691 animals occurred within the Conservation Assessment area (compare Figs. 7.24 and Fig. 5.2). Many of these animals occupy communities other than sagebrush (e.g., salt-scrub) within the landscape mosaic.

Approximately 40,000 free-roaming horses occur in ten western U.S. states (Table 7.17), and >90% occurs within the Assessment area. Herd sizes are largest in regions where sagebrush cover is most extensive, namely Nevada, Wyoming, and Oregon (Table 7.17; Fig. 7.24, 5.2).

The ratio of livestock to equid AUMs across the range of wild horses and burros averaged 23:1 in 1982 (Wagner 1983). Within Nevada, the state with over half of the continent's free-roaming equids, this ratio averaged 4.8:1 livestock to equid AUMS and ranged from 23:1 in the Elko District to 2:1 in the Las Vegas District (Wagner 1983). Introduction of burros and especially horses to western ecosystems had the effect that fewer areas were not grazed by non-native herbivores; horses often segregate elevationally from (i.e., above) and use steeper slopes than cattle (Pellegrini 1971, Ganskopp and Vavra 1986). Because of physiological differences, a horse consumes 20-65% more forage than would a cow of equivalent body mass (Hanley 1982, Wagner 1983, Menard et al. 2002). Horses also represent a grazing disturbance in sagebrush ecosystems comparable neither to cattle nor native ungulates (Beever 2003) because of their non-uniform use of the landscape, as well as their management status (horses are neither hunted nor fenced).

Horse-occupied areas exhibited lower cover of grasses, shrubs, and total vegetative cover, as well as lower species richness at sites, a less contiguous shrub canopy (and lower sagebrush cover), and, at higher elevations only, greater forb cover (Beever 1999, Beever et al. 2003). Alterations of spring or other mesic areas (Beever and Brussard 2000) may be of particular concern for sage-grouse management, especially during brooding.

Deer and Elk

Sagebrush is very important forage for mule deer in many parts of the assessment area (Wambolt 1996) and as such it may suffer damage from browsing. Although research on the topic is limited, there is some evidence that mule deer population changes in portions of the assessment area may be related to declining sagebrush densities. In Utah, Smith (1949) and Austin et al. (1986) showed declines in sagebrush density in areas protected from livestock grazing but open to mule deer browsing. Similarly, McArthur et al. (1988) reported lower densities of mountain big sagebrush in locations exposed to mule deer during heavy snowfall years when little other vegetation was available, while densities of sagebrush were higher in locations protected from mule deer browsing.

Wambolt (1996) observed that deer and elk in the northern Yellowstone area preferred mountain big sagebrush over Wyoming big sagebrush, basin big sagebrush and black sagebrush. Heavy browsing damaged or even killed some sagebrush. Patten (1993) concluded that elk browsing reduced sagebrush in the same area. Laycock (1967) has observed mortality of three-tip sagebrush with heavy browsing. The big sagebrush subspecies tend to be susceptible to heavy browsing because they lack the ability to activate buds on branches that exceed 1 year of age (Bilbrough and Richards 1992).

Declines of big sagebrush densities due to heavy deer or elk browsing (Smith 1949, Austin et al. 1986, McArthur et al. 1988, Patten 1993, Wambolt 1996) suggest that, if warranted, ungulate browsing may be used as a biocontrol for reducing the densities of sagebrush. Bork et al. (1998), Laycock (1967) and Mueggler (1950) have shown long-term declines of three-tip sagebrush from livestock browsing with recovery of herbaceous vegetation on high elevation sites in Idaho, suggesting that livestock may also serve a similar function. Although browsing of shrubs may increase the herbaceous component, this response needs further study in big sagebrush communities because it has only been noted in threetip sagebrush communities.

Nonrenewable Energy Development

Background

Oil and gas development in habitats used by sage-grouse and construction of accompanying powerlines, roads, and pipelines began in the late 1800s with the discovery of oil in the Interior West. Oil was the major resource developed in the Interior West until the 1960s. Across the primary oil and gas regions, development began in the 1880s in Wyoming, the 1920s in Colorado, and 1940s in Alberta (Braun et al. 2002). Since the 1960's, development of natural gas resources has dominated the industry. Most oil and gas resources have been developed in 17 geologic basins that overlap the sagebrush biome (Fig. 7.25). The production of natural gas from the central Rocky Mountain area has in the past been constrained by lack of transportation facilities (Doelger and Barlow 1989). However, additional interstate pipelines have been constructed to transport natural gas to California and to Midwestern markets during the past 20 years.

The Energy Policy and Conservation Act (Chapter 1, Table 1.2), signed into law in 1975 and reauthorized in 2000, was passed to regulate domestic energy consumption, establish a Strategic Petroleum Reserve, and inventory onshore oil and natural gas reserves. The 2000 reauthorization of the act authorized the re-inventory of Federal oil and gas reserves and directed a study of the extent and nature of any restrictions or impediments to the development of oil and gas resources. Phase I of the inventory and impediments study was conducted in five geologic basins: Uinta-Piceance of Colorado and Utah, southwestern Wyoming (Greater Green River Basin), San Juan Basin of New Mexico and Colorado, Montana Thrust Belt, and the Powder River Basin of Wyoming and Montana (Fig. 7.25) (U.S. Departments of Interior, Agriculture, and Energy 2003). These basins contain most of the onshore natural gas and much of the oil under Federal ownership within the 48 contiguous states. Additional Federal lands and resources remain to be inventoried in the ongoing study.

Intensity and location of development depends on energy demands and supplies, economic costs/benefit for extraction, and technological advances for extraction. Presently, the Interior West and the basins within this area support much of the oil and onshore natural gas under Federal stewardship within the 48 contiguous states (U.S. Departments of Interior, Agriculture, and Energy 2003). Oil production and exploration in the Rocky Mountain region is expected to remain constant or slightly decrease but will be focused on five basins to meet the energy needs of the United States (National Petroleum Council 2003). Natural gas development within those basins is expected to increase for the next 15 to 20 years. Within the Greater Green River Basin, development of natural gas is expected to increase by 40% by 2015.

Development of oil and gas resources requires construction of well pads and access roads, subsequent drilling and extraction, and transport of oil and gas. Before drilling a well on public lands, a lessee must file an application for permit to drill. An application for permit to drill contains information on well specifics, emergency procedures, protection of groundwater and other resources, mitigation to wildlife and archeological resources, and weed control measures. Upon receipt by the U.S. Bureau of Land Management, the application is reviewed by the administering field office prior to approval to determine the application's adequacy for operational and environmental provisions. Specialists involved may include geologists, petroleum engineers, surface reclamation specialists, biologists, archeologists, and others. The reviews may recommend relocation of proposed wells or other modifications to mitigate impacts. Conditions of approval, including mitigation measures, are attached to every permit. On state or private lands the lessee/owner must apply to the state's oil and gas agency or board for the development of oil and gas resources.

The majority of applications filed for permits to drill were approved (Table 7.18). Of 122,496 applications filed from 1929 to 2004, 95.7% were authorized, 3% are pending, 1.2% were withdrawn, and <0.1% have been rejected (Table 7.18).

Well pads vary in size from 0.10 ha for coalbed natural gas wells on flat ground in the Powder River Basin (U.S. Bureau of Land Management 2003*c*) developing reservoirs 600 m below ground to >7 ha for deep gas (7,000 m below ground) in the Madden Field of the Wind River Basin (U.S. Bureau of Land Management 2000*a*). Typical well pads are currently 0.8 – 1.9 ha for extracting oil and gas from reservoirs that are 4,300 m below ground and >1.9 ha for development of reservoirs >5,500 m below the ground surface. Pads for compressor stations, located along pipelines require 5-7 ha.

Well density and spacing are critical components in developing oil and gas reservoirs; their placement ensures that each well will sufficiently recover oil and gas from an allocated area. Spacing of oil and gas wells is the responsibility of each state's oil and gas agency or board, except on Native American lands, which are administered by the U.S. Bureau of Land Management. Oil and gas companies along with mineral estate managers in the U.S. Bureau of Land Management provide input on spacing requirements to the state oil and gas agency or board during the development phase of oil or gas fields. In most cases the mineral estate developer will submit a request to the state oil and gas agency or board to decrease well spacing to more efficiently drain each reservoir. There is standard set spacing based on geologic basins unless spacing is exempted after petition to the state oil and gas agency or board. Generally, well spacing decisions define the minimum number of wells per acre (5, 10, 20, 40, 80, 160, 320, 640 acres; or metric equivalent 2, 4, 8, 16, 32, 65, 130 ha). Although rare, no limits on spacing (e.g., spacing exemptions) are possible in some fields. Where one oil and gas reservoir lies below another, multiple wells may be drilled within the same spacing unit.

On lands administered by the U.S. Bureau of Land Management, well density is determined by land use plans and by accepting and applying the spacing decisions of individual state oil and gas agencies and boards. Land use plans include a scenario of foreseeable development and a corresponding resource analysis to forecast future impacts. Appropriate mitigation is identified and implemented at the leasing or the application for permit to drill stage. Mitigation may include the specific location of wells, access roads, and pipelines, depending on the impacts of a resource. The well is then appropriately situated within the spacing unit to protect correlative lease rights. If no accommodation can be made, then alternatives, such as directional or extended reach drilling or duel completions, are considered.

Development fields may expand slightly as they mature but development continues within the field. For example, a given field may have an initial spacing of 65 ha/well that decreases to 32 ha/well as the field matures.

Ecological Influences and Pathways

Development of oil and gas resources includes direct loss of habitat for well pads, roads, and pipelines connecting well locations. A typical well has a well pad and an access road. Depending upon the individual oil and gas fields, wells may also have pump jacks, separators, storage tanks, electrical lines, produced water ponds/pits or water discharge pipelines, associated with development. In addition, ancillary development for flow lines, other roads, compressor stations, pumping stations, and electrical facilities are necessary to develop a field (Gerding 1986).

Roads constructed to connect well pads were typically 4-7-m wide, not including drainage ditches (U.S. Bureau of Land Management and U.S. Forest Service 1989). Construction of new access roads averages 7 days of heavy equipment/1.6 km of road constructed. Vehicle traffic and noise disturbance on roads and at well sites was highest during drilling phase. Female sage-grouse moved greater distances from leks and had lower rates of nest initiation in areas disturbed by vehicle traffic (1-12 vehicles/day) (Lyon and Anderson 2003). Surface disturbance created by roads also facilitates spread of exotic plant species (Forman and Alexander 1998, Trombulak and Frissell 2000, Gelbard and Belnap 2003).

Construction of a well pad and drilling requires heavy equipment (>5 tons) and can generate intense noise and exhaust. The U.S. Bureau of Land Management generally does not regulate aboveground noise to mitigate effects on sage-grouse and other wildlife from compressors, traffic, drilling rigs, and pumping units. As an example, six U.S. Bureau of Land Management field offices occur in the Greater Green River geologic basin in Wyoming and Colorado. Of these six primary field offices, no current U.S. Bureau of Land Management land use plans address noise as it relates to sage-grouse or wildlife (U.S. Bureau of Land Management 1985*a*, 1985*b*, 1986, 1988*a*, 1988*b*, 1996). One environmental impact statement for the Pinedale Anticline oil and natural gas field has specifically addressed noise issues and limited noise relating to sage-grouse or wildlife considerations (U.S. Bureau of Land Management 2000*b*). Disturbances within 200 m of lek sites resulted in loss of attendance at sage-grouse leks (Braun et al. 2002). Sage-grouse continued to use highly fragmented habitats in some oil fields and reclaimed areas, but population levels were below numbers prior to disturbance (Braun et al. 2002).

Pipelines to transport oil and gas range from 5-15 cm diameter for flow lines that may be buried. Trunk lines (15-20 cm) and transmission lines (25-66 cm) are buried. Width of the surface area required to construct lines ranges from 0.4 m for smaller diameter pipes to >23 m for the larger transmission lines.

Stipulations regulate timing of construction activities and are largely directed to avoid disturbance to sage-grouse near leks during breeding or nesting periods. However, effects on habitats from roads, powerlines, compressor stations, and pipelines remain following construction. The expected economic production life of coalbed natural gas wells is 12-18 years with advanced technology and 20 - 100 years for oil and gas wells.

Powerlines create perches and nesting platforms for raptors and corvids (Knight and Kawashima 1993, Steenhof et al. 1993). An estimated 9,656 km of overhead power lines have been developed for coalbed natural gas production in Powder River Basin and another 8,046 km are expected with continued development over the next 10 years (Braun et al. 2002).

The soil surface of disturbed areas is prepared and reseeded. The primary objectives of the reclamation are to control soil erosion, establish desirable vegetation and prepare for natural processes to restore the site. Although sage-grouse may repopulate reclaimed areas, their numbers may not return to levels prior to disturbance (Braun et al. 2002).

Current Status

Annual demand for petroleum will increase annually by 1.4% and natural gas consumption will increase annually by 2.3% between now and 2020 (National Petroleum Council 1999, U.S. Dept. Energy 2004*a*); consumption of natural gas is projected to increase from 22.6 trillion cubic feet in 2002 to an estimated range between 29.1 and 34.2 trillion cubic feet in 2025 (U.S. Department of Energy 2004*b*). The United States National Energy Policy projected an increase in oil consumption by 33%, in natural gas consumption by >50%, and in electricity by 45% over the next 20 years (National Energy Policy Development Group 2001).

Five geologic basins contain most of the significant onshore oil and gas reserves in the United States (Table 7.19) (U.S. Departments of Interior, Agriculture, and Energy 2003). Individual oil and gas wells within all producing regions were located primarily in sagebrush-dominated landscapes (Fig. 7.26-29; Fig. 5.2) (Knick et al. 2003).

Stipulations to mitigate the effects of oil and gas development on wildlife and habitats have been developed. Stipulations on 14.6% of the federal lands restrict the timing of construction activities area intended to reduce disturbance during periods critical to sage-grouse, raptors, pronghorn antelope (*Antilocapra americana*), mule deer (*Odocoileus hemionus*), elk, or other wildlife. Exceptions to the stipulations can be granted; rates of exceptions granted vary by U.S. Bureau of Land Management Field Office. However, habitat effects persist once the well and access roads are constructed. Leasing is not permitted because of administrative restrictions on 36% of the federal lands within the 5 major geological basins; leasing under standard terms is permitted on 39% (Table 7.20).

At least 3,497 wells have been approved in the Greater Green River Basin (Table 7.21) but a potential of 9,697 wells is being considered. In the Powder River Basin, 15,811 wells have been approved and an additional 65,635 are being considered to potentially develop the reservoirs (Table 7.22). In the Uinta/Piceance Basin, 3,501 wells have been drilled and an additional 2,597 are potential (Table 7.23).

We overlaid the locations of oil and natural gas wells over land stewardship status to determine the proportion of wells on public and private lands within the 5 primary geologic basins. The proportion of wells on private lands (total number of producing, pending, abandoned, and unknown status) was 50% (36,329) in the Paradox-San Juan Basin, 42% (20,795) in the Uinta-Piceance Basin, 33% (19,034) in the Greater Green River Basin, 74% (67,668) in the Powder River Basin, and 77% (220) in the Montana Thrust Belt. Development on private lands does not require mitigation efforts. Increased development on private lands potentially shifts the importance of remaining sagebrush habitats towards those in public stewardship.

We mapped natural gas and oil pipelines to determine potential area influenced (Fig. 7.30). The physical loss of habitats directly attributed to well pads and pipelines was a minimum of 4,749 km^2. These features were buffered (1-km and 3-km) to consider predation, spread of exotic plant species, and noise disturbance. Including these areas, oil and gas development influenced a minimum of 500,276 km^2 (25%) of the total area within the Conservation Assessment study area. Oil and natural gas well pads, pipelines, and roads influenced 28% of the sagebrush habitats within the Conservation Assessment study area.

Renewable Energy Sources - Wind Energy

Background

The National Energy Policy established in 2001 encouraged the development of renewable energy sources (National Energy Policy Development Group 2001). Renewable energy sources include wind, solar, and geothermal energy sources. Thus, we also considered the influence of wind energy development on sagebrush habitats. Federal lands in the western United States have significant potential to produce energy from wind power (Fig. 7.31). Although wind power has been recognized for a long time (early explorers sailed around the world on wind power), the development of significant quantities of energy from wind power is relatively recent.

The U.S. Bureau of Land Management currently is writing a Wind Energy Programmatic Environmental Impact Statement. Publication of the draft statement is scheduled for August 2004, and the final Environmental Impact Statement completed in June 2005 following a public comment period from August to November 2004.

Ecological Influences and Pathways

Specific environmental concerns were noise produced by the rotor blades, aesthetic (visual), and mortality to birds flying into rotors. Although these considerations may be reduced through advanced technology, the greater influence on sagebrush ecosystems is likely to result from the roads and powerlines that are necessary to construct and maintain sites used for wind energy.

Military Training

Background

Many of the exercises conducted during military training are destructive by definition, the balance between meeting national needs and environmental conservation often are on uncertain terms. Environmental concerns began to be addressed in the 1980's through programs to monitor habitat and wildlife on lands used for military training and testing (Diersing et al. 1992).

Ecological Impacts and Pathways

The direct influences of military training and testing on shrubland habitats result from maneuvers by tracked and wheeled vehicles, and by fires from ordnance impacts. Tracking by armored military vehicles created soil and vegetation disturbances that facilitated the spread of exotic species, increased the potential for soil erosion, and potentially reduced ecosystem productivity and stability (Belcher and Wilson 1989, Shaw and Diersing 1989, Shaw and Diersing 1990, Watts 1998).

At larger spatial scales, the area used for military training in southeastern Idaho had smaller, more closely spaced shrubland patches (Knick and Rotenberry 1997). Frequent fires within the ordnance impact area had eliminated the shrub cover. Consequently, habitat changes associated with military training facilitated spread of cheatgrass, increased the frequency and severity of fires that resulted in loss of sagebrush habitats (U.S. Department of Interior 1996).

Current Status

The military trains on 87 installations within the Conservation Assessment Area (Fig. 7.32). Of those, 52 contained sagebrush habitats. Total area of sagebrush habitats was 6,815 km^2 on those installations or 26% of the total area used for military training. All military facilities use the Land

Condition Trend Analysis to assess and monitor their lands (Diersing et al. 1992, West 2003). Thirty-six percent of the Army training lands had moderately to highly erodible soils (Shaw and Kowalski 1996). Ground disturbance due to military activities was evident at 17% of the lands surveyed; >60% of those areas exceeded soil-loss tolerance (Shaw and Kowalski 1996).

Restoration and Rehabilitation

Background

Sage-grouse depend on various characteristics of sagebrush ecosystems for their survival. During the spring nesting and brood rearing, locations with a codominance of a subspecies of big sagebrush and mid to tall perennial bunchgrass species generally provide the most important habitat. Summer and autumn habitats vary from farmland to wet meadows to sagebrush lands. In winter, sage-grouse require big sagebrush cover and food, but can use low, black, fringed, or silver sagebrush for food (Connelly et al. 2000*a*).

Restoration of sagebrush habitats can take two different forms, passive and active. Passive forms of restoration do not require human-aided revegetation because desired species exist at the site as plants or seeds. Restoration of the desired plant community, including factors such as, community structure (plant height and cover) and ecosystem processes (e.g. nutrient cycling), can be achieved by changing current management practices to accomplish the relative dominance of species or the desired vegetation structure in the community through the normal successional process. If the desired species and their source of propagules are eliminated from the site and are too far for natural revegetation in a desired time frame, then active restoration may be necessary.

Provided that degradation of habitat quality has not been too severe and that the community has remained within the upper state (Figure 7.33), passive restoration may achieve the desired vegetation changes. However, the loss of dominant species, such as tall bunchgrasses, from a community, even if they are not replaced by invasive species, may require active restoration because the community no longer has an adequate seed bank to draw upon. The plant composition that defines these thresholds among states is unknown and is an active area of research in the region.

Active restoration is warranted if invasive species (e.g. cheatgrass or noxious weeds) or native species that are generally inconspicuous at a site (e.g., junipers or pinyon pines) have replaced dominant species (e.g. sagebrush, perennial bunchgrasses, and forbs) in the community. In the conceptual diagram of species dynamics (Figure 7.33), the site has progressed along a transition into a new vegetation state and degradation of the site has occurred. Note the transitions between states are unidirectional and do not return to the previous state.

In the case of pinyon-juniper tree encroachment, as the site becomes dominated by the trees, sagebrush will die out, the herb layer may decline, and seed banks become depleted (Koniak and Everett 1982, Miller et al. 2000). Natural disturbances such as fire become rare as these trees age and as they dominate a site (Miller and Tausch 2001). If fires do occur, they tend to be severe crown fires of high intensity. On relatively cool wet sites, recovery of native species often occurs slowly following these intense fires. However, on warmer sites high intensity fires are capable of causing shifts from woodlands to introduced annual communities (Tausch 1999*a*,*b*).

Following invasions of exotic annual grasses, the communities become susceptible to more frequent fires because of the increase in fuel that is more continuous across the soil surface than the pre-invasion community. In the pre-invasion community (upper most state Fig. 7.33), the fine fuels would be distributed in patches represented by the perennial bunchgrasses in the community. Exotic annual grasses tend to fill interspaces among these bunchgrasses providing greater continuity of fuels for fires (Whisenant 1990). Most species of sagebrush, except silver and threetip sagebrush, are intolerant of fires and require seed dispersal and germination to reestablish after a fire. Cheatgrass is known to be a successful competitor against native plants for resources necessary for the native plants to establish and grow (Harris 1967, Melgoza et al. 1990, Booth et al. 2003).

Site degradation, in some locations, may become so severe that soil erosion (lower left state, Fig. 7.33) removes the upper soil horizons to such an extent that the potential for the site to support its former native plant community (upper state Figure 7.33) is impossible. In this case, restoration 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 [adapted from Bradshaw 1983 and Aronson et al. 1993]) may be the only remaining alternative that might make the site usable by sage-grouse.

We examine the past and current forms of revegetation used within the assessment area, to examine alternatives (including experimental approaches) available to land managers when faced with degradation and loss of habitat for sage-grouse. The alternatives include combinations of passive and active restoration and rehabilitation techniques. Lastly, bottlenecks restricting restoration success will also be examined.

Past and Current Vegetation Manipulation Approaches

Because most lands in the assessment area are federally managed, vegetation manipulations done in the past have reflected mandated federal policies. The Public Rangelands Improvement Act (PRIA) of 1978 (Chapter 1, Table 1.2) recognized the continued need to improve rangeland conditions on public lands. The major source of measuring land condition was based on a technique that organized plants into categories based on their responses to livestock grazing (increasers, decreasers and invaders) (Pyke and Herrick 2003). Although PRIA explicitly stated the need for improvements in condition for multiple uses, the methods used to implement these improvements tended to rely on the current science of the day. The principal textbook of that time (Valentine 1971) defined range improvements as “special treatments, developments, and structures used to improve range forage resources or to facilitate their use by grazing animals.” The focus of many revegetation efforts was to increase forage production for livestock and to decrease the abundance of undesirable forage and invasive annuals that provided unreliable forage. Undesirable forage included the major invasive plant, cheatgrass (Young et al. 1972), and sagebrush which is still treated as a weed in some books (Whitson 1996).

Livestock grazing modifications. Passive rangeland improvement approaches sought improved vegetation composition and amount through adjustments livestock grazing seasons and animal unit months. Adjustments were achieved by constructing new fences or developing additional water sources which spread livestock use over larger areas. The greatest change was the shift from year-long grazing to seasonal uses by livestock throughout the sagebrush grassland biome (Crawford et al. 2004), however the seasons of use often differed between the intermountain and the Great Plains regions west and east of the Rocky Mountains with Wyoming and the Colorado Plateau being somewhat intermediate. Year-long grazing still is practiced in some parts of the sagebrush biome.

Differences between the two regions were largely dictated by the amount of cool-season (C_3 photosynthetic pathway) *vs.* warm-season (C_4 pathway) grasses respectively in the two regions. Cool-season grasses tolerate grazing from mid-summer through winter (Crawford et al. 2004) while adjustments in grazing seasons were more rotational so that no single plant life form would be detrimentally harmed. In sagebrush grasslands, herbivory of herbaceous plants during the growing season tends to favor sagebrush growth until sagebrush becomes so dense that the competition of sagebrush restricts recovery of herbaceous plants (Reichenberger and Pyke 1990).

Adjustments to grazing seasons or reductions in numbers of livestock will only show improvements in sage-grouse habitat quality if the vegetation community is a sagebrush grassland mix (middle vegetation community in upper state, Figure 7.33) before grazing changes are implemented. This community retains both the sagebrush and the tall bunchgrass necessary for quality habitat. The release from livestock grazing should allow the full expression of vegetation height for hiding cover and nest protection. Improvements could 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 assuming the site is not fully occupied by other species. Any other community whether in this vegetation state or in another state (Figure 7.32) will require either additional manipulations to the community to adjust the vegetation composition, or may require additions of life forms through revegetation to improve the habitat (see below).

Sagebrush Removal. Removal of sagebrush to increase herbaceous forage and allow grasses to dominate has been a common habitat treatment practice. Several techniques were used to accomplish this conversion with differing impacts on the structure and function of the ecosystem.

Prescribed fires kill, eliminate, or reduce the density of most sagebrush species, especially big sagebrush species, and provide a temporary flush of nutrients that may result in increases in herbaceous plant responses, but may leave sites susceptible to soil erosion during the first years after the fire (Wrobleski and Kaufmann 2003, Stubbs 2000, Blank et al. 1994). This tool is one that is currently being applied on lands where pinyons or junipers have encroached into sagebrush grasslands as a technique to eliminate the trees. It also results in a loss of sagebrush dominance from the community for 25-45 years (Watts and Wambolt 1996, Wambolt et al. 2001) depending on the location of seed sources.

Herbicide applications of 2,4-D (2,4-dichlorophenoxy acetic acid) or tebuthiuron (N-[5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2-yl]-N,N'-dimethylurea) were used to kill large expanses of sagebrush leaving the standing dead skeletons of the shrubs with low risk of soil erosion. However, herbicides, if used full strength during the growing season, also killed or injured many forbs (Crawford et al. 2004).

Mechanical techniques ranged from those designed to remove the aboveground portion of the plant (mowing, roller chopping, and rotobeating) to uprooting the plant from the soil (grubbing, bulldozing, anchor chaining, cabling, railing, raking and plowing) (Scifres 1980). Of these techniques, the uprooting techniques create the greatest soil disturbance thus adding to the risk of post-treatment soil erosion. Control of pinyon and juniper through chaining, cabling, railing, or chain saw can have moderate to little impact on the shrub canopy. The removal of tree competition should also facilitate rapid recovery of the shrub and herb understory if adequate levels are present prior to treatment. However, treatments such as mowing, roller chopping, rotobeating and plowing will have a greater and longer lasting impact on the shrub layer. Critical for the success of these techniques is that the community remains in the upper state of Fig. 7.33 and that invasive annual grasses do not exist within the community.

Tebuthiuron at low rates has been reported as a technique for thinning dense sagebrush and opening the community for herbaceous plants, including forbs, to respond (Olson and Whitson 2002). Provided the herbaceous perennial plants exist in the understory, this technique might yield immediate improvements to habitat quality, however if exotic annual grasses exist in the community then expansion and spread of these invasive plants might result. However, no empirical data are available to document the response of sage-grouse to these treatments.

Lastly, livestock may be used as a biological control of sagebrush. Bork et al. (1998), Laycock (1967) and Mueggler (1950) have shown long-term declines in threetip sagebrush with recovery of herbaceous vegetation high elevation sites in Idaho. Declines of Wyoming and mountain big sagebrush densities due to heavy deer or elk browsing have been noted in Utah and Montana (Smith 1949, Austin et al. 1986, McArthur et al. 1988, Patten 1993, Wambolt 1996). These all suggest the potential of browsing animals to be used as a biocontrol for reducing the densities of sagebrush and potentially increasing the herbaceous component. However, the response of the herbaceous component needs further study in big sagebrush communities since this has only been noted in threetip sagebrush communities.

Revegetation

Historic revegetation on most sagebrush grasslands had the goal of improving livestock forage (which included replacing invasive forbs and annual grasses such as cheatgrass and halogeton with perennial grasses) while protecting soils from erosion. Early experimental trials comparing native *vs.* introduced grasses in several locations within the assessment area found that native species often did not establish or produced less forage, therefore recommendations during the early phases of rangeland improvements favored the use of introduced grasses, such as crested wheatgrass (*Agropyron* sp.) to meet the combined goal of forage production and erosion control (see citations in Asay et al. 2001). Many of these early trials were conducted on abandoned wheat and rye fields at the end of the homestead era.

Wildfire rehabilitation is a major source of revegetation in the Great Basin. The mandated goal of these projects is to reduce the loss of soil and plant species, be palatable to livestock, and to reduce the spread of invasive species. Total restoration of the ecosystem with a complete suite of plant life forms is not a designated objective for expenditure of funds. Although federal policies have advocated the use of native plants in revegetation efforts when natives are available, only modest increases in the use of native plants were seen in a recent evaluation of the U.S. Bureau of Land Management's Emergency Fire Rehabilitation program. Out of the average of 5 species used on a rehabilitation project, the number of native species has increased from 1 to 2 and the proportional increase in the weight of native bulk seeds has been from 20 to 40 % (Pyke et al. 2003). Land managers cited the poor competitiveness and poor establishment of natives compared with introduced grasses as the main reasons why they elected to use introduced species (McArthur 2004) and the high cost of seed.

Most revegetation projects that use introduced forage grasses may not provide quality sage-grouse habitat because their goals were not focused on restoring a complete sagebrush grassland community. However, Cox and Anderson (2004) suggested methods for improving these sites: sites dominated by cheatgrass could be seeded with crested wheatgrass to control the cheatgrass. Later, sites dominated by introduced grasses could be prepared by a till, harrow, or with herbicides then reseeded with native species.

In an attempt to become proactive in its battle against invasive annual grasses and the loss of sagebrush grasslands, the U.S. Bureau of Land Management has begun the Great Basin Restoration Initiative. The strategy of this program is to use a three step process to achieve effective restoration in the region (Pellant 2003). The first step is to use spatial data to prioritize areas for conservation and restoration (Pyke and Knick 2003), with special emphasis on sage-grouse habitat needs. Second, they will coordinate protection and restoration plans with land users, scientists and interested people to ensure environmentally sound treatments that do not create undue hardships for local land users while using the best science to maximize restoration and conservation success. Lastly, restoration and conservation activities will target landscapes where native plant communities already exist to ensure maximize the retention of lands that remain within the nature dynamics of the sagebrush system (upper state Fig. 7.33). After these areas are protected, they will begin treatments to restore sites currently dominated by invasive plants.

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. Therefore methods to reduce cheatgrass densities are necessary. Proposed techniques include herbicides imazapic (Plateau) (Shinn and Thill 2002) and glyphosate (Whitson and Koch 1998), defoliation via livestock grazing (Hulbert 1955, Finnerty and Klingman 1961, Mosley 1996), pathogenic bacteria (Kennedy et al. 1991) and fungi (Meyer et al. 2001). Although prescribed fire alone is not recommended (Mosley et al. 1999), 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 (McLendon & Redente 1990, 1992, Young et al. 1996).

Immediate revegetation is required after any of these density reduction techniques, otherwise invasive annual grasses that escape treatments will grow unabated and produce large numbers of seeds and will quickly dominate a site again (Mack and Pyke 1983). No evidence for complete elimination of invasive annual grasses with control techniques and revegetation has been noted. However, successful revegetation efforts that have controlled invasive annual grass populations and have maintained perennial plants are generally rehabilitation projects sown with introduced forage grasses (Asay et al. 2001). 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 (Pyke et al. 2003, Cox and Anderson 2004). 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 (Sheley et al. 1996). Current studies are being conducted to investigate this potential in combination with cheatgrass and other invasive plants in the Great Basin.

For quality sage-grouse habitat, sagebrush and forb establishment and maturity are necessary. Techniques for reseeding sagebrush have been successfully demonstrated, but surface sowing followed by compaction of the soil may be necessary for establishment. Establishment of forbs important to sage-grouse have also shown promise, but availability of seed tends to limit their widespread use on rangeland restoration and rehabilitation projects (McArthur 2004).

Bottlenecks to Success

Availability and cost of native seed is a major obstruction to the use native seeds in revegetation projects (McArthur 2004). The difficulties and the vagaries of collecting, growing and selling native seeds that historically have not been used within sagebrush ecosystems tends to raise the price and increase the risk to both the seller and buyer (Dunne 1999, Roundy et al 1997, Currans et al. 1997, Bermant and Spackeen 1997) relative to tested and released plants that are widely available (Currans et al. 1997).

Equipment for sowing native seeds is not widely available. Most revegetation projects in the region use rangeland drills that were developed for the rough terrain of wildland environments and for the ease of seeding the introduced forage grasses. Many native seeds because of their differing sizes will require mixing within the seed boxes on the drills to insure equal proportions of all seeds are sown on a site or will require separate seed boxes to allow seeds of different sizes to be buried at different optimal depths. All these requirements will either require the purchase of new seed drills or the retrofitting of the old drills to accommodate these needs.

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Fig. 7.1. Number of fires, average fire size (ha), and total area burned (ha) within the Northern Great Basin division (Fig. 5.2) of the sagebrush biome. Regression models of changes in fire statistics from 1980 through 2003 are presented in Table 7.1.

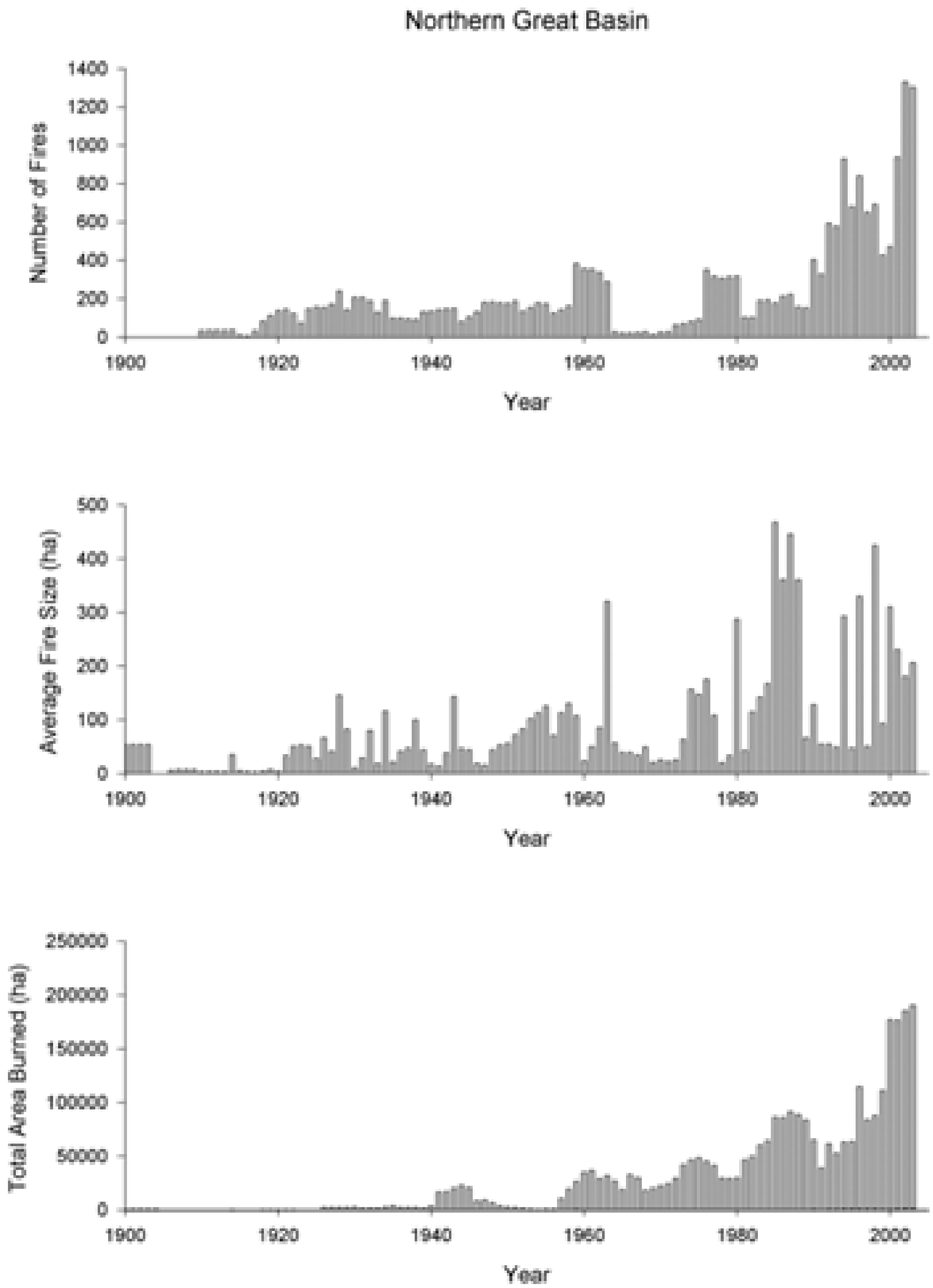

Fig. 7.2. Number of fires, average fire size (ha), and total area burned (ha) within the Southern Great Basin division (Fig. 5.2) of the sagebrush biome. Regression models of changes in fire statistics from 1980 through 2003 are presented in Table 7.1.

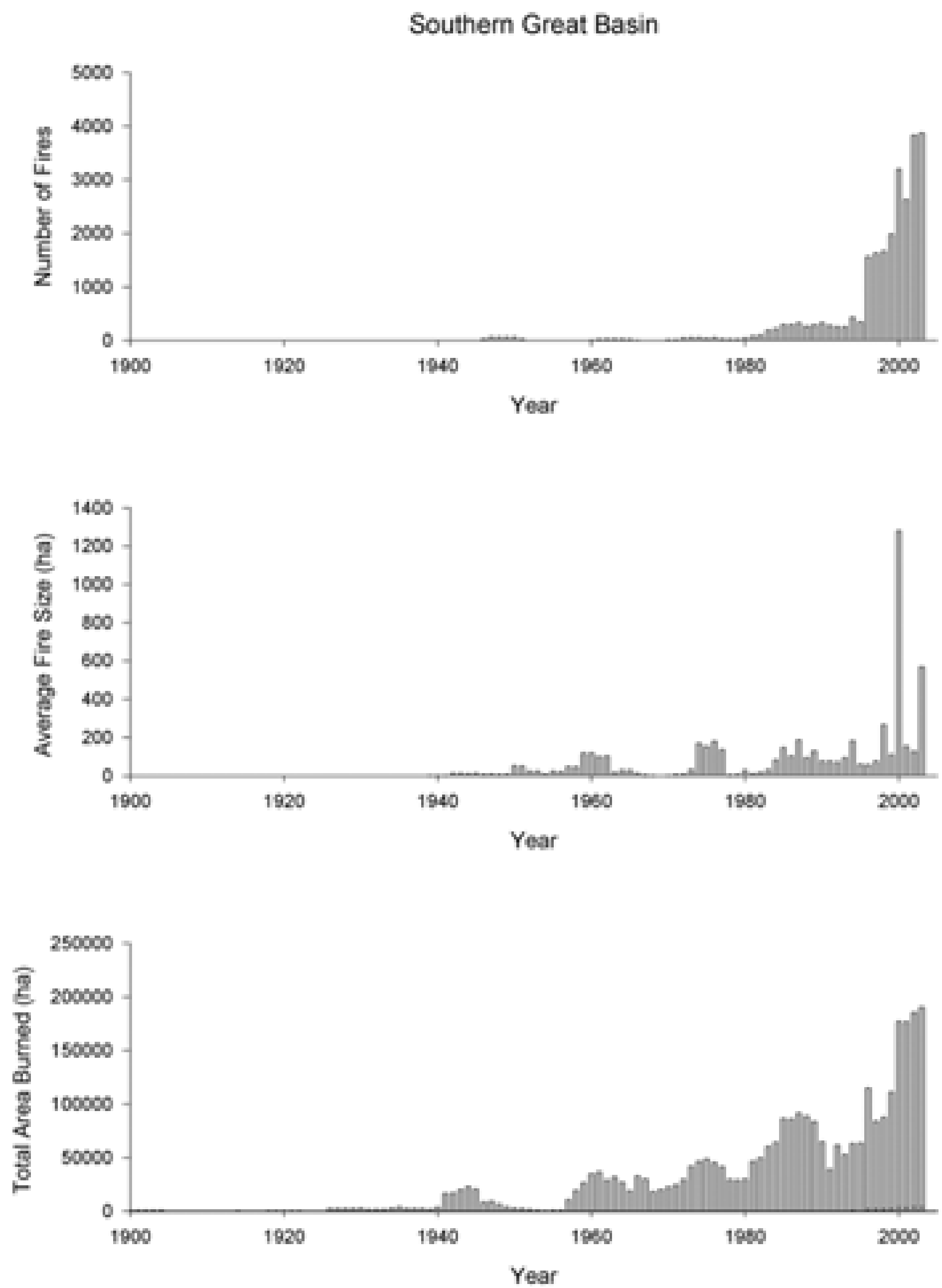

Fig. 7.3. Number of fires, average fire size (ha), and total area burned (ha) within the Silver Sagebrush division (Fig. 5.2) of the sagebrush biome. Regression models of changes in fire statistics from 1980 through 2003 are presented in Table 7.1.

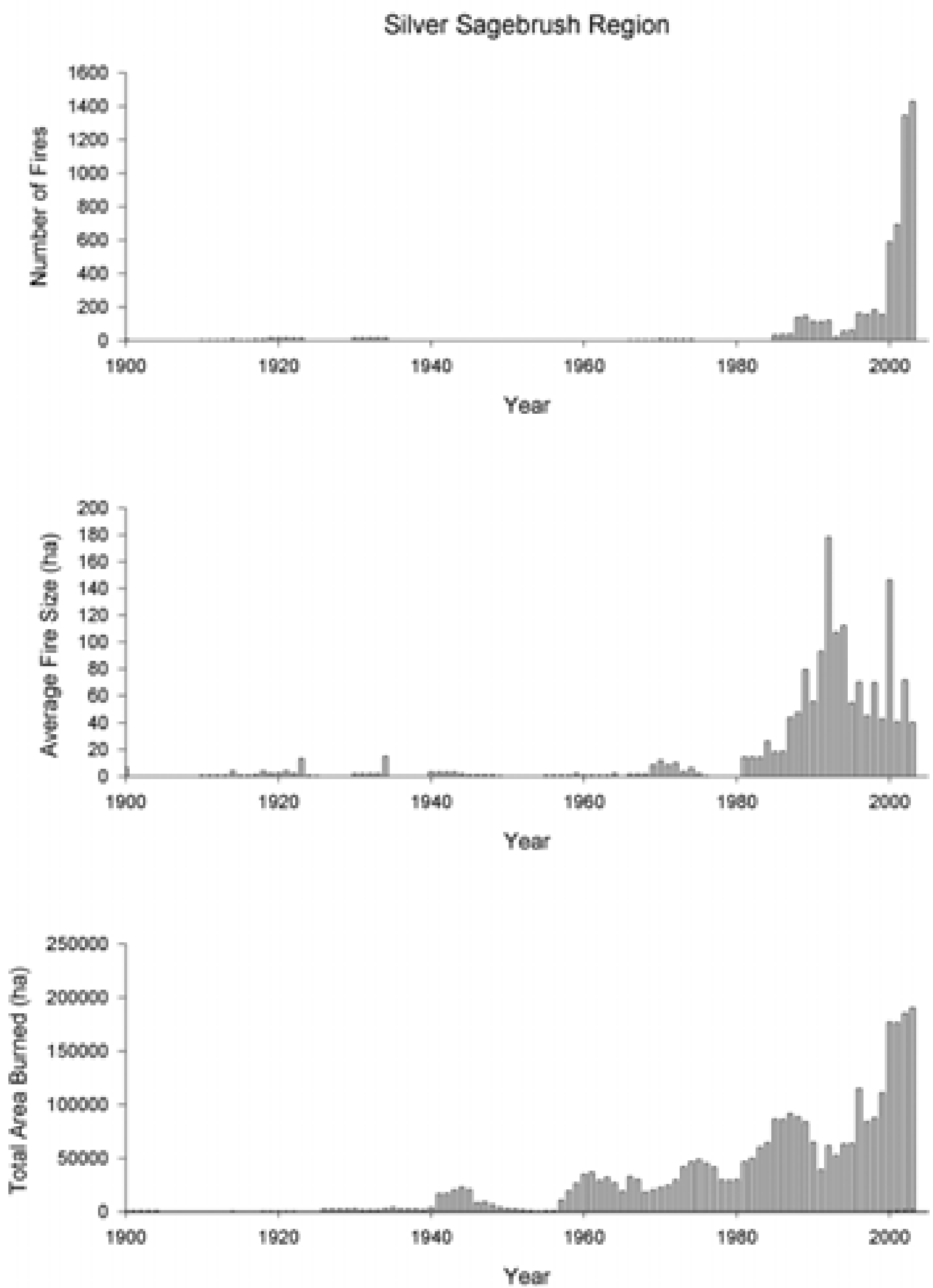

Fig. 7.4. Number of fires, average fire size (ha), and total area burned (ha) within the Snake River Basin division (Fig. 5.2) of the sagebrush biome. Regression models of changes in fire statistics from 1980 through 2003 are presented in Table 7.1.

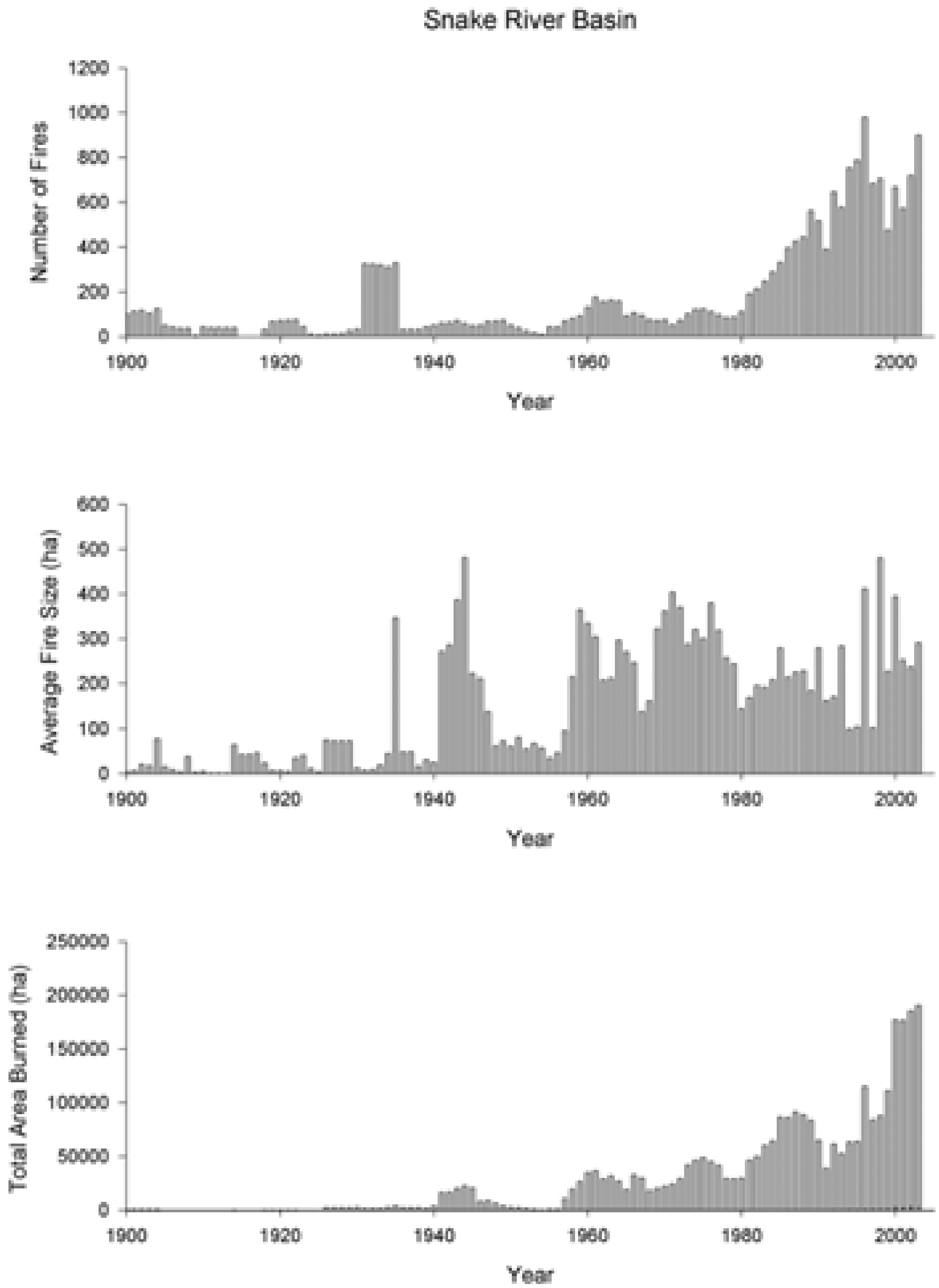

Fig. 7.5. Number of fires, average fire size (ha), and total area burned (ha) within the Wyoming Basin division (Fig. 5.2) of the sagebrush biome. Regression models of changes in fire statistics from 1980 through 2003 are presented in Table 7.1.

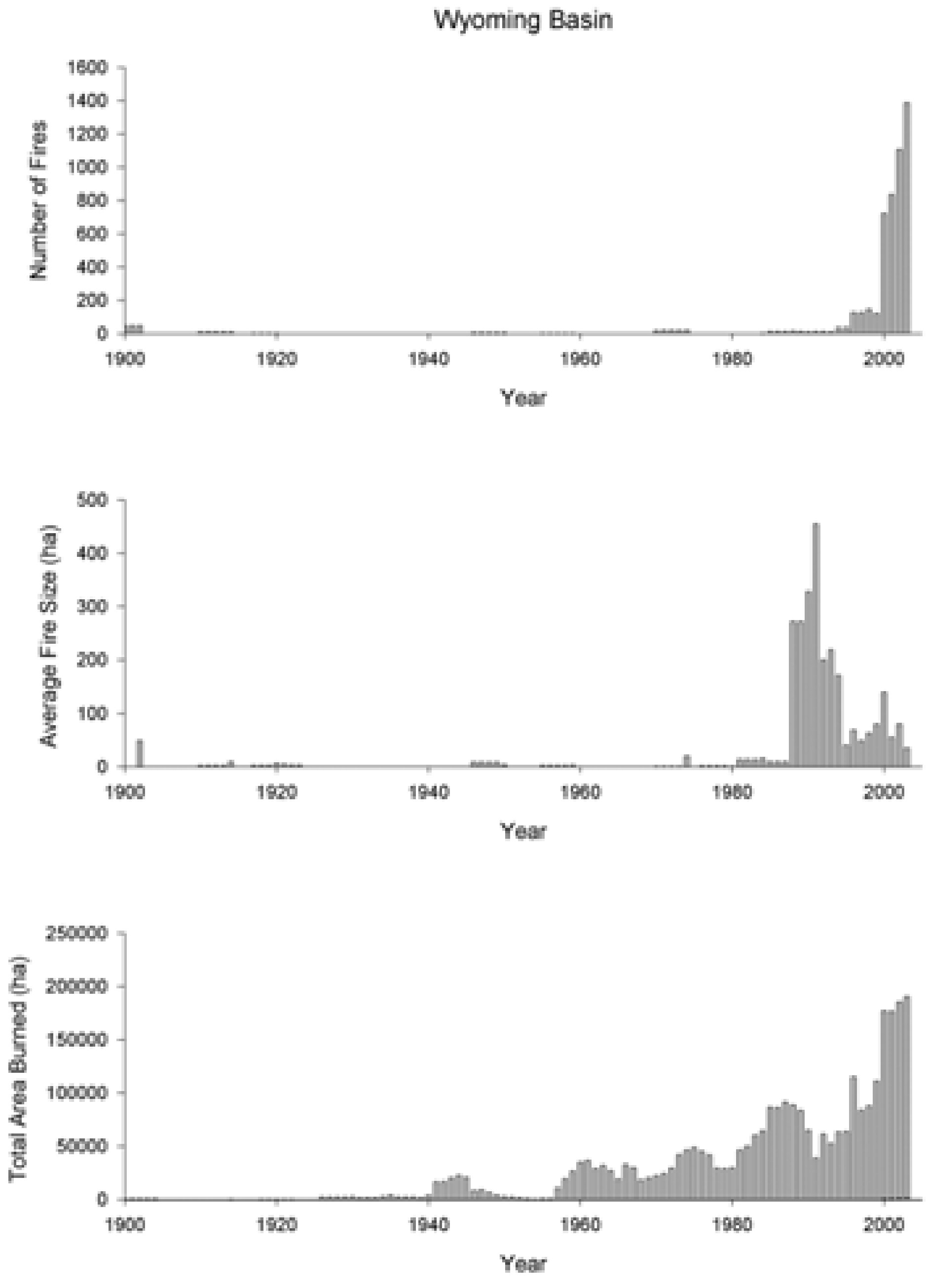

Fig. 7.6. Fires mapped in the western United States from 1960 to 2003. Fire information was obtained from approximately 500 source datasets obtained from the U.S. Bureau of Land Management, U.S. Forest Service, U.S. National Park Service, and other state and federal agencies).

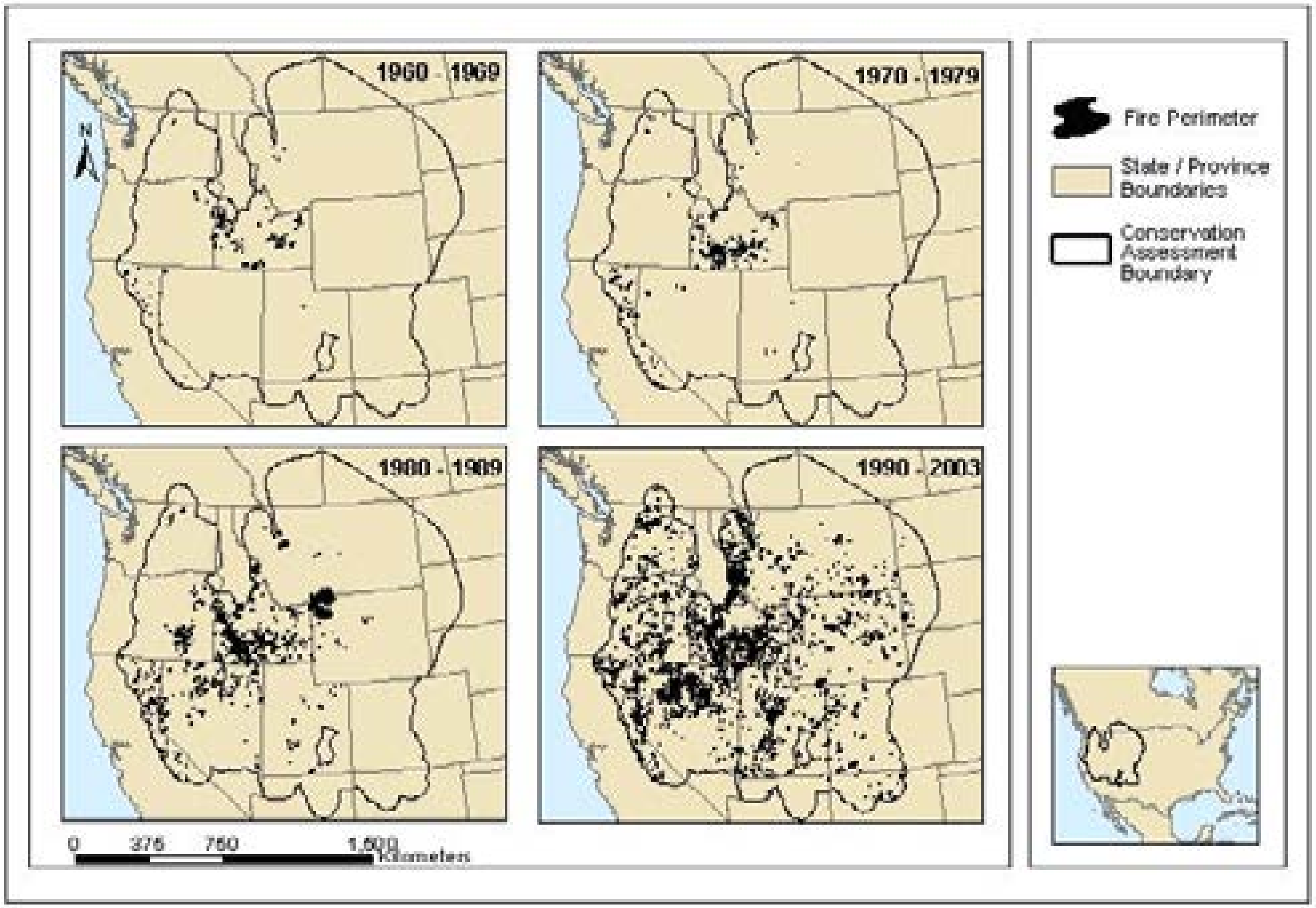

Fig. 7.7. Distribution of human-caused fires relative to roads. Fire information obtained from National Fire Occurrence Database.

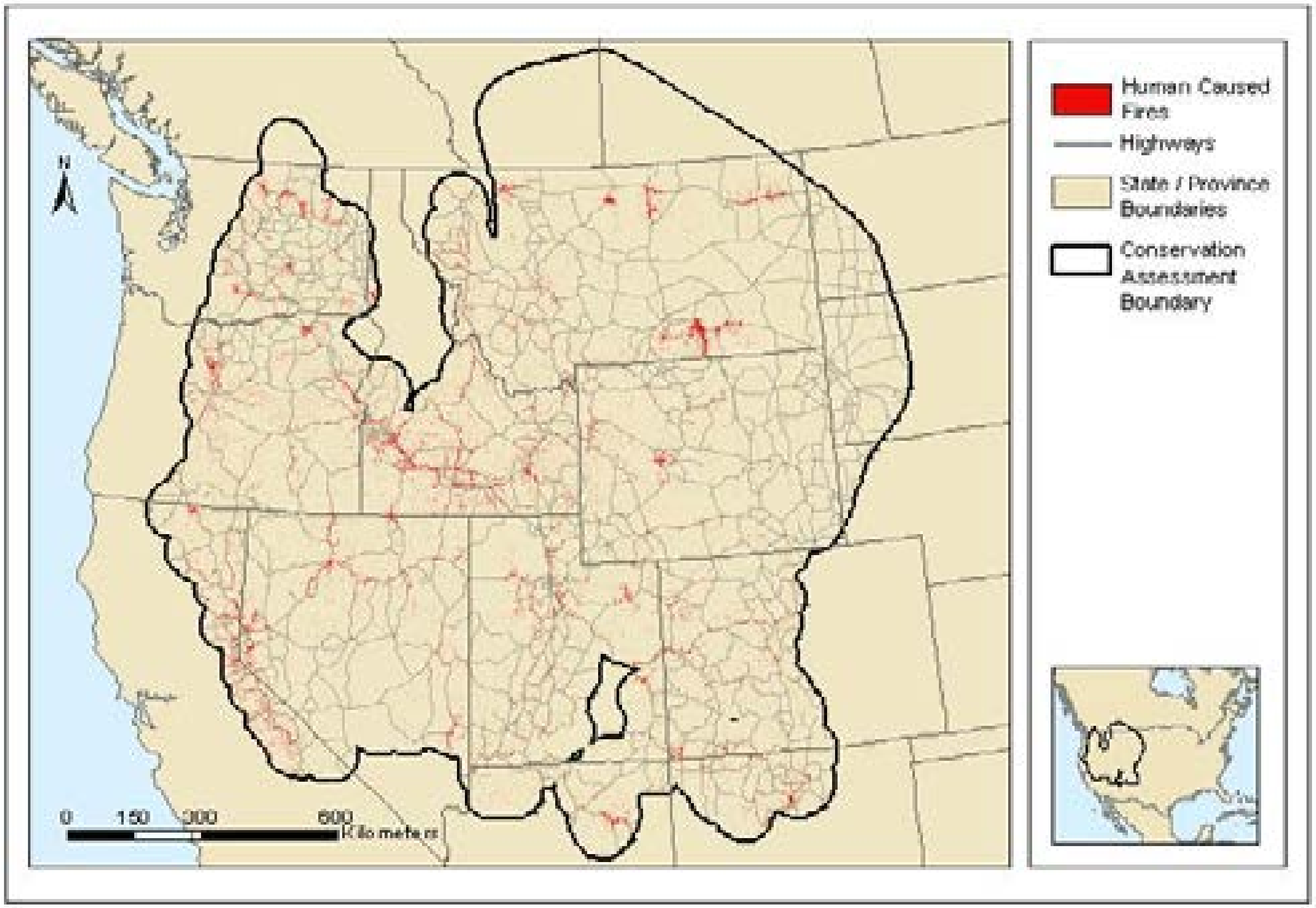

Fig. 7.8. Ecological provinces in the intermountain west, adapted from West et al. (1998) and Miller et al. (1999).

Fig. 7.9. Estimated risk of pinyon-juniper displacement of sagebrush in the Great Basin Ecoregion during the next 30 years. Levels of risk of sagebrush displacement are mapped in relation to all sagebrush cover types in the 3 Ecological Provinces. Areas considered not at risk are cover types other than sagebrush.

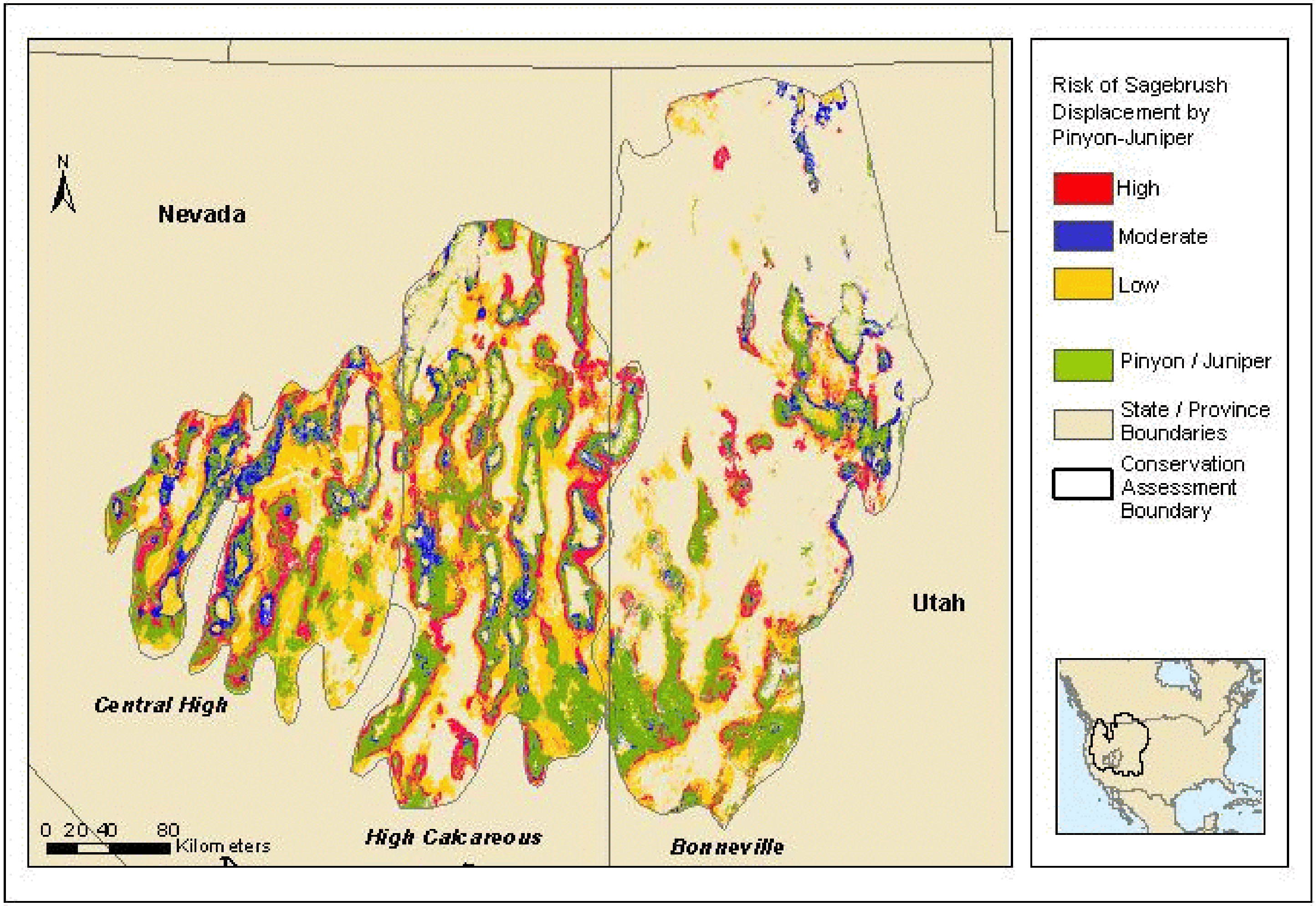

Fig. 7.10. Estimated risk of cheatgrass displacement of sagebrush and other susceptible cover types in the Great Basin Ecoregion and the state of Nevada during the next 30 years.

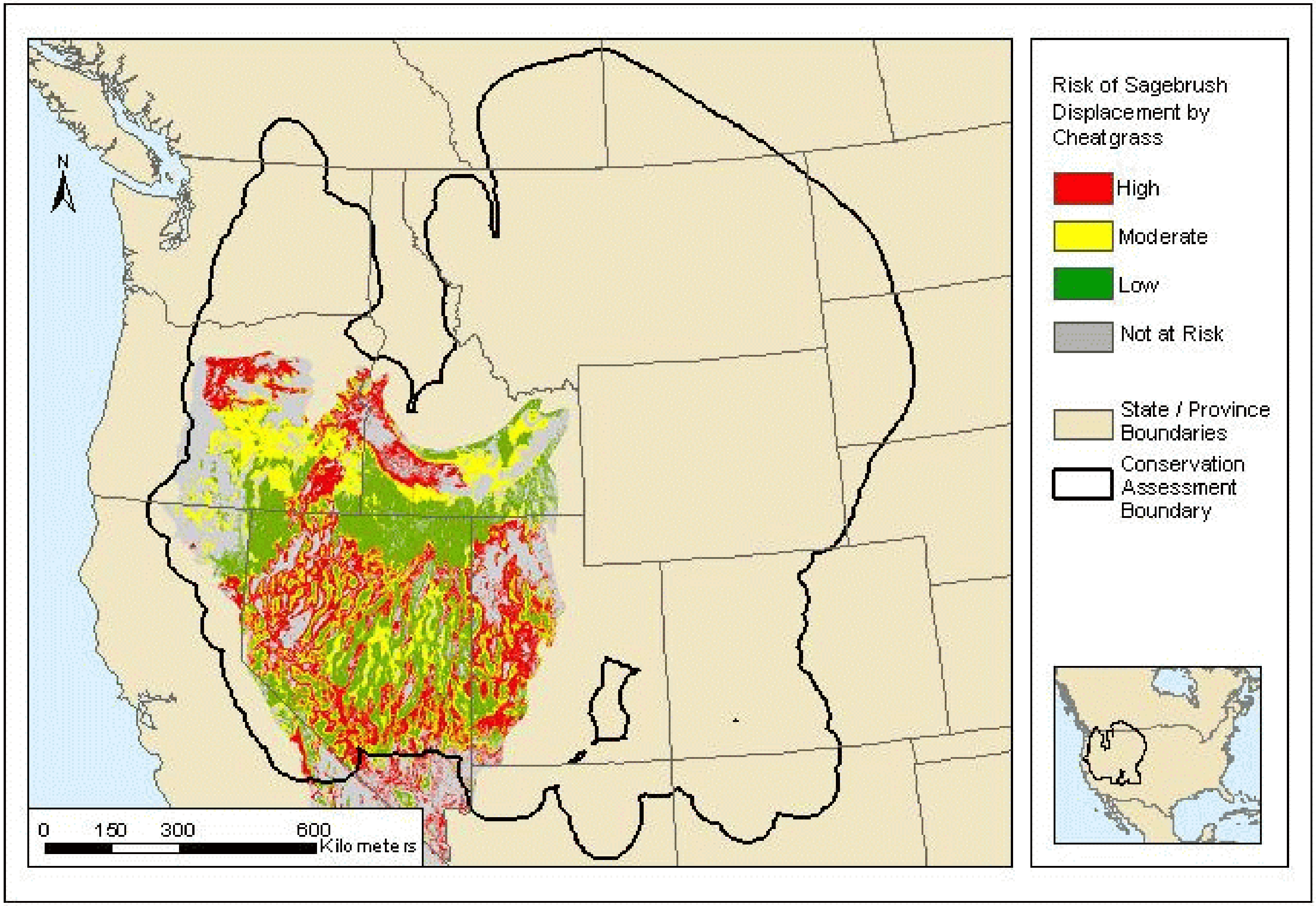

Fig. 7.11. Percent of major river basins experiencing drought conditions from 1895 to 2004 (U.S. National Drought Mitigation Center 2004). The graphs represent the Palmer Drought Severity Index (1965), which measures the degree of departure based on precipitation, temperature, and available water capacity.

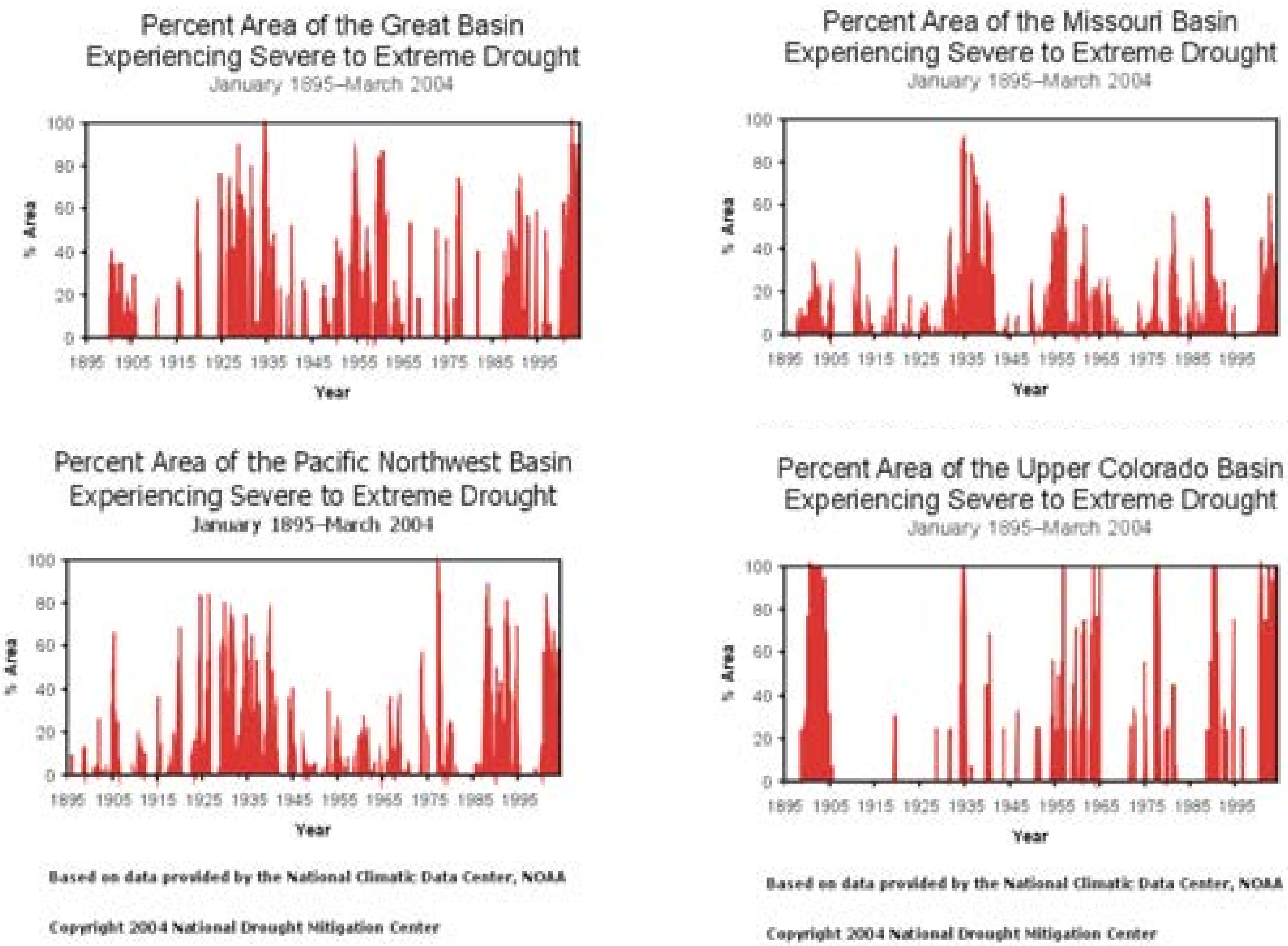

Fig. 7.12. Agricultural lands within the sagebrush biome buffered by 2.5 km. Landcover data obtained from sagebrush distribution map (Comer et al. 2002, modified in this study), individual state GAP analysis programs, and the 1992 National Land Cover Database.

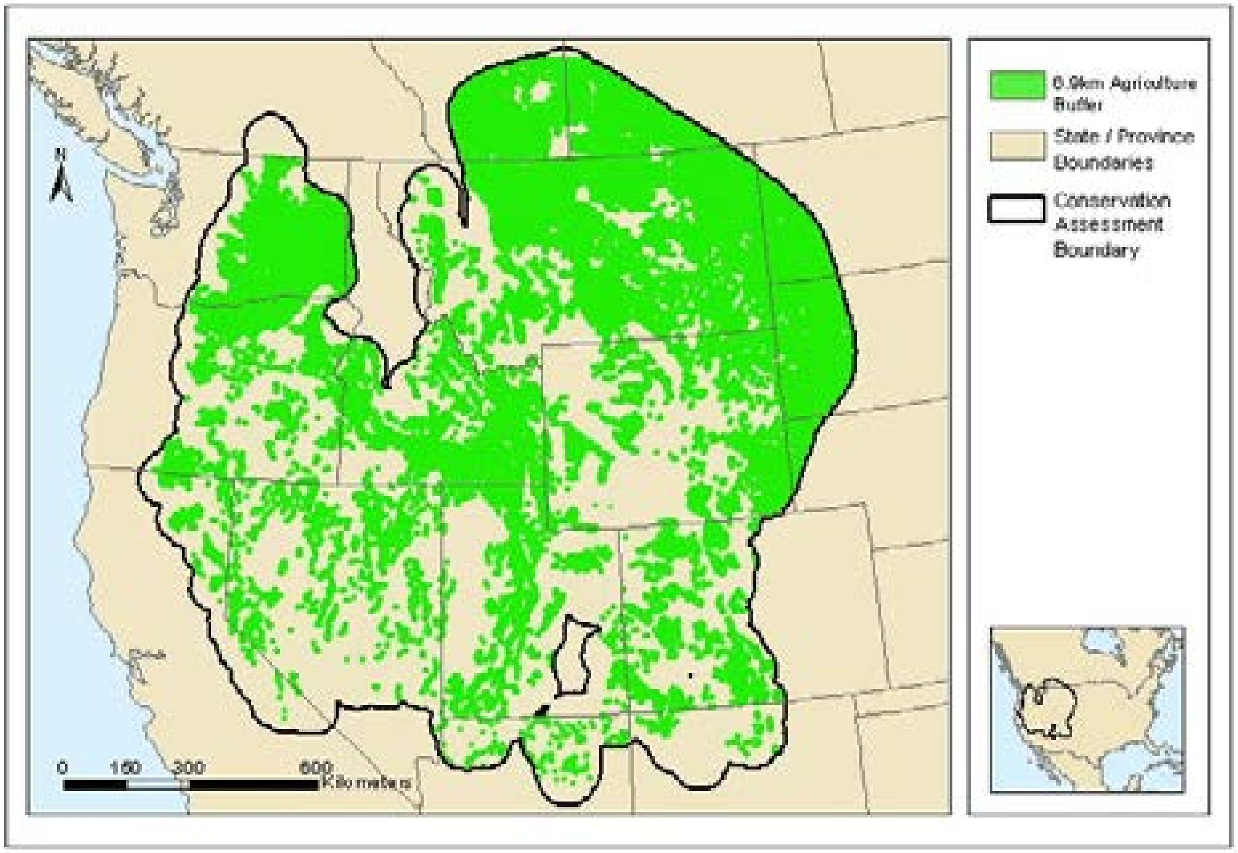

Fig. 7.13. Total area (ha) under contract within the Conservation Reserve Program for counties in the United State by 5-year intervals from 1987 through 2004. (Data provided by U.S.D.A. Farm Services Agency).

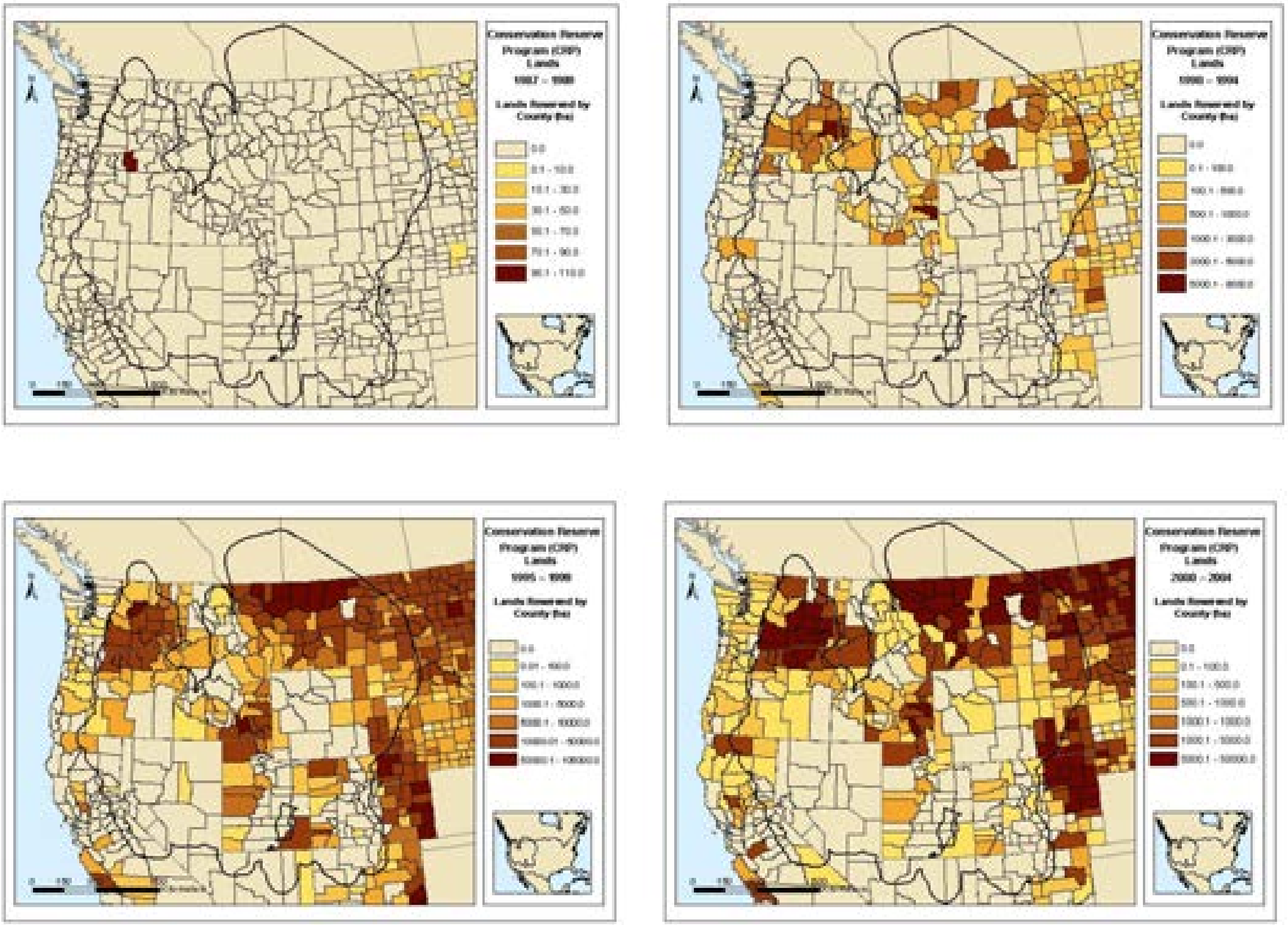

Fig. 7.14. Change in population density from 1900 to 2000. Population data obtained from U.S. Census Bureau.

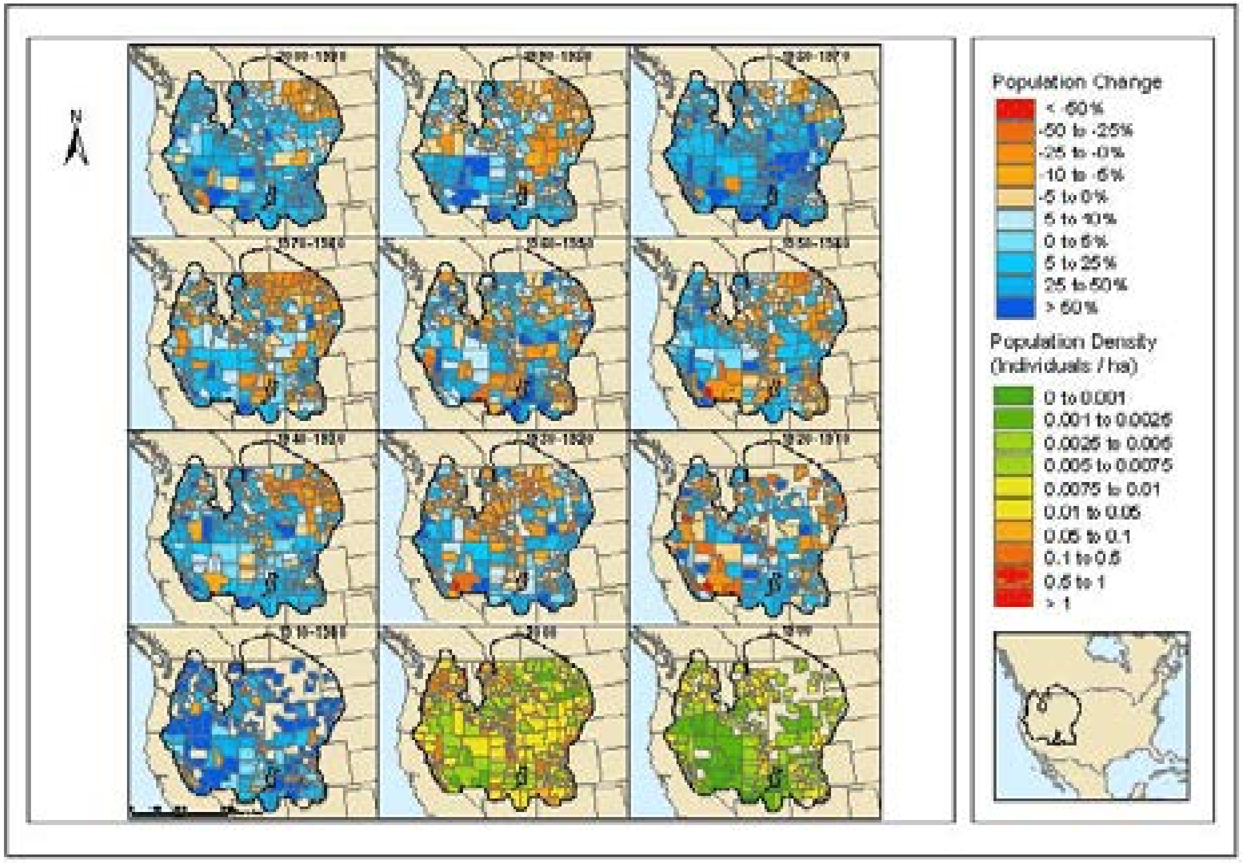

Fig. 7.15. Location of major landfills within the western United States. Landfill locations obtained from waste management agencies of individual states. Buffer distances of 6.9 km represent foraging radius of nonbreeding common ravens (Boarman and Heinrich 1999.

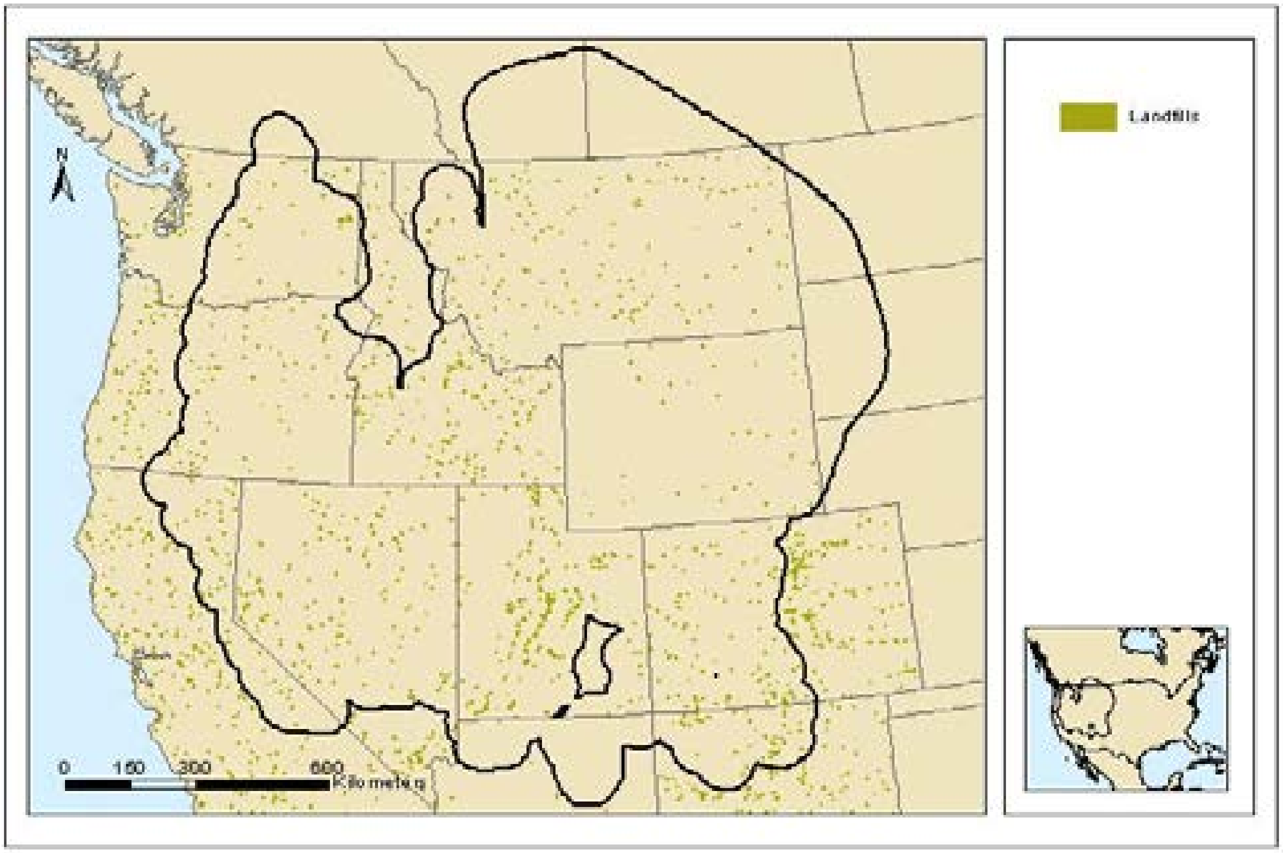

Fig. 7.16. Interstates and major highways in the sagebrush biome. (Data obtained from U.S. National Atlas and Geo Gratis for Canada).

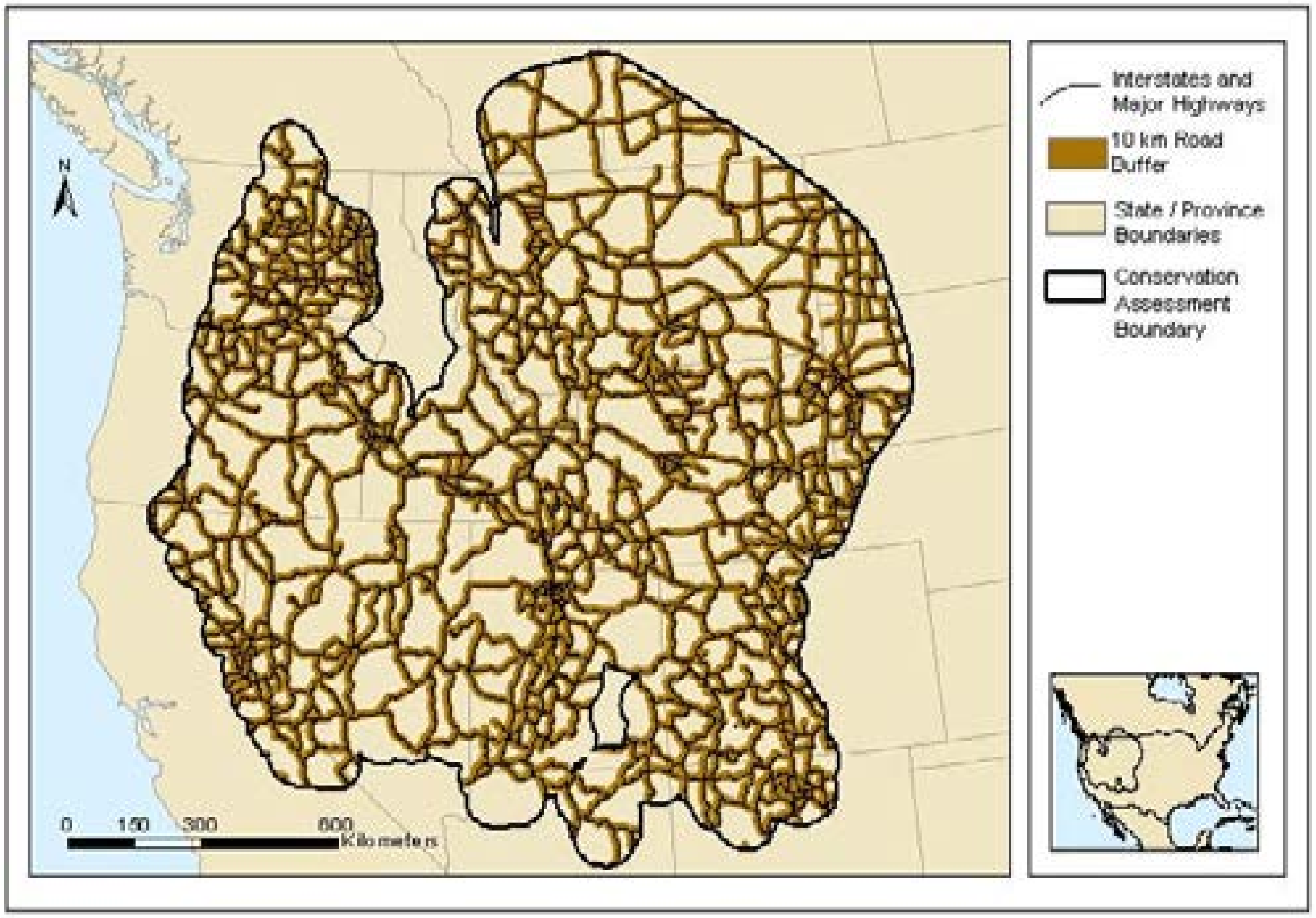

Fig. 7.17. Contoured secondary roads in the sagebrush biome (linear distance/10 km grid). Road information derived from US Census TIGER/Line Files 2002).

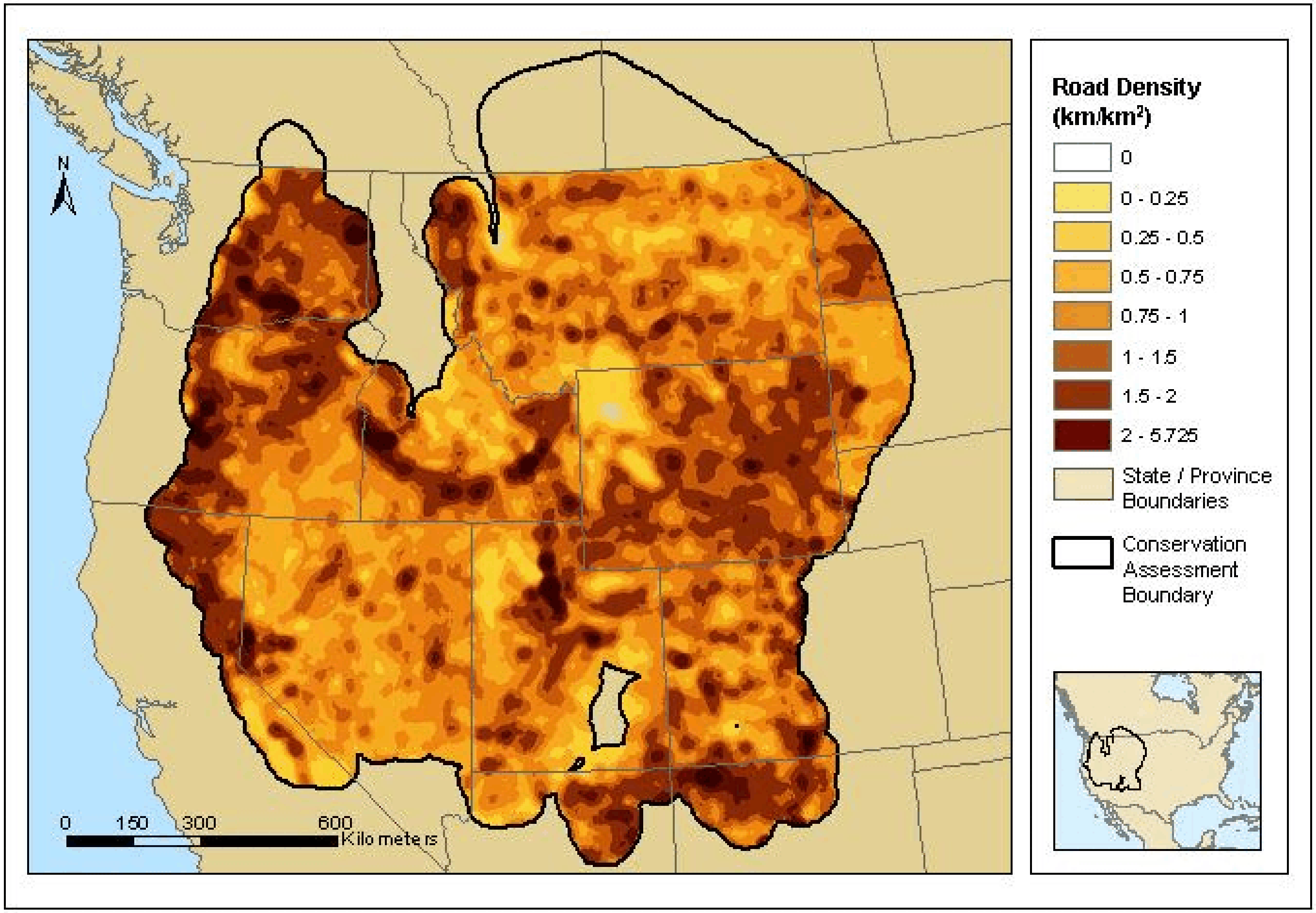

Fig. 7.18. Rest areas throughout the Conservation Assessment Study Area. Location of rest areas obtained from tranportation agencies of individual states.

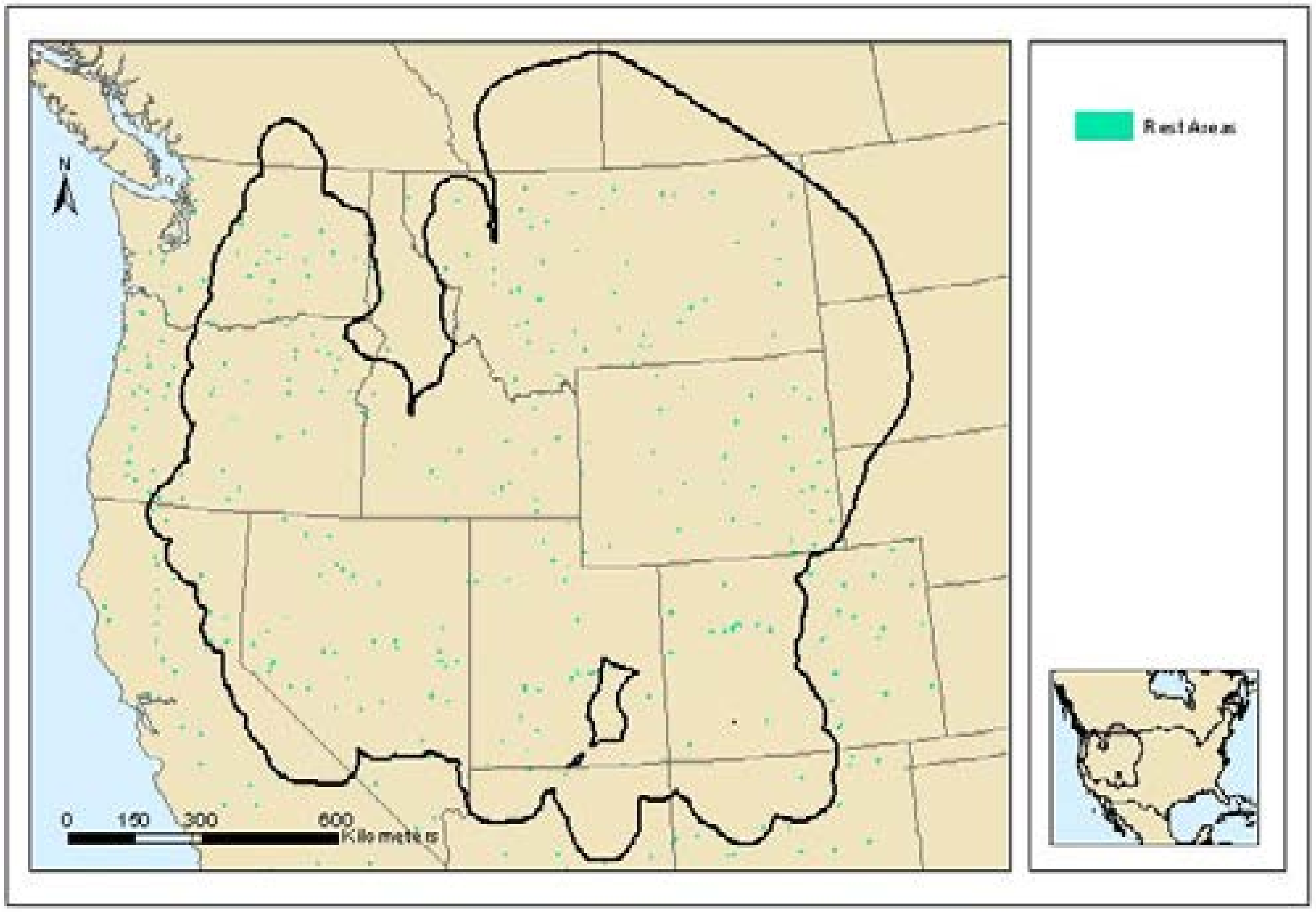

Fig. 7.19. Powerline corridors within the sagebrush biome. (Compiled from 27 GIS coverages obtained from local, state, provincial, and federal sources).

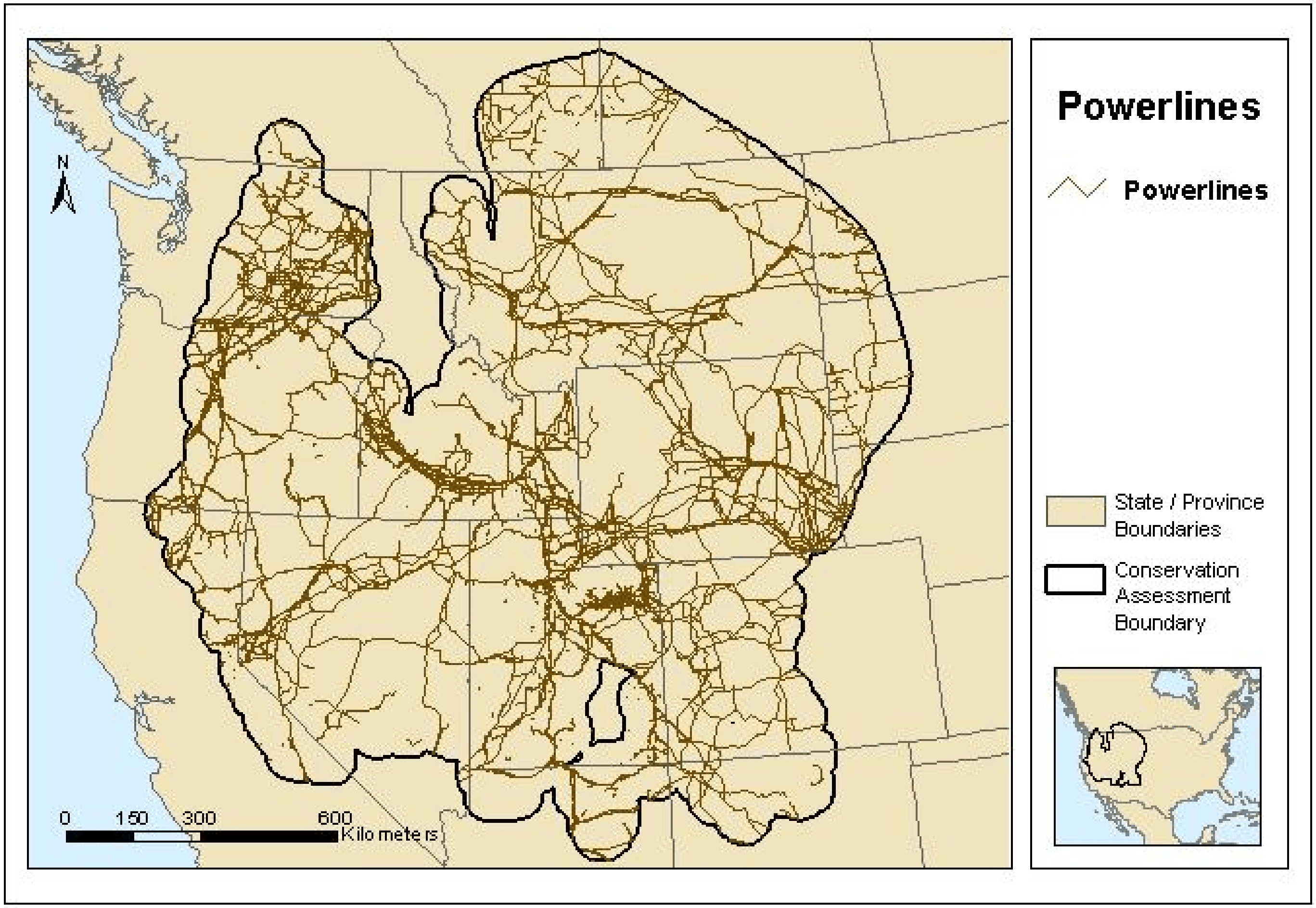

Fig. 7.20. Major railroads in the sagebrush biome. (Data obtained from U.S. National Atlas and Geo Gratis for Canada).

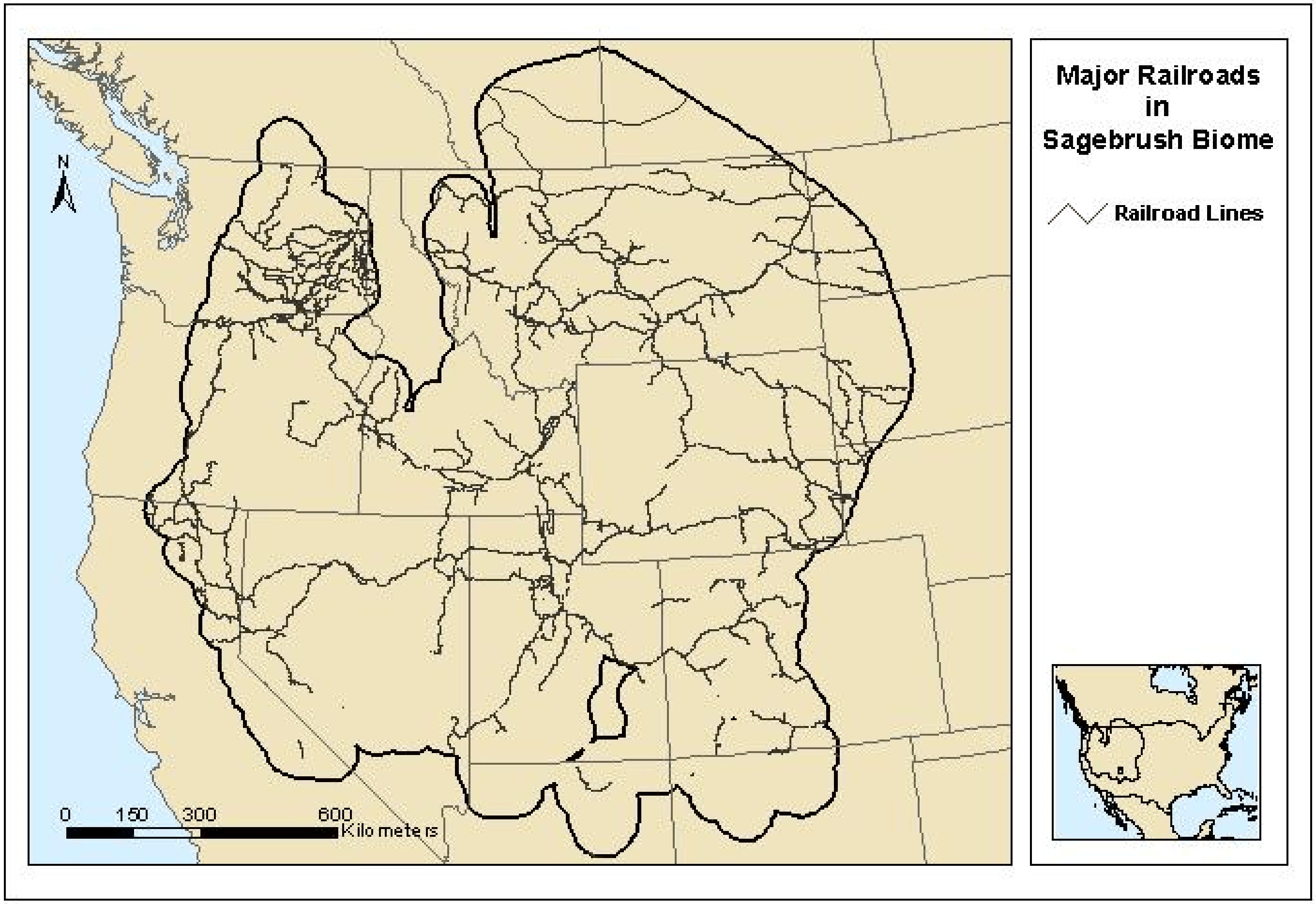

Fig. 7.21. Communications towers in the sagebrush biome. (Data obtained from Federal Communication Commission).

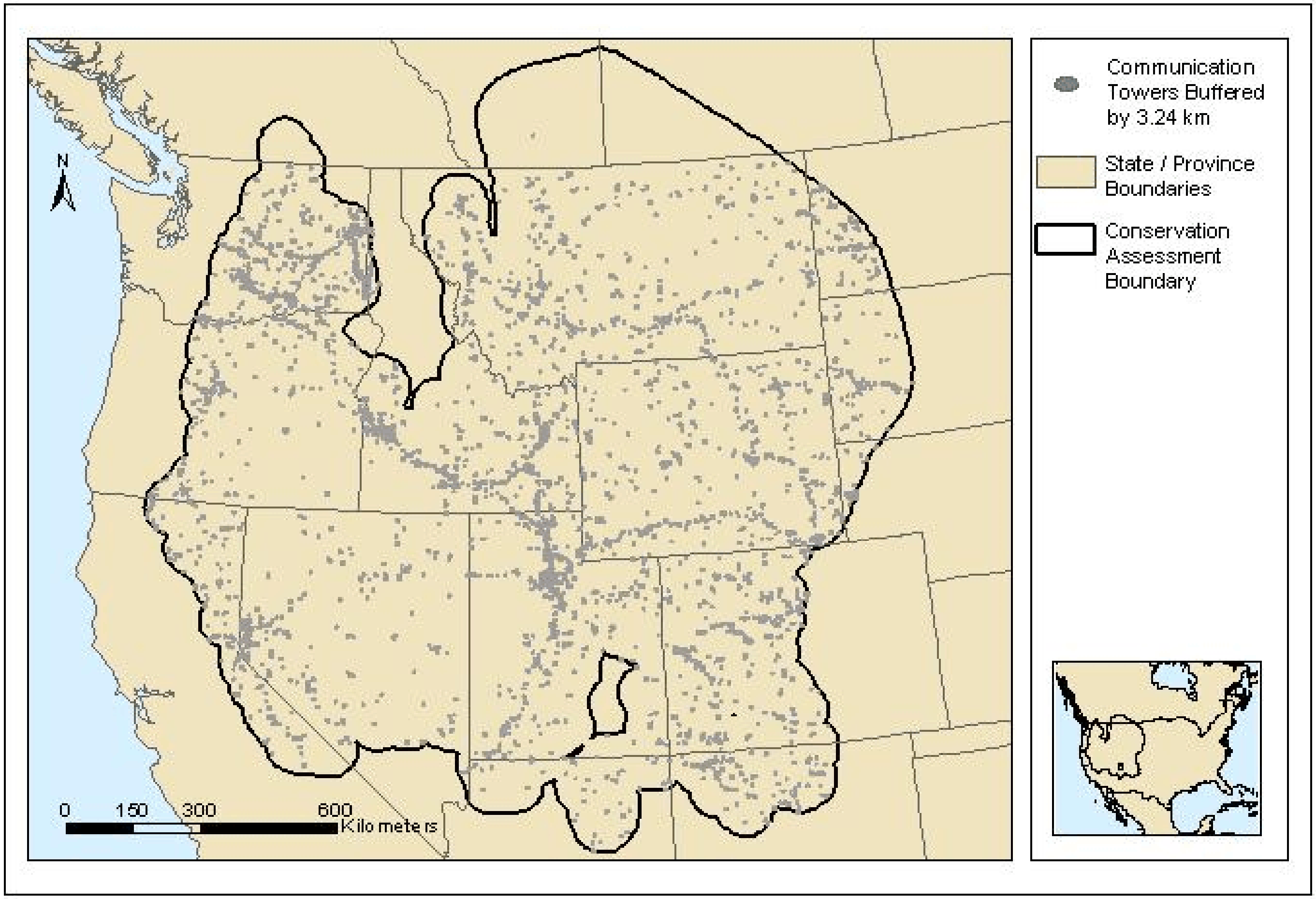

Fig. 7.22. Linear density of fences on public lands in the Conservation Assessment Study Area. (Compiled from individual GIS coverages obtained from U.S. Bureau of Land Management District Offices).

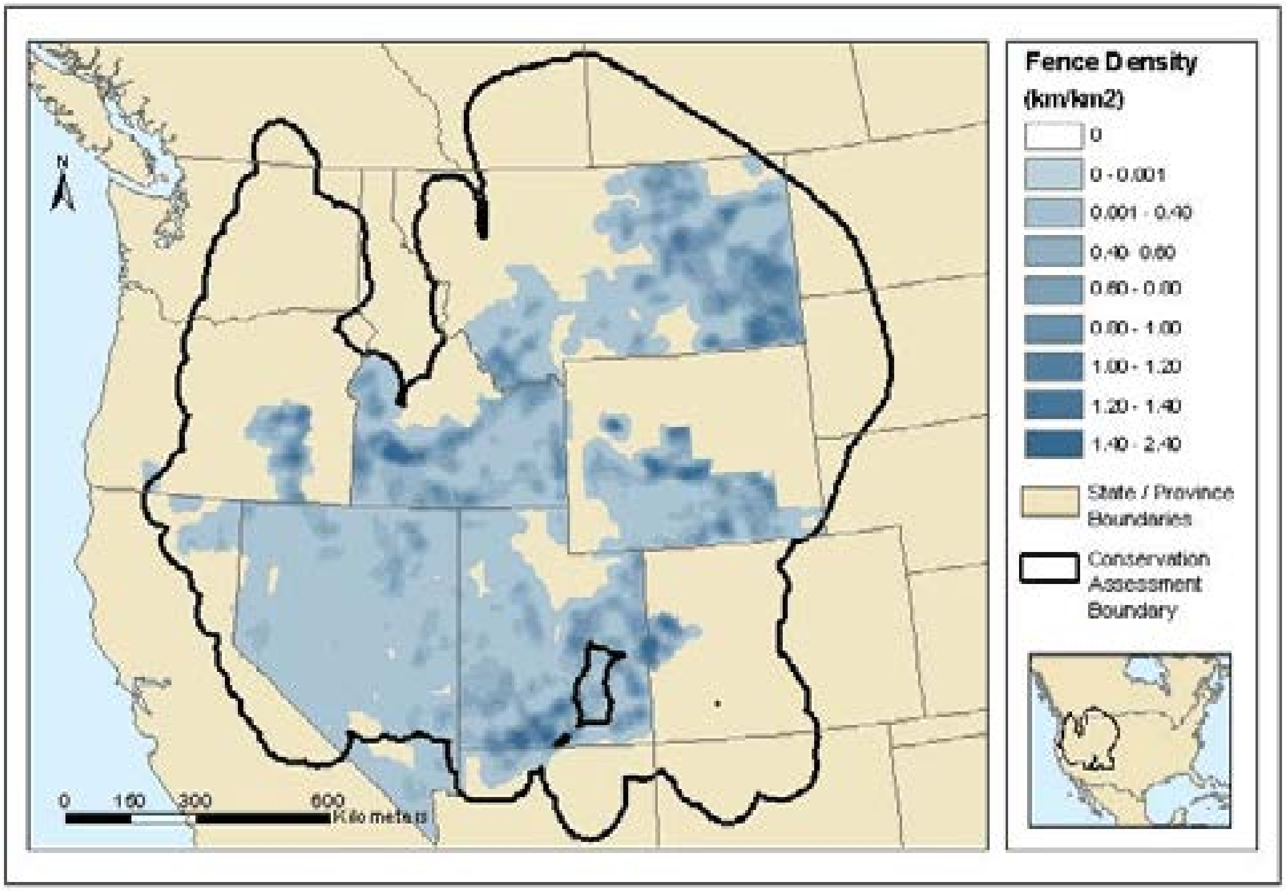

Fig. 7.23. Water developments on lands managed by the U.S. Bureau of Land Management. (U.S. Bureau of Land Management Range Improvement Database).

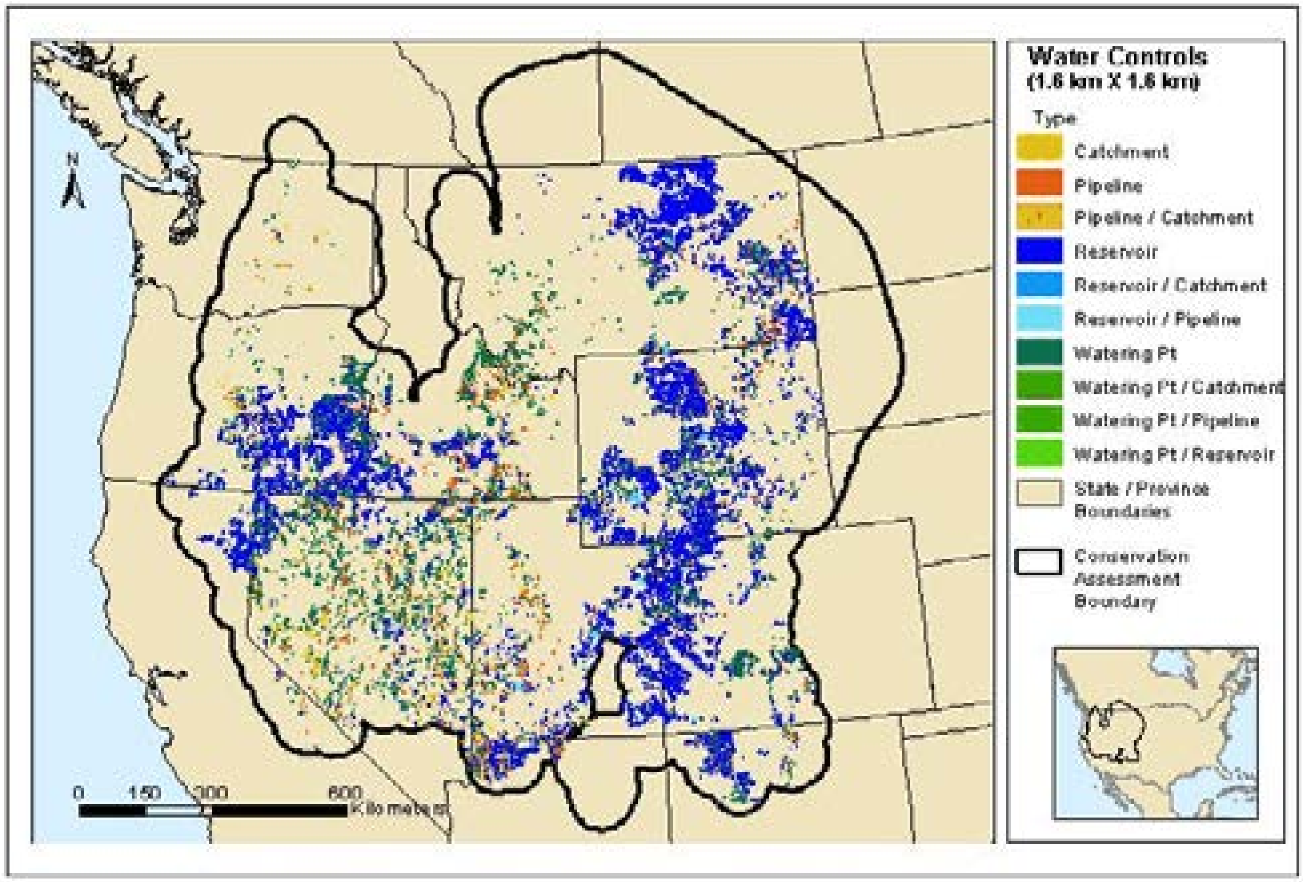

Fig. 7.24. Wild horse management units in the Conservation Assessment Area.

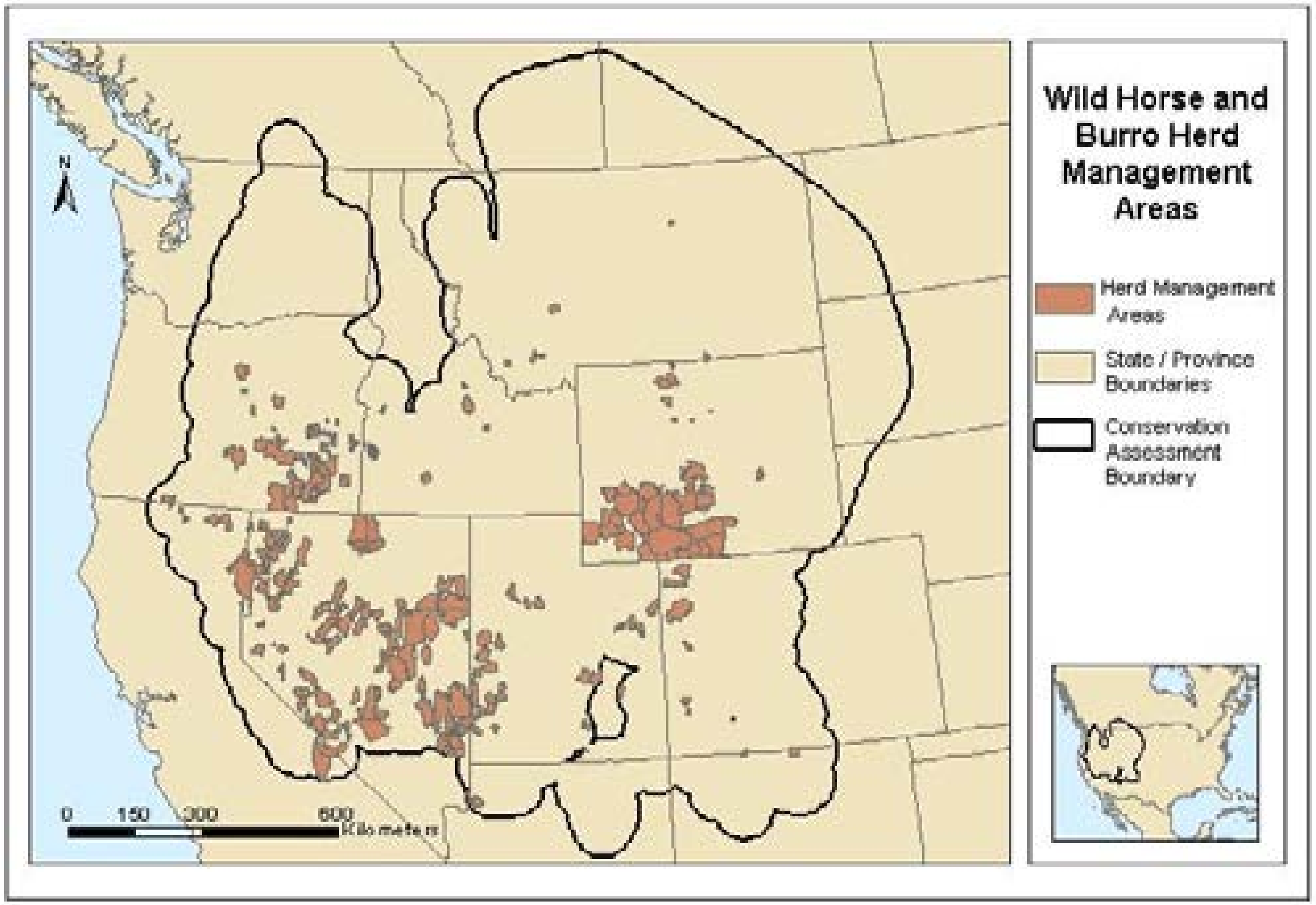

Fig. 7.25. Geologic basins in which oil and gas resources have been developed and current distribution of greater sage-grouse. (Compiled from 5 state-based Oil and Gas Conservation Commissions, 2 state Geological Surveys, and U.S. Bureau of Land Management).

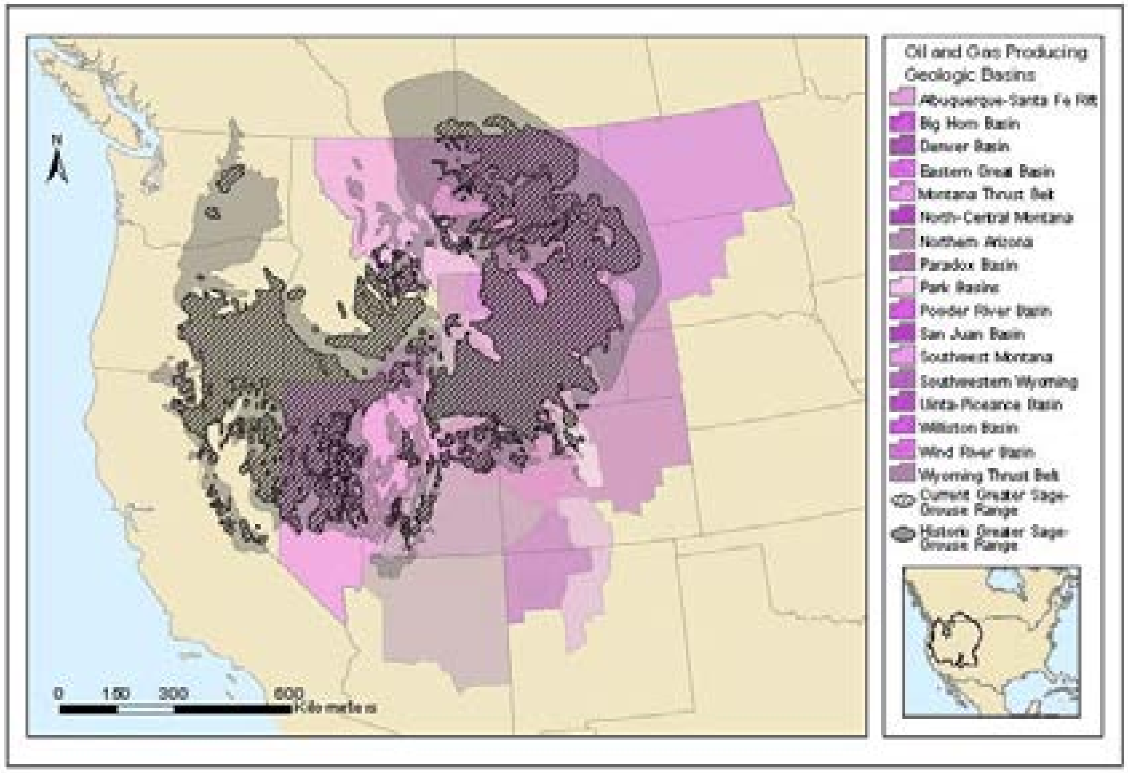

Fig. 7.26. Current producing oil wells in the sagebrush biome. (Compiled from 5 state-based Oil and Gas Conservation Commissions, 2 state Geological Surveys, and U.S. Bureau of Land Management).

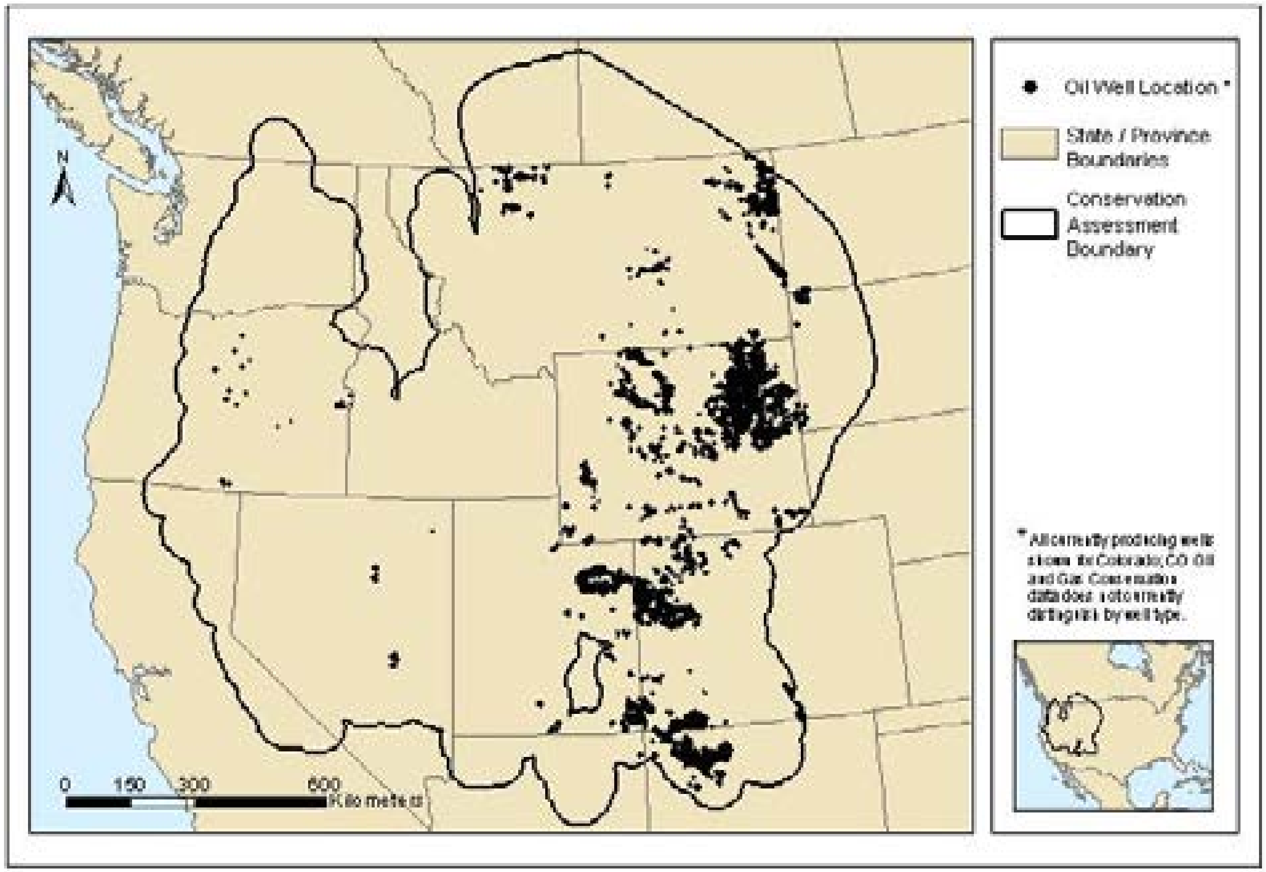

Fig. 7.27. Permitted or pending oil wells in the sagebrush biome (1929-2004). (Compiled from 5 state-based Oil and Gas Conservation Commissions, 2 state Geological Surveys, and U.S. Bureau of Land Management).

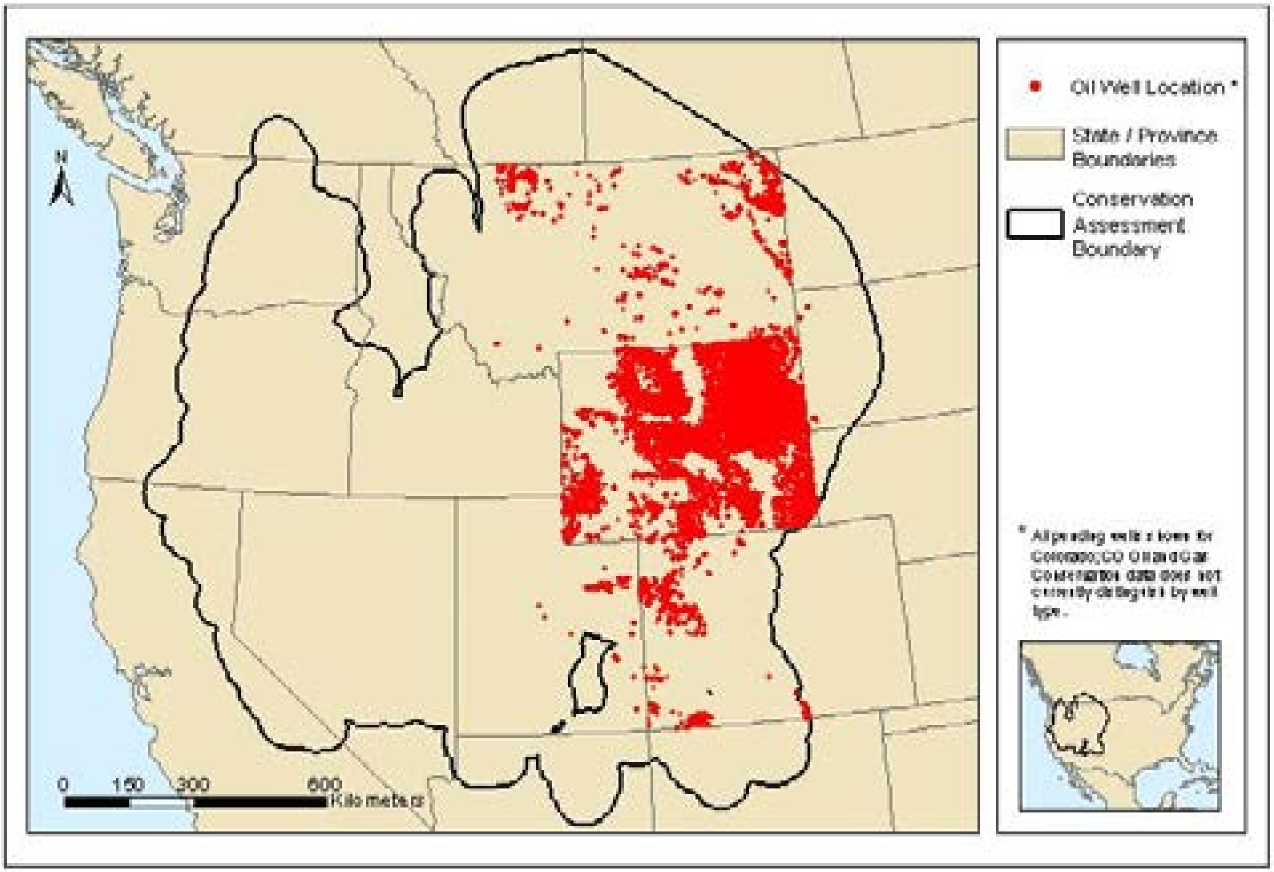

Fig. 7.28. Current producing gas wells in the sagebrush biome. (Compiled from 5 state-based Oil and Gas Conservation Commissions, 2 state Geological Surveys, and U.S. Bureau of Land Management)

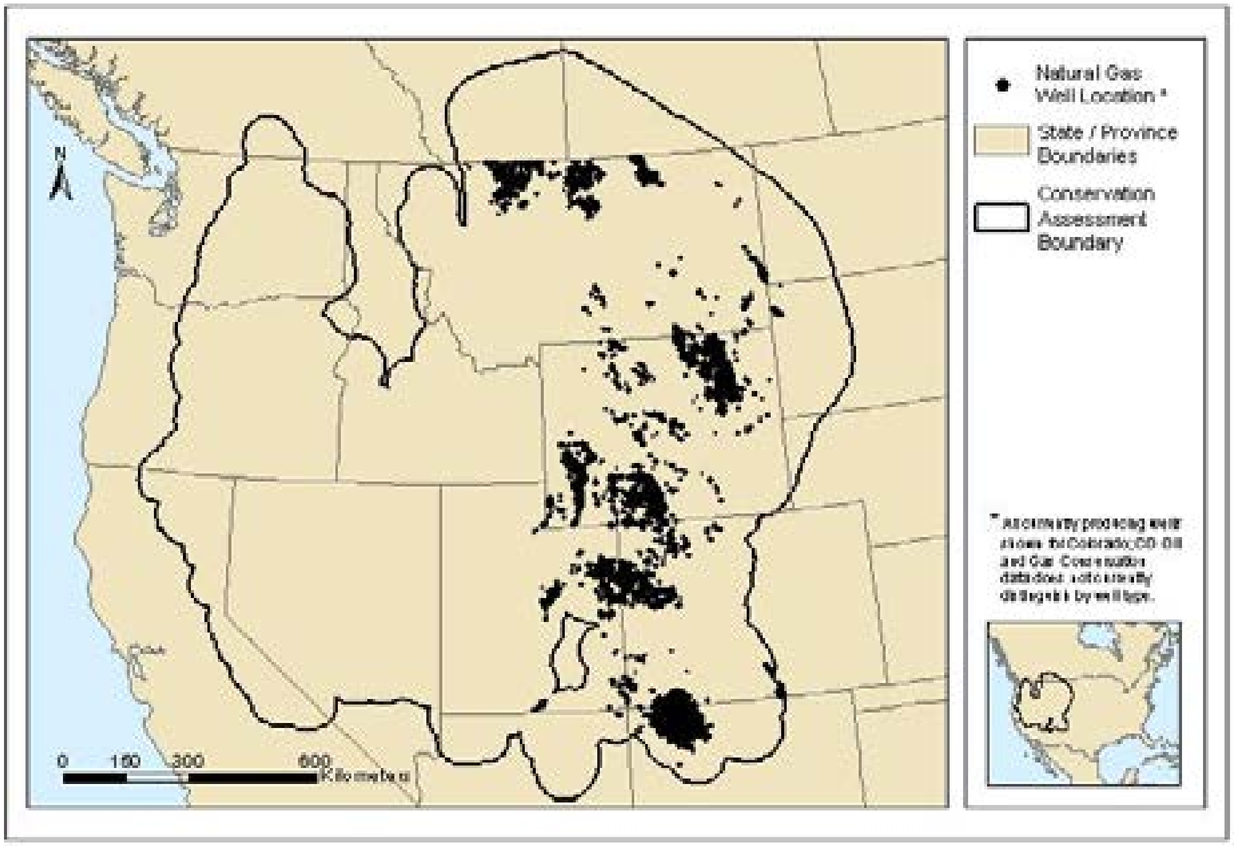

Fig. 7.29. Permitted or pending natural gas wells in the sagebrush biome (1929-2004). (Compiled from 5 state-based Oil and Gas Conservation Commissions, 2 state Geological Surveys, and U.S. Bureau of Land Management).

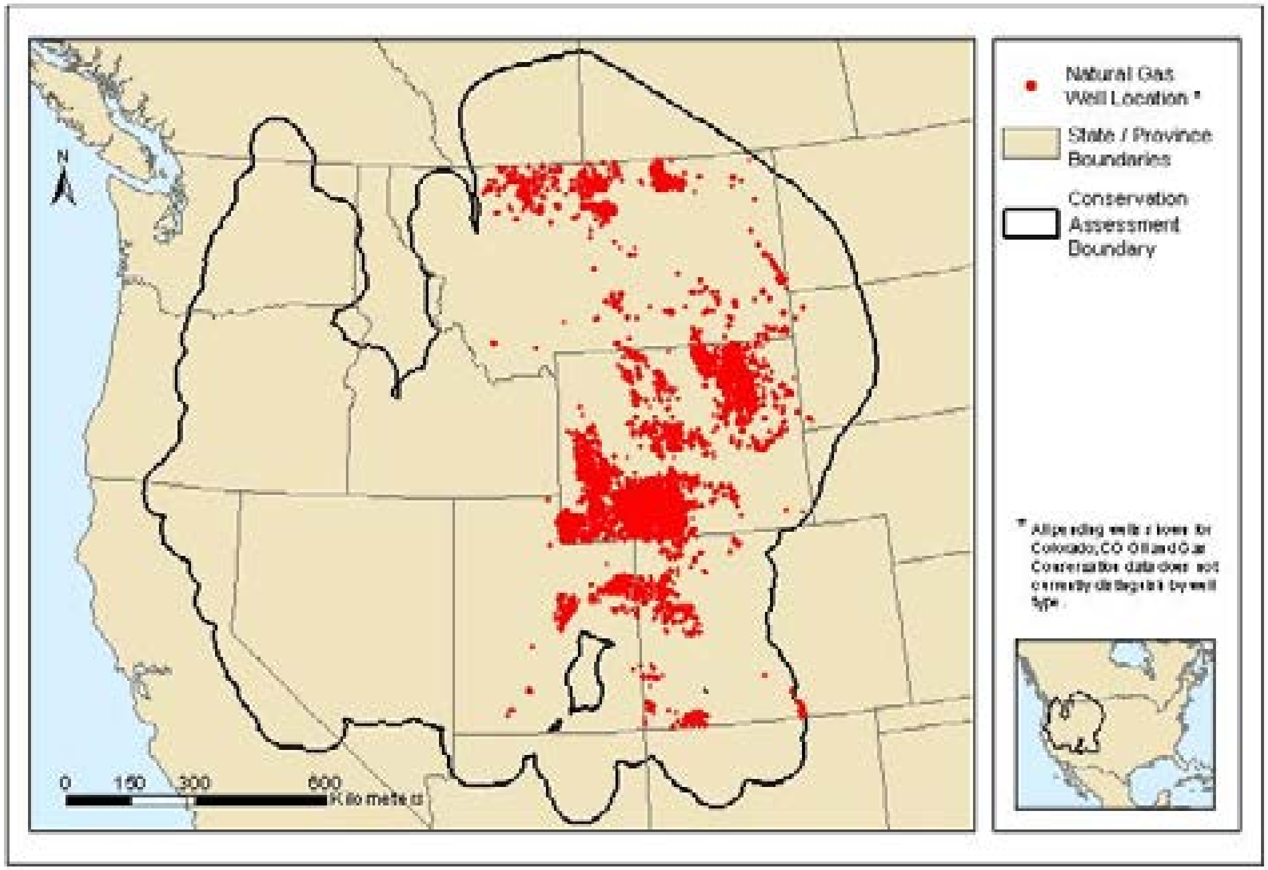

Fig. 7.30. Density of primary natural gas and oil pipelines in the Conservation Assessment study area. (Developed from U.S. Department of Transportation and 17 other sources).

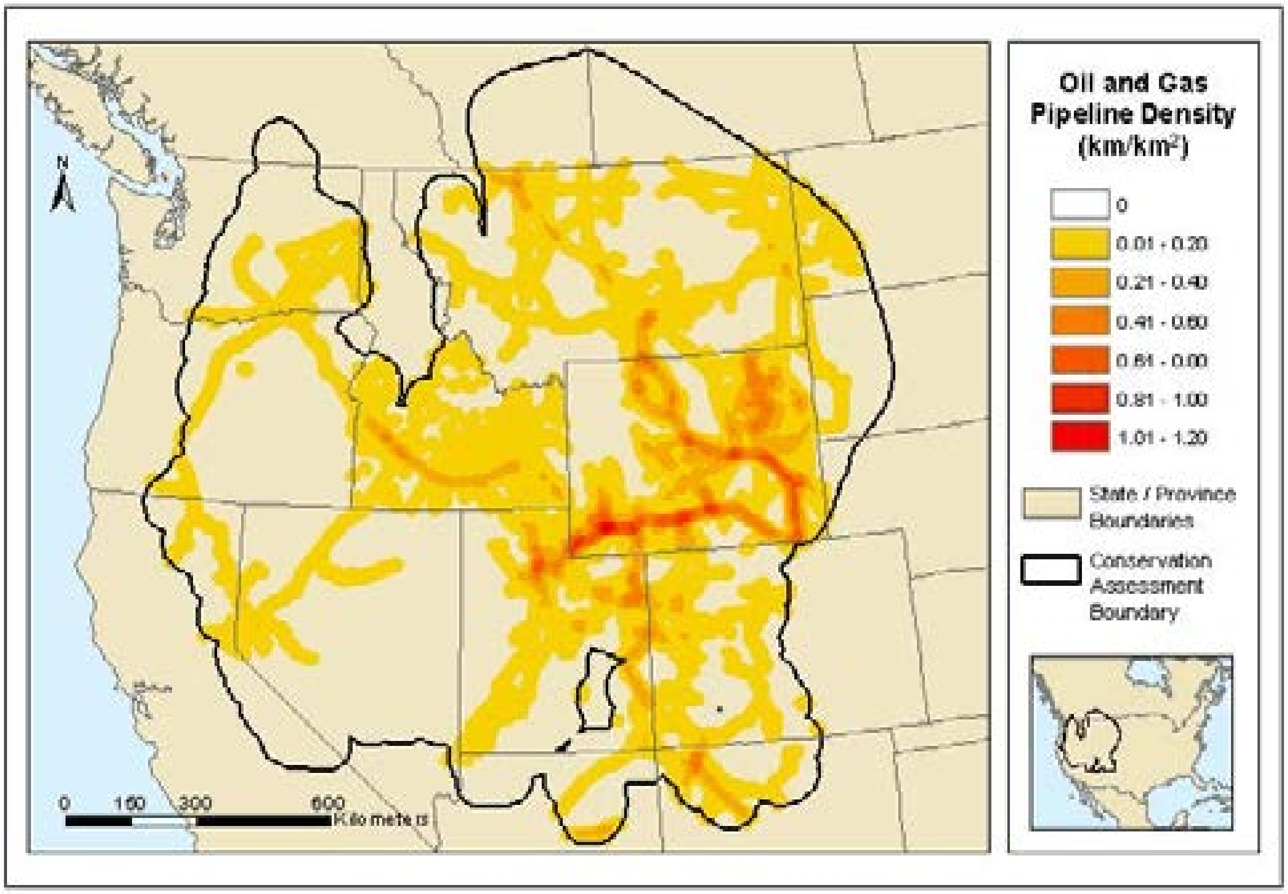

Fig. 7.31. Potential for wind energy in the western United States

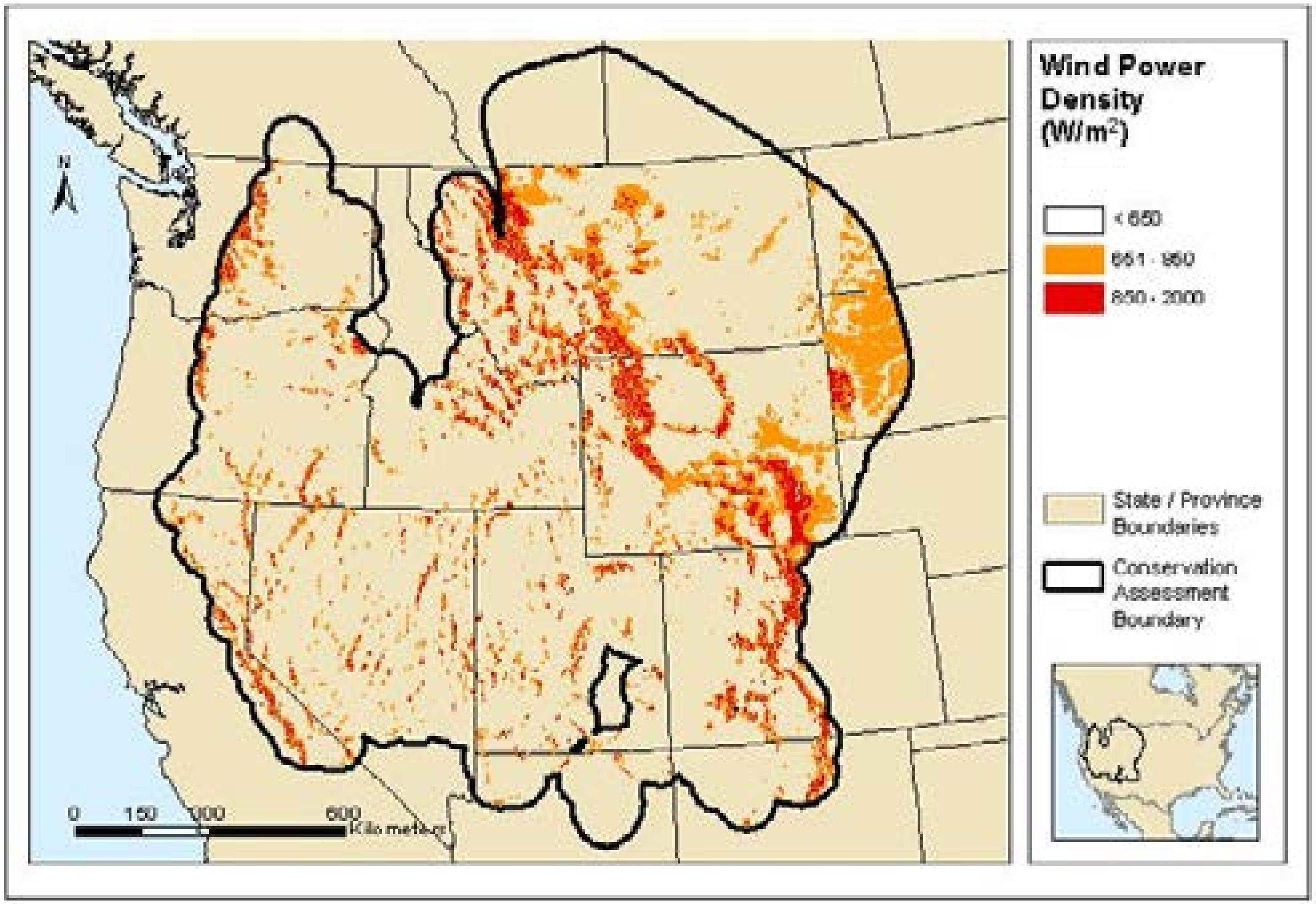

Fig. 7.32. Locations used for military training and testing within the sagebrush biome. (Compiled from U.S. Department of Defense).

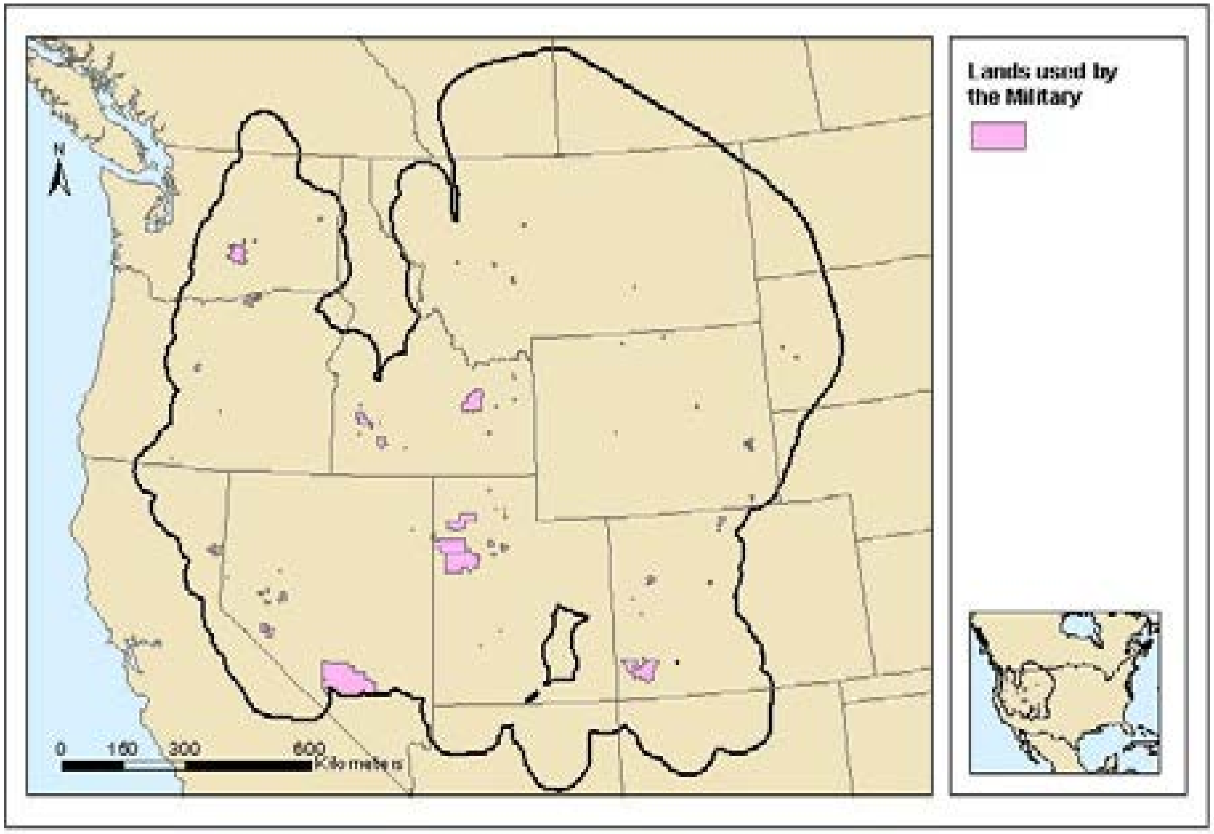

Fig. 7.33. Conceptual model showing plant dynamics using state and transition (dotted boxes and dashed lines) in a typical shrub grassland site within the sage-grouse range. Solid boxes and arrows within states are plant communities and pathways.

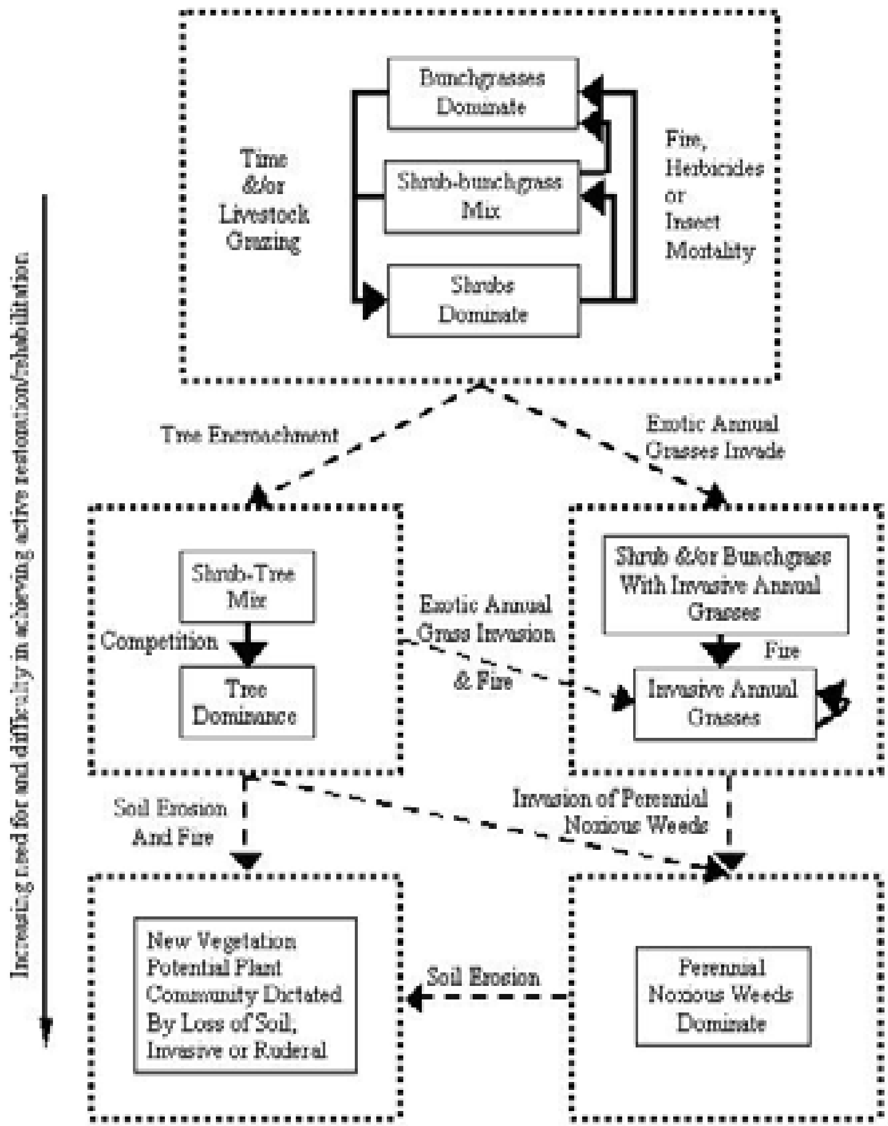

Table 7.1. Results of univariate linear regressions to detect changes in number and average size of fires, total area burned, and variation in fire size by year within floristic divisions from 1980 to 2003. The model was a time-series model regressing year by fire variables. Dependent variables were log-transformed. Dividing by mean fire size standardized variation in within-year fire size. We conducted our analysis only for fires recorded after 1980 because of more consistent reporting even though fire statistics were available prior to 1980 (Fig. 7.1-5, also 7.6).

Floristic Division (number of fire years 1980 to 2003)	Dependent Variable	F =	P =	r^2	Coefficient[1]
Northern Great Basin (23)	Fires (n)	91.00	0.01	0.81	0.10
	Avg. Size (ha)	0.13	0.72	0.04	0.00
	Total Area (ha)	9.17	0.01	0.27	0.06
	Variance	2.83	0.11	0.08	-0.04
Southern Great Basin (23)	Fires (n)	124.51	0.01	0.85	0.16
	Avg. Size (ha)	14.83	0.01	0.39	0.09
	Total Area (ha)	29.86	0.01	0.57	0.16
	Variance	20.11	0.01	0.46	-0.09
Silver Sagebrush Region (18)	Fires (n)	151.10	0.01	0.90	0.33
	Avg. Size (ha)	0.78	0.39	0.01	0.04
	Total Area (ha)	74.76	0.01	0.81	0.30
	Variance	6.73	0.02	0.25	-0.05
Snake River Basin (23)	Fires (n)	53.48	0.01	0.70	0.06
	Avg. Size (ha)	1.20	0.29	0.01	0.01
	Total Area (ha)	21.88	0.01	0.49	0.05
	Variance	1.65	0.21	0.03	-0.02
Wyoming Basins (21)	Fires (n)	38.98	0.01	0.66	0.26
	Avg. Size (ha)	5.88	0.03	0.20	0.05
	Total Area (ha)	41.96	0.01	0.67	0.26
	Variance	1.69	.0.21	0.03	-0.02

[1]Log-transformed coefficient

Table 7.2. Fires (km²) on or adjacent to lands managed by the U.S. Bureau of Land Management from 1997 through 2002 (U.S. Bureau of Land Management, Public Land Statistics 1997, 1998, 1999, 2000, 2001, 2002). Force account protection.

	2002		2001		2000		1999		1998		1997	
	BLM	nonBLM	BLM	nonBLM	BLM	nonBLM	BLM	nonBLM	BLM	nonBLM	BLM	nonBLM
Arizona	33	0	8	2	4	<1	92	6	8	<1	6	0
California	88	97	30	7	34	11	81	96	17	30	98	554
Colorado	26	24	1	<1	1	4	34	7	9	1	1	<1
Idaho	127	85	323	178	1146	460	1265	127	87	70	118	65
Montana	3	6	<1	<1	2	24	4	<1	4	<1	1	<1
Nevada	84	30	1456	168	1951	229	3995	1254	160	69	92	19
New Mexico	82	82	1,0	5	250	33	1,6	20	1	<1	18	<1
North Dakota	0	0	0	0	0	0	0	0	0	0	0	0
Oregon	413	23	600	50	600	143	46	6	130	107	43	46
South Dakota	0	0	0	0	0	0	0	0	0	0	0	0
Utah	16	16	57	62	315	99	249	119	136	79	51	10
Washington	4	16	<1	0	3	25	4	12	0	0	0	0
Wyoming	1	1	14	33	101	200	3	3	2	1	4	3
Total km²	878	380	2,502	504	4,405	1,228	5,787	1,650	554	356	432	696

Table 7.3. Fires (km^2) on or adjacent to lands managed by the U.S. Bureau of Land Management from 1997 through 2002 (U.S. Bureau of Land Management, Public Land Statistics 1997, 1998, 1999, 2000, 2001, 2002). Contract account protection.

	2002		2001		2000		1999		1998		1997	
	BLM	nonBLM	BLM	nonBLM	BLM	nonBLM	BLM	nonBLM	BLM	nonBLM	BLM	nonBLM
Arizona	<1	0	<1	0	<1	<1	0	2	0	1	0	0
California	68	106	4	50	99	181	37	73	29	71	53	133
Colorado	0	<1	0	<1	0	0	<1	2	1	0	0	0
Idaho	9	7	7	13	61	233	20	45	2	227	8	4
Montana	4	21	0	<1	2	1	2	2	4	2	0	4
Nevada	<1	10	1	190	24	84	146	202	7	56	13	9
New Mexico	<1	1	1	1	7	28	<1	3	4	7	<1	1
North Dakota	0	0	0	0	0	0	0	0	0	0	0	0
Oregon	1	1	<1	1	3	8	<1	0	29	100	2	24
South Dakota	0	0	0	0	0	0	0	0	0	0	0	0
Utah	5	15	<1	<1	43	26	47	124	5	1	2	7
Washington	8	3	3	5	0	<1	0	0	0	0	0	0
Wyoming	1	2	1	22	1	4,8	<1	0	0	<1	1	1
Total km^2	95	165	16	283	239	609	253	453	80	465	78	178

Table 7.4. Characteristics of ecological provinces in the Great Basin Ecoregion and adjacent ecoregions (Miller et al. 1999*a*).

Ecological province	Precipitation (cm) pattern	Dominant overstory species	Old-growth character	Mean annual temperature (C)	Elevation (m)
John Day	(23 – 46) Winter	Western juniper	<3%	-2 – 13	1,200 – 2,300
Mazama	(18 – 36) Winter	Western juniper	>10%	4 – 13	700 – 2,500
Snake River	(13 – 31) Winter	--	<3%	4 – 13	900 – 2,000
High Desert	(10 – 79) Winter	Western juniper	Dense <5%	5 – 10	1,200 – 3,000
Klamath	(102 – 305) Winter	--	<3%	7 – 13	460 – 2,800
Humboldt	(20 – 76) Winter	Western juniper, Utah juniper	Dense and savanna <3%	4 – 10	1,500 – 3,300
Raft River	(40 – 100) Winter	Juniper, Singleleaf pinyon pine	Dense and savanna <3%	1 – 9	--
Mono	(25 – 64) Winter	Singleleaf pinyon pine	Dense <5%	4 – 10	1,200 – 4,300
Lahontan	(10 – 30) Winter	Singleleaf pinyon pine, Utah juniper	Savanna 0%	7 – 13	1,200 – 3,000
Central High	(13 – 62) Winter and summer	Pinyon pine, juniper	Savanna <3%	4 – 10	1,500 – 4,000
High Calcereous	(13 – 62) Winter and summer	Juniper, pinyon pine	Savanna <3%	4 – 10	1,500 – 4,000
Bonneville	(10 – 25) Winter and summer	Juniper	Savanna >10%	7 – 13	1,200 – 2,400
White River	(8 – 51) Winter and summer	--	Dense and savanna	11 – 15	1,400 – 2,900

Table 7.5. Broad vegetation cover types within the assessment area and their susceptibility to invasion by non-indigenous plant species. Susceptibility to invasion is defined by four categories: (H) high – invades the cover type successfully and becomes dominant or codominant even in the absence of intense or frequent disturbances; (M) moderate - invades the cover type successfully because high intensity or frequency of disturbance impacts the soil surface or removes the normal canopy cover; (L) low – the species typically does not invade the cover type because the cover type does not provide suitable habitat for the species; (U) unknown – distribution records are limited and interpretation of the susceptibility would be difficult. Compiled from USDA FS and USDI BLM (1997*b*) and Sheley and Petroff (1999).

Cover Type	Invasive Species[1]																													
	Brte	*Canu*	*Casp.*	*Cedi*	*Cema*	*Cere*	*Ceso*	*Cevi*	*Chju*	*Chle*	*Ciar*	*Civu*	*Coma*	*Crvu*	*Eues*	*Hagl*	*Hiau*	*Hipr*	*Isti*	*Lela*	*Lida*	*Livu*	*Lysa*	*Onac*	*Pore*	*Saae*	*Saka*	*Seja*	*Sosp.*	*Taca*
UPLAND COMMUNITIES																														
Basin & Wyoming big & three-tip sagebrush *Artemisia tridentata* ssp. *tridentata*, *A.tridentata.* ssp. *wyomingensis*,*A. tripartita*	H	U	M	M	M	M	M	M	M	U	M	M	L	L	M	M	L	L	H	L	M	M	L	M	U	H	M	U	M	M
Mountain big sagebrush *A. tridentata* ssp. *vaseyana*	H	M	M	M	M	M	M	M	M	U	M	M	L	M	L	M	M	L	L	L	H	M	M	L	M	M	M	U	M	M
Low & black sagebrush *A. arbuscula*, *A.nova*	M	U	M	M	U	U	M	M	U	U	M	M	L	L	M	M	L	L	H	L	M	U	L	U	U	U	L	U	M	M
Salt desert shrub *Atriplex confertifolia*, *Eurotia lanata*, *Grayia spinosa*,*Sarcobatus vermiculatus*	M	M	M	L	L	M	L	M	L	L	M	M	L	L	M	H	L	L	L	L	L	L	L	L	L	L	M	U	M	L
Wheatgrass bunchgrass *Agropyron* sp., *Aristida longiseta*, *Elymus cinereus*, *Poa secunda*, *Pseudoroegnaria spicata*	H	M	M	H	H	M	H	M	M	M	H	M	L	M	M	M	L	L	H	L	H	M	L	M	H	H	M	U	M	M
RIPARIAN & WETLAND COMMUNITIES																														
Shrub wetlands	M	H	M	M	H	M	M	M	L	M	H	H	H	L	M	L	M	M	M	H	M	M	H	M	M	U	L	U	L	M

Cover Type	Invasive Species[1]																													
	Brte	*Canu*	*Ca*sp.	*Cedi*	*Cema*	*Cere*	*Ceso*	*Cevi*	*Chju*	*Chle*	*Ciar*	*Civu*	*Coma*	*Crvu*	*Eues*	*Hagl*	*Hiau*	*Hipr*	*Isti*	*Lela*	*Lida*	*Livu*	*Lysa*	*Onac*	*Pore*	*Saae*	*Saka*	*Seja*	*So*sp.	*Taca*
Cornus sp., *Salix* sp.																														
Cottonwood *Populus trichocarpa*	M	M	M	M	H	M	M	M	M	H	H	M	H	L	H	L	M	H	M	H	M	M	M	M	M	M	L	U	L	M
Herbaceous Wetland *Carex nebraskensis, C. rostrata, C. aquatilis*	M	M	M	M	H	M	H	M	L	H	H	M	H	L	M	L	H	M	M	H	M	M	H	H	M	H	L	U	L	M

[1] *Brte Bromus tectorum* (cheatgrass); *Canu Carduus nutans* (musk thistle); *Ca* sp. *Cardaria* species (whitetop); *Cedi Centaurea diffusa* (diffuse knapweed); *Cema Centaurea maculosa* (spotted knapweed); *Cere Centaurea repens* (Russian knapweed); *Ceso Centaurea solstitialis* (yellow starthistle); *Cevi Centaurea virgata* (squarrose knapweed); *Chju Chondrilla juncea* (Rush skeletonweed); *Chle Chrysanthemum leucanthemum* (Oxeye daisy); *Ciar Cirsium arvense* (Canada thistle); *Civu Cirsium vulgare* (bull thistle); *Coma Conium maculatum* (poison-hemlock); *Crvu Crupina vulgaris* (common crupina); *Eues Euphorbia esula* (leafy spurge); *Hagl Halogeton glomeratus* (halogeton); *Hiau Hieracium aurantiacum* (orange hawkweed); *Hipr Hieracium pratensis* (meadow hawkweed); *Isti Isatic tinctoria* (dyer's woad); *Lela Lepidium latifolium* (perennial pepperweed); *Lida Linaria dalmatica* (Dalmation toadflax); *Livu Linaria vulgaris* (yellow toadflax); *Lysa Lythrum salicaria* (purple loosestrife); *Onac Onopordum acanthium* (Scotch thistle); *Pore Potentilla recta* (sulphur cinquefoil); *Saae Salvia aethiopis* (Mediterranean sage); *Saka Salsola kali* (Russian thistle); *Seja Senecio jacobaea* (tansy ragwort); *So* sp. *Sonchus* sp. (sowthistles); *taca Taeniatherum caput-medusae* (medusahead)

Table. 7.6. Mean population density (in millions) within states from U.S. Census Statistics from 1900 to 2000.

	Year of United States Census										
State	2000	1990	1980	1970	1960	1950	1940	1930	1920	1910	1900
Arizona	3.26	2.46	1.90	1.18	0.91	0.71	0.62	0.47	0.36	0.26	0.21
California	17.70	14.56	10.71	7.46	6.05	4.88	3.72	2.75	2.34	2.32	2.21
Colorado	16.06	12.59	10.93	7.47	5.11	3.75	3.38	2.86	2.69	3.02	2.89
Idaho	10.84	8.31	7.72	5.76	5.36	4.82	4.22	3.58	3.43	4.12	2.12
Montana	2.60	2.33	2.36	2.17	2.19	1.99	1.93	1.94	2.14	2.77	2.44
North Dakota	1.49	1.60	1.73	1.64	1.81	1.76	1.85	2.17	2.02	2.15	0.93
Nebrasaka	4.18	4.18	4.48	4.23	4.30	4.16	4.11	3.94	3.06	1.79	1.04
New Mexico	11.77	10.43	10.10	8.25	7.25	6.00	2.43	1.94	1.77	1.63	1.57
Nevada	14.95	10.41	7.70	3.98	2.21	1.23	0.98	0.73	0.79	1.22	1.10
Oregon	36.75	31.69	30.06	27.44	24.92	22.18	16.58	15.41	12.67	11.55	6.03
South Dakota	2.74	2.54	2.40	2.09	2.11	1.84	1.79	1.67	1.39	2.10	1.98
Utah	29.38	23.15	19.59	14.50	11.96	8.76	6.66	6.12	5.24	4.71	3.26
Washington	30.06	25.48	22.28	19.16	16.51	13.65	9.74	8.89	8.02	7.19	3.61
Wyoming	2.17	1.99	2.01	1.48	1.52	1.35	1.18	1.07	1.03	1.16	0.79

Table. 7.7. Recreational use (number of visitors and visitor days) on lands managed by the U.S. Bureau of Land Management (U.S. Bureau of Land Management, Public Land Statistics 2000, 2001, 2002). Number of visitors represents the total of recreational visits to all lands, including those managed specifically for recreation or other opportunities. Number of visitor-days is an aggregate of 12 visitor-hours to a site or area.

	2002		2001		2000	
	Visitors	Visitor Days	Visitors	Visitor Days	Visitors	Visitor Days
Arizona[1]	1,778	4,393	4,572	14,102	4,997	15,515
California	8,947	16,747	8,312	13,684	8,400	10,610
Colorado	4,440	3,394	4,892	3,410	4,756	3,206
Idaho	6,090	4,710	5,895	4,296	6,326	4,513
Montana	2,845	2,459	2,955	3,405	3,136	3,523
Nevada	6,438	5,184	5,275	3,606	5,045	4,110
New Mexico	2,090	1,788	2,533	1,800	2,380	1,667
Oregon	7,635	6,511	8,265	7,220	8,137	9,200
Utah	5,733	7,757	5,795	7,055	6,169	7,812
Wyoming	2,022	2,282	1,931	2,043	3,655	2,285
Total	48,018	55,225	50,425	60,621	53,001	62,441

[1]Does not include 2,905 visitors and 10,036 visitor days to leased or partnership sites that previously were included in previous year's totals.

Table 7.8. Number of habitat treatments on lands managed by the U.S. Bureau of Land Management (U.S. Bureau of Land Management Range Improvement Projects database).

	State													
Habitat Treatment	AZ	CA	CO	ID	MT	ND	NM	NV	OR	SD	UT	WA	WY	Total
Unknown	9	20	15	23	30		12	40	18		91		25	283
Fences	1,489	669	3,051	3,717	5,790	48	1,445	3,489	3,451	357	3,265	239	5,278	32,288
Fence modification	1	6	9	149	17		7	23	89		17	4	23	345
Hazard reduction	1	8	28	47	9		1	5	10		2		4	115
Land treatment	30	15	253	158	82		78	77	54	4	141	11	73	976
Lake/wetland improvement	21		8	8	17		2		7	1	2	10	8	84
Misc. facility improvement	468	173	1,038	1,254	1,420		368	1,480	1,263	10	1,300	41	644	9,459
Perch/nesting structure			2	6	7		2	3	3		6	11	44	84
Stream improvement		8	35	11	13		3	1	14		17	4	22	128
Streambank stabilization structures	1		18	4	13		1	2	11		1	3	2	56
Timber stand improvement		8	23	9	1		3	1	15		1	1	12	74
Vegetation manipulation	220	207	880	1,585	445	9	681	895	1,089	14	824	66	481	7,396
Weed control		2	138	67	320	3	31	12	126	15	14	294	42	1,064
Water development/control	2,180	1,180	9,189	5,017	10,587	56	2,247	5,163	7,555	561	5,292	130	7,813	56,970
Water facilities modified		15	49	65	15		3	3	126	1	20	2	38	337
Total habitat treatments	4,420	2,311	14,736	12,120	18,766	116	4,884	11,194	13,831	963	10,995	816	14,509	109,661

Table 7.9. Summary of Grazing Permits and Leases in Force on Public Lands from 2000 through 2002 (U.S. Bureau of Land Management, Public Land Statistics 2000, 2001, 2002)

	2002		2001		2000	
	Permits	AUMs	Permits	AUMs	Permits	AUMs
Arizona	741	685,257	767	676,970	785	678,962
California	627	328,404	593	316,971	603	329,395
Colorado	1,627	648,693	1,609	644,603	1,639	691,046
Idaho	1,959	1,322,974	1,939	1,317,041	1,962	1,323,215
Montana	3,787	1,293,498	3,755	1,286,197	3,832	1,302,641
Nebraska	20	622	18	592	21	622
Nevada	658	2,241,596	661	2,221,140	683	2,225,814
New Mexico	2,313	1,875,361	2,312	1,872,958	2,321	1,875,181
North Dakota	75	9,389	75	9,389	74	9,379
Oklahoma	4	138	4	138	4	138
Oregon	1,395	1,059,047	1,296	1,036,769	1,359	1,041,707
South Dakota	462	74,126	466	74,387	456	73,448
Utah	1,569	1,234,136	1,550	1,236,840	1,589	1,246,639
Washington	363	29,475	324	30,536	298	28,239
Wyoming	2,782	1,973,653	2,773	1,972,836	2,767	1,984,061
Total	18,382	12,776,369	18,142	12,697,367	18,393	12,810,487

Table 7.10. Percent of lands by seral condition on public lands managed by the U.S. Bureau of Land Management in 2003[a,b]. Potential natural climax = 76-100% similar; late seral = 51-75% similar; mid-seral = 26-50% similar; early seral = 0-25% similar[c].

	% Inventoried	Potential Natural Community	Late Seral	Mid-Seral	Early-Seral	Unclassified
Arizona	76	6	32	26	7	30
California	16	3	20	43	30	4
Colorado	58	5	20	40	18	27
Idaho	54	3	27	38	27	5
Montana, North Dakota, South Dakota	78	7	58	21	1	13
Nevada	54	3	27	31	8	31
New Mexico	80	4	30	41	22	3
Oregon and Washington	82	1	21	45	11	23
Utah	62	11	28	42	13	6
Wyoming	56	24	34	27	5	10
Total	61	7	29	34	11	18

[a]http://www-a.blm.gov/natacq/pls03/

[b]Statistics are reported for 2003 although surveys to classify sites may have been conducted in other years.

[c]A common qualitative characterization of plant communities is for potential natural communities to be considered as in "excellent condition", late seral stages to be "good", mid-seral to be "fair", and early seral to be "poor". Because these terms may not accurate describe current understanding of succession dynamics in sagebrush ecosystems (West 2003), the U.S. Bureau of Land Management is considering changes to standards and guidelines that indicate "rangeland health" (U.S. Bureau of Land Management 2003*b*) (Table 7.11).

Table 7.11. Total area (km^2 and %) of lands managed by the U.S. Bureau of Land Management meeting standards and guidelines established for rangeland health[1]. Category A = meeting standards or making significant progress towards meeting standards. Category B = Not meeting all standards or making significant progress but appropriate action has been taken to ensure significant progress towards meeting standards (livestock is a significant factor). Category C = Not meeting standards or making significant progress towards meeting the standards and no appropriate action has been taken to ensure significant progress (livestock is a significant factor). Category D = Not meeting standards or making significant progress towards meeting the standards due to causes other than livestock grazing. Category E = total area assessed. Category F = total area that has not been assessed.

	Category A		Category B		Category C		Category D		Category E		Category F		
	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	Total
Arizona	20,264	42	1,434	3	859	2	109	0	22,667	47	25,605	53	48,272
California	12,287	39	2,637	8	1,477	5	2,078	7	18,479	58	13,392	42	31,870
Colorado	14,588	41	5,787	16	99	0	1,404	4	21,878	62	13,597	38	35,475
Idaho	8,384	18	12,575	26	1,826	4	2,525	5	25,310	53	22,449	47	47,760
Montana/Dakotas	19,745	60	1,919	6	599	2	661	2	22,924	70	10,032	30	32,957
Nevada	25,914	14	39,174	21	2,720	1	4,167	2	71,975	39	113,035	61	185,010
New Mexico	1,466	3	374	1	0	0	24	0	1,864	4	48,529	96	50,392
Oregon/Washington	12,904	23	4,335	8	2,044	4	1,133	2	20,416	36	35,835	64	56,251
Utah	19,487	21	4,640	5	1,052	1	5,016	6	30,194	33	60,722	67	90,917
Wyoming	25,592	35	14,278	20	2,654	4	778	1	43,303	60	29,074	40	72,377
Total	160,629	25	87,154	13	13,331	2	17,897	3	279,011	43	372,270	57	651,281

[1]www.blm.gov/nstc/rangeland/rangelandindex.html

Table 7.12. Total area (km^2) seeded on lands managed by the U.S. Bureau of Land Management for wildlife habitat improvement (U.S. Bureau of Land Management, Public Land Statistics 1997, 1998, 1999, 2000, 2001, 2002).

	Year					
State	2002	2001	2000	1999	1998	1997
Arizona	1	<1	57	1	0	16
California	0	1	0	<1	5	4
Colorado	25	21	3	1	1	<1
Idaho	0	0	0	4	0	283
Montana	<1	8	0	2	1	3
Nevada	265	202	1,582	90	45	23
New Mexico	<1	1	2	6	0	<1
North Dakota	0	0	0	0	0	0
Oregon	0	93	20	1	3	6
South Dakota	0	0	0	0	0	0
Utah	14	0	7	67	0	74
Washington	0	0	0	0	0	0
Wyoming	1	12	2	<1	0	0
Total area (km^2)	306	339	1,673	173	55	410

Table 7.13. Total area (km^2) treated with disking and chaining for wildlife improvements on lands managed by the U.S. Bureau of Land Management from 1997 through 2002. (U.S. Bureau of Land Management, Public Land Statistics 1997, 1998, 1999, 2000, 2001, 2002).

	Year					
State	2002	2001	2000	1999	1998	1997
Arizona	<1	0	2	1	<1	1
California	0	0	0	0	<1	<1
Colorado	37	0	<1	<1	1	1
Idaho	0	0	0	0	0	0
Montana	0	0	0	1	1	3
Nevada	13	0	8	0	3	0
New Mexico	3	1	13	9	0	<1
North Dakota	0	0	0	0	0	0
Oregon	0	10	1	<1	4	3
South Dakota	0	0	0	0	0	0
Utah	23	0	4	9	0	190
Washington	0	0	0	0	0	0
Wyoming	1	5	0	2	0	23
Total area (km^2)	77	16	29	23	9	221

Table 7.14. Total area (km^2) treated by prescribed fire projects on lands managed by the U.S. Bureau of Management from 1997 to 2002. (U.S. Bureau of Land Management, Public Land Statistics 1997, 1998, 1999, 2000, 2001, 2002)

	Year					
State	2002	2001	2000	1999	1998	1997
Arizona	52	48	40	94	95	8
California	7	3	22	21	45	16
Colorado	24	2	2	46	56	3
Idaho	59	100	43	102	64	11
Montana	12	34	15	31	18	3
Nevada	28	10	<1	3	2	10
New Mexico	53	34	15	71	131	60
North Dakota	0	0	0	0	0	0
Oregon	223	86	55	225	169	72
South Dakota	0	0	0	0	0	0
Utah	19	64	33	30	24	11
Washington	0	0	4	1	0	0
Wyoming	20	11	23	138	117	59
Total area (km^2)	497	393	251	764	720	253

Table 7.15. Nonfire fuels treatment (km^2) on lands managed by the U.S. Bureau of Land Management (U.S. Bureau of Land Management. Public Land Statistics1999, 2000, 2001, 20002). Includes mechanical (e.g, chaining), hand (e.g., thinning and pulling), and chemical (e.g., herbicide) treatments.

	Year			
State	2002	2001	2000	1999
Arizona	46	32	10	<1
California	14	4	5	7
Colorado	60	74	9	15
Idaho	228	324	277	20
Montana	15	0	<1	0
Nevada	40	6	123	0
New Mexico	150	4	5	4
North Dakota	0	0	0	0
Oregon	160	55	27	6
South Dakota	0	0	0	0
Utah	63	0	1	14
Washington	1	1	0	0
Wyoming	54	52	2	0
Total area (km^2)	832	553	460	66

Table 7.16. Number of projects (n), total area (km²) treated, and funding (USD) approved for emergency fire rehabilitation on lands managed by the U.S. Bureau of Land Management from 1997 through 2002 (U.S. Bureau of Land Management, Public Land Statistics 1997, 1998, 1999, 2000, 2001, 2002).

	Year																	
	2002			2001			2000			1999			1998			1997		
State	n	km²	Funding ($)	n	km²	Funding ($)	n	km²	Funding ($)	n	km²	Funding ($)	n	km²	Funding ($)	n	km²	Funding ($)
Arizona										1	67	1,021,000		0			0	
California	19	34	574,591	6	2,3	1,647,405	7	168	1,148,647	5	11	18,154,000		0			0	
Colorado	16	52	2,816,202	4	2,3	167,484	15	77	1,094,737	5	45	295,000	9	6	93,859		0	
Idaho	85	2,900	2,292,252	31	389	4,577,439	38	1,216	10,605,954	41	1,267	7,802,576	8	53	410,379	14	45	473,195
Montana	3	8	1,064,471	0	0		5	65	461,000		0			0			0	
Nevada	103	11,226	17,061,510	74	1,570	23,475,497	41	1,924	24,371,902	40	4,696	44,102,482	7	81	1,367,045	14	204	1,734,644
New Mexico		0			0			0			0			0			0	
North Dakota		0			0			0			0			0			0	
Oregon	50	1,392	4,794,421	22	856	6,656,000	9	450	4,841,000	10	48	655,000	7	94	880,000		0	
South Dakota		0			0			0			0			0			0	
Utah	27	409	2,309,559	29	65	1,969,000	32	262	8,111,398	20	188	6,050,054	21	121	2,577,474	7	32	439,300
Washington		0			0			0			0			0			0	
Wyoming	5	113	423,125	1	2	590,300	8	460	1,353,671	1	1	49,315		0			0	
Total	308	16,135	31,336,131	167	2,927	39,083,125	155	4,621	51,988,309	123	6,324	78,129,427	52	341	5,328,757	35	281	2,647,139

Table 7.17. Estimates of numbers of free-roaming horses and burros on lands administered by the U.S. Bureau of Land Management in the western United States in 2003, and the sum of Appropriate Management Levels [AMLs] (summed across all Herd Management Areas [HMAs] in the state). AML is determined through a multiple-use process, in which animal-unit months (AUMs) are distributed among free-roaming equids, livestock, and native ungulates. Estimated population sizes do not include animals occurring on other public jurisdictions.

	Horses		Burros		Total Equid
State	Population	AML	Population	AML	Population
Arizona	250	210	2,243	1,580	2,493
California	2,607	1,733	810	416	3,417
Colorado	516	812	0	0	516
Idaho	697	999	0	0	697
Montana	170	105	0	0	170
Nevada	16,857	13,648	919	976	17,776
New Mexico	68	50	0	0	68
Oregon	2,870	2,615	15	25	2,885
Utah	2,155	1,835	126	120	2,281
Wyoming	5,035	3,725	0	0	5,035
Total	31,225	25,732	4,113	3,117	38,815

Table 7.18. Number of applications for oil and gas leases received by federal agencies[a] from 1929 to 2004. (Data obtained from U.S. Bureau of Land Management LR2000 Lands and Records database).

State	Number of leases								
	Authorized		Pending		Rejected		Withdrawn		Total
Wyoming	50,852		606		61		1,098		52,617
Arizona	128		7		0		0		135
California	1,510		134		4		104		1,752
Colorado	13,498		308		8		102		13,916
Idaho	16		0		0		0		16
Montana	10,180		1,360		4		48		11,592
New Mexico	25,695		140		4		84		25,923
Nevada	2,272		201		20		8		2,501
Oregon	73		9		0		0		82
Washington	500		0		0		0		500
Utah	8,475		362		2		42		8,881
South Dakota	467		100		0		2		569
North Dakota	3,568		440		0		4		4,012
Total (%)	117,234	(95.7)	3,667	(3.0)	103	(0.1)	1,492	(1.2)	122,496

[a]Prior to 1972, the U.S. Geological Survey was the approval authority for applications for permit to drill. The U.S. Bureau of Land Management was authorized to provide input into the decision process in 1972 and became the approving authority in 1982 following a merger with the Minerals Management Service.

Table 7.19. Geologic basins (Fig. 7.25) where significant oil and gas resources have been developed within the sagebrush biome, total area (km^2), and area (km^2) of sagebrush included within the basin. (Data obtained from U.S. Geological Survey National Oil and Gas Assessment).

Basin Name	State(s)	Total Area (km^2)	Sagebrush Area (km^2)
1. Greater Green River Basin	Wyoming, Colorado, Utah	114,137	49,118
2. Powder River Basin	Wyoming, Montana	89,178	21,061
3. Uinta-Piceance Basin	Utah, Colorado	78,493	13,393
4. Wyoming Thrust Belt	Wyoming, Utah, Idaho	29,211	11,934
5. Paradox - San Juan Basin	Utah	158,779	18,498
Wind River Basin	Wyoming	30,312	15,787
Bighorn Basin	Wyoming, Montana	34,087	10,567
Williston Basin	Montana, North Dakota, South Dakota	89,178	9,442
North-Central Montana	Montana	123,102	6,273

Table 7.20. Major category of stipulations on oil/gas development, total area (km^2) and percent of federal lands affected in five major and priority geological basins (adapted from Table ES-1; U.S. Departments of the Interior, Agriculture and Energy 2003)

Access Category	Area (km^2)	Federal Lands (%)
No Leasing (Statutory/Executive Order)	40,745	16.9
No Leasing (Administrative) pending NEPA or LUP action.	24,310	10.1
No Leasing (Administrative), general	20,632	8.6
Leasing, no surface occupancy	10,984	4.6
Leasing, cumulative timing limitations on drilling >9 months	101	0
Leasing, cumulative timing limitations on drilling 6-9 months	10,202	4.2
Leasing, cumulative timing limitations on drilling 3-6 months	22,024	9.2
Leasing, cumulative timing limitations on drilling <3 months	2,821	1.2
Leasing, controlled surface use	15,188	6.3
Leasing, standard lease terms	93,449	38.9

Table 7.21. Area (km^2) of major oil and gas producing fields and number of wells and spacing (ha/well) for the Greater Green River Basin, Wyoming (Fig. 7.25). (Compiled from U.S. Geological Survey National Oil and Gas Assessment and U.S. Bureau of Land Management state and field office personnel).

		Number of wells			Spacing (ha/well)	
Name of Field/Area	Area (km^2)	Approved	Drilled	Potential	Current Range	Potential
Continental Divide (many fields)	4,293	2,100		3,000	32-65	32
Jonah II	241	497		497	16-32	16
Jonah II+ (proposed addition) no final proposal to date	118	Proposed but not approved		3,100 additional wells	2-16	2-16
Pinedale Anticline	638	900		900	16-65	8-65
Big/Piney LaBarge per CAP	797	No limit		No limit	4-16	4-16
Moxa Arch	1,930		≈1,500	≈2,200	65	Unknown
Fontenelle	725					
Blue Gap			81		65	Unknown
Great Divide Field, Colorado					16	16
Big Hole Field, Colorado					16	16
Little Snake Field Office, Colorado	497	2,327	1,979	3,327	16-130	8

Table 7.30. Area (km^2) of major oil and gas producing fields and number of wells and spacing for the Powder River Basin, Wyoming and Montana (Fig. 7.25). Area was determined from individual Environmental Impact Statements or mapped estimates. (Compiled from U.S. Geological Survey National Oil and Gas Assessment and U.S. Bureau of Land Management state and field office personnel).

	Area	Number of wells			Spacing (ha/well)	
Name of Field/Area	(km^2)	Approved	Drilled	Potential	Range	Potential
Powder River Basin, Wyoming (Coalbed Methane)	32,376	15,300	≈15,000	51,000	32	32
Powder River Basin, Montana (Coalbed Methane)	11,028	511	480	15,635	32	32
Salt Creek and Teapot Dome			1,600		4-16	4-16
Sand Dunes Field			30-40		16	16
Fina/Fly Draw/Phillips Cr.			30-40		65	65

Table 7.31. Area (km^2) of major oil and gas producing fields, number of wells and spacing (ha/well) for the Uinta/Piceance Basin in Utah and Colorado (Fig. 7.25). Area was determined from individual Environmental Impact Statements (EIS), Environmental Assessments (EA) or mapped estimates. (Compiled from U.S. Geological Survey National Oil and Gas Assessment and U.S. Bureau of Land Management state and field office personnel).

	Area	Number of wells			Spacing	
Name of Field/Area	(km^2)	Approved	Drilled	Potential	Range	Potential
Red Wash/Greater Deadman Bench	595	New EIS in progress	1,464	1,249	16-65	16
Natural Buttes	316	875 - EA	704	171	16-32	16
Castlegate CBM	103	154 - EIS	30	124	65	16-32
Myton Bench	265	New EIS in Progress	714	933	16-32	16
Prickly Pear	265	160 - EA	40	120	65	16-32
Altamont-Bluebell	1396		549	Unk	16-65	16

Chapter 8

Greater Sage-Grouse Genetics

CHAPTER 8

Greater Sage-grouse Genetics

Abstract. Recent research on greater sage-grouse (*Centrocercus urophasianus*) genetics has helped identify some important characteristics. First, the greater sage-grouse is genetically distinct from the congeneric Gunnison sage-grouse (*C. minimus*). Second, the Mono Lake area of California (but also Washington to a lesser extent), appears to have some unique genetic characteristics. Third, the previous delineation of western (*C. u. phaios*) and eastern (*C. u. urophasianus*) subspecies was not supported. Fourth, the two isolated populations in Washington are showing indications that genetic heterogeneity has been lost due to population declines and isolation. Additional work is being planned and conducted that will help address unanswered questions.

Introduction

Genetic techniques have only recently begun to be used in studies of wildlife. With recent technological advances, straightforward and rather inexpensive genetic techniques have emerged which can be directly applied to wildlife studies. Genetic techniques are currently being used to investigate relationships among species, populations, family groups, and individuals.

Genetic studies all examine portions of DNA at some scale. Two different genomes are used in genetic studies of animals. The nuclear genome is biparentally inherited and is found in the cell nucleus. It is large and not well mapped in most species. The mitochondrial genome is housed in the mitochondrion, an organelle involved in cellular metabolism. It is small compared to the nuclear genome and is a circular, maternally inherited molecule that has been well mapped in many species. Nuclear DNA on average evolves slowly, although some portions (e.g., microsatellites) evolve quickly. Mitochondrial DNA (mtDNA) on average evolves more quickly than the nuclear genome and some areas (e.g., control region) evolve very rapidly. These features make mtDNA and some regions of nuclear DNA suitable targets for certain genetic studies (Avise 1994).

Since it is virtually impossible to look at the entire genome, we typically target different areas throughout the genome to use in analyses. These "targeted" areas are called molecular markers. Because different stretches of DNA evolve (or mutate) at different rates, one can address different questions by targeting different regions of the genome. Systematic relationships are often investigated using the more slowly evolving cytochrome-*B* gene of the mtDNA whereas population genetic studies often target the hypervariable control region of mtDNA.

Published genetic studies of greater sage-grouse have focused on several questions. A number of authors have investigated the phylogenetic relationship of greater sage-grouse to other grouse species (Ellsworth et al. 1996, Gutierrez et al. 2000, and Drovetski 2002). These studies will not be described here as their focus is on evolutionary relationships among species rather than relationships within the greater sage-grouse. One study investigated the mating system of greater sage-grouse by comparing behavioral data and genetic data (Semple et al. 2001). Three other studies have looked at taxonomic boundaries and population genetics (Kahn et al. 1999, Oyler-McCance et al. 1999, and Benedict et al. 2003).

Investigating the Lek Mating System

Behavioral aspects of the lek mating system of greater sage-grouse have been studied (Wiley 1973, Gibson and Bradbury 1986, Gibson et al. 1991) and suggest that few males do most of the mating and that females typically only mate with one dominant male. There is some evidence, however, to suggest that there exists a component to the mating system that was unobserved by previous studies whereby females may be mating off the lek with purportedly non-dominant males. To investigate this issue, Semple et al. (2001) used six hypervariable microsatellite markers to identify the parentage of 10 broods of greater sage-grouse in Long Valley, California.

In this study, Semple et al. (2001) made behavioral observations on attendance, territoriality and mating behavior of males on one lek. Each male was uniquely identified using colored leg bands. Both males and females associated with this lek were trapped and blood samples collected. Females were radio-collared and followed throughout the breeding process. After 7 – 9 days of incubation, nests were located and all eggs were collected and embryos were sacrificed for genetic analysis.

Six variable microsatellite loci were used to identify the parentage of all embryos. The genotypes of the embryos were compared to the associated mother to identify which alleles were maternal and which were paternal. The number of paternal alleles was then used to determine whether the brood had been sired by more than one male. An attempt was made to exclude potential males as fathers based on their microsatellite genotypes.

Semple et al. (2001) found that all broods had genotypes consistent with their putative mother and that eight of ten broods showed results consistent with only one father. Two broods did show results suggesting that at least two males had fathered the brood. Of the ten females with broods, only four mated on the lek from which behavioral observations were made and thus were seen copulating. Three of those four females mated with males that had been genotyped and in each case the putative father could not be excluded as the father based on genotype data. The fourth female mated with a male who was not banded or genotyped. Therefore, Semple et al. (2001) concluded that in all four cases, the genetic data were consistent with behavioral data. While previous studies suggested very low instances of multiple paternity, this study showed that it may be more prevalent than was once thought.

Population Genetics of Sage-grouse in Colorado

Two published studies have investigated population genetic structure in Colorado (Kahn et al. 1999, Oyler-McCance et al. 1999). These two studies were initiated after morphological (Hupp and Braun 1991) and behavioral (Young et al. 1994) evidence suggested the sage-grouse in southwestern Colorado and southeastern Utah were different from sage-grouse elsewhere. The goal of the two genetic studies was to determine whether there were genetic differences between sage-grouse from Southwestern Colorado and those from Northwestern Colorado.

Kahn et al. (1999) sequenced 141 base pairs of the rapidly evolving control region of mtDNA from approximately 20 individuals each from seven populations. Six of those populations were the larger-bodied greater sage-grouse sampled from Eagle, North Park, Cold Springs, Middle Park, and Blue Mountain in Colorado and from Rich County in Utah. The remaining population was the smaller-bodied yet to be described Gunnison sage-grouse sampled from the Gunnison Basin. All DNA samples were extracted from muscle tissue obtained from hunter-killed sage-grouse.

This study revealed that all the greater sage-grouse populations had many mtDNA haplotypes present per population (average of 7.3 haplotypes per population) while the Gunnison sage-grouse population had only 2 haplotypes present. The greater sage-grouse populations were dominated by four haplotypes (Table 8.1) A, B, C, and D. The Gunnison sage-grouse population had only one of those four haplotypes (A) and had a haplotype G, which was unique to the Gunnison sage-grouse. Kahn et al. (1999) noted the dramatically different haplotype frequencies between the Gunnison sage-grouse and the greater sage-grouse. They performed a maximum parsimony analysis on the haplotype sequences using a heuristic search algorithm and constructed a phylogram (Fig. 8.1). This analysis revealed that all haplotypes fell into one of two deep clades. All populations of greater sage-grouse had individuals from both clades. They suggest that the two deep clades began diverging at least 850,000 years ago during the Pleistocene. Under this hypothesis the two clades subsequently intermixed as these populations re-converged.

Oyler-McCance et al. (1999) expanded on the study by Kahn et al. (1999) by including data from four nuclear microsatellites and also adding approximately 20 samples each from three additional populations of Gunnsion sage-grouse (Crawford, Dry Creek Basin, and Dove Creek). Their additional samples were DNA extracted from blood taken from trapped Gunnison sage-grouse.

This study showed similar results to those found by Kahn et al. (1999) in that the greater sage-grouse showed much more genetic diversity in mitochondrial haplotypes with an average of 7.3 haplotypes per population compared to an average of 2.3 haplotypes per population among the four populations of Gunnison sage-grouse. Further, the microsatellite data is concordant with this pattern with higher levels of allelic diversity among the greater sage-grouse (average of 5.7 alleles per locus) than among the Gunnison sage-grouse (average of 2.6 alleles per locus). Population genetic analysis revealed a lack of evidence of gene flow among the two groups of sage-grouse (Fig. 8.2) which is consistent with the idea that the Gunnison sage-grouse should be recognized as a new species (Young et al. 2000). Further, Oyler-McCance et al. (1999) showed that there was much less gene flow among populations of Gunnison sage-grouse (F_{ST} = 0.0266, 95% CI 0.0016-0.0528) than among populations of greater sage-grouse (F_{ST} = 0.2153, 95% CI 0.1230-0.3339).

Evaluation of the Eastern & Western Subspecies

Historically, sage-grouse were classified into two subspecies: eastern *(C. u. urophasianus*) and western sage-grouse *(C. u. phaios*) based on plumage and coloration differences in eight individuals collected from Washington, Oregon and California (Aldrich 1946). Western sage-grouse presumably occurred in southern British Columbia, central Washington, east-central Oregon, and northeastern California (Aldrich 1946). Populations in other areas of the range are considered to be eastern sage-grouse. The validity of this taxonomic distinction has since been questioned (Johnsgard 1983). While this species has recently been the target of extensive conservation efforts, the taxonomic/genetic relationships between populations/subspecies remain poorly understood. Because the distinction between the eastern and western subspecies has been questioned (Johnsgard 1983), Benedict et al. (2003) investigated whether there was evidence at the genetic level to support it. While genetic data alone can neither prove nor disprove a subspecies distinction, Benedict et al. (2003) surmise, as does Young et al. (2000), that morphological, behavioral and genetic data when used in conjunction, can help clarify such taxonomic questions. In addition, Benedict et al. (2003) were interested in providing information relevant to an understanding of gene flow, genetic diversity and evolutionary history between sage-grouse populations in Washington, Oregon, Nevada and California.

Benedict et al. (2003) collected sage-grouse tissue (muscle or blood) samples from sixteen populations in California, Nevada, Oregon, and Washington (Fig. 8.3), crossing the boundary separating the eastern and western subspecies, as described by Aldrich (1946, 1963). Approximately 20 birds were sampled from each population and sequence in the 141 base pair mitochondrial DNA control-region (following Kahn et al. 1999) was obtained. Benedict et al. (2003) found 38 haplotypes (Table 8.2) among the 332 birds assayed. All haplotypes fell into one of the two monophyletic clades (Clade 1 and Clade 2) described by Kahn et al. (1999) (Fig. 8.4). In this study all populations, except Yakima (WA), contained multiple haplotypes from both clades.

Benedict et al. (2003) found that most populations had a combination of common, rare, and novel haplotypes. Five common, widespread haplotypes (A, B, Q, T and X), were found in at least 6 and as many as 14 of the 16 populations sampled. Of the birds sampled, 221 (66.6%) had one of these five haplotypes. The X haplotype was found in all populations sampled except the Lyon/Mono population. This widespread haplotype was the only one found in the Yakima (WA) population and constituted the majority of the haplotypes in Douglas/Grant (WA) birds. The distribution of widespread, common haplotypes showed that there was no obvious genetic subdivision between the eastern and western subspecies. In addition, 42% of birds in the study by Benedict et al. (2003) shared five haplotypes with populations from Colorado and Utah (Kahn et al. 1999). The Washington populations and the Lyon/Mono population were found to be exceptions to this overall pattern.

Although Benedict et al. (2003) found that novel haplotypes were not uncommon, they were found normally to occur in low frequency within populations, typically fewer than 10% of the individuals. In the Lyon/Mono population, however, 87.5% of the haplotypes found were novel, constituting 97.7% of the birds sampled (Fig. 8.5). The only haplotype that Lyon/Mono shared with another population is from a single individual possessing the widespread Q haplotype. Thus, Benedict et al. (2003) suggest that the Lyon/Mono population has been isolated from neighboring populations for a considerable amount of time. Since this population has closely related, but novel haplotypes belonging to each clade, they believe that it is likely that this isolation occurred after the intermixing of populations representing the two major haplotype clades and that over tens of thousands of years, factors such as mutation, genetic drift, and the fixation of rare haplotypes have resulted in the significant divergence of the Lyon/Mono population from other sage-grouse populations.

In addition, Benedict et al. (2003) found that the Washington populations contain the lowest level of haplotype diversity observed. Two haplotypes were found to be unique to the Douglas/Grant population, yet a single haplotype (X) is found in the majority of individuals (88.6%). Benedict et al. (2003) suggest that Washington's low genetic diversity may be related to a genetic bottleneck given that these populations now occupy between 8 and 10 percent of their original range (Friedman and Carlton 1999). A neighbor-joining tree (Fig. 8.6) produced by Benedict et al. (2003) showed a lack of dichotomy between the populations representing the eastern and western subspecies (Fig. 4). Further, they suggest that the long branch length of the Lyon/Mono population is attributable to the unique allelic composition of these birds, while the long branch representing the Washington populations can be explained by their relative low level of haplotype diversity.

In summary, Benedict et al. (2003) found no evidence to support the subspecies delineation proposed by Aldrich (1946). Their data, however, did uncover the distinctiveness of the Washington and Lyon/Mono populations. The low genetic diversity found in the Washington populations is likely a reflection of population declines and isolation (Schroeder et al. 2000). The possibility that this type of isolation could be having an effect in other portions of the range has not been addressed yet. Benedict et al. (2003) suggested that the probable loss of genetic variation caused by this bottleneck and its potentially long-term adverse impact (Le Page et al. 2000, Bouzat et al. 1998) should be addressed as management strategies are developed for these populations. They advocate that active management, such as translocation of birds, may be justified to ensure their continued persistence.

Benedict et al. (2003) also indicated that the preservation of genetic diversity represented by the unique allelic composition of the Lyon/Mono population is also of particular importance for conservation. Since the likelihood that the distinctiveness of neutral genetic markers extends to genes under adaptive selection, Benedict et al. (2003) believe that this population should be managed independently to avoid the translocation of other sage-grouse into this area. They also maintain that it will be critical that additional morphological and behavioral studies of the Lyon/ Mono population be undertaken to address taxonomic questions. Sound conservation strategies require that multiple and mutually supportive lines of evidence be used to make prudent delineations at the species and subspecies level.

Current and Future Work

Further genetics research on greater sage-grouse is ongoing to complete a large-scale range-wide genetic survey of greater sage-grouse. This survey involves microsatellite genotyping at 7 loci and collecting mtDNA sequence data from approximately 1200 individuals from 45 populations. Data from nuclear introns are also being collected. There may also be additional efforts to consider genetic flow among and within populations. The delineations of these populations had not been established (Chapter 6) at the time the current data were collected and analyzed.

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Table 8.1. Sampling locations and haplotype frequencies of sage-grouse (from Kahn et al. 1999).

Location	Haplotype name																					*n*
	A	B	C	D	E	F	G	H	L	S	X	Z	AA	AC	AD	AE	AF	AG	AI	AL	AM	
Gunnison	30	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	31
Blue Mountain	1	8	1	1	0	0	0	0	0	1	0	3	1	1	1	2	1	0	0	0	0	21
Cold Springs	3	7	10	1	0	0	0	0	2	0	0	1	0	1	0	0	0	0	0	0	0	25
Eagle	2	2	15	4	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	26
Middle Park	0	7	9	2	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	21
North Park	4	5	6	3	2	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	1	23
Rich County, UT	3	5	3	0	0	1	0	0	0	0	0	0	3	0	0	0	0	1	1	0	0	17

Table 8.2. Individual sage-grouse sampled in Benedict et al. (2003) by population and haplotype. Alpha represents the proportion of haplotypes from clade I. N* represents the number of individuals sequenced. The bottom two rows show the Lyon/Mono population vs. all others.

			Haplotype (clade I underlined and in italics, clade II in bold)																																					
Location	N*	α	*A*	**B**	*F*	**O**	**Q**	**T**	**U**	**X**	*AG*	*AU*	**AV**	*AZ*	*BB*	**BD**	**BF**	**BO**	*BX*	*CH*	**CJ**	**CK**	**CN**	**CR**	*DA*	*DB*	*DC*	*DD*	**DH**	**DL**	**DM**	**DS**	*DT*	**BJ**	**BK**	*BL*	*BM*	**BQ**	**DP**	*DQ*
Yakima WA	18	0.00								18																														
Douglass/Grant WA	18	0.11								13																						3	2							
Whitehorse OR	33	0.42	14	1			1	10	4	2											1																			
Steens OR	21	0.29	5	1	1		1	8		4																					1									
Warner OR	19	0.37	4		1		2	2		5													1	2				1					1							
Wagontire OR	19	0.42	6		1		4	2	1	1										1			3																	
Beatty's OR	21	0.29	4				1			6			4							2	2	1		1																
Churchill NV	18	0.06		2				5		8				1		1	1																							
Washoe NV	20	0.40	6				3		1	6				1				1	1		1																			
Elko NV	20	0.50	3	3						5	1	4	2												1		1													
Humboldt NV	21	0.67	10	2			1	1		3																4														
Sheldon NWR NV	19	0.16	2			1	2	6	1	6					1																									
Nye NV	20	0.30		4				1		9	1			4	1																									
Lyon NV	20	0.20					1																											5	9	3	1	1		
Mono CA	23	0.65																																7		4	8		1	3
Lassen CA	22	0.23	5				4			8			1																1	1	2									
Others	289	0.31	59	13	3	1	19	35	7	94	2	4	7	6	2	1	1	1	1	3	4	1	4	3	1	4	1	1	1	1	3	3	3							
Lyon/Mono	43	0.44					1																											12	9	7	9	1	1	3

Fig. 8.1. Phylogram (from Kahn et al. 1999) shows that the 21 mitochondrial DNA haplotypes are separated into two deep monophyletic clades. Bootstrap values greater than 50% are shown along the branches of the tree. Haplotypes from clade I and clade II are found in all greater sage-grouse populations sampled.

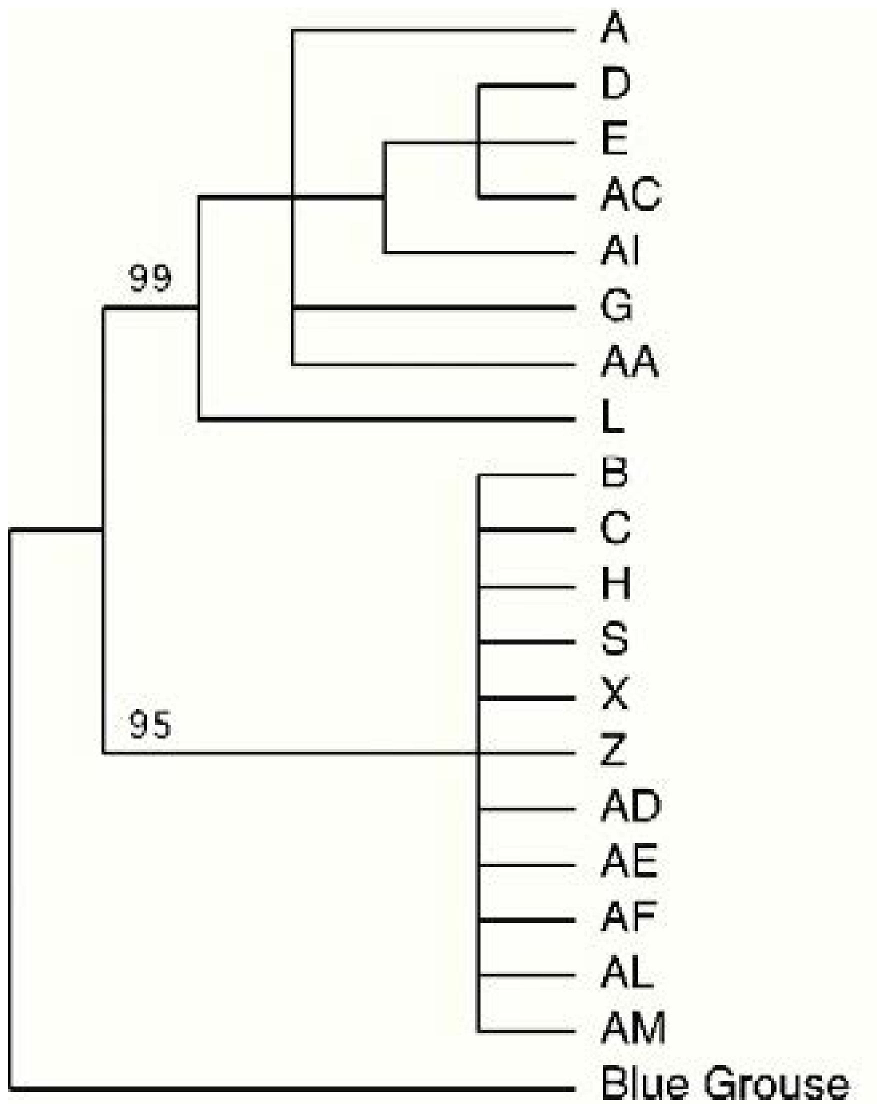
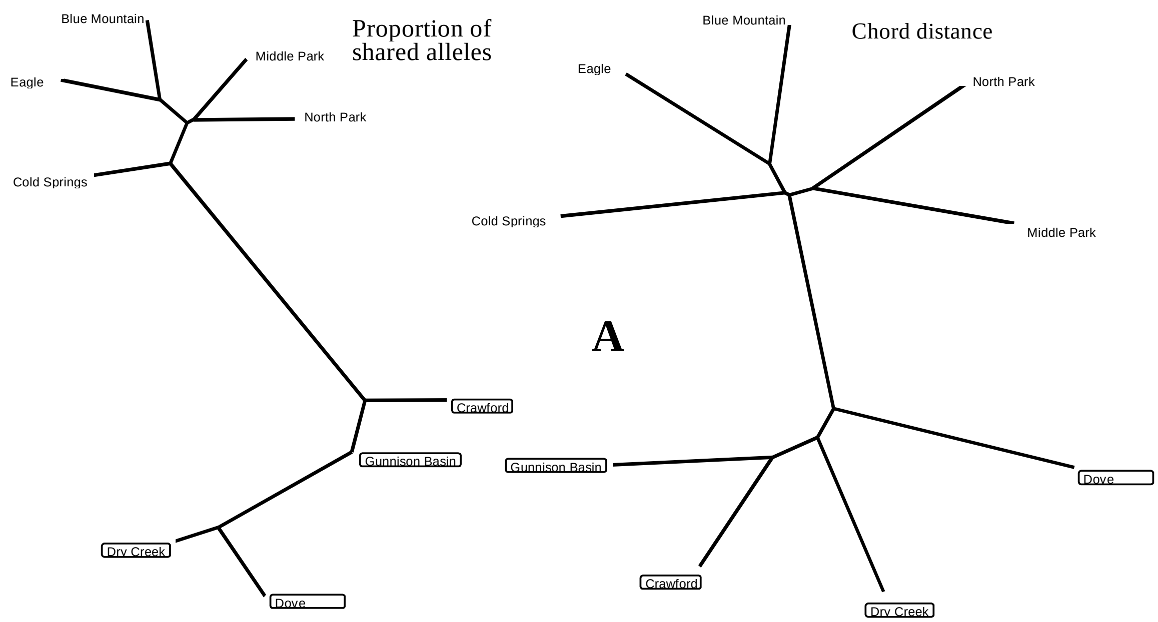


Fig. 8.2. From Oyler-McCance et al. (1999) (A) Neighbor-joining trees of microsatellite data using two different genetic distance measures. (B) Neighbor-joining tree of mitochondrial DNA genetic distances using allele frequencies and haplotype distances. Gunnison Sage-Grouse are identified with boxes.



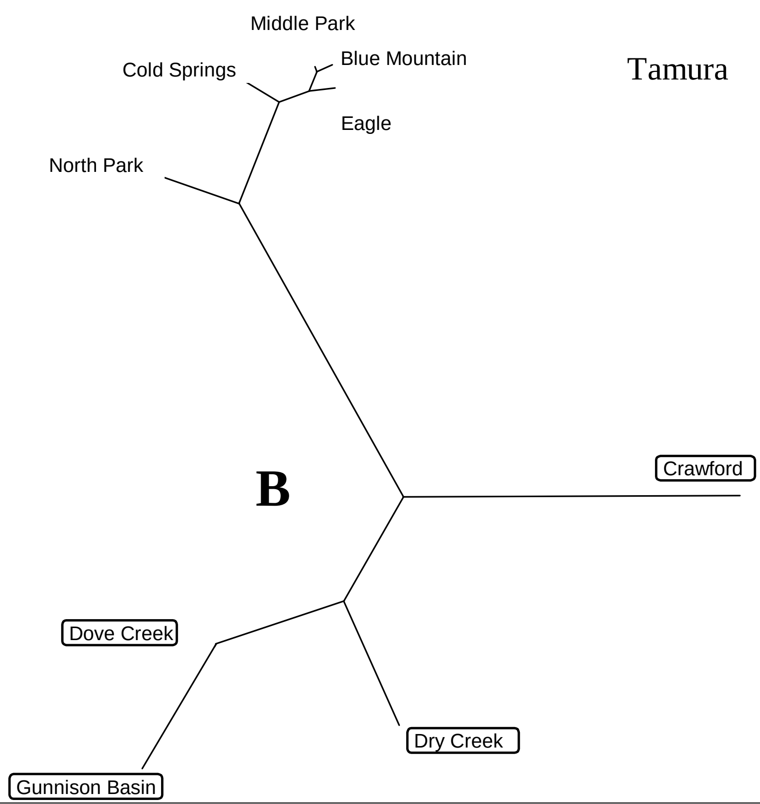


Fig. 8.3. Location of study populations from Benedict et al. (2003). The solid line denotes the delineation between the eastern and western subspecies of sage-grouse as proposed by Aldrich (1946).

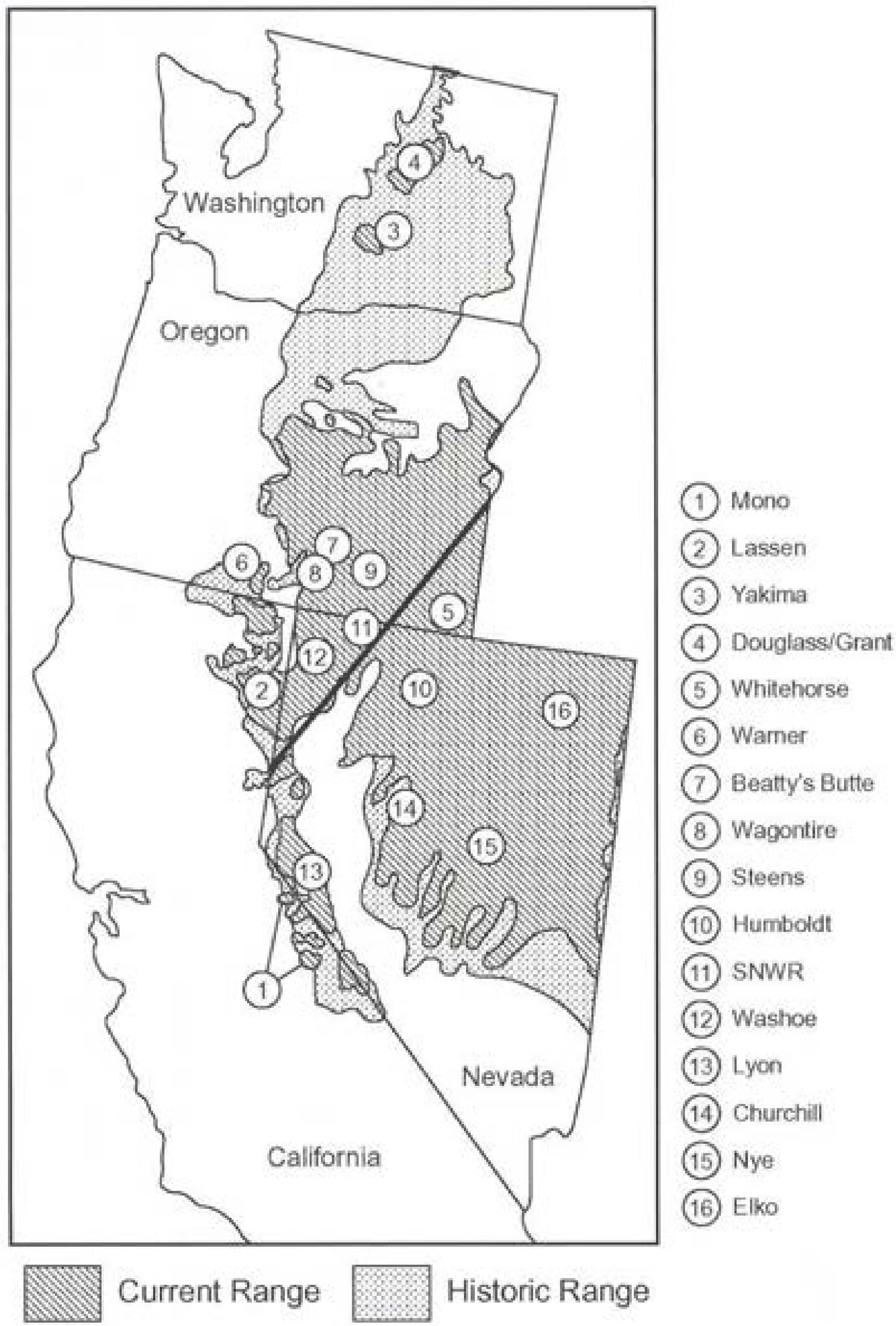


Fig. 8.4. Phylogram of the strict-consensus tree of all sage-grouse haplotypes represented (from Benedict et al. 2003). Bootstrap values greater than 50% are shown along branches of the tree.

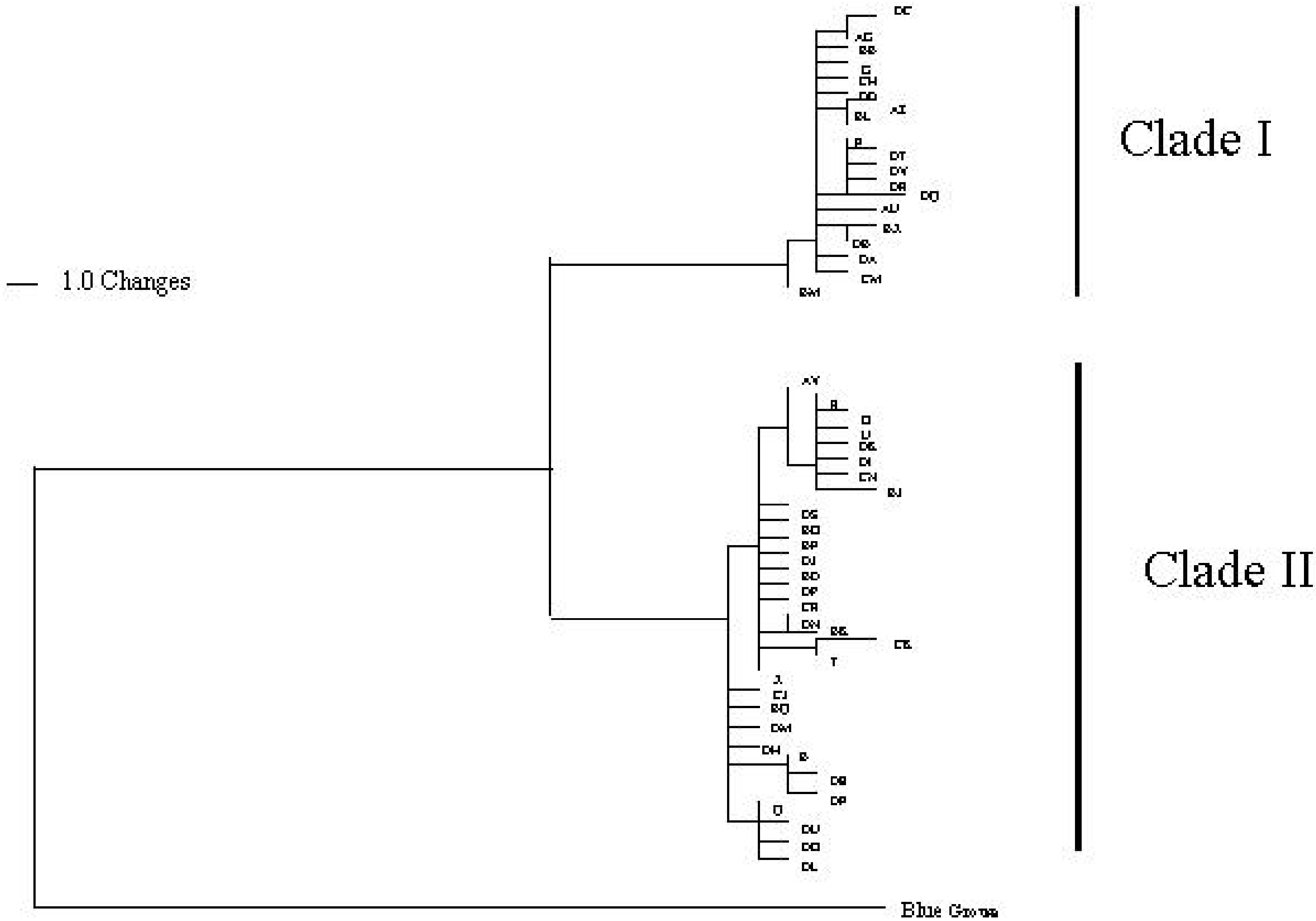


Fig. 8.5. Proportion of individual sage-grouse per population with novel haplotypes (from Benedict et al. 2003).

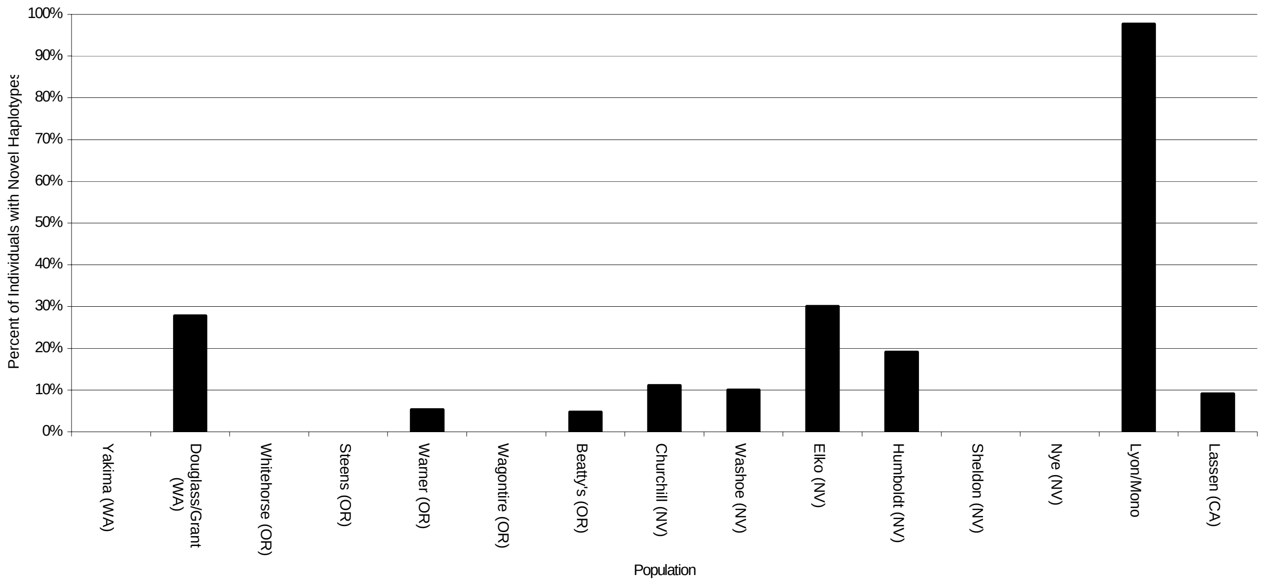
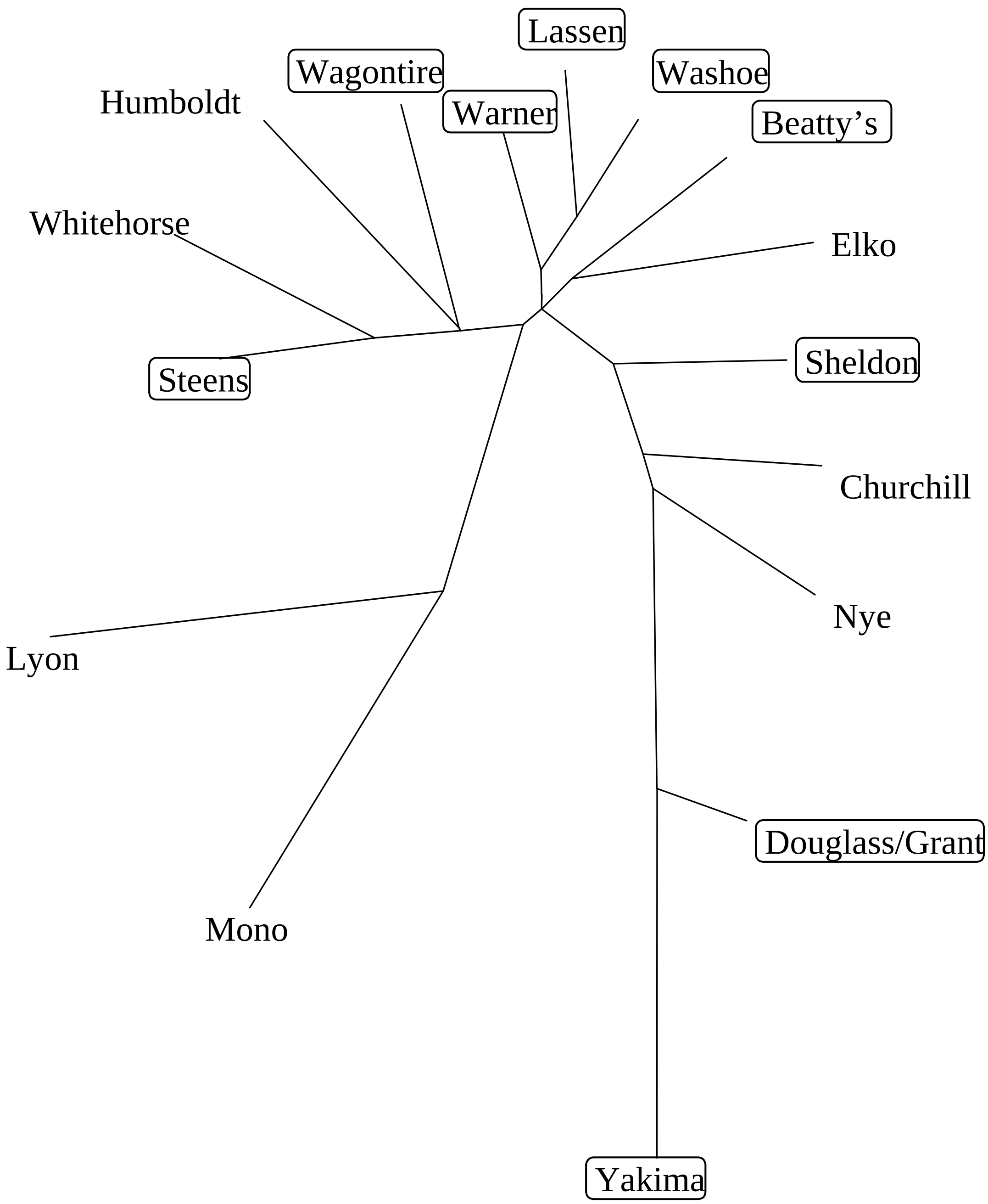


Fig. 8.6. Neighbor-joining tree constructed using Wright's (1978) modification of Roger's genetic distance. Populations with boxes around them represent those populations that lie within the bounds of the Western sage-grouse. Populations without boxes lie within the bounds of Eastern sage-grouse.



Chapter 9

Effect of Harvest on Greater Sage-grouse

CHAPTER 9

Effect of Harvest on Greater Sage-grouse

Abstract. Harvest of greater sage-grouse (*Centrocercus urophasianus*) has occurred throughout recorded history, but relatively few studies address the impact of harvest on sage-grouse numbers. Harvest of greater sage-grouse occurs in 10 of the 11 western states in which they reside. Here we discuss current season structures and recent changes in sage-grouse hunting seasons. We also review the likely effects of hunting on sage-grouse populations based on published studies. Based on recent research, it appears that because greater sage-grouse experience low mortality over winter, mortality from hunter harvest in September and October may not be compensatory to a large extent. However, no studies have demonstrated that hunting is a primary cause of reduced numbers of greater sage-grouse.

Introduction

The impact of harvest on populations of many wildlife species remains uncertain (Guttierrez 1994, Roy and Woolf 2001, Otis 2002, Williams et al. 2004). While there are numerous examples of sustainable harvest within high-quality habitats (Potts 1986, Hudson and Dobson 2001, Sutherland 2001, Willebrand and Hornell 2001, Gonzalez Voyer et al. 2003), excessive harvest can reduce spring breeding population size of gamebird species (Anderson and Burnham 1976, Small et al. 1991, Williams et al. 2004). Harvest of greater sage-grouse (*Centrocercus urophasianus*) has occurred throughout recorded history (Patterson 1952, Autenreith 1981), but relatively few studies address the impact of harvest on sage-grouse numbers. Given the declines in sage-grouse populations across the West (Connelly and Braun 1997), many hunters and biologists have expressed concern over the possible impacts of continued sport hunting of the species.

Harvest of greater sage-grouse occurs in 10 of the 11 western states in which they reside; only Washington prohibits harvest. Neither Alberta nor Saskatchewan allows harvest.

In the 1800s, heavy harvests decimated populations of greater sage-grouse, most states then prohibited harvest and populations increased (Patterson 1952). Harvest regulations have fluctuated over the past 50 years as populations increased or declined (Autenreith 1981, Wambolt et al. 2002).

In 2003, opening dates for hunting greater sage-grouse ranged from 1 September to 11 October, but most states began their seasons in mid-September (Table 9.1). Season lengths ranged from 2 to 62 days with a mean length of 9 days. Only two states, Montana and Idaho, retain seasons of more than 9 days, while 5 of the 10 states have open seasons of 5 days or less. Recently, hunting seasons, and bag and possession limits, for greater sage-grouse have become more restrictive as states respond to declining populations (Table 9.1). For example, in the last 10 years, regulatory changes in 5 states reduced harvest opportunities. Wyoming shifted the opening date of the hunting season from early September to mid September in 1995 and changed the opening day to the last Saturday of September in 2002. Wyoming also shortened the season by 7 days, and reduced the bag and possession limits from 3 and 6 to 2 and 4, respectively. In

1999, Nevada closed hunting seasons in 6 counties and in 7 hunt units in two other counties; the state also shortened the season from 16 to 9 days in other areas. Nevada closed two more hunt units in 2001 and an additional unit in 2003. Utah changed the regulations for greater sage-grouse from a general hunt with 1 and 2 bird bag and possession limits, respectively, to permit-only hunts in 4 units. Total number of permits was 954 for a 9-day season with a 2-bird season limit. California reduced the number of permits by nearly half, from 325 to 175. South Dakota retained a 2-day season, but reduced the season limit from 2 birds to 1.

In contrast, Oregon increased hunter opportunity through increasing the number of greater sage-grouse permits. In 2003, Oregon offered 1275 permits, 10 more than in 2001. This is less than a 1% increase in permit numbers. Regulations in Montana, Idaho, Nevada, Colorado, and North Dakota remained constant the past 3 years, but each state may have opened or closed specific hunting areas not reported here. For example, an area in Idaho closed to harvest since 1996 was opened in 2002.

Harvest

Currently available data indicate that the states of Montana, Idaho, Wyoming and Nevada, all with general greater sage-grouse hunting seasons, lead in estimated numbers of harvested greater sage-grouse (Table 9.2). Annual harvest estimates range from 12 in South Dakota to 7,576 in Idaho. Total annual harvest across the 10 states approximates 24,000 greater sage-grouse.

State wildlife management agencies reduce harvest opportunities to minimize the possibilities that hunting may have a negative impact on populations of greater sage-grouse. Over the past 50 years, wildlife managers and scientists have recognized that sport hunting can reduce wildlife species (Bergerud 1985, Ellison 1991, Dixon et al. 1996, Williams et al. 2004) unless prevented through harvest regulations. For upland gamebird populations, harvest mortality that reduces the population for the subsequent spring breeding season is termed “additive” (Anderson and Burnham 1976, Williams et al. 2004). Each bird harvested is in addition to those killed naturally through disease, starvation, accidents or predation. Additive hunting mortality results in a spring breeding population lower than if harvest had not occurred. In contrast, mortality from hunters could reduce natural mortality through a number of density dependent mechanisms including reduced depredation or lower competition for food or shelter, such that total mortality is no higher than without harvest. This “compensatory” mortality does not reduce subsequent spring breeding population size below what it would have been due to natural mortality (Anderson and Burnham 1976). Partial compensation could also occur in hunted populations (Anderson and Burnham 1976). Robertson and Rosenberg (1988) also addressed the issue of compensatory and additive mortality and concluded that in natural populations hunting mortality usually falls between the 2 extremes of being totally additive or totally compensatory.

Life history characteristics of greater sage-grouse differ from many other upland game birds. Many of these species have an r-selected strategy (Anderson 2002:54). They have high fecundity with large clutch sizes of 10-17 eggs, high annual rates of natural mortality, especially over winter (40-70%), and short life spans of 1-2 years (Gullion 1984, Potts 1986, Petersen et al. 1988, Christensen 1996, Giudice and Ratti 2001). Removal of individuals through hunting likely compensates for the many birds that would die naturally during their first or second winter (Kokko 2001, Sutherland 2001). Greater sage-grouse, however, tend toward K-selected life history features, with relatively low productivity through clutch sizes of 6-9 eggs, low over-winter mortality rates of 2-20%, and long life spans of 3-6 years (Schroeder et al. 1999). "Hunting will not have as large an impact on a population exhibiting an r strategy as it will on one having a K strategy" (Anderson 2002:55). Although effects of hunting may be independent of life history features (r-K continuum) if post harvest survival or reproduction is largely density dependent, there is no published information suggesting that this is the case for sage-grouse. Moreover, Ellison (1991) concluded there is little evidence for density-dependent breeding in tetraonids and that hunting may result in an age structure that lowers a population's productivity. Compensatory survival has been characterized as dogma within the field of wildlife management (Romesburg 1981, Warner 1992).

The appropriate harvest rate, expressed as percentage of the autumn population, remains elusive for greater sage-grouse. Several studies have addressed this during recent decades. A harvest rate of 30% for greater sage-grouse in Idaho was deemed allowable in a non peer-reviewed report by Autenreith (1981), but he believed that this high rate was never reached in any area. In addition, he emphasized that in xeric areas close to urban centers, harvest should be more conservative than in more mesic areas. Forbs are readily available to grouse throughout mesic ranges and grouse do not congregate in restricted feeding areas in August and September as they do in xeric ranges with limited mesic sites. Autenreith (1981) argued that dispersed birds in more mesic ranges are not as vulnerable to harvest as aggregated birds in xeric ranges nearer urban centers.

Crawford (1982), Crawford and Lutz (1985), and Braun and Beck (1985) all examined impacts of harvest through use of harvest figures, lek count trends, brood counts or with band recoveries. In Oregon, Crawford (1982:376) analyzed 20 years of data and reported that "the mortality from harvest may have been compensatory." Crawford and Lutz (1985) concluded that while harvest may have short-term effects on greater sage-grouse populations through increased mortality, sport hunting was not responsible for the long-term decline in sage-grouse numbers in Oregon. Braun and Beck (1985) determined that 7 to 11% of the fall population was harvested in an area of Colorado and that harvest had no measurable effect on sage-grouse densities in the spring. They concluded that hunting mortality could remove 20 to 25% of the autumn population without being additive.

While greater sage-grouse populations declined further in the 1990s and states reduced harvest opportunities, influence of harvest received little attention until late in the decade. Robert Gibson (personal communication) of the University of Nebraska examined population dynamics of 2 populations of greater sage-grouse in Mono County, California using data over a 45-year span. One population was isolated and the other was contiguous with populations in Nevada. Data used consisted of lek counts, numbers of birds shot per hunter in autumn, juveniles per hen in brood counts and in the fall bag, and number of birds inspected at check stations. He reported that the population contiguous with Nevada fluctuated independently of hunting mortality. However, the isolated population fluctuated, in part, with number of birds examined at check stations the previous autumn. Gibson (1998) concluded in an unpublished abstract that hunting mortality could "depress and hold population levels of sage grouse well below carrying capacity" and that this "should be of widespread concern in the light of long term population declines and range fragmentation in this species."[1]

Johnson and Braun (1999) used 23 years of hunter harvest and lek count data in a population viability analysis (PVA) for greater sage-grouse in North Park, Colorado. A PVA can assess the risk of a species population going extinct based on its survival and production values (Boyce 1992). As a mortality factor for greater sage-grouse, Johnson and Braun (1999) reported that hunting mortality was compensatory up to a threshold level, and at or beyond that threshold, harvest may be additive. The level where harvest becomes additive was not reported.

In another 20-year study, Connelly et al. (2000*a*) monitored mortality of radio-marked adult sage-grouse in Idaho. Adult females (n = 363) were affected by harvest more than adult males (n = 141). Autumn harvest caused 15% of known male mortality and 42% of known female mortality. Forty-six percent of all female mortality occurred during the hunting season (September and October) and harvest accounted for 91% of all female deaths. The sage-grouse hunting season in Idaho occurred in September and October. Post-hunting mortality (November-December) was only 1% for both genders combined (n = 103). Moreover, during the 4 post-hunting season months of November through February only 2% of the deaths of either sex occurred. The low over-winter mortality rate supports the contention that winter is not typically a difficult season for greater sage-grouse (Beck and Braun 1978, Remington and Braun 1988, Sherfy 1992).

Recognizing the typically low over-winter mortality of sage-grouse is vital to understanding impacts of harvest. Robertson (1991) monitored radio-marked sage-grouse over 3 winters in southeastern Idaho. Sample sizes were small, 7, 7 and 9 birds, respectively, but only 1 death occurred over 3 winters, and the average survival over 3 winters was 96%. A recent analysis of 20 years of band-recovery data from North Park, Colorado provides more evidence that winter is not a period of great loss for sage-grouse.

Zablan et al. (2003) found no evidence that variation in winter precipitation affected annual survival rates of sage-grouse, further support for winter as a time of minimal mortality. In southwestern Idaho, Wik (2002) determined seasonal survival rates of radio-marked greater sage-grouse from spring 1999 through fall 2001. Six estimates of winter survival rates of various age and sex classes of grouse produced point estimates of 0.85, 0.87, 0.88, 0.90, 1.00, and 1.00, all exceeding the 0.80 minimum over-winter survival reported in the review by Schroeder et al. (1999).

[1] Although we have generally avoided the use of personal communications and non peer-reviewed reports, there is a general lack of information on effects of hunting on sage-grouse. After careful consideration and discussions with the author and a California Department of Fish and Game biologist, we decided it was in the best interest of the resource to provide information from the Gibson report.

Because greater sage-grouse experience low mortality over winter, mortality from hunter harvest in September and October may not be compensatory to a large extent. What is the threshold point where harvest becomes additive? Based on a review of the literature, Connelly et al. (2000*c*) suggested that no more than 10% of the autumn population be removed through harvest. What are current harvest rates from autumn populations of greater sage-grouse? In Idaho, the harvest rate of adult females over 15 years averaged 6%, but in 6 of those years exceeded 10%, therefore Connelly et al. (2000*a*:230) concluded that for adult females "hunting losses are likely additive to winter mortality and may result in lower breeding populations." More recently, Wik (2002) reported harvest rates in an area of southwestern Idaho to be 5% for males, 6% for adult females and 18% for sub-adult females, and suggested limiting harvest, especially of females. Zablan et al. (2003) considered recovery rates to reflect harvest rates since most recovered bands from greater sage-grouse were from hunter-killed birds. They reported that recovery rates varied from 14.0% to 18.7% in North Park, Colorado, but found no correlation between population fluctuations, as measured by lek counts and recovery data over the period of study. However, Zablan et al. (2003) suggested that lek counts were inadequate to index population changes or that sample sizes of banded birds were low. Wildlife management agencies have reduced hunting pressure and harvest rates through regulatory changes, but have not universally accepted a specific harvest rate. There may not be one harvest rate appropriate for all populations of greater sage-grouse.

Zunino (1987) conducted a non-replicated experiment on 2 areas of Nevada to determine the response of sage-grouse density to harvest. Using autumn helicopter surveys that were assumed to count a constant proportion of the population, autumn sage-grouse densities were estimated in 1984 and 1985. Autumn sage-grouse populations on both the control (non-hunted) and treatment (hunted) areas increased between years. However, the increase in fall population density was 4 times greater on the control area than on the treatment area. These differences "were attributed to the treatments no harvest and harvest" (Zunino 1987:26).

Connelly et al. (2003) conducted the only other experimental study of greater sage-grouse response to harvest. They used lek counts from 1996 to 2001 on 19 lek routes to assess response to 3 levels of harvest. All lek routes were in areas with the same harvest regulations in 1996 (30-day season, 3 bird bag, 6 in possession). In 1997 and continuing through 2001, regulations changed to either no hunting, a restrictive 7-day season with 1 bird bag, 2 in possession, or a moderate 23-day season with 2 bird bag, 4 in possession. Treatments (no hunting, restrictive, or moderate seasons) consisted of 5, 7 and 7 lek routes, respectively. Lek routes were also categorized as being in lowland areas close ($\leq$ 1.5 hours drive) to major cities and towns or in high elevation mountain valleys farther from urban centers. After reducing harvest opportunities, areas that remained open to hunting had lower rates of increase than did areas with no hunting (Connelly et al. 2003). Both the moderate and restrictive hunting seasons produced harvests that apparently slowed population recovery (Connelly et al. 2003). Populations in low elevation habitats, close to urban centers and isolated because of habitat fragmentation, may be less able to withstand a harvest rate that has little or no effect on populations in more extensive, contiguous, remote, or mesic areas (Gibson 1998, Connelly et al. 2003).

Regulations for harvest of greater sage-grouse should minimize the possibility of negative effects. Therefore, units with dissimilar vegetative, physical or ecological attributes may differ in appropriate hunting season length and bag limits. However, several investigators have suggested that hunting seasons should be established with low rates of harvest (Schroeder et al. 1999, Connelly et al. 2000*c*, Wambolt et al. 2002), which should allow populations to increase if habitat quality is not limiting population numbers (Connelly et al. 2003). States do not presently determine autumn population sizes of greater sage-grouse.

No studies have demonstrated that hunting is a primary cause of reduced numbers of greater sage-grouse. Many studies support habitat-based reasons for sage-grouse declines (Swenson et al. 1987, Connelly and Braun 1997, Connelly et al. 2000*b*, Leonard et al. 2000, Aldridge and Brigham 2002, Pedersen et al. 2003). While eliminating harvest as a source of mortality did not produce any increase in greater sage-grouse numbers in Washington, likely because of habitat issues (Schroeder et al. 2000), reducing harvest may aid in population recovery in specific cases (Connelly et al. 2003).

Harvest of greater sage-grouse provides population data not easily obtained except through costly radio-telemetry studies of specific populations. Wings from hunter-harvested birds allow determination of sex and age ratios, average production per hen, and percentage of successful hens (see Chapter 6 for additional information). In conjunction with population trend counts, these data contribute to understanding the dynamics of sage-grouse in each management unit.

An appropriate harvest rate has not been determined for greater sage-grouse populations. Harvest equal to 5-10% of the autumn population may be appropriate, but assumes detailed and specific knowledge of population size in September or October. Given the uncertainty in abundance estimates for breeding season populations, expecting any state to adequately determine size of any population of greater sage-grouse in fall may not be realistic.

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Table 9.1. Calendar year 2003 hunting seasons for greater sage-grouse.

State	Opening date for state or area	Number of days	Bag and possession limit
Montana	1 September	62	3/6
Idaho (1)	20 September	23	2/4
(2)	20 September	7	1/2
Wyoming	27 September	9	2/4
Nevada (1)	11 October	9	2/4
(2)	20 September	2	3/6 with 75 permits
(3)	27 September	2	3/6 with 75 permits
(4)	27 September	2	3/6 with 75 permits
Utah (1)	20 September	9	2/2 season limit, 431 permits
(2)	20 September	9	2/2 season limit, 211 permits
(3)	20 September	9	2/2 season limit, 200 permits
(4)	20 September	9	2/2 season limit, 112 permits
Colorado	13 September	7	2/4
Oregon	6 September	5	2/2 season limit, 12 areas with 10 to 225 permits, total of 1,275 permits
California (1)	13 September	2	2/2 season limit, 100 permits
(2)	13 September	2	2/2 season limit, 40 permits
(3)	13 September	2	1/1 season limit, 25 permits
(4)	13 September	2	1/1 season limit, 10 permits
South Dakota	24 September	2	1/1 season limit
North Dakota	15 September	3	1/1

Table 9.2. Annual statewide harvest estimates for greater sage-grouse.

State	Year[1]	Harvest
Montana	2002	5,475
Idaho	2002	7,576
Wyoming	2002	4,835
Nevada	2002	3,940
Utah	2003	1,049
Colorado	2002	307
Oregon	2003	979
California	2003	170
South Dakota	2003	12
North Dakota	2002	45

[1] Most recent data available.

Chapter 10

Predation, Parasites and Parasites

CHAPTER 10

Predation, Parasites and Pathogens

Abstract. In this chapter we discuss the diversity of predators, parasites and diseases-causing pathogens that influence greater sage-grouse *(Centrocercus urophasianus*) the species range. Despite the prevalence of organisms that effect individual birds, population-level effects have been rarely documented. Concerns about predation have generally been indirectly addressed with habitat management. The newest observed disease, West Nile virus, has shown greater virulence than any infection or infestation noted so far.

Predation

As with most species of game birds, sage-grouse have many predators. Throughout most of the species' range, coyotes (*Canis latrans*), badgers, (*Taxidea taxus*), bobcats (*Felis rufus*) and several species of raptors are common predators of juvenile and adult sage-grouse (Patterson 1952, Schroeder et al. 1999, Schroeder and Baydack 2001). Additionally, coyotes, badgers, ground squirrels (*Spermophillus* spp.), common ravens (*Corvus corax*), and magpies (*Pica pica*) commonly prey on sage-grouse eggs (Patterson 1952, Schroeder et al. 1999, Schroeder and Baydack 2001). Many additional predators can kill and consume younger birds including the common raven, northern harrier (*Circus cyaneus*), and weasel (*Mustella* spp.) (Schroeder et al. 1999). The abundance of red fox (*Vulpes vulpes*) and raccoon (*Procyon lotor*) may have substantially increased in sage-grouse habitats because of landscape changes (Fichter and Williams 1967, Bunnell 2000, Connelly et al. 2000*a*).

Although there is little published information supporting the notion that predation is a major limiting factor on sage-grouse Connelly and Braun 1997, Connelly et al. 2000*b*, Schroeder and Baydack 2001), arguments continue to be made supporting predator control as an important management action (Wambolt et al. 2002). Two non peer-reviewed studies (Batterson and Morse 1948, Autenrieth 1981) suggested that nest predation due to corvids may limit sage-grouse numbers. More recently, numerous investigators have documented sage-grouse survival and nest success (Gregg 1991, Robertson 1991, Connelly et al. 1993, Gregg et al. 1994, Holloran 1999, Lyon 2000, Wik 2002). Only two of these studies (Gregg 1991, Gregg et al. 1994) indicated that predation was limiting sage-grouse populations by decreasing nest success, but both of these indicated that low nest success due to predation was ultimately related to poor nesting habitat. Most reported nest success rates are >40% (see chapter 3), suggesting that nest predation is not a widespread problem. Additionally, relatively high survival of adult birds (Zablan et al. (2003) and recent results demonstrating that coyote control in an area of Wyoming failed to produce an effect on nesting success (Slater 2003), further reinforce the idea that predation is not a widespread factor acting to depress sage-grouse populations. Thus, rigorous field studies using radio telemetry have generally failed to support these early findings.

In order to understand the possible impacts of predators on sage-grouse, it is important to understand the dynamics and behavior of predator populations. There are no predators within the range of sage-grouse that depend on sage-grouse as their primary food source; many depend primarily on rodents and lagomorphs and feed on sage-grouse opportunistically (see Bump et al. 1947, Angelstam 1986, Marcström et al. 1988, and Myrberget 1988 for examples). Consequently, the dynamics of a predator population and its primary food source can have observable impacts on a grouse population (Schroeder and Baydack 2001). When the primary food source is relatively rare, then a predator may spend more time searching for food, and consequently may be more likely to encounter a grouse or its nest (Angelstam 1983).

Predation may influence the population dynamics of grouse by reducing nest success, survival of juveniles (especially during the first few weeks after hatch), and annual survival of breeding-aged birds. The low survival of sage-grouse in the Strawberry Valley of Utah has been attributed to an unusually high density of red foxes (Bunnell 2000). Nest success is extremely variable and differences in success have been attributed to variation in habitat and management strategy (Connelly et al. 1991, Gregg et al. 1994, Connelly et al. 2000*b*). Although sage-grouse may partly compensate for predation pressure on nests by renesting (Schroeder 1997), habitat in sufficient quality and quantity often has been stated as an important goal for reducing the effects of predation (Connelly et al. 1991, 2000*b*). Survival of juveniles is clearly low, but is also difficult to accurately assess (Crawford et al. 2004). Unlike nesting habitat, management of brood-rearing habitat has focused on increasing the density and diversity of forbs (Klott and Lindzey 1990, Pyle and Crawford 1996, Sveum et al. 1998*b*), rather than improving vegetation to reduce predation (Edelmann et al. 1998). Although there have been many observations and recommendations concerning the importance of suitable habitat for reducing predation pressure on adults, detailed statistics have been difficult to obtain (Schroeder and Baydack 2001).

The quantity, quality, and configuration of habitat clearly has the potential to impact predator behavior and dynamics (Chapters 4, 12). These considerations include, but are not limited to, escape cover at nests (Connelly et al. 1991, Gregg et al. 1994) and visibility at leks (Hartzler 1974). In addition, several investigators have suggested that adequate feeding areas may minimize risks associated with increased travel and time spent in riskier habitats (Gregg et al. 1993, Fischer et al. 1996, Pyle and Crawford 1996).

Landscape fragmentation, agricultural habitats, and human populations have the potential to increase predator populations, and hence, predation pressure on grouse populations, as shown for corvids, domestic cats, and dogs (see Chapter 12). This potential for increased predation pressure in fragmented habitats is similar to what has been observed for grouse in Europe, where the pattern is well documented (Andrén et al. 1985, Andrén and Angelstam 1988, Bernard-Laurent and Magnani 1994, Kurki et al. 1997).

Although predator controls have been tried within the range of sage-grouse (Batterson and Morse [1948] removed many common ravens on an area in Oregon and there was a short-term increase in nest success), the cost effectiveness and long-term impacts of the removal on the behavior, genetics, and abundance of sage-grouse have not been examined (Schroeder and Baydack 2001). There also has been a more recent recognition of the broader financial and political cost to removing predators (Messmer et al. 1999). Because of these considerations, predator management for sage-grouse has generally been addressed with the "manipulation of habitat, because it is believed to be the most economical, efficient, and viable long-term strategy to enhance populations" (Schroeder and Baydack 2001:28).

Parasites and Pathogens

Introduction

Greater sage-grouse host a variety of potentially pathogenic organisms. However, the mere presence of such organisms does not necessarily indicate a population level effect. During the 1940's and 50's there was widespread belief, among both managers and laypersons, that diseases were responsible for heavy annual losses of gamebirds, including greater sage-grouse (Patterson 1952, Herman 1963). To the average person, the presence of parasites and disease producing organisms was synonymous with "disease" itself. Herman (1963) also reported that his review of the pertinent literature uncovered frequent references to 'the' greater sage-grouse disease, as though the authors were convinced that one specific agent could be the "universal root of all losses". In reality, some background level of parasites and disease producing organisms is normal and does not typically cause any significant alteration in structure or function of the population of the host species (Patterson 1952, Herman 1963). However, under certain circumstances a disease-causing agent may increase to a level that local populations are impacted (Herman 1963). Such circumstances may include extremes in precipitation, poor nutrition or the introduction of an exotic disease agent.

The parasites and diseases affecting greater sage-grouse have conformed to this pattern. Most of these parasites and diseases (Table 10.1) have not resulted in widespread population level impacts to greater sage-grouse. Those outbreaks of disease that have resulted in noteworthy morbidity and mortality were typically discovered under conditions of high numbers of grouse being concentrated due to dry conditions when water was scarce and large numbers of grouse likely contaminated water and soil with fecal material. Even in these cases there was insufficient evidence to conclude that disease was responsible for major declines across any extensive area of the birds' range (Patterson 1952).

Even so, there have been few systematic surveys for parasites or infectious diseases completed in greater sage-grouse, especially in recent years. This problem has been exacerbated by the ineffectiveness of some techniques for detecting pathogens that are present, or for inappropriately labeling a pathogen as present when it is absent. Therefore, the role diseases and parasites play in population declines across their range is essentially unknown. This fact, coupled with the emergence of new infectious diseases and the increasing numbers of small, isolated populations of greater sage-grouse that may be more vulnerable to population level effects, suggests this field deserves further study.

Of current concern is the emergence of the West Nile virus (WNV). This disease was exotic to the United States but is now considered endemic. It was first documented on the east coast in 1999 and has advanced rapidly across the nation. Greater sage-grouse were first documented to have contracted and succumbed to WNV in northeast Wyoming, eastern Montana and southeast Alberta in the summer of 2003, although one Wyoming bird actually died in 2002 but was not diagnosed until 2003 (D.E. Naugle, Personal Communication).

The following describes known parasites and diseases of greater sage-grouse and their implications to the bird. Many sources of information have not been published in the scientific literature, but instead appear in agency reports and proceedings. Where necessary we have included the references to provide a more through understanding of parasites and diseases in greater sage-grouse.

Macro-Parasites

Endoparasites. *Protozoa.* Coccidiosis. Coccidiosis in greater sage-grouse is the disease caused by the effect of one or more species of the protozoan genus Eimeria (Jolley 1982). Three species of the genus are known in greater sage-grouse: *E. angusta, E. centrocerci,* and *E. pattersoni. E. angusta* has been the most frequently documented species associated with coccidiosis (Thorne 1969). Diarrhea, which is characteristic of the disease, is caused when the parasites develop in and damage the mucosal lining of portions of the digestive tract. Transmission occurs when susceptible birds ingest feed or water contaminated by sporulated oocysts from feces of infected birds (Jolley 1982). When large numbers of oocysts are found in the feces of live birds concurrent with diarrhea and emaciation, coccidiosis should be suspected as the cause of illness (Friend and Franson 1999). However, a diagnosis as a cause of death requires a necropsy as well. Diarrhea is caused by a number of pathogens and oocysts observed in feces may or may not be those of the pathogenic species.

Historically, coccidiosis was the most important known parasitic disease of greater sage-grouse and has been, to date, the most prevalent of all known diseases affecting greater sage-grouse. In some site-specific locations, significant losses of young greater sage-grouse were documented in Wyoming, Colorado and Idaho from 1932-1953 (Honess and Post 1968). Most cases of coccidiosis in greater sage-grouse were discovered in areas where large numbers of birds were congregated resulting in fecal contamination of soil and water.

Greater sage-grouse mortalities attributed to coccidiosis have not been documented since the early 1960s. This may be the result of decreased greater sage-grouse densities. Investigators estimated 2,000 greater sage-grouse occupied a 2.59 km^2 (1 mi^2) alfalfa field prior to a coccidiosis outbreak in 1932 (Honess and Post 1968). There are no reports of similar densities being observed in more recent history. Any attempt to hold captive greater sage-grouse (or other game birds) should include a plan to prevent coccidiosis before clinical signs appear (Jolley 1982).

Hemosporidiosis (avian malaria). Hemosporidia are microscopic, intracellular parasitic protozoans found within the blood cells and tissues of avian hosts (Atkinson 1999). Three closely related genera, *Plasmodium*, *Haemoproteus* and *Leucocytozoon* are commonly found in wild birds (Atkinson 1999), including greater sage-grouse (Stabler et al. 1977, Gibson 1990, Boyce 1990, Deibert and Boyce 1997, Johnson and Boyce 1991, Dunbar et al. 2003). Hemosporidia are transmitted from bird to bird by a variety of insects including mosquitoes, black flies and biting midges (Atkinson 1999). A distantly related flagellate parasite, *Trypanosoma avium*, has been documented in greater sage-grouse (Stabler et al. 1977) but neither clinical signs nor implications to individual hosts or populations have been described.

In highly susceptible bird species and age classes, infections may result in death (Atkinson 1999). In greater sage-grouse, significant mortalities have not been well documented and most of the research conducted has centered around the effect of blood parasites (*Plasmodium*, *Haemoproteus*) on mate selection and breeding success. The traditional view that blood parasites are only slightly or not pathogenic has been challenged with recent research (Bennett et al. 1993, Nordling et al. 1998, Raidal and Jaensch 2000). The need for control of hemosporidiosis in greater sage-grouse populations has not been indicated. Such control would require control of insect vectors rather than treatment or prevention in birds themselves.

Sarcocystis. Salt (1958) described *Sarcocystis releyi* in greater sage-grouse however Jolley (1982) maintained that reports of *S. releyi* in gallinaceous and other birds were probably incorrect and involved other species of Sarcocystis. Regardless, there is no evidence to suggest significant implications or the need for control in greater sage-grouse. Sarcocysts are ubiquitous in wild species and generally cause little or no pathology.

Trichomoniasis. Although large numbers of *Tritrichomonas simoni* were found in all healthy greater sage-grouse examined (Honess 1955), the parasite is not known to be pathogenic (Jolley 1982).

Platyhelminthes. Tapeworms. Greater sage-grouse are the only known host of the cestode tapeworm *Raillietina centrocerci* (Honess 1982*a*). *Raillietina cesticillus* and *Rhabdometra nullicollis* have also been reported (Keller et al. 1941, Simon, 1940). Leidy (1887) reported the cestode *Hymenolepis microps,* however Simon (1940) believed this to be a mistaken identification. Greater sage-grouse show no apparent clinical signs of parasitism even though the tapeworms may distend the intestine and even dangle from the vent (Honess 1982*a*). Even the most heavily parasitized birds will be in good physical condition (Thorne 1969). Although Kerwin (1971, cited in the Canadian Sage Grouse Recovery Strategy) felt the role of *R. centrocerci* was overlooked as a mortality factor, Honess (1982*a*) suggested the relationship was an almost perfect adjustment between the host and its parasite. While unsightly, these tapeworms cannot parasitize humans and do not affect the quality of greater sage-grouse meat.

Nematoda. Gizzard worms. Gizzard worms belong to the genera Habronema and Acuaria (Cheilospirura). While Bergstrom (1982) indicated severe infestations of *A. centrocerci* could threaten greater sage-grouse in local areas, gizzard worms have only been seen occasionally and thus are not thought to present an important threat.

Cecal worms. Cecal worms of the genus *Heterakis* have been reported in greater sage-grouse (Simon 1940, Honess and Winter 1956). *H. gallinarum* is an important avian parasite because its eggs carry *Histomonas meleagridis*, the protozoan that causes the disease, avian blackhead. This disease is important in domestic fowl and game bird farms (Bergstrom 1982). Because development of *H. gallinarum* can be influenced by temperature fluctuations, fluctuations in parasite abundance could be linked with weather conditions (Saunders et al. 2002). The documentation of cecal worms in greater sage-grouse suggests the possibility of exposure to *H. meleagridis,* however the documentation cited by Simon (1940) consisted solely of a personal communication with an investigator that identified *H. gallinae* in a single greater sage-grouse specimen. Since blackhead disease has never been diagnosed in greater sage-grouse (free-ranging or captive), neither the disease nor its parasite vector is considered a threat to greater sage-grouse. Cecal worms (*Trichostrongylus tenuis*) in red grouse (*Lagopus lagopus scoticus*) have been linked with the population fluctuations or cycles (Hudson et al. 1998).

Filarid worms. A filarial nemotode, *Ornithofilaria tuvensis*, has been identified in greater sage-grouse but its presence is rare (Hepworth 1962). These white, hairlike worms were found in the connective tissue between the skin and breast muscles. Some birds observed where the known infected specimens were taken appeared unable to fly. However none of these birds were collected for analysis.

Ectoparasites. *Mallophaga (lice)*. Three species of chewing (bird) lice have been identified on greater sage-grouse: *Gonoides centrocerci*, *Lagopoecus gibsoni* and *L. perplexus*. Chewing lice are small, wingless and dorso-ventrally flattened insects that feed on fragments of skin and feathers. Heavy infestations may cause irritation to the host but there are no known serious implications for wildlife (Honess 1982*b*). Studies have been made regarding the presence and prevalence of lice and other parasites as it relates to mate selection and breeding success by birds, including greater sage-grouse (Boyce 1990, Deibert 1995). These relationships may be important to the long-term ecology and co-evolution of greater sage-grouse and the parasites; however, they have not been shown to be significant to the immediate status of greater sage-grouse populations.

Acarina (ticks). Two species of hard tick have been identified on greater sage-grouse: *Haemaphysalis chordeilis* and *Haemaphysalis leporis-palustris*. Early authors described the species *Haemaphysalis cinnabarina* (Parker et al. 1932, Simon 1940); however, Kingston and Honess (1982) report that *H. cinnabarina* does not occur in North America, and all references to this species should be referred to *H. chordeilis*. Ticks have been implicated in one greater sage-grouse mortality event that is discussed under the bacterial disease tularemia. Ticks can also cause "tick paralysis" in some hosts which is caused by a neurotoxin released by feeding adult female ticks (Honess and

Bergstrom 1982). Ticks are not considered a threat to greater sage-grouse populations because they are so infrequently observed on greater sage-grouse. Given the number of greater sage-grouse handled by researchers and hunters each year, and the relative ease in which ticks can be observed, more ticks would be reported if their presence was significant.

Diptera (flies, mosquitoes, midges, keds). Many diptera are pests and cause distress to the hosts on which they feed. Some diptera are important vectors of animal diseases, some of which will be discussed below e.g. West Nile virus. High numbers of keds were detected on about 25% of approximately 200 greater sage-grouse captured in the Montana Mountains of northern Nevada in 2002 (M.R. Dunbar, personal communication). These keds were identified as *Ornithomyia anchineuria*. Keds have not been documented on greater sage-grouse elsewhere.

Pathogens

Bacteria. *Salmonellosis*. Bacteria of the genus *Salmonella* are responsible for several diseases of birds such as pullorum and fowl typhoid (Thorne 1969). However, the only instance in which a *Salmonella sp.* has been documented as a disease agent in greater sage-grouse was a Wyoming case of dysentery characterized by debilitation and occasional death caused by stagnant and contaminated water supplies (Post 1960). This was probably a different species of *Salmonella* than those that cause pullorum and typhoid (W. Cook, personal communication). Studies have disclosed a much lower infection rate in wild birds than those in captivity and have caused investigators to conclude that, in general, salmonellosis is not an important disease of free-ranging wild birds (Friend 1999*a*).

Tularemia. Tularemia is primarily a tick-borne disease of mammals, but natural infections by *Francisella tularensis* have caused die-offs of ruffed grouse (*Bonasa umbellus*) and other grouse species (Friend 1999*b*). Parker et al. (1932) studied an epizootic in greater sage-grouse in Montana. These grouse were heavily infested with a bird tick, *Haemaphysalis chordeilis* and infected with *Francisella tularensis*. Hopla (1974) concluded that it was likely the tularemia, the tick, or both were major factors in the mortalities of these grouse. There have been no other reports of tularemia in greater sage-grouse and it is not currently a disease of concern because of the lack of tick infestations as noted above in the tick section.

Colibacillosis. Honess and Post (1968) reported that captured sick greater sage-grouse associated with an outbreak of coccidiosis were also bacteremic with *Escherichia coli*. *E. coli* has many strains that vary in virulence and pathogenicity (Thorne 1982). It is found in the lower intestine and is frequently a secondary invader that complicates primary infections. Diarrhea is clinical indicator of a possible pathogenic infection. While the presence of *E. coli* is probably under-reported, it is not believed to be a threat to wild populations of greater sage-grouse because it has only been shown to cause acute mortality in captive birds kept in unsanitary conditions (Friend 1999*b*).

Botulism, Avian Tuberculosis and Avian Cholera. While greater sage-grouse are almost certainly capable of being infected with these avian diseases (Thorne 1969), they have not been diagnosed in greater sage-grouse and are not considered a significant threat because the potential for exposure is low.

Mycoplasma. Mycoplasma synoviae was tentatively identified in greater sage-grouse in Moffat County Colorado, but there was concern for the possibility of 'false positives' on the tests (Hausleitner 2003). As with many other infectious diseases, greater sage-grouse may be susceptible to the disease but the potential for exposure is low.

Fungi. *Aspergillosis*. This disease is generally caused by the saprophytic mold *Aspergillus fumigatus,* a fungus which is highly pathogenic in many birds (Thorne 1969). The fungus is closely associated with agriculture and human activities such as damp, decaying vegetation and grain (Friend 1999c). It is believed clinical manifestation of the disease is stress related (W.Cook, personal communication). One case of aspergillosis has been documented in greater sage-grouse (Honess and Winter 1956) but there is no evidence to suggest aspergillosis plays a significant role in greater sage-grouse ecology. Greater sage-grouse habitats and feeding habits are not generally compatible with the ecology of this fungal disease.

Viruses.

The USGS Field Manual of Wildlife Disease (Anonymous 1999) states, "Historically, viral diseases have not been recognized as major causes of illness and death in North American wild birds. significant concern about viral diseases in wild birds has primarily occurred since the 1970's. This timeframe is consistent with an apparent increase of emerging infectious diseases and emerging viruses in other species It seems likely that viral diseases will assume even greater future importance as causes of disease in wild birds." The emergence of West Nile virus in North America has proven these statements prophetic.

West Nile Virus. Since its introduction to the northeastern U.S. in 1999, WNV has spread rapidly across North America, infecting and killing wild and domestic birds, horses, humans and other animals (CDC 2003). At least 208 species of birds, including greater sage-grouse, 29 species of mammals and two species of reptiles have been infected with WNV in North America (USGS 2003).

WNV is a member of the Japanese encephalitis antigenic complex [Family Flaviviridae]. Like many other flaviviruses, WNV is an arbovirus, meaning that it is transmitted between hosts by arthropod (insect) vectors. Mosquitoes are thought to be the primary vectors of WNV although other ectoparasites such ticks, lice, fleas and midges are also being evaluated as possible vectors of the disease (Marra et al. In Review, D.E. Naugle et al. 2004). Mosquito species in the genus *Culex* appear to be the main vectors implicated in the avian amplification cycle of WNV (Marra et al. In Review). In western North America, the mosquito, *Culex tarsalis,* is believed to be the most likely vector of WNV in birds (D.E. Naugle et al. 2004). Many birds act as reservoirs and amplifying hosts for the virus, infecting mosquitoes that then may transmit the virus back to more birds or on to other hosts. Host species differ in the degree to which the virus is replicated and circulated in the blood at levels high enough to infect mosquitoes. Humans and horses are considered "dead end" hosts because they do not typically accumulate the virus at a high enough concentration to infect mosquitoes that bite them. It is not known if greater sage-grouse are dead end or amplifying hosts.

Some birds can also become infected by means other than insect bites (Komar et al. 2003). In laboratory conditions, birds were infected via ingestion of WNV in aqueous solution or being fed infected mosquitoes, house sparrows or mice. Transmission was also documented after uninfected birds were placed in physical contact with infected cagemates, possibly through oral-fecal transmission or allopreening. The importance of these routes of transmission in free-ranging birds is not known.

Information regarding impacts of WNV on survival of native, wild birds is sparse (Malakoff 2003). Data from wild populations are limited to American crow (*Corvus brachyrhynchos*), in which mortality of marked individuals may reach 40% (Caffrey et al. 2003), and greater sage-grouse where (D.E. Naugle et al. 2004) a substantial decline in survival of greater sage-grouse hens were recently documented. The impact of WNV on populations of other birds over larger geographic regions is just now being studied using Christmas Bird Count (CBC) and North American Breeding Bird Surveys (BBS) (Marra et al. In Review). These data indicate most species show only local declines, suggesting common species are not at risk of extinction (Caffrey and Peterson 2003).

Elevated late-summer mortality of greater sage-grouse was reported across the eastern edge of their range (SE Alberta, E Montana, NE Wyoming) during an initial outbreak of WNV in 2003 (D.E. Naugle et al. 2004). Data from three study locations show survival declining an average of 25% between pre-WNV years and the first year WNV was detected (2003), whereas survival did not decline in a study area where WNV was not detected. Additionally survival in four study areas with WNV induced mortality in 2003 was, on average, 26% lower than the study site where WNV was not detected. Overall, individuals in populations exposed to the virus were 3.3 times more likely to die during the two-month WNV period (July-August) than birds in uninfected populations.

In addition, serum samples taken from grouse live-captured in the known WNV infected study areas all tested seronegative for WNV antibodies. The absence of seropositive live birds coming from areas with confirmed WNV deaths suggests that greater sage-grouse rarely survive WNV infection (T.E. Cornish et al. unpublished report., D.E. Naugle et al. 2004). However, the duration and variability of immunity among animals surviving WNV infection is essentially unknown (Marra et al. In Review).

The availability of surface water in sagebrush habitats may directly influence exposure of greater sage-grouse to WNV. Throughout their range, greater sage-grouse hens and broods congregate in mesic habitats in mid-late summer (Connelly et al. 2000*a*). The WNV vector mosquito, *Culex tarsalis*, exploits mesic habitats as breeding sites (Goddard et al. 2002). Risk of greater sage-grouse exposure to WNV may be particularly acute when WNV outbreaks coincide with environmental factors that aggregate birds around remaining water sources (e.g., drought) (D.E. Naugle et al. 2004).

Greater sage-grouse have suffered population declines and constricted distribution as a result of anthropogenic changes (Connelly and Braun 1997, Connelly et al. 2000*a*, Connelly et al. 2000*b*, Knick et al. 2003). The impacts of WNV, in combination with ongoing habitat loss and degradation, may pose a significant threat to some greater sage-grouse populations. However, WNV has only been known to infect and kill greater sage-grouse since August 2003. Determining the level of resistance greater sage-grouse have to the virus, the epidemiology of the disease in greater sage-grouse, and how land-use practices (especially the addition of late-summer surface water) influence prevalence and transmission of the disease will be required prior to being able to determine the impact WNV will ultimately have on greater sage-grouse populations.

Consideration of the positive and negative ramifications of disease management is also important. These considerations include control of adult mosquitoes, control of larval mosquitoes, integrated pest management, and vaccinations (Marra et al. In Review). A collaborative, multi-disciplinary approach in monitoring, reporting, and quantifying the impacts of WNV to greater sage-grouse and other wildlife will likely be needed to effectively manage the disease.

Newcastle Disease. Newcastle Disease has been a focus of concern for domestic poultry. The disease is highly contagious and there is great variation in the severity of disease caused by different strains of the virus. Prior to 1990, this disease had rarely been reported in free-ranging native birds in North America (Docherty and Friend 1999). However, in the last decade there have been repeated large-scale losses of double-crested cormorants (*Phalacrocorax auritus*) in the wild (Docherty and Friend 1999). Newcastle Disease has never been documented in greater sage-grouse. They may be susceptible but the potential for exposure is low.

Avian Pox. Avian pox is transmitted primarily by mosquitoes. The disease can also be transmitted via abraded skin or the conjunctiva or mucous membranes of the eyes (Hansen 1999*a*). Some strains of the virus are species specific but others have the ability to infect several species. "Dry pox" is the most common form of avian pox that consists of warty nodules that develop on the featherless parts of the birds. These lesions usually regress on their own but can become enlarged and cause sight, breathing and feeding impairment. Secondary bacterial and fungal infections are common and can lead to mortality. The internal form of the disease ("wet pox") is less observable because the lesions are internal. It has only occasionally been reported in wild birds but probably occurs more frequently than has been reported. The internal form causes greater morbidity and mortality than the cutaneous form (Hansen 1999*a*).

Pox outbreaks are common in captive situations. Species that would not ordinarily have contact with avian pox virus in the wild often become infected in captivity. Avian pox has been documented in one captive greater sage-grouse (DuBose 1965). Little is known about the disease's prevalence in wild birds. The increased frequency of reported cases of this highly visible disease and the involvement of new bird species during recent years suggests that avian pox is an emerging viral disease (Hansen 1999*a*).

Avian Infectious Bronchitis. Avian infectious bronchitis virus (AIB), a coronavirus, causes acute, highly contagious upper respiratory disease, decreased egg production and quality, decreased growth rates and chick mortality as high as 25% in domestic chickens (Cavanagh and Naqi 1997). Coronaviruses have also been isolated in domestic turkeys (Nagaraja and Pomeroy 1997), pheasants (*Phasianus colchicus*) (Spackman and Cameron 1983; Gough et al. 1996) and lesser prairie chickens (*Tympanuchus pallidicinctus*) (Peterson et al. 2002). As yet unpublished research (Dunbar and Gregg unpublished report), detected positive antibody titers to AIB in greater sage-grouse surveyed in southeastern Oregon and northwestern Nevada in 2003. The researchers concurrently observed unexplained early chick mortality in a portion of their study area.

While both greater sage-grouse and lesser prairie chickens have shown positive antibody titers to the virus, no clinical signs of disease have been observed. Peterson et al. (2002) suggested the first step to understanding the implications of the virus in lesser prairie chickens would be to challenge captive-reared birds with the virus, then describe the pathogenesis and transmission should the species prove susceptible. Further, they suggested that if clinically ill birds could be obtained from the wild, virus isolation and characterization should be attempted. Such recommendations could apply to greater sage-grouse as well.

Avian Influenza. Avian influenza virus (AI), also known as "bird flu" in the popular press, can cause significant problems for the commercial poultry industry. There are many different strains with varying virulence and recent outbreaks in Asia have had human implications. A variety of bird species are susceptible. However, the only mortality event known in wild birds killed common terns in South Africa in 1961 (Hansen, 1999*b*). The viruses are maintained in wild birds by fecal-oral routes of transmission.

Discussion

Prior to the very recent emergence of West Nile virus in greater sage-grouse, there was little evidence to suggest pathogens and/or parasites were major threats to greater sage-grouse. Pathogens and parasites are not mentioned in the 1997 Idaho Sage Grouse Management Plan (Idaho Department of Fish and Game, 1997) nor were they addressed in the "Guidelines to manage sage grouse populations and their habitats" (Connelly et al. 2000*a*). Disease issues are briefly discussed in the Nevada, Montana, Wyoming and Canadian greater sage-grouse conservation plans/strategies but these documents largely concluded diseases were not a priority issue at the time the documents were prepared from 2000-2003. However, in spite of being prepared prior to the 2003 emergence of WNV in greater sage-grouse, both the Montana and Wyoming plans mention the disease and suggest it be monitored.

Although it is clear that the long-term impacts of pathogens and parasites on greater sage-grouse populations remain to be fully explored, other species of grouse may offer some useful insights. Fox and Hudson (2001) found that cecal nematodes (*Trichostrongylus tenuis*) could impact territoriality in red grouse. The louping ill virus, which is transmitted by the tick Ixodes ricinus from sheep to red grouse can result in lower densities, higher mortality, and reduced harvest (Hudson 1992). The timing and effects of each parasite on the population dynamics of red grouse has also been examined in detail (Dobson and Hudson 1992, Hudson and Dobson 2001). It should be noted that the populations of red grouse studied by Fox and Hudson were found at substantially higher densities (> 200 birds km2, Hudson 1986) than typical populations of greater sage-grouse (< 5 birds km^2, Patterson 1952). Hudson et al. (1998) further argued that control of parasites in red grouse resulted in the elimination of their normally cyclic tendencies; this last observation was subsequently debated by Lambin et al. (1999) and Hudson et al. (1999).

The possibility that shared parasites may result in competition between species has also been considered. For example, *Heterakis gallinarum* can parasitize ring-necked pheasants with few effects. However if grey partridge (*Perdix perdix*) are infected, the adverse impacts are much more severe. Consequently, grey partridge populations may be reduced or eliminated in some areas where they overlap ring-necked pheasants (Tompkins et al. 2000*a*, *b*). Clearly, the potential for significant impacts of pathogens on populations remains an issue.

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Table 10.1. Known parasites and diseases of greater sage-grouse (*Centrocercus urophasianus*). Adapted in part from Gill (1966) and Schroeder et al. (1999).

Parasite/disease[a]	Reference(s)[b]
Endoparasites	
Protozoa (coccidians, malarians, others)	
Eimeria angusta	10, 11, 14, 15, 17, 19, 27, 28
Eimeria centrocerci	10, 11, 13, 14, 15, 17, 19, 27, 28
Eimeria pattersoni	18, 27, 28
Haemonproteus canachites	32, 37
Leucocytozoon lovati (*bonasae*)	32, 41
Plasmodium pediocetti	32, 36, 38, 39, 40, 41
Sarcocystis releyi	20, 35
Trichomonas simony	18
Trichomanas sp.	11, 12, 13, 24
Trypanosoma avium	26, 32
Platyhelminthes (tapeworms)	
Brachylaema fuscata	16
Raillietina centrocerci	7, 8, 11, 12, 13, 15, 24, 34
Raillietina cesticillus	12, 13, 24
Rhabdometra nullicollis	1, 5, 7, 11, 12, 13, 15, 16, 34
Nematoda (roundworms)	
Acuaria (*Cheilospirura*) *centrocerci*	9, 11, 15, 33
Acuaria (*Cheilospirura*) *spinosa*	4, 6
Habronema urophasiana	2, 9, 11, 15, 33
Heterakis gallinae	6, 9, 11
Heterakis gallinarum	19
Ornithofilaria tuvensis	23, 30
Oxyspirura lumsdeni	29
Microfilaria sp.	32, 41
Ectoparasites	
Mallophaga (lice)	
Gonoides centrocerci	11, 19, 36, 38
Lagopoecus gibsoni	19, 21, 36, 38
Lagopoecus perplexus	11, 12, 19
Acarina (ticks)	
Haemaphysalis chordeilis (*cinnabarina*)	3, 11, 31
Haemaphysalis leporis-palustris	3, 11
Diptera (true flies, mosquitoes, midges, keds)	

Ornithomyia anchineuria (ked)	45
Infectious diseases	
Bacterial diseases	
Salmonella sp. (Salmonellosis)	22, 30
Fracisella (*Pasturella*) *tularensis* (Tularemia)	3, 19, 31
Escherichia coli (Colibacillosis)	28
Fungal diseases	
Aspergillus fumigatus (Aspergillosis)	19, 30
Viral diseases	
Flavivirius (West Nile Virus)	42, 43
Avipoxvirus (Avian Pox)	25
Coronavirus (Avian Infectious Bronchitis)	44

[a]While we have attempted to include the most current taxonomic nomenclature, this was not the purpose of this section and some referenced nomenclature may be out of date. Genus and/or species names in parentheses were previously reported in the literature but are now considered outdated or incorrect. Also, some of the parasitic infections listed above (e.g. *Brachylaema fuscata*) have little documentation other than brief mention in a published document. As a result, they are not discussed in the text.

[b]References: 1) Ransom 1909; 2) Wehr 1931; 3) Parker et al. 1932; 4) Wehr 1933; 5) Boughton 1937; 6) Shillinger and Morley 1937; 7) Simon 1937; 8) Griner 1939; 9) Simon 1939*a*; 10) Simon 1939*b*; 11) Simon 1940; 12) Keller et al. 1941; 13) Dargan et al. 1942; 14) Honess 1942; 15) Patterson 1952; 16) Babero 1953; 17) Levine 1953; 18) Honess 1955; 19) Honess and Winter 1956; 20) Salt 1958; 21) Malcomson 1960; 22) Post 1960; 23) Hepworth 1962; 24) Rogers 1964; 25) DuBose 1965; 26) Stabler et al. 1966; 27) Honess 1968; 28) Honess and Post 1968; 29) Addison and Anderson 1969; 30) Thorne 1969; 31) Hopla 1974; 32) Stabler et al. 1977; 33) Bergstrom 1982; 34) Honess 1982*a*; 35) Jolley 1982; 36) Boyce 1990; 37) Gibson 1990; 38) Johnson and Boyce 1991; 39) Deibert 1995; 40) Deibert and Boyce 1997; 41) Dunbar et al. 2003; 42) Cornish et al. *in prep*; 43) Naugle et al. *in review*; 44)Dunbar and Gregg *in prep*., 45) Dunbar, pers. comm.

Chapter 11

Monitoring Sage-grouse Habitats and Populations

CHAPTER 11

Monitoring Sage-Grouse Habitats and Populations

Abstract. Most studies of sage-grouse relied on published techniques for assessing range vegetation, monitoring, and trapping sage-grouse. However, published methods for assessing vegetation were not developed specifically for sage-grouse habitats. Some population monitoring techniques have not been described in detail while others were based on work done in a single area or over a relatively short time. Here we discuss a recent peer-reviewed report completed as a part of the conservation assessment process and provide a general overview of techniques for population and habitat monitoring. This report described various techniques suitable for assessing sage-grouse habitat characteristics, monitoring populations, and capturing and marking sage-grouse. This report also attempted to standardize techniques where variations may have existed and made recommendations about the use of some techniques.

Introduction

Numerous studies have reported on characteristics of greater sage-grouse (*Centrocercus urophasianus*) populations and habitats throughout the species' range (Gregg et al. 1994, Fischer et al. 1996, Schroeder 1997, Apa 1998, Sveum et al. 1998, Commons et al. 1999, Lyon 2000, Nelle et al. 2000, Smith 2003, and others). Additionally, Connelly et al. (2000*b*) provided guidelines for managing sage-grouse populations and habitats and identified monitoring as an important component of a sage-grouse management program.

Most studies of sage-grouse relied on published techniques for assessing range vegetation, monitoring, and trapping sage-grouse (Canfield 1941, Daubenmire 1959, Floyd and Anderson 1982, Giesen et al. 1982, Emmons and Braun 1984, Wakkinen et al. 1992, Burkepile et al. 2002, Connelly et al. 2000*a*, and others). However, published methods for assessing vegetation were not developed specifically for sage-grouse habitats. Some population monitoring techniques have not been described in detail while others were based on work done in a single area or over a relatively short time.

Because of declines in sage-grouse populations (Connelly and Braun 1997) and continuing threats to these species and its habitats (Connelly and Braun 1997, Wambolt et al. 2002), standard techniques for monitoring populations and habitats are necessary to allow valid comparisons among areas and years and provide rigorous and consistent data sets (Connelly et al. 2003). Until recently, no effort has been made to compile and standardize all major monitoring techniques useful for assessing sage-grouse habitats and populations. As part of the Conservation Assessment, Connelly et al. (2003) described various techniques suitable for assessing sage-grouse habitat characteristics, monitoring sage-grouse populations, and capturing and marking sage-grouse. They attempted to standardize techniques where variations may have existed and made recommendations about the use of some techniques. They also provided a glossary to help standardize terms used in sage-grouse management. Connelly et al. (2003) intended their report to be used with the guidelines to manage sage-grouse populations and their habitats (Connelly et al. 2000*b*). The purpose of this chapter is to review the monitoring techniques discussed in Connelly et al. (2003) and provide a general overview of population and habitat monitoring methods that are currently considered appropriate for sage-grouse.

Habitat Assessment

Sagebrush (*Artemisia spp*.) habitats have changed markedly over the last 25 to 50 years and fire and agricultural development have played major roles in this change in many portions of the west (Knick and Rotenberry 1997, Connelly et al. 2000*a*, Wambolt et al. 2002). In other areas, energy development has impacted sagebrush rangeland (Lyon 2000, Braun et al. 2002). Recently, revised guidelines for managing greater sage-grouse populations and habitats were published (Connelly et al. 2000*b*). These guidelines strongly suggested that management decisions should be based on the best available data. Therefore, the quality and quantity of sage-grouse habitats must be documented to make appropriate management decisions. There are four general reasons for assessing habitats: 1) to document current condition and trend of habitat; 2) to evaluate impacts of a land treatment; 3) to assess the success of a habitat restoration program; and 4) to evaluate the ability of habitat to support a reintroduced population. Connelly et al. (2003) provided information on how sage-grouse habitat assessments may be made for any of these reasons and discussed techniques used to make these measurements. Chapter 7 also provides information on modeling techniques designed to assess risk to sagebrush communities.

Connelly et al. (2003) reported that in virtually all cases, habitat characterization should follow habitat selection processes described by Johnson (1980). Therefore, habitat assessment should initially reflect first-order selection or the geographic range of the sage-grouse population of interest. Within this range, second-order selection of habitat should be examined based on home ranges of individuals or subpopulations (e.g., birds associated with a lek or lek complex). Assessing the condition of various habitat components within the home range describes third-order selection and further refines the habitat assessment process (e.g., breeding habitat). Finally, if necessary, assessment can be made at the fourth-order selection level that involves the quality and quantity of food or cover at particular use sites. All of these approaches were described in Connelly et al. (2003).

Insects are an important component of early brood-rearing habitat (Patterson 1952, Klebenow and Gray 1968, Johnson and Boyce 1990). A complete assessment of early brood-rearing habitat should include an evaluation of insect abundance. Several methods exist for estimating insect numbers including sweep nets, beating sheets, and pitfall traps (Fischer 1994). Ants and beetles are often the most important groups of insects for young sage-grouse chicks (Johnson and Boyce 1990, Fischer et al. 1996), and their abundance can easily be assessed with pitfall traps. Pitfall traps can vary in size and shape. A common method of using this technique in sage-grouse habitat is to place test tubes so that they are flush with the ground in a grid arrangement (e.g., a 4x4 grid with tubes placed 50 cm apart) (Nelle 1998). Connelly et al. (2003) provided additional information on insect sampling.

Population Monitoring and Assessment

Sage-grouse populations in various parts of western North America have been monitored for well over 50 years (Patterson 1952, Dalke et al. 1963). Unfortunately, even within a given state, monitoring techniques have varied among areas and years. This variation complicates attempts to understand sage-grouse population trends and make comparisons among areas. Incomplete information on sage-grouse seasonal movements and the juxtaposition of various seasonal habitats also inhibits a manager's ability to understand population trends and effects of habitat changes. Until recently, in a non peer-reviewed report for the Western States Sage and Columbian Sharp-tailed Grouse Technical Committee, only Autenrieth et al. (1982) attempted to standardize population data collection techniques and describe methods available for documenting sage-grouse population characteristics. The sage-grouse guidelines (Connelly et al. 2000*b*) stressed the importance of population monitoring and collecting quality data in sage-grouse management programs. Thus, Connelly et al. (2003) described methods for routine monitoring as well as techniques for capturing and marking sage-grouse if more detailed information is necessary.

Monitoring

Breeding Populations. Because sage-grouse gather on traditional display areas (leks) each spring, wildlife biologists are afforded relatively easy methods for tracking breeding populations. These methods include lek censuses (annually counting the number of male sage-grouse attending leks in a given area), lek routes (annually counting the number of male sage-grouse on a group of leks that are relatively close and represent part or all of a single breeding population or deme), and lek surveys (annually counting the number of active leks in a given area). All monitoring procedures are conducted during early morning (1/2 hour before to 1 hour after sunrise), with reasonably good weather (light or no wind, partly cloudy to clear) from early March to early May. Timing is dependent on elevation of leks and persistence of winter conditions. Sage-grouse will begin displaying in late February at lower elevations with milder climates and in years with mild winter weather (e.g., southern Washington). Lek attendance will persist into early or mid-May at higher elevations (see Chapter 6 for additional information on censusing sage-grouse).

Although lek counts are widely used, concern over their usefulness was expressed > 20 years ago in a non peer-reviewed report to the Western Association of Fish and Wildlife Agencies (Beck and Braun 1980). Additional criticism was recently published (Walsh et al. 2004) but this work was short-term (1 field season), confined to one area (a high elevation basin in Grand County Colorado) and did not appear to address the use of lek counts as an indicator of population trend. Techniques for correctly conducting lek counts have also been described (Jenni and Hartzler 1978, Emmons and Braun 1984) and problems generally seem to be related to disregarding accepted techniques. Although lek counts are normally conducted in the manner described in the preceding paragraph, a recent review of raw data recorded while conducting lek counts in Idaho indicated that leks were sometimes counted when conditions were windy, ceiling was overcast, and during rainstorms; in some cases counts were begun greater than 1.5 hours after sunrise (M. L. Commons- Kemner, personal communication). Connelly et al. (2003) provided detailed information on monitoring breeding populations of sage-grouse.

Production. The use of brood observations, brood routes, and wing surveys (see Chapter 6 for additional information on wing surveys) to assess sage-grouse production has been described in non-peer reviewed reports (Autenrieth et al. 1982, Willis et al. 1993). Brood observations, sometimes called random brood routes, are simply records of all sage-grouse broods observed in a given area by any field personnel that find themselves in that area. This information provides some idea of the juvenile to adult ratio and percent of hens observed with broods. Thus, it is somewhat better than anecdotal data. However, it is not easily replicated and comparisons among years can be difficult to interpret.

In non peer-reviewed reports, Autenrieth (1981) and Willis et al. (1993) indicated that brood routes were commonly conducted in many states during the 1960's and 1970's and routinely used in Oregon. Routes are usually driven at speeds less than 32 kph in the morning (sunrise to about 0900) and evening (1800 to sunset) during late June, July, and early August. Routes may also be walked or conducted from horseback, trail bike, mountain bike or ATV. Brood routes are normally established in areas known to have concentrations of sage-grouse (Autenrieth et al. 1982). These areas are often in or adjacent to wet meadows, riparian zones and agricultural fields. Each brood is recorded separately and the presence of a hen is also recorded. Groups of unsuccessful females and males are also normally tallied. Because chicks are quite secretive it is usually necessary to flush the brood to obtain an accurate count. A trained bird dog can increase the efficiency of this procedure. If sufficient numbers of grouse are observed such that the sample size is adequate, this technique may provide an indication of trends in production. Brood routes provide the following information: birds/km, broods/km, average brood size, and chick:adult hen ratio. For non-hunted populations or populations subject to very light hunting where relatively few wings can be collected, brood routes are the only method available for assessing production, short of using radio telemetry. Additional information on monitoring production is provided in Connelly et al. (2003).

Winter Populations. Unlike breeding populations and production, there are no widely accepted methods for assessing winter populations (Connelly et al 2003). In part, this is because birds may be spread out over large areas during mild winters but clumped in less than 10% of the available habitat in severe winters (Beck 1977).

Beck (1977) searched probable winter use areas in Colorado by 4-wheel drive vehicles, snowmobiles, and snowshoes to document sage-grouse winter habitat. Similarly, Connelly (1982) used survey routes traversed by 4-wheel drive truck or snowmobile, depending on conditions, to document winter habitat use of sage-grouse in southeastern Idaho. Flock size, location, cover type, snow depth, and temperature were recorded along these routes (Connelly 1982).

Aerial surveys using either a fixed-wing aircraft or helicopter may be effective in identifying sage-grouse winter habitats and can often be done in conjunction with surveys for pronghorn (*Antilocapra americana*; Patterson 1952). If aerial surveys are used, data should be acquired over a series of years with different snow conditions to give a more complete picture of sage-grouse distribution in winter (Connelly et al. 2003).

Trapping and marking

Trapping. The capture and subsequent marking of sage-grouse has been employed as a method of assessing and delineating populations for well over 50 years (Patterson 1952). Over the years, techniques have been modified and the quality of radio transmitters has improved considerably. Nevertheless, there remains two main periods for effectively capturing sage-grouse, spring and late summer, although Colorado biologists have had success trapping sage-grouse during winter (A. D. Apa, personal communication). Techniques used will vary depending on terrain, access, weather, and population size. Detailed information on trapping techniques for sage-grouse have been described by Connelly et al. (2003).

Marking. A variety of techniques have been used to identify individual sage-grouse including numbers and patterns of tail feathers (Wiley 1973), leg-bands (Patterson 1952, Dalke et al. 1963), wing markers (Connelly 1982), ponchos (Wallestad 1975), colored back-tags (Autenrieth 1981), and radio-transmitters (Wallestad 1975, Autenrieth 1981). Two researchers even resorted to shooting off tips of tail feathers of displaying males as a means of identifying individual birds (Hartzler and Jenni 1988). Generally leg bands and radio-transmitters are the most common methods currently used for marking sage-grouse. Patagial tags may also have some value in providing movement and distribution data at a relatively low cost. Connelly et al. (2003) presented additional information on marking different gender and age classes of sage-grouse.

Habitat and Population Assessment

We are not aware of any approaches that would allow assessment of sage-grouse habitats and populations over relatively broad scales. The techniques so far developed appear most appropriate for mid to small-scale assessments and do not incorporate a simultaneous approach that includes both habitats and populations. However, Pedersen et al. (2003) recently described a model that simulates the effect of grazing and fire on temporal and spatial aspects of sagebrush vegetation and sage-grouse population dynamics. Although the model was used to assess a single population in eastern Idaho (Pedersen et al. 2003), this approach appears appropriate for application at broader scales.

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Chapter 12

The Human Footprint Across the Sage-Grouse Conservation Assessment Area

A Large-scale Analysis of Anthropogenic Impacts

CHAPTER 12

The Human Footprint Across the Sage-Grouse Conservation Assessment Area: A Large-Scale Analysis of Anthropogenic Impacts.

Authors' note: The published version is a shortened version from the review draft. The paper in its entirety will be published in a peer-reviewed journal.

Abstract. We developed the human footprint, an accumulation of anthropogenic impacts on ecological processes, across sagebrush habitats. To evaluate the spatial distribution of anthropogenic disturbance patterns, we incorporated four models evaluating the influence of anthropogenic features on sagebrush habitats, and three models on the spatial distribution of sage-grouse nest predators. The extent of the human footprint varies greatly across sagebrush habitats. Overall, 5% and 49% of the sage-grouse conservation assessment area and 1% and 49% of sagebrush habitats are within the three highest (i.e., class 8-10) and lowest (i.e., class 1-3) human footprint classes, respectively. The model suggests that the extent and intensity of the human footprint differs among areas currently occupied and those where sage-grouse (*Centrocercus urophasianus)* have been extirpated. The high human footprint classes (7-10) comprise less than 5% of the area within the current range compared to 25% for the extirpated range; low human footprint classes (1-4) encompass 72% of the current compared to 46% of the extirpated range.

Introduction

The human footprint developed in this chapter is not an exhaustive model of anthropogenic factors influencing sagebrush habitats; rather, we selected anthropogenic factors for which we were able to acquire sufficient spatial data sets to model anthropogenic effects on ecological processes across large spatial scales (i.e., the sage-grouse habitat conservation assessment area). As a result, the human footprint does not include models on how (1) air pollution (Vitousek et al. 1997, Smith et al. 2000) and livestock grazing (for reviews see Fleischner 1994, Freilich et al. 2003) influence exotic plant dispersal, and (2) anthropogenic resources influence the spatial distribution of non-domesticated mammalian sage-grouse predators. The spatial data sets required to model the influence of these anthropogenic disturbance patterns are simply not available. This is particularly puzzling for livestock grazing, because approximately 70% of the area covered by the 11 western states is grazed by cattle (Fleischner 1994) and has been altering ecological processes in the Intermountain West since the arrival of European settlers (Freilich et al. 2003).

The human footprint model developed in this chapter is based on how anthropogenic features influence sagebrush habitat and how synanthropic predators, predators that are symbionts of humans (Johnston 2001), utilize anthropogenic resources. The four habitat models evaluate how the invasion of exotic plants, human-caused fires, energy extraction, and anthropogenic fragmentation influence the spatial distribution and fragmentation of sagebrush habitats. These anthropogenic disturbance patterns often act in synergy and may cause irreversible loss of sagebrush habitats (Knick and Rotenberry 1997). Synergistic processes occur in sagebrush habitats when invading exotic plants [e.g., cheat grass (*Bromus tectorum*)] alter fire regimes such that resultant post-fire plant communities are dominated by exotic plants (for review see Knick 1999). Roads may directly influence exotic plant dispersal via disturbance during road construction or via alterations in soil regimes (for review see Tyser and Worley 1992, Forman and Alexander 1998, Safford and Harrison 2001, Gelbard and Belnap 2003). For example, in Californian serpentine soil ecosystems exotic plant species can be found up to 1km from the nearest road (Gelbard and Harrison 2003), and Russian thistle (*Salsola kali*), an exotic forb growing along roads, is wind-dispersed over distances greater than 4km (Stallings et al. 1995). Roads may also indirectly facilitate the dispersal of exotic grasses, such as crested wheatgrass (*Agropyron cristatum*), via human seeding along road verges or in burned areas near roads as a management strategy to curb the establishment of less desirable exotic grass species (Evans and Young 1978). The habitat models also evaluate fragmentation of sagebrush habitats. The addition of roads, railroads, and power lines to sagebrush habitats, and the conversion of sagebrush habitat to agricultural land and/or urban areas, induces fragmentation of sagebrush habitats. Fragmentation of these habitats has been shown to affect presence of sagebrush obligate passerine species (Knick and Rotenberry 1995).

The three predator models evaluate how anthropogenic activities at the landscape level influence the spatial distribution of synanthropic predators. Synanthropic predators can have detrimental effects on populations of threatened or endangered species because these predators are subsidized by anthropogenic resources and, in contrast to predators that do not utilize these resources, will prey on wildlife populations even if they occur in very low numbers (for review see Kristan III and Boarman 2003). The addition of anthropogenic features induces numerical and functional responses in synanthropic predator populations (Kristan III and Boarman 2003, Kristan III et al. 2004), thereby exposing wildlife populations to higher rates of incidental predation (Schmidt et al. 2001), and facilitating the expansion of these predators into sagebrush habitats where they are, in the absence of anthropogenic features, either found only at low densities or are absent (Restani et al. 2001, Kristan III and Boarman 2003). For sagebrush wildlife populations in general, we modeled the spatial distribution of house cats (*Felis cattus*) and domestic dogs (*Canis familiaris*). These predators, particularly house cats, can have detrimental effects on wildlife populations (Alterio et al. 1998). The domestic predator models are based on human populated areas, where the process of converting ranches to ranchettes will introduce these predators to wild landscapes even at low human densities (Odell and Knight 2001, Maestas et al. 2003). For sage-grouse populations, we modeled habitat utilization for synanthropic avian predators: common ravens (*Corvus corax*), American crows (*Corvus brachyrhynchos*), and black-billed magpies (*Pica hudsonia*). The former two species show increasing nation-wide population trends (Sauer et al. 2003), and common ravens in the Mojave desert have been shown to have detrimental effects on threatened desert tortoise (*Gopherus agassizii*) populations (Kristan III and Boarman 2003). Common ravens are also potential predators of sage-grouse nests (for review see Connelly et al. 2000), and all corvid species benefit from the addition of anthropogenic features to sagebrush habitats. For example, power lines are used by common ravens and other raptors for nesting and as hunting perches (Gilmer and Wiehe 1977, Knight and Kawashima 1993, Steenhof et al. 1993). Linear features such as railroads, primary and secondary roads, and irrigation channels often serve as travel routes for these predators (Knight et al. 1995), and expand their movements into previously unused regions. Numbers of synanthropic avian predators increase in areas surrounding rural human developments (Tewksbury et al. 1998), campgrounds (Neatherlin and Marzluff *In Press*), landfills (Kristan III et al. 2004), roads (Case 1978, Rolley and Lehman 1992, Knight and Kawashima 1993) rest stops, and agricultural lands because they provide reliable and often highly abundant food sources.

The objectives of this chapter are to: (1) define the human footprint by modeling anthropogenic factors that influence ecological processes in sagebrush habitats, (2) map the extent of the human footprint across sagebrush habitats; (3) demonstrate the effects of the human footprint on sagebrush habitats and sage-grouse populations.

Methods

To model the human footprint across sagebrush habitats, we first developed 13 spatial data sets (cell size 0.09 km) of anthropogenic disturbance factors: railroads, power lines, three road layers, campgrounds, rest stops, landfills, irrigation canals, oil-gas wells, human-induced fires, agricultural land, populated areas. Second, these spatial data sets were combined and/or manipulated further into grid layers before being used in the human footprint model development. Third, the spatial data sets and/or grid layers were then used to produce seven input models. Last, we used the seven input models in the development of the human footprint model. To investigate the effects of the human footprint, we used spatial data sets representing political and ecological provinces boundaries (western state boundaries, sagebrush floristic provinces, and land ownership). Below we describe the methods used to: 1) develop the seven input models, 2) combine the seven models into the human footprint model, and 3) assess the effects of the human footprint on sagebrush habitats and sage-grouse populations.

Input Models

***Exotic plant invasion risk*.** The inputs for this model are road type (Parendes and Jones 2000, Gelbard and Belnap 2003), distance from road (Gelbard and Harrison 2003), forest - non-forest vegetation (Parendes and Jones 2000), and proximity to rural-urban and agricultural areas. Three road-based models were built based on the classification of road type and four distance risk classes: (1) high risk for the roadway; (2) medium risk within a 0.09 km buffer directly adjacent to a road; 3) low risk (value = 1) between the 0.09 km road buffer and 1 km from a road; and 4) negligible risk (value = 0) for distance >1km from road. The high and medium risk values were scaled based on road type with interstates, federal and state highways receiving a higher risk (high = 3, medium = 2) than secondary roads (high = 2, medium = 1) because secondary roads with shallow road verges are poorer exotic plant dispersers compared to the other two road types (Gelbard and Belnap 2003). We classified the area 0.135 km on either side of the centerline as roadway (the interstate pixel and one pixel on either side of the interstate, 0 – 0.135 km) for interstate highways. For state and federal highways and secondary roads, the area

0.045 km on either side of the centerline was classified as roadway. Because exotic plant invasion in forest areas is restricted to roads and riparian corridors (Parendes and Jones 2000), we included only the high risk areas for all road types within forested areas. Last, because urbanized and agricultural areas act as exotic plant sources (Vitousek et al. 1996), we classified the populated areas and agricultural land spatial data sets assigning each pixel a high risk (value = 3). The three road models and the populated areas and agricultural land models were merged by selecting the maximum value at each pixel location from the five input grids using the MAX command in ARC/INFO.

***Synanthropic avian predators*.** We modeled the utilization of sagebrush landscapes by synanthropic avian predators using five spatial data sets: 1) populated areas, 2) campgrounds, 3) rest stops, 4) agricultural land, and 5) landfills; and one input layer: density of linear features. All spatial data sets and the one input layer were converted to distance input layers. Each GIS layer was buffered with a probability function derived from the daily movement patterns of corvid species. Daily forays of common ravens differ by region and breeding status. Non-breeding ravens traveled daily an average 6.9 km in Idaho (up to 62.5 km) to 27 km in Michigan (range 0.8 – 147 km) from roost sites to distant food sources (Boarman and Heinrich 1999). For breeding birds, pairs hunted on average 0.57 km ± 0.71 km se from the nest (Boarman and Heinrich 1999). Using these daily movement patterns, we developed a decaying probability function (P = 100-100/1+e $^{5 - 0.3 \text{ Distance}}$) which weighs areas near anthropogenic features more heavily (probability of occurrence = 90% < 8 km from an anthropogenic source) than far (probability of occurrence = < 0.001% at 30 km). Using each of the five-buffered layers, we created a composite layer by adding the probabilities of occurrence of each layer. This composite layer, therefore, is a measure of synergistic effects that enhance synanthropic predator dispersal in relation to the spatial distribution of anthropogenic resources.

Domestic mammalian predators. These models are both based on how domestic predators, house cats and dogs, utilize wildlands near human habituation (Odell and Knight 2001, Maestas et al. 2003). We based our models on the data collected by Odell and Knight (2001) that investigated habitat utilization of these predators with regard to distance from housing and on the probability for a homeowner to possess either a house cat or a dog.

For the house cat model, we buffered the populated areas distance layer in ARC/INFO using a probability function [P = 0.216 - 0.96 * Distance (km)] where any cell with distance <180 m received a probability between 0.216 to 0. All distances ≥ 0.18 km from populated areas were assigned a probability of 0.

The dog model included both the populated areas and the campground spatial data sets, where the populated areas and campground grids were buffered using probability functions [P = 0.548 - 1.4589 * Distance (km) and P = 0.566 - 0.001572 * Distance (km)], respectively). Any cell with distance < 0.36 km received a probability based on the function (0.556 to 0.0001572) and all distances ≥ 0.36 km from populated areas or campgrounds were assigned a probability of

0. We combined the two models into the dog model by selecting the maximum value at each pixel location from the 2 models using the MAX command in ARC/INFO.

Anthropogenic fragmentation. We used the following spatial data sets to model anthropogenic fragmentation: agricultural land, populated areas, power lines, railroads, and road spatial data sets. Because we were interested in the spatial arrangements of wildland patches and how anthropogenic fragmentation affects wildlife dispersal, we buffered some of these spatial data sets according to their area of influence. For example, the area of influence of interstate highways extends beyond the traffic lanes (Rowland et al. 2000, Brotons and Herrando 2001, Rheindt 2003), we therefore buffered each interstate highway by 1km. Similarly, we reasoned that the area of influence of federal and state highways, railroads and power lines extends well beyond the actual line feature; each anthropogenic feature was therefore buffered by 0.5km. All of these buffered spatial data sets were combined in ARC/INFO using the MERGE command and reclassified to 1 = anthropogenic feature and 0 = wildland. Using this input layer, we performed a moving window analysis to calculate the percentage of cells occupied by anthropogenic features using a 54.5 km quadrate as the analysis window (303 * 303 cells, area = 2975 km^2). This analysis window size approximates the upper home range size of a sage-grouse (Connelly et al. 2000).

Energy extraction. To model the influence of oil and gas wells and associated supporting structures, we calculated density of oil and gas wells, using the oil and gas wells spatial data set, by employing the POINTDENSITY command in ARC/INFO within a circle of 1km radius.

Human induced fire ignition density. To model the influence of human induced fires, we calculated density of human caused fire ignitions, using the human induced fires spatial data set by using the POINTDENSITY command in ARC/INFO within a circle of 1 km radius.

The Human Footprint Model

The human footprint model. We combined the seven models into one human footprint model using a summation approach. Six of the seven models were built with cell size 0.09 km, the fragmentation model was built with 0.18 km cell size because of the computing time required for a model based on 0.09 km cell size. Therefore, all models with 0.09 km cell size were re-sampled to 0.18 km using RESAMPLE in ARC/INFO with the option set for bilinear. The bilinear approach evaluates the four nearest cells surrounding each center cell (ESRI 1998). In order to weigh each model equally in the final summation, we standardized each model between 0 and 1 by dividing by the maximum value in each grid. We then summed up all models for the final model. Due to the input spatial layers used to produce each of the input models, certain anthropogenic features were given more weight in the final model due to their influence on multiple ecological processes. For example, the populated areas spatial data set was used five times, the roads and agricultural land three times, the power lines and campgrounds twice and six of the 11 spatial data sets were used only once to develop the input models. For analysis, the final model was classified into 10 classes: 1 = 0 – 0.333; 2 = 0.333 – 0.666; 3 = 0.666 - 1; 4 = 1 – 1.333; 5 = 1.333 – 1.666; 6 = 1.666 - 2; 7 = 2 – 2.333; 8 = 2.333 – 2.666; 9 = 2.666 - 3; 10 = > 3.

The influence of the human footprint on sagebrush habitats. We evaluated the human footprint area of influence for each of the six sagebrush floristic provinces (Miller and Eddleman 2001) within the conservation assessment area. We created a grid layer of the floristic provinces and used the COMBINE command in ARC/INFO to get the sum of all cells within each combination of floristic province and human footprint class. Using a similar approach, we evaluated the human footprint with regard to Federal, State and Private land holdings. For each land holding we calculated the total area within the conservation assessment area, average elevation, and area within each human footprint class.

The influence of the human footprint on the sage-grouse. We evaluated the influence of the human footprint in the current and extirpated sage-grouse ranges. We converted the historic and extirpated range to grid and used the ZONALSTATS command in ARC/INFO to get the percentage of each range covered by each human footprint class.

Results and Discussion

Input models

For the exotic plant invasion risk model, 28% of the conservation assessment area is within the high and medium class whereas 36% are within the negligible class (Figs. 12.1 and 12.2). The high and medium risk classes are found mainly in the Columbia Plateau, western Montana, central Utah, and along the Snake River Plain in Idaho.

The corvid presence risk model suggests that over 58% of the conservation assessment area is within the high and medium corvid presence risk class (Figs. 12.2 and 12.3). The high and medium corvid presence risk classes are found mainly in the Columbia Plateau, western Montana, central Utah, the I-80 corridor in Nevada, and along the Snake River Plain in Idaho. The negligible corvid presence risk class covers 7% of the conservation assessment area.

The domestic predator models have small-localized areas of influence within the conservation assessment area. Cat and dog presence affects 0.8% and 1.2%, respectively, of the conservation area (Figs. 12.2, 12.4, and 12.5).

The fragmentation of the conservation assessment area by anthropogenic features varies regionally. For example, percent anthropogenic features is high within the Columbia Plateau, the Snake River Plain in Idaho, northwestern Montana, southeastern Wyoming, and central Utah; but relatively low in Nevada and southeastern Oregon (Fig. 12.6). Overall, < 2% of the conservation assessment area is within landscapes that have >70% human features; in contrast 46% of the conservation assessment area contained < 10% human features (Fig. 12.2).

The extent of the oil and gas well density model covers 7% of the conservation assessment area and is limited to Colorado, Montana, Utah, Wyoming, and New Mexico (Figs. 12.2 and 12.7). However, the number of currently undeveloped oil and gas leases that could be developed in the future may increase the spatial extent of oil and gas wells in sagebrush habitats (see Chapter 7 for review).

The extent of the human-induced fire ignition model covers 5% of the conservation assessment area (Figs. 12.2 and 12.8). High densities of human induced fires are found in the Columbia Plateau, northeastern Oregon, western Montana, central Utah, the I-80 corridor in Nevada, and along the Snake River Plain in Idaho. Because this model documents the ignition points of human-induced fires, and not the extent of the fires, our model is a conservative estimation of the impact that human-induced fires have on sagebrush habitats.

The Human Footprint

The human footprint area of influence varies greatly across the conservation assessment area (Fig. 12.9). Areas of high human footprint are found throughout the Columbia Plateau in Washington, near Missoula, Montana, the Snake River Plain in Idaho, along I-15 corridor in Utah, and along the I-80 corridor in Nevada. Not surprisingly, most areas of low human footprint are within National Parks and Wilderness Areas, as well as the Owyhee region of southeastern Oregon and Nevada. Overall, 5% and 49% of the conservation assessment area are within the three highest (i.e., class 8-10) and lowest (i.e., class 1-3) human footprint classes, respectively (Fig. 12.10). Within current sagebrush habitat, 1% and 49% are in the three highest (i.e., class 8-10) and lowest (i.e., class 1-3) human footprint classes, respectively (Fig. 12.10).

The human footprint area of influence also varies among landowners (Fig. 12.11). The most prominent human footprint influences are found on Bureau of Reclamation (BOR), and Bureau of Indian Affairs (BIA), as well as private and state lands; they all have < 5% of their land in the lowest human footprint class. For other federal land holdings, the U.S. Bureau of Land Management (BLM), Department of Energy (DOE) and U.S. Forest Service (USFS) have between 6 – 8% and the Department of Defense (DOD), National Park Service (NPS), and U.S. Fish and Wildlife Service (USFWS) have between 14 - 42% of their land in the lowest human footprint class. Furthermore, higher human footprint classes dominate most of the private and BOR lands whereas the human footprint least affects NPS and DOD lands. On private land holdings throughout the West, particularly Colorado, Utah, New Mexico, and Montana, large-acreage ranches are being broken up into smaller ranchettes which increases human densities and brings along associated infrastructure, exotic plants, domestic predators and a subsequent change in biodiversity (Maestas et al. 2003, Odell et al. 2003). This shift in land use could potentially increase the influence of the human footprint within and near these areas of development.

The human footprint also affects six floristic provinces within the conservation assessment area differently (Fig. 12.12). Four of the six floristic provinces have < 5% area in the lowest human footprint class; the Northern and Southern Great Basin have 5% and 9%, respectively, of their land cover within the lowest human footprint class. Moreover, the Columbia Basin floristic province is affected most severely by the human footprint with over 30% of its land cover distributed in the top four human footprint classes.

In areas that have a low human footprint, the addition of anthropogenic features could potentially have drastic effects. Habitat fragmentation may occur in small increments, but if there are thresholds in habitat – extinction rates, then even a small increment in habitat fragmentation near thresholds could have a detrimental effects on sage-grouse population viability (With and Crist 1995, With and King 1999, Fahrig 2001). Development within human footprint classes 1 –3 and connectivity between these areas may be a critical issue for sagebrush habitat conservation.

The human footprint and sage-grouse. We find that the human footprint intensity differs between areas currently occupied and those where sage-grouse have been extirpated (Fig. 12.13). The high human footprint classes (7-10) comprise less than 5% of the area within the current range compared to 25% for the extirpated range. For the low human footprint classes (1-4), these encompass 72% of the area within the current range compared to 46% of the extirpated range. Although the human footprint is more rampant within the extirpated range other influences may also play a role in this range reduction.

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Fig. 12.1: Spatial extent of exotic plant invasion risk across the sage-grouse conservation assessment area.

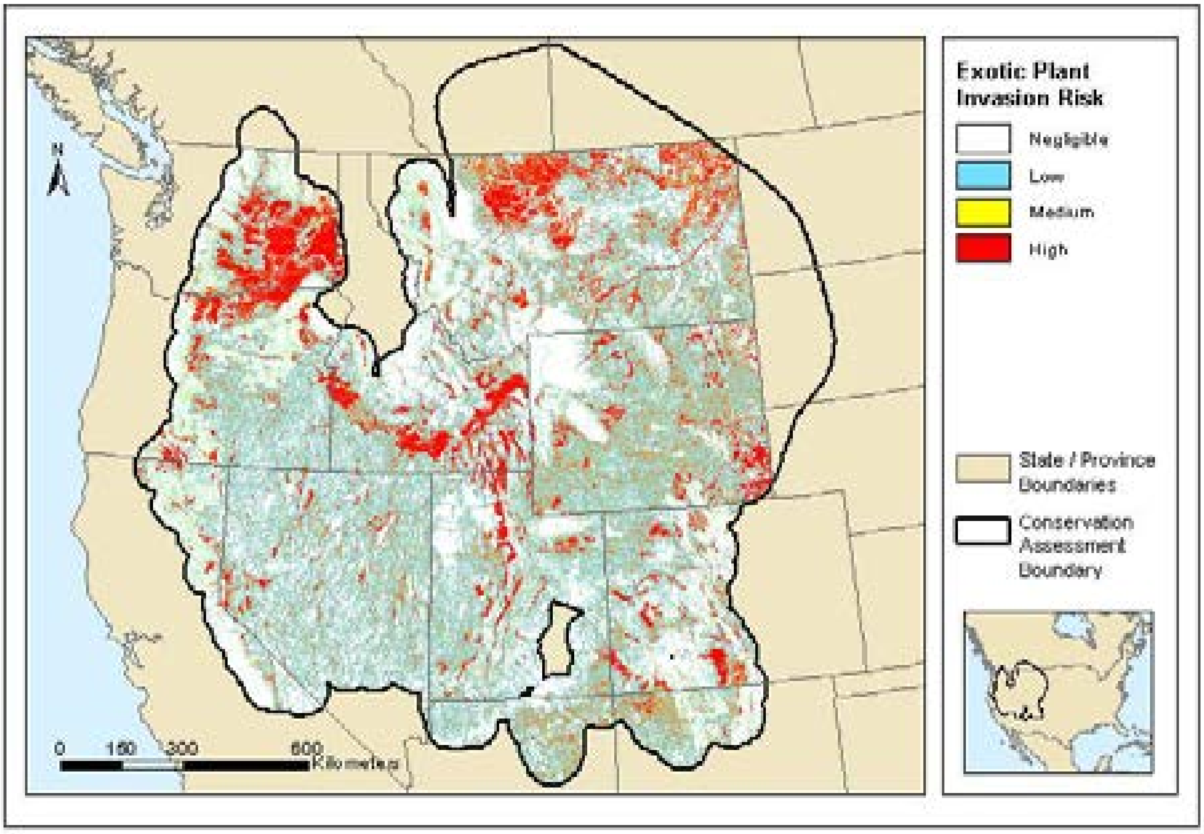


Fig. 12.2: Area of influence for the seven input model across the sage-grouse conservation assessment area (percentage of total conservation assessment area shown above bars).

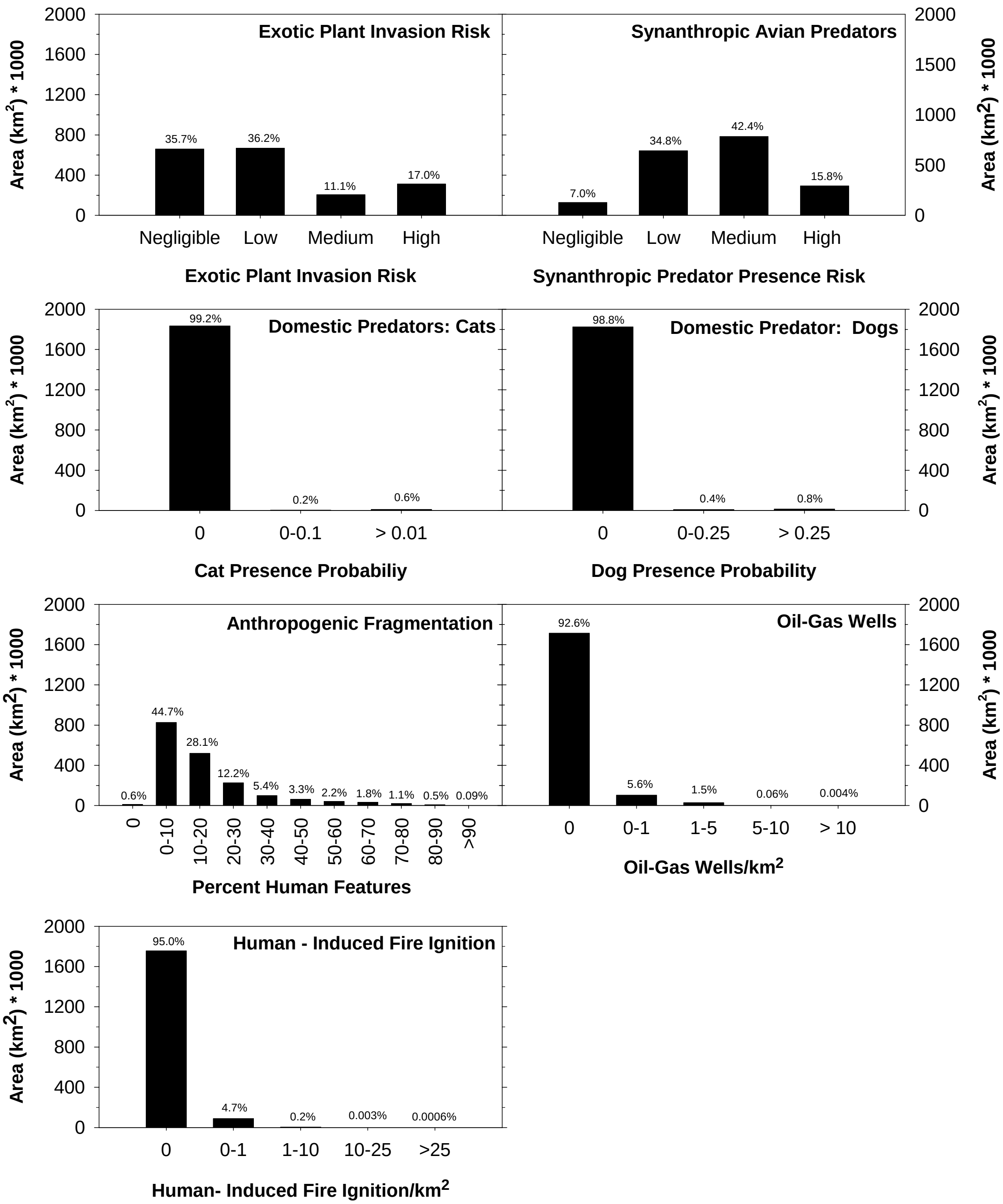


Fig. 12.3: Spatial extent of synanthropic corvid presence risk (American crow, *Corvus brachyrhynchos*; common raven, *Corvus corax*; black-billed magpie, *Pica hudsonia*), across the sage-grouse conservation assessment area. Areas with negligible corvid presence risk (white areas) represent corvid population levels in the absence of anthropogenic resources.

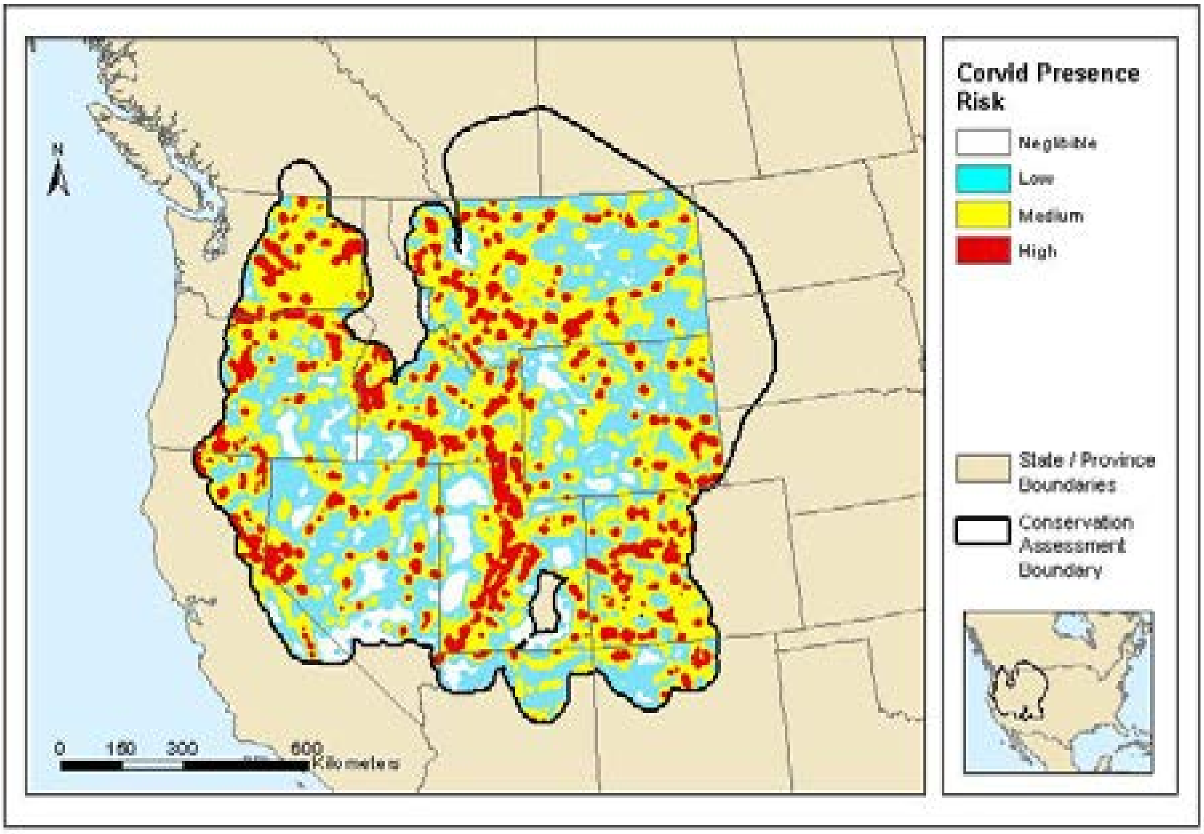


Fig. 12.4: Spatial extent of house cat (*Felis cattus*) presence risk across the sage-grouse conservation assessment area.

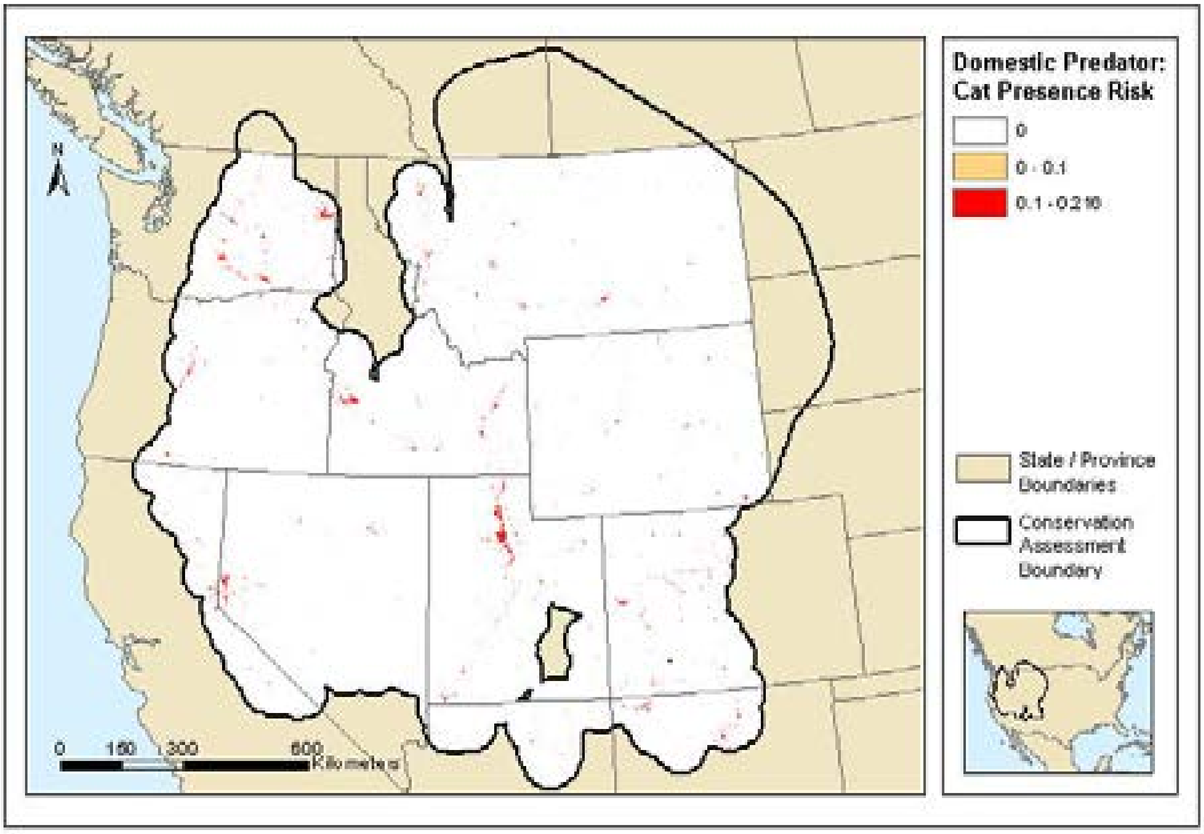


Fig. 12.5: Spatial extent of domestic dog (*Canis familiaris*) presence risk across the sage-grouse conservation assessment area.

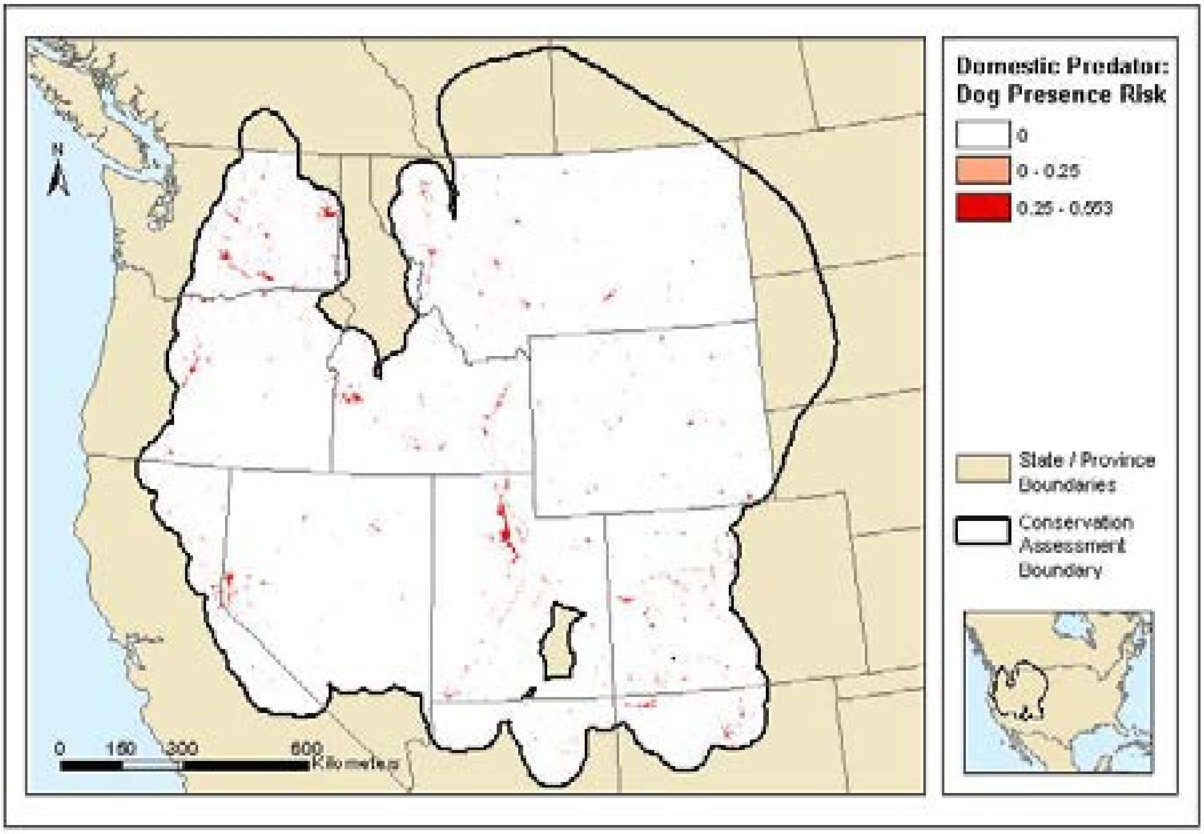


Fig. 12.6: Spatial extent of percent area containing anthropogenic features (agricultural land, populated areas, power lines, railroads, and roads) within a 2,975 km^2 (area approximates a sage-grouse home range) moving window.

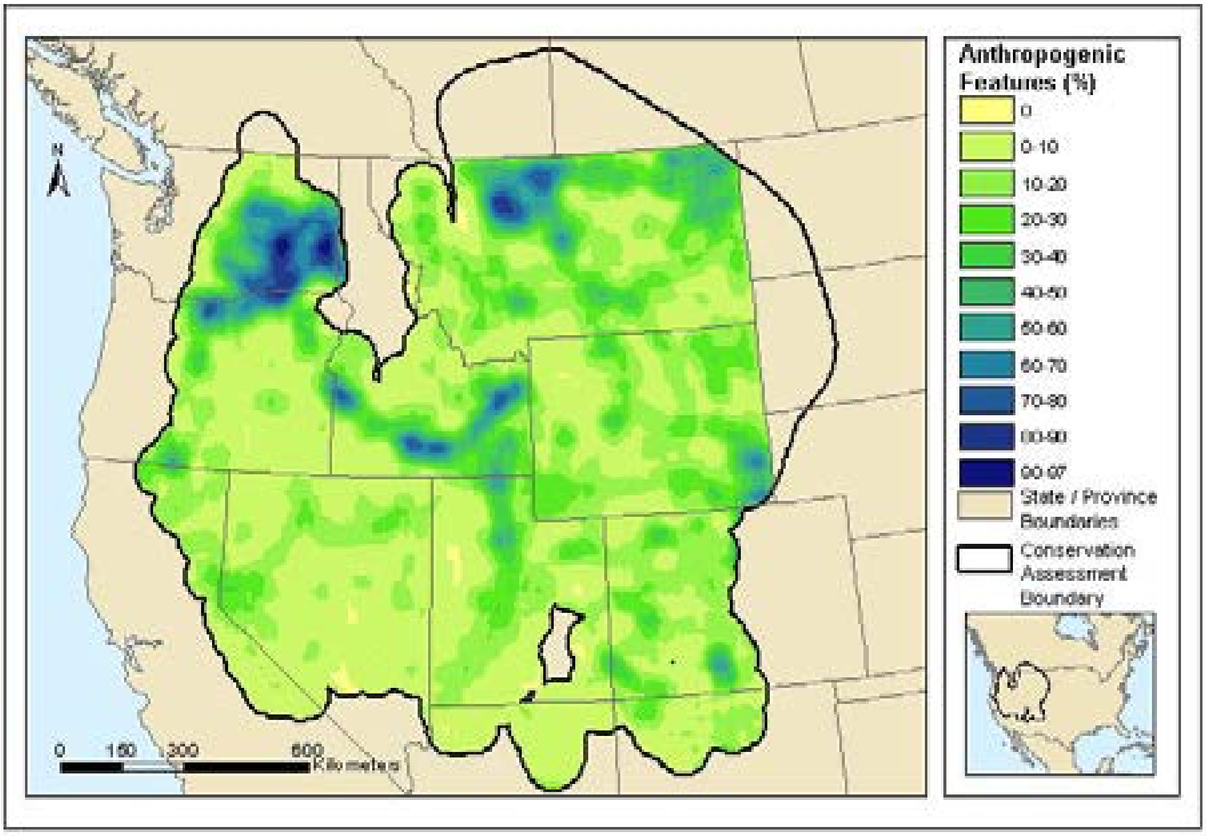


Fig. 12.7: Spatial extent of active and abandoned energy extraction operations (oil and gas well densities) across the sage-grouse conservation assessment area.

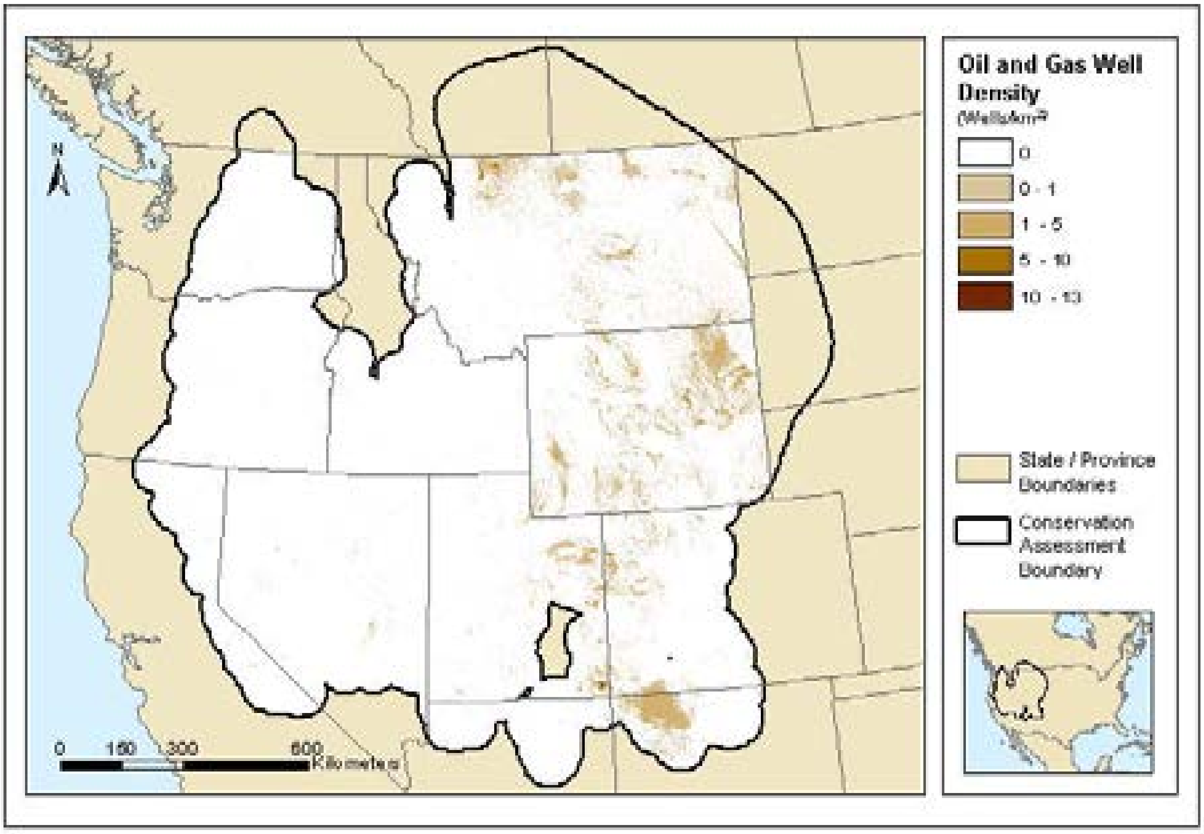


Fig. 12.8: Spatial extent of human-caused fire ignition densities across the sage-grouse conservation assessment area. Human fire ignition densities are based on point locations where fires were ignited by humans but do not reflect the full spatial extent of these fires.

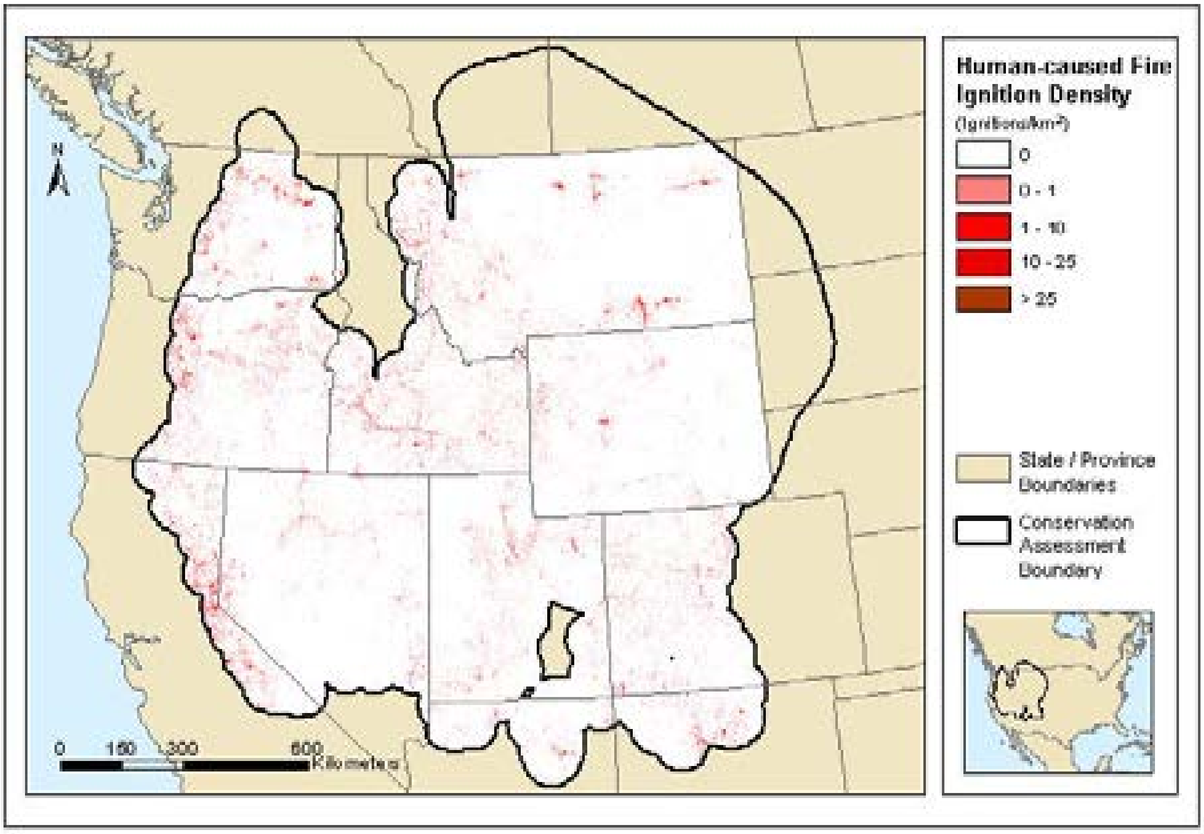


Fig. 12.9: Spatial extent of the human footprint across the sage-grouse conservation assessment area. Human footprint classes range from 1 (lowest human footprint influence) to 10 (highest human footprint influence).

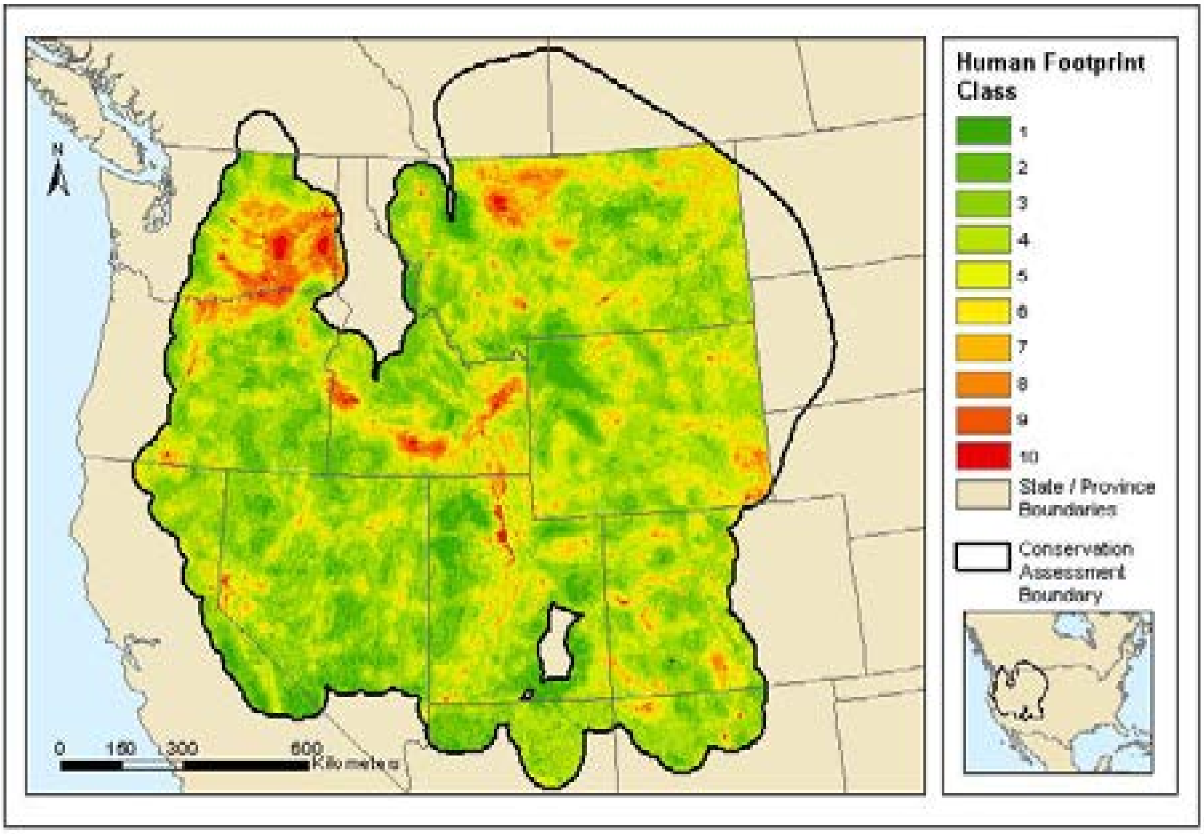


Fig. 12.10: Area of influence for the 10 human footprint classes within the sage-grouse conservation assessment and sagebrush habitat areas (percentage of total conservation assessment area shown above bars). Human footprint classes range from 1 (lowest human footprint influence) to 10 (highest human footprint influence).

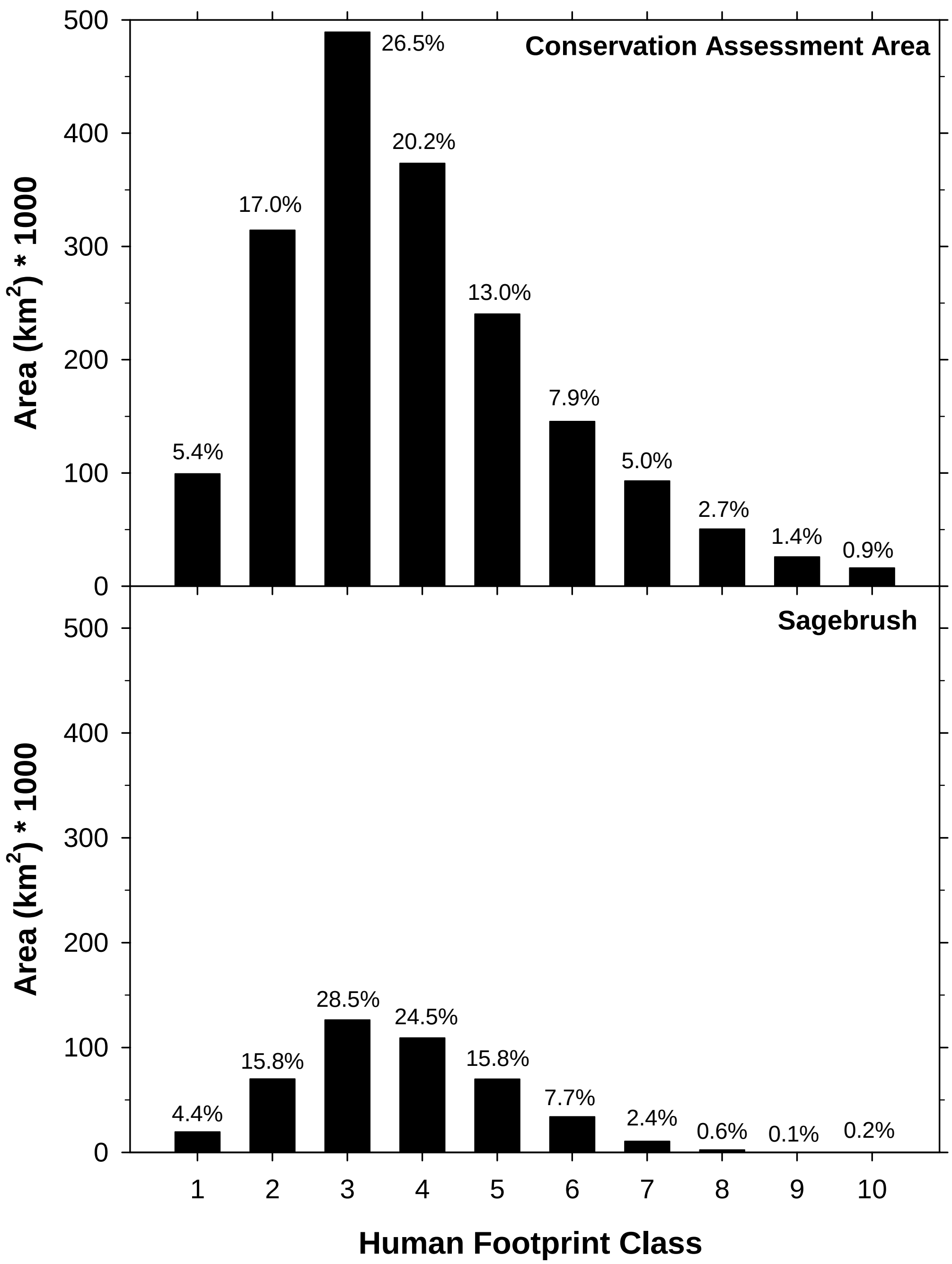
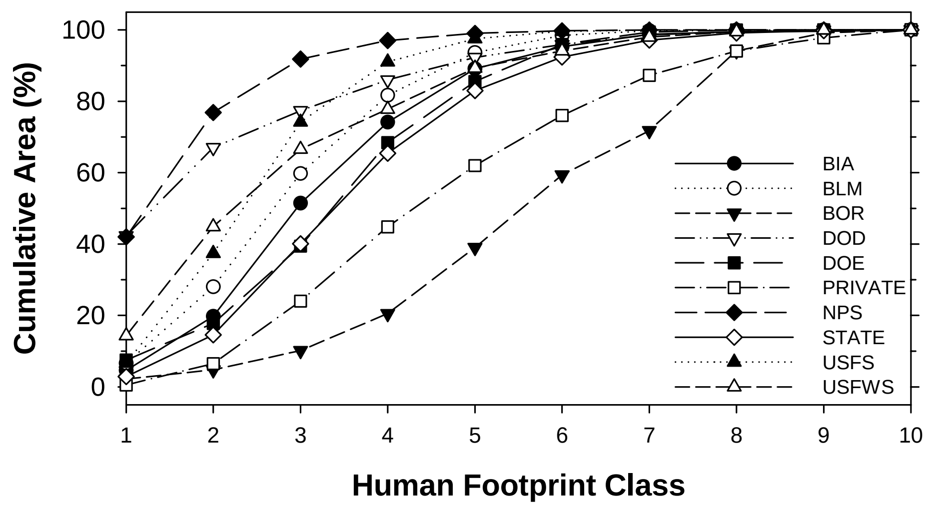


Fig. 12.11: Top graph depicts cumulative percent area for federal, state, and private landholdings versus human footprint classes. Human footprint classes range from 1 (lowest human footprint influence) to 10 (highest human footprint influence). Bottom graph depicts total area for federal, state, and private landholdings (BIA = Bureau of Indian Affairs, BLM = U.S. Bureau of Land Management, BOR = Bureau of Reclamation, DOD = Department of Defense, DOE = Department of Energy, NPS = National Park Service, USFS = U.S. Forest Service, and USFWS = U.S. Fish and Wildlife Service) within the sage-grouse conservation assessment area.



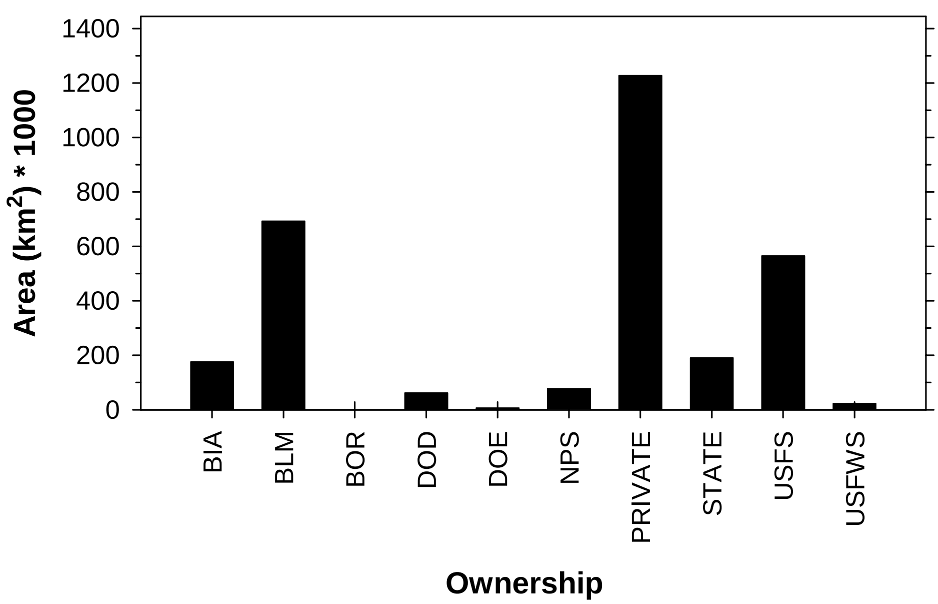


Fig. 12.12: Cumulative percent area for six floristic provinces (Miller and Eddleman 2001) versus human footprint classes within the sage-grouse conservation assessment area. Human footprint classes range from 1 (lowest human footprint influence) to 10 (highest human footprint influence).

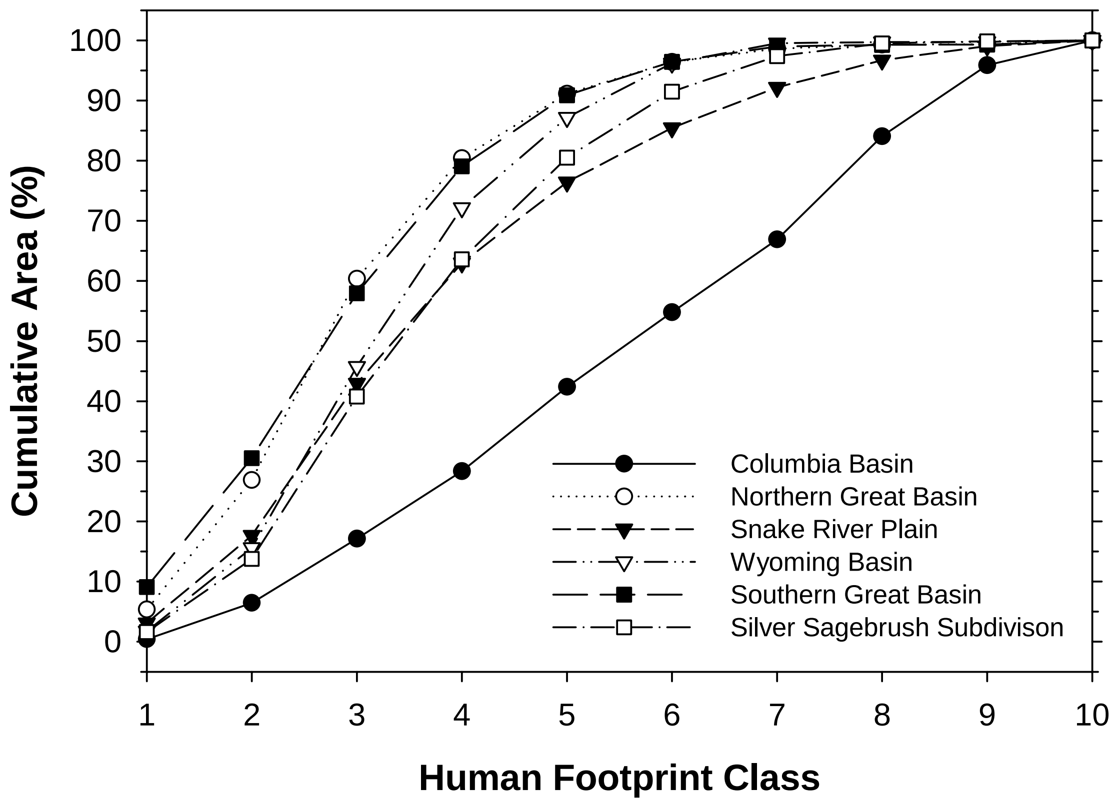
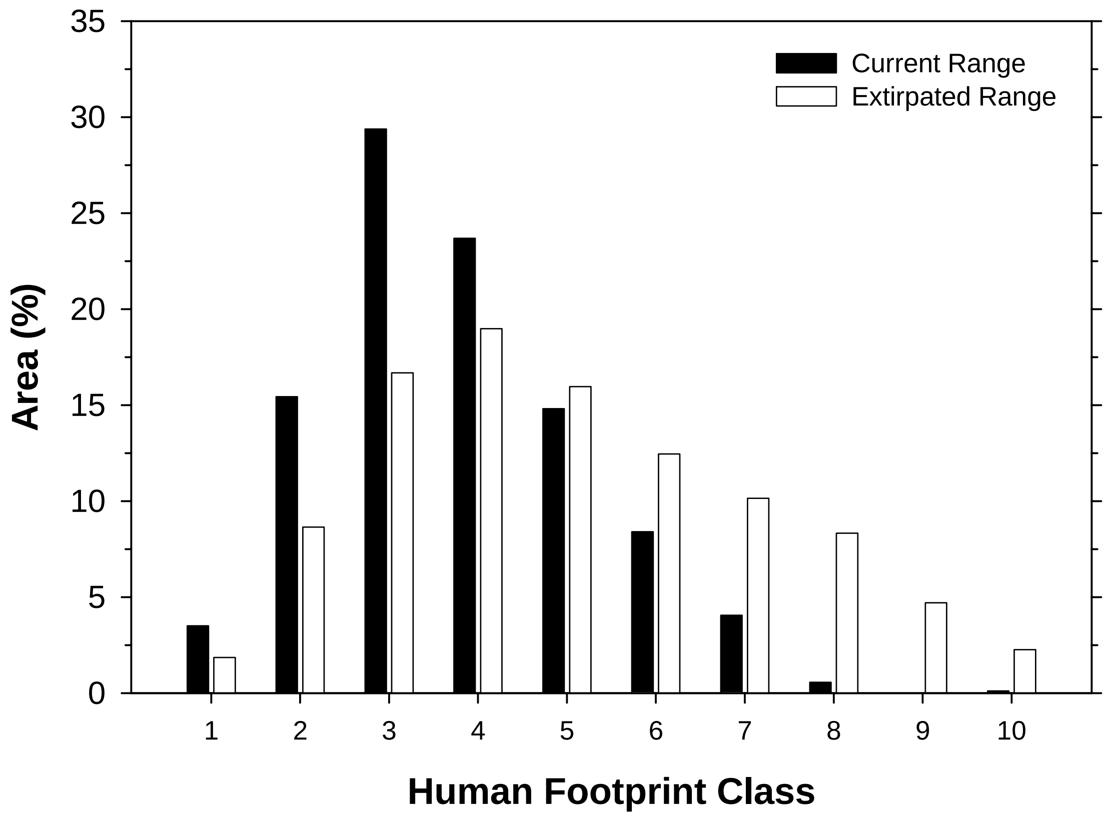


Fig. 12.13: Percent area of influence within each of the 10 human footprint classes versus current and extirpated ranges of sage-grouse. Human footprint classes range from 1 (lowest human footprint influence) to 10 (highest human footprint influence).



Chapter 13

Synthesis

CHAPTER 13

Synthesis

Abstract. This is the first range-wide assessment of greater sage-grouse (*Centrocercus urophasianus*) using the vast amount of data collected since the 1950s. The objective of this chapter was to synthesize information on greater sage-grouse habitats and populations. Our analysis of the entire sage-grouse population indicated that the abundance and distribution of greater sage-grouse declined dramatically in North America from the 1960s to the mid-1980s and then tended to stabilize. In a relatively brief ecological moment, the western landscape has been subjected to a suite of intense, frequent, or continuous disturbances, many of which have had negative impacts on sage-grouse. Here we provide information on the effects of fences on sage-grouse and the development of an interstate on the distribution and persistence of sage-grouse leks. We also discuss the effects of various habitat alterations on sage-grouse populations. Sagebrush (*Artemisia* spp*.)* habitats cannot return to some pre-settlement condition because many of the parts no longer are present or the sagebrush ecosystem has gone past a threshold from which recovery may not be possible. Recovery from these conditions may require periods longer than a century in lower elevation, more xeric conditions; longer than the 2-, 5-, or 10-year horizons of most management plans. Although we identified the dominant patterns, our analyses suggest we now need to better understand the underlying processes of sagebrush ecosystems at multiple scales. The value of this assessment may not be in what we have written, but in the data that we have presented that now can be used for advancing our understanding of the ecology of sagebrush-dominated landscapes and species that depend upon them.

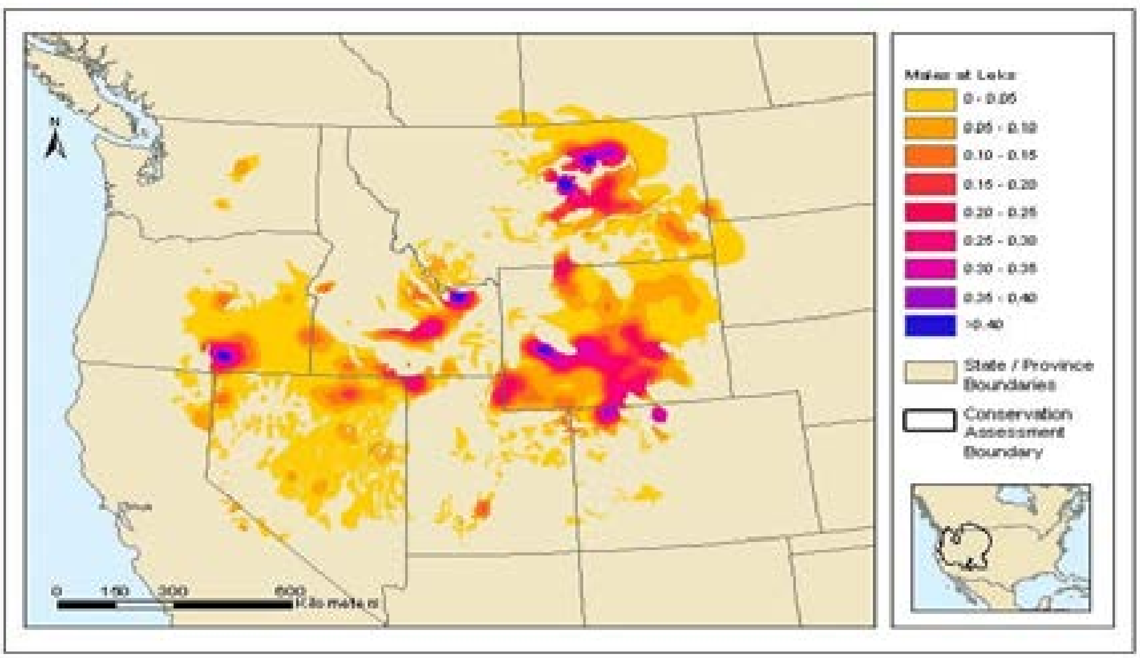
Overview

The objective of this report was to describe the status and trends of greater sage-grouse habitats and populations. In doing so we included literature spanning the last 200 years, landscape information dating back 100 years, and population data collected over the last 60 years. We attempted to include as much information as possible while eliminating data that were clearly flawed (e.g., lek counts made on non-standard dates, counts where gender was not recorded, etc.). We avoided making recommendations or suggesting policy based on our assessment of the data because those actions were beyond the scope of this report.

In many cases, actions taken >100 yrs ago strongly influenced today's situation. These actions included settlement patterns, agricultural practices, livestock use, and federal land use policy. Until fairly recently there was little concern over welfare of the sagebrush system, and species dependent on it. Sagebrush was largely seen as a constraint on productivity of the landscape. Therefore, large areas of sagebrush were eradicated and often replaced with monocultures of crested wheatgrass (*Agropyron cristatum*). By the early 1980s, widespread concerns by many interested publics necessitated a paradigm shift and large-scale sagebrush eradication programs were curtailed. However, losses were still occurring due to wildfire, agricultural practices, suburban development, and the spread of annual exotic grasses.

Sage-grouse population trends reflected these changes in landscape condition and management because populations in most states declined rather sharply from the mid-1960s to the mid-1980s and from that point on often tended to fluctuate around a much lower level compared to earlier data. This analysis also suggested that if trends characteristic of the 1960s through the mid-1980s continued, sage-grouse had a relatively high likelihood of being extirpated. However, for many populations, those trends have not continued (see Appendix 4, 5, 6). As a result, data suggest sage-grouse populations in many areas have been relatively stable for the last 15-20 years and some areas presently could be considered population strongholds (Figure 13.1). Chapter 5 (Figure 5.18) provided additional data on the distribution of sagebrush habitat that supported the information on population strongholds. However, some populations are still declining rather precipitously (Chapter 6, Table 6.23; Appendix 4, 5). Additionally, habitat change (Chapter 7) and other factors (e.g., West Nile Virus, see Chapter 10) could significantly influence the future abundance and distribution of these strongholds.

Figure 13.1. Strongholds for breeding populations of sage-grouse in western North America.



The darker shades represent the greatest densities of males/km^2.

Population Trends

The desire for data to monitor long-term trends in sage-grouse populations is a recurring theme in the scientific literature (Connelly et al. 2000*b*, Connelly et al. 2003). Most previous research has focused on the average number of males per active lek and on documentation of areas where sage-grouse have been extirpated (Aldridge and Brigham 2003, Beck et al. 2003, Braun 1995, Schroeder et al. 2004). In one case, investigators estimated annual rates of population change and applied those estimates to the interpretation of long-term population trends (Schroeder et al. 2000).

In this report, we applied several different techniques to evaluate trends for greater sage-grouse populations in North America. These techniques included: 1) changes in the average and median number of males per active lek; 2) changes in the average and median number of males per lek (including leks that are inactive); 3) annual changes in the number of males attending leks monitored in consecutive years (rate of change data); 4) evaluation of spatial patterns of lek extirpation; 5) evaluation of spatial patterns of range extirpation; and 6) delineation and evaluation of distinct breeding populations.

Most of the techniques depend on a basic understanding of leks, our ability to evaluate male attendance at leks, and the validity of applying those observations to long-term trends. For example, sage-grouse attendance at leks varies by sex, age, time of year, time of day, and weather (Jenni and Hartzler 1978, Emmons and Braun 1984, Walsh 2002). Observer bias also may influence counts (Walsh 2002). In addition, there may be annual variation in size of leks monitored, their accessibility, and the number of counts per lek.

Although the correlation between short-term changes in lek attendance and population size is sometimes difficult to support, long-term changes in lek attendance were more reliable. For example, there are many areas throughout the pre-settlement distribution of sage-grouse where localized populations or subpopulations have been extirpated. Analysis of sage-grouse numbers in these areas (regardless of the technique) inevitably shows a distinct (and usually significant) downward trend (see Schroeder et al. 2000 for example, also Appendix 4). Similarly, data on average or median numbers of leks is usually consistent with estimates of trend, regardless of the directionality of the trend. These estimates of trend have also been supported with random sampling of simulated populations (Appendix 3). Nevertheless, we have attempted to be conservative in our analysis of long-term trends throughout this report. This conservative approach included a hesitancy to interpret the magnitude of presumed increases and declines.

The abundance and distribution of greater sage-grouse have declined dramatically in North America. This statement is supported by published literature (summarized by Connelly and Braun 1997, Schroeder et al. 2004), anecdotal observations (Patterson 1952), and the analysis presented in this report. Many of the long-term changes are reflected in the regional extirpations of sage-grouse that have resulted in substantial differences between occupied habitat (668,412 km^2) and habitats that appear to have been occupied in the past (532,071 km^2) (Schroeder et al. 2004). However, regular surveys of sage-grouse leks did not begin until the 1950s or later in most areas, well past the time when many of the changes in distribution likely occurred. Therefore, most of the analyses of sage-grouse numbers were focused on the 1965-2003 period. Although most states and provinces collected data prior to 1965, this 39-year range from 1965-2003 provided an opportunity to analyze data after a sample of leks had been identified and protocols for data collection had been established and implemented (Patterson 1952).

Range-wide numbers of male sage-grouse counted on leks have declined substantially from 1965 through 2003. Although the overall trends have been downward, the declines have not been consistent throughout the 1965-2003 period. Analysis of the 39-year period indicates that these changes were often not density-independent. Sage-grouse numbers declined most between 1965 and the mid-1980s. In contrast, between the mid-1980s and 2003 populations fluctuated or declined slightly. In many areas numbers increased between 1995 and 2003. Sage-grouse trends also have varied dramatically on an annual basis. Although some of this variation was related to sampling technique and intensity (particularly in early years when fewer leks were surveyed), much of this variation also may be due to unexplored factors such as weather.

In general, overall trends were reflected in the 1965-2003 data for each state, province, region, population, and subpopulation. Some populations of sage-grouse did not fit the general pattern. For example, sage-grouse in California increased from 1965 to 2003. Within the Wyoming Basin population (the largest), sage-grouse numbers in 2003 were almost identical to their level in 1985. However, when the population was divided into subpopulations, the SW WY/NW CO/NE UT/SE ID subpopulation (primarily in SW Wyoming) increased while the NE WY/SE MT subpopulation (primarily in NE Wyoming) decreased (see Appendix 3).

On a local basis, declines largely reflected the disappearance of active leks. Although there were clear indications that the average size of active leks has declined (Chapter 6, Fig. 6.41), this difference was larger when the inactive leks are included in the analysis. This same trend was apparent in the loss and declines of populations. Forty-one populations were defined for this analysis. Five populations are now extirpated or have numbers too small to monitor. Four of these populations demonstrated significant downward trends in male attendance at leks preceding extirpation and a fifth did not have enough data for analysis. An additional 14 populations face a high risk of extirpation, due largely to their small populations (often with only 1 active lek). Three of these populations illustrated significant downward trends and one had a significant positive trend. Twelve populations were relatively small, with 7 to 18 known active leks. Nine of these populations have exhibited significant downward trends in male attendance at leks. The vast majority (92%) of active leks (and birds) were in the remaining 8 populations. Five of those populations were so large and expansive that they were divided into an additional 24 subpopulations to facilitate analysis. Data for these populations were presented in Appendix 5.

Although we did not attempt to estimate a range-wide population size, we examined the plausibility of a previous estimate of 142,000 (1997 data) for the combination of both greater and Gunnison sage-grouse (*C. minimus*) published in a non-peer reviewed report (Braun 1998). More than 50,000 male greater sage-grouse were counted on leks in 2003. Because many leks were not surveyed in 2003, but were active when they were previously surveyed, we obtained an estimate of about 25,000 males on those leks (by extrapolating a lek's last known count with the observed rate of change on leks that were counted). Because it is also likely that some males do not visit leks each day (Walsh et al. 2004) and females likely outnumber males in the population (Swenson 1986), the number of greater sage-grouse in western North America is probably much greater than the previous estimate. In part, this may be due to the apparent range-wide population increase between 1997 and 2003 (Chapter 6, Fig. 6.42). Expanded efforts to conduct surveys for sage-grouse leks also have resulted in improved information and expanded databases.

This is the first range-wide assessment of greater sage-grouse populations using the vast amount of data collected since the 1950s. Efforts to collect these data were initiated by the states and provinces because of a prescient desire to develop a system for monitoring sage-grouse populations and thus prepare to document population changes and impacts of habitat alteration. By analyzing these data now, we will be better prepared to document changes in the future. We can also learn from past mistakes in adjusting and/or re-affirming established protocols. For example, it is critical that biologists differentiate between males and females when surveying leks. Although this was clearly part of the established protocol (Connelly et al. 2003), not all biologists have followed these methods. Consequently, an unnecessary source of uncertainty was introduced into analyses conducted across large areas that incorporated these surveys. It is also important that leks that have not been surveyed for many years (particularly those known to be active on the last visit) be revisited. There are a substantial number of these leks in the overall database that have been categorized as 'unknown'. Reducing these unknowns will reduce variation in methods and increase our ability to assess population change.

Sagebrush Habitats: Trajectories of Patterns and Processes

Our focus throughout this report has been an evaluation of the array of dominant disturbances that influence sagebrush ecosystems. Periodic disturbance is a normal part of ecosystem functioning. Disturbance releases nutrients, creates space, reduces competition for resources, and maintains long-term stability among colonizing (*r*-selected) and climax (*k*-selected species) (Shugart 1998). The kinds of disturbance and their impact form a gradient from severe events at specific sites to more diffuse pressures across spatially diverse regions. The ability of a system to respond to disturbance depends on the severity, frequency, and intensity of the disturbance relative to its resiliency (Holling 1973, Southwood 1977, Urban et al. 1987). The individual and cumulative disturbances that now influence the sagebrush ecosystem may be too large, too frequent, and too intense for these systems to function as they once did.

Disturbance was a periodic component of pre-settlement sagebrush ecosystems. Although fire frequencies varied regionally, the underlying process of fire disturbance reduced sagebrush cover, promoted perennial grass and forb growth, facilitated nutrient cycling, and maintained a structurally diverse mosaic within the sagebrush-dominated landscape (Young et al. 1979). The number of fires and their frequency has increased across the sagebrush biome in recent decades (Chapter 7). Periods of extreme drought or excessive moisture also influenced sagebrush ecosystems, which over long periods changed the balance of shrub, grass, and woodlands (Tausch et al. 1993, West 1996). Other disturbances affecting sagebrush ecosystems included insect outbreaks and diseases. Grazing, primarily by buffalo, in the *Bouteloua* regions east of the Rocky Mountains was locally intense but spatially dispersed (Mack and Thompson 1982). Grazing disturbance in the eastern part of the sagebrush biome developed with sod-forming grasses compared to development of bunchgrasses in the relative absence of grazing pressures in the western intermountain region (Mack and Thompson 1982). Other herbivores, such as black-tailed jackrabbits (*Lepus californicus*) may have influenced vegetation communities during periodic fluctuations (Anderson and Shumar 1986). Therefore, disturbance and recovery in various forms affected a continuum of scales ranging from effects on single plants in the sagebrush community to influences on entire regions and resulted in a complex mosaic of landscape conditions that changed spatially as well as temporally (Miller and Eddleman 2001).

The ability of sagebrush ecosystems to respond to disturbance is a function of precipitation and water availability, elevation, vegetation cover, and soils (Anderson and Inouye 2001, West and Young 2000). For much of the sagebrush biome, the relatively low resilience to disturbance was offset by long recovery times between disturbance events. Fires in mountain big sagebrush communities in higher elevation and higher precipitation zones likely burned at 12-25 year intervals (Miller and Rose 1999, Miller and Tausch 2001). For Wyoming big sagebrush regions in more xeric, shallower soils, and lower elevation, fires may have been events occurring from 30 to >100 years (Wright and Bailey 1982).

In a brief ecological moment, the western landscape has been subjected to a new suite of intense, frequent, or continuous disturbances (West and Young 2000, Young and Sparks 2002, Griffin 2002). Prime agriculture areas were developed by the 1920s in Washington State (Buss and Dziedic 1955); agriculture and irrigation canals cover over 250,000 km^2 of the Conservation Assessment study area and influence 45% of the Conservation Assessment area and 32% of the sagebrush habitats (Table 13.1). Lands continue to be converted to agriculture because technological advances in irrigation methods now permit expansion into steeper terrains further from river flood plains. An estimated 10-20% of the sagebrush range had been treated to increase forage production for livestock by the 1960s (Schneegas 1967, Vale 1974). A minimum of 14,272 km^2 are now covered by interstate highways and roads; when all roads are included, <15% of the Conservation Assessment area and <5% of the existing sagebrush habitats are >2.5 km from a road (Table 13.1). The density of secondary roads exceeds 5 km/km^2 in some regions, opening up access for recreation, management actions, human-caused fire ignitions, and facilitating spread of exotic plant species. Within recent years, 9,510 communication towers (>62 m) have been built and provide perches for raptors that previously were not in this landscape. Oil and gas wells and pipelines collectively influence 28% of the sagebrush habitats (Table 13.1). Approximately 3,000 more permits are to be issued annually for oil and gas development in Montana, Wyoming, Colorado, and Utah. Accompanying the oil and gas construction are changes in hydrology that alter ground water tables and may influence spread of diseases such as West Nile Virus (see Chapter 10). The pipeline networks required to move oil and gas create ground disturbance, fragment remaining landscapes dominated by sagebrush, and facilitate spread of exotic plants over almost 116,000 km^2 and >6% of the sagebrush habitats (Table 13.1). Noise disturbance influences the rate of nest initiation in sage-grouse hens >3 km from construction activities surrounding oil and gas development (Lyon and Anderson 2003). Approximately 12 million AUMs are permitted for grazing on public lands in the western states. The density of fences exceeds >2 km/km^2 of habitat in some regions and influence movements of livestock, vehicles, predators, and exotic plants (Chapter 7)

The rapidity with which we can transform and develop an entire western landscape is significantly greater than the natural disturbances that previously influenced the dynamics in sagebrush ecosystems. In response to increasing demand for oil and gas resources (U.S. Departments of Interior, Agriculture, and Energy 2003) and requests by the Executive Branch for agencies to "expedite their review of permits or take other actions as necessary to accelerate the completion of such projects, while maintaining safety, public health, and environmental protections" (White House 2001), the U.S. Bureau of Land Management anticipates receiving large numbers of applications for permits to drill (4,279 applications were filed in FY2002 [1 October 2001-30 September 2003] (U.S. Bureau of Land Management 2003b) and attempts to meet a 30 to 45-day turnaround on the approval process (U.S. Bureau of Land Management 2003a). Planners estimate that 7 days are required for construction of 1.6 km of roads using heavy equipment (dozer, grader, and backhoe) to connect oil and gas well sites; the duration of most drilling operations is 2 weeks during which the well bore is drilled, cased, and prepared for completion operations (U.S. Bureau of Land Management, Draft Document of Oil and Gas Screen for sage-grouse, Colorado). Given the high likelihood of permit approval (117,234 of 122,496 applications [96%] have been authorized since 1929 [Chapter 7]), the frequency and extent of oil and gas development on sagebrush ecosystems are likely to increase in a brief period of ecological time.

The consequences of our earlier use of sagebrush habitats continue to influence current patterns and processes. Our previous history of livestock grazing has influenced soils and plant composition. Depleted plant communities facilitated invasion by cheatgrass (Young et al. 1972, Young and Allen 1997), a problem that consumes vast amounts of personnel, financial, and logistic resources today in control and in fire suppression. The large blocks of sagebrush removal of the 1950s to the 1970s that were replanted with crested wheatgrass influence distribution of wildlife populations (Reynolds and Trost 1981, Swenson et al. 1987). Prescribed fires conducted 20 years ago still influence movements of greater sage-grouse (Byrne 2002).

The first rule of conservation is to preserve all the parts (Leopold 1966, Stein et al. 2000). Sagebrush habitats cannot return to some pre-settlement condition because many of the parts no longer are present or the sagebrush ecosystem has gone past a threshold from which recovery is not possible. Recovery from these conditions may require periods longer than a century in lower elevation, more xeric conditions (Hemstrom et al. 2002, Billings 1986); longer than the 2-, 5-, or 10-year horizons of most management plans. New invasions by exotic plants and animals (including those facilitated by human development) into the sagebrush community, long-term climate changes, and increased CO_2 further compromise restoration efforts.

For many western arid lands, the energy and resources required to restore the ecosystem are too great or the political will is not present (Allen and Jackson 1992). We also have few management options available for habitat management or restoration at large-scales because of rapid and large-scale transformations. Releasing a system dominated by cheatgrass or other exotic plants from disturbance due to livestock grazing does not ensure that these systems will return to one dominated by native perennial grasses (Young and Allen 1997) but depends on site-specific characteristics that include previous disturbance regime, soils, and climate variables (West and Yorks 2002). Controlling the spread of exotic plants may require herbicides, mechanical means, or proper grazing management over large treatment areas; failure to do so may further condemn the landscape to irreversible changes (Young et al. 1981). In some arid regions containing poor soils, non-native plants may be the best option to stabilize soils and prevent further erosion (Asay et al. 2001). Without adequate soil conditions, long-term objectives to establish a sustainable sagebrush community are not possible (Society for Range Management 1995).

We presented the dominant disturbances influencing the sagebrush biome. In most cases, we were able to quantify the changes, the regional distribution of a factor, or the area influenced by the disturbance. Some factors, such as military training, may have very intense effects on habitats (Shaw and Diersing 1990, Watts 1998) that are restricted to relatively small regions across the entire sagebrush biome. Others factors, such as fences, influence sagebrush ecosystems across the entire biome but at lower intensities. The cumulative impacts of the disturbances, rather than any single source, may be the most significant influence on the trajectory of sagebrush ecosystems.

The collective human footprint was greatest in those areas that also were the most resilient because of higher precipitation and deeper soils. Many of those regions have been converted to cropland and remaining sagebrush habitats are interspersed in small patches across the landscape. In contrast, the areas in which larger patches of sagebrush remained received lower precipitation and had drier and shallower soils; those regions were the least resilient to disturbance. Those remaining landscapes of sagebrush habitats most important to sage-grouse also are the most sensitive to disturbance impacts and also will require the longest recovery periods.

We could not conduct a meaningful test for effects of livestock grazing across regions or biome-wide because we lacked the appropriate variables for the question (Milchunas and Lauenroth 1993). The question of effects of livestock grazing at large spatial scales is difficult because we lack control areas large enough to include landscape processes (Bock et al. 1993). Compounding site-specific results does not give us a cumulative estimate of effect nor tell us what the landscape would be like in the absence of grazing (National Research Council 1994). We also lack an understanding of the way sagebrush ecosystems functioned prior to the addition of livestock grazing in the 1800s (Freilich et al. 2003). Because we could not test for an effect does not mean that livestock grazing has no effect or is a compensatory use of sagebrush habitats and therefore should be ignored. Concluding no effect when one exists (Type II error) is as significant an error as concluding an effect when none exists (Type I error) (Eberhardt and Thomas 1991, Wiens and Parker 1995).

Livestock grazing influences sagebrush habitats although we do not know the full extent of that influence. Although pathways following grazing disturbance may be less predictable (Anderson and Inouye 2001, West and Yorks 2002), grazing influences vegetation components and differences exist between grazed and ungrazed regions. Additionally, infrastructure to support grazing programs, including fencing (see following section on landscape features) and water developments (Chapter 4, 7), may have both direct and indirect influences on the ecosystem.

Livestock grazing differs from herbivory in natural systems because the interaction between food availability and number of grazers is largely decoupled. Stocking rates derived by livestock managers are based on a conceptual understanding of system response to disturbance, environmental guidelines, and on external factors such as economics. Ultimately, livestock function as a keystone species: grazing and management actions to manipulate habitats do not preclude wildlife and vegetation, but they influence the ecological pathways and frequently determine which species will persist (Bock et al. 1993).

Current assessments of "rangeland health" evaluate the integrity of soil, vegetation, water and air for land areas based on comparison to ecological site descriptions or ecological reference areas (National Research Council 1994, U.S. Department of the Interior 2000, West 2003*a*). However, the ecological condition of large areas of public lands is unknown or is not surveyed with a statistically designed approach that permits an assessment over large regions (Mitchell 2000). New changes proposed to regulate livestock grazing would require more quantitative data in making management decisions and would implement grazing changes over 5 consecutive years, and make public input on public lands decisions optional (U.S. Bureau of Land

Management 2003*c*). Until we collect the appropriate quantitative data on livestock numbers, grazing intensity, timing, location, and vegetation response at the relevant spatial and temporal scales, the issue will remain unresolved (West 2003*b*).

Large numbers of habitat treatments are conducted on sagebrush habitats each year across the biome (Chapter 7). We have changed the semantics of our actions to include objectives other than increasing forage for livestock. Nonetheless, multiple use still mandates our management of sagebrush ecosystems and simply doing nothing is rarely, if ever, considered (Wambolt and Payne 1986, Wambolt et al. 2001). Unfortunately, the effects of habitat treatments are rarely monitored at the spatial and temporal scales appropriate to the wildlife response. Without objective assessment of results, the value of these treatments to better understand ecosystem response is lost. Similarly, a true program of adaptive management necessitates unbiased feedback to evaluate the influence of actions in achieving the stated objectives (Walters 1986).

Not all disturbances threaten ecosystems because the form of disturbance differs, as does the resiliency of systems to respond (Rapport et al. 1985). Disturbance is required for many ecosystem processes – it is the change in frequency or intensity of the disturbance that is important in affecting the ecosystem's response. Changes in the quantity and composition of sagebrush habitats, and their configuration in the landscape have altered natural disturbance regimes and they are now vastly different from pre-settlement conditions (West 1999, West and Young 2000). Cheatgrass is present across much of the sagebrush biome, and has resulted in increased fire frequencies and area burned, thus decreasing available sage-grouse habitat (Chapter 7). At higher elevation sites, absence of fire has permitted juniper woodland expansion into sagebrush habitats (Miller and Rose 1999), effectively reducing the suitability of those regions for greater sage-grouse.

The primary anthropogenic stresses on natural ecosystems include (1) harvesting renewable resources resulting in loss of productivity, (2) physical restructuring of the landscape from land use, (3) introduction of exotic species, and (4) discharge of toxic substances to air, land, and water (Rapport and Whitford 1999). Ecosystems that are heavily stressed lack the capacity to maintain normal function, initiating a process of degradation and lowered resilience for further disturbance (Milton et al. 1994). Many regions of the sagebrush biome now exist in an ecological state past thresholds from which recovery is likely (West 1999).

All primary anthropogenic stresses are present throughout the sagebrush biome (Table 13.1). Although public opinion is polarized on the disposition, perception, and use of public land resources (Donahue 1999), management goals are developed to streamline processing, approve permit applications, and remove impediments to commodity extraction on public lands (U.S. Departments of Interior, Agriculture, and Energy 2003, U.S. Bureau of Land Management 2003*b*).

Productivity of many areas now is less than pre-settlement (Young et al. 1981, West 1983, Holechek et al. 1999). Alteration, loss, and fragmentation of sagebrush landscapes are widespread conservation concerns (Hemstrom et al. 2002, Knick et al. 2003). Consequences of fragmentation in sagebrush habitats are increased rates of habitat loss, spread of exotic plants, and increased risk of regional extirpation of wildlife species (Knick and Rotenberry 1997, Raphael et al. 2002). Use of herbicides, insecticides, prescribed fire, and proper management of livestock grazing may be the tools best suited for the some of the large-scale actions now required to manage sagebrush habitats. However, these treatments may have negative effects on sage-grouse or other species or responses may not be monitored. We also must recognize that to benefit sage-grouse, the best approach for some habitats is to do nothing (Wambolt and Payne 1986, Wambolt et al. 2001).

Patterns emerged about the relationship between human use of western landscapes and current distribution of sagebrush habitats. Sagebrush habitats were fragmented at multiple scales from single or multiple features. Some barriers were significant, such as the separation of sage-grouse populations and sagebrush habitats by the Snake River corridor. Other disturbances, such as roads, powerlines, pipelines, and communication towers, modify habitats and landscapes to spread exotic plant species, influence predator movements and distributions, and facilitate human activities. Although we identified the dominant patterns, our analyses suggest we lack the information needed to better understand the underlying processes of sagebrush ecosystems at multiple scales.

Sage-grouse and Shrub-steppe Relationships

Habitat

The scientific literature has clearly demonstrated that sage-grouse populations are dependent on relatively large expanses of sagebrush-dominated shrub steppe. However, there is some uncertainty associated with the appropriate patch size needed for winter and breeding habitats. It is likely that this patch size is not a fixed amount but varies depending on migration patterns (Wallestad and Schladweiler 1974, Connelly et al. 1988) and productivity of the habitat (Schroeder 1997).

Numerous investigators documented the negative effects of herbicide application to sagebrush stands on sage-grouse populations (Enyeart 1956, Klebenow 1970, Martin 1970). Swenson et al. (1987) reported that the number of breeding males on their study area declined by 73% after 16% of the area was plowed. Connelly et al. (2000*a*) reported that fire resulted in a significantly greater decline in a sage-grouse breeding population in the burned area compared to a nearby control area. Byrne (2002) and Pedersen et al. (2003) also provided evidence of negative effects of fire on sage-grouse populations. Finally, Leonard et al. (2000) documented a relationship between sage-grouse habitat loss to agricultural development and an accompanying decline in sage-grouse numbers.

Landscape features

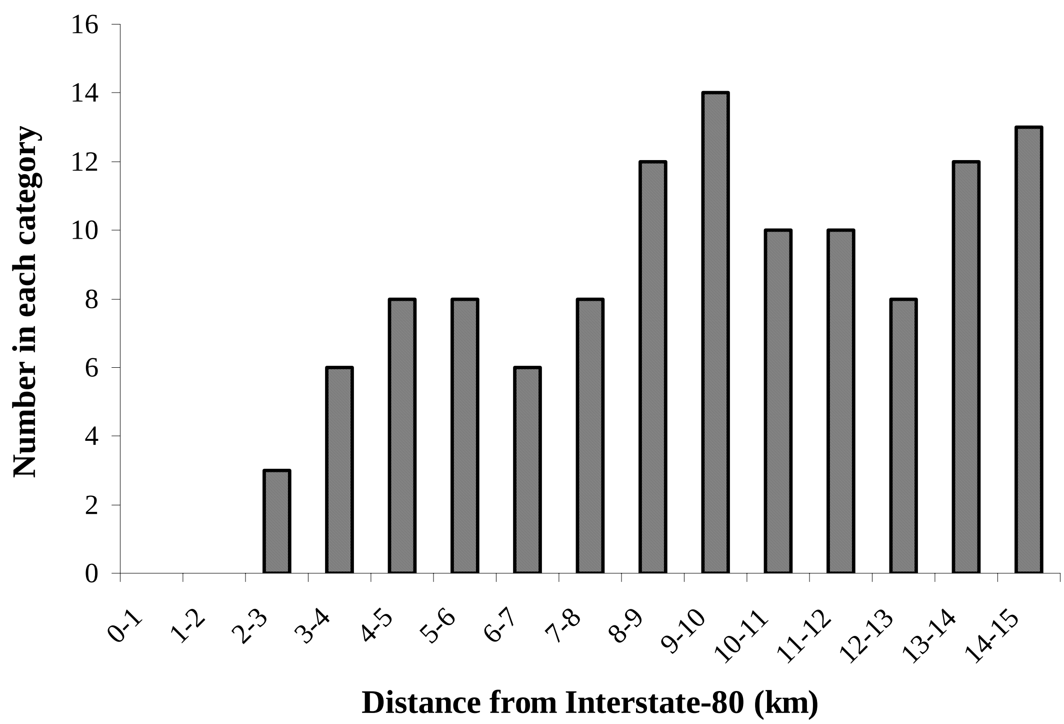
Numerous anthropogenic features have an influence on sage-grouse habitats including: farms; housing developments; roads, highways and interstates; fences; power poles; cell phone towers and other elevated structures (Chapter 7). Expanding agricultural land and housing developments have resulted in the addition of domestic dog, cat, and red fox (*Vulpes vulpes*) to sage-grouse habitats (Connelly et al. 2000*b*). Sage-grouse are adapted to a landscape with few vertical obstructions but now occupy areas that commonly have many kilometers of fences and powerlines. From 1962 to 1997, >51,000 km of fence were constructed on land administered by the U.S. Bureau of Land Management in states supporting sage-grouse populations (Connelly et al. 2000*b*); >1,000 km of fences were constructed each year from 1996 through 2002 (Chapter 7) and density of fences exceeds 2 km/km^2 in some regions of the sage-grouse range. Structures such as powerlines and fences pose hazards to sage-grouse because they provide perch sites for raptors. However, predator control is rarely recommended for sage-grouse and other species of prairie grouse for a variety of reasons including long-term consequences, relatively high cost, and public attitudes (Messmer et al. 1999, Schroeder and Baydack 2001). Grouse may also be injured or killed when they fly into these structures. In further support of this concern, an informal report was provided by the Bureau of Land Management documenting (with GPS locations and photographs) 21 incidents of sage-grouse striking a barbed wire fence in Sublette County Wyoming during spring 2003 (T. Rinkes, U.S. Bureau of Land Management, Lander, WY). Although we have a great deal of empirical data on effects of habitat change, much less is known about effects of landscape features on sage-grouse populations. Lyon and Anderson (2003) assessed the impact of energy development on sage-grouse and reported that traffic disturbance associated with natural gas developments may reduce nest initiation rates and increase distances moved from leks to nest sites.

Recent data on the effects of an interstate highway on the distribution of sage-grouse breeding populations further support Lyon and Anderson's (2003) findings. We examined the distribution of 804 leks within 100 km of Interstate 80 (I-80) across southern Wyoming and the northeastern portion of Utah. There were no leks within 2 km of the interstate (4-km wide band) and only 9 leks between 2 and 4 km of the interstate. There was only one equivalent-sized area 62-64 km from the interstate that had 8 leks; all other intervals had more leks. The average 4-km interval (2 km on either side of interstate) had 16.1 leks (95% C.I. = 14.7 – 17.5 leks). We documented 34 leks within 7.5 km of I-80 and 84 leks in an equivalent amount of area between 7.5 and 15 km of the interstate (Figure 13.2).

The majority of I-80 in Wyoming opened for traffic during the 1959-1970 period. Because most thorough lek surveys did not begin until after that time, it is likely that changes in the distribution of leks would have occurred prior to their being monitored. It is also possible that I-80 was placed in habitat less suitable for greater sage-grouse. Consequently, we also examined the persistence of known leks in relation to their distance to the interstate. An examination of lek activity in a logistic regression indicated that distance was a significant predictor (χ^2 = 3.88, P = 0.0489) of lek activity for leks within 15 km of I-80. Leks closer to the interstate were less likely to be active in 2003. An examination of long-term changes in the population between 1970 and 2003 (Figure 13.3) showed similar trends. The leks within 7.5 km of I-80 appeared to decline at a higher rate than leks 7.5 to 15.0 km from I-80.

The analysis of I-80 is preliminary and does not consider the effects of other major highways in the area. It also does not address local variation in habitat and the possibility that some leks may have moved from the area close to the interstate to an area further away, thus compensating for some of the apparent loss. Many other types of land-use activities remain to be examined including powerlines, pipelines, communication towers, and oil and gas wells. This type of analysis offers potential for examining past effects as well as for making predictions about future impacts.

Figure 13.2. Distribution of greater sage-grouse leks with respect to Interstate 80 in Wyoming and northeastern Utah, 2003.



Conclusion

The value of this assessment may not be in what we have written, but in the data that we have presented that now can be used for advancing our understanding of the ecology of sagebrush-dominated landscapes and species that depend upon them. Concerns for the conservation of sage-grouse and sagebrush ecosystems have been expressed for a long time (Patterson 1952, Braun et al. 1976). However, the inability to quantify and address the primary issues across the entire sagebrush biome limited those concerns because they lacked the breadth and geographic and temporal scope of information that we have presented in this assessment. Other large-scale, highly contentious natural resource issues, such as those surrounding conservation of spotted owls (*Strix occidentalis*), ultimately have resulted in significant contributions to conservation, ecology, and management (Noon and Franklin 2002). Similarly, we hope that the data that we have presented in this assessment will permit effective conservation plans to be developed that will ensure the species survival for generations to come.

Figure 13.3. Population trends for leks relatively close to and far from I-80 in Wyoming and northeastern Utah.

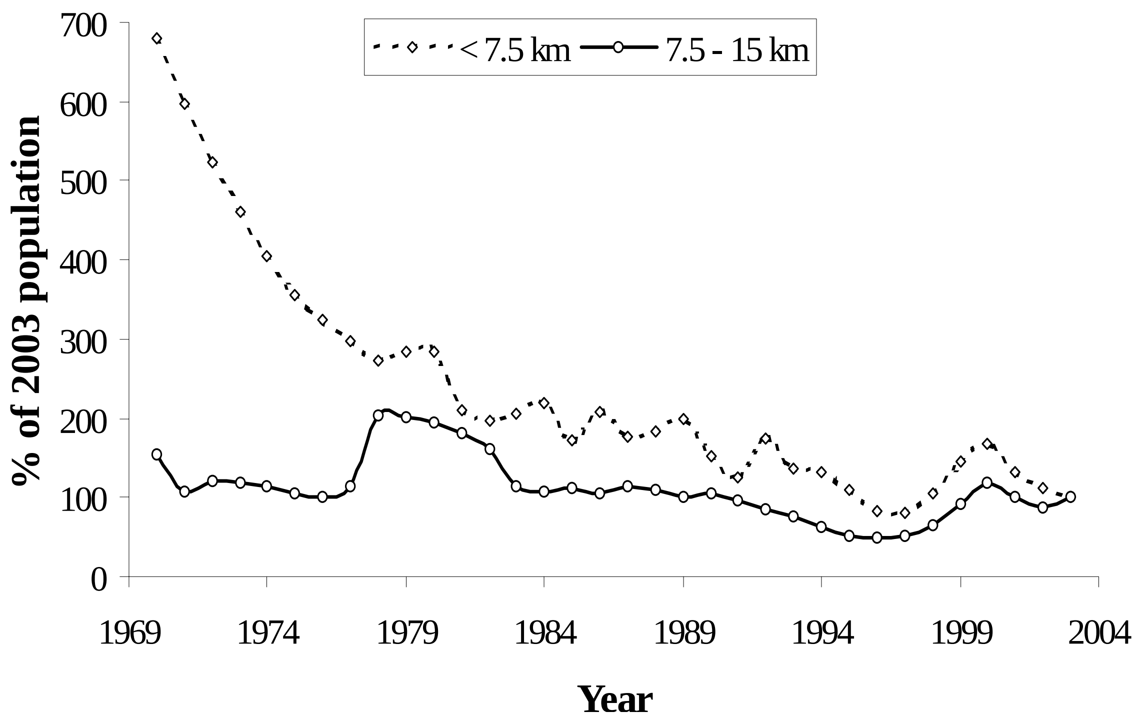


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Table 13.1. Summary of influences across the range-wide distribution of greater sage-grouse and sagebrush habitats. Only influences for which we could develop a spatial estimate of area of influence were included in the summary. Other influences, such as livestock grazing, have more diffuse effects exerted unevenly across the sagebrush biome and could not be included in this analysis. Two buffer sizes were considered, based on ranges reported in literature sources; the high value was reported when only one buffer size was calculated.

				Area of Influence					
				Effective Area[a] (km^2)		Conservation Assessment Area (%)		Sagebrush Area (%)	
	Total Area (km^2)	Conservation Assessment Area[b] (%)	Buffer Size (low/high)	Low	High	Low	High	Low	High
Sagebrush Area[c]	481,384	21.6							
"Natural" Disturbance									
Fires (1990-present)	79,755	3.9							
Shrub brown out	2,544	0.1					0.1		0.2
Agriculture									
Cropland	248,975	12.1	2.5/6.9 km[d]	740,760	1,152,157	35.8	55.6	23.7	48.5
Irrigation canals	6,916	0.3	2.5/6.9 km	193,175	462,146	9.3	22.3	8.2	23.2
Urban									
Urban development	1981	0.1	6.9 km		75,191		3.6		2.1
Landfills	196	>0.1	2.5/6.9 km	22,650	137,954	1.1	6.7	1.0	6.1
Interstates/highways[e]	14,272	1.0	7 km		841,927		40.6		37.2
All roads	159,279	7.7	2.5/6.9 km	1,768,793	1,912,463	85.4	92.3	95.1	99.6

Railroads	137	>0.1	3 km	183,915	183,915		8.9		6.1
Powerlines	15,296	1.0	5/6.9 km	672,344	837,390	32.5	40.4	32.4	40.3
Communication towers	95	>0.1	3.2 km		99,135		4.8		3.6
Energy Development									
Oil/gas wells[f]	1960	0.1	3 km	393,744			19.0		21.0
Pipelines[g]	2,790	0.1	1 km	116,571			5.6		6.9
Wind energy[h]	35,354	1.7	5 km	383,705			18.5		14.9
Military Training									
Installations/training	26,043	1.3					1.3		1.5
Recreation									
Rest Areas	8	<0.1	2.5/6.9	5,313	94,396	0.3	4.6	0.2	4.4
Campgrounds	84	<0.1	2.5/6.9	46,143	238,811	2.2	11.5	1.1	7.3
Cumulative Total									
Low Influence				2,027,516		98.2		99.5	
High influence					2,057,209		99.7		99.9

[a]Effective area was the total area of the individual effect plus the area included surrounding in the buffer zone.
[b]The total area included in the Conservation Assessment Study Area was 2,062,872 km^2 (Chapter 1).
[c](Comer et al. 2002, this study) (Chapter 1). Sagebrush area in the eastern portion of the Conservation Study area, including Montana, North Dakota, and South Dakota are likely underrepresented (Chapter 1, 5).
[d]Range of foraging areas for nonbreeding common raven (*Corvus corax*) (Boarman and Heinrich 1999).
[e]Buffer value determined from this study of I-80 (Chapter 13).
[f]Effective distance of influence of oil/gas construction activities on sage-grouse nesting (Lyon and Anderson 2003).
[g]Distance for spread of invasive plant species away from road disturbance (Gelbard and Belnap 2003).
[h]Percent of potential area in high category

Appendix

APPENDIX 1

Sage-grouse Memorandums of Understanding

Conservation and Management of Sage Grouse in North America
Western Associations of Fish and Wildlife Associations
1999

Western Association of Fish and Wildlife Agencies
U.S. Department of Agriculture, Forest Service
U.S. Department of the Interior, Bureau of Land Management
U.S. Department of the Interior, Fish and Wildlife Service
2000

MEMORANDUM OF UNDERSTANDING

AMONG

MEMBERS OF WESTERN ASSOCIATION OF

FISH AND WILDLIFE AGENCIES

Conservation and Management of Sage Grouse in North America

I. Purpose

The purpose of this Memorandum of Understanding (MOU) is to provide guidance for conservation and management of sage grouse (Centrocercus spp.) and sagebrush (Artemisia spp., primarily A. tridentata tridentata, A. t. vaseyana, A. t. wyomingensis, A. tripartita) shrub-steppe habitats upon which the species depends. Sage grouse historically occurred in at least 15 states and 3 provinces. This species has become extirpated in 5 states (Arizona, Kansas, Nebraska, New Mexico, Oklahoma) and 1 province (British Columbia). The current distribution of sage grouse is reduced throughout the species' historic range. Reasons for the reduction in area occupied from presettlement periods relate to habitat loss, habitat degradation, and habitat fragmentation. The long-term trend in sage grouse abundance is downward. The members of the Western Association of Fish and Wildlife Agencies agree that cooperative efforts are necessary to collect and analyze data on sage grouse and their habitats so that cooperative plans may be formulated and initiated to maintain the broadest distribution and greatest abundance possible within the fiscal realities of the member agencies and cooperating partners.

II. Objectives

All member affected agencies agree that sage grouse are an important natural component of the sagebrush shrub-steppe ecosystem. As such, sage grouse serve as an indicator of the overall health of this important habitat type in western North America. Further, the presence and abundance of sage grouse reflects humankind's commitment to maintaining all natural components of the sagebrush shrub-steppe ecosystem so that all uses of this type are sustainable over time. Specific objectives are:

1. Maintain and increase where possible the present distribution of sage grouse.
2. Maintain and increase where possible the present abundance of sage grouse.
3. Develop strategies using cooperative partnerships to maintain and enhance the specific habitats used by sage grouse throughout their annual cycle.

4. Conduct management experiments on a sufficient scale to demonstrate that management of habitats can stabilize and enhance sage grouse distribution and abundance.
5. Collect and analyze population and habitat data throughout the range of sage grouse for use in preparation of conservation plans.

III. Actions

It is the intent of the members of the Western Association of Fish and Wildlife Agencies to sustain and enhance the distribution and abundance of sage grouse through responsible collective management programs. These programs will include:

1. Identification of the present distribution of sage grouse in each member state/province.
2. Collection of sage grouse population data following standardized protocols throughout the range of the species..
3. Continuation of development of Conservation Plans based on the local working group concept.
4. Validation of habitat evaluation models.
5. Completion of genetic analyses across the range of sage grouse to more effectively define and manage individual populations.
6. Development of cooperative partnerships with interested individuals, and private, state, and federal land managers.
7. Support and implement the revised sage grouse population and habitat management guidelines.

IV. Responsibilities

1. Each state/province will collect data as recommended by the Western States Sage Grouse and Columbian Sharp-tailed Grouse Technical Committee within the constraints of their budgetary process.
2. All member states/provinces will work cooperatively to maintain and enhance sage grouse and their habitats.

V. Approval

We, the undersigned designated officials, do hereby approve this Memorandum of Understanding as recommended by resolution at the Summer Meeting of the Western Association of Fish and Wildlife Agencies in Durango, Colorado on 14 July 1999.

MEMORANDUM OF UNDERSTANDING

AMONG

WESTERN ASSOCIATION OF FISH AND WILDLIFE AGENCIES

and

U.S. DEPARTMENT OF AGRICULTURE, FOREST SERVICE

and

U.S. DEPARTMENT OF THE INTERIOR, BUREAU OF LAND MANAGEMENT

and

U.S. DEPARTMENT OF THE INTERIOR, FISH AND WILDLIFE SERVICE

I. Purpose

The purpose of this Memorandum of Understanding (MOU) is to provide for cooperation among the participating state and federal land and wildlife management agencies in the development of a rangewide strategy for the conservation and management of sage grouse (Centrocercus spp.) and their sagebrush (Artemisia) habitats. The sage grouse is an obligate sagebrush habitat species that requires large tracts of sagebrush habitat for its survival. Sage grouse historically occurred in at least 16 states and three provinces. This species has been extirpated in five states (Arizona, Kansas, Nebraska, New Mexico, and Oklahoma) and one Canadian province (British Columbia). Its current range includes portions of California, Oregon, Washington, Nevada, Idaho, Utah, Montana, Wyoming, Colorado, North Dakota and South Dakota. The long-term trend in sage grouse abundance is downward throughout its range.

Member state agencies ("State Agencies") of the Western Association of Fish and Wildlife Agencies (WAFWA) have signed a "Memorandum of Understanding Among Members of the Western Association of Fish and Wildlife Agencies for the Conservation and Management of Sage Grouse in North America." That MOU, signed in July of 1999, and attached hereto as Appendix A, outlines the purpose, objectives, actions and responsibilities for cooperation among WAFWA States.

The Bureau of Land Management, United States Department of the Interior (BLM), the Forest Service, United States Department of Agriculture (FS) and the Fish and Wildlife Service, U.S. Department of the Interior (FWS), and WAFWA, (collectively, "the Parties") herein agree that cooperative efforts among the Parties, consistent with the applicable statutory requirements, are necessary to conserve and manage the nation's sagebrush ecosystems for the benefit of sage grouse and all other sagebrush dependent species.

II. Objectives

The Parties agree that sage grouse are an important natural component of the sagebrush ecosystem. Sage grouse serve as an indicator of the overall health of this important ecosystem in Western North America. Providing for the presence and abundance of sage grouse reflects the Parties commitment to maintaining all natural components and ecological processes within sagebrush ecosystems. Specific objectives are to:

1. Maintain, and increase, where possible, the present distribution of sage grouse.
2. Maintain, and increase, where possible, the present abundance of sage grouse.
3. Identify the impacts of major land uses and hunting on sage grouse, and determine the primary causes for declines in sage grouse populations.
4. Develop a Rangewide Conservation Framework to provide for cooperation and integration in the development of Conservation Plans to address conservation needs across geographic scales as appropriate.
5. Develop partnerships with agencies, organizations, tribes, communities, individuals and private landowners to cooperatively accomplish the preceding objectives.

III. Actions

The States will convene Working Groups to develop State or Local Conservation Plans. Working Groups will be comprised of representatives of local, state, federal and tribal governments, as appropriate. Participation will be open to all other interested parties. Federal participation in working groups will operate in a manner consistent with the Federal Advisory Committee Act. Working groups will be convened within 60 days of the effective date of this agreement.

The Parties will establish a Conservation Planning Framework Team consisting of four (4) representatives from WAFWA and one (1) representative each from BLM, FS and FWS. The Framework Team will develop a Range-wide Conservation Framework and provide recommendations and guidance to the working groups concerning the contents of State and Local Conservation Plans.

The Parties will collect, analyze and distribute sage grouse population and habitat data to the working groups for conservation planning. These data include, at a minimum: data on fire history, habitat composition and trend, known wintering and nesting habitat, and lek locations. Population data will be collected as recommended by the Western States Sage Grouse and Columbian Sharptailed Grouse Technical Committee.

Each State Conservation Plan will provide recommendations:

1. To protect and improve important sage grouse sagebrush habitats.
2. To actively manage to improve degraded sagebrush ecosystems.
3. To reduce the fragmentation and isolation of sagebrush habitats.
4. To address non-habitat issues, such as hunting, if such issues are identified to limit sage grouse populations in an area.
5. For desired population levels, distribution and habitat conditions.

The BLM, FS and FWS will provide for habitat protection, conservation and restoration, as appropriate, consistent with the National Environmental Policy Act and other applicable laws, regulations, directives and policies. In doing so, the BLM, FS, and FWS will consider the WAFWA Guidelines for Management of Sage Grouse Populations and Habitats, State and Local Conservation Plans, and other appropriate information in their respective planning processes.

Parties to this agreement will work together to identify research needs and strategies and conduct joint assessments, monitoring and research.

IV. Authorities

This MOU is among the FWS, BLM, FS, and WAFWA under the provisions of the following laws:

Federal Land and Policy Management Act of 1976 (43 U.S.C. 1701 et seq.)
Fish and Wildlife Act of 1956 (16 U.S.C. 742 et seq.)
Fish and Wildlife Coordination Act (16 U.S.C. 661-667)
Multiple-Use Sustained-Yield Act [of 1960] (16 U.S.C. 528-531)
Forest and Rangeland Renewable Resources Research Act of 1978 (16 U.S.C. 1641-48)
National Forest Management Act of 1976 (16 U.S.C. 1600 et seq.)
Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.)
National Wildlife Refuge Administration Act of 1966, as amended by the National Wildlife
Refuge System Improvement Act of 1997 (16 U.S.C 668dd et seq.)

V. Approval

It is mutually agreed and understood by and between the Parties that:

1. This MOU is neither a fiscal nor a funds obligation document. Nothing in this agreement may be construed to obligate Federal Agencies or the United States to any current or future expenditure of resources in advance of the availability of appropriations from Congress. Any endeavor involving reimbursement or contribution of funds between the Parties to this MOU will be handled in accordance to applicable

regulations, and procedures including those for federal government procurement and printing. Such endeavor will be outlined in separate agreements that shall be made in writing by representatives of the Parties and shall be independently authorized in accordance with appropriate statutory authority. This MOU does not provide such authority.

2. This MOU in no way restricts the Parties from participating in similar activities with other public or private agencies, organizations and individuals.

3. This MOU is executed as of the last date shown below and expires five years from the execution date, at which time it will be subject to review, renewal or expiration.

4. Modifications within the scope of this MOU shall be made by the issuance of a mutually executed modification prior to any changes being performed.

5. Any party to this MOU may withdraw with a 60-day written notice.

6. Any press releases with reference to this MOU, the Parties, or the relationship established between the Parties of this MOU, shall be reviewed and agreed upon by all of the Parties.

7. In any advertising done by any of the Parties, this MOU should not be referred to in a manner that states or implies that any Party approves of or endorses unrelated activities of any other.

8. During the performance of the MOU the participants agree to abide by the terms of Executive Order 11246 on nondiscrimination and will not discriminate against any person because of race, age, color, religion, gender, national origin or disability.

9. No member of, or delegate to Congress, or resident Commissioner, shall be admitted to any share or part of this agreement, or to any benefit that may arise from, but these provisions shall not be construed to extend to this agreement if made with a corporation for its general benefits.

10. The Parties agree to implement the provisions of this MOU to the extent personnel and budgets allow. In addition, nothing in the MOU is intended to supercede any laws, regulations or directives by which the Parties must legally abide.

IN WITNESS THEREOF, the parties hereto have executed this Memorandum of Understanding as of the last written date below.

APPENDIX 2

Policy for Evaluation of Conservation Efforts
PECE

preferred the rulemaking petition. The coordinates for Channel 287C3 at Alamo are 32–19–29 North Latitude and 82–43–23 West Longitude. This allotment has a site restriction of 20.4 kilometers (12.7 miles) north of Alamo.

DATES: Effective April 28, 2003.

FOR FURTHER INFORMATION CONTACT: R. Barthen Gorman, Media Bureau, (202) 418–2180.

SUPPLEMENTARY INFORMATION: This is a synopsis of the Commission's Report and Order, MM Docket No. 01–111, adopted March 12, 2003, and released March 14, 2003. The full text of this Commission decision is available for inspection and copying during normal business hours in the FCC's Reference Information Center at Portals II, 445 12th Street, SW., Room CY–A257, Washington, DC, 20554. The document may also be purchased from the Commission's duplicating contractor, Qualex International, Portals II, 445 12th Street, SW., Room CY–B402, Washington, DC, 20554, telephone 202 863–2893. facsimile 202 863–2898, or via e-mail *qualexint@aol.com.*

List of Subjects in 47 CFR Part 73

Radio, Radio broadcasting.

■ Part 73 of Title 47 of the Code of Federal Regulations is amended as follows:

PART 73—RADIO BROADCAST SERVICES

■ 1. The authority citation for Part 73 reads as follows:

Authority: 47 U.S.C. 154, 303, 334 and 336.

§ 73.202 [Amended]

■ 2. Section 73.202(b), the Table of FM Allotments under Georgia, is amended by adding Alamo, Channel 287C3.

Federal Communications Commission.

John A. Karousos,

Assistant Chief, Audio Division Media Bureau.

[FR Doc. 03–7470 Filed 3–27–03; 8:45 am]

BILLING CODE 6712–01–P

FEDERAL COMMUNICATIONS COMMISSION

47 CFR Part 73

[DA 03–629; MB Docket No. 02–120; RM–10442]

Radio Broadcasting Services; Owen, Wisconsin

AGENCY: Federal Communications Commission.

ACTION: Final rule.

SUMMARY: The Audio Division, at the request of Starboard Broadcasting, Inc., allots Channel 242C3 at Owen, Wisconsin, as the community's first local FM service. Channel 242C3 can be allotted to Owen, Wisconsin, in compliance with the Commission's minimum distance separation requirements with a site restriction of 12.9 km (8.0 miles) northeast of Owen. The coordinates for Channel 242C3 at Owen, Wisconsin, are 45–03–08 North Latitude and 90–29–21 West Longitude. A filing window for Channel 242C3 at Owen, WI, will not be opened at this time. Instead, the issue of opening this allotment for auction will be addressed by the Commission in a subsequent Order.

DATES: Effective April 28, 2003.

FOR FURTHER INFORMATION CONTACT: Deborah Dupont, Media Bureau, (202) 418–2180.

SUPPLEMENTARY INFORMATION: This is a synopsis of the Commission's Report and Order, MB Docket No. 02–120, adopted March 12, 2003, and released March 14, 2003. The full text of this Commission decision is available for inspection and copying during normal business hours in the FCC Information Center, Portals II, 445 12th Street, SW., Room CY–A257, Washington, DC 20554. The complete text of this decision may also be purchased from the Commission's duplicating contractor, Qualex International, Portals II, 445 12th Street, SW., Room CY–B402, Washington, DC, 20554, (202) 863–2893, facsimile (202) 863–2898, or via e-mail *qualexint@aol.com.*

List of Subjects in 47 CFR Part 73

Radio, Radio broadcasting.

■ Part 73 of title 47 of the Code of Federal Regulations is amended as follows:

PART 73—RADIO BROADCAST SERVICES

■ 1. The authority citation for part 73 continues to read as follows:

Authority: 47 U.S.C. 154, 303, 334 and 336.

§ 73.202 [Amended]

■ 2. Section 73.202(b), the Table of FM Allotments under Wisconsin, is amended by adding Owen, Channel 242C3.

Federal Communications Commission.

John A. Karousos,

Assistant Chief, Audio Division, Media Bureau.

[FR Doc. 03–7472 Filed 3–27–03; 8:45 am]

BILLING CODE 6712–01–P

DEPARTMENT OF THE INTERIOR

Fish and Wildlife Service

DEPARTMENT OF COMMERCE

National Oceanic and Atmospheric Administration

50 CFR Chapter IV

[Docket No. 000214043–2227–02; I.D. 011603A]

RIN 1018–AF55, 0648–XA48

Policy for Evaluation of Conservation Efforts When Making Listing Decisions

AGENCIES: Fish and Wildlife Service, Interior; National Marine Fisheries Service, NOAA, Commerce.

ACTION: Announcement of final policy.

SUMMARY: We, the Fish and Wildlife Service (FWS) and the National Marine Fisheries Service (NMFS) (the Services), announce a final policy for the evaluation of conservation efforts when making listing decisions (PECE) under the Endangered Species Act of 1973, as amended (Act). While the Act requires us to take into account all conservation efforts being made to protect a species, the policy identifies criteria we will use in determining whether formalized conservation efforts that have yet to be implemented or to show effectiveness contribute to making listing a species as threatened or endangered unnecessary. The policy applies to conservation efforts identified in conservation agreements, conservation plans, management plans, or similar documents developed by Federal agencies, State and local governments, Tribal governments, businesses, organizations, and individuals.

DATES: This policy is effective April 28, 2003.

ADDRESSES: Chief, Division of Conservation and Classification, U.S. Fish and Wildlife Service, 4401 North Fairfax Drive, Arlington, VA 22203 (Telephone 703/358–2171, Facsimile 703/358–1735); or Chief, Endangered Species Division, National Marine Fisheries Service, Office of Protected Resources, 1315 East-West Highway, Silver Spring, MD 20910 (Telephone 301/713–1401, Facsimile 301/713–0376).

FOR FURTHER INFORMATION CONTACT: Chris Nolin, Chief, Division of Conservation and Classification, U.S. Fish and Wildlife Service at the above address, telephone 703/358–2171 or facsimile 703/358–1735, or Margaret Lorenz, Endangered Species Division, National Marine Fisheries Service at the above address, telephone 301/713–1401 or facsimile 301/713–0376.

SUPPLEMENTARY INFORMATION:

Background

This policy provides direction to Service personnel in determining how to consider a conservation agreement when making a decision on whether a species warrants listing under the Act. It also provides information to the groups interested in developing agreements or plans that would contribute to making it unnecessary for the Services to list a species under the Act.

On June 13, 2000, we published in the **Federal Register** (65 FR 37102) a draft policy for evaluating conservation efforts that have not yet been implemented or have not yet demonstrated effectiveness when making listing decisions under the Act. The policy establishes two basic criteria: (1) The certainty that the conservation efforts will be implemented and (2) the certainty that the efforts will be effective. The policy provides specific factors under these two basic criteria that we will use to direct our analysis of the conservation effort. At the time of making listing determinations, we will evaluate formalized conservation efforts (i.e., conservation efforts identified in a conservation agreement, conservation plan, management plan, or similar document) to determine if the conservation effort provides certainty of implementation and effectiveness and, thereby, improves the status, as defined by the Act, of the species such that it does not meet the Act's definition of a threatened or endangered species.

When we evaluate the certainty of whether the formalized conservation effort will be implemented, we will consider the following: Do we have a high level of certainty that the resources necessary to carry out the conservation effort are available? Do the parties to the conservation effort have the authority to carry it out? Are the regulatory or procedural mechanisms in place to carry out the efforts? And is there a schedule for completing and evaluating the efforts? If the conservation effort relies on voluntary participation, we will evaluate whether the incentives that are included in the conservation effort will ensure the level of participation necessary to carry out the conservation effort. We will also evaluate the certainty that the conservation effort will be effective. In making this evaluation, we will consider the following: Does the effort describe the nature and extent of the threats to the species to be addressed and how these threats are reduced by the conservation effort? Does the effort establish specific conservation objectives? Does the effort identify the appropriate steps to reduce threats to the species? And does the effort include quantifiable performance measures to monitor for both compliance and effectiveness? Overall, we need to be certain that the formalized conservation effort improves the status of the species at the time we make a listing determination.

This policy is important because it gives us a consistent set of criteria to evaluate formalized conservation efforts. For states and other entities that are developing agreements or plans, this policy informs them of the criteria we will use in evaluating formalized conservation efforts when making listing decisions, and thereby guides States and other entities that wish to develop formalized conservation efforts that may contribute to making listing unnecessary.

In the notice of the draft policy, we specifically requested comments on the criteria that we would use to evaluate the certainty that a formalized conservation effort will be implemented. Also, we requested comments on the timing of the development of conservation agreements or plans. We have learned that timing is the most critical element when developing a successful conservation agreement or plan. Encouraging and facilitating early development of conservation agreements or plans is an important objective of this policy. Last-minute agreements (i.e., those that are developed just before or after a species is proposed for listing) often have little chance of affecting the outcome of a listing decision. Once a species is proposed for listing under the Act, we may have insufficient time to include consideration of a newly developed conservation plan in the public notice and comment process and still meet our statutory deadlines. Last-minute efforts are also less likely to be able to demonstrate that they will be implemented and effective in reducing or removing threats to the species. In addition, there are circumstances in which the threats to a species are so imminent and/or complex that it will be almost impossible to develop an agreement or plan that includes conservation efforts that will result in making the listing unnecessary. Accordingly, we encourage the early development of formalized conservation efforts before the threats become too extreme and imminent and when there is greater flexibility in sufficiently improving a species' status to the point where listing the species as threatened or endangered is unnecessary.

Summary of Comments and Recommendations

In response to our request for comments on the draft policy, we received letters from 44 entities. Thirty-five were in support of the policy and nine were against. We reviewed all comments received and have incorporated accepted suggestions or clarifications into the final policy text. Because most of these letters included similar comments (several were form letters) we grouped the comments according to issues. The following is a summary of the relevant comments and our responses. We also received comments that were not relevant to the policy and, therefore, outside the policy's scope. We responded to some of these comments where doing so would clarify the process for determining whether a species is endangered or threatened (the listing process) or clarify the nature of conservation plans, agreements, and efforts.

Policy Scope Issues

Issue 1: Many commenters felt that this policy should also apply to downlisting species from endangered to threatened status and delisting actions, or else parties to an agreement where the final decision is to list the species would not have any incentives to take action on a listed species until a recovery plan is developed. In addition, one commenter suggested that the policy scope should be expanded to include the process of designating critical habitat.

Response 1: We believe that the immediate need is to develop criteria that will guide consistent and predictable evaluation of conservation efforts at the time of a listing determination. We may consider such a policy for downlisting or delisting actions in the future. However, we note that a recovery plan is the appropriate vehicle to provide guidance on actions necessary to delist a species. Also, we may consider developing a similar policy for critical habitat designations.

Issue 2: Two commenters stated that our estimates of time needed to develop, implement, monitor, and report on conservation efforts are underestimated.

Response 2: We agree that our original estimates were too low. We have increased our estimate to an average of 2,500 person-hours to complete a conservation agreement (with a range of 1,000 to 4,000 person-hours). We also increased our estimate of the average number of person-hours to conduct monitoring and to prepare a report to

320 and 80 hours, respectively. We expect the amount of time will vary depending on several factors including, but not limited to, the number of species addressed, amount of biological information available on the species, and the complexity of the threats. Therefore, we have provided an average to assist interested parties in their planning efforts.

Issue 3: One commenter questioned whether we would evaluate proposed agreements or plans using the stated criteria automatically or only upon request. The commenter also questioned whether we will consider agreements or plans that we previously determined were not sufficient to prevent the need for listing in combination with ''new'' proposed agreements or plans when we evaluate whether to list a species.

Response 3: If a listing proposal is under review, we will consider any conservation effort. We will evaluate the status of the species in the context of all factors that affect the species' risk of extinction, including all known conservation efforts whether planned, under way, or fully implemented. However, for formalized conservation efforts not fully implemented, or where the results have not been demonstrated, we will consider the PECE criteria in our evaluation of whether, and to what extent, the formalized conservation efforts affect the species' status under the Act.

Issue 4: One commenter asked the length of time for which a plan is approved.

Response 4: The PECE is not a plan-approval process, nor does it establish an alternative to listing. PECE outlines the criteria we will consider when evaluating formalized conservation efforts that have not yet been fully implemented or do not yet have a record of effectiveness at the time we make a listing decision. Should the status of a species decline after we make a decision not to list this species, we would need to reassess our listing decision. For example, there may be situations where the parties to a plan or agreement meet their commitments, but unexpected and/or increased threats (e.g., disease) may occur that threaten the species' status and make it necessary to list the species.

Issue 5: One commenter asked if the ''new information'' reopener is operative at any time.

Response 5: Yes, because section 4(b)(1) of the Act requires us to use the best available scientific and commercial data whenever making decisions during the listing process. In making a decision whether to list a species, we will take into account all available information, including new information regarding formalized conservation efforts. If we receive new information on a formalized conservation effort that has not yet been implemented or not yet demonstrated effectiveness prior to making a listing decision, we will evaluate the conservation effort in the context of the PECE criteria. If we receive new information on such an effort after we have decided to list a species, then we will consider this new information along with other measures that reduce threats to the species and may use this information in downlisting the species from endangered to threatened status or delisting. However, PECE will not control our analysis of the downlisting of the species.

Issue 6: One commenter stated that it is unrealistic and unreasonable to expect agreements to be in place at the time the conservation effort is evaluated. In addition, the commenter stated that it is particularly unrealistic and unreasonable to expect that conservation agreements or plans be submitted within 60 days of publication of a proposed rule.

Response 6: We strongly encourage parties to initiate formalized conservation efforts prior to publication of a proposal to list a species under the Act. If a formalized conservation effort is submitted during the public comment period for a proposed rule, and may be significant to the listing decision, then we may extend or reopen the comment period to allow time for comment on the new conservation effort. However, we can extend the public comment period only if doing so does not prevent us from completing the final listing action within the statutory timeframe.

Issue 7: One commenter stated that most existing conservation agreements are ineffective, and furthermore that we are unable to determine their effectiveness for several years.

Response 7: We agree that it could take several years for some conservation efforts to demonstrate results. However, the PECE criteria provide the framework for us to evaluate the likely effectiveness of such formalized conservation efforts. Some existing conservation efforts have proven to be very effective and have justifiably influenced our listing decisions.

Issue 8: Several commenters stated that funds are better spent to list species, designate critical habitat, and implement recovery efforts rather than to develop conservation agreements.

Response 8: Conservation agreements can be seen as early recovery efforts. Early conservation efforts to improve the status of a species before listing is necessary may cost less than if the species' status has already been reduced to the point where it needs to be listed. Early conservation of candidate species can reduce threats and stabilize or increase populations sufficiently to allow us to use our resources for species in greater need of the Act's protective measures.

Issue 9: Some commenters questioned the 14 conservation agreements that we cited which contributed to making listing the covered species as threatened or endangered unnecessary. Commenters requested information on each plan to better allow the public to evaluate the adequacy of the agreements.

Response 9: We referenced the 14 conservation agreements in the Paperwork Reduction Act section of the draft policy and used them solely to estimate the information collection and recordkeeping burden that would result from our draft policy if it were made final. Therefore, we do not recommend using these to comment on the new policy.

Biological Issues

Issue 10: One commenter questioned our method for evaluating a conservation plan that addresses only a portion of a species' range.

Response 10: Using the PECE criteria, we will evaluate all formalized conservation efforts that have yet to be implemented or have yet to demonstrate results at the time we make our listing decision. This is true for efforts that are applicable to all or only a portion of the species' range. The PECE does not set standards for how much conservation is needed to make listing unnecessary. The significance of plans that address only a portion of a species' range will be evaluated in the context of the species' overall status. While a formalized conservation effort may be effective in reducing or removing threats in a portion of the species' range, that may or may not be sufficient to remove the need to list the species as threatened or endangered. In some cases, the conservation effort may lead to a determination that a species warrants threatened status rather than endangered.

In addition, parties may have entered into agreements to obtain assurances that no additional commitments or restrictions will be required if the species is listed. A landowner or other non-Federal entity can enter into a Candidate Conservation Agreement with Assurances (CCAA) (64 FR 32726, June 17, 1999), which are formal agreements between us and one or more non-Federal parties that address the conservation needs of proposed or candidate species, or species likely to become candidates. These agreements provide assurances to non-Federal property owners who voluntarily agree to manage their lands or waters to remove threats to candidate or proposed species, or to species likely to become candidates. The assurances are authorized under the CCAA regulations (50 CFR 17. 22(d)(5) and 17.32(d)(5)) and provide non-Federal property owners assurances that their conservation efforts will not result in future regulatory obligations in excess of those they agree to at the time they enter into the Agreement. Should the species eventually be listed under the Act, landowners will not be subjected to increased property use restrictions as long as they conform to the terms of the agreement. While one of these agreements may not remove the need to list, several such agreements, covering a large portion of the species' range, may.

Issue 11: Several commenters suggested that the Services should consider conservation efforts developed for species other than the species for which a listing decision is being made when the species have similar biological requirements and the conservation effort addresses protection of habitat of the species for which a listing decision is being made.

Response 11: We agree. When a decision whether or not to list a species is being made, we will consider all conservation efforts that reduce or remove threats to the species under review, including conservation efforts developed for other species. However, for all formalized conservation efforts that have not yet been implemented or have yet to demonstrate results, we will use the PECE criteria to evaluate the conservation effort for certainty of implementation and effectiveness for the species subject to the listing decision.

Issue 12: One commenter stated the "biology/natural history" of the species should be adequately known and explained in order to evaluate the effectiveness of the effort.

Response 12: When we consider the elements under the effectiveness criterion, we will evaluate whether the formalized conservation effort incorporates the best available information on the species' biology and natural history. However, due to variation in the amount of information available about different species and the threats to their existence, the level of information necessary to provide a high level of certainty that the effort will be effective will vary.

We believe it is important, however, to start conservation efforts as early as possible even if complete biological information is lacking. Regardless of the extent of biological information we have about a species, there will almost always be some uncertainty about threats and the most effective mechanisms for improving the status of a species. We will include the extent of gaps in the available information in our evaluation of the level of certainty that the formalized conservation effort will be effective. One method of addressing uncertainty and accommodating new information is the use of monitoring and the application of adaptive management principles. The PECE criteria note that describing the threats and how those threats will be removed, including the use of monitoring and adaptive management principles, as appropriate, is critical to determining that a conservation effort that has yet to demonstrate results has reduced or removed a particular threat to a species.

Issue 13: Several commenters suggested that affected party(ies) should work with the Services to identify species that will be proposed for listing in the near future to help concentrate and direct efforts to those species that most warrant the protection, and help make the party(ies) aware of when and what actions should be taken to help conserve species in need.

Response 13: We do identify species in need of protection. The FWS publishes a Candidate Notice of Review (CNOR) in which the FWS identifies those species of plants and animals for which they have sufficient information on the species' biological status and threats to propose them as endangered or threatened under the Act, but for which development of a proposed listing regulation is precluded by other higher priority listing activities. NMFS, which has jurisdiction over marine species and some anadromous species, defines candidate species more broadly to include species whose status is of concern but more information is needed before they can be proposed for listing. NMFS candidate species can be found on their web site at *http://www.nmfs.noaa.gov*. The FWS's CNOR is published in the **Federal Register** and can also be found on their web site at *http://endangered.fws.gov.*

We agree that it is important to start developing and implementing conservation efforts and coordinating those efforts with us as early as possible. Early conservation helps preserve management options, minimizes the cost of reducing threats to a species, and reduces the potential for land use restrictions in the future. Addressing the needs of species before the regulatory protections associated with listing under the Act come into play often allows greater management flexibility in the actions necessary to stabilize or restore these species and their habitats. Early implementation of conservation efforts may reduce the risk of extinction for some species, thus eliminating the need for them to be listed as threatened or endangered.

Issue 14: One commenter stated that requiring an implementation schedule/timeline for conservation objectives is not feasible when baseline data on a species is poorly understood. The policy should recognize that variation in patterns of species distribution and land ownership will cause variation in the difficulty of developing conservation efforts. Thus, some conservation efforts should be allotted more time for their completion.

Response 14: Biological uncertainty is a common feature of any conservation effort. Nevertheless, some conservation actions can proceed even when information on the species is incomplete. Implementation schedules are an important element of all formalized conservation planning efforts (e.g., recovery plans). The implementation schedule identified in PECE criterion A.8. establishes a timeframe with incremental completion dates for specific tasks. In light of the information gaps that may exist for some species or actions, schedules for completing certain tasks may require revision in response to new information, changing circumstances, and the application of adaptive management principles. Including an implementation schedule in a formalized conservation effort is critical to determining that the effort will be implemented and effective and has improved the status of the species under the Act at the time we make our listing determination.

We acknowledge that the amount of time required to develop and implement formalized conservation efforts will vary. Therefore, we encourage early development and implementation of conservation efforts for species that have not yet become candidates for listing and for those species that are already candidates. This policy does not dictate timeframes for completing conservation efforts. However, the Act mandates specific timeframes for many listing decisions, and we cannot delay final listing actions to allow for the development and signing of a conservation agreement or plan. We and participants must also acknowledge that, for species that are poorly known, or whose threats are not well understood, it is unlikely that conservation efforts that have not been implemented or that have yet to yield results will have improved the status of the species sufficiently to play a significant role in the listing decision.

Issue 15: One commenter stated that the Services, when evaluating the certainty of conservation efforts while making listing decisions, should factor into the analysis the Services' ability to open or reopen the listing process at any time, and to list the species on an emergency basis if necessary.

Response 15: We will initiate or revisit a listing decision if information indicates that doing so is warranted, and on an emergency basis if there is an imminent threat to the species' well-being. However, we do not make any listing determinations based on our ability to change our decisions. We base our listing decisions on the status of the species at that time, not on some time in the future.

Criteria Issues

Issue 16: Several commenters requested that we further explain the criteria for both implementation and effectiveness. The commenters claim that our criteria are too vague and are subject to interpretation by the Services. One commenter said that, by stating ''this list should not be considered comprehensive evaluation criteria,'' the policy allows the Services to consider criteria not addressed in the agreement, and allows for too much leeway for the Services to reject conservation efforts of an agreement, even if all criteria listed in the draft policy are satisfied.

Response 16: PECE establishes a set of criteria for us to consider when evaluating formalized conservation efforts that have not yet been implemented or have not yet demonstrated effectiveness to determine if the efforts have improved the status of the species. At the time of the listing decision, we must find, with minimal uncertainty, that a particular formalized conservation effort will be implemented and will be effective, in order to find that the effort has positively affected the conservation status of a species. Meeting these criteria does not create an approval process. Some conservation efforts will address these criteria more thoroughly than others. Because, in part, circumstances vary greatly among species, we must evaluate all conservation efforts on a case-by-case basis at the time of listing, taking into account any and all factors relevant to whether the conservation effort will be implemented and effective.

Similarly, the list of criteria is not comprehensive because the conservation needs of species will vary greatly and depend on species-specific, habitat-specific, location-specific, and action-specific factors. Because conservation needs vary, it is not possible to state all of the factors that might determine the ultimate effectiveness of formalized conservation efforts. The species-specific circumstances will also determine the amount of information necessary to satisfy these criteria. Evaluating the certainty of the effectiveness of a formalized conservation effort necessarily includes an evaluation of the technical adequacy of the effort. For example, the effectiveness of creating a wetland for species conservation will depend on soil texture, hydrology, water chemistry, and other factors. Listing all of the factors that we would appropriately consider in evaluations of technical adequacy is not possible.

Issue 17: One commenter suggested that we consider conservation plans in the development stage rather than waiting until finalized due to the possible benefits that may result from initial efforts.

Response 17: Plans that have not been finalized and, therefore, do not conform to the PECE criteria, may have some conservation value for the species. For example, in the process of developing a plan, participants and the public may become more informed about the species and its conservation needs. We will consider any benefits to a species that have accrued prior to the completion of an agreement or plan in our listing decision, under section 4(b)(1)(A) of the Act. However, the mere existence of a planning process does not provide sufficient certainty to actually improve the status of a species. The criteria of PECE set a rigorous standard for analysis and assure a high level of certainty associated with formalized conservation efforts that have not been implemented, or have yet to yield results, in order to determine that the status of the species has improved.

We encourage parties to involve the appropriate Service during the development stage of all conservation plans, whether or not they are finalized prior to a listing decision. Sharing of the best available information can lead to developing better agreements. In the event that the focus species is listed, these planning efforts can be utilized as the basis for development of Safe Harbor Agreements or Habitat Conservation Plans, through which we can permit incidental take under Section 10(a) of the Act, or provide a basis for a recovery plan.

Issue 18: Several commenters stated that the policy should provide more sufficient, clear criteria by which the implementation and effectiveness of conservation efforts is monitored and assessed. One commenter also suggested that we require a specific reporting format to help show effectiveness of conservation efforts.

Response 18: When evaluating formalized conservation efforts under PECE, we will consider whether the effort contains provisions for monitoring and reporting implementation and effectiveness results (see criterion B.5).

Regarding a standard reporting format, the nature of the formalized conservation efforts we evaluate will probably vary a great deal. Efforts may range from complex to single-threat approaches. Therefore, for us to adopt a one-size-fits-all approach to report on monitoring efforts and results would be inappropriate.

Issue 19: One commenter stated that PECE is too demanding with respect to identification and commitment of resources ''up-front,'' and that these strict requirements and commitments on conservation efforts harm the voluntary nature of agreements.

Response 19: Addressing the resources necessary to carry out a conservation effort is central to establishing certainty of plan implementation and effectiveness. Accordingly, we believe that PECE must establish a minimum standard to assure certainty of implementation and effectiveness. This certainty is necessary in determining whether the conservation effort has improved the status of species.

It is our intention and belief that the PECE criteria will actually increase the voluntary participation in conservation agreements by increasing the likelihood that parties' voluntary efforts and commitments that have yet to be implemented or have yet to demonstrate results will play a role in a listing decision.

Issues Related to Specific Changes

Several commenters recommended specific changes to the evaluation criteria. The recommended additions in language to the criteria are italicized and deletions are shown in strikeout to help the reader identify the proposed changes.

Issue 20: Commenters stated that there is potential confusion between evaluation criteria A.2. (authority) and A.3.(authorization) as they believed some Service staff may have difficulty distinguishing between an ''authority,'' and an ''authorization.'' To help eliminate this potential confusion, commenters requested that criterion A.2. be changed to read: ''the legal authority of the party(ies) to the agreement or plan to implement the conservation effort and the legal procedural requirements necessary to implement the effort are described.'' They also requested that we change criterion A.3. to read: The legal requirements (e.g. permits, environmental review documents) necessary to implement the conservation effort are identified, and an explanation of how the party(ies) to the agreement or plan that will implement the effort will fulfill these requirements is provided.''

Response 20: We agree with adding the word ''legal'' and also have incorporated additional language and separated this criterion (former criterion A.2) into two criteria (A.2. and A.3.). Evaluation Criterion A.2. now reads, ''The legal authority of the party(ies) to the agreement or plan to implement the formalized conservation effort, and the commitment to proceed with the conservation effort are described.'' New evaluation Criterion A.3. reads, ''The legal procedural requirements necessary to implement the effort are described, and information is provided indicating that fulfillment of these requirements does not preclude commitment to the effort.'' In making these changes, we recognize that there may be overlap between new criterion A.3. and the criterion on authorizations (now A.4.), but our intent is to separate a criterion on procedural requirements from substantive authorizations (e.g. permits). We believe that we need to specifically determine that the parties to the agreement will obtain the necessary authorizations. We also recognize that parties may not be able to commit to some conservation efforts until they have fulfilled procedural requirements (e.g. under the National Environmental Policy Act) since some laws preclude commitment to a specific action until certain procedures are completed. Additionally, in creating a new criterion A.3., we find it unnecessary to incorporate the suggested changes to old A.3. (now A.4.).

Issue 21: Commenters requested the following change to Criterion A.4. (now Criterion A.5.): ''The level of voluntary participation (e.g., permission to enter private land or other contributions by private landowners) necessary to implement the conservation effort is identified, and an explanation of how the party(ies) to the agreement or plan that will implement the conservation effort will obtain that level of voluntary participation is provided (e.g., an explanation of why incentives to be provided are expected to result in the necessary level of voluntary participation)''.

Response 21: We do not believe that including ''an explanation of how the party(ies) * * * will obtain that level of voluntary participation * * *'' will provide us with enough information in order to determine that necessary voluntary participation will, in fact, be obtained. Evaluation Criterion A.5. (formerly A.4.) now reads: ''The type and level of voluntary participation (e.g., number of landowners allowing entry to their land, or number of participants agreeing to change timber management practices and acreage involved) necessary to implement the conservation effort is identified, and a high level of certainty is provided that the party(ies) to the agreement or plan that will implement the conservation effort will obtain that level of voluntary participation (e.g., an explanation of how incentives to be provided will result in the necessary level of voluntary participation).''

Issue 22: Commenters suggested that Evaluation Criterion A.5. (now criterion A.6.) be changed to read as ''Any statutory or regulatory deficiency or barrier to implementation of the conservation effort is identified and an explanation of how the party(ies) to the agreement or plan that will implement the effort will resolve the deficiency or barriers is provided.''

Response 22: We do not agree with the suggested language change. We believe that all regulatory mechanisms, including statutory authorities, must be in place to ensure a high level of certainty that the conservation effort will be implemented.

Issue 23: The suggested change to Evaluation Criterion A.6. (now A.7.) is ''A fiscal schedule and plan is provided for the conservation effort, including a description of the obligations of party(ies) to the agreement or plan that will implement the conservation effort, and an explanation of how they will obtain the necessary funding is provided.''

Response 23: We do not agree with the suggested language change since we believe that there must be a high level of certainty that the party(ies) will obtain the necessary funding to implement the effort. While we agree that including a fiscal schedule, a description of the obligations of the party(ies), and an explanation of how they will obtain the funding is important, this information, by itself, does not provide enough certainty for us to consider a formalized conservation effort that has not yet been implemented as contributing to a listing decision. Also see our response to Issue 41.

Issue 24: One commenter suggested that the Services should consider an incremental approach to evaluating implementation dates for the conservation effort.

Response 24: We agree with the commenter's suggested change. Evaluation Criterion A.8. (formerly A.7.) now reads as: ''An implementation schedule (including incremental completion dates) for the conservation effort is provided.''

Issue 25: Commenters suggested that Criterion A.8. (now A.9.) be revised to read: ''The conservation agreement or plan that includes the conservation effort include a commitment by the party(ies) to apply their legal authorities and available resources as provided in the agreement or plan.''

Response 25: The participation of the parties through a written agreement or plan establishes each party's commitment to apply their authorities and resources to implementation of each conservation effort. Therefore, it is unnecessary to include the suggested language; criterion A.9. (formerly A.8.) remains unchanged.

Issue 26: A commenter also suggested adding a criterion: ''Evidence that other conservation efforts have been implemented for sympatric species within the same ecosystem that may provide benefits to the subject species is provided.''

Response 26: We do not think it is necessary to add such a criterion. At the time of listing, we will take into consideration all relevant information, including the effect of other conservation efforts for sympatric species on the status of the species we are considering for listing.

Issue 27: Several commenters recommended that we make specific changes to the Criterion B.1. language to read as: ''The nature and extent of threats being addressed by the conservation effort are described, and how the conservation effort will reduce the threats are defined.'' In addition, commenters suggested we change Criterion B.2. to read as: ''Explicit incremental objectives for the conservation effort and dates for achieving them should be stated.''

Response 27: We agree that, in addition to identifying threats, the plan should explain how formalized conservation efforts reduce threats to the species. Therefore, Evaluation Criterion B.1. now reads as: ''The nature and extent of threats being addressed by the conservation effort are described, and how the conservation effort reduces the threats is described.'' We agree that conservation efforts should include incremental objectives. This allows the parties to evaluate progress toward the overall goal of a conservation effort, which is essential for adaptive management. In addition, setting and achieving interim objectives is helpful in maintaining support for the effort. Therefore, Evaluation Criterion B.2. now reads as: ''Explicit incremental objectives for the conservation effort and dates for achieving them are stated.''

Issue 28: Some commenters recommended that the party's (ies') prior record with respect to development and implementation of conservation efforts be recognized towards their credibility and reliability to implement future conservation efforts. A commenter also suggested adding a criterion to read as: ''Demonstrated ability of the party(ies) to develop and implement effective conservation efforts for this or other species and habitats.'' Another comment suggested that the history and momentum of a program should be taken into account (e.g., watershed council programs) when considering the certainty of effectiveness and implementation. These considerations would help ensure a high level of certainty that regulatory mechanisms, funding authorizations, and voluntary participation will be adopted by a specified date adequate to provide certainty of implementation.

Response 28: Although it would be beneficial for the party(ies) to demonstrate their past abilities to implement effective formalized conservation efforts for the focus species or other species and habitats, we do not believe that this is necessary to demonstrate a high level of certainty that the conservation effort will be implemented. In addition, a criterion that emphasizes previous experience in implementing conservation efforts may limit formalized conservation efforts to only those party(ies) that have a track record and would unjustifiably constrain consideration of efforts by those who do not satisfy this criterion. Such parties can provide certainty in other ways. We agree that a party's (ies') prior record and history with respect to implementation of conservation efforts should be recognized towards their credibility and reliability. Information concerning a party's experience in implementing conservation efforts may be useful in evaluating how their conservation effort satisfies the PECE criteria. The momentum of a project is a good indication of the progress that is being made towards a party's (ies') conservation efforts, but momentum can decrease, and thus cannot be solely relied upon to determine the certainty that a formalized conservation effort will be implemented or effective.

Issue 29: One commenter stated that our use of ''must'' in meeting the criteria is inappropriate in the context of a policy, and the policy should rather be treated as guidance.

Response 29: The only mandatory statements in the policy refer to findings that we must make. In order for us to find that a particular formalized conservation effort has improved the status of the species, we must be certain that the formalized conservation effort will be implemented and will be effective. No party is required to take any action under this policy. Rather the policy provides us guidance on how we will evaluate formalized conservation efforts that have yet to be implemented or have yet to demonstrate effectiveness at the time of our listing decision.

Legal Issues

Issue 30: Many commenters mentioned past litigation (i.e., decisions on coho salmon and Barton Springs salamander) in which the courts have ruled against the Services in cases that have involved Candidate Conservation Agreements or other conservation efforts, and question how the PECE policy addresses this issue. Commenters question how this policy will keep the Services from relying on speculative conservation efforts.

Response 30: We referenced past adverse decisions when we published the draft policy. The purpose of PECE, in part, is to address situations similar to those in which some courts found past conservation efforts insufficient. We developed the PECE to establish a set of consistent standards for evaluating certain formalized conservation efforts at the time of a listing decision and to ensure with a high level of certainty that formalized conservation efforts will be implemented and effective. We agree that we may not rely on speculative promises of future action when making listing decisions.

Issue 31: Several commenters questioned the legality of considering private party's (ies') input when section 4(b)(1)(A) of the Act states ''* * * and after taking into account those efforts, if any, being made by any State or foreign nation, or any political subdivision of a State or foreign nation, to protect such species * * *'' In addition, commenters stated that the PECE policy is inconsistent with the plain language and the congressional intent of the Act by allowing agencies to evaluate any private measures. They also stated that this was inconsistent with considering section 4(a)(1)(D), which only permits agencies to evaluate ''existing regulatory mechanisms.'' They also stated that the Services incorrectly conclude that section 4(a)(1)(E), ''other natural or manmade factors affecting [the species'] continued existence,'' allows the Services to consider actions of ''any other entity'' in making listing determinations. One commenter stated that there are no provisions to authorize the Services to consider voluntary conservation agreements by other Federal agencies. In 1982, the Act omitted 1973 language for listing determinations made with ''other interested Federal agencies.'' In addition, the commenters stated that the Act imposes conservation duties on all Federal agencies only after the Services have taken the initial step in listing the species.

Response 31: Please refer to the Policy Scope section for an explanation of our authority under section 4 of the Act to assess all threats affecting the species status as well as all efforts that reduce threats to the species.

Issue 32: One commenter suggested that we formalize this policy by codifying it in the Code of Federal Regulations. They suggest that by adopting this policy as agency regulation, we can make the policy more binding, provide a basis for judicial deference, and thus hopefully reduce the amount of litigation.

Response 32: We believe that codifying PECE in the Code of Federal Regulations is not necessary because it is intended as a policy to guide how we will evaluate formalized conservation efforts when making listing decisions.

Issue 33: Some commenters believe that all regulatory mechanisms must be in place prior to finalizing a conservation plan, while other commenters feel that this requirement may dissuade voluntary conservation efforts of private landowners. One commenter stated that, based on the amount of time usually needed to enact most regulatory mechanisms, it seems appropriate to set this minimum standard for evaluating formalized conservation efforts. This criterion should prompt more serious political consideration of adopting a regulatory mechanism sooner rather than later. Another commenter suggested that, instead of requiring regulations, we should require cooperators to identify and address any regulatory deficiencies affecting the species.

Response 33: In order for us to determine with a high level of certainty that a formalized conservation effort will be implemented, among other things, all regulatory mechanisms necessary to implement the effort must be in place at the time we make our listing decision. However, there may be situations where regulatory mechanisms are not necessary for implementing the conservation effort due to the nature of the action that removes threats, or there may be situations where necessary regulatory mechanisms are already in place.

Issue 34: One commenter stated that only when an alternative regulatory mechanism provides the same or higher protections than listing can the threat factors be said to be alleviated. A high level of certainty over future funding or voluntary participation might be acceptable if alternative regulatory mechanisms to prevent take in the interim are in place.

Response 34: Determinations to list species under the Act are based solely on whether or not they meet the definitions of threatened or endangered as specified by the Act. Through PECE, we will evaluate, at the time of our listing decision, whether a formalized conservation effort adequately reduces threats and improves the status of the species to make listing unnecessary. Additional alternative regulatory mechanisms to prevent take are not necessary if the threats to the species are reduced to the point that the species does not meet the definitions of threatened or endangered.

Issue 35: One commenter stated concern that the Services would not be able to provide assurances to private landowners because no specific provisions in the Act authorize conservation agreements in lieu of listing, and that third party lawsuits also undermine the Services' assurances. One commenter asked what future protection of their ongoing actions participants would receive.

Response 35: Satisfying the PECE criteria does not provide assurances that we will not decide to list a species. Also, because of the individual nature of species and the circumstances of their status, PECE does not address how much conservation is required to make listing unnecessary. Because of the numerous factors that affect a species' status, we may list a species despite the fact that one or more formalized conservation efforts have satisfied PECE. However, assurances can be provided to non-Federal entities through an approved Candidate Conservation Agreement with Assurances (CCAA) and in an associated enhancement of survival permit issued under section 10(a)(1)(A) of the Act. Many property owners desire certainty with regard to future regulatory restrictions to guarantee continuation of existing land or water uses or to assure allowance for future changes in land use. By facilitating this kind of individual land use planning, assurances provided under the CCAA policy can substantially benefit many property owners. These agreements can have significance in our listing decisions, and we may also evaluate them according to the criteria in the PECE if they are not yet implemented or have not demonstrated results. However, we will make the determination of whether these CCAAs preclude or remove any need to list the covered species on a case-by-case basis in accordance with the listing criteria and procedures under section 4 of the Act.

Issue 36: Several commenters stated that the PECE does not always provide incentives to conserve species and is, therefore, not supported by the Congressional finding of section 2(a)(5) of the Act. The commenters stated that the parties lack incentives to develop conservation programs until after the species is listed (e.g., *Building Industry Association of Southern California* v. *Babbitt*, where listing the coastal California gnatcatcher encouraged enrollment in conservation programs.) In addition, they stated that PECE provides a means for the listing process to be avoided entirely, and, therefore, may often fail to provide incentives that Congress referred to in its findings in section 2(a)(5). They stated that the ''system'' of incentives to which that Congressional finding refers is already found in incidental take provisions in section 10 of the Act, which will better ensure development and implementation of successful conservation programs.

Response 36: PECE is not ''a way to avoid listing'' or an ''in lieu of listing'' policy. This policy outlines guidance on the criteria we will use to evaluate formalized conservation efforts in determining whether to list a species. Knowing how we will evaluate any unimplemented or unmeasured formalized conservation efforts may help parties draft more effective agreements. However, there is a conservation incentive because, if a species becomes listed, these efforts can contribute to recovery and eventual delisting or downlisting of the species. Also, see our response to Issue 35.

Issue 37: Several commenters stated that relying on unimplemented future conservation measures is inconsistent with the definitions of ''threatened species'' and ''endangered species'' as provided in section 3 of the Act, and that PECE's evaluation of future, unimplemented conservation efforts in listing determinations is inconsistent with both the plain language of the Act and Congressional intent. Also, the commenters stated that the PECE erroneously claims that the definitions of ''threatened species'' and ''endangered species'' connote future status, not present status.

Response 37: We agree that, when we make a listing decision, we must determine the species' present status which includes, in part, an evaluation of current threats. However, deciding or determining whether a species meets the definition of threatened or endangered also requires us to make a prediction about the future persistence of a species. Central to this concept is a prediction of future conditions, including consideration of future negative effects of anticipated human actions. The language of the Act supports this approach. The definitions for both ''endangered species'' and ''threatened species'' connote future condition, which indicates that consideration of whether a species should be listed depends in part on identification and evaluation of future actions that will reduce or remove, as well as create or exacerbate, threats to the species. We cannot protect species without taking into account future threats to a species. The Act does not require that, and species conservation would be compromised if, we wait until a threat is actually impacting populations before we list the species as threatened or endangered. Similarly, the magnitude and/or imminence of a threat may be reduced as a result of future positive human actions. Common to the consideration of both the negative and positive effects of future human actions is a determination of the likelihood that the actions will occur and that their effects on the species will be realized. Therefore, we consider both future negative and future positive impacts when assessing the listing status of the species. The first factor in section 4(a)(1)—''the present or threatened destruction, modification, or curtailment of [the species'] habitat or range''—identifies how analysis of both current actions affecting a species' habitat or range and those actions that are sufficiently certain to occur in the future and affect a species' habitat or range are necessary to assess a species' status. However, future Federal, state, local, or private actions that affect a species are not limited to actions that will affect a species' habitat or range. Congress did not intend for us to consider future actions affecting a species' habitat or range, yet ignore future actions that will influence overutilization, disease, predation, regulatory mechanisms, or other natural or manmade factors. Therefore, we construe Congress' intent, as reflected by the language of the Act, to require us to consider both current actions that affect a species' status and sufficiently certain future actions—either positive or negative—that affect a species' status.

Issue 38: Several commenters stated that PECE's ''sufficient certainty'' standard is inconsistent with the Act's ''best available science'' standard. They stated that courts have ruled that any standard other than ''best available science'' violates the plain language and the Congressional intent of the Act. The commenters also stated that the ''sufficient certainty'' standard violates Congressional intent because it weakens the standard required by the Act to list species and can result in unnecessary, and potentially harmful, postponement of affirmative listing.

Response 38: We agree that our listing decisions must be based on the best available science. PECE does not address or change the listing criteria and procedures established under section 4 of the Act. Listing analyses include the evaluation of conservation efforts for the species under consideration. PECE is designed to help ensure a consistent and rigorous review of formalized conservation efforts that have yet to be implemented or efforts that have been implemented but have not yet shown effectiveness by establishing a set of standards to evaluate the certainty of implementation and effectiveness of these efforts.

Issue 39: Several commenters stated that PECE reduces or eliminates public comment on proposed rules to list species and is in violation of the Administrative Procedure Act (APA). Further, they stated that PECE violates the APA by allowing submission of formalized conservation measures after the proposed rule is issued to list species as threatened or endangered. Receiving ''conservation agreements or plans before the end of the comment period in order to be considered in final listing decision'' encourages landowners to submit conservation agreements at the last minute to avoid public scrutiny, and the PECE process could be a potential delay tactic used by landowners to postpone the listing of species. They stated that the Courts agree that failure of the Services to make available to the public conservation agreements on which listing decisions are based violates the public comment provision of the APA.

Response 39: All listing decisions, including those involving formalized conservation agreements, will comply with the requirements of the APA and ESA. If we receive a formalized conservation agreement or plan during an open comment period and it presents significant new information relevant to the listing decision, we would either extend or reopen the public comment period to solicit public comments specifically addressing that plan or agreement. We recognize, however, that there may be situations where APA requirements must be reconciled with the ESA's statutory deadlines.

Issue 40: Several commenters expressed their concern that conservation efforts do not have binding obligations.

Response 40: While PECE does not require participants to have binding obligations, the policy does require a high level of certainty that a conservation effort will be implemented and effective at the time we make our listing decision. Furthermore, any subsequent failure to satisfy one or more PECE criteria would constitute new information and, depending on the significance of the formalized conservation effort to the species' status, may require a reevaluation of whether there is an increased risk of extinction, and whether that increased risk indicates that the species' status is threatened or endangered.

Funding Issues

Issue 41: Several commenters requested that we further specify our criteria stating that ''a high level of certainty that the party(ies) to the agreement or plan that will implement the conservation effort will obtain the necessary funding is provided.'' In addition, one commenter questioned whether ''a high level of certainty'' for authorizations or funding was really an improvement over the status quo and suggested that we either list the required elements we will use to evaluate completeness of the conservation efforts or quantitatively define an evaluation standard.

Response 41: A high level of certainty of funding does not mean that funding must be in place now for implementation of the entire plan, but rather, it means that we must have convincing information that funding will be provided each year to implement relevant conservation efforts. We believe that at least 1 year of funding should be assured, and we should have documentation that demonstrates a commitment to obtain future funding, e.g., documentation showing funding for the first year is in place and a written commitment from the senior official of a state agency or organization to request or provide necessary funding in subsequent budget cycles, or documentation showing that funds are available through appropriations to existing programs and the implementation of this plan is a priority for these programs. A fiscal schedule or plan showing clear links to the implementation schedule should be provided, as well as an explanation of how the party(ies) will obtain future necessary funding. It is also beneficial for entities to demonstrate that similar funding was requested and obtained in the past since this funding history can show the likelihood that future funding will be obtained.

Issue 42: One commenter suggested that the PECE policy holds qualifying conservation efforts to a higher standard than recovery plans. The commenter quoted several existing recovery plans that included disclaimers about budget commitments associated with specific tasks. Therefore, the commenter concluded that it is unrealistic and unreasonable to mandate that funding be in place when a conservation effort is evaluated.

Response 42: The Act does not require that certainty of implementation be provided for recovery management actions for listed species or conservation efforts for nonlisted species. Likewise, the PECE does not require that certainty of implementation be provided for during development of conservation efforts for nonlisted species. It is inappropriate to consider the PECE as holding conservation plans or agreements to a higher standard than the standard that exists for recovery plans because the PECE does not mandate a standard for conservation plans or agreements at the time of plan development. Rather, the PECE provides us guidance for the evaluation of conservation efforts when making a listing decision for a nonlisted species.

Recovery plans for listed species and conservation plans or agreements for nonlisted species identify needed conservation actions but may or may not provide certainty that the actions will be implemented or effective. However, when making a listing decision for nonlisted species, we must consider the certainty that a conservation effort will be implemented and effective. The PECE establishes criteria for us to use in evaluating conservation efforts when making listing decisions.

It is possible that we would evaluate a management action identified in a recovery plan for a listed species using the PECE. If, for example, a yet-to-be-implemented task identified in a recovery plan for a listed species would also benefit a nonlisted species, we, in making a listing decision for the nonlisted species, would apply the PECE criteria to that task to determine whether it could be considered as contributing to a decision not to list the species or to list the species as threatened rather than endangered. In this situation, we would evaluate the management task identified in a recovery plan using the PECE criteria in the same way as other conservation efforts for the nonlisted species. That is, the recovery plan task would be held to the same evaluation standard in the listing decision as other conservation efforts.

Foreign Species Issues

Issue 43: One commenter asked why the proposed policy excluded conservation efforts by foreign governments, even though section 4(b)(1)(A) of the Act requires the Services to take such efforts into account. This commenter also stated that the proposed policy is contrary to ''The Foreign Relations Law of the United States,'' which he argues requires the United States to defer to other nations when they have a ''clearly greater interest'' regarding policies or regulations being considered by the United States that could negatively affect their nations.

Response 43: As required by the Act, we have taken and will continue to take into account conservation efforts by foreign countries when considering listing of foreign species (sections 4(b) and 8 of the Act). Furthermore, whenever a species whose range occurs at least in part outside of the United States is proposed for a listing action (listing, change in status, or delisting), we communicate with and solicit the input of the countries within the range of the species. At that time, countries are provided the opportunity to share information on the status of the species, management of the species, and on conservation efforts within the foreign country. We will take those comments and information provided into consideration when evaluating the listing action, which by law must follow the analysis outlined in sections 4(a) and 4(b) of the Act. Thus, all listing decisions for foreign species will continue to comply with the provisions of the Act.

Issues Outside Scope of Policy

We received several comments that were outside of the scope of PECE. Below, we have briefly addressed these comments.

Issue 44: A comment was made that the Services should not list foreign species under the Act when such listing is in conflict with the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES).

Response 44: Considerations regarding CITES are outside the scope of the PECE. However, we do not believe there is a conflict with CITES and listing of a foreign species under the Act. When evaluating the status of foreign species under the Act, we take into consideration whether the species is listed under CITES (and if listed, at what level) and all available information regarding the listing. If you have questions regarding CITES, please contact the FWS Division of Scientific Authority at 4401 N. Fairfax Drive, Room 750, Arlington, VA 22203 or by telephone at 703–358–1708.

Issue 45: One commenter stated that all conservation agreements/plans should be subject to independent scientific peer review. This commenter also argued that any conservation agreement or plan for a candidate species should remove all known major threats for the species and convey a reasonably high certainty that the agreement or plan will result in full conservation of the species.

Response 45: We believe that scientific review can help ensure that formalized conservation efforts are comprehensive and effective, and we expect that most or all participants will seek scientific review, but we will not require a formal independent peer review of conservation plans at the time of development. If a formalized conservation plan is presented for a species that has been proposed for listing, all relevant information, including formalized conservation efforts, will be subject to independent scientific review consistent with our policy on peer review (59 FR 34270). We will also solicit public comments on our listing proposals.

The amount or level of conservation proposed in a conservation plan (e.g., removal of all versus some of the major threats) is outside the scope of PECE. Assuming that all of the PECE criteria have been satisfied for the efforts to which they apply, it stands to reason that plans that comprehensively address threats are likely to be more influential in listing decisions than plans that do not thoroughly address the conservation of the species. We believe that by establishing the PECE criteria for certainty of implementation and effectiveness, we are promoting the development of plans that improve the status of species. We expect that in some cases this improvement will reduce the risk of extinction sufficiently to make listing under the Act unnecessary, to result in listing a species as threatened rather than endangered, or to make classifying a species as a candidate for listing unnecessary.

Issue 46: Several commenters questioned the extent of state involvement in the development of conservation efforts. One commenter said that the policy should mandate that States be involved with plan development, and that states approve all conservation efforts.

Response 46: It is outside the scope of PECE to establish standards to determine who participates in the development of conservation efforts and at what level. In many cases, states play a crucial role in the conservation of species. For formalized conservation efforts to be effective, it is logical for the states to play an integral role. To that end, we highly encourage state participation to help ensure the conservation of the species, but we do not believe that states should be mandated to participate in the development of all conservation plans. In some cases, states may not have the resources to participate in these plans, and in other situations, individuals or non-state entities may have the ability to develop an effective and well-implemented plan that does not require state participation, but that contributes to the conservation of a species. Through our listing process, we will work with state conservation agencies, and, if the listing decision involves a public comment period, states have a formal opportunity to comment on any conservation efforts being considered in the listing decision.

Issue 47: Several comments were made regarding the feedback mechanisms to correct a party's (ies') inadequate or ineffective implementation of a conservation effort. It was suggested that the Services specify clearly, and based on scientific information, those factors which the Services believe indicate that a conservation effort is either not being implemented or not being effective. Comments also suggested that party(ies) be given reasonable time (e.g., 90–120 days) to respond to the Service's findings by either implementing actions, achieving objectives, or providing information to respond to the Services.

Response 47: PECE is not a regulatory approval process, and establishing a formal feedback mechanism between the Services and participants is not within the scope of PECE. The final determination whether to list a species under the Act will rest solely upon whether or not the species under consideration meets the definition of threatened or endangered as specified by the Act, which will include consideration of whether formalized conservation efforts that meet PECE criteria have enhanced the status of the species. We will provide guidance to improve conservation efforts when possible, but we cannot delay listing decisions in order to participate in a corrective review process when the best scientific and commercial data indicate that a species meets the definition of threatened or endangered.

Issue 48: One commenter requested that we clarify how significant the conservation agreement must be to the species, and describe the anticipated overall impact/importance to the species and the estimated extent of the species' overall range that the habitat conservation agreement might cover.

Response 48: PECE does not establish standards for how much or what kind of conservation is required to make listing a species under the Act unnecessary. We believe that high-quality formalized conservation efforts should explain in detail the impact and significance of the effort on the target species. However, at the time of our listing decision, we will evaluate formalized conservation efforts using PECE to determine whether the effort provides certainty of implementation and effectiveness and improves the status of the species. Through our listing process, we will determine whether or not a species meets the definition of threatened or endangered.

Issue 49: Several commenters wrote that states do not have additional resources to be pro-active on candidate conservation efforts, and suggested that funding for conservation plans or efforts should be provided by the Federal Government.

Response 49: This comment is outside the scope of the PECE. This policy establishes a set of standards for evaluating formalized conservation efforts in our listing decisions and does not address funding sources to develop and implement these efforts.

Summary of Changes From the Proposed Policy

We have slightly revised some of the evaluation criteria as written in the proposed policy. We made the following changes to reflect comments that we received during the public comment period. We added the word ''legal'' to criterion A.2., incorporated additional language (''the commitment to proceed with the conservation effort is described.''), and separated this criterion into two criteria (A.2. and A.3.). We revised criterion A.3. (formerly part of A.2.) to recognize that parties cannot commit to completing some legal procedural requirements (e.g. National Environmental Policy Act) since some procedural requirements preclude commitment to a proposed action before the procedures are actually completed. We changed criterion A.5. (formerly A.4.) by adding ''type'' and ''(e.g., number of landowners allowing entry to their land, or number of participants agreeing to change timber management practices and acreage involved)'' and by replacing ''why'' with ''how'' and ''are expected to'' with ''will.'' We deleted the word ''all'' at the beginning of criterion A.6. as we felt it was redundant. We added ''(including incremental completion dates)'' to criterion A.8. (formerly A.7.). To criterion B.1. we added ''and how the conservation effort reduces the threats is described.''

Also in the proposed policy we stated that if we make a decision not to list a species, or to list the species as threatened rather than endangered, based in part on the contributions of a formalized conservation effort, we will monitor the status of the species. We have clarified this in the final policy to state that we will monitor the status of the effort, including the progress of implementation of the formalized conservation effort.

Required Determinations

Regulatory Planning and Review

In accordance with Executive Order 12866, this document is a significant policy and was reviewed by the Office of Management and Budget (OMB) in accordance with the four criteria discussed below.

(a) This policy will not have an annual economic effect of $100 million or more or adversely affect an economic sector, productivity, jobs, the environment, or other units of government. The policy for the evaluation of conservation efforts when making listing decisions does not pertain to commercial products or activities or anything traded in the marketplace.

(b) This policy is not expected to create inconsistencies with other agencies' actions. FWS and NMFS are responsible for carrying out the Act.

(c) This policy is not expected to significantly affect entitlements, grants, user fees, loan programs, or the rights and obligations of their recipients.

(d) OMB has determined that this policy may raise novel legal or policy issues and, as a result, this action has undergone OMB review.

Regulatory Flexibility Act (5 U.S.C. 601 et seq.)

Under the Regulatory Flexibility Act (5 U.S.C. 601 *et seq.*, as amended by the Small Business Regulatory Enforcement Fairness Act (SBREFA) of 1996), whenever an agency is required to publish a notice of rulemaking for any proposed or final rule, it must prepare and make available for public comment a regulatory flexibility analysis that describes the effect of the rule on small entities (i.e., small businesses, small organizations, and small government jurisdictions), unless the agency certifies that the rule will not have a significant economic impact on a substantial number of small entities.

SBREFA amended the Regulatory Flexibility Act to require Federal agencies to provide the statement of the factual basis for certifying that a rule will not have a significant economic impact on a substantial number of small entities. The following discussion explains our determination.

We have examined this policy's potential effects on small entities as required by the Regulatory Flexibility Act and have determined that this action will not have a significant economic impact on a substantial number of small entities since the policy will not result in any significant additional expenditures by entities that develop formalized conservation efforts. The criteria in this policy describe how we will evaluate elements that are already included in conservation efforts and do not establish any new implementation burdens. Therefore, we believe that no economic effects on States and other entities will result from compliance with the criteria in this policy.

Pursuant to the Regulatory Flexibility Act, at the proposed policy stage, we certified to the Small Business Administration that this policy would not have a significant economic impact on a substantial number of small entities, since we expect that this policy will not result in any significant additional expenditures by entities that develop formalized conservation efforts. We received no comments regarding the economic impacts of this policy on small entities. Thus, we certify that this final policy will not have a significant adverse impact on a substantial number of small entities and conclude that a regulatory flexibility analysis is not necessary.

We have determined that this policy will not cause (a) any effect on the economy of $100 million or more, (b) any increases in costs or prices for consumers; individual industries; Federal, State, or local government agencies; or geographical regions, or (c) any significant adverse effects on competition, employment, investment, productivity, innovation, or the ability of U.S.-based enterprises to compete with foreign-based enterprises (see Economic Analysis below).

Executive Order 13211

On May 18, 2001, the President issued an Executive Order (E.O. 13211) on regulations that significantly affect energy supply, distribution, and use. Executive Order 13211 requires agencies to prepare Statements of Energy Effects when undertaking certain actions. Although this policy is a significant action under Executive Order 12866, it is not expected to significantly affect energy supplies, distribution, or use. Therefore, this action is not a significant energy action and no Statement of Energy Effects is required.

Unfunded Mandates Reform Act (2 U.S.C. 1501 et seq.)

In accordance with the Unfunded Mandates Reform Act (2 U.S.C. 1501 *et seq.*):

(a) This policy will not ''significantly or uniquely'' affect small governments. A Small Government Agency Plan is not required. We expect that this policy will not result in any significant additional expenditures by entities that develop formalized conservation efforts.

(b) This policy will not produce a Federal mandate on state, local, or tribal governments or the private sector of $100 million or greater in any year; that is, it is not a ''significant regulatory action'' under the Unfunded Mandates Reform Act. This policy imposes no obligations on state, local, or tribal governments (see Economic Analysis below).

Takings

In accordance with Executive Order 12630, this policy does not have significant takings implications. While state, local or Tribal governments, or private entities may choose to directly or indirectly implement actions that may have property implications, they would do so as a result of their own decisions, not as a result of this policy. This policy has no provision that would take private property.

Federalism

In accordance with Executive Order 13132, this policy does not have significant Federalism effects. A Federalism assessment is not required. In keeping with Department of the Interior and Commerce policy, we requested information from and coordinated development of this policy with appropriate resource agencies throughout the United States.

Civil Justice Reform

In accordance with Executive Order 12988, this policy does not unduly burden the judicial system and meets the requirements of sections 3(a) and 3(b)(2) of the Order. With the guidance provided in the policy, requirements under section 4 of the Endangered Species Act will be clarified to entities that voluntarily develop formalized conservation efforts.

Paperwork Reduction Act of 1995 (44 U.S.C. 3501 et seq.)

This policy contains collection-of-information requirements subject to the Paperwork Reduction Act (PRA) and which have been approved by Office of Management and Budget (OMB). The FWS has OMB approval for the collection under OMB Control Number 1018–0119, which expires on December 31, 2005. The NMFS has OMB approval for the collection under OMB Control Number 0648–0466, which expires on December 31, 2005. We may not conduct or sponsor, and a person is not required to respond to, a collection of information unless it displays a currently valid OMB control number. Public reporting burden for FWS collections of information is estimated to average 2,500 hours for developing one agreement with the intent to preclude a listing, 320 hours for annual monitoring under one agreement, and 80 hours for one annual report. The FWS expects that six agreements with the intent of making listing unnecessary will be developed in one year and that four of these will be successful in making listing unnecessary, and therefore, the entities who develop these four agreements will carry through with their monitoring and reporting commitments. Public reporting burden for NMFS collections of information is estimated to average 2,500 hours for developing one agreement with the intent to preclude a listing, 320 hours for annual monitoring under one agreement, and 80 hours for one annual report. The NMFS expects that two agreements with the intent of making listing unnecessary will be developed in one year and that one of these will be successful in making listing unnecessary, and therefore, the entities who develop this agreement will carry through with their monitoring and reporting commitments. These estimates include the time for reviewing instructions, searching existing data sources, gathering and maintaining the data needed, and completing and reviewing the collection of information. Send comments regarding this burden estimate, or any other aspect of this data collection, including suggestions for reducing the burden, to the FWS and NMFS (see **ADDRESSES** section of this policy).

National Environmental Policy Act

We have analyzed this policy in accordance with the criteria of the National Environmental Policy Act (NEPA), the Department of the Interior Manual (318 DM 2.2(g) and 6.3(D)), and National Oceanic and Atmospheric Administration (NOAA) Administrative Order 216–6. This policy does not constitute a major Federal action significantly affecting the quality of the human environment. The FWS has determined that the issuance of the policy is categorically excluded under the Department of the Interior's NEPA procedures in 516 DM 2, Appendix 1 (1.10) and 516 DM 6, Appendix 1. NOAA has determined that the issuance of this policy qualifies for a categorical exclusion as defined by NOAA Administrative Order 216–6, Environmental Review Procedure.

ESA Section 7 Consultation

We have determined that issuance of this policy will not affect species listed as threatened or endangered under the Endangered Species Act, and, therefore, a section 7 consultation on this policy is not required.

Government-to-Government Relationship With Tribes

In accordance with the President's memorandum of April 29, 1994, ''Government-to-Government Relations with Native American Tribal Governments'' (59 FR 22951), E.O. 13175, and the Department of Interior's 512 DM 2, this policy does not directly affect Tribal resources. The policy may have an indirect effect on Native American Tribes as the policy may influence the type and content of conservation plans and efforts implemented by Tribes, or other entities. The extent of this indirect effect will be determined on a case-by-case basis during our evaluation of individual formalized conservation efforts when we make a listing decision. Under Secretarial Order 3206, we will, at a minimum, share with the entity that developed the formalized conservation effort any information provided by the Tribes, through the public comment period for the listing decision or formal submissions. During the development of conservation plans, we can encourage the incorporation of conservation efforts that will restore or enhance Tribal trust resources. After consultation with the Tribes and the entity that developed the formalized conservation effort and after careful consideration of the Tribe's concerns, we must clearly state the rationale for the recommended final listing decision and explain how the decision relates to our trust responsibility. Accordingly:

(a) We have not yet consulted with the affected Tribe(s). We will address this requirement when we evaluate formalized conservation efforts that have yet to be implemented or have recently been implemented and have yet to show effectiveness at the time we make a listing decision.

(b) We have not yet worked with Tribes on a government-to-government basis. We will address this requirement when we evaluate formalized conservation efforts that have yet to be implemented or have recently been implemented but have yet to show effectiveness at the time we make a listing decision.

(c) We will consider Tribal views in individual evaluations of formalized conservation efforts.

(d) We have not yet consulted with the appropriate bureaus and offices of the Department about the identified effects of this policy on Tribes. This requirement will be addressed with individual evaluations of formalized conservation efforts.

Information Quality

In Accordance with section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001 (Public Law 106–554), OMB directed Federal agencies to issue and implement guidelines to ensure and maximize the quality, objectivity, utility, and integrity of Government information disseminated to the public (67 FR 8452). Under our Information Quality guidelines, if we use a conservation plan or agreement as part of our decision to either list or not list a species under the Act, the plan or agreement is considered to be disseminated by us and these guidelines apply to the plan or agreement. The criteria outlined in this policy are consistent with OMB, Department of Commerce, NOAA, and Department of the Interior. FWS information quality guidelines. The Department of the Interior's guidelines can be found at *http://www.doi.gov/ocio/guidelines/515Guides.pdf*, and the FWS's guidelines can be found at *http://irm.fws.gov/infoguidelines/.* The Department of Commerce's guidelines can be found at *http://www.osec.doc.gov/cio/oipr/iqg.html*, and the NOAA/NMFS's guidelines can be found at *http://www.noaanews.noaa.gov/stories/iq.htm*. Under these guidelines, any affected person or organization may request from FWS or NMFS, a correction of information they believe to be incorrect in the plan or agreement. ''Affected persons or organizations'' are those who may use, be benefitted by, or be harmed by the disseminated information (i.e., the conservation plan or agreement). The process for submitting a request for correction of information is found in the respective FWS and NOAA guidelines.

Economic Analysis

This policy identifies criteria that a formalized conservation effort must satisfy to ensure certainty of implementation and effectiveness and for us to determine that the conservation effort contributes to making listing a species unnecessary or contributes to forming a basis for listing a species as threatened rather than endangered. We developed this policy to ensure consistent and adequate evaluation of agreements and plans when making listing decisions. The policy will also provide guidance to States and other entities on how we will evaluate certain formalized conservation efforts during the listing process.

The criteria in this policy primarily describe elements that are already included in conservation efforts and that constitute sound conservation planning. For example, the criteria requiring identification of responsible parties, obtaining required authorizations, establishment of objectives, and inclusion of an implementation schedule and monitoring provisions are essential for directing the implementation and affirming the effectiveness of conservation efforts. These kinds of ''planning'' requirements are generally already included in conservation efforts and do not establish any new implementation burdens. Rather, these requirements will help to ensure that conservation efforts are well planned and, therefore, increase the likelihood that conservation efforts will ultimately be successful in making listing species unnecessary.

The development of an agreement or plan by a state or other entity is completely voluntary. However, when a state or other entity voluntarily decides to develop an agreement or plan with the specific intent of making listing a species unnecessary, the criteria identified in this policy can be construed as requirements placed on the development of such agreements or plans. The state or other entity must satisfy these criteria in order to obtain and retain the benefit they are seeking, which is making listing of a species as threatened or endangered unnecessary.

The criteria in the policy require demonstrating certainty of implementation and effectiveness of formalized conservation efforts. We have always considered the certainty of implementation and effectiveness of conservation efforts when making listing decisions. Therefore, we believe that no economic effects on states and other entities will result from using the criteria in this policy as guidance.

Furthermore, publication of this policy will have positive effects by informing States and other entities of the criteria we will use in evaluating formalized conservation efforts when making listing decisions, and thereby guide states and other entities in developing voluntary formalized conservation efforts that will be successful in making listing unnecessary. Therefore, we believe that informational benefits will result from issuing this policy. We believe these benefits, although important, will be insignificant economically.

Authority

The authority for this action is the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq.*).

Policy for Evaluation of Conservation Efforts When Making Listing Decisions

Policy Purpose

The Fish and Wildlife Service and National Marine Fisheries Service developed this policy to ensure consistent and adequate evaluation of formalized conservation efforts (conservation efforts identified in conservation agreements, conservation plans, management plans, and similar documents) when making listing decisions under the Act. This policy may also guide the development of conservation efforts that sufficiently improve a species' status so as to make listing the species as threatened or endangered unnecessary.

Definitions

''Adaptive management'' is a method for examining alternative strategies for meeting measurable biological goals and objectives, and then, if necessary, adjusting future conservation management actions according to what is learned.

''Agreements and plans'' include conservation agreements, conservation plans, management plans, or similar documents approved by Federal agencies, State and local governments, Tribal governments, businesses, organizations, or individuals.

''Candidate species,'' as defined by regulations at 50 CFR 424.02(b), means any species being considered for listing as an endangered or a threatened species, but not yet the subject of a proposed rule. However, the FWS includes as candidate species those species for which the FWS has sufficient information on file relative to status and threats to support issuance of proposed listing rules. The NMFS includes as candidate species those species for which it has information indicating that listing may be warranted, but for which sufficient information to support actual proposed listing rules may be lacking. The term ''candidate species'' used in this policy refers to those species designated as candidates by either of the Services.

''Conservation efforts,'' for the purpose of this policy, are specific actions, activities, or programs designed to eliminate or reduce threats or otherwise improve the status of a species. Conservation efforts may involve restoration, enhancement, maintenance, or protection of habitat; reduction of mortality or injury; or other beneficial actions.

''Formalized conservation efforts'' are conservation efforts identified in a conservation agreement, conservation plan, management plan, or similar document. An agreement or plan may contain numerous conservation efforts.

Policy Scope

When making listing decisions, the Services will evaluate whether formalized conservation efforts contribute to making it unnecessary to list a species, or to list a species as threatened rather than endangered. This policy applies to those formalized conservation efforts that have not yet been implemented or have been implemented, but have not yet demonstrated whether they are effective at the time of a listing decision. We will make this evaluation based on the certainty of implementing the conservation effort and the certainty that the effort will be effective. This policy identifies the criteria we will use to help determine the certainty of implementation and effectiveness. Listing decisions covered by the policy include findings on petitions to list species, and decisions on whether to assign candidate status, remove candidate status, issue proposed listing rules, and finalize or withdraw proposed listing rules. This policy applies to formalized conservation efforts developed with or without a specific intent to influence a listing decision and with or without the involvement of the Services.

Section 4(a)(1) of the Endangered Species Act of 1973, as amended (16 U.S.C. 1533(a)(1)), states that we must determine whether a species is threatened or endangered because of any of the following five factors:(A) the present or threatened destruction, modification, or curtailment of its habitat or range; (B) overutilization for commercial, recreational, scientific, or educational purposes; (C) disease or predation; (D) the inadequacy of existing regulatory mechanisms; or (E) other natural or manmade factors affecting its continued existence.

Although this language focuses on impacts negatively affecting a species, section 4(b)(1)(A) requires us also to ''tak[e] into account those efforts, if any, being made by any State or foreign nation, or any political subdivision of a State or foreign nation, to protect such species, whether by predator control, protection of habitat and food supply, or other conservation practices, within any area under its jurisdiction, or on the high seas.'' Read together, sections 4(a)(1) and 4(b)(1)(A), as reflected in our regulations at 50 CFR 424.11(f), require us to take into account any State or local laws, regulations, ordinances, programs, or other specific conservation measures that either positively or negatively affect a species' status (i.e., measures that create, exacerbate, reduce, or remove threats identified through the section 4(a)(1) analysis). The manner in which the section 4(a)(1) factors are framed supports this conclusion. Factor (D) for example—ldquo;the inadequacy of existing regulatory mechanisms''—indicates that overall we might find existing regulatory mechanisms adequate to justify a determination not to list a species.

Factor (E) in section 4(a)(1) (any ''manmade factors affecting [the species'] continued existence'') requires us to consider the pertinent laws, regulations, programs, and other specific actions of any entity that either positively or negatively affect the species. Thus, the analysis outlined in section 4 of the Act requires us to consider the conservation efforts of not only State and foreign governments but also of Federal agencies, Tribal governments, businesses, organizations, or individuals that positively affect the species' status.

While conservation efforts are often informal, such as when a property owner implements conservation measures for a species simply because of concern for the species or interest in protecting its habitat, and without any specific intent to affect a listing decision, conservation efforts are often formalized in conservation agreements, conservation plans, management plans, or similar documents. The development and implementation of such agreements and plans has been an effective mechanism for conserving declining species and has, in some instances, made listing unnecessary. These efforts are consistent with the Act's finding that ''encouraging the States and other interested parties * * * to develop and maintain conservation programs * * * is a key * * * to better safeguarding, for the benefit of all citizens, the Nation's heritage in fish, wildlife, and plants'' (16 U.S.C. 1531 (a)(5)).

In some situations, a listing decision must be made before all formalized conservation efforts have been implemented or before an effort has demonstrated effectiveness. We may determine that a formalized conservation effort that has not yet been implemented has reduced or removed a threat to a species when we have sufficient certainty that the effort will be implemented and will be effective.

Determining whether a species meets the definition of threatened or endangered requires us to analyze a species' risk of extinction. Central to this risk analysis is an assessment of the status of the species (i.e., is it in decline or at risk of decline and at what rate is the decline or risk of decline) and consideration of the likelihood that current or future conditions or actions will promote (see section 4(b)(1)(A)) or threaten a species' persistence. This determination requires us to make a prediction about the future persistence of a species, including consideration of both future negative and positive effects of anticipated human actions. The language of the Act supports this approach. The definitions for both ''endangered species'' and ''threatened species'' connote future condition, which indicates that consideration of whether a species should be listed depends in part on identification and evaluation of future actions that will reduce or remove, as well as create or exacerbate, threats to the species. The first factor in section 4(a)(1)—''the present or *threatened* destruction, modification, or curtailment of [the species'] habitat or range''—identifies how analysis of both current actions affecting a species' habitat or range and those actions that are sufficiently certain to occur in the future and affect a species' habitat or range are necessary to assess a species' status. However, future Federal, State, local, or private actions that affect a species are not limited to actions that will affect a species' habitat or range. Congress did not intend for us to consider future actions affecting a species' habitat or range, yet ignore future actions that will influence overutilization, disease, predation, regulatory mechanisms, or other natural or manmade factors. Therefore, we construe Congress' intent, as reflected by the language of the Act, to require us to consider both current actions that affect a species' status and sufficiently certain future actions—either positive or negative—that affect a species' status. As part of our assessment of future conditions, we will determine whether a formalized conservation effort that has yet to be implemented or has recently been implemented but has yet to show effectiveness provides a high level of certainty that the effort will be implemented and/or effective and results in the elimination or adequate reduction of the threats.

For example, if a state recently designed and approved a program to eliminate collection of a reptile being considered for listing, we must assess how this program affects the status of the species. Since the program was just designed, an implementation and effectiveness record may not yet exist. Therefore, we must evaluate the likelihood, or certainty, that it will be implemented and effective, using evidence such as the State's ability to enforce new regulations, educate the public, monitor compliance, and monitor the effects of the program on the species. Consequently, we would determine that the program reduces the threat of overutilization of the species through collecting if we found sufficient certainty that the program would be implemented and effective.

In another example, a state could have a voluntary incentive program for protection and restoration of riparian habitat that includes providing technical and financial assistance for fencing to exclude livestock. Since the state has already implemented the program, the state does not need to provide certainty that it will be implemented. If the program was only recently implemented and no record of the effects of the program on the species' status existed, we would evaluate the effectiveness of this voluntary program at the time of our listing decision. To assess the effectiveness, we would evaluate the level of participation (e.g., number of participating landowners or number of stream-miles fenced), the length of time of the commitment by landowners, and whether the program reduces the threats on the species. We would determine that the program reduces the threat of habitat loss and degradation if we find sufficient certainty that the program is effective.

In addition, we will consider the estimated length of time that it will take for a formalized conservation effort to produce a positive effect on the species. In some cases, the nature, severity, and/or imminence of threats to a species may be such that a formalized conservation effort cannot be expected to produce results quickly enough to make listing unnecessary since we must determine at the time of the listing decision that the conservation effort has improved the status of the species.

Federal agencies, Tribal governments, state and local governments, businesses, organizations, or individuals contemplating development of an agreement or plan should be aware that, because the Act mandates specific timeframes for making listing decisions, we cannot delay the listing process to allow additional time to complete the development of an agreement or plan. Nevertheless, we encourage the development of agreements and plans even if they will not be completed prior to a final listing decision. Such an agreement or plan could serve as the foundation for a special rule under section 4(d) of the Act, which would establish only those prohibitions necessary and advisable for the conservation of a threatened species, or for a recovery plan, and could lead to earlier recovery and delisting.

This policy provides us guidance for evaluating the certainty of implementation and effectiveness of formalized conservation efforts. This policy is not intended to provide guidance for determining the specific level of conservation (e.g., number of populations or individuals) or the types of conservation efforts (e.g., habitat restoration, local regulatory mechanisms) specifically needed to make listing particular species unnecessary and does not provide guidance for determining when parties should enter into agreements. We do encourage early coordination in conservation measures to prevent the species from meeting the definition of endangered or threatened.

If we make a decision not to list a species or to list the species as threatened rather than endangered based in part on the contributions of a formalized conservation effort, we will track the status of the effort including the progress of implementation and effectiveness of the conservation effort. If any of the following occurs: (1) a failure to implement the conservation effort in accordance with the implementation schedule; (2) a failure to achieve objectives; (3) a failure to modify the conservation effort to adequately address an increase in the severity of a threat or to address other new information on threats; or (4) we receive any other new information indicating a possible change in the status of the species, then we will reevaluate the status of the species and consider whether initiating the listing process is necessary. Initiating the listing process may consist of designating the species as a candidate species and assigning a listing priority, issuing a proposed rule to list, issuing a proposed rule to reclassify, or issuing an emergency listing rule. In some cases, even if the parties fully implement all of the conservation efforts outlined in a particular agreement or plan, we may still need to list the species. For example, this may occur if conservation efforts only cover a portion of a species' range where the species needed to be conserved, or a particular threat to a species was not anticipated or addressed at all, or not adequately addressed, in the agreement or plan.

Evaluation Criteria

Conservation agreements, conservation plans, management plans, and similar documents generally identify numerous conservation efforts (i.e., actions, activities, or programs) to benefit the species. In determining whether a formalized conservation effort contributes to forming a basis for not listing a species, or for listing a species as threatened rather than endangered, we must evaluate whether the conservation effort improves the status of the species under the Act. Two factors are key in that evaluation: (1) for those efforts yet to be implemented, the certainty that the conservation effort will be implemented and (2) for those efforts that have not yet demonstrated effectiveness, the certainty that the conservation effort will be effective. Because the certainty of implementation and effectiveness of formalized conservation efforts may vary, we will evaluate each effort individually and use the following criteria to direct our analysis.

A. The certainty that the conservation effort will be implemented:

1. The conservation effort, the party(ies) to the agreement or plan that will implement the effort, and the staffing, funding level, funding source, and other resources necessary to implement the effort are identified. 2. The legal authority of the party(ies) to the agreement or plan to implement the formalized conservation effort, and the commitment to proceed with the conservation effort are described.3. The legal procedural requirements (e.g. environmental review) necessary to implement the effort are described, and information is provided indicating that fulfillment of these requirements does

not preclude commitment to the effort. 4. Authorizations (e.g., permits, landowner permission) necessary to implement the conservation effort are identified, and a high level of certainty is provided that the party(ies) to the agreement or plan that will implement the effort will obtain these authorizations. 5. The type and level of voluntary participation (e.g., number of landowners allowing entry to their land, or number of participants agreeing to change timber management practices and acreage involved) necessary to implement the conservation effort is identified, and a high level of certainty is provided that the party(ies) to the agreement or plan that will implement the conservation effort will obtain that level of voluntary participation (e.g., an explanation of how incentives to be provided will result in the necessary level of voluntary participation). 6. Regulatory mechanisms (e.g., laws, regulations, ordinances) necessary to implement the conservation effort are in place. 7. A high level of certainty is provided that the party(ies) to the agreement or plan that will implement the conservation effort will obtain the necessary funding. 8. An implementation schedule (including incremental completion dates) for the conservation effort is provided. 9. The conservation agreement or plan that includes the conservation effort is approved by all parties to the agreement or plan.

B. The certainty that the conservation effort will be effective:

1. The nature and extent of threats being addressed by the conservation effort are described, and how the conservation effort reduces the threats is described. 2. Explicit incremental objectives for the conservation effort and dates for achieving them are stated. 3. The steps necessary to implement the conservation effort are identified in detail. 4. Quantifiable, scientifically valid parameters that will demonstrate achievement of objectives, and standards for these parameters by which progress will be measured, are identified. 5. Provisions for monitoring and reporting progress on implementation (based on compliance with the implementation schedule) and effectiveness (based on evaluation of quantifiable parameters) of the conservation effort are provided.6. Principles of adaptive management are incorporated.

These criteria should not be considered comprehensive evaluation criteria. The certainty of implementation and effectiveness of a formalized conservation effort may also depend on species-specific, habitat-specific, location-specific, and effort-specific factors. We will consider all appropriate factors in evaluating formalized conservation efforts. The specific circumstances will also determine the amount of information necessary to satisfy these criteria.

To consider that a formalized conservation effort(s) contributes to forming a basis for not listing a species or listing a species as threatened rather than endangered, we must find that the conservation effort is sufficiently certain to be implemented and effective so as to have contributed to the elimination or adequate reduction of one or more threats to the species identified through the section 4(a)(1) analysis. The elimination or adequate reduction of section 4(a)(1) threats may lead to a determination that the species does not meet the definition of threatened or endangered, or is threatened rather than endangered. An agreement or plan may contain numerous conservation efforts, not all of which are sufficiently certain to be implemented and effective. Those conservation efforts that are not sufficiently certain to be implemented and effective cannot contribute to a determination that listing is unnecessary or a determination to list as threatened rather than endangered. Regardless of the adoption of a conservation agreement or plan, however, if the best available scientific and commercial data indicate that the species meets the definition of ''endangered species'' or ''threatened species'' on the day of the listing decision, then we must proceed with appropriate rule-making activity under section 4 of the Act.

Dated: September 16, 2002.

Steve Williams,

Director, Fish and Wildlife Service.

December 23, 2002.

William T. Hogarth,

Assistant Administrator for Fisheries, National Marine Fisheries Services.

[FR Doc. 03–7364 Filed 3–27–03; 8:45 am]

BILLING CODES 4310–55–S and 3510–22–S

DEPARTMENT OF COMMERCE

National Oceanic and Atmospheric Administration

50 CFR Part 679

[Docket No. 021212306–2306–01; I.D. 032403A]

Fisheries of the Exclusive Economic Zone Off Alaska; Pollock in Statistical Area 610 of the Gulf of Alaska

AGENCY: National Marine Fisheries Service (NMFS), National Oceanic and Atmospheric Administration (NOAA), Commerce.

ACTION: Modification of a closure.

SUMMARY: NMFS is reopening directed fishing for pollock in Statistical Area 610 of the Gulf of Alaska (GOA) for 24 hours. This action is necessary to fully use the B season allowance of the total allowable catch (TAC) of pollock specified for Statistical Area 610.

DATES: Effective 1200 hrs, Alaska local time (A.l.t.), March 26, 2003, through 1200 hrs, A.l.t., March 27, 2003.

FOR FURTHER INFORMATION CONTACT: Mary Furuness, 907–586–7228.

SUPPLEMENTARY INFORMATION: NMFS manages the groundfish fishery in the GOA exclusive economic zone according to the Fishery Management Plan for Groundfish of the Gulf of Alaska (FMP) prepared by the North Pacific Fishery Management Council under authority of the Magnuson-Stevens Fishery Conservation and Management Act. Regulations governing fishing by U.S. vessels in accordance with the FMP appear at subpart H of 50 CFR part 600 and 50 CFR part 679.

NMFS closed the B season directed fishery for pollock in Statistical Area 610 of the GOA under § 679.20(d)(1)(iii) on March 19, 2003 (68 FR 13857, March 21, 2003).

NMFS has determined that, approximately 986 mt of pollock remain in the B season directed fishing allowance. Therefore, in accordance with 679.25(a)(2)(i)(C) and (a)(2)(iii)(D), and to fully utilize the B season allowance of pollock TAC specified for Statistical Area 610, NMFS is terminating the previous closure and is reopening directed fishing for pollock in Statistical Area 610 of the GOA. In accordance with § 679.20(d)(1)(iii), the Regional Administrator finds that this directed fishing allowance will be reached after 24 hours. Consequently, NMFS is prohibiting directed fishing for pollock in Statistical Area 610 of the GOA effective 1200 hrs, A.l.t., March 27, 2003.

APPENDIX 3

Evaluation of Lek Counts Using Simulations

Introduction

The use of lek counts to evaluate populations has been controversial with greater sage-grouse (Walsh 2002, Walsh et al. 2004) and with other species of prairie grouse (Applegate 2000). A central issue involves the use of lek counts to estimate populations, even without supporting data on the population's actual parameters (Anderson 2001, Walsh et al. 2004). Some of this concern is based on attendance rates for males on leks. There are little range-wide data on these rates but in Colorado rates varied between 42% to 92% for adult males and 19% to 86% for yearling males (Emmons and Braun 1984, Dunn and Braun 1985, Walsh et al. 2004). Some of this variation may be due to monitoring design, but some may be due to annual and/or regional variation (Walsh et al. 2004). Walsh et al. (2004) also expressed concerns that estimation of long-term trends using lek counts might be faced with similar problems but did not examine the relationship between lek counts and trends.

Therefore, to better understand the reliability of lek data for assessing population trends in this conservation assessment, we tested the lek count procedure using simulated populations. Simulated populations have an advantage over real populations in this type of analysis because their characteristics can be defined and their rates of population change accurately monitored. Hence the lek count procedure employed throughout the sage-grouse range in North America can be examined and its probability of detecting real trends assessed.

Simulated populations

We designed simulated populations with relatively conservative characteristics. Each simulated population contained 20 leks in the first year and was monitored for 20 additional years (21 years total with 20 annual rates of population change). Because each of the actual populations or subpopulations contained an average of 93 leks (Appendix 4 and 5), this approach was believed to be conservative. The average observed attendance at 5,585 leks on 35,919 different yearly occasions was 26.4 males (SD = 27.4 males). Because attendance rates on leks may be relatively low (42% for adult males and 19% for yearling males, Walsh et al. 2004), we attempted to simulate leks with an average attendance of 50 males. Although there are few data to verify the actual size of sage-grouse leks, there are numerous data elucidating the distribution of leks by size category (Fig. A3.1). Because lek size appears to approximate an exponential distribution, we used an exponential equation to randomly assign the starting attendance at each lek with numbers varying between 1 and 300.

Following the initial year, simulated populations were allowed to vary annually based on an assigned overall rate of change (0.0 in this case) and an annual yearly variance around the overall rate of change (randomly assigned with SD = 0.1). Individual leks were also assigned a random annual variance (SD = 0.1) so that they could vary independently to a certain extent. Hence, some leks within a simulated population could increase while others were declining, even though most leks were likely to fluctuate in the same direction. Similarly, a simulated population with an overall increase could display occasional years with decreases. All leks were assigned the nearest whole number of males. An initial assessment of simulated populations with these characteristics suggested that they were consistent with observed populations and with demographic characteristics obtained with radio telemetry. For example, simulated populations did not vary much beyond what normal survival and productivity estimates would suggest is possible (Schroeder et al. 1999, Crawford et al. 2004). Annual rates of change for each simulated population were estimated as the natural logs of the result of the populations in year x divided by the populations in year x-1.

Sampling of simulated populations

An analysis of survey intensity by year illustrated a dramatic increase in effort over time (Fig. A3.2). Thus we used an equation (25% + 2.5%*year) to approximate the increase in survey intensity from about 25% to 75% during the last 21 years (1983 was considered year 0 in the equation). We also attached a random sampling variance (SD = 10%) to each year to reflect some of the normal annual variation in sampling intensity.

A randomly selected annual survey effort was used to determine the number of surveyed leks in each year for each simulated population. The only deviation from this procedure is that leks surveyed in the previous year were more likely to be surveyed (approximately 65% of leks surveyed in one year are surveyed the following year). The proportion of simulated males actually observed was based in part on average male attendance at leks (values ranging from 19-92%; Emmons and Braun 1984, Dunn and Braun 1985, Walsh et al. 2004). Because of the lack of solid and consistent information on the topic, we designed the program to conservatively select 50% of the males actually present with an assigned random variance (SD = 15%). We bounded all observations by 0 and the actual lek size and all observations were rounded to the nearest whole number. This variance ensured that some leks would be observed to have zero males, which often happens under field conditions. Annual rates of change for each sampled population were estimated as the natural logs of the result of males counted on leks in year x divided by the males counted on the same leks in year x-1 (only leks counted in consecutive years were used for each annual estimate). Although this procedure was conservative, we designed it to reflect actual sampling and analysis techniques.

Results

Average initial lek size for 10,000 simulated populations was 49.7 males. The overall average instantaneous rate of population change for the actual simulated populations was 0.0003 (SD = 0.0413). The rate of change based on surveys of these simulated populations was 0.0003 (SD = 0.0418). Neither value was significantly different than zero. The annual rate of 10,000 estimated population changes deviated from the actual rate of population changes by an average of 0.0384 (SD = 0.0308); this number was not correlated with the actual rate of population change (r^2 = 0.0001). An evaluation of accuracy suggested that accuracy increased with the observed rate of population change (Fig. A3.3). Accuracy was generally greater than 80% for populations with an observed annual rate of change of at least 0.03 and greater than 95% with rates of a least 0.07.

We used these accuracy statistics to conduct a preliminary evaluation of the 41 populations (Appendix 4) and 24 subpopulations (Appendix 5) identified in this conservation assessment. Most of the populations or subpopulations fit at least one of the following categories: 1) too few years for analysis; 2) too few data for analysis; and/or 3) population changes during the last 20 years were not apparent or not significant. Fifteen of the 65 populations or subpopulations had 20-year trends in data (1983-2003) that were either significant, or close to being significant (Table A3.1). The two largest areas with significant changes (both subpopulations) were E-Interior MT/NE Tip WY (> 100 leks) and NE WY/SE MT (> 200 leks). Although these subpopulations were grouped into different populations (Yellowstone Watershed and Wyoming Basin, respectively), they are geographically close and appeared to be declining at similar rates.

Discussion

Although this analysis is preliminary, it suggests population trends may be appropriately assessed using lek counts. In previous research the only way to verify long-term trends was with regional or local extirpations (Schroeder et al. 2000, 2004). For example, evidence presented in this document (Appendix 4) illustrates trends for five populations followed until extirpation.

This analysis may be better applied to specific populations in the future by using some of the applicable parameters for that particular population. It is impossible to include every source of variation and there may always be effects such as annual variation in the number of counts per lek. Because most of these sources of variation have not been examined in detail, these results should be viewed with caution (see Walsh et al. 2004). Nevertheless, this is the first indication that the significance of trends using lek counts can be supported by data other than with the finality of localized extirpations (Schroeder et al. 2004).

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Table A3.1. Populations and subpopulations exhibiting significant trends (based on simulation analysis) either upward or downward between 1983 and 2003.

Population (subpopulation)	Trend direction	Significance[a]
Red Rock MT	Down	Significant
Belt Mountains MT	Down	Trend
Yellowstone Watershed (E-Enterior MT/NE Tip WY)	Down	Significant
Moses Coulee	Down	Trend
NE-Interior UT	Down	Significant
S Mono Lake CA	Up	Trend
Snake, Salmon, and Beaverhead (Big Lost ID)	Down	Significant
Sanpete/Emery UT	Down	Significant
Wyoming Basin (NE WY/SE MT)	Down	Significant
Summit/Morgan UT	Up	Significant
Tooele/Juab UT	Up	Significant
Weiser ID	Up	Significant
Yakima WA	Down	Significant
E-Central ID	Down	Significant
Snake, Salmon, and Beaverhead (Little Lost ID)	Up	Trend

[a]Trend represents an 80% or higher probability of being accurate and significant represents at 95% or higher probability of being accurate.

Fig. A3.1. Distribution of 5,585 greater sage-grouse leks in North America by size category.

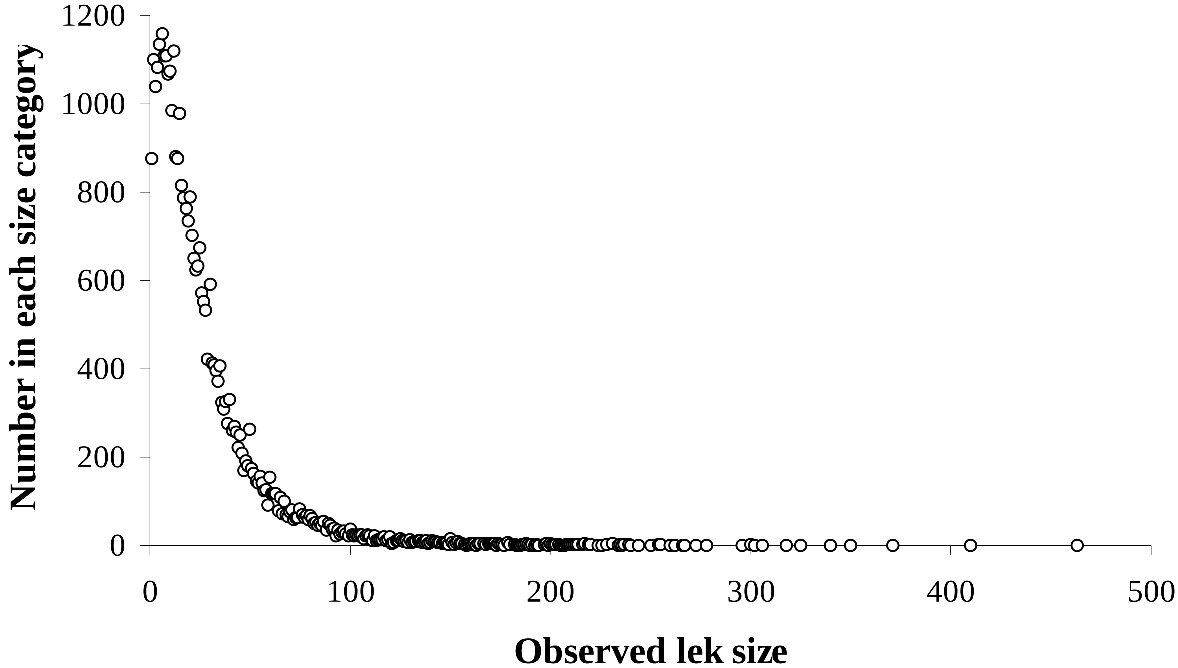


Fig. A3.2. Survey intensity of greater sage-grouse leks by year in North America (also see Table 6.18).

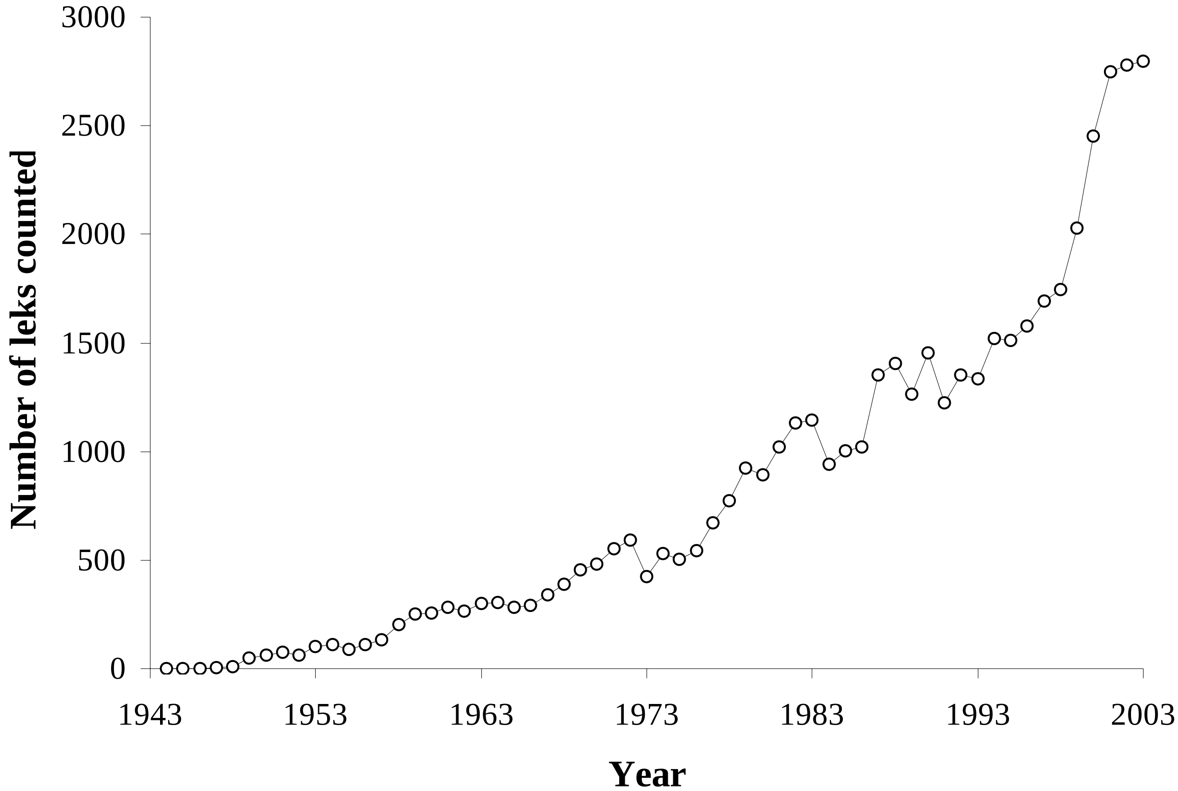
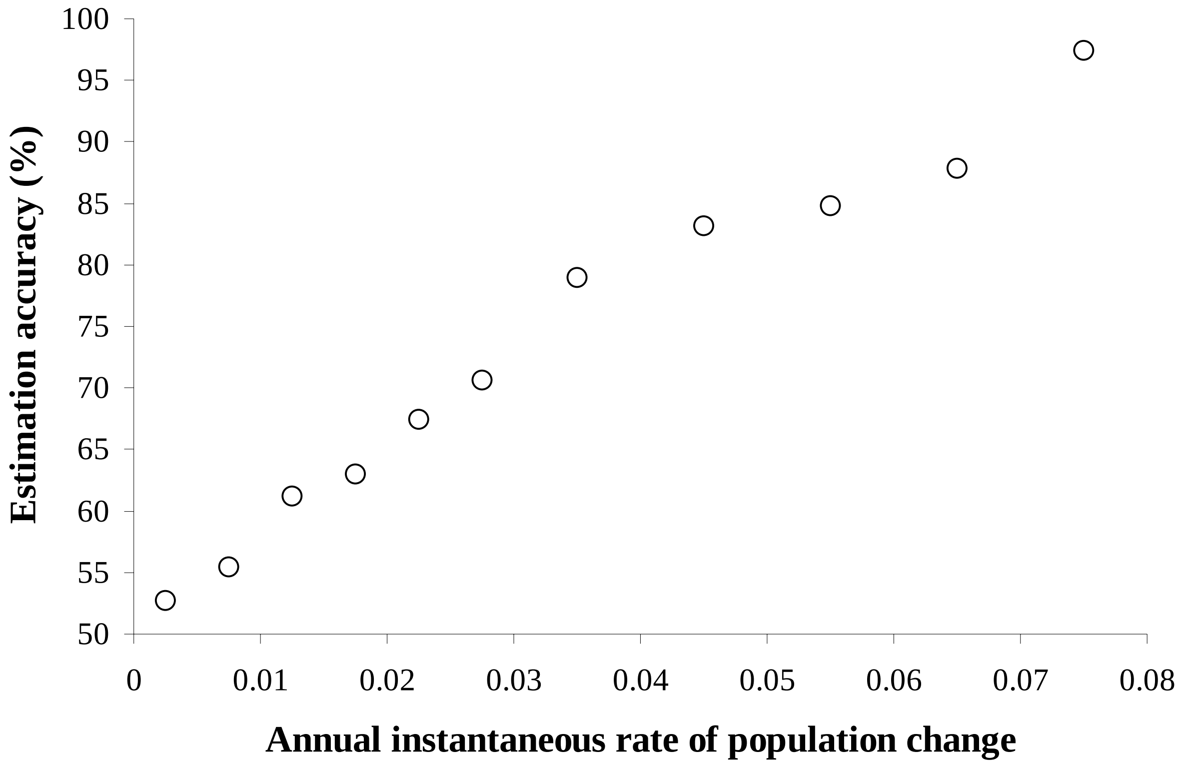


Fig. A3.3. Accuracy of trend estimation in relation to the average annual instantaneous rates of population change for 10,000 simulated populations, each containing 20 leks.



APPENDIX 4

Characteristics of Greater Sage-Grouse Populations

Methods

Forty-one distinct populations of greater sage-grouse were defined on the basis of isolation-by-distance and/or isolation-by-topography rather than political and/or jurisdictional boundaries (Table 6.16, Fig. 6.37). Five of these populations were further divided into an addition 24 subpopulations based on their large size, expansive distribution, differences in region, and a relatively small degree of separation (Appendix 5).

In general, lek attendance data was analyzed identically to the methods used in Chapter 6. However, because many of the populations were extremely small and data were limited, presentation of the data in the figures below was expanded in some situations to include multi-year intervals. For example in a situation where lek counts were not obtained in one year (year x-1), we estimated the instantaneous rate of change based on the lek count on either end of the 2-year interval (natural log of [lek count in year x/lek count in year x-2]). The instantaneous rates were then divided by the number of intervals (2 in this case) to obtain the instantaneous rate of change for the given yearly interval. To estimate the lek count in the intervening year (x-1) the following formula was used: (lek count in year x –2) * (exponential of the instantaneous rate of change for a single year). This procedure enabled us to track long-term changes in situations were lek counts not completely thorough; back to 1965 when possible. Nevertheless, we were unable to apply this procedure in situations where the starting or ending lek count was zero because the instantaneous rate of change was undefined in those cases. In the following figures, years without data do not have points.

Because many of the populations and subpopulations have very small sample sizes of leks, care should be taken when interpreting the data. The small sample sizes appear to be reflected in the dramatic year-to-year changes in some areas. This volatility is clearly more apparent in the smaller data sets than in the larger data sets. Consequently, more attention should be paid to the overall trends, rather than annual fluctuations and/or the magnitude of changes.

Results

Table A4.1. Sage-grouse monitoring and population trends in Baker OR population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	8	6	2	1	0	0	0	0
Number of active leks[1]	8	5	2	1	0	0	0	0
Percent active leks	100	86	89	100				
Average males/lek	25	23	22	36				
Median males/lek	24	23	19	41				
Average males/active lek	25	27	25	36				
Median males/active lek	24	28	26	41				

[1] Averaged over each year for each period.

Fig. A4.1. Change in the population index for Baker OR population, 1989-2003.

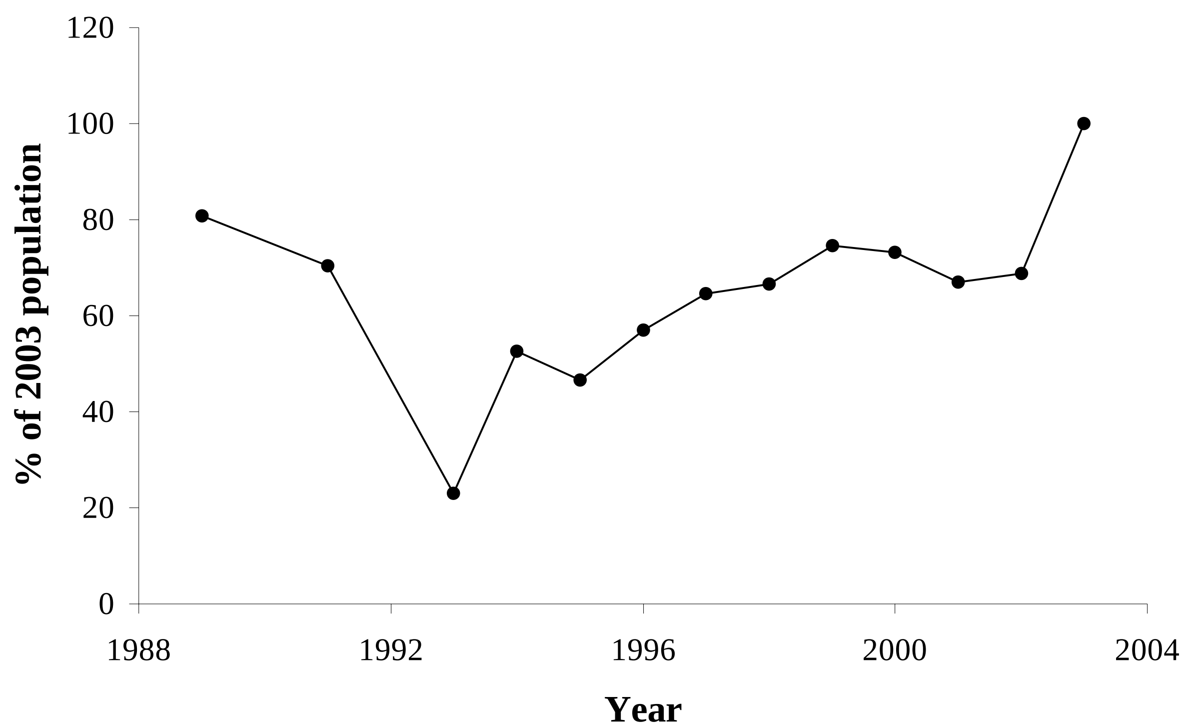


Table A4.2. Sage-grouse monitoring and population trends in Bannack MT population, summarized over 5-year periods, 1972 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	12	3	2	4	4	4	2	2
Number of active leks[1]	10	3	2	4	4	4	2	2
Percent active leks	83	100	100	100	100	100	100	100
Average males/lek	13	14	19	23	30	39	30	18
Median males/lek	9	14	20	22	28	36	34	13
Average males/active lek	15	14	19	23	30	39	30	18
Median males/active lek	14	14	20	22	28	36	34	13

[1] Averaged over each year for each period.

Fig. A4.2. Change in the population index for Bannack MT population, 1972-2003.

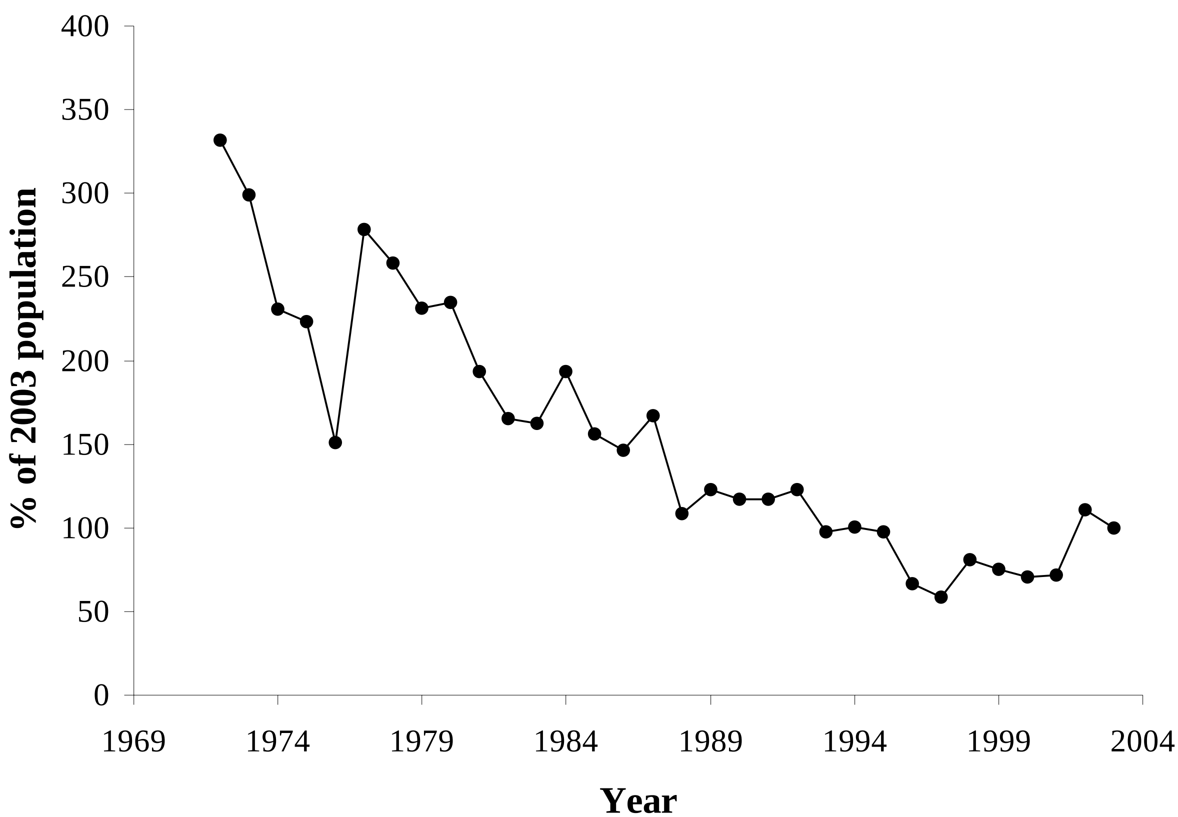


Table A4.3. Sage-grouse monitoring and population trends in Belt Mountains MT population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	14	10	8	8	9	10	6	1
Number of active leks[1]	8	5	7	7	9	10	6	1
Percent active leks	58	51	80	92	98	100	100	100
Average males/lek	10	7	9	13	25	24	37	88
Median males/lek	12	16	17	16	28	37	73	
Average males/active lek	18	15	11	14	26	24	37	88
Median males/active lek	14	20	17	16	28	37	73	

[1] Averaged over each year for each period.

Fig. A4.3. Change in the population index for Belt Mountains MT population, 1969-2003.

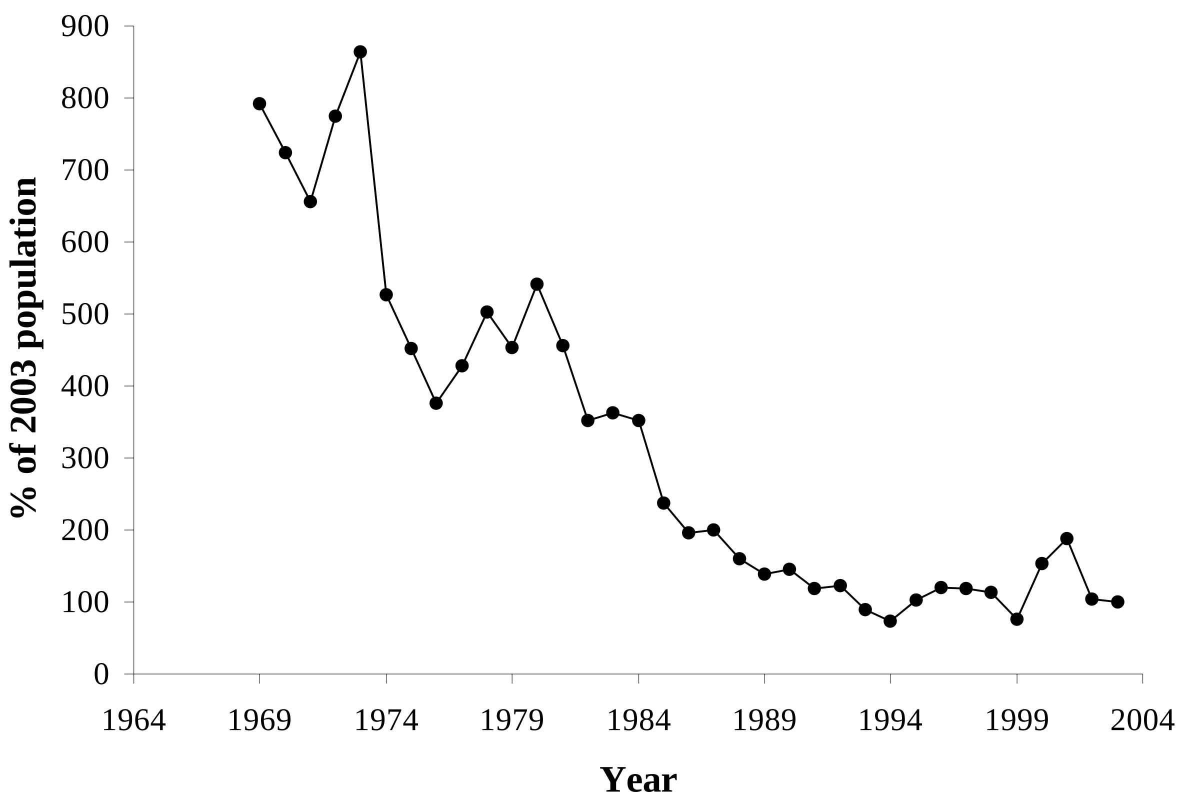


Table A4.4. Sage-grouse monitoring and population trends in Central OR population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	64	47	30	22	18	10	12	14
Number of active leks[1]	52	42	26	20	13	6	10	13
Percent active leks	81	89	86	90	69	62	80	96
Average males/lek	12	13	14	14	11	11	13	24
Median males/lek	7	11	12	12	7	6	10	18
Average males/active lek	14	14	16	16	16	17	16	25
Median males/active lek	11	12	14	13	15	14	12	21

[1] Averaged over each year for each period.

Fig. A4.4. Change in the population index for Central OR population, 1965-2003.

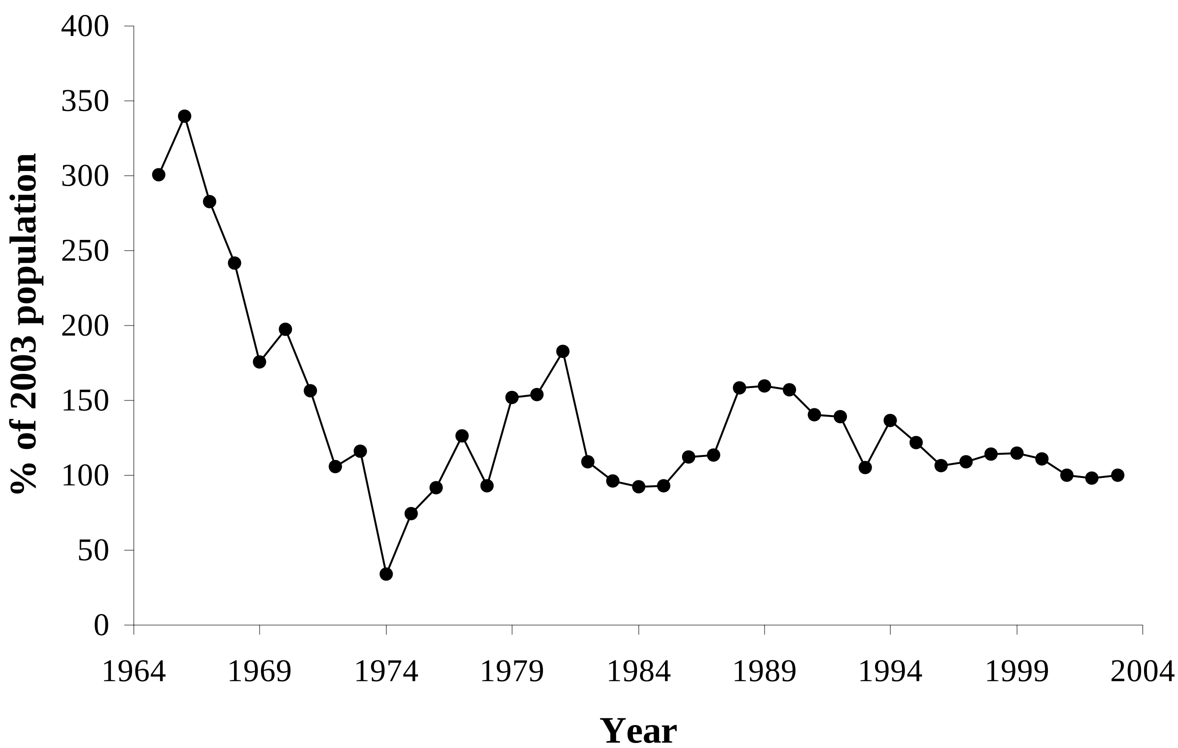


Table A4.5. Sage-grouse monitoring and population trends in Eagle/S Routt CO population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	59	23	37	23	32	13	6	8
Number of active leks[1]	31	12	23	15	27	12	5	7
Percent active leks	53	54	62	63	84	97	90	90
Average males/lek	9	5	8	7	14	13	23	24
Median males/lek	0	0	0	0	1		23	22
Average males/active lek	18	9	13	12	16	14	25	26
Median males/active lek	9	13	4	3	9		26	26

[1] Averaged over each year for each period.

Fig. A4.5. Change in the population index for Eagle/S Routt CO population, 1965-2003.

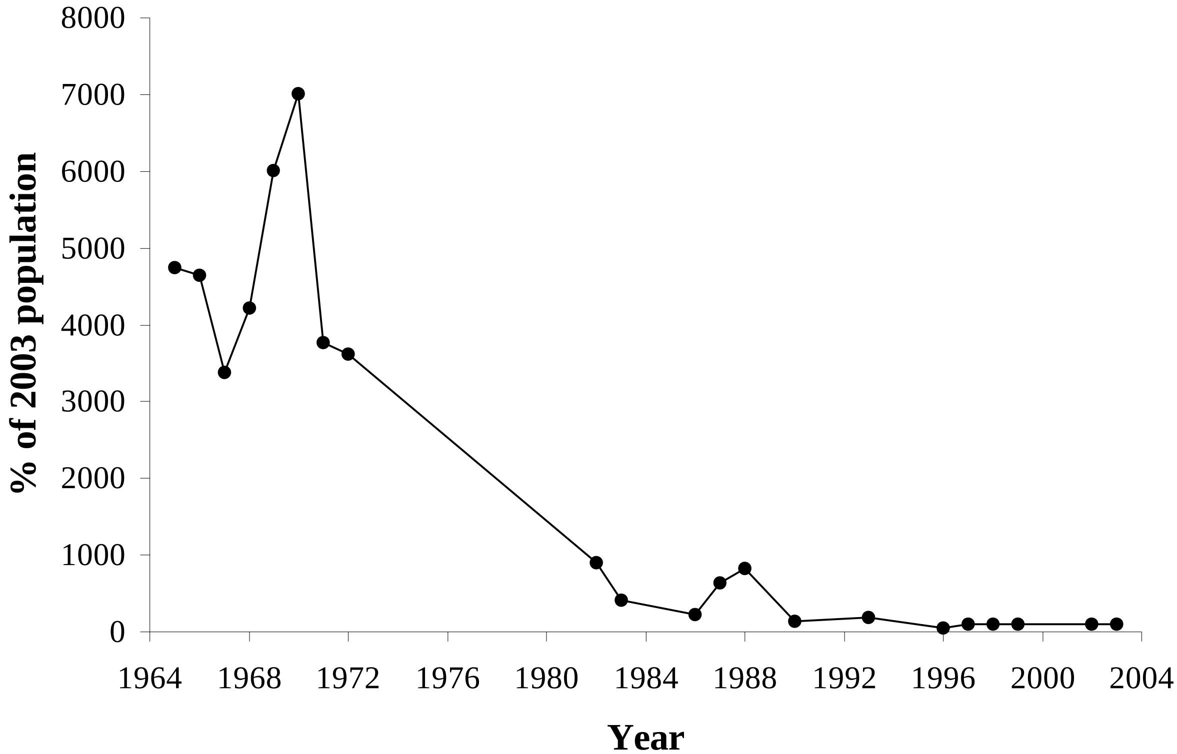


Table A4.6. Sage-grouse monitoring and population trends in East Tavaputs Plateau UT population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	0	0	2	2	2	1	1	0
Number of active leks[1]	0	0	1	2	1	1	1	0
Percent active leks		100	75	80	63	100	100	
Average males/lek		7	4	3	3	14	8	
Median males/lek		7	5	3	1	13	7	
Average males/active lek		7	6	3	4	14	8	
Median males/active lek		7	7	4	3	13	7	

[1] Averaged over each year for each period.

Fig. A4.6. Change in the population index for East Tavaputs Plateau UT population, 1971-2000.

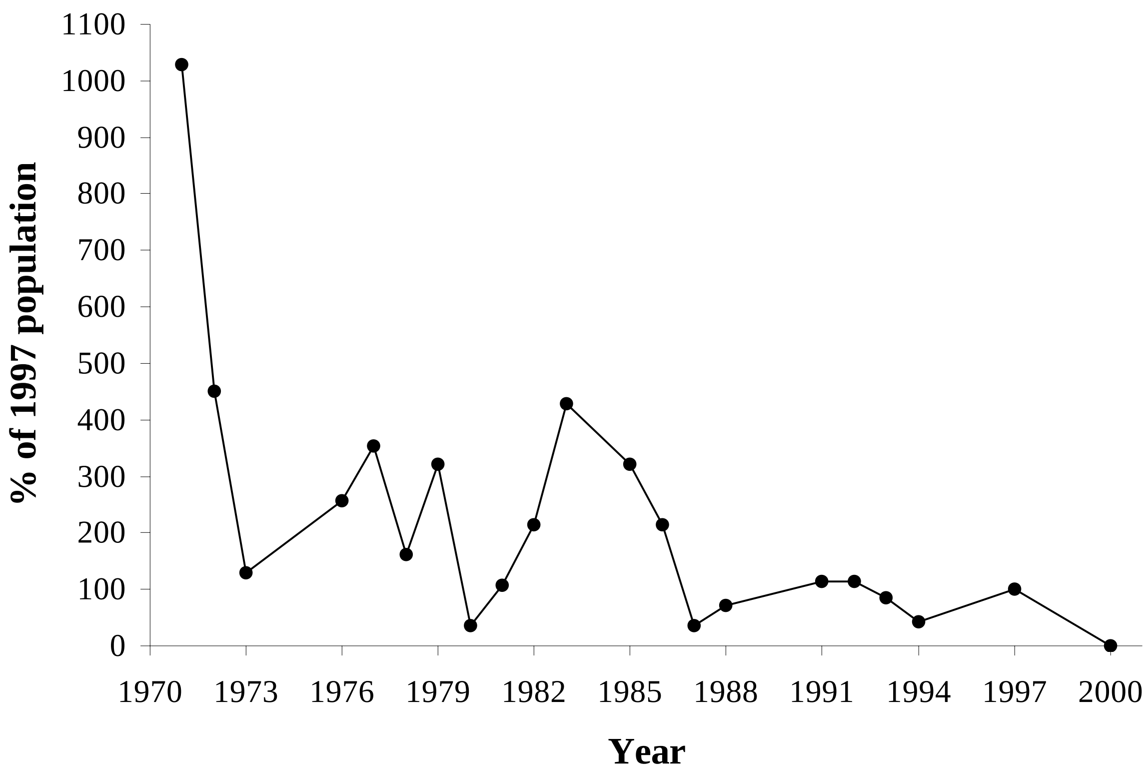


Table A4.7. Sage-grouse monitoring and population trends in E-Central ID population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	7	5	3	4	5	3	3	2
Number of active leks[1]	5	4	3	3	4	3	3	2
Percent active leks	74	83	100	80	91	100	81	89
Average males/lek	9	7	13	14	15	15	19	15
Median males/lek	5	6	10	12	12	13	18	10
Average males/active lek	12	8	13	17	17	15	23	17
Median males/active lek	10	6	10	15	15	13	20	13

[1] Averaged over each year for each period.

Fig. A4.7. Change in the population index for E-Central ID population, 1966-2002.

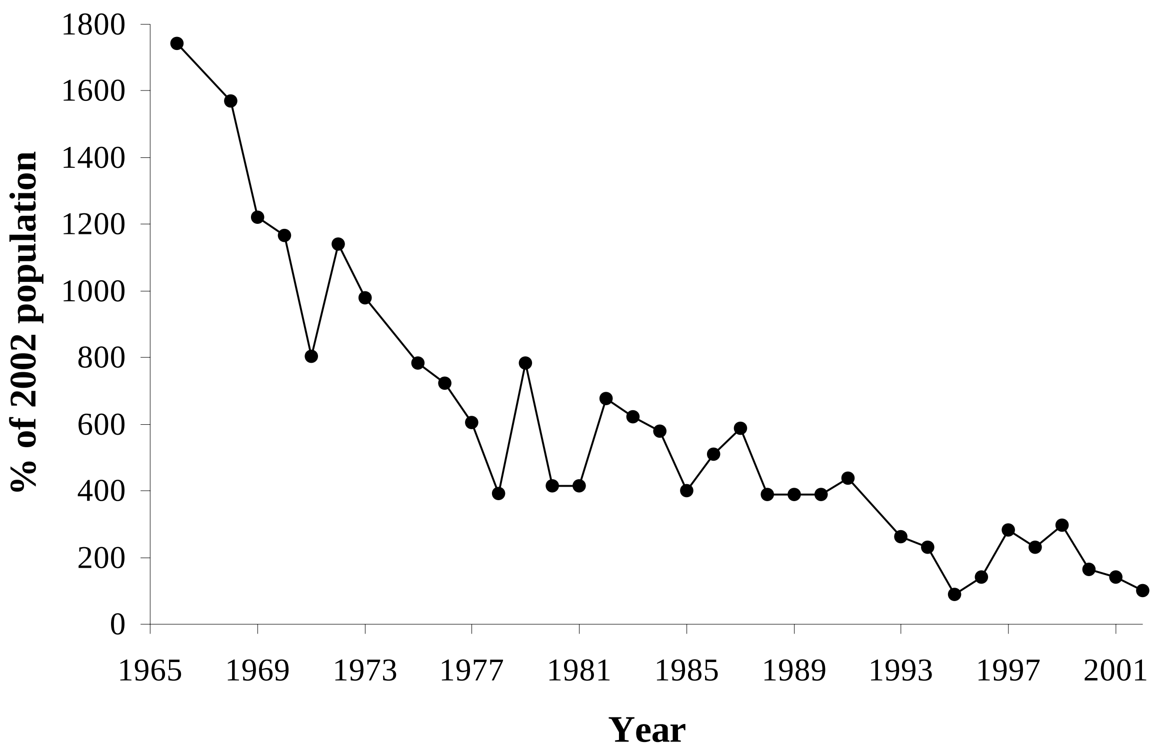


Table A4.8. Sage-grouse monitoring and population trends in E Garfield CO population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	0	0	1	1	1	0	1	1
Number of active leks[1]	0	0	0	0	0	0	0	1
Percent active leks			0	0	0		0	60
Average males/lek			0	0	0		0	4
Median males/lek			0	0	0		0	5
Average males/active lek								6
Median males/active lek								6

[1] Averaged over each year for each period.

Fig. A4.8. Change in the population index for E Garfield CO population, 1962-1969.

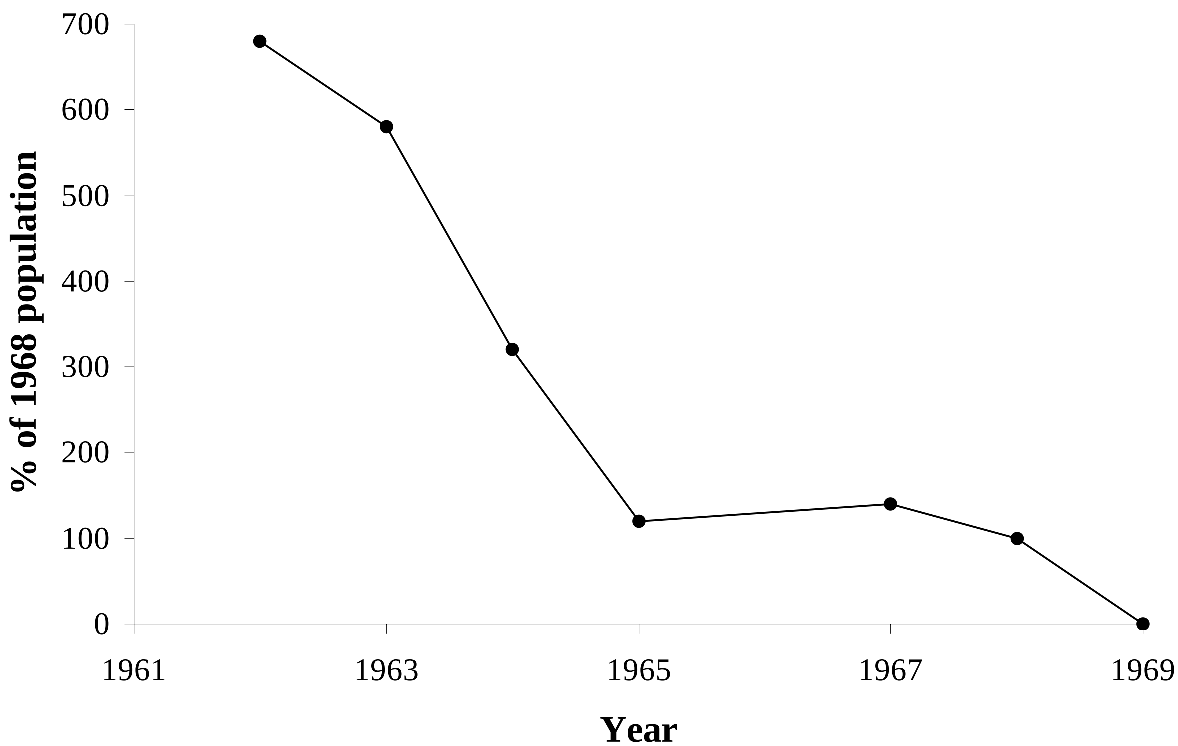


Table A4.9. Sage-grouse monitoring and population trends in Great Basin Core population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	438	294	179	148	167	100	93	46
Number of active leks[1]	359	249	164	124	144	83	76	42
Percent active leks	82	85	92	84	86	83	81	90
Average males/lek	19	16	25	24	23	24	24	28
Median males/lek	12	11	16	15	15	15	15	18
Average males/active lek	23	19	27	29	26	29	30	31
Median males/active lek	16	13	18	20	18	21	20	21

[1] Averaged over each year for each period.

Fig. A4.9. Change in the population index for Great Basin Core population, 1965-2003.

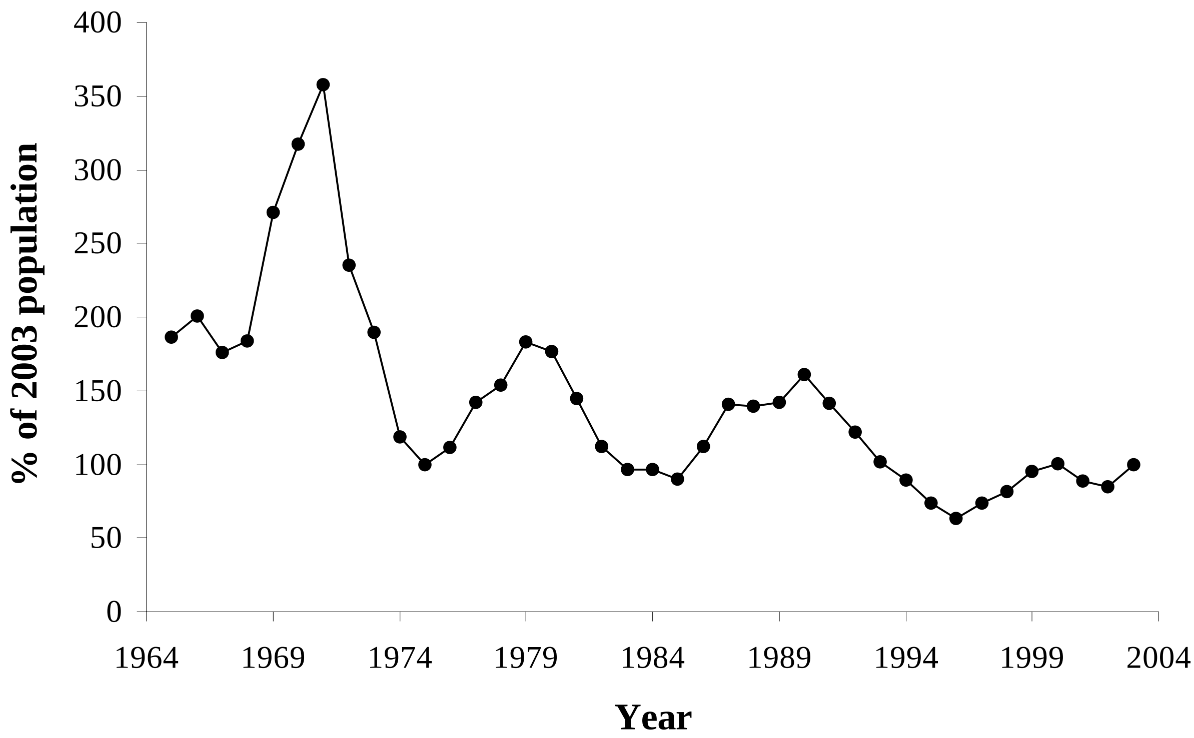


Table A4.10. Sage-grouse monitoring and population trends in Gunnison Range UT population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	0	1	1	1	0	0	0	0
Number of active leks[1]	0	0	1	1	0	0	0	0
Percent active leks		40	100	100				
Average males/lek		1	7	10				
Median males/lek		0	6	11				
Average males/active lek		2	7	10				
Median males/active lek		2	6	11				

[1] Averaged over each year for each period.

Fig. A4.10. Change in the population index for Gunnison Range UT population, 1984-1997.

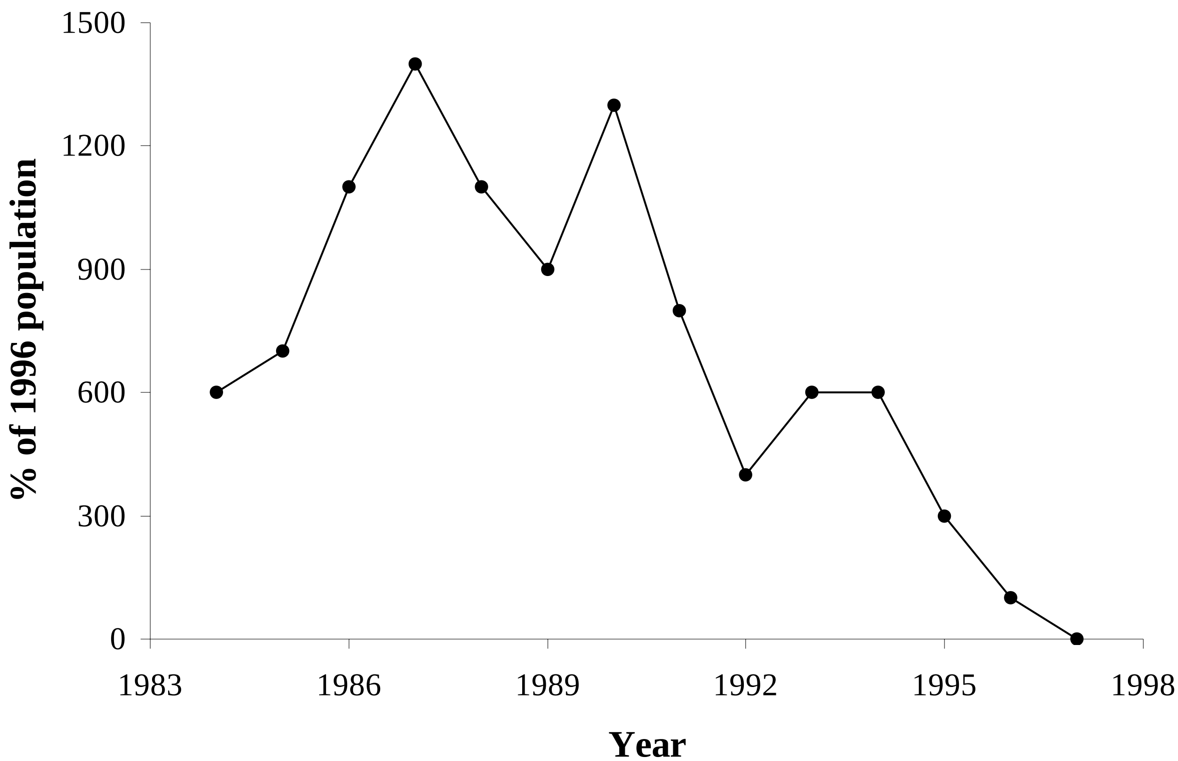


Table A4.11. Sage-grouse monitoring and population trends in Jackson Hole WY population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	7	7	6	4	0	0	0	0
Number of active leks[1]	4	5	6	4	0	0	0	0
Percent active leks	57	77	100	100				
Average males/lek	10	11	27	26				
Median males/lek	5	7	16	23				
Average males/active lek	17	15	27	26				
Median males/active lek	19	10	16	23				

[1] Averaged over each year for each period.

Fig. A4.11. Change in the population index for Jackson Hole WY population, 1986-2003.

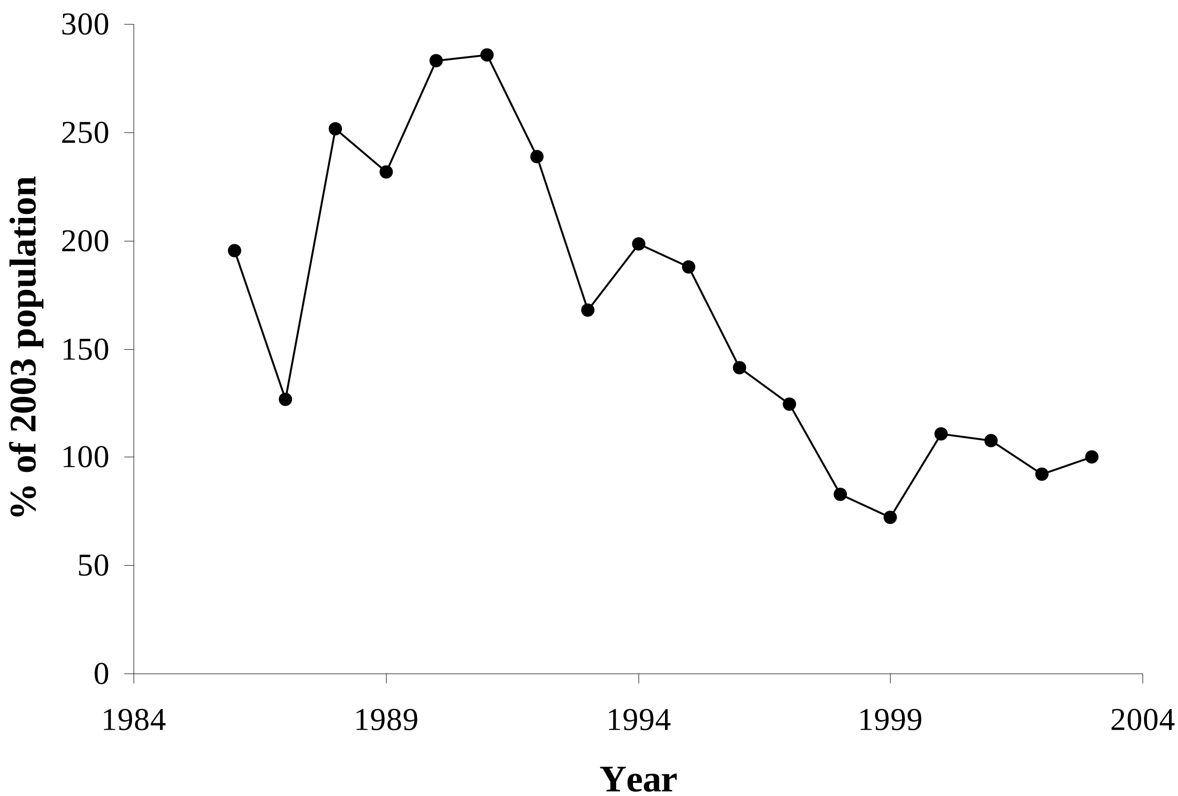


Table A4.12. Sage-grouse monitoring and population trends in Klamath OR/CA population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	3	2	2	0	0	0	1	0
Number of active leks[1]	1	1	0	0	0	0	1	0
Percent active leks	40	33	17				100	
Average males/lek	4	6	0				5	
Median males/lek	0	0	0				4	
Average males/active lek	10	17	3				5	
Median males/active lek	10	14	3				4	

[1] Averaged over each year for each period.

Fig. A4.12. Change in the population index for Klamath OR/CA, 1996-2003.

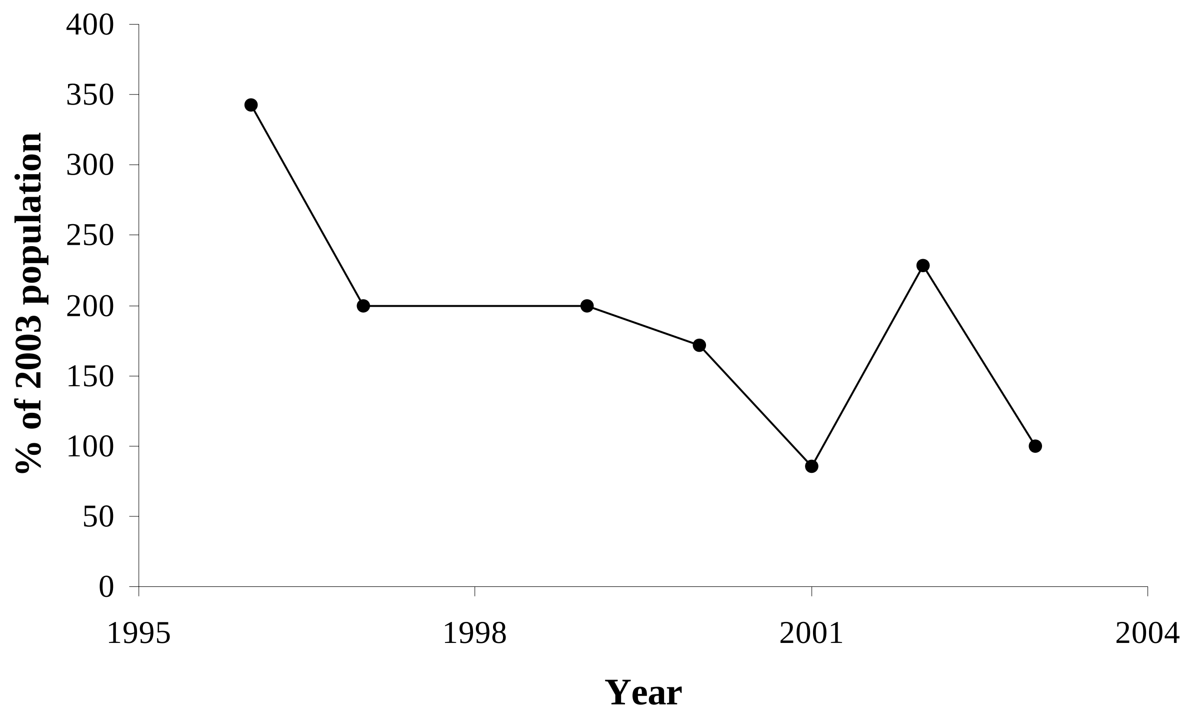


Table A4.13. Sage-grouse monitoring and population trends in Laramie WY population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	2	2	1	1	1	2	2	2
Number of active leks[1]	0	0	0	0	0	1	2	2
Percent active leks	14	0	67	29	20	25	92	100
Average males/lek	1	0	1	0	1	5	19	52
Median males/lek	0	0	1	0	0	0	15	46
Average males/active lek	5		2	2	4	19	21	52
Median males/active lek	5		2	2	4	15	16	46

[1] Averaged over each year for each period.

Fig. A4.13. Change in the population index for Laramie WY population, 1965-2003.

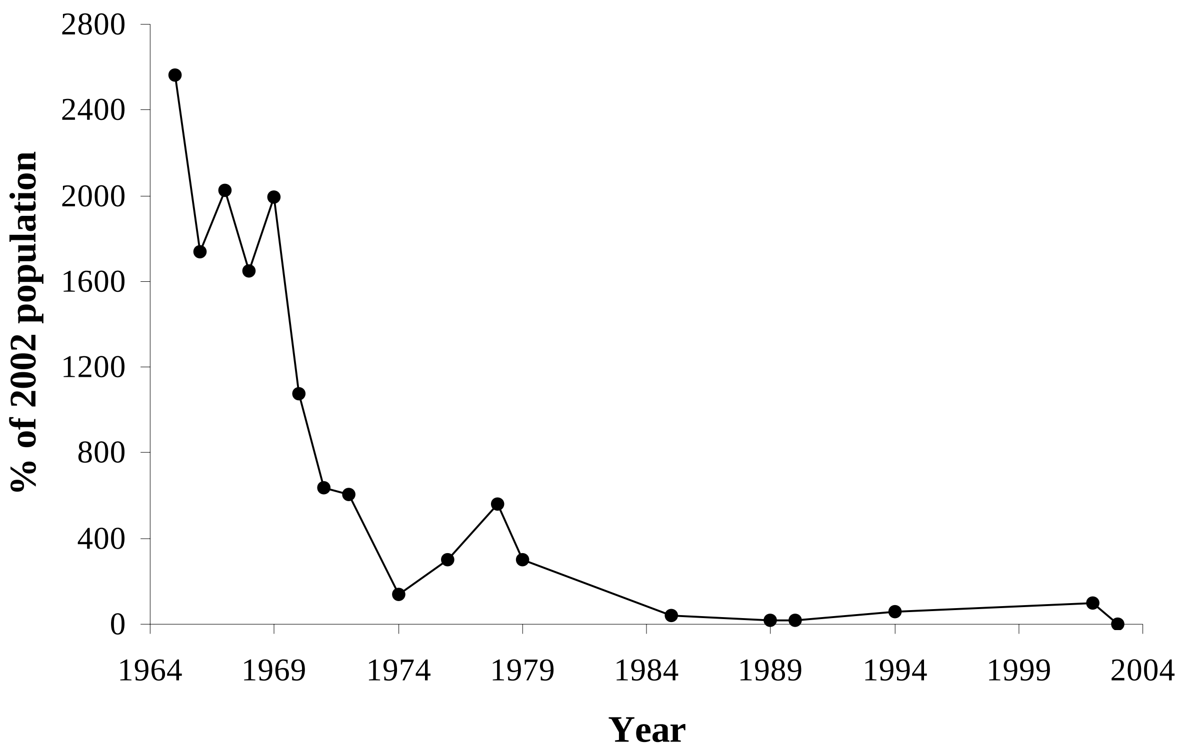


Table A4.14. Sage-grouse monitoring and population trends in Middle Park CO population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	11	11	10	10	6	8	4	6
Number of active leks[1]	11	8	6	7	5	7	4	5
Percent active leks	100	75	62	71	74	92	95	80
Average males/lek	18	11	10	11	16	26	20	17
Median males/lek	19	10	9	6	12	22	11	9
Average males/active lek	18	15	16	15	21	28	21	21
Median males/active lek	19	14	12	9	15	23	13	15

[1] Averaged over each year for each period.

Fig. A4.14. Change in the population index for Middle Park CO population, 1972-2002.

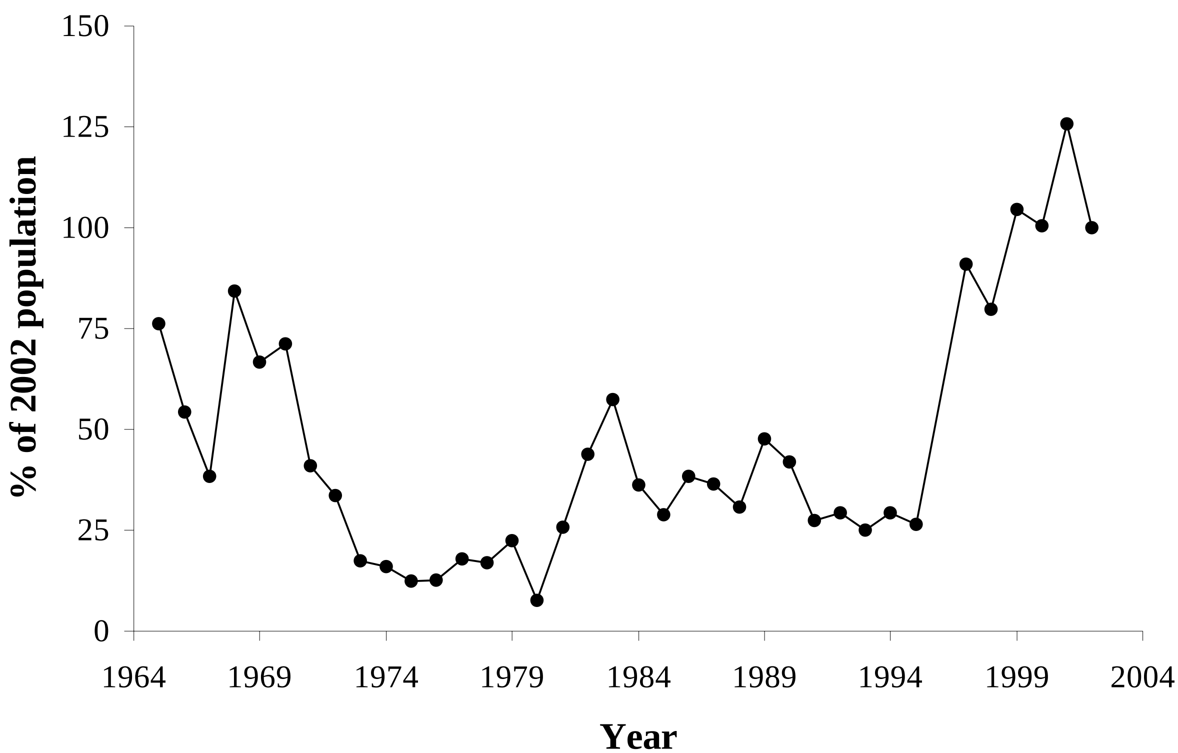


Table A4.15. Sage-grouse monitoring and population trends in Moses Coulee WA population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	37	24	18	16	15	13	13	3
Number of active leks[1]	12	12	11	10	13	11	12	3
Percent active leks	32	48	59	66	90	90	98	100
Average males/lek	6	10	11	14	22	14	25	33
Median males/lek	0	0	2	5	21	13	21	31
Average males/active lek	19	20	18	21	24	15	26	33
Median males/active lek	17	19	11	12	23	14	21	31

[1] Averaged over each year for each period.

Fig. A4.15. Change in the population index for Moses Coulee WA population, 1967-2003.

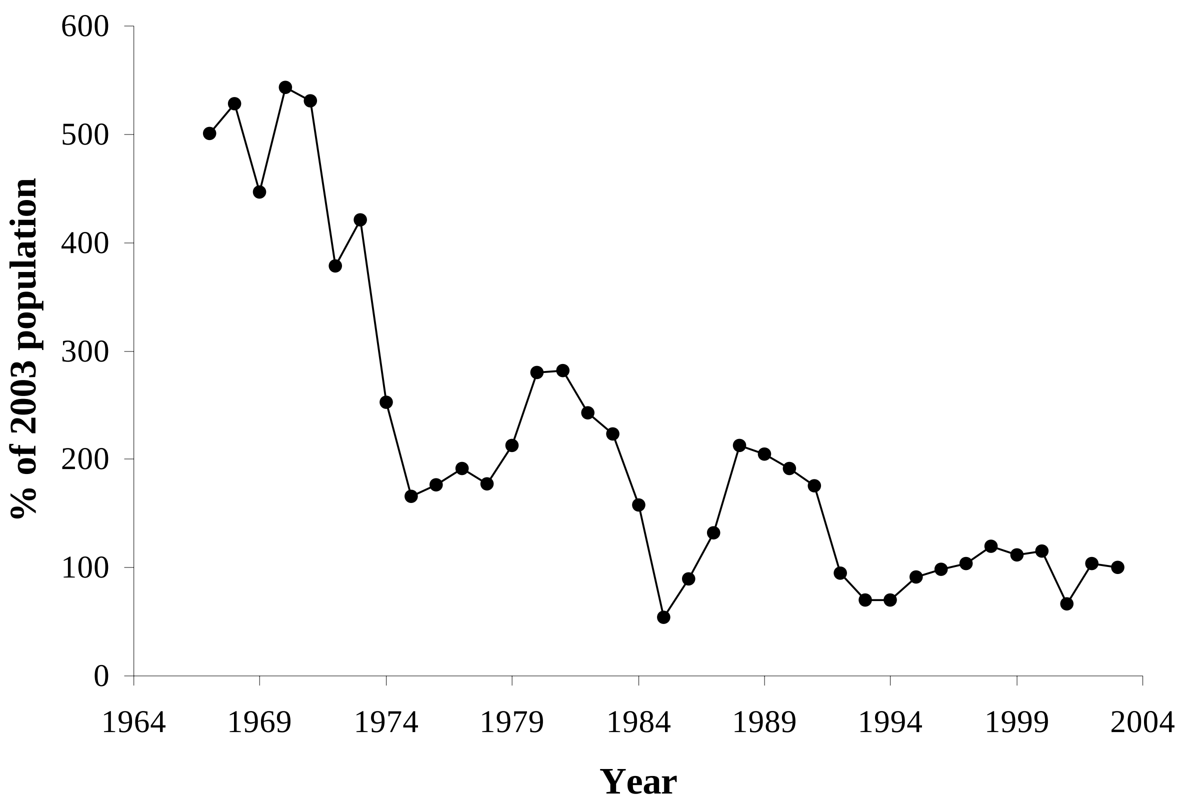


Table A4.16. Sage-grouse monitoring and population trends in MT/ND/NW SD population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	48	34	38	28	33	26	23	19
Number of active leks[1]	32	25	32	24	28	23	21	17
Percent active leks	66	74	83	86	87	89	92	90
Average males/lek	11	8	12	11	14	12	17	13
Median males/lek	7	5	9	8	10	11	17	10
Average males/active lek	16	10	14	13	16	14	19	15
Median males/active lek	12	9	12	10	14	12	18	12

[1] Averaged over each year for each period.

Fig. A4.16. Change in the population index for MT/ND/NW SD population, 1965-2003.

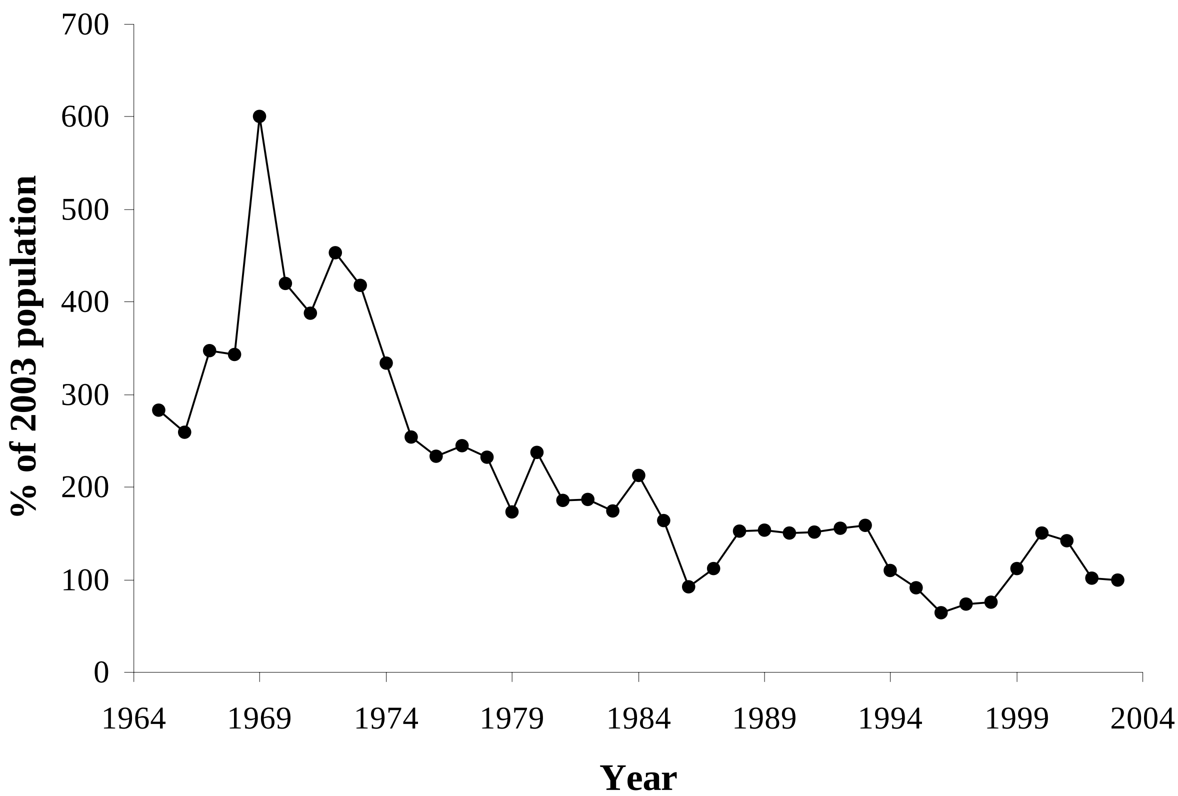


Table A4.17. Sage-grouse monitoring and population trends in N Mono Lake CA/NV population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	13	9	12	12	8	7	6	4
Number of active leks[1]	12	6	10	10	6	6	6	4
Percent active leks	90	67	88	83	67	83	90	95
Average males/lek	17	13	23	22	25	28	33	32
Median males/lek	11	10	22	18	19	14	26	23
Average males/active lek	19	19	27	26	37	33	36	33
Median males/active lek	13	13	23	23	33	24	32	24

[1] Averaged over each year for each period.

Fig. A4.17. Change in the population index for N Mono Lake CA/NV population, 1966-2003.

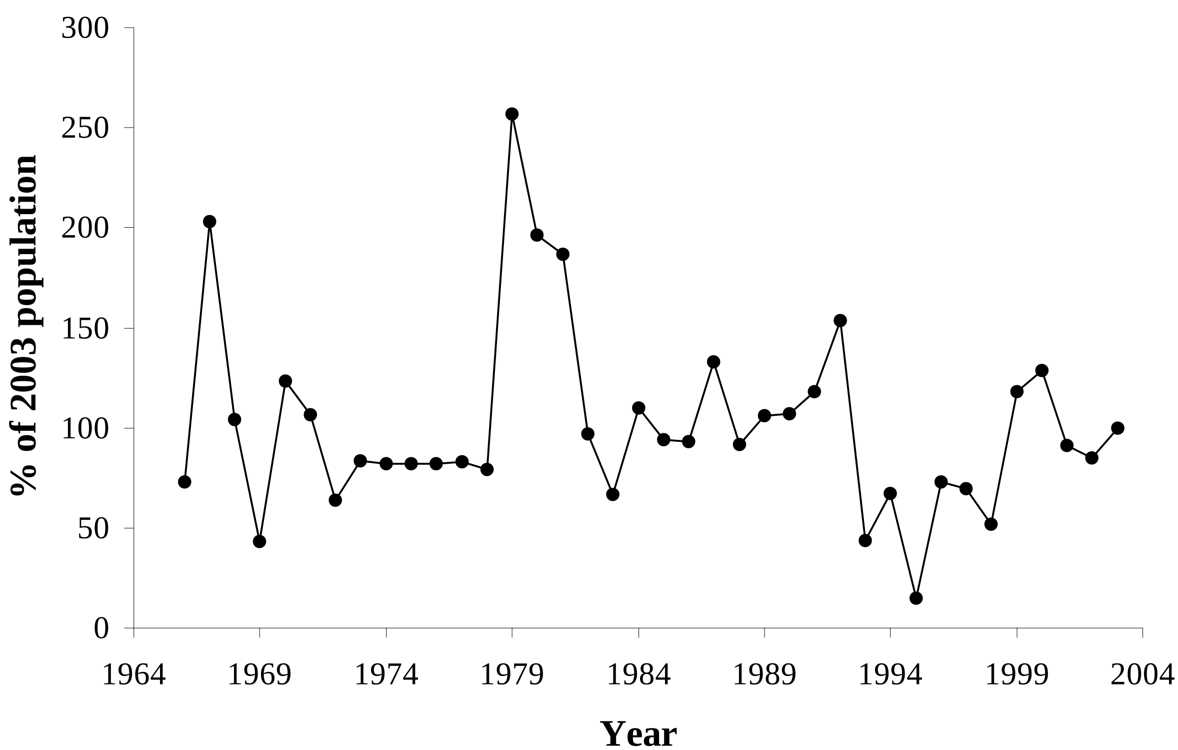


Table A4.18. Sage-grouse monitoring and population trends in NE Interior UT population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	13	10	15	12	14	18	9	1
Number of active leks[1]	10	8	11	9	9	17	8	1
Percent active leks	77	75	74	76	68	94	85	83
Average males/lek	14	10	12	19	11	25	20	8
Median males/lek	6	5	8	13	8	13	14	3
Average males/active lek	18	14	16	24	17	26	24	9
Median males/active lek	14	7	13	17	16	14	21	3

[1] Averaged over each year for each period.

Fig. A4.18. Change in the population index for NE Interior UT, 1970-2003.

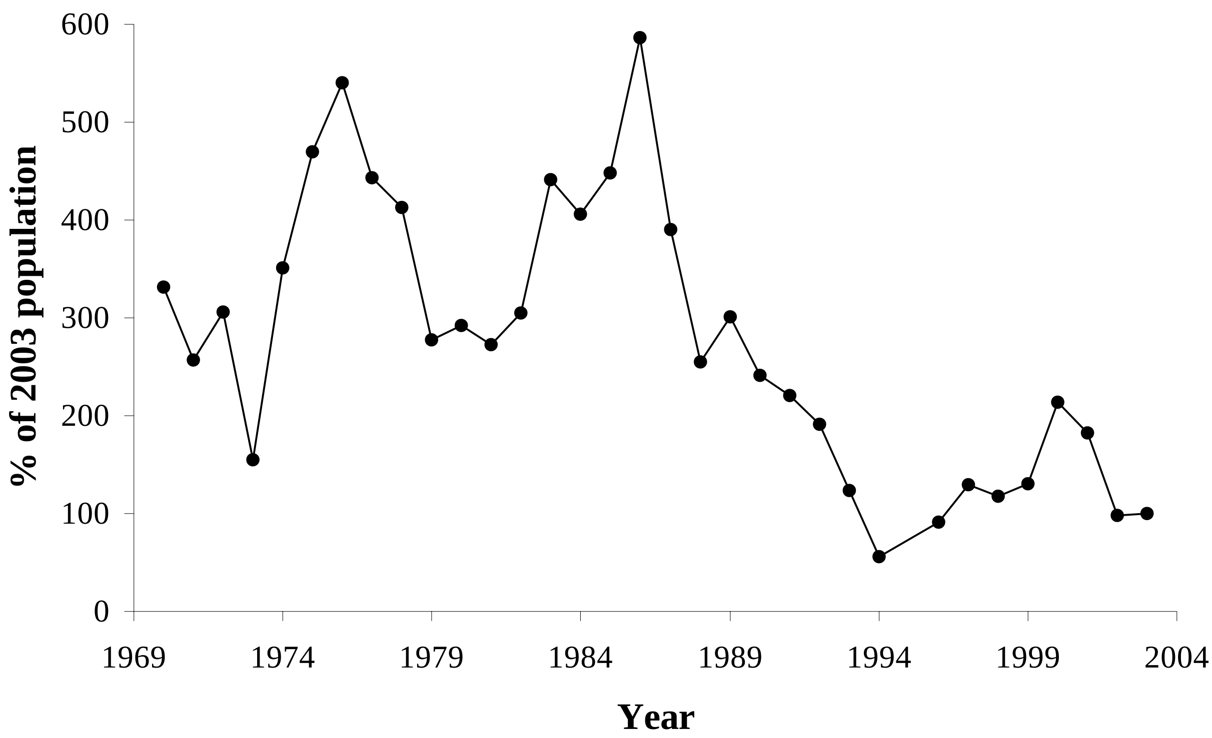


Table A4.19. Sage-grouse monitoring and population trends in Northern Montana population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	156	63	25	29	14	18	2	10
Number of active leks[1]	114	36	18	27	14	18	2	9
Percent active leks	73	58	73	94	100	100	100	96
Average males/lek	19	9	12	23	27	18	28	29
Median males/lek	13	3	10	20	25	16	28	24
Average males/active lek	26	15	17	24	27	18	28	30
Median males/active lek	21	11	14	21	25	16	28	24

[1] Averaged over each year for each period.

Fig. A4.19. Change in the population index for Northern MT population, 1966-2003.

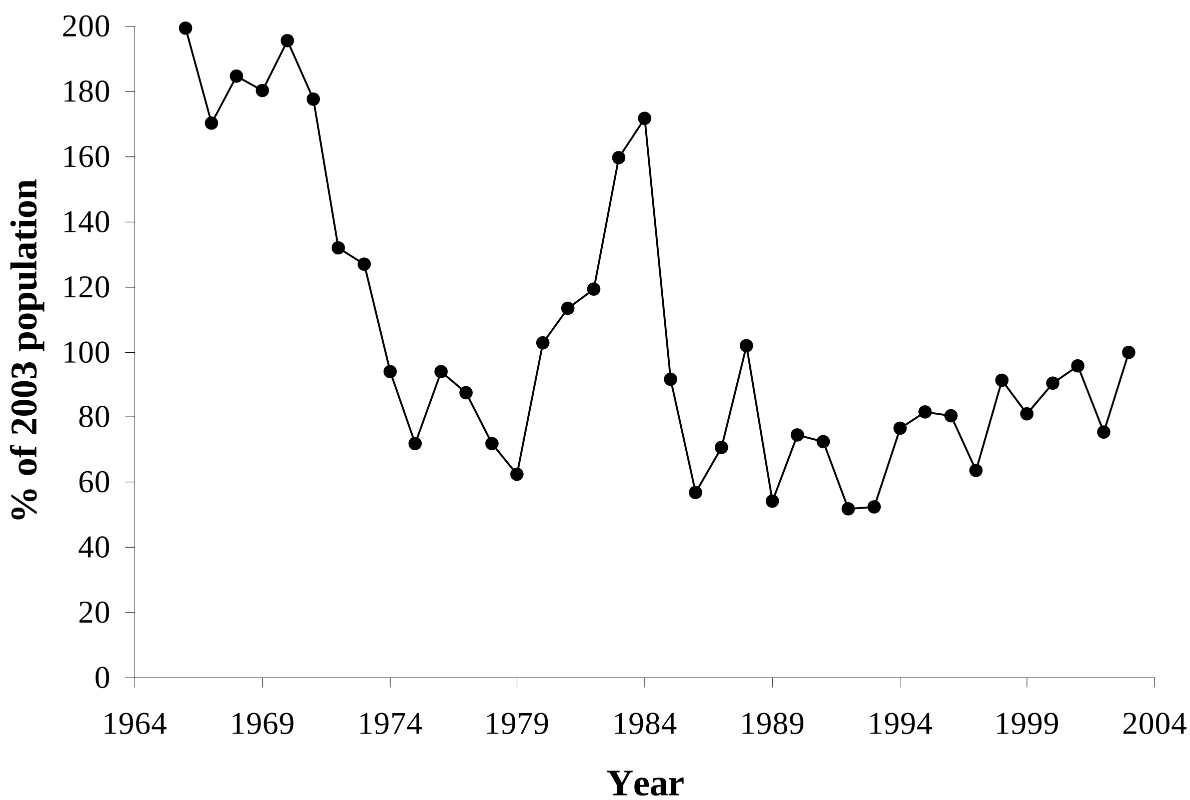


Table A4.20. Sage-grouse monitoring and population trends in NW-Interior NV population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	11	4	4	0	0	0	0	0
Number of active leks[1]	7	2	4	0	0	0	0	0
Percent active leks	64	57	100		100			
Average males/lek	9	8	11		4			
Median males/lek	2	1	11		4			
Average males/active lek	14	15	11		4			
Median males/active lek	7	7	11		4			

[1] Averaged over each year for each period.

Fig. A4.20. Change in the population index for NW-Interior NV population, 1992-2002.

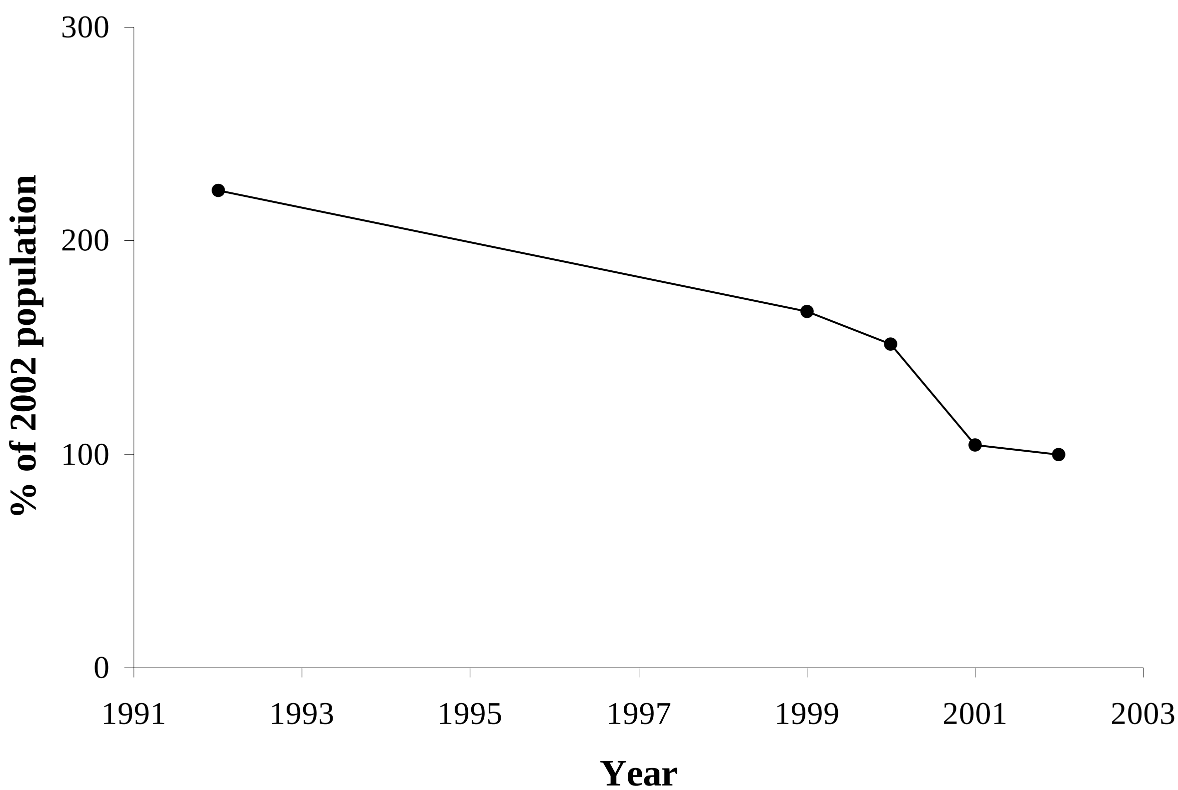


Table A4.21. Sage-grouse monitoring and population trends in Piceance CO population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	3	1	5	7	6	0	1	1
Number of active leks[1]	2	1	0	0	5	0	1	0
Percent active leks	80	100	0	0	87		100	67
Average males/lek	7	4	0	0	7		16	2
Median males/lek	6	3	0	0	5		9	2
Average males/active lek	9	4			8		16	3
Median males/active lek	8	3			6		9	3

[1] Averaged over each year for each period.

Fig. A4.21. Change in the population index for Piceance CO population, 1971-2003.

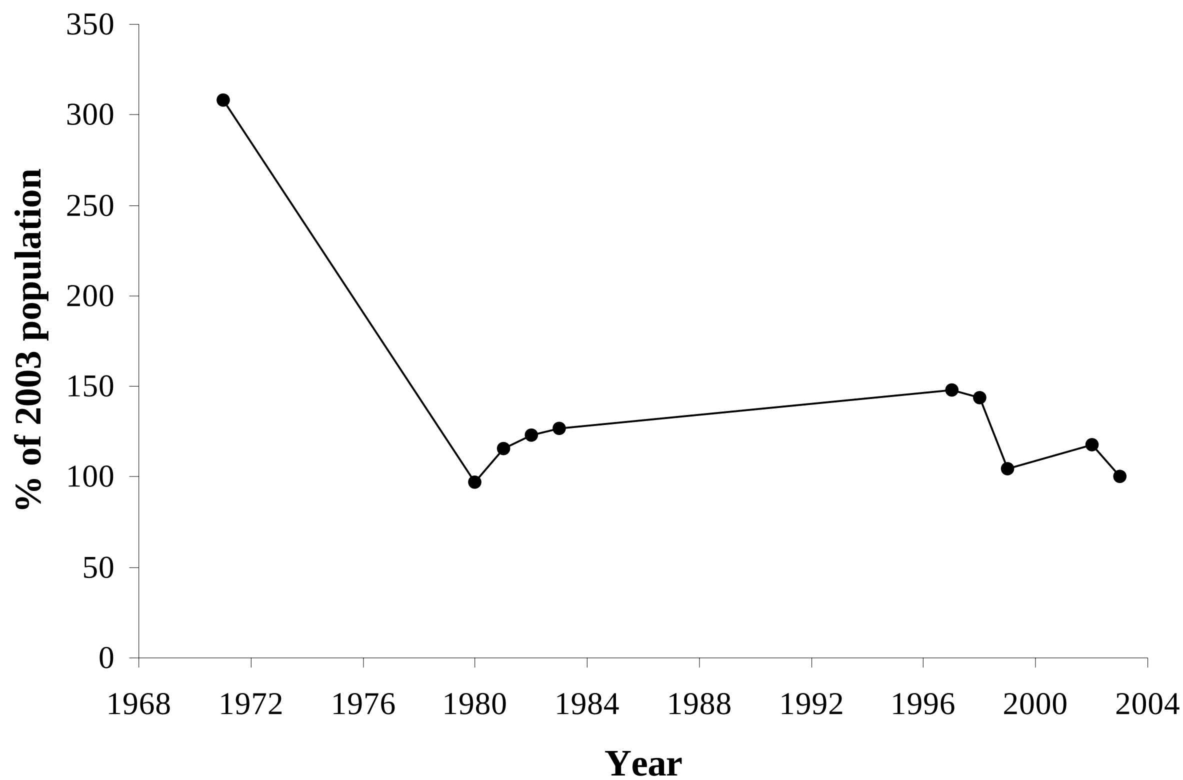


Table A4.22. Sage-grouse monitoring and population trends in Pine Nut NV population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	1	0	0	0	0	0	0	0
Number of active leks[1]	1	0	0	0	0	0	0	0
Percent active leks	100							
Average males/lek	21							
Median males/lek	21							
Average males/active lek	21							
Median males/active lek	21							

[1] Averaged over each year for each period.

Table A4.23. Sage-grouse monitoring and population trends in Quinn Canyon Range NV population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	0	0	0	0	0	0	0	0
Number of active leks[1]	0	0	0	0	0	0	0	0
Percent active leks								
Average males/lek								
Median males/lek								
Average males/active lek								
Median males/active lek								

[1] Averaged over each year for each period.

Table A4.24. Sage-grouse monitoring and population trends in Red Rock MT population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	13	2	1	1	1	1	1	1
Number of active leks[1]	10	1	1	1	1	1	1	1
Percent active leks	76	50	83	100	100	100	100	100
Average males/lek	11	11	19	47	71	57	73	104
Median males/lek	7	3	16	43	62	69	70	100
Average males/active lek	15	21	22	47	71	57	73	104
Median males/active lek	12	22	17	43	62	69	70	100

[1] Averaged over each year for each period.

Fig. A4.22. Change in the population index for Red Rock MT population, 1965-2003.

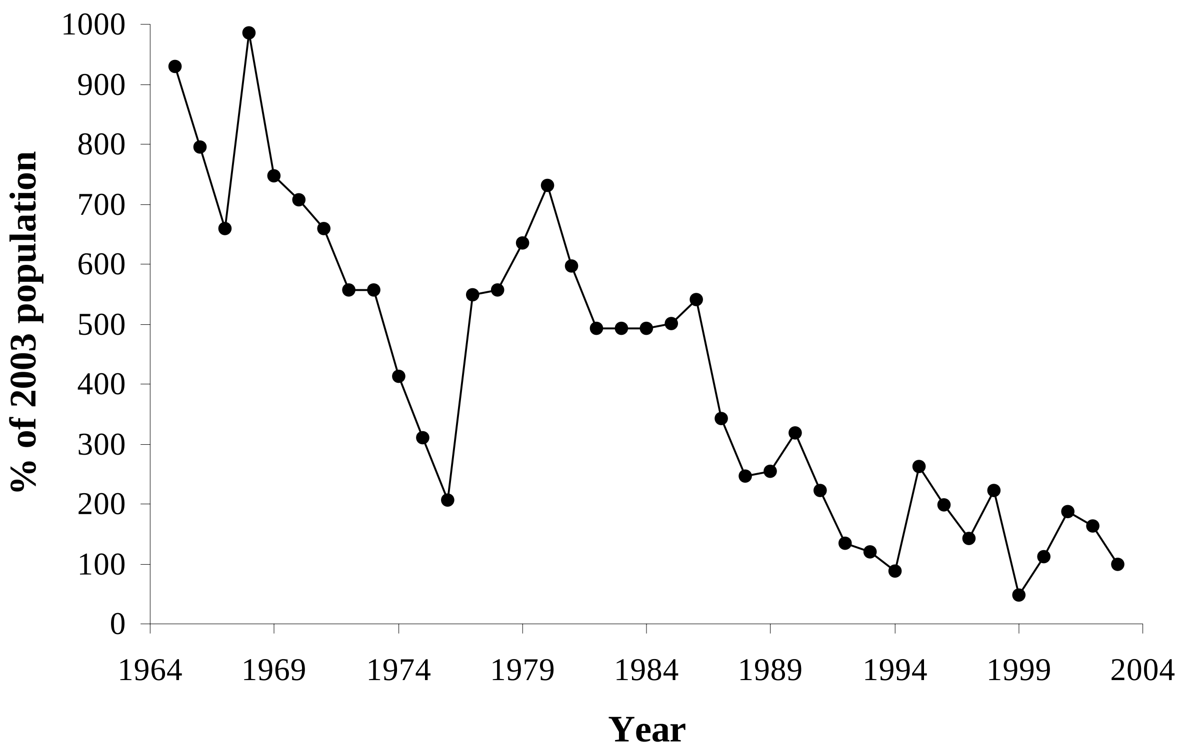


Table A4.25. Sage-grouse monitoring and population trends in S Mono Lake CA population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	9	8	9	7	6	7	6	5
Number of active leks[1]	8	8	8	7	6	5	5	4
Percent active leks	91	95	89	97	97	82	75	81
Average males/lek	29	24	24	45	20	16	20	13
Median males/lek	14	21	12	28	16	19	14	7
Average males/active lek	32	26	27	47	20	20	26	16
Median males/active lek	15	22	15	29	16	22	21	7

[1] Averaged over each year for each period.

Fig. A4.23. Change in the population index for S Mono Lake CA population, 1965-2003.

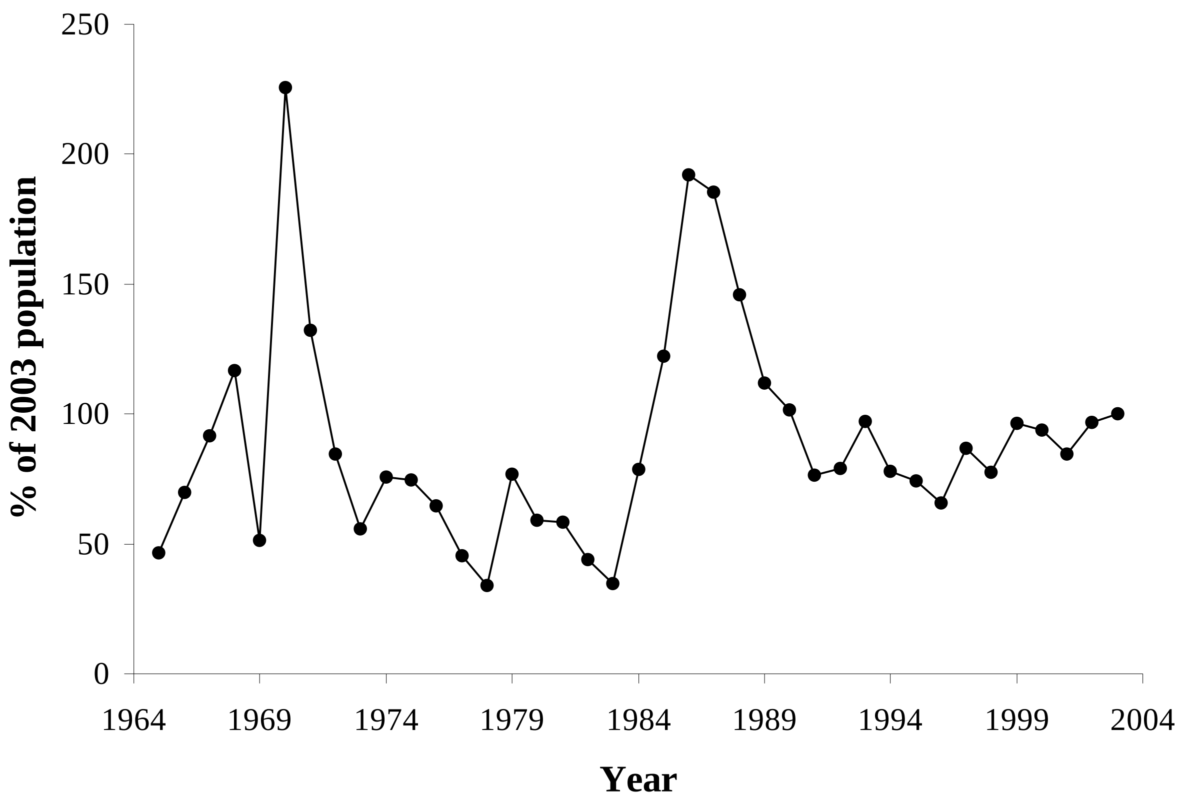


Table A4.26. Sage-grouse monitoring and population trends in S White River UT population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	1	1	2	2	0	0	0	0
Number of active leks[1]	1	1	2	2	0	0	0	0
Percent active leks	100	71	100	100	100			
Average males/lek	19	21	16	27	19			
Median males/lek	19	28	16	27	19			
Average males/active lek	19	30	16	27	19			
Median males/active lek	19	29	16	27	19			

[1] Averaged over each year for each period.

Fig. A4.24. Change in the population index for S White River UT population, 1983-2000.

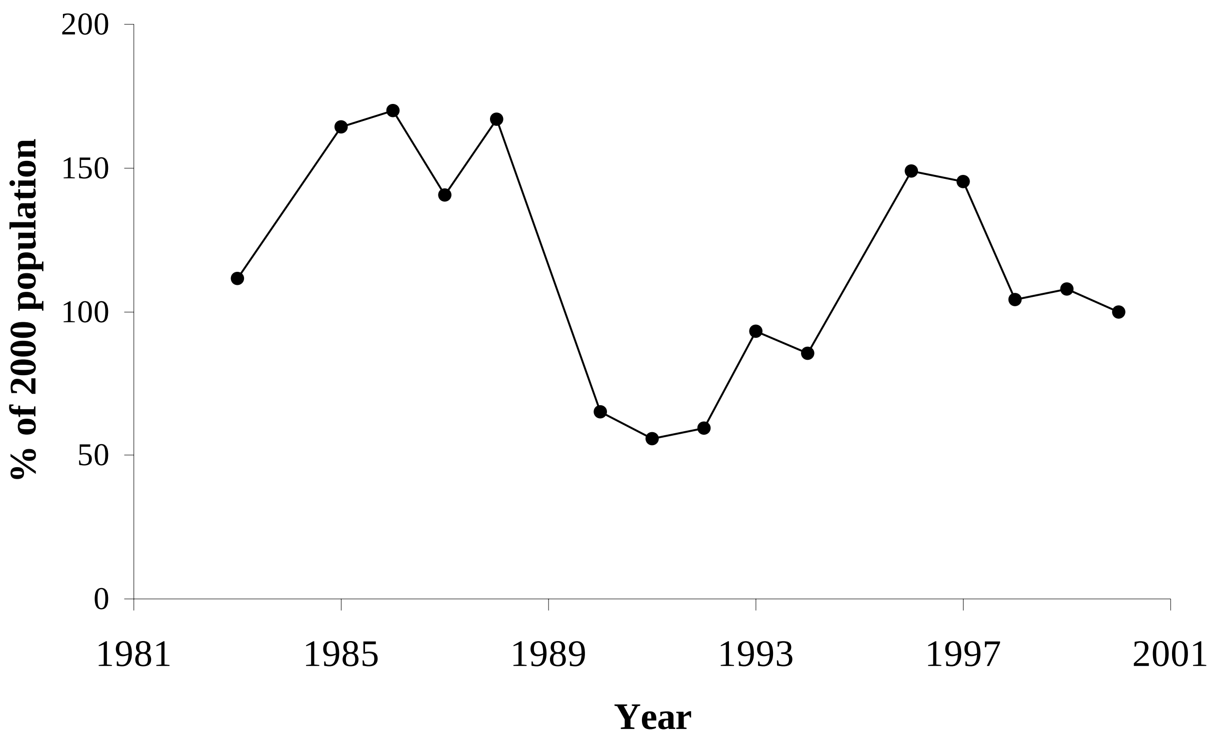


Table A4.27. Sage-grouse monitoring and population trends in Sanpete/Emery UT population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	3	3	3	2	0	1	2	1
Number of active leks[1]	1	1	3	2	0	1	1	1
Percent active leks	36	33	81	100	100	100	50	100
Average males/lek	3	5	5	9	9	15	2	7
Median males/lek	0	0	3	10	9	15	1	8
Average males/active lek	9	16	6	9	9	15	4	7
Median males/active lek	10	15	3	10	9	15	4	8

[1] Averaged over each year for each period.

Fig. A4.25. Change in the population index for Sanpete/Emery UT population, 1968-2003.

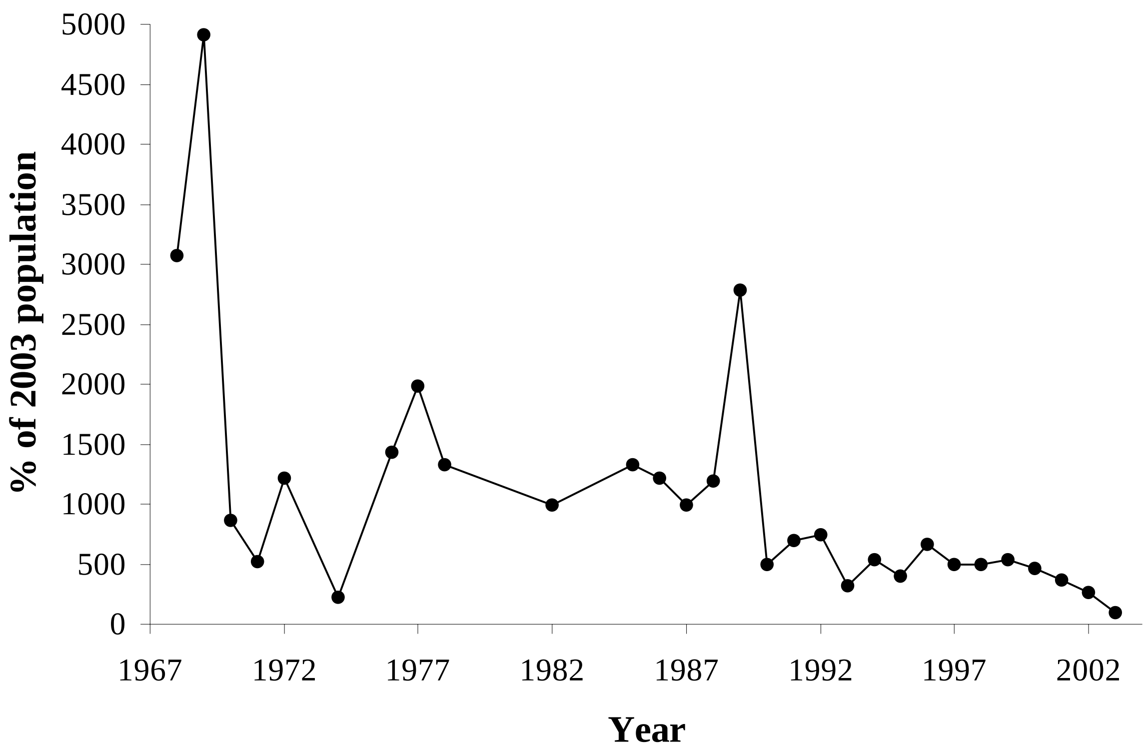


Table A4.28. Sage-grouse monitoring and population trends in Sawtooth ID population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	0	0	1	2	3	0	0	0
Number of active leks[1]	0	0	1	1	2	0	0	0
Percent active leks			75	33	53			
Average males/lek			3	1	4			
Median males/lek			3	0	1			
Average males/active lek			4	2	8			
Median males/active lek			3	2	5			

[1] Averaged over each year for each period.

Fig. A4.26. Change in the population index for Sawtooth ID population, 1980-1999.

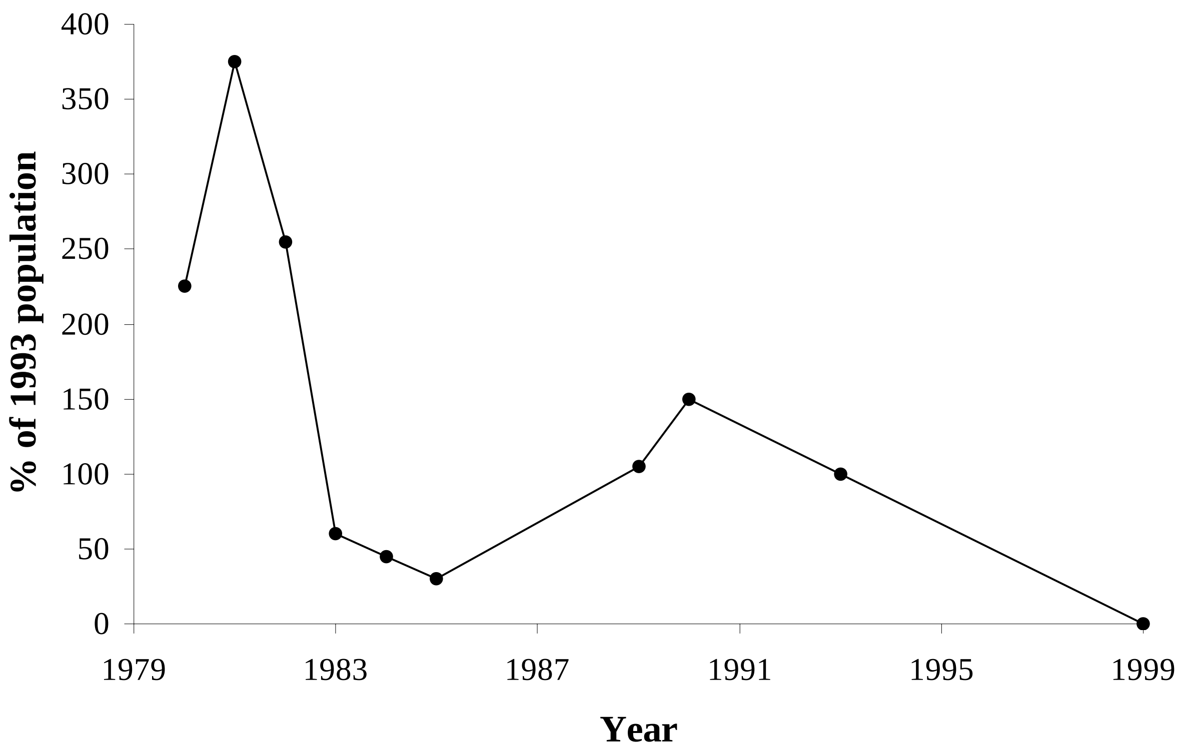


Table A4.29. Sage-grouse monitoring and population trends in S-Central UT population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	28	29	19	23	23	28	24	10
Number of active leks[1]	23	22	14	17	19	21	21	9
Percent active leks	82	74	77	77	81	77	86	90
Average males/lek	30	19	21	21	21	19	27	33
Median males/lek	17	10	17	14	15	12	16	21
Average males/active lek	36	25	28	28	26	24	31	37
Median males/active lek	27	16	25	24	19	16	26	26

[1] Averaged over each year for each period.

Fig. A4.27. Change in the population index for S-Central UT population, 1967-2003.

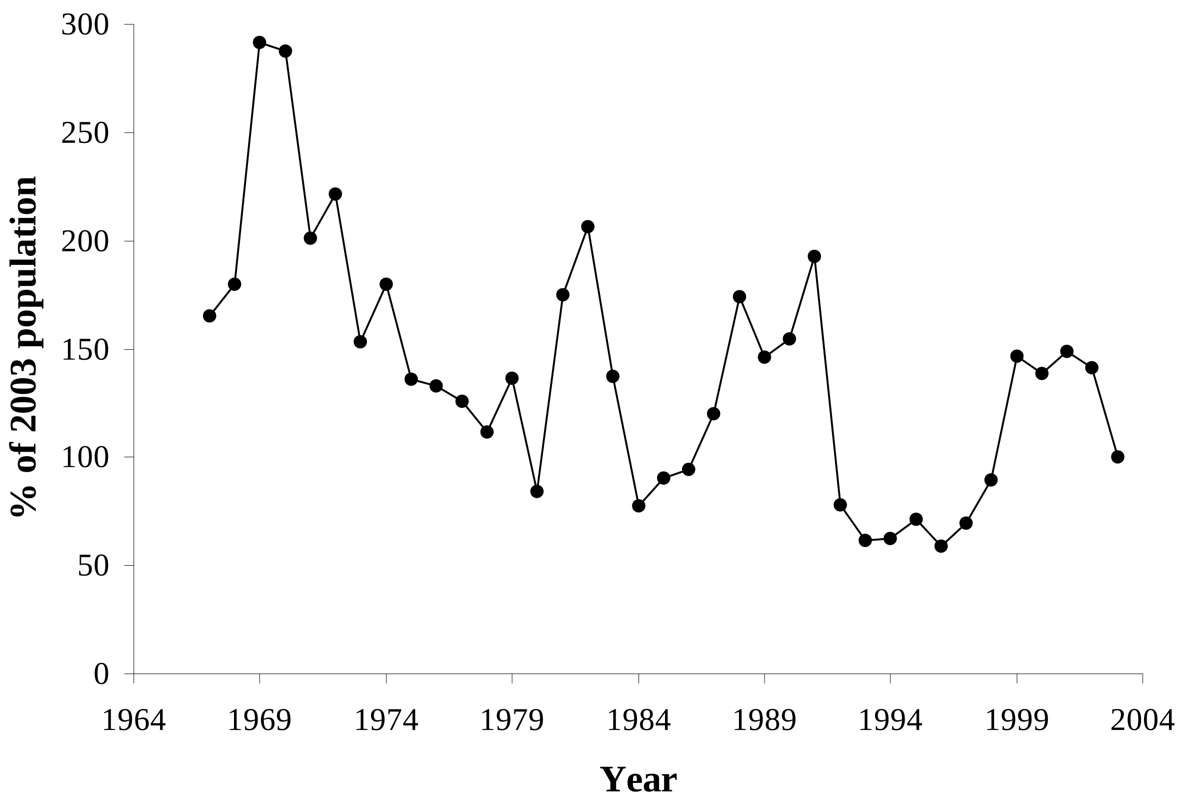


Table A4.30. Sage-grouse monitoring and population trends in Snake, Salmon, and Beaverhead population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	171	130	95	80	100	98	86	64
Number of active leks[1]	129	92	70	70	83	87	77	61
Percent active leks	76	71	74	88	83	89	90	94
Average males/lek	25	18	21	35	22	34	36	44
Median males/lek	17	10	12	22	13	21	27	34
Average males/active lek	33	25	29	39	26	38	40	47
Median males/active lek	25	18	19	26	18	26	31	35

[1] Averaged over each year for each period.

Fig. A4.28. Change in the population index for Snake, Salmon, and Beaverhead population, 1965-2003.

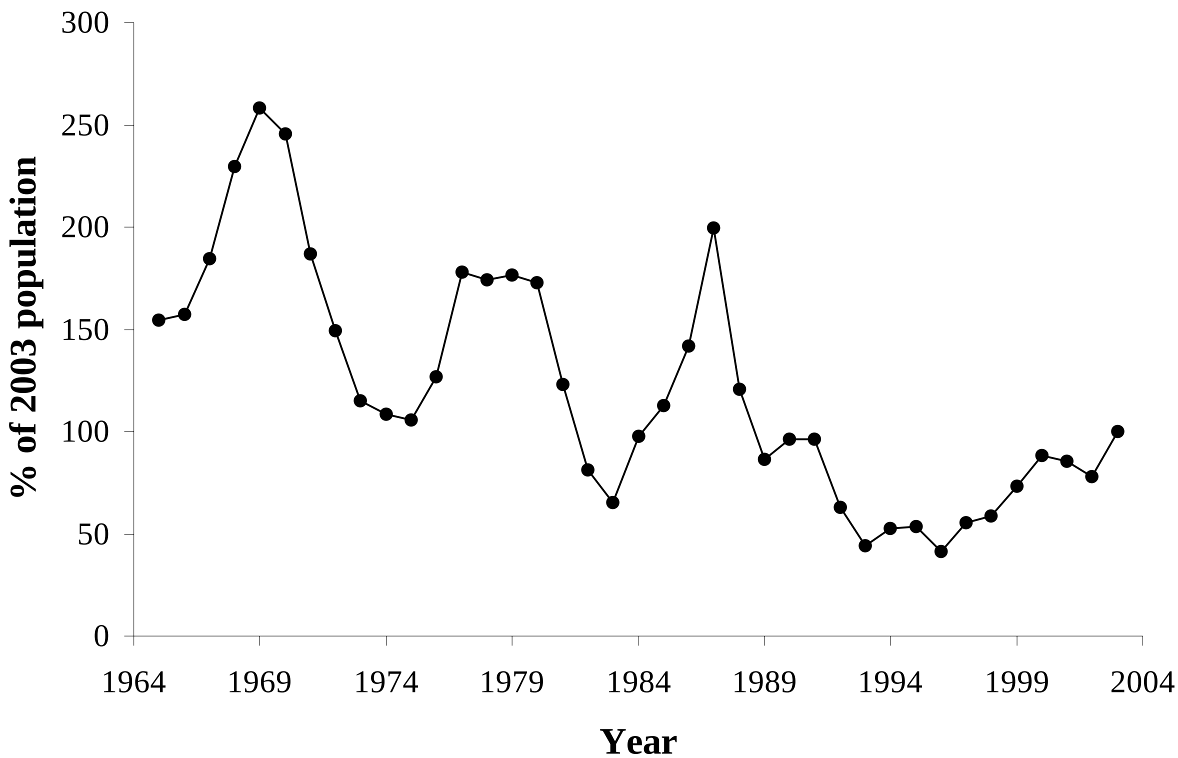


Table A4.31. Sage-grouse monitoring and population trends in Summit/Morgan UT population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	6	2	3	5	5	8	7	5
Number of active leks[1]	3	1	3	3	4	6	7	4
Percent active leks	54	67	87	56	85	82	100	78
Average males/lek	12	25	14	7	16	22	31	19
Median males/lek	1	24	8	1	12	16	30	17
Average males/active lek	21	37	16	12	19	26	31	25
Median males/active lek	14	30	14	13	15	22	30	18

[1] Averaged over each year for each period.

Fig. A4.29. Change in the population index for Summit/Morgan UT population, 1965-2003.

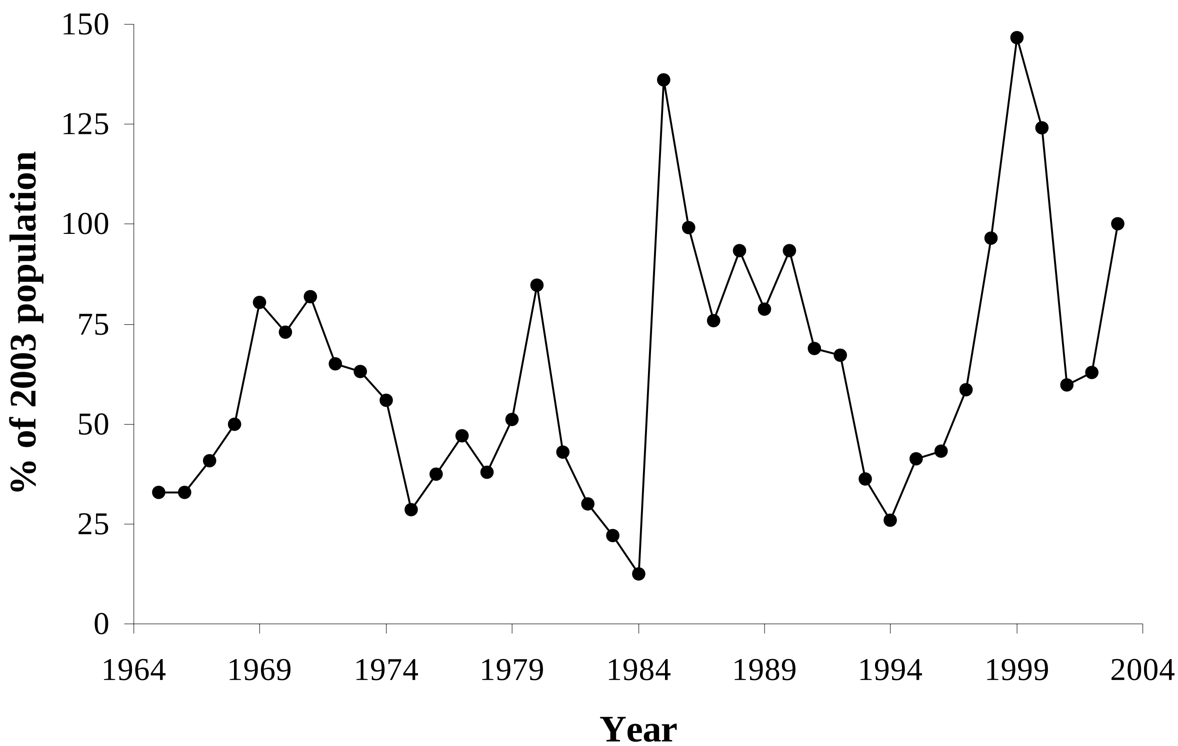


Table A4.32. Sage-grouse monitoring and population trends in Tooele/Juab UT population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	4	2	3	4	3	4	4	1
Number of active leks[1]	3	2	3	4	2	3	4	1
Percent active leks	81	73	81	82	71	77	100	100
Average males/lek	28	13	14	16	7	18	25	28
Median males/lek	25	8	10	10	3	12	20	26
Average males/active lek	34	18	18	19	11	23	25	28
Median males/active lek	32	12	15	13	7	17	20	26

[1] Averaged over each year for each period.

Fig. A4.30. Change in the population index for Tooele/Juab UT population, 1968-2003.

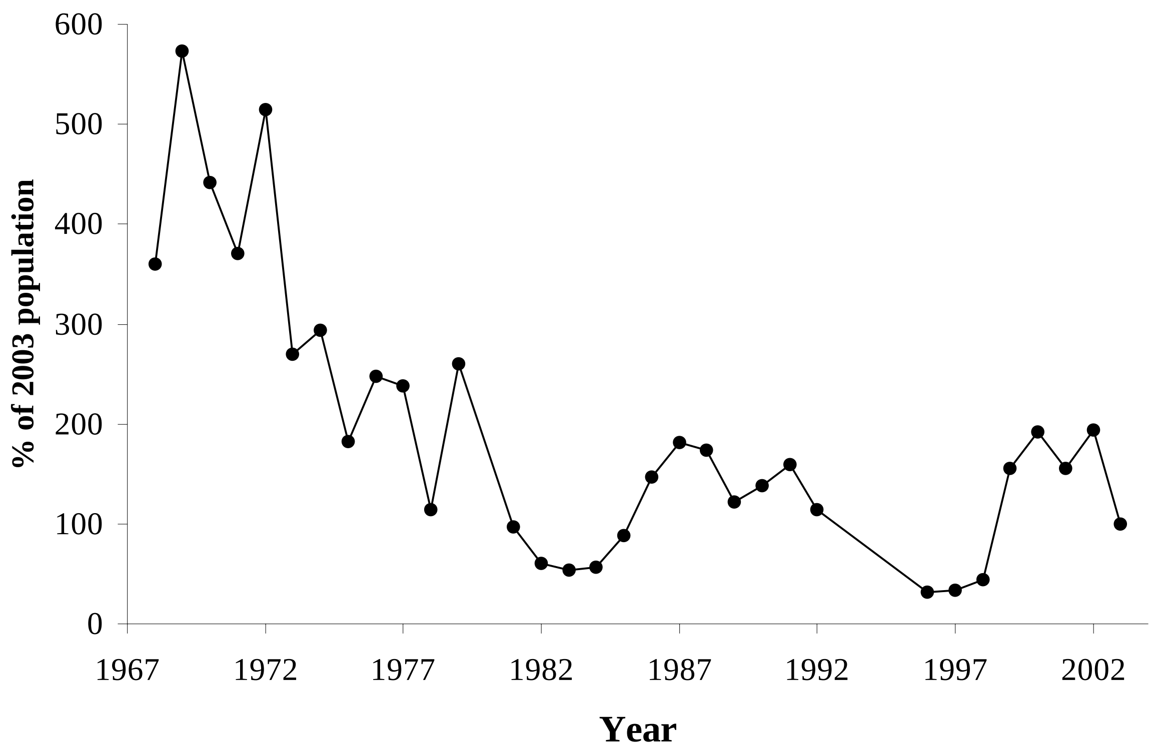


Table A4.33. Sage-grouse monitoring and population trends in Twin Bridges MT subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	1	0	0	0	0	0	0	0
Number of active leks[1]	0	0	0	0	0	0	0	0
Percent active leks	50	0						
Average males/lek	6	0						
Median males/lek	6	0						
Average males/active lek	11							
Median males/active lek	11							

[1] Averaged over each year for each period.

Table A4.34. Sage-grouse monitoring and population trends in Warm Springs Valley NV population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	0	0	0	0	0	0	0	0
Number of active leks[1]	0	0	0	0	0	0	0	0
Percent active leks	100							
Average males/lek	24							
Median males/lek	24							
Average males/active lek	24							
Median males/active lek	24							

[1] Averaged over each year for each period.

Table A4.35. Sage-grouse monitoring and population trends in Weiser ID population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	11	5	2	1	1	3	3	0
Number of active leks[1]	11	5	2	1	1	3	3	0
Percent active leks	100	100	100	100	100	100	100	
Average males/lek	25	21	47	33	25	21	13	
Median males/lek	23	17	35	19	23	24	13	
Average males/active lek	25	21	47	33	25	21	13	
Median males/active lek	23	17	35	19	23	24	13	

[1] Averaged over each year for each period.

Fig. A4.31. Change in the population index for Weiser ID population, 1973-2002.

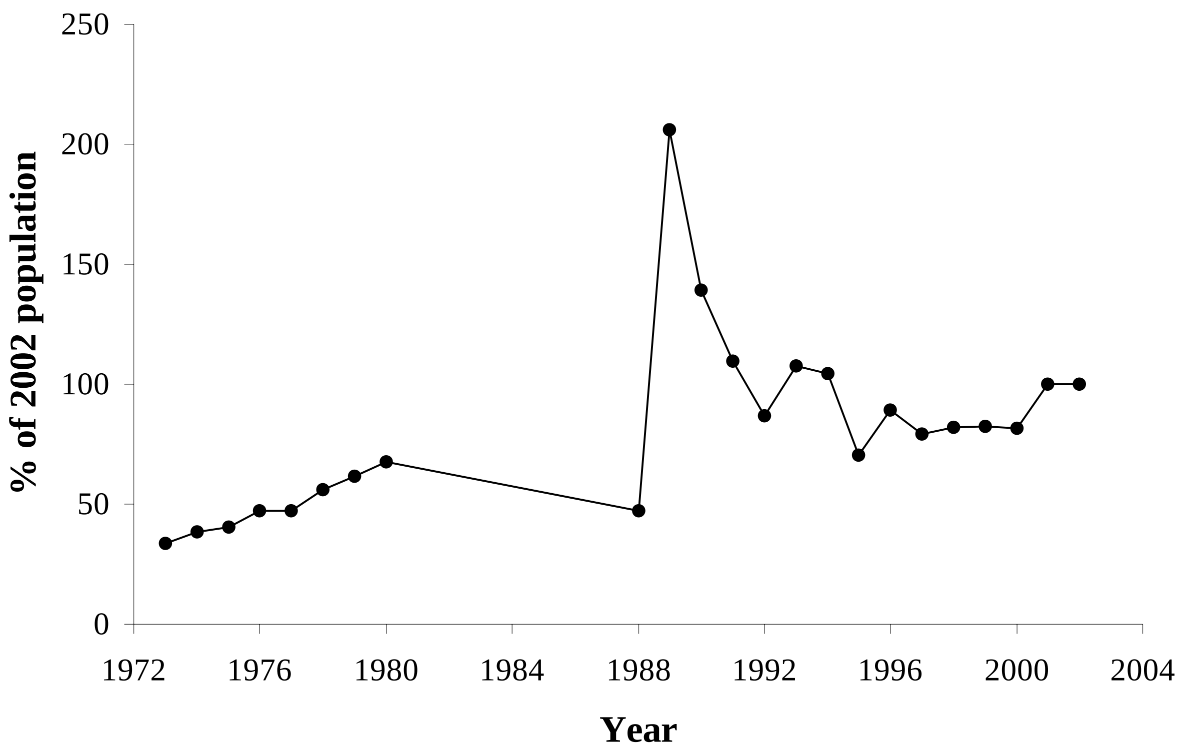


Table A4.36. Sage-grouse monitoring and population trends in White Mountains NV/CA population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	0	0	0	0	0	0	0	0
Number of active leks[1]	0	0	0	0	0	0	0	0
Percent active leks								
Average males/lek								
Median males/lek								
Average males/active lek								
Median males/active lek								

[1] Averaged over each year for each period.

Table A4.37. Sage-grouse monitoring and population trends in White River CO population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	1	0	0	0	0	0	0	0
Number of active leks[1]	1	0	0	0	0	0	0	0
Percent active leks	100							
Average males/lek	29							
Median males/lek	29							
Average males/active lek	29							
Median males/active lek	29							

[1] Averaged over each year for each period.

Table A4.38. Sage-grouse monitoring and population trends in Wisdom MT population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	3	0	0	0	0	0	0	0
Number of active leks[1]	3	0	0	0	0	0	0	0
Percent active leks	100							
Average males/lek	23							
Median males/lek	18							
Average males/active lek	23							
Median males/active lek	18							

[1] Averaged over each year for each period.

Fig. A4.32. Change in the population index for Wisdom MT population, 2000-2003.

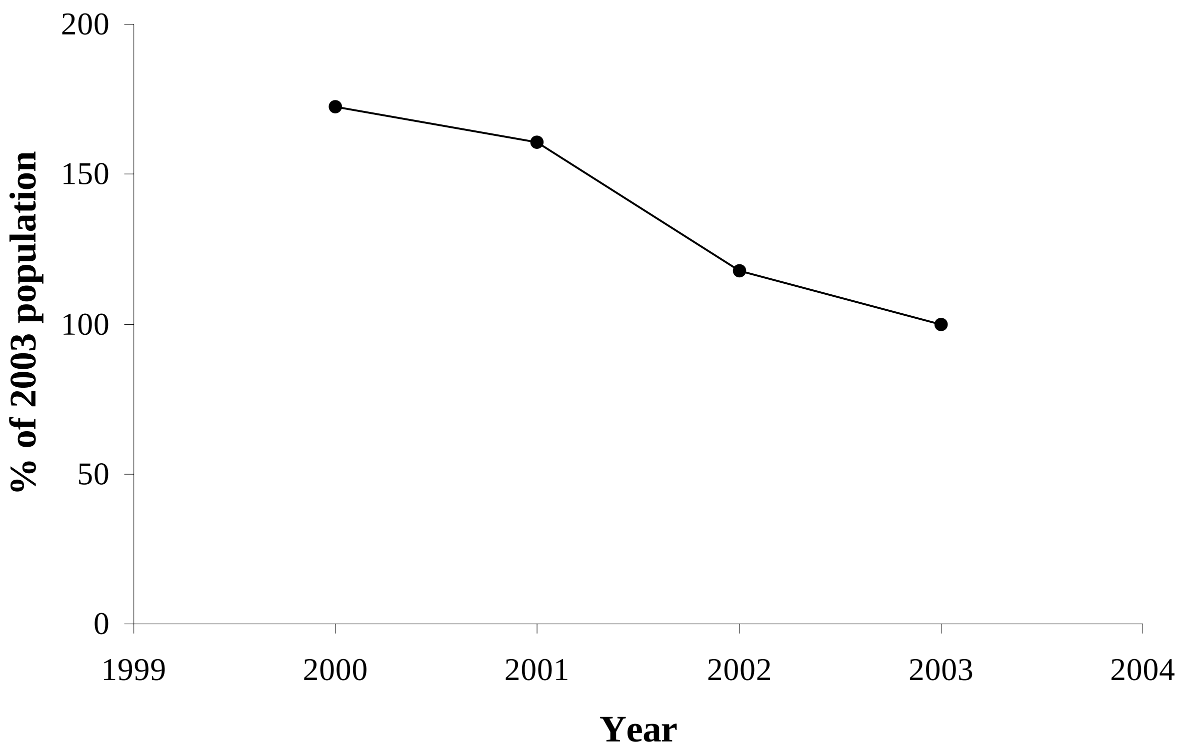


Table A4.39. Sage-grouse monitoring and population trends in Wyoming Basin population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	1193	832	682	584	441	224	144	138
Number of active leks[1]	842	538	454	428	339	173	114	103
Percent active leks	71	65	67	73	77	77	79	75
Average males/lek	19	14	15	17	22	29	28	36
Median males/lek	11	7	8	11	14	18	19	24
Average males/active lek	27	21	23	24	29	38	35	48
Median males/active lek	20	14	16	18	21	27	26	34

[1] Averaged over each year for each period.

Fig. A4.33. Change in the population index for Wyoming Basin population, 1965-2003.

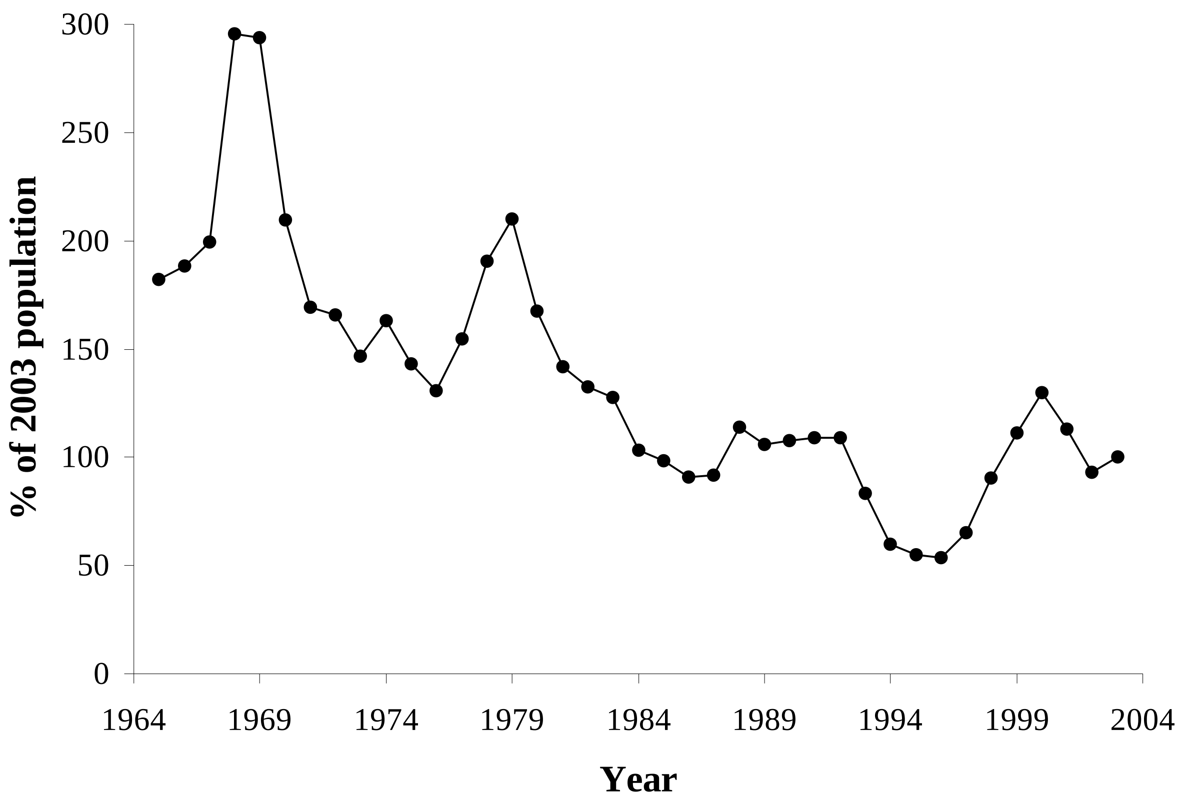


Table A4.40. Sage-grouse monitoring and population trends in Yakima WA population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	10	7	7	4	3	2	1	0
Number of active leks[1]	8	6	5	4	3	2	1	0
Percent active leks	76	86	73	95	94	100	100	
Average males/lek	13	16	18	29	44	19	12	
Median males/lek	14	14	15	27	28	13	12	
Average males/active lek	18	19	25	31	47	19	12	
Median males/active lek	18	17	23	27	30	13	12	

[1] Averaged over each year for each period.

Fig. A4.34. Change in the population index for Yakima WA population, 1970-2003.

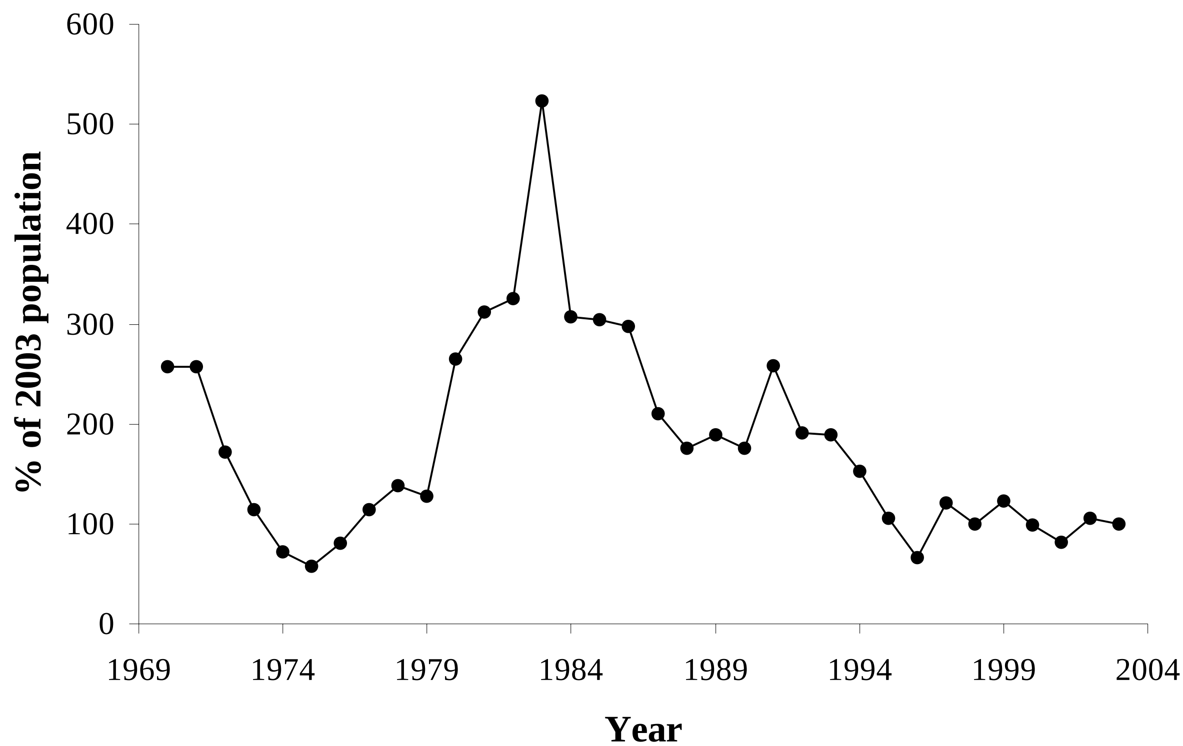
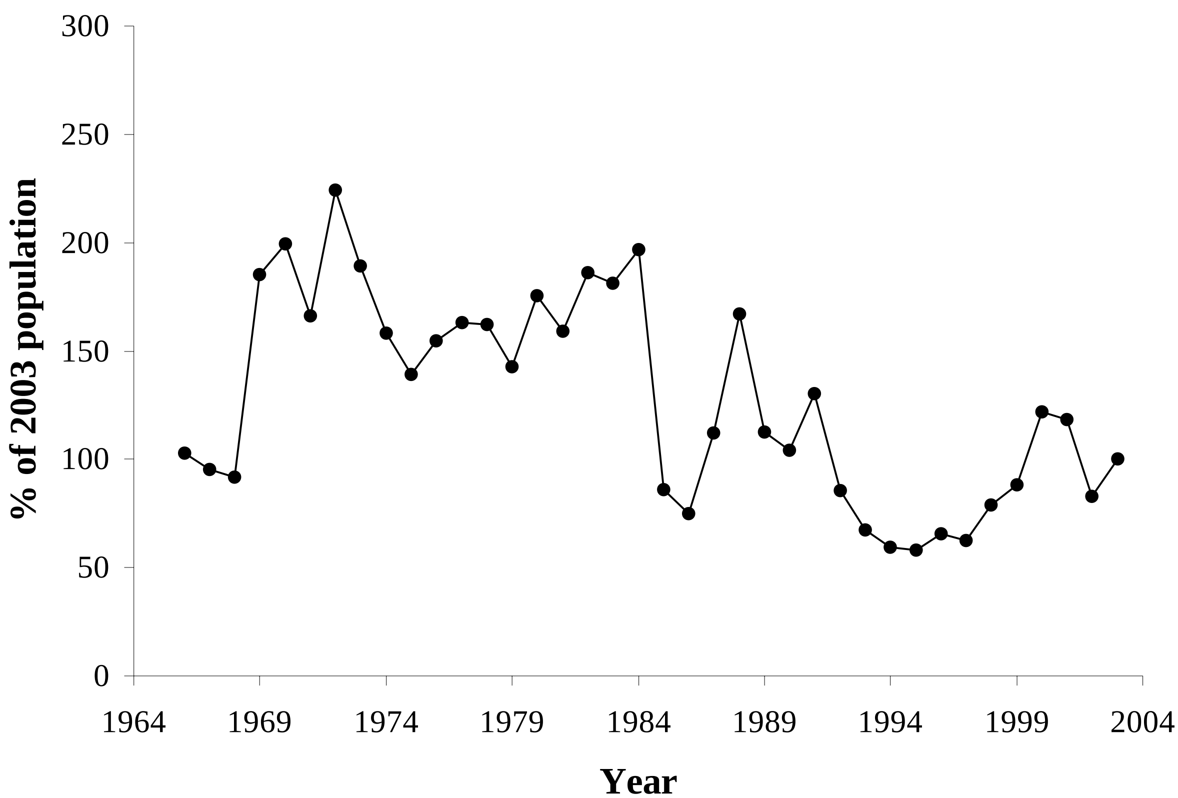


Table A4.41. Sage-grouse monitoring and population trends in Yellowstone Watershed population, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	341	130	129	140	129	88	54	8
Number of active leks[1]	236	89	95	109	116	80	49	8
Percent active leks	69	68	74	78	91	91	91	98
Average males/lek	15	13	14	16	24	21	22	23
Median males/lek	9	8	9	11	19	17	18	15
Average males/active lek	22	19	19	20	26	24	25	23
Median males/active lek	16	15	13	15	22	19	20	15

[1] Averaged over each year for each period.

Fig. A4.35. Change in the population index for Yellowstone Watershed population, 1965-2003.



APPENDIX 5

Characteristics of Greater Sage-Grouse Subpopulations

Methods

Five populations were further divided into an addition 24 subpopulations based on their large size, expansive distribution, differences in region, and a relatively small degree of separation (Table 6.16). The same methods used to define and describe populations (Appendix 4) were used here for subpopulations.

Great Basin Core Population

Table A5.1. Sage-grouse monitoring and population trends in Central NV subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	60	22	18	16	19	12	12	14
Number of active leks[1]	47	20	17	16	18	11	9	13
Percent active leks	79	91	95	98	95	90	79	94
Average males/lek	18	19	20	23	28	30	21	26
Median males/lek	13	16	16	16	21	24	11	16
Average males/active lek	22	21	21	23	30	34	27	28
Median males/active lek	17	17	18	17	22	24	15	19

[1] Averaged over each year for each period.

Fig. A5.1. Change in the population index for Central NV subpopulation, 1974-2003.

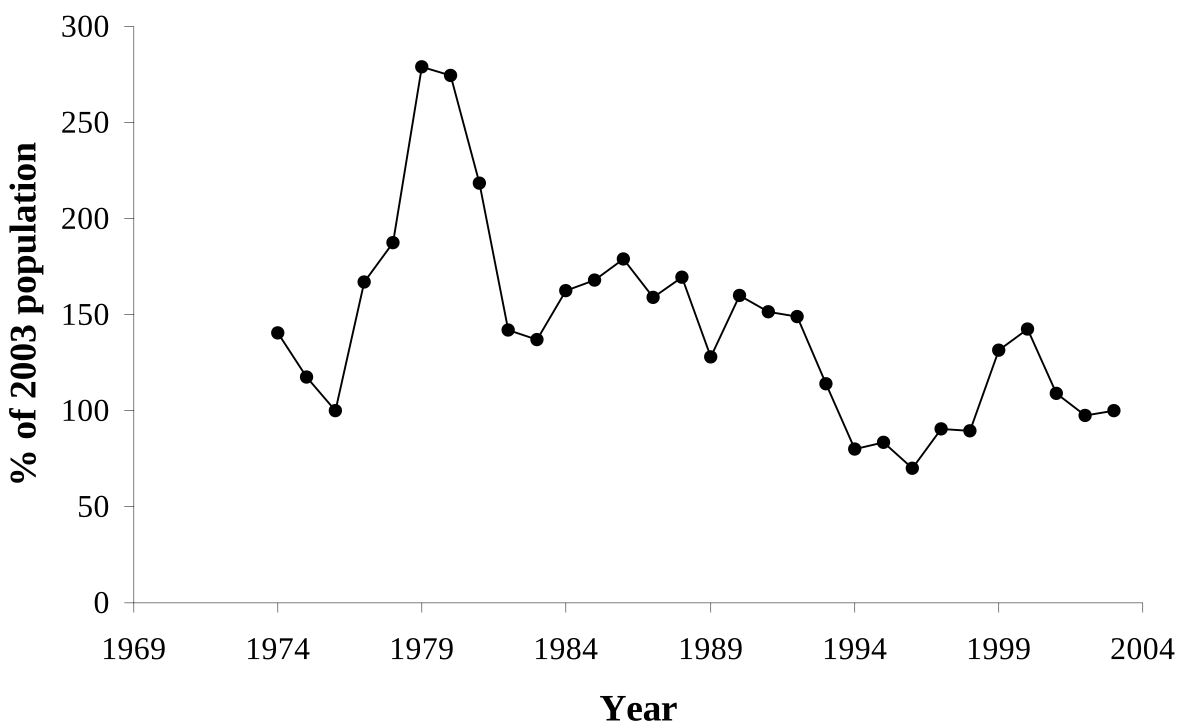


Table A5.2. Sage-grouse monitoring and population trends in E-Central OR subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	15	20	9	6	4	1	1	1
Number of active leks[1]	15	19	9	5	3	1	1	1
Percent active leks	100	92	96	81	59	100	100	100
Average males/lek	22	19	20	20	12	26	64	8
Median males/lek	16	17	17	20	10	27	54	8
Average males/active lek	22	20	21	24	20	26	64	8
Median males/active lek	16	18	19	25	20	27	54	8

[1] Averaged over each year for each period.

Fig. A5.2. Change in the population index for E-Central OR subpopulation, 1965-2003.

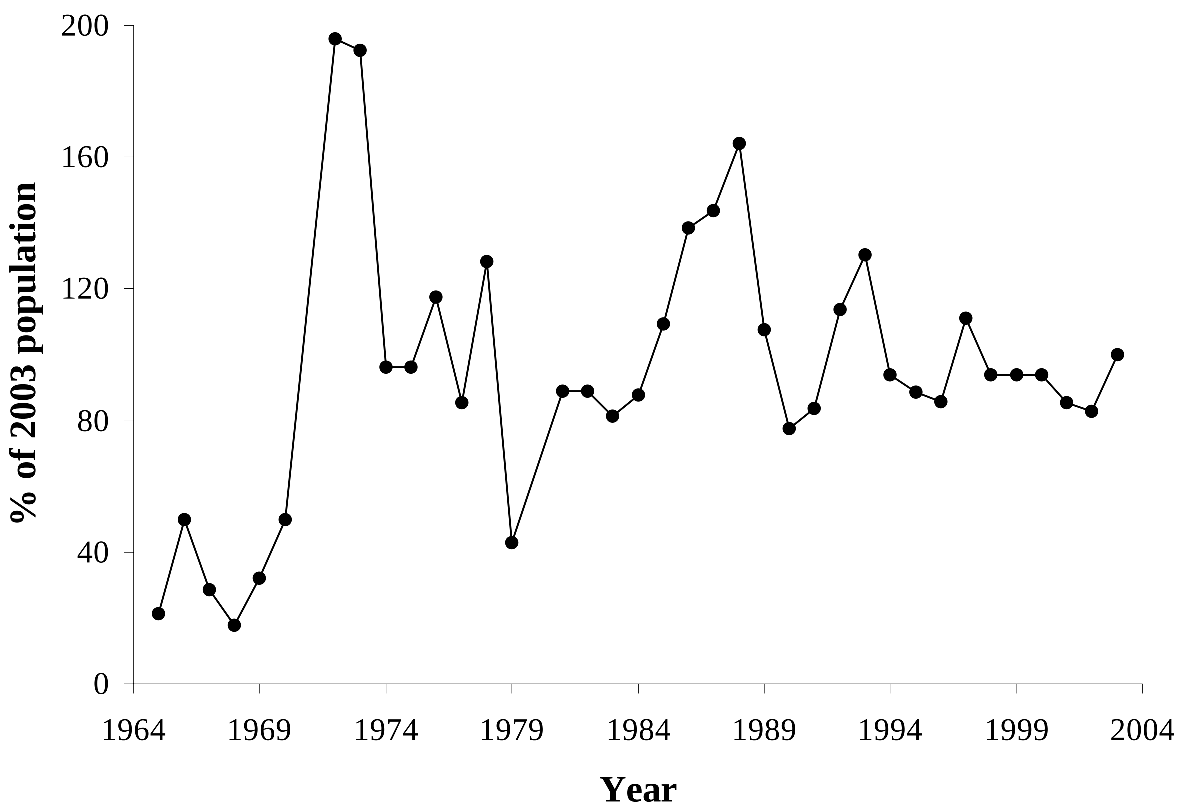


Table A5.3. Sage-grouse monitoring and population trends in Lake Area OR/NE CA/NW NV subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	88	54	40	26	14	11	20	4
Number of active leks[1]	74	45	37	24	12	4	15	3
Percent active leks	84	84	94	92	83	38	78	74
Average males/lek	30	25	44	42	29	7	26	20
Median males/lek	22	18	27	29	18	0	16	9
Average males/active lek	36	29	47	45	35	19	33	27
Median males/active lek	26	22	28	32	23	11	23	12

[1] Averaged over each year for each period.

Fig. A5.3. Change in the population index for Lake Area OR/NE CA/NW NV subpopulation, 1965-2003.

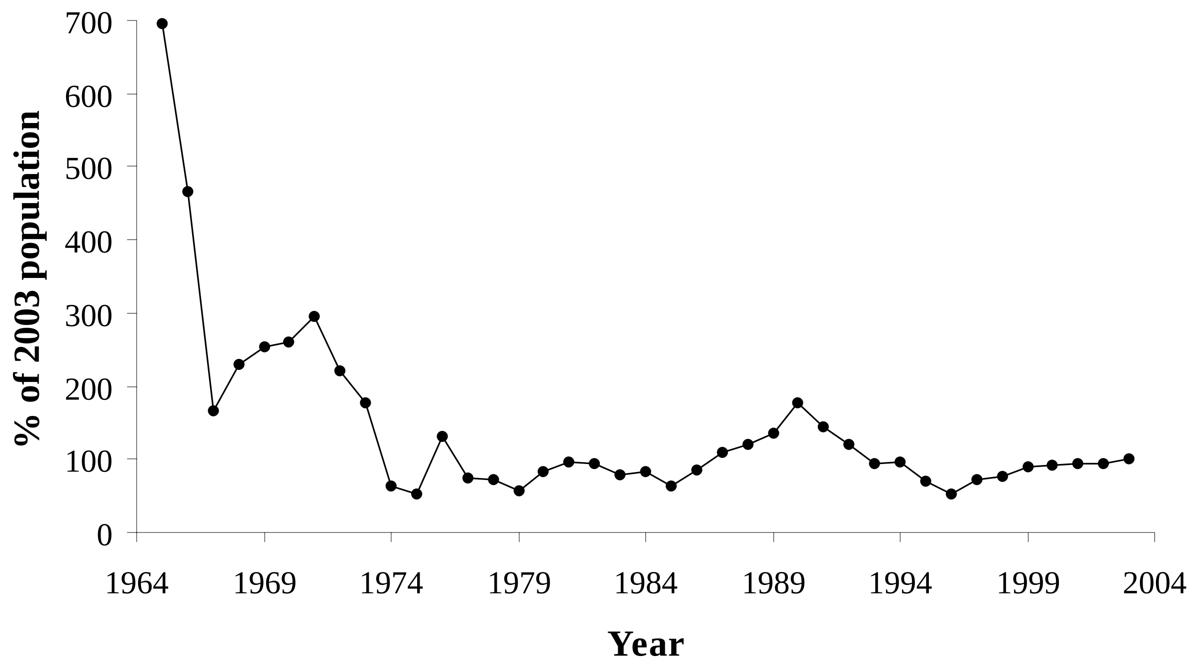


Table A5.4. Sage-grouse monitoring and population trends in N-Central NV/SE OR/SW ID subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	28	38	20	11	9	11	4	1
Number of active leks[1]	26	35	18	9	9	10	4	1
Percent active leks	93	91	92	87	96	93	95	100
Average males/lek	20	15	25	29	29	32	23	34
Median males/lek	18	9	17	24	29	22	18	28
Average males/active lek	22	16	27	33	31	35	24	34
Median males/active lek	20	11	18	29	29	28	18	28

[1] Averaged over each year for each period.

Fig. A5.4. Change in the population index for N-Central NV/SE OR/SW ID subpopulation, 1967-2003.

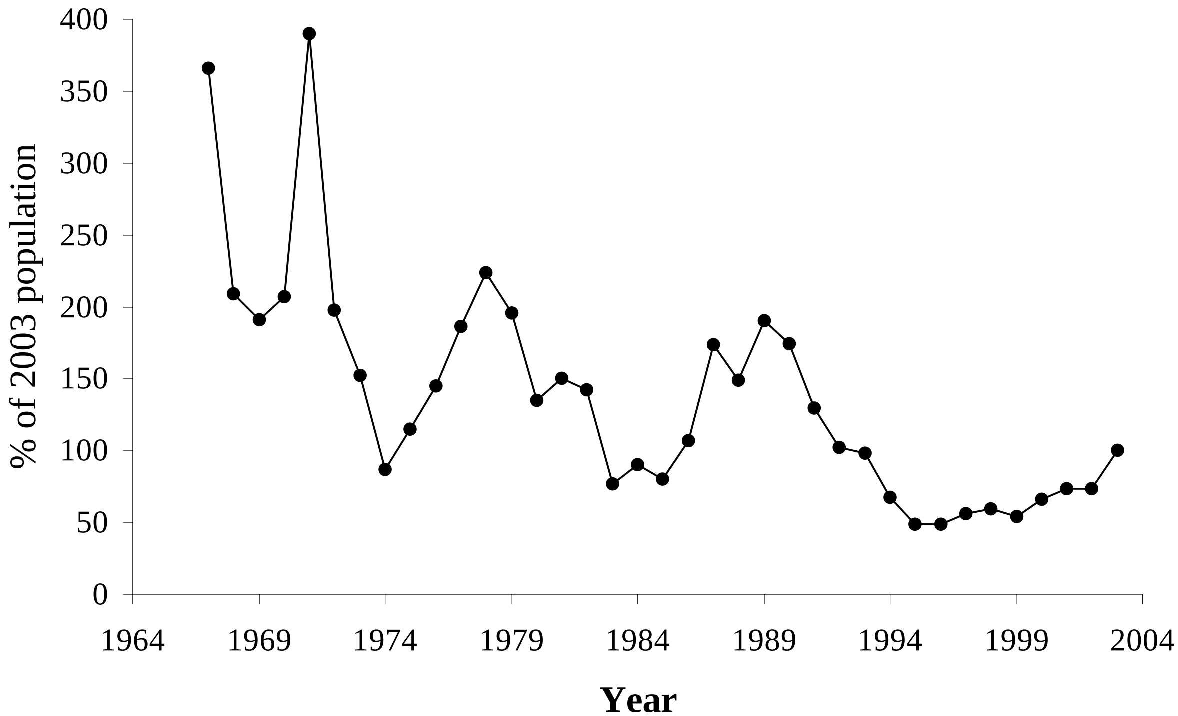


Table A5.5. Sage-grouse monitoring and population trends in NE NV/S-Central ID/NW UT subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	179	118	57	57	87	51	47	25
Number of active leks[1]	145	89	48	39	70	46	38	23
Percent active leks	81	76	84	68	81	90	81	90
Average males/lek	15	13	20	21	22	28	24	31
Median males/lek	9	7	14	10	12	17	15	21
Average males/active lek	19	17	23	31	27	31	29	34
Median males/active lek	14	12	18	24	17	20	19	24

[1] Averaged over each year for each period.

Fig. A5.5. Change in the population index for NE NV/S-Central ID/NW UT subpopulation, 1965-2003.

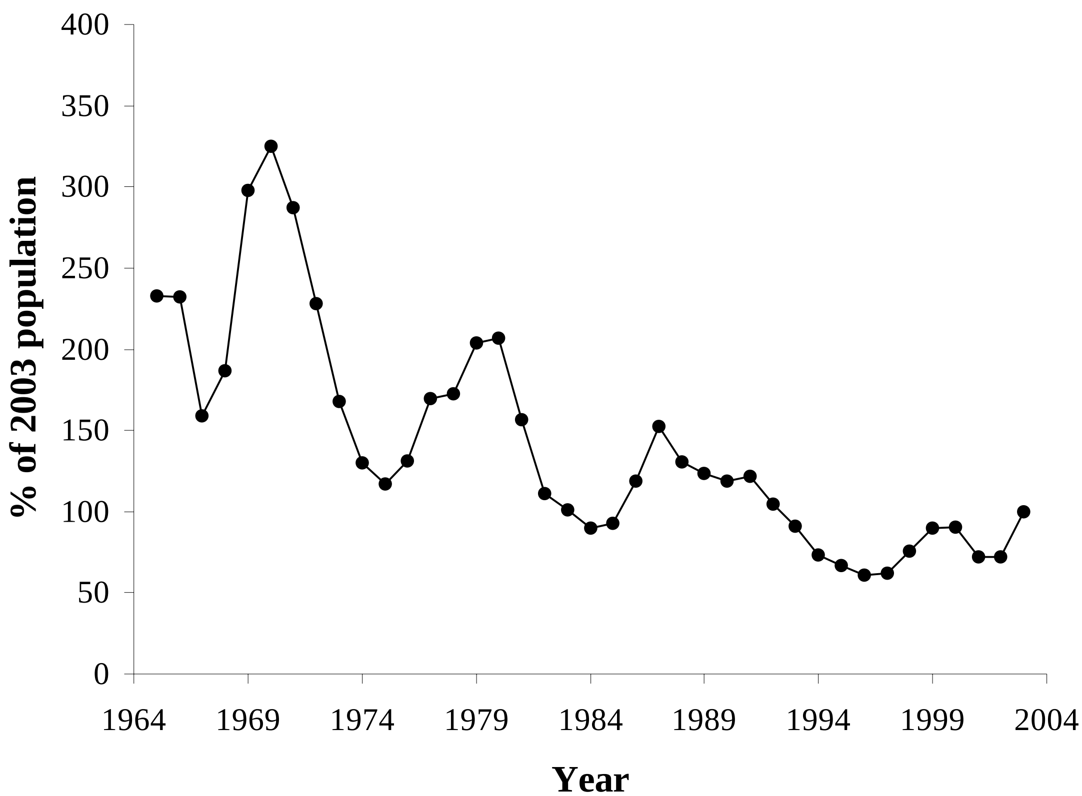


Table A5.6. Sage-grouse monitoring and population trends in S-Central OR/N-Central NV subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	1	4	1	0	0	0	0	0
Number of active leks[1]	1	4	1	0	0	0	0	0
Percent active leks	67	95	100					
Average males/lek	2	8	16					
Median males/lek	1	5	17					
Average males/active lek	3	8	16					
Median males/active lek	3	5	17					

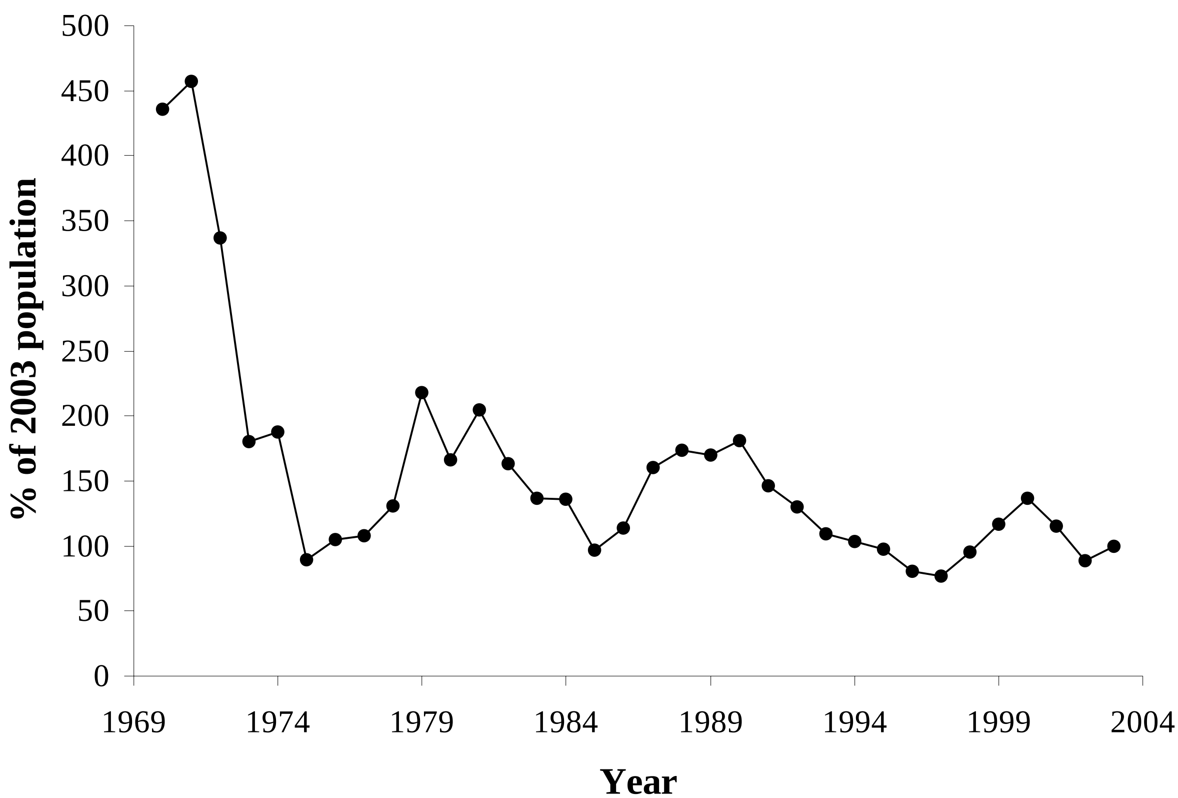
[1] Averaged over each year for each period.

Table A5.7. Sage-grouse monitoring and population trends in SE NV/SW UT subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	68	38	35	32	33	14	10	1
Number of active leks[1]	52	37	35	31	33	11	8	1
Percent active leks	77	97	98	98	99	77	88	100
Average males/lek	12	12	15	15	18	12	24	24
Median males/lek	9	9	10	12	13	7	18	22
Average males/active lek	15	12	15	16	19	16	28	24
Median males/active lek	11	10	11	13	13	12	23	22

[1] Averaged over each year for each period.

Fig. A5.6. Change in the population index for SE NV/SW UT subpopulation, 1970-2003.



Northern Montana Population

Table A5.8. Sage-grouse monitoring and population trends in AB/SK/MT subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	39	31	9	13	10	14	0	8
Number of active leks[1]	13	13	5	13	10	14	0	7
Percent active leks	33	41	57	94	100	100		95
Average males/lek	4	5	8	21	28	18		29
Median males/lek	0	0	2	21	24	15		23
Average males/active lek	12	11	14	23	28	18		31
Median males/active lek	11	8	11	22	24	15		24

[1] Averaged over each year for each period.

Fig. A5.7. Change in the population index for AB/SK/MT subpopulation, 1968-2003.

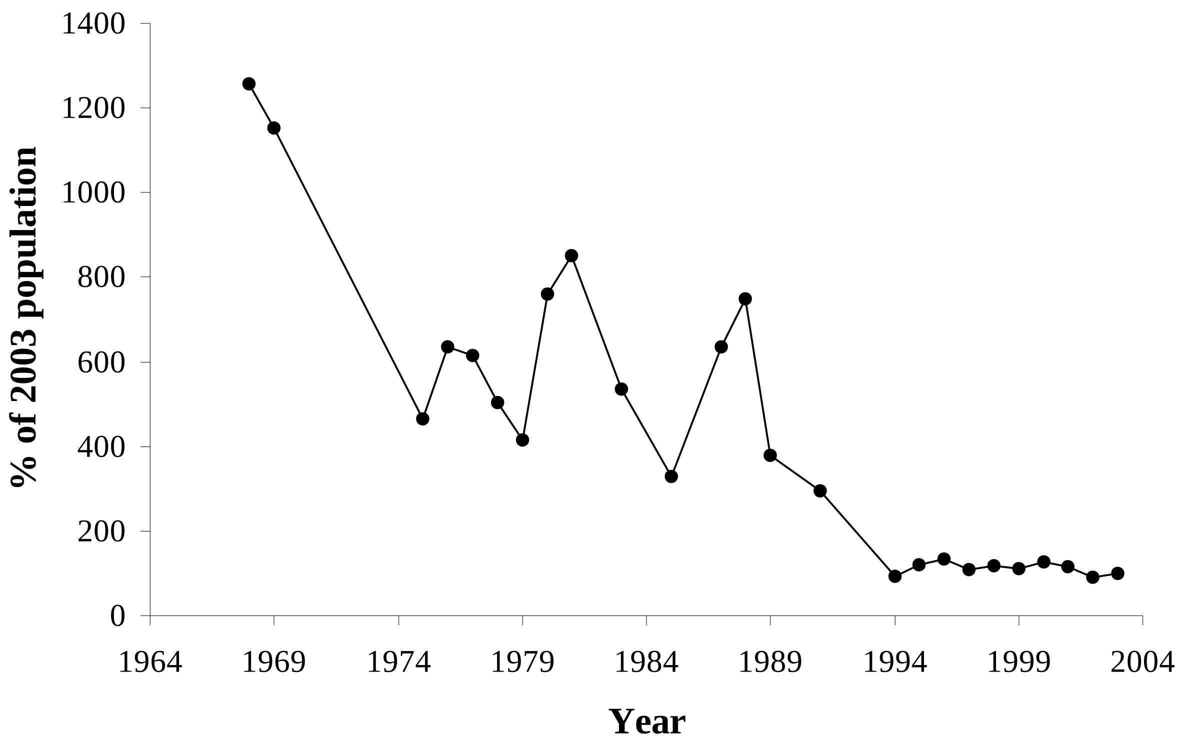


Table A5.9. Sage-grouse monitoring and population trends in N-Central MT subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	99	18	11	8	5	4	2	2
Number of active leks[1]	87	17	11	8	5	4	2	2
Percent active leks	88	94	100	98	100	100	100	100
Average males/lek	27	18	20	18	26	19	28	28
Median males/lek	22	16	18	15	25	22	28	25
Average males/active lek	31	19	20	18	26	19	28	28
Median males/active lek	26	17	18	16	25	22	28	25

[1] Averaged over each year for each period.

Fig. A5.8. Change in the population index for N-Central MT subpopulation, 1965-2003.

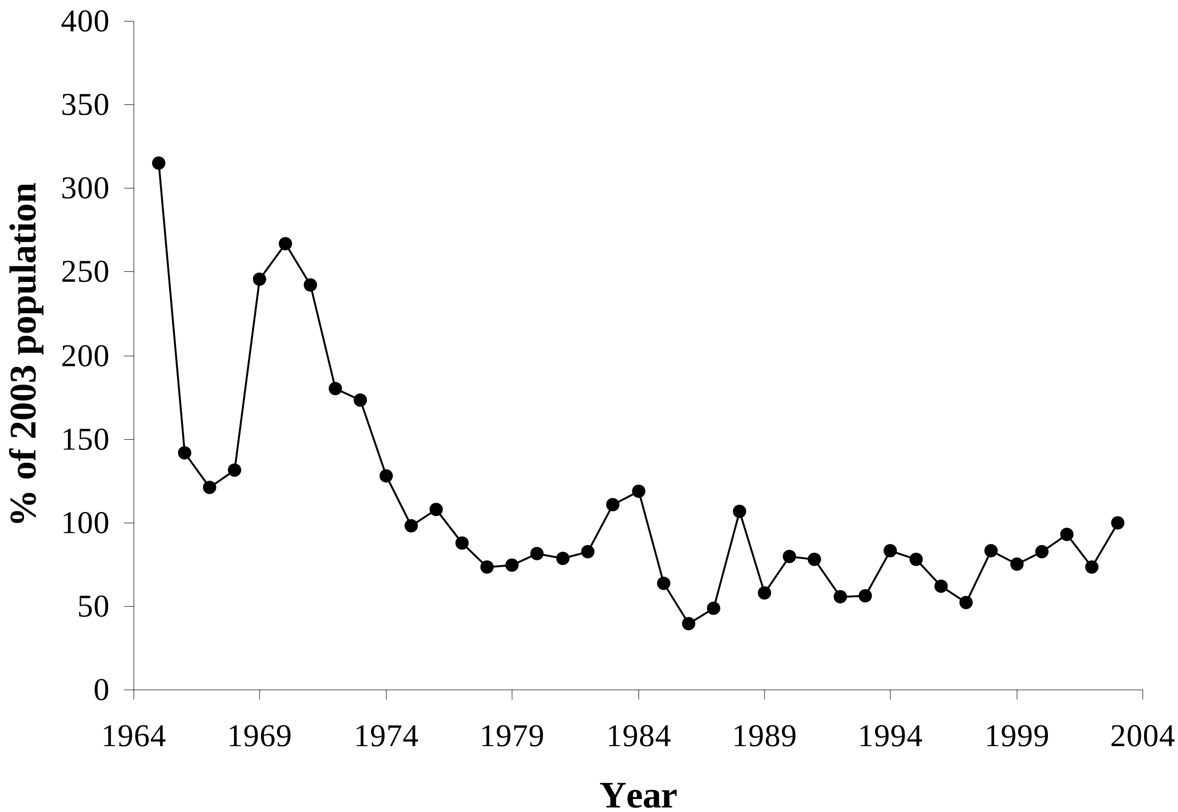
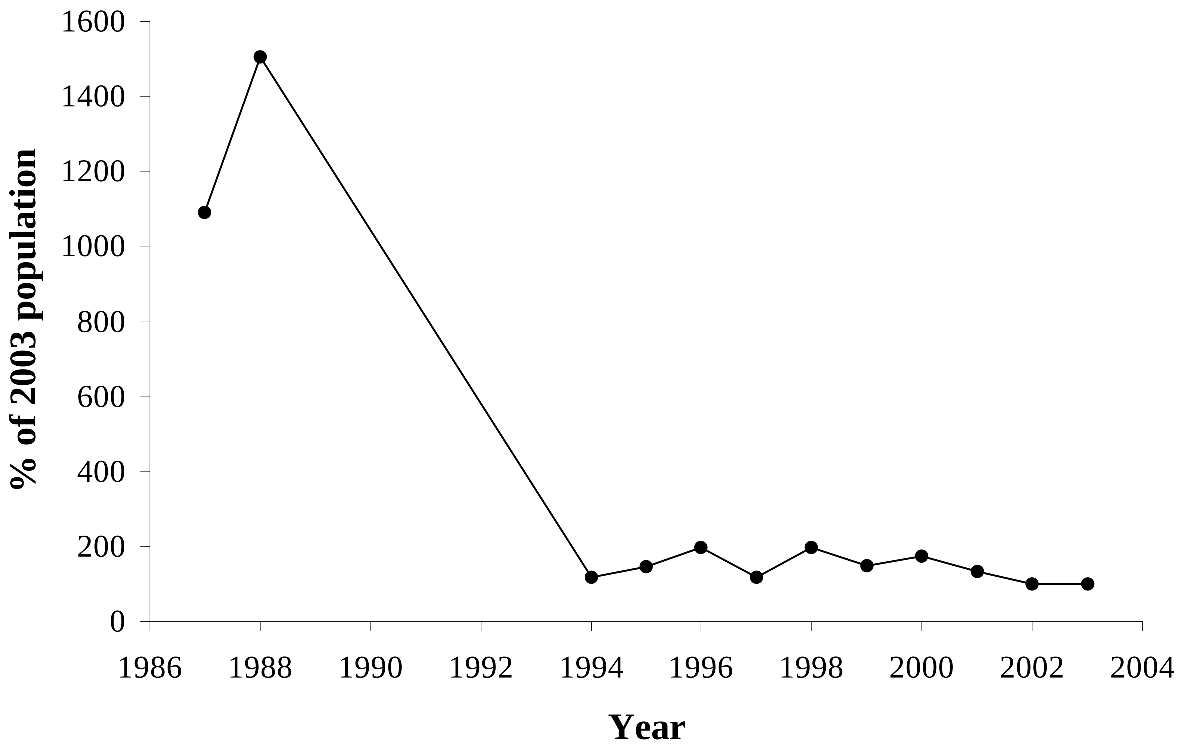


Table A5.10. Sage-grouse monitoring and population trends in S-Central SK/MT subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	18	14	4	7	0	0	0	0
Number of active leks[1]	14	7	2	7	0	0	0	0
Percent active leks	78	48	41	92				
Average males/lek	10	6	3	30				
Median males/lek	10	0	0	24				
Average males/active lek	13	12	6	33				
Median males/active lek	12	11	3	26				

[1] Averaged over each year for each period.

Fig. A5.9. Change in the population index for S-Central SK/MT subpopulation, 1987-2003.



Snake, Salmon, and Beaverhead Population

Table A5.11. Sage-grouse monitoring and population trends in Big Lost ID subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	12	8	1	1	4	1	3	0
Number of active leks[1]	11	6	1	1	4	1	3	0
Percent active leks	91	82	71	100	90	100	100	100
Average males/lek	17	12	6	12	22	47	54	97
Median males/lek	13	8	3	12	19	30	50	97
Average males/active lek	19	15	9	12	24	47	54	97
Median males/active lek	17	11	4	12	20	30	50	97

[1] Averaged over each year for each period.

Fig. A5.10. Change in the population index for Big Lost ID subpopulation, 1969-2003.

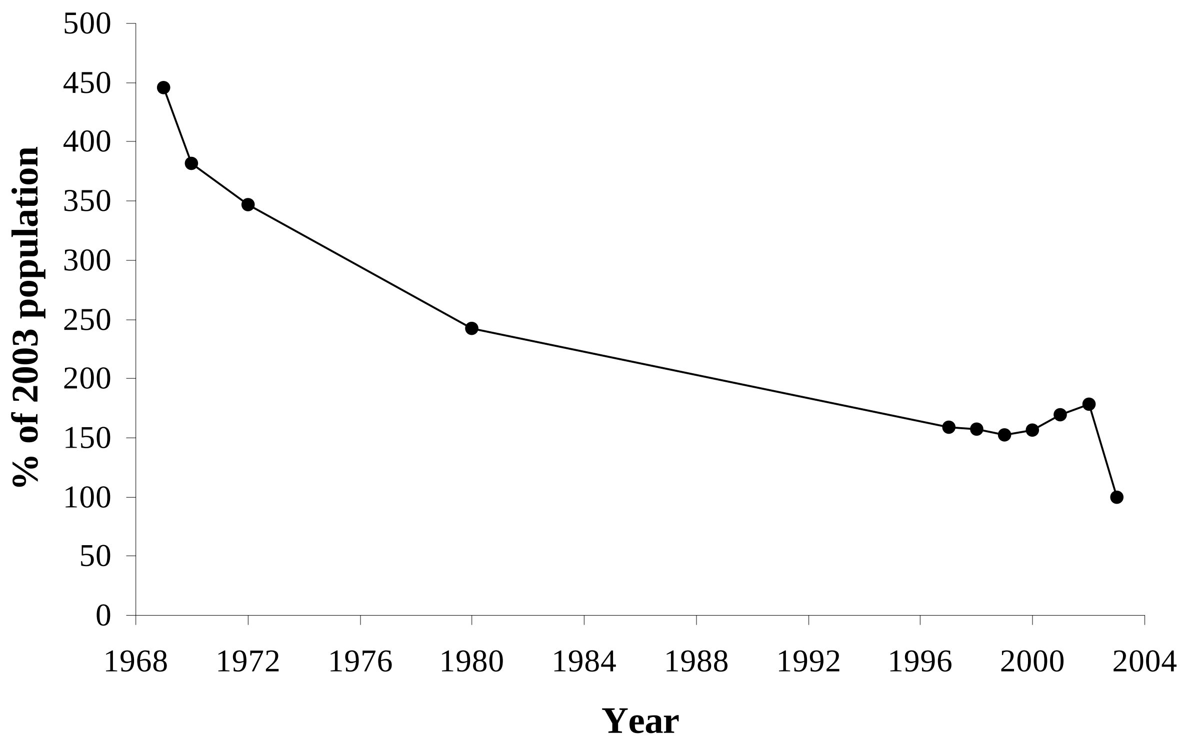


Table A5.12. Sage-grouse monitoring and population trends in Lemhi-Birch ID subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	19	14	13	12	5	7	6	6
Number of active leks[1]	12	7	10	11	4	5	6	5
Percent active leks	61	51	72	98	79	67	97	93
Average males/lek	13	8	16	28	12	13	20	29
Median males/lek	6	1	9	19	11	4	18	30
Average males/active lek	21	16	22	28	15	20	21	31
Median males/active lek	16	14	14	19	18	16	18	32

[1] Averaged over each year for each period.

Fig. A5.11. Change in the population index for Lemhi-Birch ID subpopulation, 1965-2003.

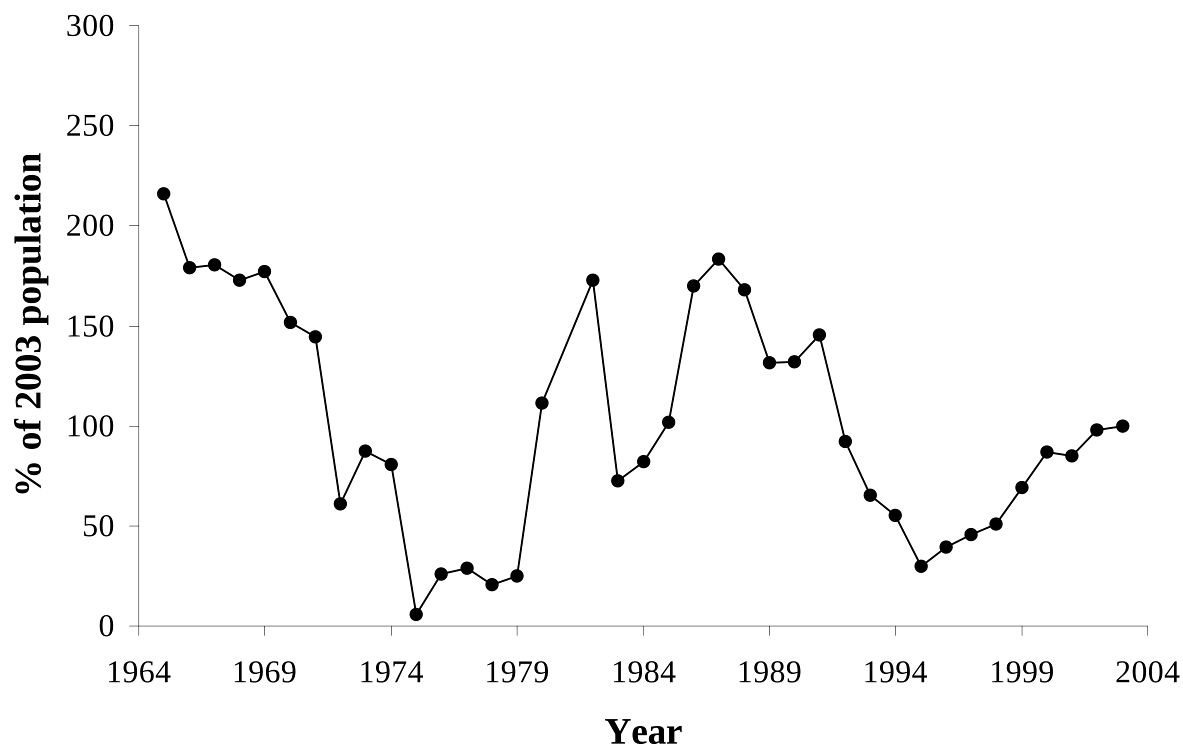


Table A5.13. Sage-grouse monitoring and population trends in Little Lost ID subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	11	5	4	3	2	2	4	2
Number of active leks[1]	10	4	4	2	2	1	3	2
Percent active leks	95	80	95	80	73	70	78	100
Average males/lek	34	26	30	30	17	16	38	32
Median males/lek	26	18	28	27	8	14	12	19
Average males/active lek	36	32	31	38	23	23	49	32
Median males/active lek	27	35	29	37	9	17	42	19

[1] Averaged over each year for each period.

Fig. A5.12. Change in the population index for Little Lost ID subpopulation, 1973-2003.

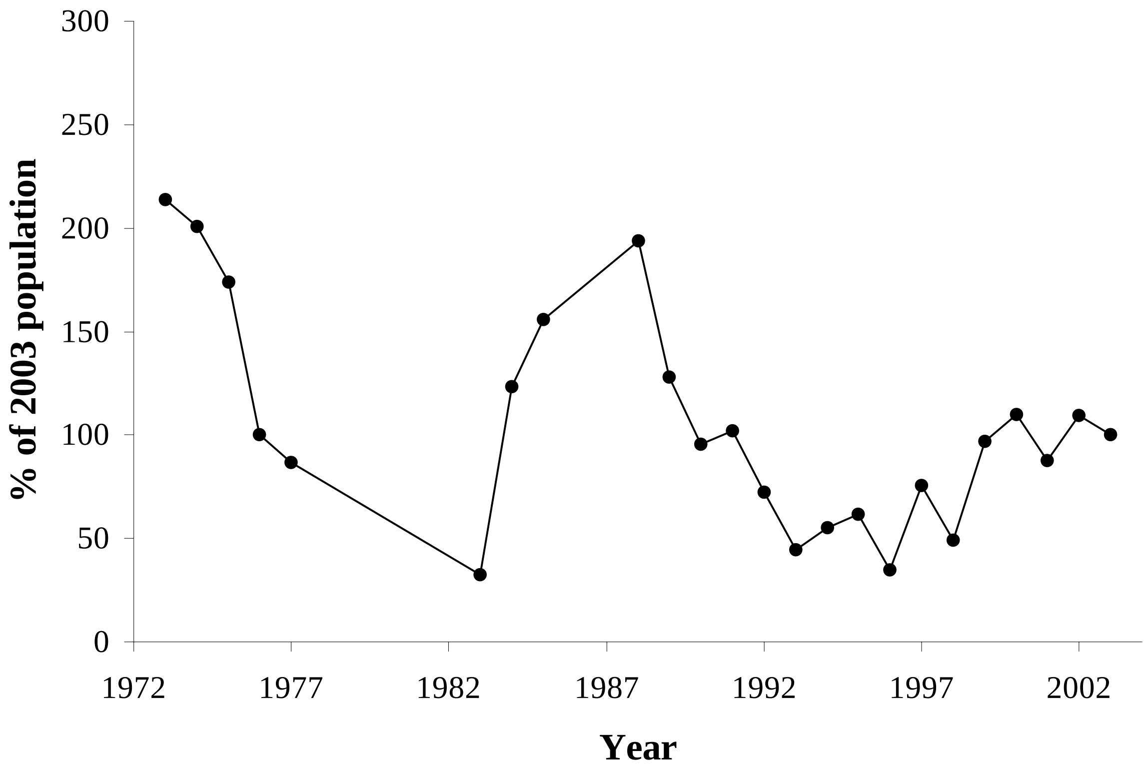


Table A5.14. Sage-grouse monitoring and population trends in N Side Snake ID subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	13	9	12	12	8	7	6	4
Number of active leks[1]	12	6	10	10	6	6	6	4
Percent active leks	90	67	88	83	67	83	90	95
Average males/lek	17	13	23	22	25	28	33	32
Median males/lek	11	10	22	18	19	14	26	23
Average males/active lek	19	19	27	26	37	33	36	33
Median males/active lek	13	13	23	23	33	24	32	24

[1] Averaged over each year for each period.

Fig. A5.13. Change in the population index for N Side Snake ID subpopulation, 1965-2003.

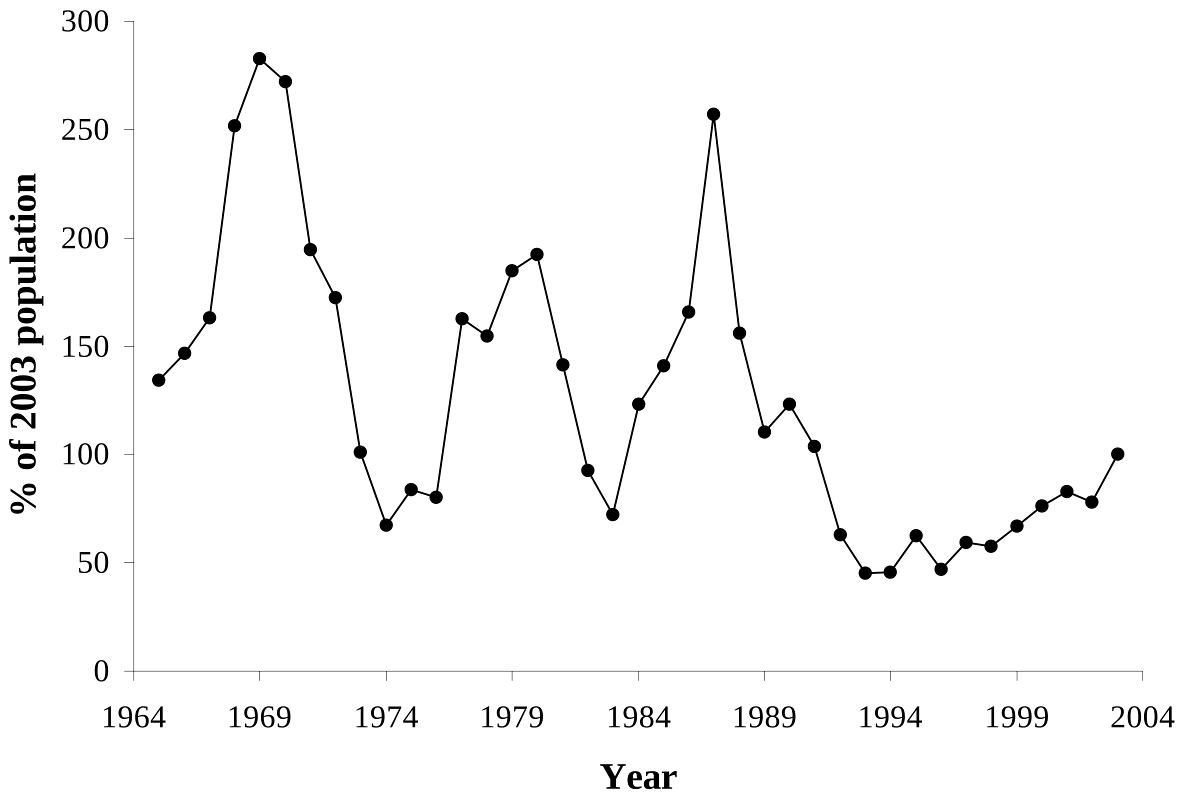
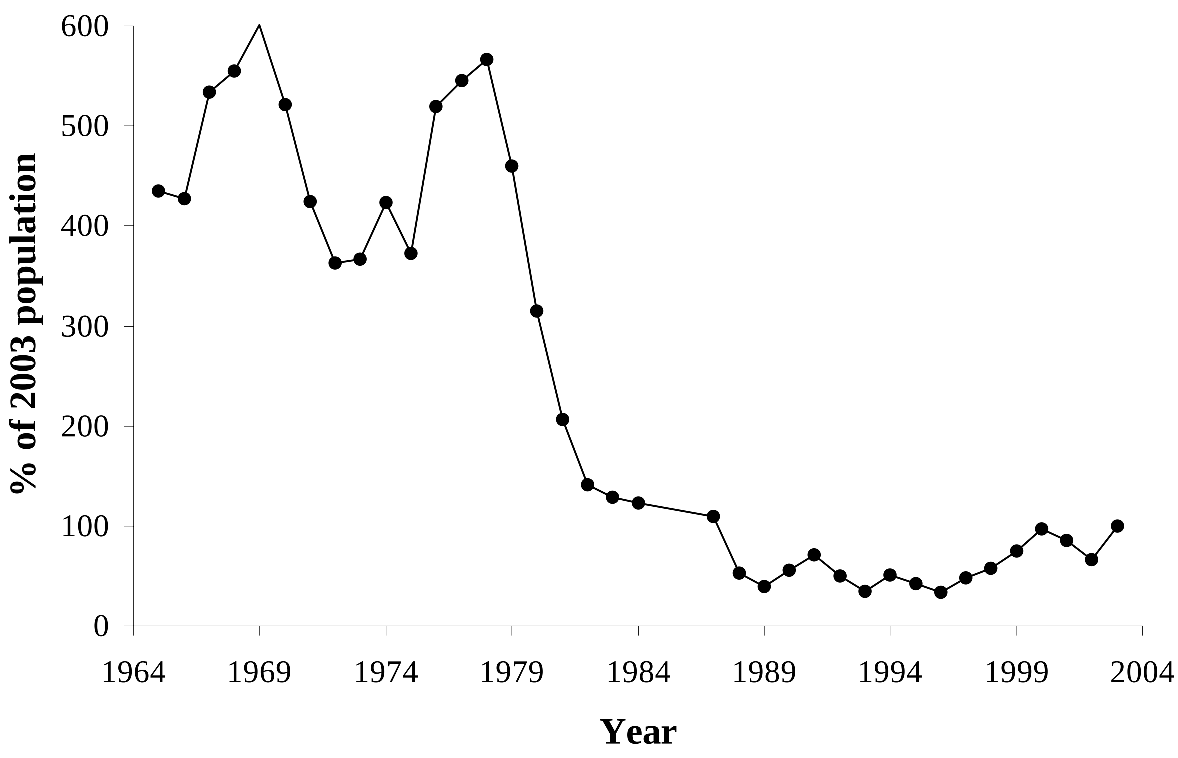


Table A5.15. Sage-grouse monitoring and population trends in Upper Snake ID subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	39	40	32	19	20	26	25	26
Number of active leks[1]	35	32	24	16	18	24	23	24
Percent active leks	88	80	75	85	91	92	93	93
Average males/lek	42	23	23	27	26	53	46	49
Median males/lek	25	16	13	16	17	36	38	36
Average males/active lek	47	29	31	32	28	58	49	53
Median males/active lek	30	24	21	21	18	39	41	39

[1] Averaged over each year for each period.

Fig. A5.14. Change in the population index for Upper Snake ID subpopulation, 1965-2003.



Wyoming Basin Population

Table A5.16. Sage-grouse monitoring and population trends in Dinosaur UT/CO subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	27	19	17	13	8	9	9	5
Number of active leks[1]	18	17	14	11	6	8	7	4
Percent active leks	69	86	86	82	79	93	84	96
Average males/lek	23	29	20	17	21	34	24	51
Median males/lek	7	13	11	11	12	15	15	34
Average males/active lek	33	34	23	21	26	37	29	53
Median males/active lek	18	19	12	14	16	16	16	40

[1] Averaged over each year for each period.

Fig. A5.15. Change in the population index for Dinosaur UT/CO subpopulation, 1965-2003.

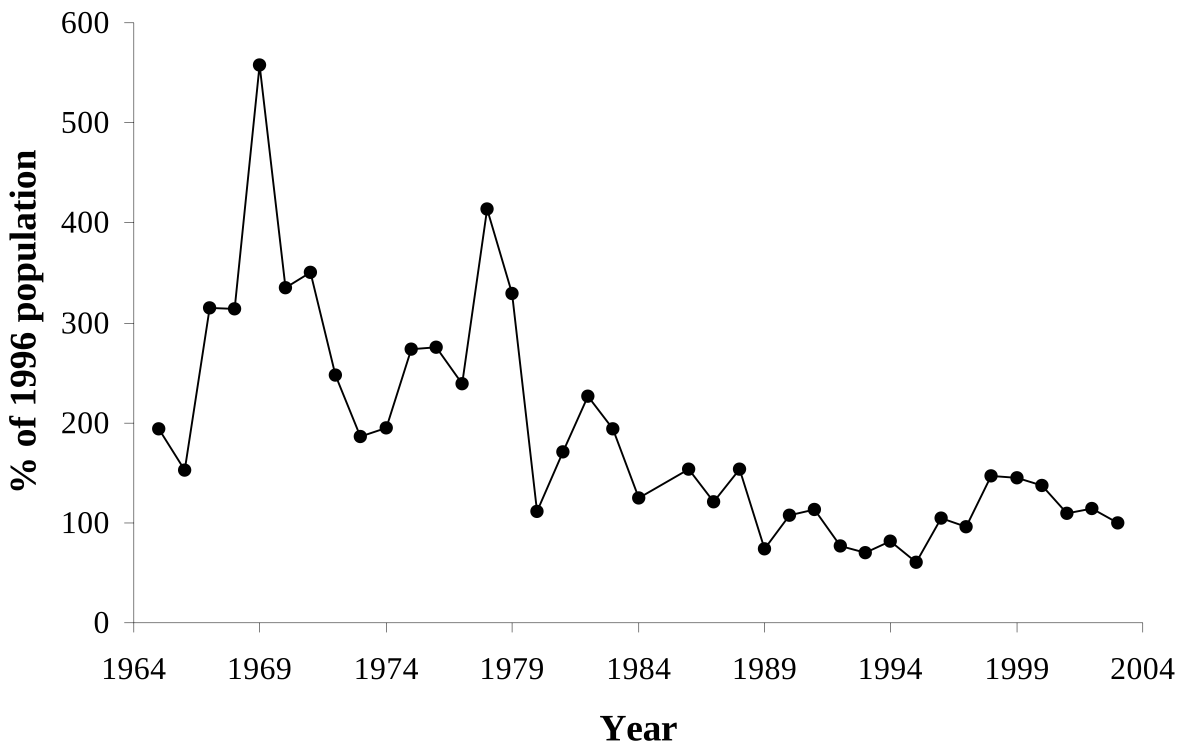


Table A5.17. Sage-grouse monitoring and population trends in Fall River SD/E Edge WY subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	5	1	1	1	2	0	0	0
Number of active leks[1]	5	1	1	1	2	0	0	0
Percent active leks	100	100	100	100	100			
Average males/lek	16	9	7	6	25			
Median males/lek	8	9	6					
Average males/active lek	16	9	7	6	25			
Median males/active lek	8	9	6					

[1] Averaged over each year for each period.

Fig. A5.16. Change in the population index for Fall River SD/E Edge WY subpopulation, 1982-2003.

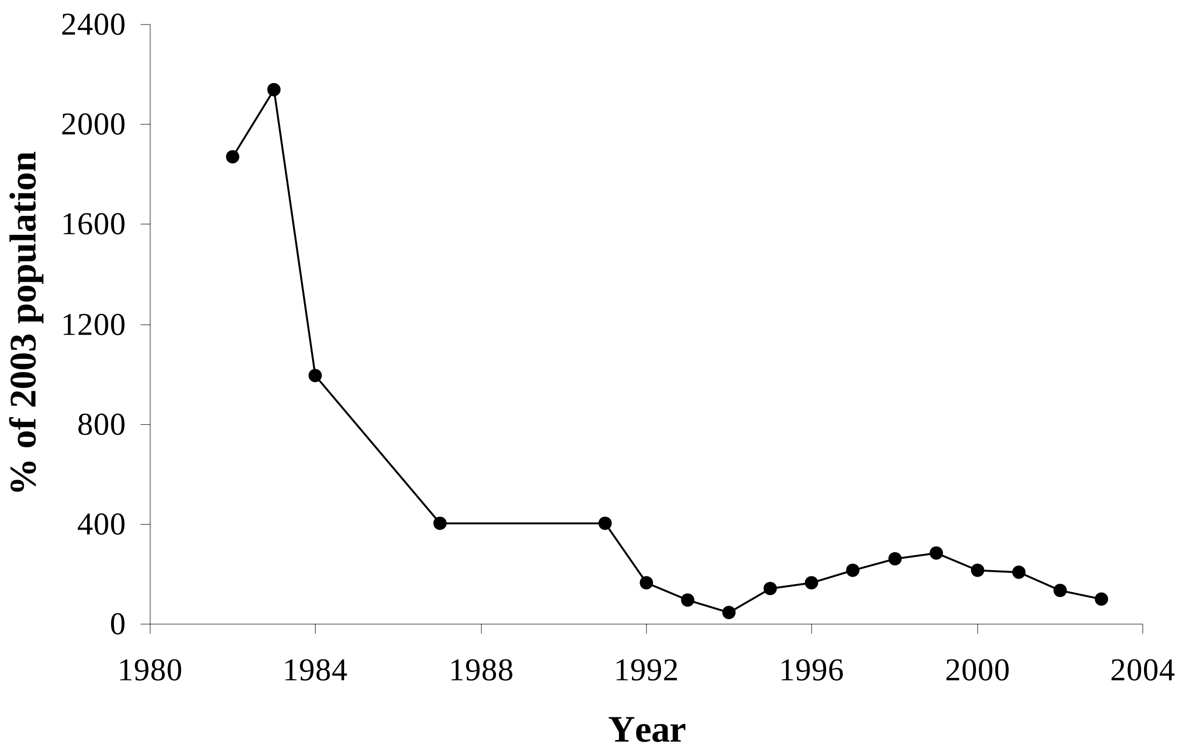


Table A5.18. Sage-grouse monitoring and population trends in NE WY/SE MT subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	197	87	77	76	78	25	15	7
Number of active leks[1]	132	52	56	62	71	23	15	7
Percent active leks	67	59	73	81	92	91	97	100
Average males/lek	11	8	13	16	22	25	23	36
Median males/lek	6	3	8	11	17	19	20	31
Average males/active lek	17	14	18	20	24	28	24	36
Median males/active lek	12	10	13	15	18	21	21	31

[1] Averaged over each year for each period.

Fig. A5.17. Change in the population index for NE WY/SE MT subpopulation, 1967-2003.

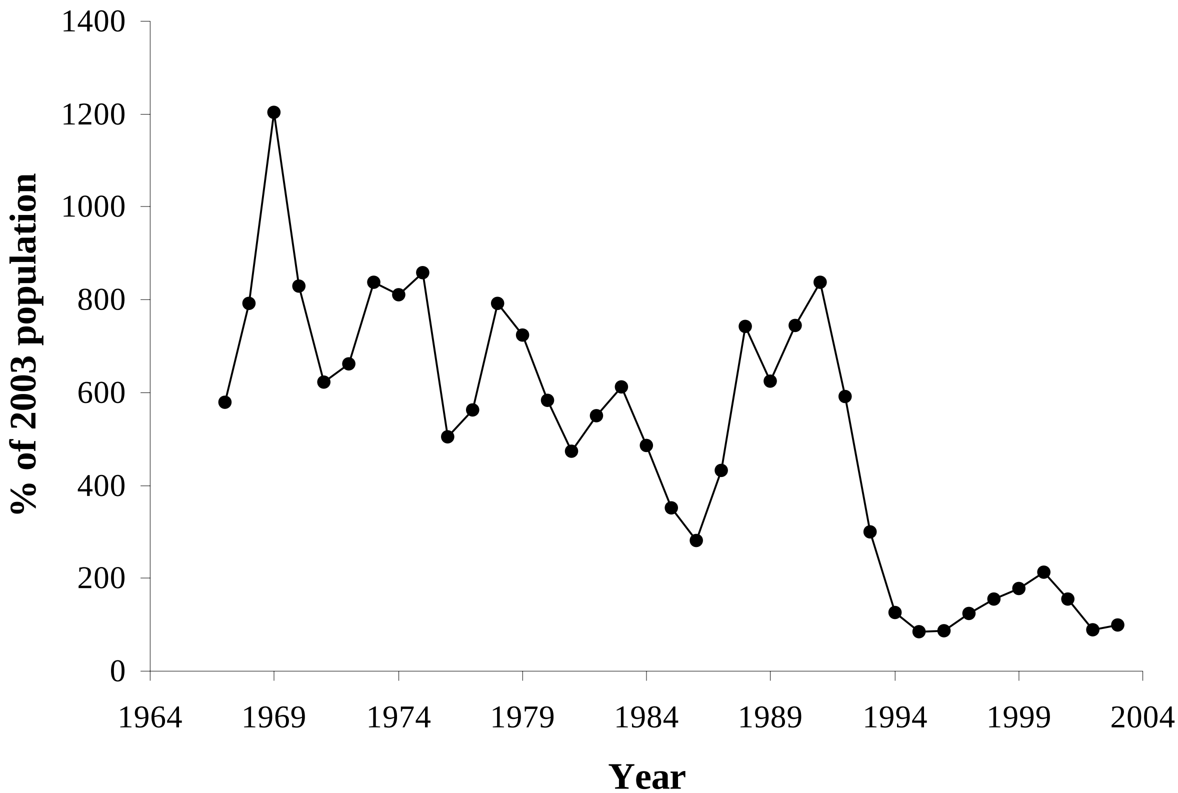


Table A5.19. Sage-grouse monitoring and population trends in North Park CO/WY subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	33	32	29	33	29	23	12	14
Number of active leks[1]	28	26	23	24	26	20	11	13
Percent active leks	85	82	80	72	91	89	95	90
Average males/lek	39	24	24	21	39	42	47	44
Median males/lek	19	15	21	13	30	33	39	44
Average males/active lek	46	30	30	29	42	47	49	49
Median males/active lek	23	22	28	24	34	41	44	50

[1] Averaged over each year for each period.

Fig. A5.18. Change in the population index for North Park CO/WY subpopulation, 1965-2003.

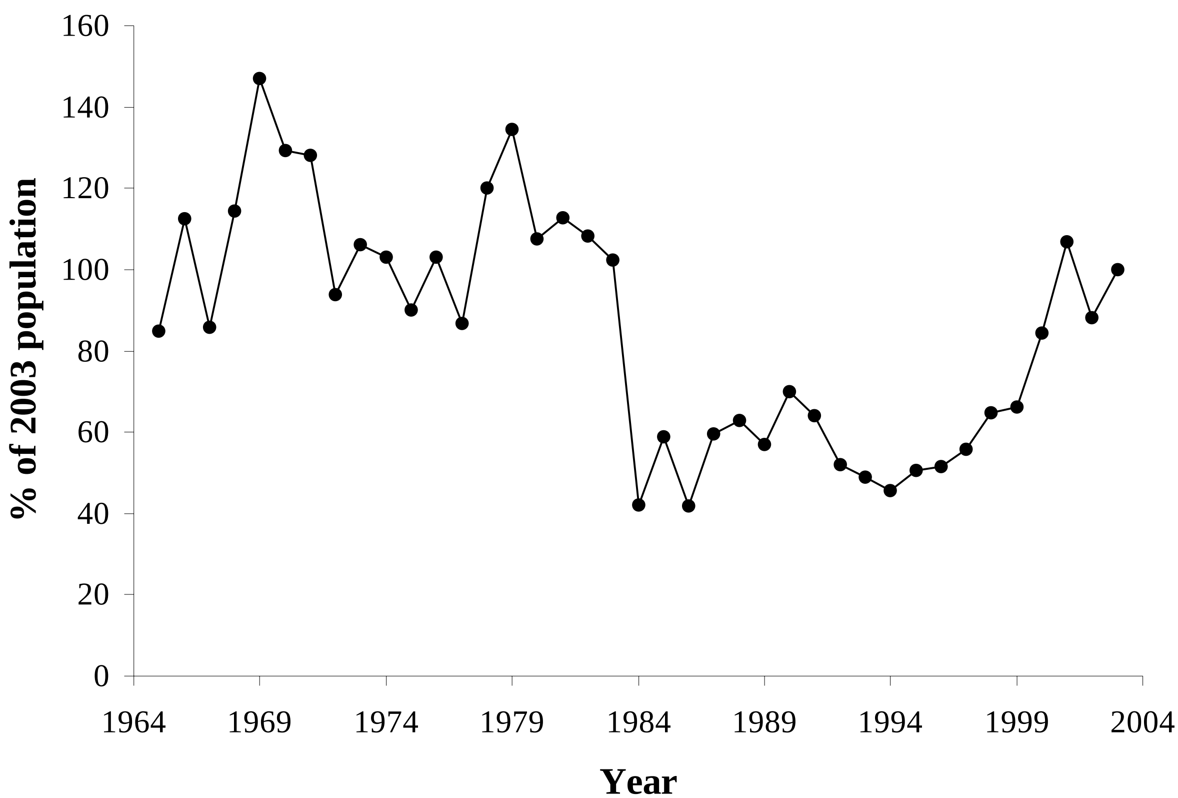


Table A5.20. Sage-grouse monitoring and population trends in S-Central MT/N-Central WY subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	159	114	108	93	68	30	24	20
Number of active leks[1]	133	86	85	78	62	25	14	6
Percent active leks	84	75	78	83	91	82	57	28
Average males/lek	16	13	15	16	19	18	12	8
Median males/lek	12	9	10	11	14	15	4	0
Average males/active lek	19	17	19	19	21	22	20	26
Median males/active lek	15	12	14	14	15	18	15	19

[1] Averaged over each year for each period.

Fig. A5.19. Change in the population index for S-Central MT/N-Central WY subpopulation, 1969-2003.

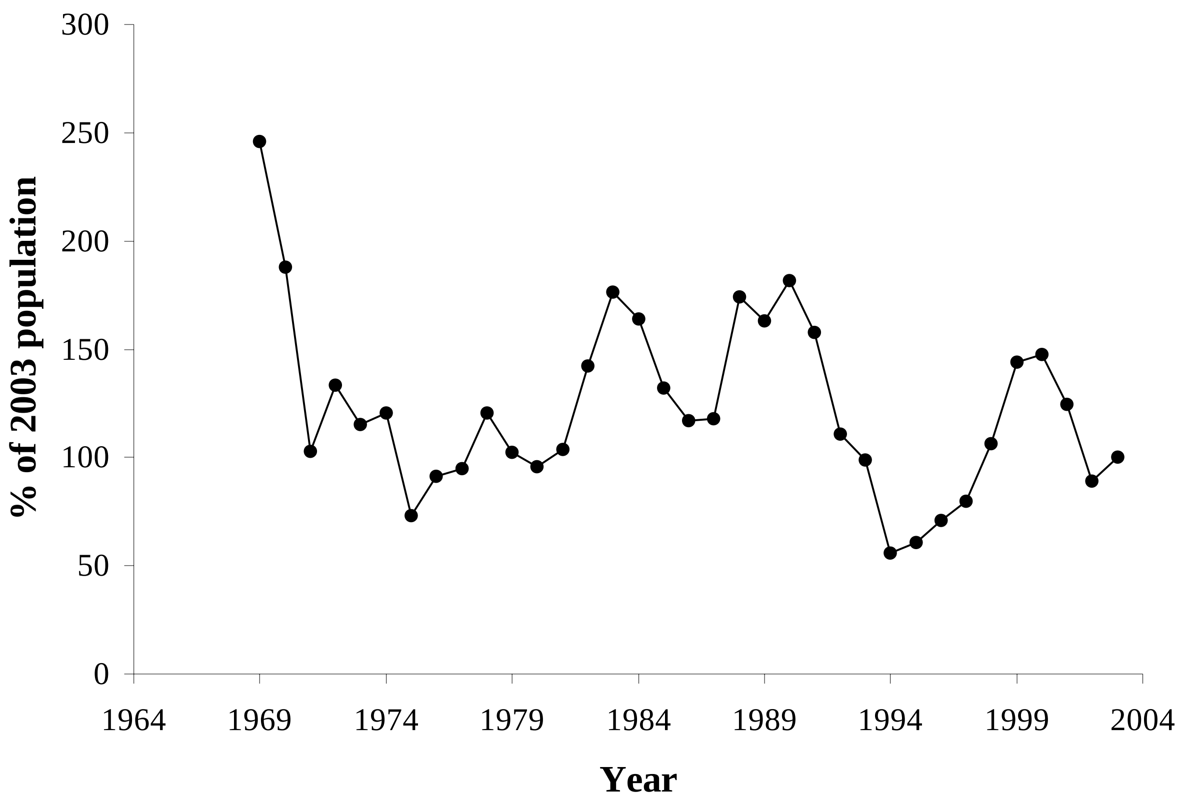


Table A5.21. Sage-grouse monitoring and population trends in S-Central WY/N-Central CO subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	491	372	283	268	199	106	66	81
Number of active leks[1]	323	225	180	174	118	70	52	64
Percent active leks	66	60	63	65	59	66	79	79
Average males/lek	20	12	15	16	20	27	32	41
Median males/lek	11	5	8	10	8	16	24	27
Average males/active lek	31	19	23	25	34	41	41	51
Median males/active lek	25	14	18	20	26	34	32	36

[1] Averaged over each year for each period.

Fig. A5.20. Change in the population index for S-Central WY/N-Central CO subpopulation, 1965-2003.

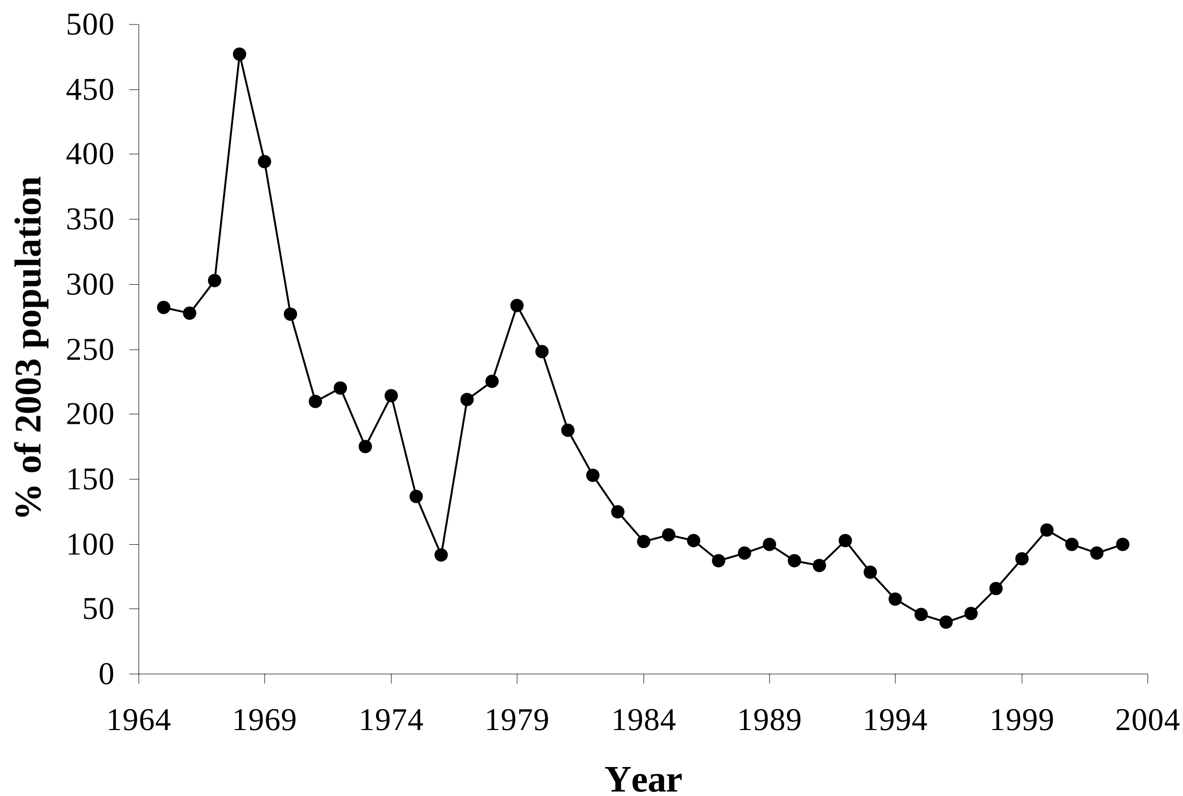
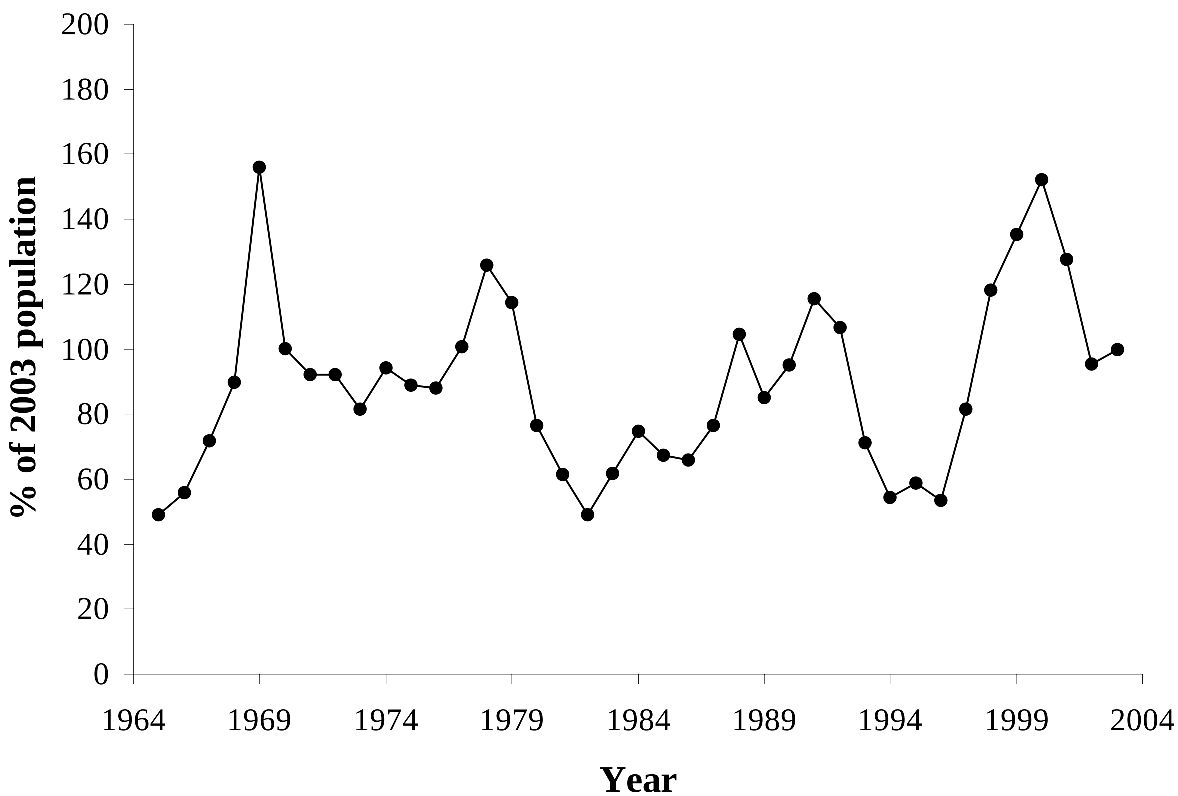


Table A5.22. Sage-grouse monitoring and population trends in SW WY/NW CO/NE UT/SE ID subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	282	207	166	100	58	31	17	10
Number of active leks[1]	203	132	95	79	53	26	15	9
Percent active leks	72	64	57	79	92	86	84	86
Average males/lek	22	18	14	22	26	38	29	34
Median males/lek	15	8	5	16	19	21	21	26
Average males/active lek	31	28	25	28	28	44	34	39
Median males/active lek	24	20	19	22	20	29	26	31

[1] Averaged over each year for each period.

Fig. A5.21. Change in the population index for SW WY/NW CO/NE UT/SE ID subpopulation, 1965-2003.



Yellowstone Watershed Population

Table A5.23. Sage-grouse monitoring and population trends in Central MT subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	239	111	95	116	96	75	52	8
Number of active leks[1]	175	79	72	94	88	67	47	8
Percent active leks	73	71	76	81	91	90	91	97
Average males/lek	17	14	16	17	27	23	23	23
Median males/lek	12	10	11	13	23	19	18	14
Average males/active lek	24	20	21	21	30	25	25	23
Median males/active lek	19	16	15	16	26	21	20	15

[1] Averaged over each year for each period.

Fig. A5.22. Change in the population index for Central MT subpopulation, 1966-2003.

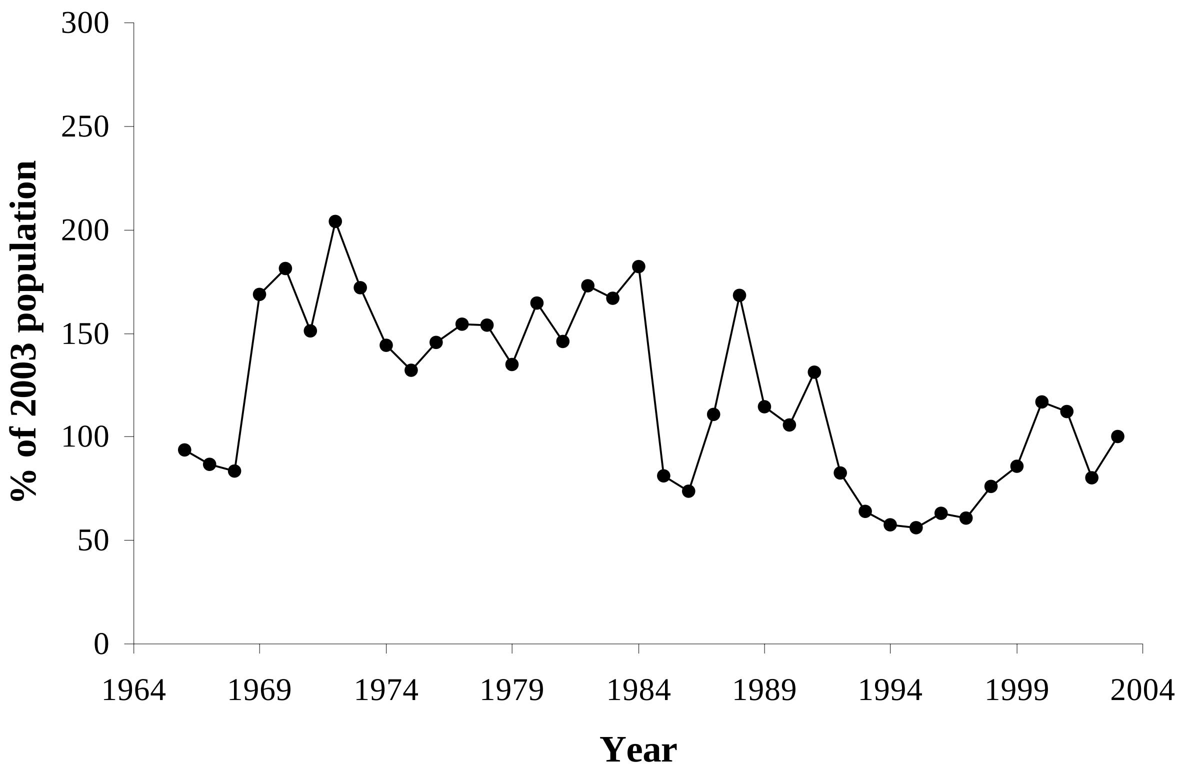
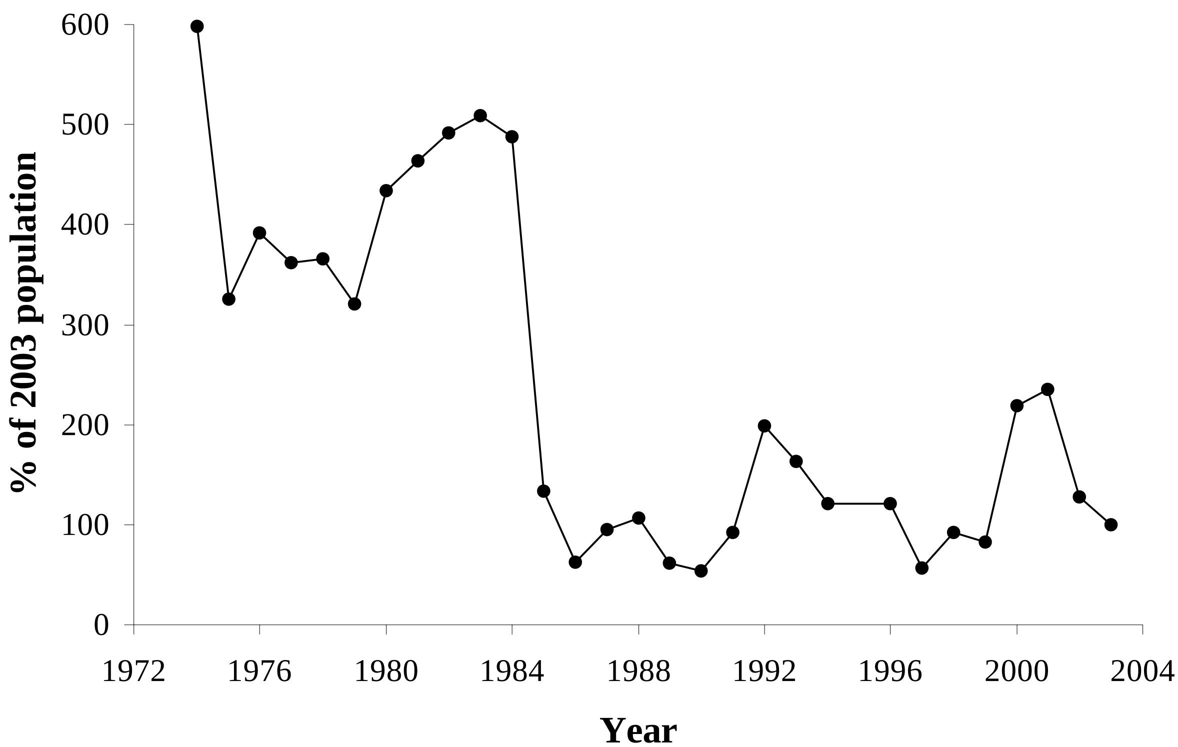


Table A5.24. Sage-grouse monitoring and population trends in E Interior MT/NE tip WY subpopulation, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	102	19	35	23	32	13	2	0
Number of active leks[1]	61	10	24	15	28	13	2	0
Percent active leks	60	53	69	66	88	97	100	100
Average males/lek	9	4	9	8	15	14	18	32
Median males/lek	3	2	7	6	13	11	20	32
Average males/active lek	15	8	13	13	17	14	18	32
Median males/active lek	10	7	12	10	14	12	20	32

[1] Averaged over each year for each period.

Fig. A5.23. Change in the population index for E Interior MT/NE tip WY subpopulation, 1974-2003.



APPENDIX 6

Characteristics of Greater Sage-Grouse Within Floristic Regions

Methods

The identified populations (Table 6.16, Appendix 4) and subpopulations (Table 6.16, Appendix 5) were divided into six floristic regions (Miller and Eddleman 2001) for a general analysis. An additional Great Plains Region was defined to include populations east and north of the area encompassed by Miller and Eddleman's research. Because some populations/subpopulations straddled the lines between floristic regions, they were included within the region where they had most of their leks.

Literature Cited

Miller, R. F., and L. L. Eddleman. 2001. Spatial and temporal changes of sage grouse habitat in the sagebrush biome. Oregon State University, Agricultural Experiment Station, Technical Bulletin 151, Corvallis, Oregon.

Table A6.1. Sage-grouse monitoring and population trends in Colorado Plateau region, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	4	7	10	12	9	1	2	2
Number of active leks[1]	3	6	4	4	7	1	1	1
Percent active leks	80	86	39	34	78	67	67	63
Average males/lek	11	9	4	5	6	10	8	3
Median males/lek	8	5	0	0	5	10	5	3
Average males/active lek	14	11	10	14	8	14	12	5
Median males/active lek	13	7	8	11	6	13	8	5

[1] Averaged over each year for each period.

Fig. A6.1. Change in the population index for Colorado Plateau region, 1980-2003.

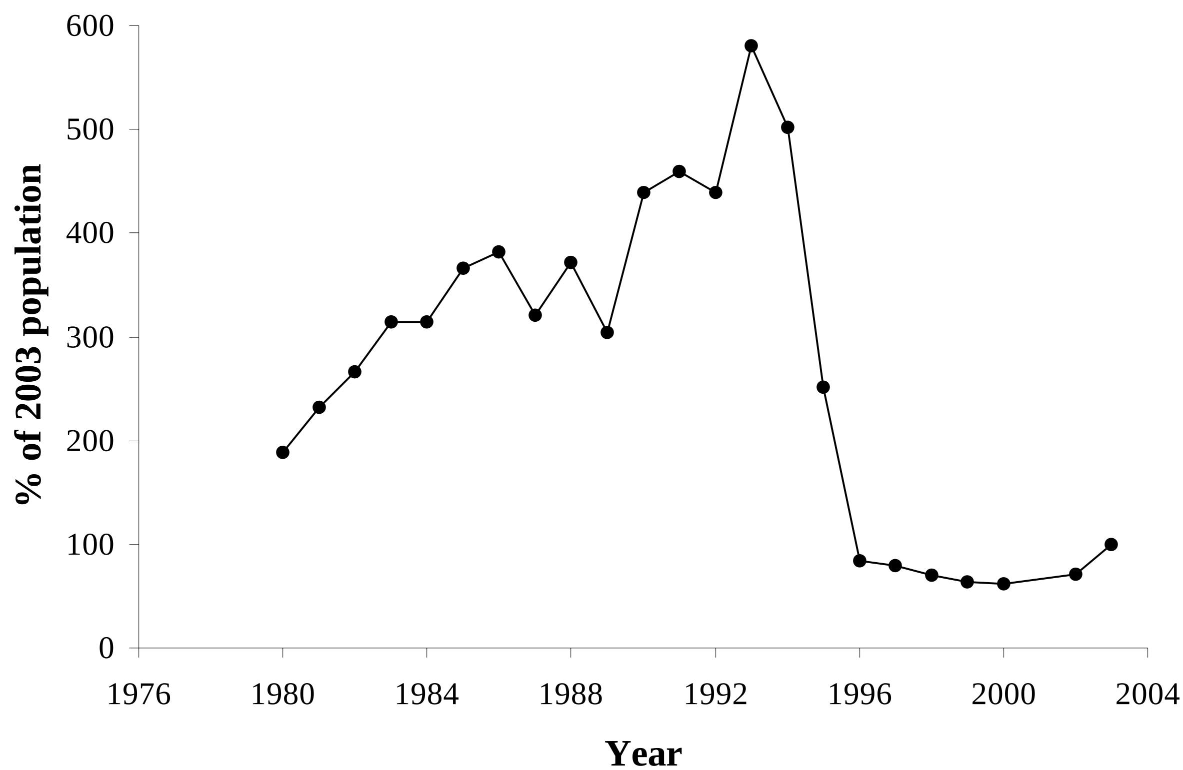


Table A6.2. Sage-grouse monitoring and regional trends in Columbia Basin region, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	47	31	25	20	18	15	14	3
Number of active leks[1]	20	18	16	14	16	14	13	3
Percent active leks	41	56	62	71	91	92	99	100
Average males/lek	8	11	13	17	26	15	24	33
Median males/lek	0	5	4	6	23	13	20	31
Average males/active lek	18	20	20	23	28	16	25	33
Median males/active lek	17	18	14	14	24	14	20	31

[1] Averaged over each year for each period.

Fig. A6.2. Change in the population index for Columbia Basin region, 1965-2003.

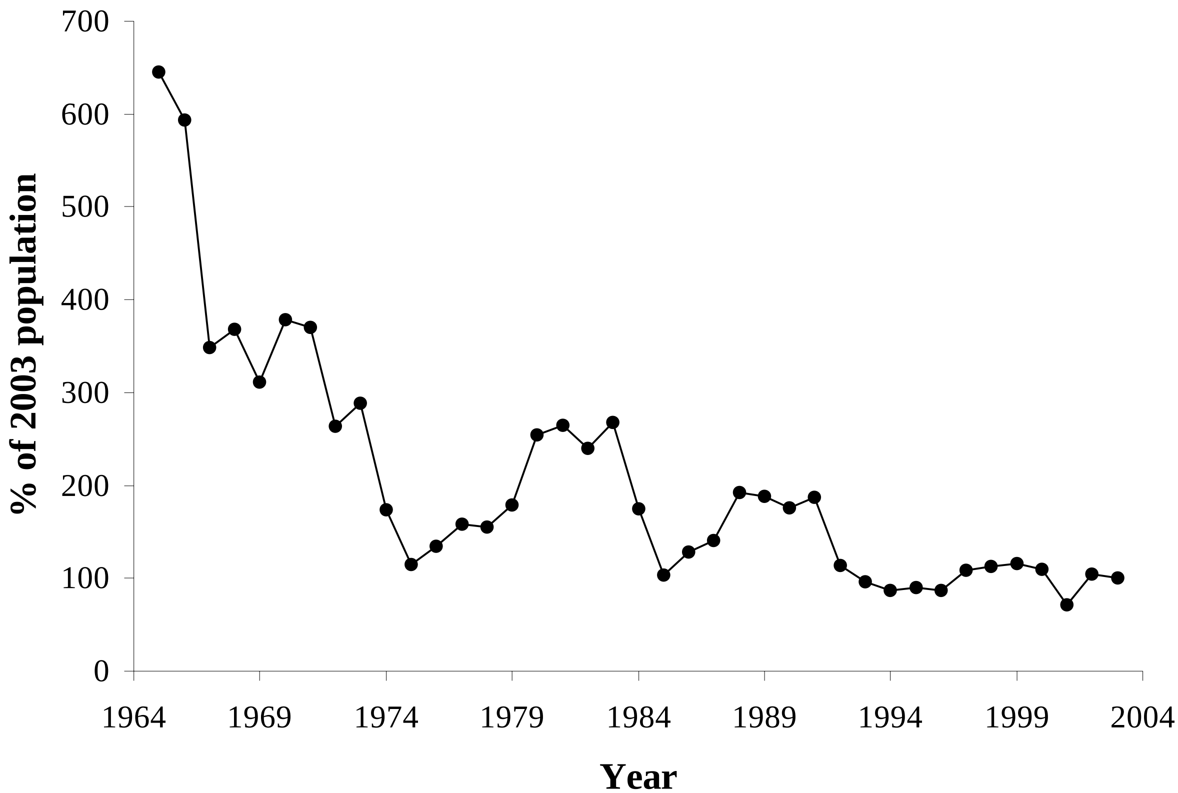


Table A6.3. Sage-grouse monitoring and regional trends in Great Plains region, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	747	315	271	274	255	158	94	44
Number of active leks[1]	519	203	203	224	232	145	87	42
Percent active leks	69	64	75	82	91	92	93	94
Average males/lek	15	10	13	16	22	20	21	22
Median males/lek	9	5	9	11	17	16	18	16
Average males/active lek	21	16	18	20	25	22	23	23
Median males/active lek	16	12	13	15	20	18	20	18

[1] Averaged over each year for each period.

Fig. A6.3. Change in the population index for Great Plains region, 1965-2003.

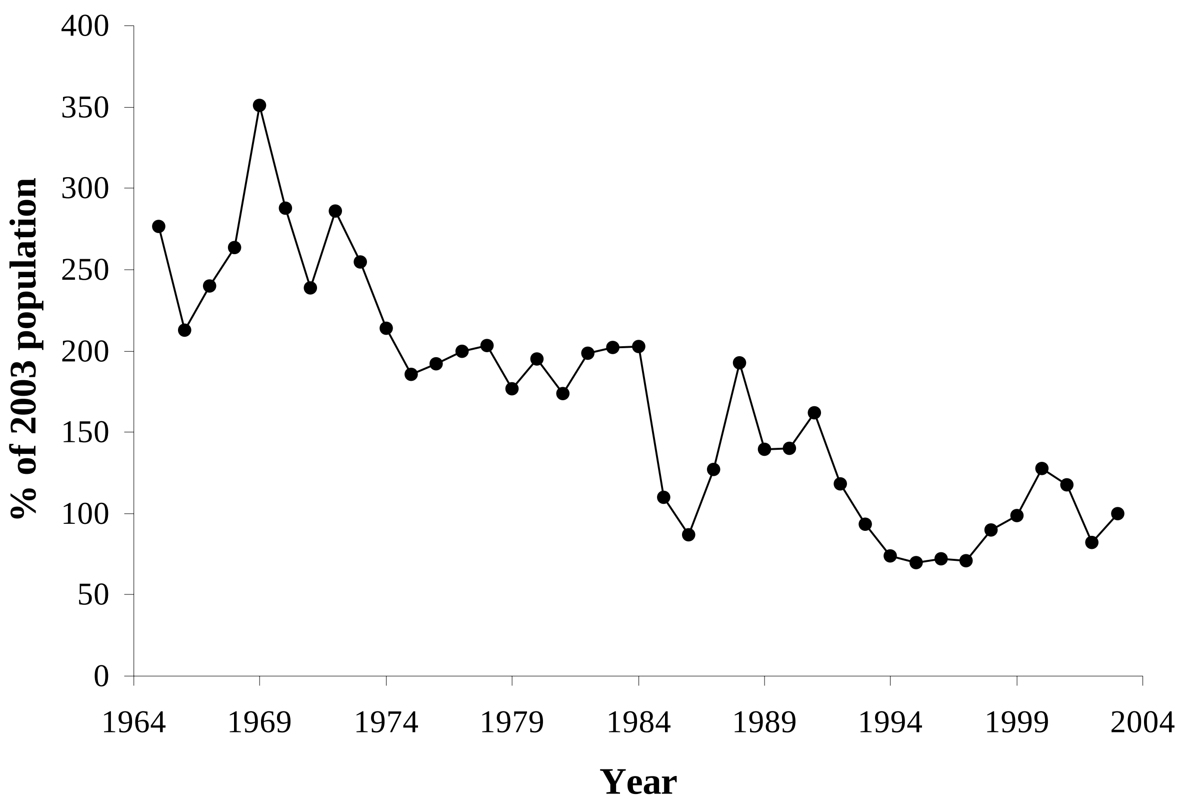


Table A6.4. Sage-grouse monitoring and regional trends in Northern Great Basin region, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	18	32	22	33	48	72	107	156
Number of active leks[1]	16	25	11	25	44	64	92	127
Percent active leks	91	79	50	76	91	88	86	82
Average males/lek	23	21	9	19	29	30	19	22
Median males/lek	14	11	1	10	19	17	12	13
Average males/active lek	25	26	17	25	32	34	22	27
Median males/active lek	16	16	14	17	20	20	15	19

[1] Averaged over each year for each period.

Fig. A6.4. Change in the population index for Northern Great Basin region, 1965-2003.

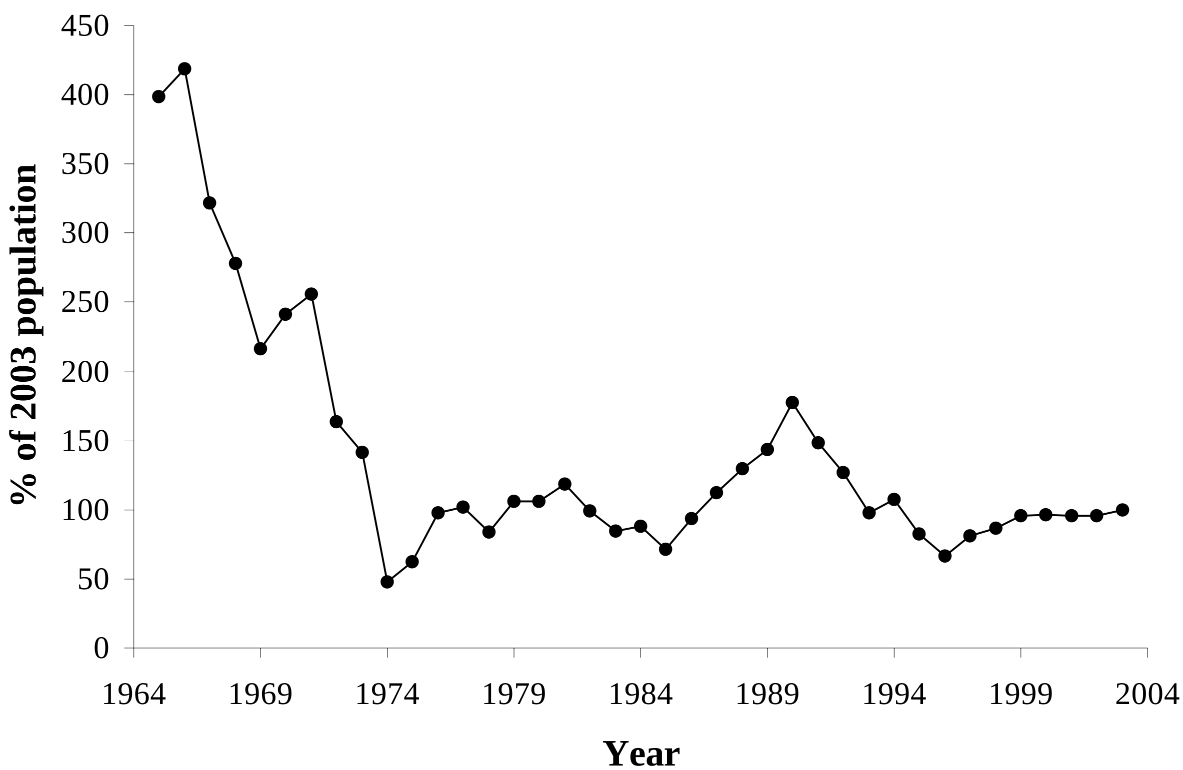


Table A6.5. Sage-grouse monitoring and regional trends in Snake River Basin region, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	450	334	195	169	215	172	149	97
Number of active leks[1]	364	256	158	136	177	155	130	91
Percent active leks	81	77	81	81	83	90	87	93
Average males/lek	20	15	21	28	22	32	31	39
Median males/lek	13	9	14	18	14	21	21	29
Average males/active lek	25	20	26	34	27	36	36	42
Median males/active lek	18	15	18	25	18	24	26	30

[1] Averaged over each year for each period.

Fig. A6.5. Change in the population index for Snake River Basin region, 1965-2003.

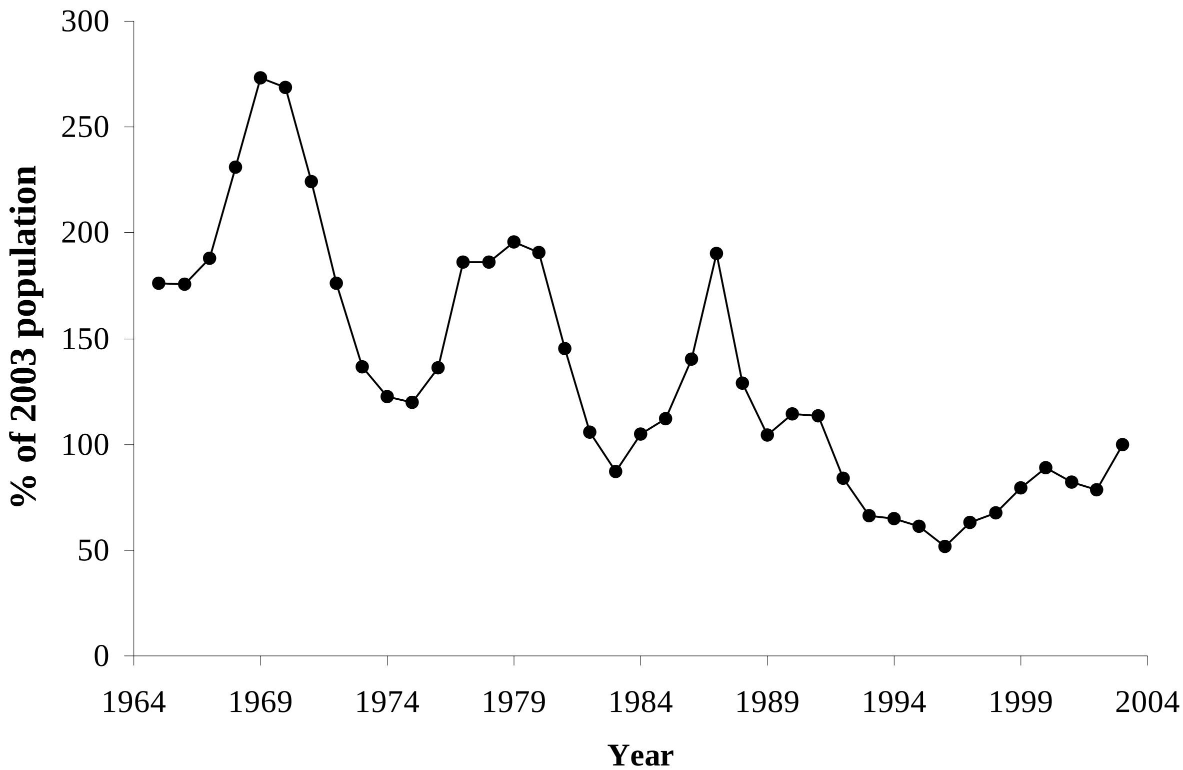


Table A6.6. Sage-grouse monitoring and regional trends in Southern Great Basin region, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	213	128	121	113	112	98	81	42
Number of active leks[1]	166	106	107	99	97	82	69	38
Percent active leks	78	83	89	87	87	83	85	90
Average males/lek	17	15	17	20	20	21	24	25
Median males/lek	10	10	11	14	14	13	16	16
Average males/active lek	22	18	19	23	23	25	28	28
Median males/active lek	15	13	14	16	17	18	22	19

[1] Averaged over each year for each period.

Fig. A6.6. Change in the population index for Southern Great Basin region, 1965-2003.

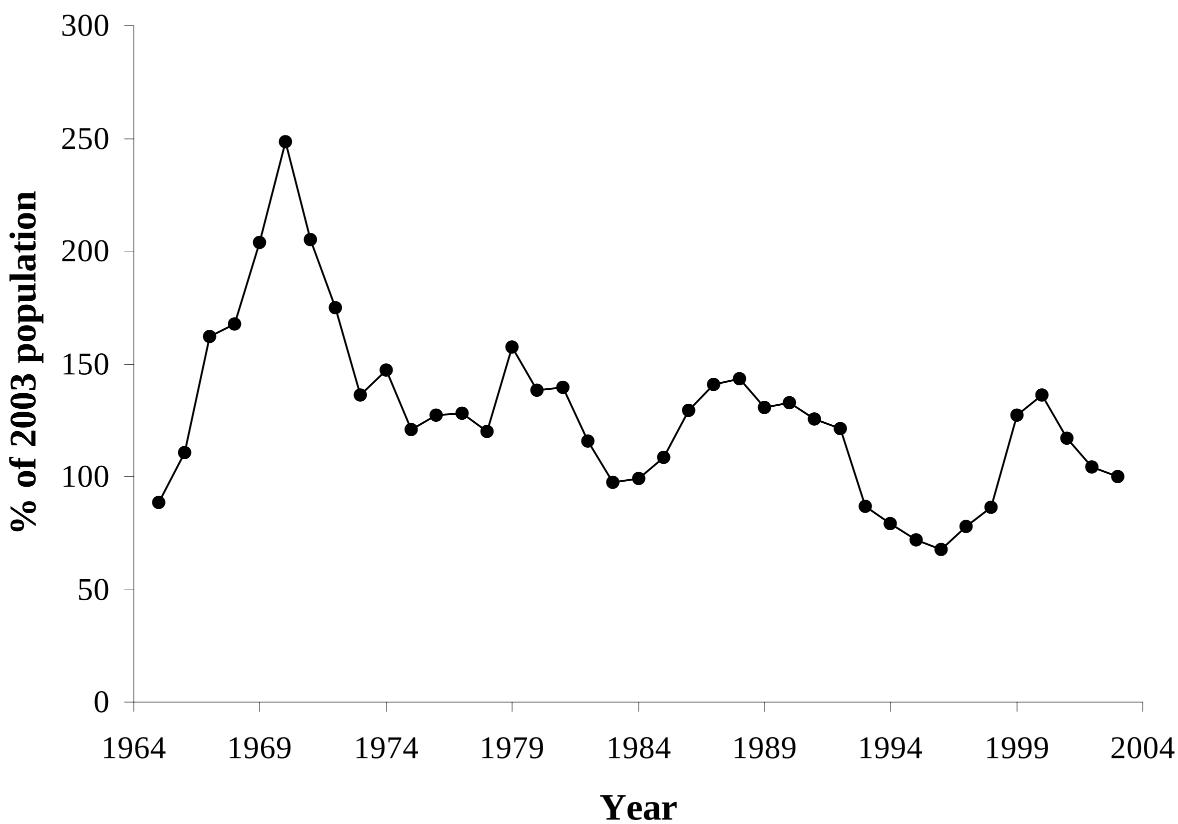


Table A6.7. Sage-grouse monitoring and regional trends in Wyoming Basin region, summarized over 5-year periods, 1965 - 2003.

Parameter	00-03	95-99	90-94	85-89	80-84	75-79	70-74	65-69
Leks counted[1]	1229	1100	819	693	587	381	287	242
Number of active leks[1]	892	751	565	511	449	312	237	199
Percent active leks	73	68	69	74	76	82	83	82
Average males/lek	20	15	16	20	22	30	30	37
Median males/lek	12	7	8	11	13	18	19	21
Average males/active lek	28	21	24	27	29	37	36	45
Median males/active lek	22	15	17	19	22	28	27	33

[1] Averaged over each year for each period.

Fig. A6.7. Change in the population index for Wyoming Basin region, 1965-2003.