

CHAPTER TEN

Characteristics of Sagebrush Habitats and Limitations to Long-Term Conservation

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Abstract. The distribution of sagebrush (*Artemisia* spp.) within the Sage-Grouse Conservation Area (SGCA, the historical distribution of sage-grouse buffered by 50 km) stretches from British Columbia and Saskatchewan in the north, to northern Arizona and New Mexico in the south, and from the eastern slopes of the Sierra Nevada and Cascade mountains to western South Dakota. The dominant sagebrush (sub)species as well as the composition and proportion of shrubs, grasses, and forbs varies across different ecological sites as a function of precipitation, temperature, soils, topographic position, elevation, and disturbance history. Most important to Greater Sage-Grouse (*Centrocercus urophasianus*) are three subspecies of big sagebrush (*Artemisia tridentata*) (basin big sagebrush [*A. t.* ssp. *tridentata*], Wyoming big sagebrush [*A. t.* ssp. *wyomingensis*], and mountain big sagebrush [*A. t.* ssp. *vaseyana*]); two low or dwarf forms (little sagebrush [*A. arbuscula*] and black sagebrush [*A. nova*]); and silver sagebrush (*A. cana*), which occurs primarily in the northeast portion of the sage-grouse range. Invasive plant species, wildfires, and weather and climate change are major influences on sagebrush habitats and present significant challenges to their long-term conservation. Each factor is spatially pervasive across the Greater Sage-Grouse Conservation Area

and has significant potential to influence processes within sagebrush communities. Cheatgrass (*Bromus tectorum*), the most widespread exotic annual grass, has invaded much of the lower-elevation, more xeric sagebrush landscapes across the western portion of the Greater Sage-Grouse Conservation Area. A large proportion of existing sagebrush communities are at moderate to high risk of invasion by cheatgrass. Juniper (*Juniperus* spp.) and pinyon (*Pinus* spp.) woodlands have expanded into sagebrush habitats at higher elevations creating an elevational squeeze on the sagebrush ecosystem from both extremes. Number of fires and total area burned have increased since 1980 throughout the SGCA except in the Snake River Plain, which has a long-term history of high fire disturbance. Climate change scenarios for the sagebrush region predict increasing trends in temperature, atmospheric carbon dioxide, and frequency of severe weather events that favor cheatgrass expansion and increased fire disturbance resulting in a decline in sagebrush. Approximately 12% of the current distribution of sagebrush is predicted to be replaced by expansion of other woody vegetation for each 1°C increase in temperature. Periodic drought regularly influences sagebrush ecosystems; drought duration and severity have increased throughout the 20th century in much of the interior western

Miller, R. F., S. T. Knick, D. A. Pyke, C. W. Meinke, S. E. Hanser, M. J. Wisdom, and A. L. Hild. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. Pp. 145–184 in S. T. Knick and J. W. Connelly (editors). Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology (vol. 38), University of California Press, Berkeley, CA.

United States. Synergistic feedbacks among invasive plant species, fire, and climate change, coupled with current trajectories of habitat changes and rates of disturbance (natural and human-caused), will continue to change sagebrush communities and create challenges for future conservation and management.

Key Words: *Artemisia*, *Bromus tectorum*, climate change, community dynamics, drought, exotic plant species, juniper, pinyon, sagebrush, weather, wildfire.

Características de los Hábitats de Artemisa y Limitaciones para la Conservación a Largo Plazo

Resumen. La distribución de artemisa (*Artemisia* spp.) dentro del área de conservación del Greater Sage-Grouse (SGCA; la distribución histórica del sage-grouse amortiguada por 50 km) se extiende desde la British Columbia y Saskatchewan, en el norte, al norte de Arizona y New Mexico en el sur, y desde las laderas orientales de la Sierra Nevada y Montañas Cascade, al oeste de Dakota del Sur. Las (sub)especies de artemisa dominantes, así como la composición y la proporción de arbustos, gramíneas, y malezas varían a lo largo de los diferentes sitios ecológicos en función de las precipitaciones, la temperatura, los suelos, la posición topográfica, la altitud, y el historial de disturbios en el área. Lo más importante para el Greater Sage-Grouse (*Centrocercus urophasianus*) son tres subespecies de big sagebrush (*Artemisia tridentata*) (basin big sagebrush [*A. t.* spp. *tridentata*], Wyoming big sagebrush [*A. t.* spp. *wyomingensis*], y mountain big sagebrush [*A. t.*, spp. *vaseyana*]); dos formas bajas (o enanas, la little sagebrush [*A. arbuscula*] y black sagebrush [*A. nova*]); y silver sagebrush (*A. cana*) que ocurre principalmente en la porción nordeste de la zona de Greater Sage-Grouse. Especies de plantas invasoras, incendios forestales, el tiempo, y el cambio climático son las principales influencias sobre los hábitats de la artemisa y también presentan desafíos importantes para su conservación a largo plazo. Cada uno de ellos está presente a través de

la zona de SGCA y tiene un importante potencial para influir en los procesos dentro de las comunidades de artemisa. Cheatgrass (*Bromus tectorum*), la más extendida hierba anual exótica, ha invadido gran parte de las altitudes más bajas, los paisajes de artemisa más adaptados a las condiciones desérticas a través de la parte occidental de la SGCA. Una gran proporción de las comunidades existentes de artemisa están entre un moderado a alto riesgo de invasión por cheatgrass. Bosques de enebro (*Juniperus* spp.) y piñón (*Pinus* spp.) se han expandido a los hábitats de artemisa en elevaciones más altas creando así presión sobre el ecosistema de artemisa en ambos extremos. El número de incendios y el área total incendiada se han incrementado desde 1980, en toda la SGCA, excepto en el área del Snake River Plain, el cual tiene un largo historial de alteraciones por fuegos grandes. Escenarios de cambios climáticos en la región de la artemisa predicen tendencias de incremento en la temperatura, dióxido de carbono atmosférico, y frecuencia de eventos severos de clima que favorecen la expansión del cheatgrass e incrementan la alteración por fuego que produce una disminución de artemisa. Se predice que aproximadamente el 12% de la actual distribución de artemisa será remplazada, por la expansión de otra vegetación leñosa, por cada grado centígrado de incremento en temperatura. Sequías periódicas influyen regularmente el ecosistema de la artemisa; la duración y severidad de las sequías se han incrementado a lo largo del siglo XX en la mayoría del interior del oeste de EE.UU. Las retroalimentaciones climáticas sinérgicas entre especies de plantas invasivas, fuego y cambios climáticos, sumados a las actuales trayectorias en cambios de hábitat y los grados de alteración (naturales y causadas por el hombre) continuarán cambiando las comunidades de artemisa y creando desafíos para la futura conservación y manejo.

Palabras Clave: artemisa, *Artemisia*, *Bromus tectorum*, cambio climático, clima, dinámica de la comunidad, especies de plantas exóticas, enebro, incendios forestales, pinos, sequía.

Ecological sites supporting sagebrush (*Artemisia* spp.) within the Sage-Grouse Conservation Area (SGCA, the historical distribution of sage-grouse buffered by 50 km [Connelly et al. 2004, Schroeder et al. 2004]) represent some of the largest and most imperiled ecosystems in North America (Noss et al. 1995; Center for Science, Economics, and Environment 2002). The primary patterns, processes, and components of sagebrush ecosystems have been altered significantly since Euro-American settlement in the late 1800s (West and Young 2000, Bunting et al. 2003). Few, if any, landscapes remain intact and unchanged throughout the SGCA (Miller et al. 1994, West 1996, Miller and Eddleman 2001). It is unlikely that we can return to pre-settlement conditions because size of the area and magnitude of changes far exceed any financial or logistical resources available (Hemstrom et al. 2002, Wisdom et al. 2005c, Meinke et al. 2009). In addition, loss of available parts from native systems (West 1996, Longland and Bateman 2002) coupled with continual short- and long-term changes in climate further complicate our ability to recreate previous sagebrush communities. However, learning how sagebrush communities function and the potential effects of primary disrupters can help increase understanding how these systems respond to land use (Knick et al., this volume, chapter 12) or management actions, such as restoration (Pyke, this volume, chapter 23) and may help maintain at least a portion of these ecosystems.

We describe the distribution of sagebrush within the SGCA followed by a general description of the characteristics of sagebrush alliances (a physiognomically uniform group of plant associations sharing one or more dominant or diagnostic species, which, as a rule, are found in the uppermost stratum of the vegetation; Grossman et al. 1998) and plant associations (a plant community type of definite floristic composition, uniform habitat conditions, and uniform physiognomy; Daubenmire 1978, Grossman et al. 1998). We then describe the extent and potential for invasive plant species, wildfire, and weather and climate change to alter the character of sagebrush systems and further affect our ability to conserve and manage these habitats. Sagebrush ecosystems are influenced by numerous stressors: Wisdom et al. (2005b:30–35) list more than 25 potential disrupters. Of these, we focused on exotic plants, wildfire, and climate because they are spatially pervasive agents having

a significant potential to influence long-term changes in patterns and processes across the SGCA. Other disrupters that more directly originate from human actions are considered elsewhere (Knick et al., this volume, chapter 12; Leu et al. 2008; Leu and Hanser, this volume, chapter 13).

The SGCA includes Küchler's (1970) three sagebrush vegetation types in addition to a portion of the northern Great Plains, which also supports stands of upright woody sagebrush. Sagebrush is a dominant land cover across much of the unforested parts of this region. However, many areas now contain only islands of sagebrush habitats embedded within larger expanses of highly altered landscapes. Sage-grouse have been extirpated from many of these islands (Schroeder et al. 2004; Aldridge et al. 2008; Wisdom et al., this volume, chapter 18); other sagebrush-dependent wildlife, such as Sage Sparrows (*Amphispiza belli*), Brewer's Sparrows (*Spizella breweri*), and Sage Thrashers (*Oreoscoptes montanus*), continue to use these sagebrush habitats (Wisdom et al. 2000b, Knick and Rotenberry 2002, Dobkin and Sauder 2004, Wisdom et al. 2005b). These remnants also may retain critical plant and wildlife components that could be valuable in restoration of adjacent potential or at-risk sagebrush communities (West 1996, West and Young 2000, Longland and Bateman 2002, Bunting et al. 2003). These remaining sagebrush islands continue to interact with other habitats in the landscape matrix by providing seed sources and habitat for resident and transient wildlife.

GEOGRAPHIC DISTRIBUTION OF SAGEBRUSH WITHIN THE GREATER SAGE-GROUSE CONSERVATION AREA

Sagebrush habitats are distributed throughout the SGCA (Fig. 10.1). The SGCA extends from British Columbia and Saskatchewan in the north to northern Arizona and New Mexico in the south, and from the eastern slopes of the Sierra Nevada and Cascade Mountains in the west to western North and South Dakota in the east (Fig. 10.1). Küchler (1964, 1970)(Fig. 10.2) separated sagebrush in this region into two potential natural vegetation types: (1) sagebrush steppe (type 55; Küchler 1970), where sagebrush is frequently a codominant with perennial bunchgrasses under potential natural conditions; and (2) Great Basin sagebrush (type 38), where sagebrush can often be the dominant plant

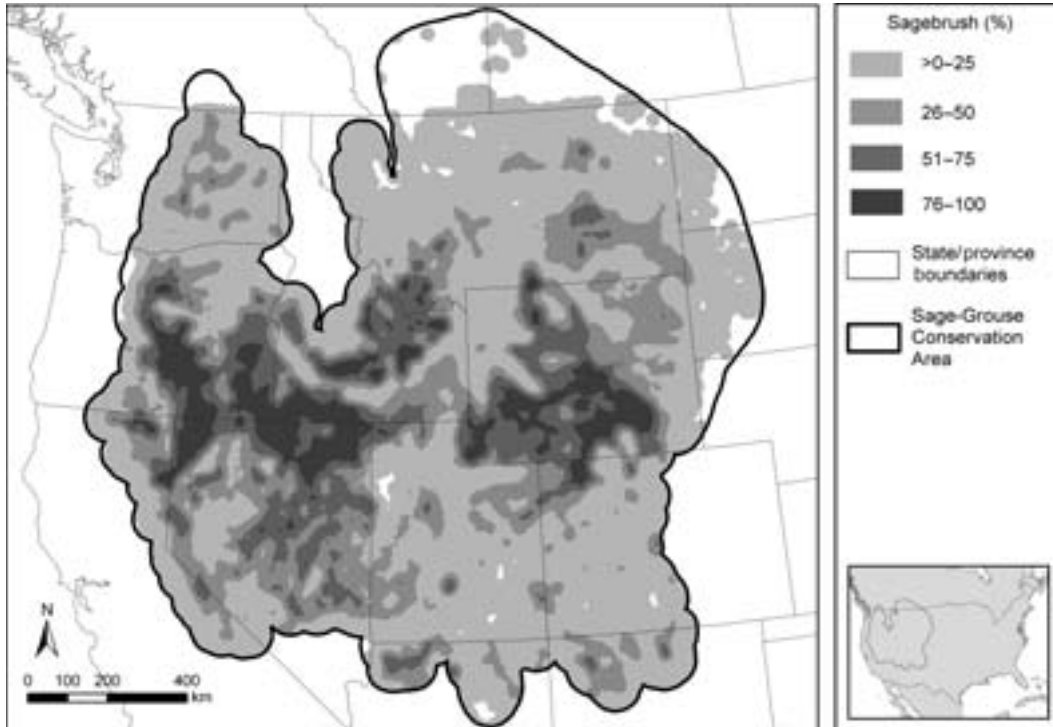


Figure 10.1. Current distribution of sagebrush habitats within western North America. The map represents the percent of the landscape dominated by sagebrush habitats and not site-specific values of ground cover. As such, it is intended to be a general representation of sagebrush distribution.

layer with a sparse understory. These two vegetation types, which exclude the Great Plains region, occupy >500,000 km² (Table 10.1)(Küchler 1970). West (1983a), naming Küchler's Great Basin sagebrush type the Great Basin–Colorado Plateau sagebrush semidesert, addressed some of the differences between these two subregions. Sagebrush steppe occupies parts of British Columbia, the Columbia Basin, northern Great Basin, Snake River Plain, Montana, Wyoming Basin, and northern Colorado (Fig. 10.2). The Great Basin sagebrush type lies to the south below the polar front gradient (Miller and Eddleman 2001) where temperatures are warmer, summer precipitation increases, and winter precipitation decreases (Mitchell 1976). The Great Basin sagebrush type includes portions of the Colorado Plateau and extends across Nevada, Utah, southeastern Colorado, northern Arizona, northern New Mexico, and central-eastern California. A third vegetation type, the mixed desert shrubland (Knight 1994; Küchler's type 56) occurs in the Bighorn Basin in north-central Wyoming (Fig. 10.2). Portions of the northern Great Plains that Küchler (1964, 1970) mapped as *Gramineedgrass-wheatgrass* (type 64) in eastern

Montana and eastern Wyoming (Fig. 10.2) support plains silver sagebrush (*Artemisia cana* ssp. *cana*) and sand sagebrush (*A. filifolia*).

Geographic subdivisions across the SGCA have been delineated into Sage-Grouse management zones based on general similarities in climate, elevation, topography, geology, soils, and floristics (West 1983b, Miller and Eddleman 2001)(Fig. 10.3). Sagebrush habitats in the Columbia Basin, Northern Great Basin, Snake River Plain, Wyoming Basin, Southern Great Basin, and Silver Sagebrush floristic provinces (Fig. 10.3) are of primary importance to Greater Sage-Grouse (*Centrocercus urophasianus*). Boundaries of floristic provinces were used as general guides for delineating the current Sage-Grouse Management Zones (Stiver et al. 2006).

SAGEBRUSH TAXA

Intermountain Region

Shultz (2009) recognized 13 species and 12 subspecies in the genus *Artemisia* subgenus *Tridentate*. The most predominant sagebrush taxa, which provide important annual and seasonal

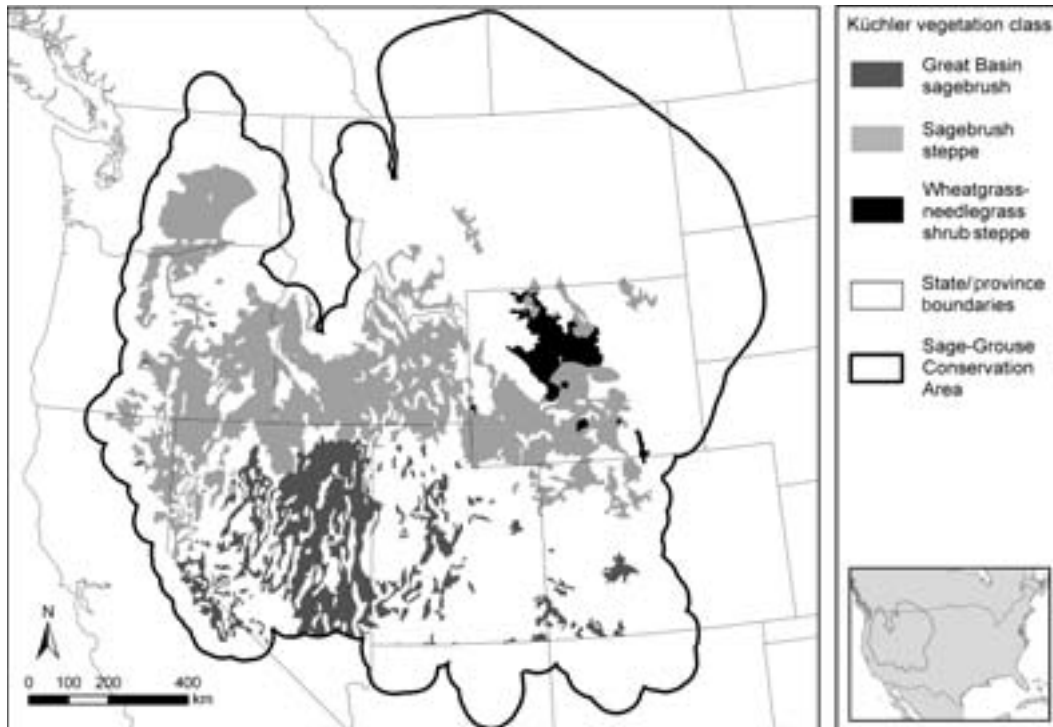


Figure 10.2. Küchler map (1970) of potential sagebrush distribution. The map illustrates the potential distribution of Great Basin sagebrush (type 38), sagebrush steppe (type 55), and wheatgrass-needlegrass mixed desert shrubland (type 56) vegetation types within the Sage-Grouse Conservation Area. The vegetation classes represent the potential vegetation that would be present in the absence of natural or human disturbance.

habitats for sage-grouse across the SGCA, are three subspecies of *Artemisia tridentata*, represented by Wyoming big sagebrush (*A. t. ssp. wyomingensis*), basin big sagebrush (*A. t. ssp. tridentata*), and mountain big sagebrush (*A. t. ssp. vaseyana*); two low (or dwarf) forms of sagebrush, little sagebrush (*A. arbuscula*) and black sagebrush (*A. nova*); and silver sagebrush (Connelly et al. 2000c, Crawford et al. 2004). The abundance and frequency of occurrence of sagebrush taxa characterizing different ecological sites is influenced primarily by soils, climate, topographic position, and disturbance history (West 1983b)(Fig. 10.4). An ecological site is defined as having specific physical characteristics that differ from other sites in the ability to produce distinctive compositions and amounts of vegetation and in response to management.

The three subspecies of big sagebrush usually occur on well-drained, moderately deep, sandy to clay loam soils. Wyoming big sagebrush typically occurs on warmer, drier sites at elevations between 150 and 2,140 m in valleys and foothills (Mahalovich and McArthur 2004). Soils are often underlain

by an argillic, caliche, or silica layer. Basin big sagebrush also occurs in the valleys and foothills on deep (often ≥ 1 m), dry, fertile soils. Mountain big sagebrush occurs on relatively cooler sites in foothills and mountains on moderate to deep well-drained soils where summer moisture is available. It typically occurs at elevations varying from 1,200 to 3,100 m.

The low (or dwarf) forms of sagebrush, including little sagebrush, black sagebrush, and rigid (stiff or scabland) sagebrush (*A. rigida*), generally occur on shallow or poorly drained soils (Eckert 1957, Fosberg and Hironaka 1964). A strong argillic horizon, duripan, or bedrock in these areas that keeps water from draining generally is present < 33 cm from the surface or < 50 cm in wet areas. Depth to the wetting horizon is usually limited when black or little sagebrush occur on deeper soils, and soils are coarse textured (Fosberg and Hironaka 1964, Sabinske and Knight 1978, Tisdale 1994). Three subspecies of little sagebrush grow in the western portion of the sagebrush region extending east into western Wyoming (Cronquist et al. 1994, Mahalovich and McArthur 2004). Little

TABLE 10.1

Area occupied by sagebrush in the Intermountain region estimated from Kuchler's map of potential vegetation (Kuchler 1970, West 1983b).

The regional boundaries used for these area estimates were delineated in Kuchler's map (1970) of potential vegetation (Fig. 10.2) and did not include eastern parts of the Sage-Grouse Conservation Area.

Type/state	Area (km ²)	% of type total
Sagebrush steppe		
Wyoming	99,642	25.5
Idaho	96,162	24.6
Oregon	75,728	19.3
Nevada	38,498	9.8
Washington	34,478	8.8
Colorado	14,399	3.7
California	12,558	3.2
Montana	11,089	2.8
Utah	8,936	2.3
Total	391,490	100.0
Great Basin sagebrush		
Nevada	96,908	70.8
Utah	26,040	19.0
Colorado	7,606	5.6
California	6,373	4.7
Total	136,927	100.0

sagebrush (*Artemisia arbuscula* spp. *arbuscula*) occurs from western Wyoming and Colorado to south-central Washington to northern California between elevations of 700 and 3,780 m. Lahontan little sagebrush (*A. a.* spp. *longicaulis*) occupies elevations between 1,050 and 2,000 m in northwest Nevada and neighboring Oregon and California (in the vicinity of old Lake Lahontan). Hot springs little sagebrush (*A. a.* spp. *thermophila*) occurs across western Wyoming, northern Utah, and eastern Idaho between 1,800 and 2,500 m. Black sagebrush extends farther south and east than little sagebrush and commonly occurs on calcareous soils between 625 and 2,990 m elevation. Early sagebrush (*A. longiloba*) was recently recognized as a fourth subspecies of little sagebrush (Shultz 2009). Early sagebrush flowers earlier than other low-statured sagebrush and is an important taxa for sage-grouse. It also is one of the most palatable sagebrush

species and often is heavily browsed (Winward 2004, Rosentreter 2005). Early sagebrush occurs from North Park, Colorado, to central Oregon and central California. Sagebrush taxa in Nevada follow an increasing gradient of soil fertility from black sagebrush, little sagebrush, early sagebrush, Wyoming big sagebrush, basin big sagebrush, and mountain big sagebrush (Jensen 1989a).

Northern Great Plains

The primary *Artemisia* species in the northeastern range of Greater Sage-Grouse, including northeastern Wyoming, eastern Montana, southeastern Alberta, southwestern Saskatchewan, and the extreme western portions of South Dakota, southwest North Dakota, and northwest Nebraska are Wyoming and basin big sagebrush, prairie sage-wort (or fringed sagebrush [*A. frigida*]), plains

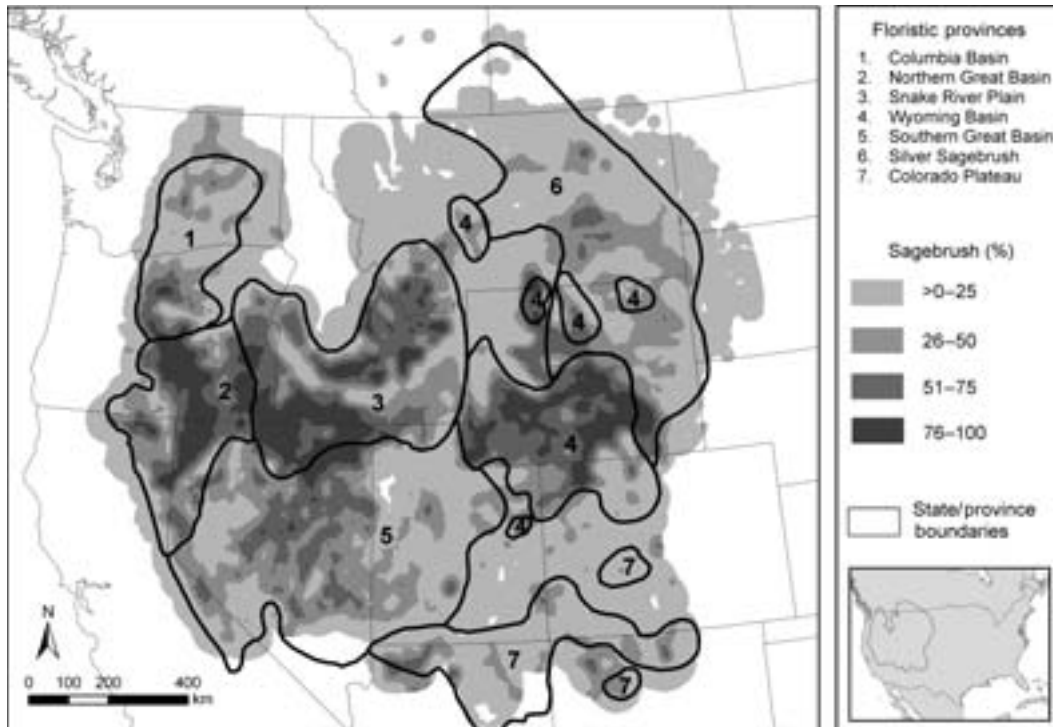


Figure 10.3. Geographic subdivisions within the sagebrush steppe are (1) Columbia Basin, (2) Northern Great Basin, (3) Snake River Plain, and (4) Wyoming Basin. The Great Basin includes (5) Southern Great Basin and (7) Colorado Plateau. The Northern Great Plains grasslands overlap the Silver Sagebrush (6) subdivision (derived from West 1983b, Küchler 1970, Miller and Eddleman 2001, and this study). Percent sagebrush habitat is the general landscape distribution of sagebrush.

silver sagebrush, and sand sagebrush. Wyoming big sagebrush is the most common subspecies of the *tridentata* group in this region, and typically occurs in marine shales and upland soils (Morris et al. 1976, Wambolt and Frisina 2002).

Prairie sagewort, a low-growing subshrub, is widely distributed and characteristic of the high plains of central North America extending west into south-central Idaho, eastern Washington, and central Utah and east throughout eastern Montana (Morris et al. 1976, Wambolt and Frisina 2002). Prairie sagewort grows in dry, open sites from plains and foothills to middle and upper elevations in the mountains up to 3,400 m and is common on disturbed sites (Bai and Romo 1996).

Plains silver sagebrush is widespread throughout the northern Great Plains at 1,200- to 2,100-m elevation and occupies well-drained, coarse-textured soils on alluvial flats, terraces, valley bottoms, and drainage ways. In Montana, plains silver sagebrush is distributed primarily throughout the central and eastern portions of the state and is the most common upright shrubby *Artemisia* species in the north and northeastern

plains (Morris et al. 1976). The silver sagebrush–western wheatgrass (*Pascopyrum smithii*) type is of major importance throughout this region.

Sand sagebrush is a widespread but low-abundance species that commonly grows in dunes and coarse soils. The species has an extensive distribution across the Great Plains, southward from the Black Hills to the Texas panhandle, and west through New Mexico and Utah to Arizona and Nevada (McKean 1976). Sand sagebrush is associated with Lesser Prairie-Chicken (*Tympanuchus pallidicinctus*) habitat in Colorado, Kansas, and Oklahoma (Cannon and Knopf 1981, Pitman et al. 2006). It is important to note that, unlike the big sagebrush subspecies, both sand and silver sagebrush are capable of resprouting following fire.

CLASSIFICATION OF ECOLOGICAL COMMUNITIES WITHIN SAGEBRUSH SYSTEMS

Sagebrush habitats within the SGCA have been described and mapped using the International Classification of Ecological Communities, which

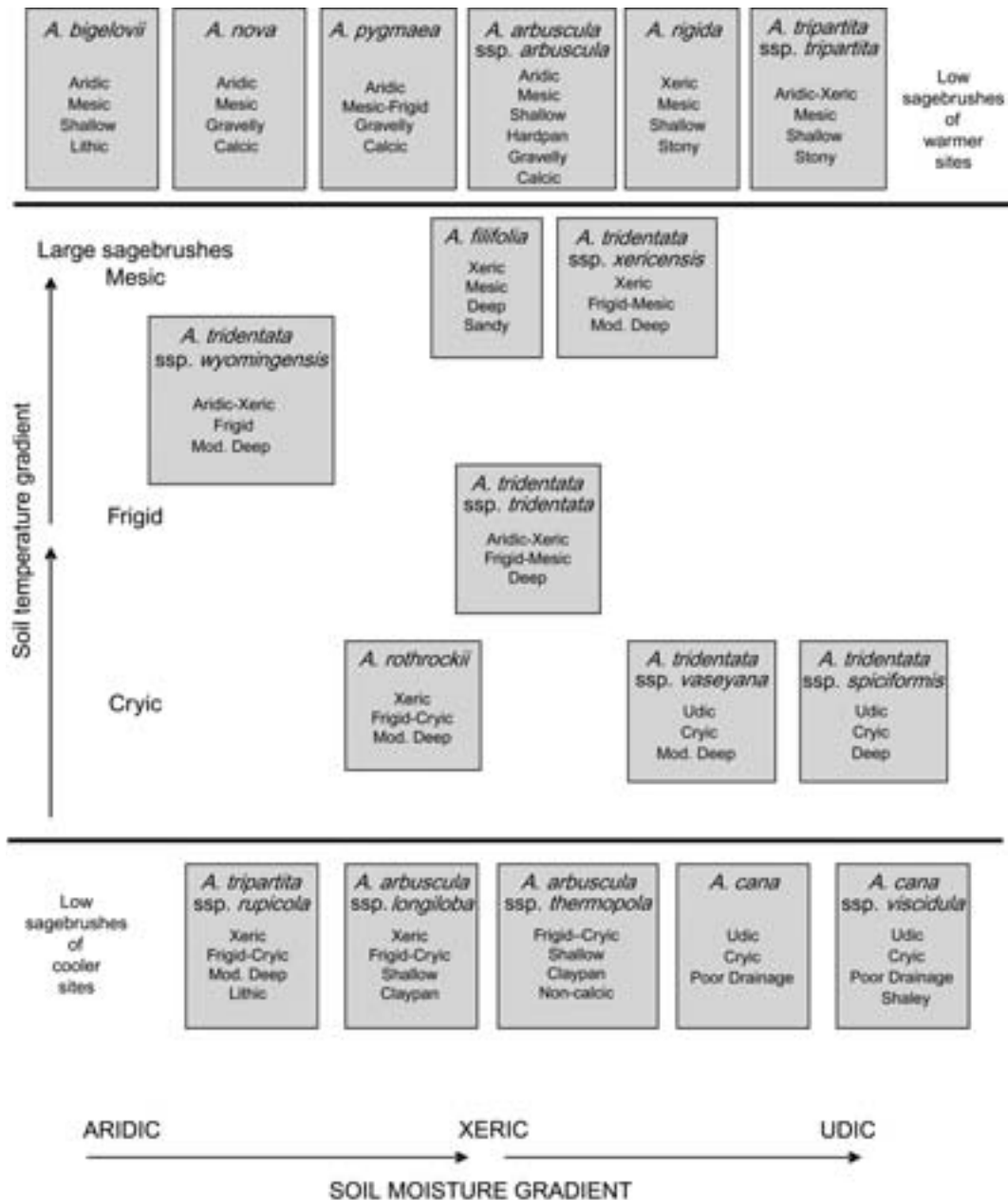


Figure 10.4. Ordination of major sagebrush taxa in the Intermountain Region against gradients of soil temperature and soil moisture (adapted from West and Young 2000; with additions from Robertson et al. 1966, McArthur 1983, and this study). Sagebrush species not shown were prairie sagewort (*Artemisia frigida*), Owyhee sage (*A. papposa*), birdfoot sagebrush (*A. pedatifida*), and bud sagebrush (*Picrothamnus desertorum*).

separates sagebrush communities based on floristics into alliances and plant associations (Reid et al. 2002). Sagebrush alliances are delineated by a species or subspecies of sagebrush and, at times, a second diagnostic shrub that may include antelope bitterbrush (*Purshia*

tridentata), snowberry (*Symphoricarpos* spp.), and saltbush (*Atriplex* spp.). Rabbitbrush (*Chrysothamnus* spp. and *Ericameria* spp.), which often increase with disturbance, are typically not used as a diagnostic species distinguishing plant alliances.

Plant associations, which further separate sagebrush alliances, are usually delineated by understory perennial grass. Tussock-forming grasses typically dominate the undergrowth in the intermountain region. Common examples include Idaho fescue (*Festuca idahoensis*), bluebunch wheatgrass (*Pseudoroegneria spicata*), Thurber's needlegrass (*Acnatherum thurberianum*), needle and thread (*Hesperostipa comata*), Columbia needlegrass (*Acnatherum nelsonii*), western needlegrass (*A. occidentale*), California brome (*Bromus carinatus*), squirreltail (*Elymus elymoides*), and Sandberg bluegrass (*Poa secunda*).

Sagebrush–grass type alliances in the northern Great Plains are composed of big sagebrush and silver sagebrush often mixed with rabbitbrush, saltbush, or winterfat (*Krascheninnikovia lanata*). The understory is dominated by rhizomatous grasses. Common understory grasses associated with sagebrush–shrub mixes are western or bluebunch wheatgrass, needle and thread, prairie Junegrass (*Koeleria macrantha*), and blue grama (*Bouteloua gracilis*). Prairie sandreed (*Calamovilfa longifolia*), sand dropseed (*Sporobolus cryptandrus*), and sand bluestem (*Andropogon hallii*) are commonly present in coarse soils, whereas alkali sacaton (*S. airoides*) is more abundant on moist bottomlands and more alkaline sites in association with greasewood (*Sarcobatus vermiculatus*). Sagebrush and other shrubs in these associations generally are more scattered and reduced in stature on the plains where wheatgrasses dominate the visual aspect. Sand sagebrush–bluestem associations commonly occur on sandy soils within the northern Great Plains. Sand sagebrush may be associated with mid and tall grasses, including sand bluestem, little bluestem (*Schizachyrium scoparium*), and prairie sandreed.

Ecological site descriptions and state-and-transitional models are currently being developed across shrublands in the western United States. These descriptions use soil physical characteristics in combination with environmental factors, hydrology, and vegetation as interrelated components to describe land units with particular kinds and amount of vegetation and their disturbances and potential successional transitions and pathways (United States Department of Agriculture 2003). In addition, ecological site descriptions incorporate temporal changes associated with management, disturbance, and subsequent plant community development on the site. Ecological site descriptions are based on physical site characteristics and

recognize the transitions between vegetative physiognomic states; thus, they are particularly well suited to sagebrush-dominated landscapes. Such site descriptions allow for the variety of plant dominance possible in shrubland systems following fire, invasive species, and site disturbances. One or more sagebrush plant associations may occupy an ecological site at different times.

The structure of the sagebrush steppe and Great Basin sagebrush vegetation types is typically characterized by four layers: (1) shrubs 0.3–1.0 m tall, (2) forbs and caespitose grasses 0.2–0.6 m tall, (3) low-growing grasses and forbs <0.2 m tall, and (4) biological soil crust. Potential plant cover varies widely with soils and moisture availability ranging from considerable exposed bare ground, but it can approach 100% in wet sagebrush communities. Biological soil crusts typically increase on more arid sites where vascular plant cover is low (Ponzetti et al. 2007). These crusts are less common in the eastern plains because vascular plant cover often increases.

Annual herbaceous production varies widely within and among communities dominated by different sagebrush species and subspecies, ranging from 120 to 2,350 kg/ha (Table 10.2). This is due primarily to moisture availability during the growing season, which in turn is related to climate, soils, and topographic position. The average number of days in Nevada when soil temperature and moisture were not limiting to herbaceous plant growth across 372 range sites varied from 28–32 days on little sagebrush sites, to 50–56 days on mountain big sagebrush sites, to 130 days on meadows (Jensen 1989b).

Species composition of vascular plants in sagebrush communities is strongly influenced by moisture availability and edaphic characteristics (Passey et al. 1982, Barker and McKell 1983, Jensen et al. 1990), particularly soil texture, nutrients, and depth to Bt horizon (Davies et al. 2007a). Floristic diversity is usually considered as moderate (West 1983b). Species numbers on sites with minimal disturbance ranged from 20 in the Columbia Basin in central Washington (Daubenmire 1975a), to 13–24 in the Snake River Plain (Tisdale et al. 1965), to 54 across several sagebrush communities in Nevada (Zamora and Tueller 1973), to 24–56 in mountain big sagebrush communities in the northern Great Basin (Miller et al. 2000). Forb species usually outnumber grass species, but forbs often constitute a much smaller portion of the biomass and ground

TABLE 10.2

Ranges of annual precipitation, elevation, soil depth, and aboveground annual herbaceous production for communities with a dominant overstory of common *Artemisia* species and subspecies.

Species/subspecies	Annual precipitation (mm)	Elevation (m)	Site adaptation and soils	Annual productivity (kg/ha)
<i>Artemisia tridentata</i> spp. <i>spiciformis</i>	>400	2,300–3,200	High mountain areas	>1,850
<i>A. t.</i> spp. <i>vaseyana</i>	350–450	1,200–3,200	Moderate to deep summer moisture	1,120–3,080
<i>A. t.</i> spp. <i>tridentata</i>	200–400	610–2,140	Deep, dry fertile soils	868–2,350
<i>A. t.</i> spp. <i>wyomingensis</i>	180–300	150–2,150	Shallower to moderately deep; hotter than <i>A.t.t.</i> , <i>A.t.v</i>	490–990
<i>A. tripartita</i>	300–400	1,100–2,300	Rocky knolls to moderately deep	560–1,370
<i>A. arbuscula</i>	200–400	700–3,780	Shallow, often alkaline, or if deep, usually an abrupt textural change between the A and B horizons	370–1,000
<i>A. nova</i>	200–300(400)	625–2,990	Shallow, often calcareous	440–620
<i>A. rigida</i>	200–400	230–2,130	Shallow, rocky scablands	120–250
<i>A. cana</i> spp. <i>cana</i>		up to 3,300	Well-watered, deep soils along stream bottoms and drainages	NA

SOURCE: Adapted from Miller and Eddleman 2001; and derived from Passey et al. 1982, Cronquist et al. 1994, Shiflet 1994, and Mahalovich and McArthur 2004.

cover. Mean perennial forb cover in five minimally disturbed Wyoming big sagebrush plant associations across southeast Oregon was usually <5%, compared to 10–20% cover of perennial grasses (Davies et al. 2007a). Forb abundance can be highly variable from year to year and is largely affected by amount and timing of precipitation.

TEMPORAL SCALES OF SAGEBRUSH DYNAMICS

Long-Term Dynamics

Long-term dynamics of sagebrush ecosystems extend over centuries or millennia. Pre-settlement shifts in potential natural vegetation were caused

primarily by long-term changes in climate and severe disturbances (e.g., volcanic eruptions and floods), resulting in a change in plant associations, alliances, and disturbance regimes. Climate has fluctuated since the end of the Pleistocene 10,000 years before present (BP), with cooler and wetter, warmer and drier, and warmer and more humid periods (Antevs 1938, 1948; Bright and Davis 1982; Davis 1982). The duration of these periods extended from centuries to several millennia and resulted in changes in abundance between sagebrush and graminoids, and the distribution of pinyon (*Pinus* spp.), juniper (*Juniperus* spp.), sagebrush, grassland, and salt desert communities (Mehring 1985, Mehring and Wigand 1987, Wigand 1987).

Severe drought and major fires followed the Neoglacial during the beginning of the late Holocene (2,500 BP) in the northern Great Basin and resulted in rapid regional declines in juniper and perennial grasses, and expansion of sagebrush at the upper elevations and salt-desert shrub at lower elevations (Mehring 1985, Mehringer and Wigand 1987, Wigand 1987). Examination of charcoal layers, pollen cores, and sediments indicate that frequent large fires in combination with climate were primary drivers of pinyon and juniper abundance and distribution (Miller et al. 2001). The Little Ice Age (700–150 years BP), which ended just prior to Eurasian settlement, was the wettest and coolest period during the last half of the Holocene. A general warming trend has occurred since the end of the Little Ice Age (ca. 1850) 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 that were synchronous with conifer expansion rates that exceed any that have occurred during a similar length of time in the southern and northern Great Basin (Miller and Wigand 1994, Miller and Tausch 2001, Miller et al. 2008).

Short-Term Dynamics

Short-term changes, usually calculated in years or decades, are a function of weather and disturbance (e.g., fire, diseases, molds, insects, and changes in herbivory) resulting in fluctuation or permanent change in relative abundance of species and structure of plant communities. Short-term climatic cycles measured in years can 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. Vegetation composition and structure that is most persistent through time within a defined climate regime or ecological site is often affected by severity and frequency of disturbance events (Miller and Heyerdahl 2008). There are two potential outcomes resulting from disturbance or climatic change: (1) plant communities shift within their range of natural variability (e.g., succession from one phase to another within a steady state), or (2) they cross a threshold and shift to a new steady state. A phase is defined as a plant community within a state that

is hypothesized to replace other communities along traditional succession-retrogression pathways; succession from one community to the next is readily reversible over short time periods (years to decades) without management intervention because they are not separated by thresholds. However, an at-risk community phase may not progress directly to the most resilient community phase without passing through an intermediate phase. A state is a suite of plant community successional phases occurring on similar soils that interact with the environment to produce resistant functional and structural attributes with a characteristic range of variability maintained through autogenic repair mechanisms. Shifts between multiple stable states represent a transition across a threshold that requires large changes to return the site to the previous state (Westoby et al. 1989b, Bestelmeyer et al. 2003, Briske et al. 2003).

CURRENT AND POTENTIAL DISTRIBUTION OF SAGEBRUSH HABITATS

Accurate estimates of the amount of sagebrush habitat that has been lost from what was present during pre-settlement are not possible because of our inability to map the historical distribution with an accuracy or resolution comparable to that in modern satellite image data. Therefore, we estimated the difference between the area that could be dominated by sagebrush in Küchler's (1970) potential vegetation map for Great Basin, sagebrush steppe, and wheatgrass-needlegrass shrub steppe (Fig. 10.2) to the current distribution of sagebrush habitats (Fig. 10.1). Küchler's map depicts the vegetation that would occur if there were no disturbances from humans or nature (Küchler 1964). Therefore, our analysis compared the difference between the vegetation type that could potentially occur and what currently was present (LANDFIRE 2006). We used only regions included in Great Basin sagebrush, sagebrush steppe, and wheatgrass-needlegrass shrub steppe types. We recognize that some of the difference is a function of the coarse resolution in Küchler's map compared to the finer resolution in the sagebrush map. We subtracted forested, water, marsh, and wetland habitats delineated in the map of current habitats (LANDFIRE 2006) from the total area for each sagebrush type in Küchler's (1970) map to partially correct for differences in thematic and spatial resolution. We

emphasize that the analysis only identified broad-scale differences between current and potential distribution in sagebrush and was not intended to identify specific locations where sage-grouse habitat had been lost. Additionally, because of scale differences between Küchler's (1970) map and our habitat map, we cannot clearly distinguish where Küchler's broad delineations may have masked smaller parcels of nonhabitat.

Fifty-five percent of the area delineated on Küchler's maps as potentially dominated by sagebrush across Washington, Montana, Wyoming (sagebrush habitats in eastern portions of Montana and Wyoming were not included), Idaho, Oregon, Nevada, Utah, California, and Colorado are currently occupied by sagebrush (Fig. 10.5, Table 10.3). Wyoming (66%) and Oregon (65%) have the highest portion of potential area that is currently mapped as sagebrush, whereas Utah (38%) and Washington (24%) had the lowest. Within the areas not currently mapped as sagebrush, agriculture made up the largest category of land cover (10% of the potential area) and was the

dominant land cover within potential sagebrush areas in Washington (42%) and Idaho (19%). Urban areas covered 1% of the potential sagebrush areas. The remaining 31% of potential sagebrush vegetation was mapped as barren, grassland, burn, exotic grassland, shrubland, and juniper woodland.

Previous estimates of potential sagebrush vegetation currently designated as urban, agriculture, or converted to land-cover types that no longer can support sagebrush vegetation were 3% for the Great Basin sagebrush type, 5% for wheatgrass-needlegrass-shrub steppe, and 15% for sagebrush steppe (Klopatek et al. 1979). Based on updated maps of urban and agriculture areas (and corrected for other nonsagebrush habitats), we estimated that 5% (6,293 km²) of the area occupied by Great Basin sagebrush now was in agriculture, urban, or industrial areas; 46% (63,379 km²) still supported sagebrush; and 49% (67,635 km²) was dominated by other vegetation types. In the wheatgrass-needlegrass-shrub steppe, 5% (1,339 km²) of the potential area that could support sagebrush

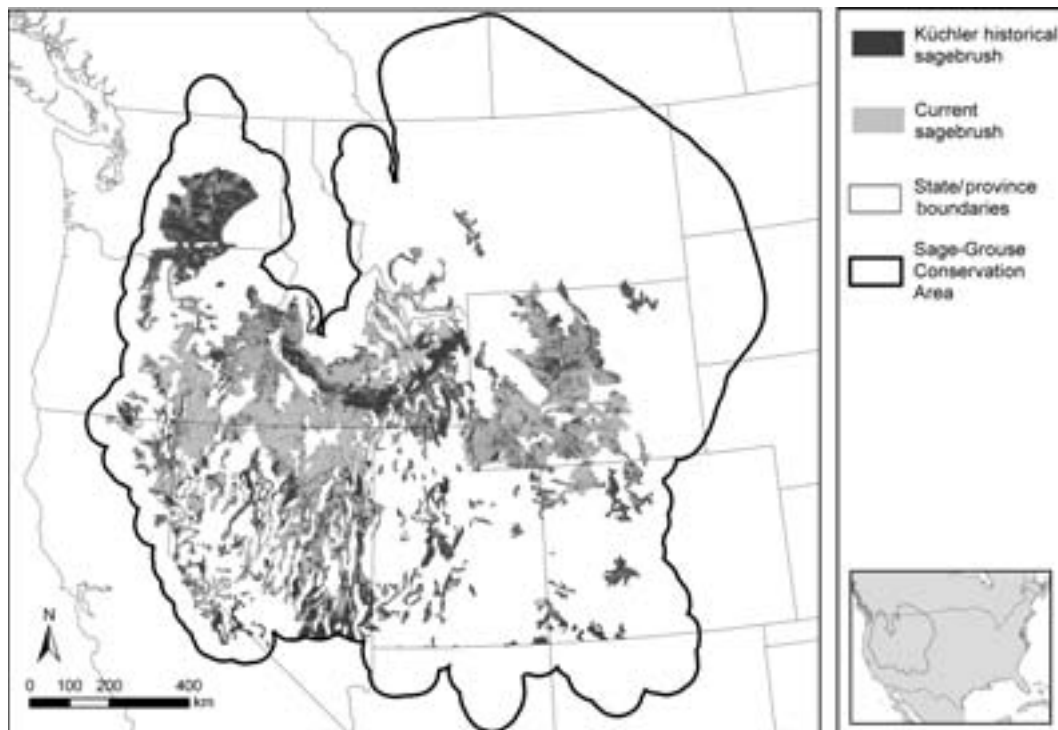


Figure 10.5. Difference between Küchler's map (1970) of potential sagebrush distribution (Fig. 10.2) and current distribution of sagebrush (Fig. 10.1). Only the distribution of Küchler's categories for Great Basin sagebrush, sagebrush steppe, and wheatgrass-needlegrass shrub steppe are used in this analysis. Sagebrush habitats also exist outside of the distribution of these habitat types.

TABLE 10.3

Percent of sagebrush in Küchler's (1970) map of potential vegetation (Fig. 10.2) that is currently in sagebrush habitat (Fig. 10.1).

Only comparisons for Küchler's categories of Great Basin sagebrush (type 38), sagebrush steppe (type 55), and wheatgrass-needlegrass shrub steppe (type 56) were used in this analysis. The remaining percentage consisted of habitat categories describing barren, burn, grassland, exotic grassland, non-sagebrush shrublands, and juniper woodland habitats (derived from the LANDFIRE 2006 Existing Vegetation Map).

State	Potential area (km ²)	Current condition	
		Sagebrush km ² (%)	Agriculture km ² (%)
Washington	34,478	8,158 (23.7)	14,618 (42.4)
Montana ^a	11,089	6,229 (56.2)	834 (7.5)
Wyoming ^a	99,642	65,809 (66.0)	3,391 (3.4)
Idaho	96,162	52,853 (55.0)	17,896 (18.6)
Oregon	76,107	49,062 (64.5)	6,582 (8.6)
Nevada	135,406	79,540 (58.7)	1,797 (1.3)
Utah	34,976	13,147 (37.6)	3,399 (9.7)
California	18,931	9,419 (49.8)	1,513 (8.0)
Colorado ^a	22,005	8,932 (40.6)	2,560 (11.8)
Total	528,796	293,194 (55.4)	52,619 (10.0)

^a Does not include sagebrush lands in eastern portions of the state outside the boundaries of the Küchler's vegetation types used in this analysis.

vegetation has been converted to agriculture, urban, or industrial habitats; 59% (15,864 km²) currently was mapped as sagebrush, and 36% (9,601 km²) was occupied by other vegetation types. In the sagebrush steppe, 14% (52,865 km²) of the area that could potentially support sagebrush has been converted to agriculture, urban, or industrial categories; 59% (213,908 km²) was occupied by sagebrush habitat; and 27% (97,913 km²) has been converted to other vegetation types. These analyses are based only on dominant land cover across large regions. Information about understory, soil, and other characteristics not mapped in satellite imagery or captured in coarse-resolution maps is not implied.

PRIMARY DISRUPTERS OF SAGEBRUSH HABITAT DYNAMICS

Invasive Species

A broad array of invasive plants influences the structure and function of habitats used by Greater Sage-Grouse (Table 10.4). An invasive species is defined as an exotic or native species that is non-

native to the specific ecosystem under consideration and whose introduction causes or is likely to cause economic or environmental harm or harm to human health (Clinton 1999); this definition also includes species native to other parts of North America; however, species that occur within the region are classified as increasers. Sheley and Petroff (1999) list 29 species of rangeland weeds. The Interior Columbia Basin Ecosystem Management Project compiled a similar list of 25 species (United States Departments of Agriculture and the Interior 1997a,b). With the exception of the snakeweeds (broom snakeweed [*Gutierrezia sarothrae*], threadleaf snakeweed [*G. microcephala*], and poison hemlock [*Conium maculatum*]), which are native to the western United States (Sterling et al. 1999) and considered increasers, the remaining species on these lists are defined as nonnative invasive in one or more of the sagebrush communities important to Greater Sage-Grouse. No scientific reports, models, or maps currently exist to provide a comprehensive list of the susceptibility of habitats within the SGCA to invasion by these weeds. Estimates of susceptibility (Table 10.4) to

TABLE 10.4

Major vegetation cover types within the Sage-Grouse Conservation Area and their susceptibility to invasion by nonindigenous plant species. Susceptibility to invasion is defined by four categories: (H) high^a, (M) moderate^b, (L) low^c, and (U) unknown^d.

Invasive species	Upland communities					
	Basin big sagebrush, Wyoming big sagebrush, three-tip sagebrush	Mountain big sagebrush	Low sagebrush, black sagebrush	Salt-desert shrub	Wheatgrass, bunchgrass	
Cheatgrass (<i>Bromus tectorum</i>)	H	M	M	M	H	
Musk thistle (<i>Carduus nutans</i>)	U	M	U	M	M	
Whiteweed (<i>Cardaria</i> spp.)	M	M	M	M	M	
Diffuse knapweed (<i>Centaurea diffusa</i>)	M	M	M	L	H	
Spotted knapweed (<i>C. maculosa</i>)	M	M	U	L	H	
Russian knapweed (<i>C. repens</i>)	M	M	U	M	M	
Yellow starthistle (<i>C. solstitialis</i>)	M	M	M	L	H	
Squarrose knapweed (<i>C. virgata</i>)	M	M	M	M	M	
Rush skeletonweed (<i>Chondrilla juncea</i>)	M	M	U	L	M	
Oxeye daisy (<i>Chrysanthemum leucanthemum</i>)	U	U	U	L	M	
Canada thistle (<i>Cirsium arvense</i>)	M	M	M	M	H	
Bull thistle (<i>C. vulgare</i>)	M	M	M	M	M	
Poison hemlock (<i>Conium maculatum</i>)	L	L	L	L	L	
Common crupina (<i>Crupina vulgaris</i>)	L	M	L	L	M	
Leafy spurge (<i>Euphorbia esula</i>)	M	L	M	M	M	
Halogeton (<i>Halogeton glomeratus</i>)	M	M	M	H	M	
Orange hawkweed (<i>Hieracium aurantiacum</i>)	L	M	L	L	L	
Meadow hawkweed (<i>H. pratensis</i>)	L	L	L	L	L	

Dyer's woad (<i>Isatis tinctoria</i>)	H	L	H	L	H
Perennial pepperweed (<i>Lepidium latifolium</i>)	L	L	L	L	L
Dalmatian toadflax (<i>Linaria dalmatica</i>)	M	H	M	L	H
Yellow toadflax (<i>L. vulgaris</i>)	M	M	U	L	M
Purple loosestrife (<i>Lythrum salicaria</i>)	L	M	L	L	L
Scotch thistle (<i>Onopordum acanthium</i>)	M	L	U	L	M
Sulphur cinquefoil (<i>Potentilla recta</i>)	U	M	U	L	H
Mediterranean sage (<i>Salvia aethiopsis</i>)	H	M	U	L	H
Russian thistle (<i>Salsola kali</i>)	M	M	L	M	M
Tansy ragwort (<i>Senecio jacobaea</i>)	U	U	U	U	U
Sowthistles (<i>Sonchus</i> spp.)	M	M	M	M	M
Medusahead (<i>Taeniatherum caput-medusae</i>)	M	M	L	M	M

SOURCE: Compiled from United States Departments of Agriculture and the Interior 1997b, Sheley and Petroff 1999.

- a Invades the cover type successfully and becomes dominant or codominant even in the absence of intense or frequent disturbances.
- b Invades the cover type successfully because high intensity or frequency of disturbance alters the soil surface or removes the normal canopy cover.
- c The species typically does not invade the cover type because the cover type does not provide suitable habitat for the species.
- d Distribution records are limited and interpretation of the susceptibility would be difficult.

invasion are based upon the knowledge of experts and written descriptions of the types of vegetation communities where infestations or colonization populations currently exist.

Estimates of the size of infestations of any of these species are subjective because of the lack of a definition of what constitutes an infestation. For example, cited estimates of diffuse (white) knapweed (*Centaurea diffusa*) infestation in Idaho ranged from 410 to 5,670 km² (Roché and Roché 1999). Thus, it is extremely difficult to ascertain a reasonable estimate of the area of lands currently occupied or level of dominance by invasive plants within any area across the range-wide distribution of Greater Sage-Grouse. Sheley and Petroff (1999) and United States Departments of Agriculture and the Interior (1997a,b) relied heavily on distribution maps of counties in the five-state area (Idaho, Montana, Oregon, Washington, and Wyoming) covered by the Invaders Database (Rice 2004), where counties are considered 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.

Many of the species listed (Table 10.4) may be widely distributed across the SGCA, but infestations are localized because of the narrow environmental needs of the invasive species. For example, diffuse knapweed is estimated to reach its greatest competitiveness within shrub-grassland communities where antelope bitterbrush may dominate or codominate along the eastern side of the Cascade Range in Washington (Roché and Roché 1999).

Invasions into native plant communities also may be sequential, as the initial invaders are replaced by a series of new exotics or by species adapting to new habitats within their range (Young and Longland 1996). Areas that once were dominated by cheatgrass (*Bromus tectorum*) in some locations along the Snake River Plain and the Boise Front Range in Idaho have been replaced by medusahead (*Taeniatherum caput-medusae*). Rush skeletonweed (*Chondrilla juncea*), which originally was localized to disturbed areas in the drier sagebrush-grassland communities, now is invading areas dominated by medusahead (Sheley et al. 1999) and following wildfire (Kinter et al. 2007).

Exotic Annual Grasses

Cheatgrass and medusahead have become the most problematic of the exotic annual grasses within the SGCA. These Eurasian annual grasses were introduced in the 1890s and have continued to expand their range (Mack 1981). Both grasses are winter annuals that rely on winter precipitation to invade and dominate lands. They tend to be more dominant in the Intermountain West (Washington, Oregon, Idaho, Nevada, and Utah) than in the Rocky Mountain states that receive more summer precipitation (parts of Montana, Wyoming, and Colorado can have local infestations of cheatgrass or other *Bromus* species). In particular, cheatgrass invasion can result in a dominant near-monoculture in the more arid, lower-elevation, Wyoming big sagebrush communities (Chambers et al. 2007). Annual-dominated communities can be considered a new steady state (Laycock 1991) over much of eastern Washington, eastern Oregon, southern Idaho, Nevada, and Utah.

Cheatgrass likely was first introduced within the intermountain region of the United States via contaminated imports of grains and expanded along transportation routes and in locations of documented severe livestock grazing and reductions in native perennial grasses (Young and Evans 1973, Mack 1981). Cheatgrass reached most of its current range expansion during the 1930s (Billings 1990) but has continued to expand southward into the Mojave Desert (Hunter 1991) and eastward.

Cheatgrass has been a major factor in loss of Wyoming big sagebrush communities (Chambers et al. 2007). Medusahead is filling a similar niche in more mesic communities with heavier clay soils (Dahl and Tisdale 1975). Since the initial occurrences in the late 1800s and early 1900s, medusahead has continued to spread and occupy new locations (Miller et al. 1999). These communities now include little sagebrush and mountain big sagebrush communities, as well as some Wyoming big sagebrush communities at lower elevations.

Cheatgrass Distribution in the Intermountain Western United States

One estimate of land area in sagebrush ecosystems dominated by introduced annual grasses comes from a qualitative survey conducted by the Bureau of Land Management (BLM) in 1991 (Pellant and

Hall 1994). This survey covered 400,000 km² of BLM-managed lands in Washington, Oregon, Idaho, Nevada, and Utah. Cheatgrass and medusa-head now either dominate or have a significant presence (estimated >10% composition based on biomass) on 70,000 km² of public land within these five states.

Recent surveys using combinations of field data and remote imagery estimated that cheatgrass now dominates >20,000 km², or 7% of land cover, within a portion of the northern Great Basin (Bradley and Mustard 2005, 2006; Peterson 2005). Whisenant (1990) indicated that cheatgrass has become a major herbaceous species in the West, dominating over 400,000 km², but his estimate was actually a major overestimate and a misinterpretation of the original citation that indicates cheatgrass now dominates on many rangelands within 410,000 km² of potential steppe vegetation in the intermountain western United States (Mack 1981). This incorrect figure has been repeated in other prominent review papers on the topic (d'Antonio and Vitousek 1992).

We mapped the probability of presence of cheatgrass within five floristic provinces in the

Intermountain West. Using environmental variables from field surveys in Washington, Oregon, Idaho, Utah, and Nevada (Connelly et al. 2004), we developed a statistical function predicting the probability of presence by cheatgrass using logistic regression and Akaike's Information Criterion to select the best model (Burnham and Anderson 2002). Initial predictor variables in the model were elevation (m), slope (%), aspect (degree), annual precipitation (cm), depth to rock (cm), soil pH, salinity (mmhos/cm), and available water capacity (cm) (Meinke et al. 2009). We mapped the probability of cheatgrass presence across each floristic province based on the values of predictor variables within each 2-km grid cell (Fig. 10.6). A moderate to high probability of presence by cheatgrass was predicted for 281,000 km², or almost half of the area within the mapped region of the Intermountain West (Fig. 10.6) (Meinke et al. 2009). Approximately 65% the Great Basin ecoregion (1,500,000 km²) has environmental conditions suitable for moderate to high risk of cheatgrass invasion; 38% of the existing sagebrush was at moderate risk and 20% at high risk (Suring et al. 2005b).

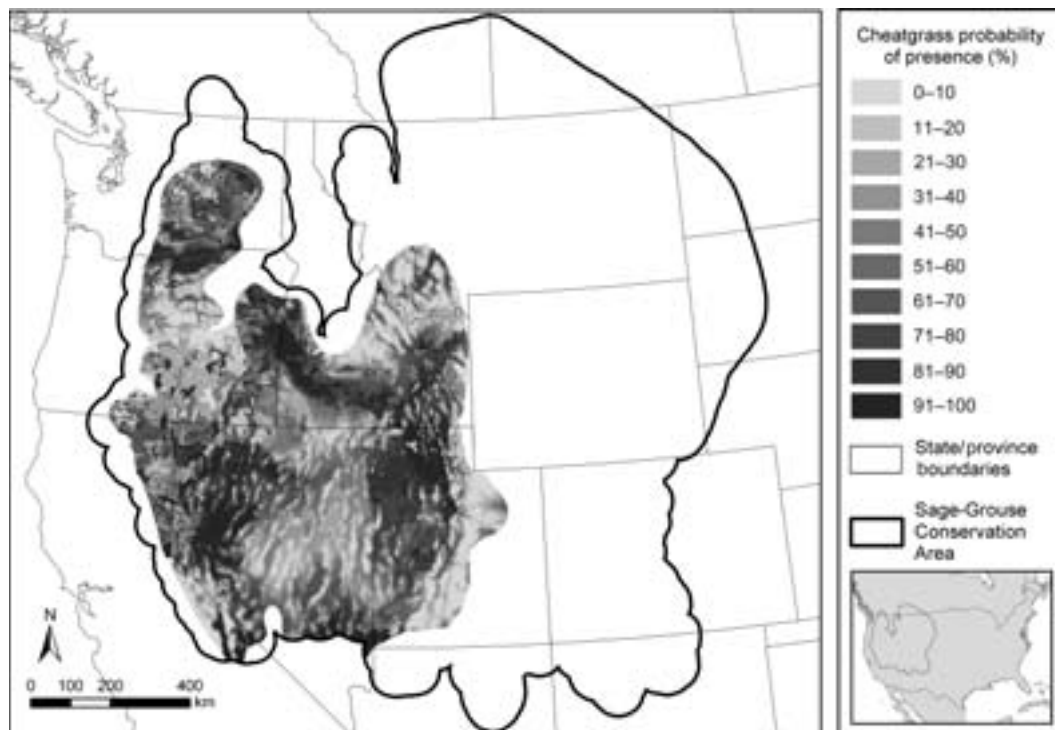


Figure 10.6. Predicted distribution of cheatgrass modeled from logistic regressions of presence/absence of cheatgrass and environmental factors measured at survey points (Meinke et al. 2009).

Postsettlement 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 SGCA and are encroaching and infilling into large portions of sagebrush steppe-dominated communities at higher elevations (Cottam and Stewart 1940, Tausch et al. 1981, Knapp and Soulé 1998, Tausch and Nowak 1999, Miller and Tausch 2001, Weisberg et al. 2007, Miller et al. 2008). To a lesser extent, Rocky Mountain juniper (*J. scopulorum*) also is expanding into sagebrush communities in portions of its range. The increase in woodlands after Euro-American settlement, which began in the late 1800s, is occurring at rates exceeding that of any expansions during the Holocene (Miller and Wigand 1994). The expansion of pinyon and juniper co-occurred with introduction of livestock and surface fire exclusion (Tausch et al. 1981, Miller and Rose 1999, Miller and Tausch 2001, Swetnam et al. 2001).

Cover of sagebrush and other associated shrubs decline with increasing tree dominance (Adams 1975, Miller et al. 2000, Roberts and Jones 2000, Schaefer et al. 2003)(Fig. 10.7). Juniper and pinyon woodlands occupy approximately 189,000 km² in the Intermountain West (Miller and Tausch 2001).

As much as 90% of the areas currently dominated by pinyon and juniper in the sagebrush steppe and Great Basin sagebrush vegetation types were predominantly persistent sagebrush vegetation types prior to the late 1800s (Tausch et al. 1981, Johnson and Miller 2006, Miller et al. 2008). However, the proportion varies spatially across the SGCA. A greater proportion of extensive old woodlands occupy the Colorado Plateau (Floyd et al. 2000, 2008; Eisenhart 2004) and the Mazama ecological province in central Oregon (Waichler et al. 2001, Miller et al. 2005). The greatest proportion of woodland encroachment has occurred at higher elevations in mountain big sagebrush plant associations and little and black sagebrush plant associations that occur on moderate to deep soils. Expansion and infill have also occurred in little and black sagebrush plant associations. Millions of hectares of potential sagebrush vegetation types are considered at high risk of displacement by juniper and pinyon woodlands (Suring et al. 2005b) by both encroachment and infill (Weisberg et al. 2007). We have limited documentation, but other conifer species such as Douglas fir (*Pseudotsuga menziesii*) have been expanding into mountain big sagebrush in Montana (Sindelar 1971, Dando and Hansen 1990, Hansen et al. 1995, Heyerdahl et al. 2006). Pinyon and juniper currently occupy far less land

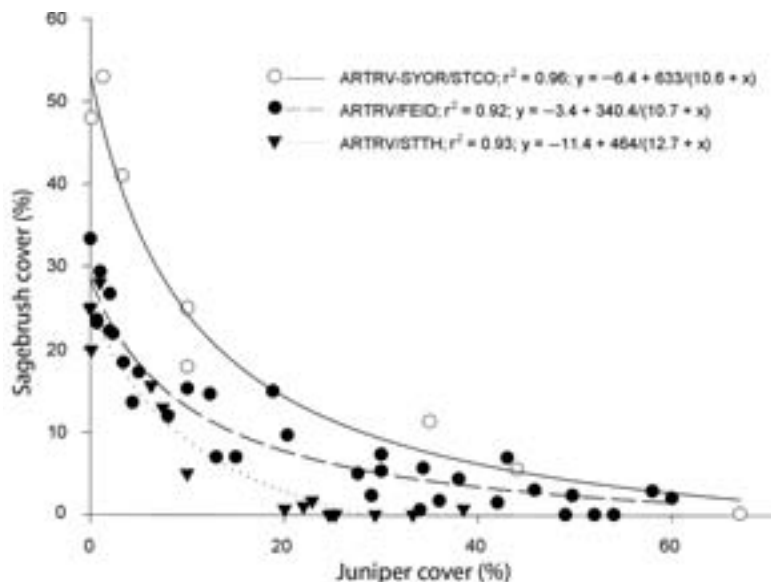


Figure 10.7. Relationship between juniper and mountain big sagebrush canopy cover in three plant associations: *Artemisia tridentata* ssp. *vaseyana*-*Symphoricarpos oreophilus*/*Stipa columbiana* (ARTRV-SYOR/STCO), *A. tridentata* ssp. *vaseyana*/*Festuca idahoensis* (ARTRV/FEID), and *A. tridentata* ssp. *vaseyana*/*Stipa thurberiana* (ARTRV/STTH) (Miller et al. 2000).

than their potential under current climatic conditions (Betancourt 1987, West and Van Pelt 1987, Miller et al. 2001). In addition, many of these woodlands are in different phases of development, where tree densities and cover are continuing to increase, resulting in the continued loss of sagebrush communities (Miller et al. 2000, 2008).

Mechanisms of Ecosystem Disruption

The effect of invasive species may be evaluated in how they alter the community structure (type, number, and relative abundance of species) and ecological function (nutrient, energy, and water cycles). In addition, invasive plant species can influence or disrupt the functional relationships among organisms in a community.

The altered structure of plant communities that results when invasive species replace native plant species can have significant consequences on the dynamics of sagebrush ecosystems. For example, the invasion of cheatgrass into the Wyoming big sagebrush alliance changes the structure of the understory to provide more complete and continuous ground cover in comparison to sparse, discontinuous cover of native perennial grasses that often occur in this alliance (Klemmedson and Smith 1964, Billings 1990). Consequently, these continuous and extremely flammable fuels result in fires in cheatgrass-dominated systems that are more frequent and often less complex, with few unburned patches. Sagebrush species are intolerant of fire and easily killed, with the exception of threetip (*Artemisia tripartita*), silver, and sand sagebrush, which are sprouters (Billings 1990, West and Young 2000). Reestablishment of most sagebrush species following fire depends on seed from the soil seed bank and dissemination from unburned areas in addition to appropriate conditions for germination and establishment (Hemstrom et al. 2002, Lesica et al. 2007). Seeds of mountain big sagebrush can remain in the seed bank over multiple years, compared to the seeds of Wyoming big sagebrush, which are not viable for more than 1 year unless covered by soil (Wijayratne and Pyke 2009). Dissemination of sagebrush seed is primarily from wind, the majority falling within 9–12 m (and under rare conditions, up to 33 m) of the parent plant (Blaisdell 1953, Mueggler 1956, Johnson and Payne 1968, Daubenmire 1975b, Frischknecht 1978). More complete fires in cheatgrass-dominated communities result in fewer, more

widely dispersed seed sources in remaining unburned islands. Cheatgrass is highly competitive for resources in the community, making it difficult for new perennial grass and shrub seedlings to establish (Harris 1967, Francis and Pyke 1996, Beckstead and Augspurger 2004, Chambers et al. 2007). Consequently, the primary mechanisms to reestablish native plants are severely compromised or eliminated. A sequence is started in which native species die, are not replaced and possibly eventually eliminated from the species pool.

A second structural change that may occur with infestations of invasive plants is a change in life-forms represented in the community. Elimination of woody plants, such as sagebrush, may be more permanent in communities dominated by annual grasses than if the herbaceous understory consisted of native perennial bunchgrasses and forbs. This structural alteration becomes extremely important for sagebrush-obligate animals that require sagebrush for nesting or sparse ground cover for foraging (Rotenberry 1998, Dobkin and Sauder 2004, Knick et al. 2005, Suring et al. 2005a, Beck et al. 2009).

A third form of structural change can occur belowground in the form of the composition and distribution of roots and mycorrhiza, which may affect nutrient cycling and organic matter content in the soil. Shrub grassland communities that are dominated by perennial plants have a mixture of shallow and deep roots, which are associated with mycorrhiza, and have varying forms of carbon that decompose at different rates. As a native shrub grassland community is converted to a community dominated by cheatgrass with few mycorrhiza associations, which has roots that contain fewer structural cells and decompose more quickly than woody plant roots, the distribution of amount and type of organic matter in the soil is changed. Surface soils of cheatgrass communities tend to contain more of the easily decomposed organic matter, while little of the recalcitrant forms of organic matter are found at any depth. In native shrub grasslands, less of the easily decomposed form exists near the surface and more of the recalcitrant form occurs throughout the soil depth (Norton et al. 2004).

Functional relationships may also change with conversion of a community from a diverse native plant system to dominance by an invasive plant. Postfire communities dominated by cheatgrass become net sources of carbon to the atmosphere because of decreased net carbon exchange compared to native sagebrush communities, which are

net carbon sinks (Prater et al. 2006). Communities dominated by cheatgrass also have lower evapotranspiration compared to native sagebrush systems; lower soil surface moisture and increased surface temperatures can increase the potential severity of summer droughts and stress to native plants (Prater et al. 2006). Shifts from fibrous to taprooted forb species can result in reduced water infiltration in some soils (Tisdall and Oades 1982).

The native sagebrush steppe and Great Basin sagebrush mosaics important for Greater Sage-Grouse often are characterized by a discontinuous spatial arrangement of perennial plants within the community. The interspaces among perennial plants may be partially filled with biological soil crusts (lichens, mosses, and cyanobacteria; Johansen 1993, Belnap and Lange 2001, Belnap et al. 2006, Muscha and Hild 2006), but percent cover can be highly variable across the SGCA. The cover of crusts in ungrazed Wyoming big sagebrush communities was <12% in Wyoming (Muscha and Hild 2006); crusts were more prominent on finer-textured soils and in 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). In addition, soil surface temperatures are moderated and soil water availability is greater beneath sagebrush canopies compared to the interspaces (Davies et al. 2007b).

The shift from a native shrub grassland community to a near monoculture of annual invasive grasses changes the temporal availability of water. Stands of cheatgrass reduced growth of native perennial plants, which was a function of significant reduction in water availability and reduced native plant water content (Melgoza et al. 1990, Booth et al. 2003). Paradoxically, conversion of sagebrush habitats to a cheatgrass monoculture can close the community to reestablishment of native plants (Robertson and Pearse 1945), but open the community to subsequent invasions by other exotics, such as medusahead, Russian thistle (*Salsola kali*), and rush skeletonweed (Kinter et al. 2007), and can lead to dominance by peren-

nial weed species (Young and Longland 1996, Shaw and Hild 2007).

There is considerable speculation regarding the effect of cheatgrass on nitrogen cycles and how nitrogen levels relate to native plant maintenance and establishment in communities. Evans et al. (2001) speculated that cheatgrass monocultures would lead to reduced nitrogen availability at a site, but others have found no evidence for this relationship (Bolton et al. 1990, Svejcar and Sheley 2001). Increased native perennial establishment with reduced nitrogen availability has occurred 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.

Maintaining resistance to cheatgrass invasion may be a function of containing nitrogen and other resources within the native plant community (Beckstead and Augspurger 2004, Humphrey and Schupp 2004). Cheatgrass and other invasive species are characterized as benefiting from disturbance, but a disturbance event is not required for cheatgrass germination and dominance to occur (Roundy et al. 2007). Thus, the primary effect of disturbance may be expressed primarily through reductions in native plant communities that release resources better exploited by invasive plants (Chambers et al. 2007) and allow invasion into more resistant plant communities. Cheatgrass then can dominate the site until resources become limiting and perennial species can reestablish dominance (Mata-González et al. 2008).

Wildfire

Characteristics of Fire and Fire Regimes

Characteristics of individual fire events as well as the collective fire regime are important drivers of structure, composition, and abundance of vegetation within sagebrush communities. At broader spatial scales, fire events and regime are dominant determinants of habitat configuration within the landscape. Individual fires are described by severity (the level of biological and physical effect of fire on all plant layers, soils, and animals), intensity (the amount of energy released during a fire), season, extent or size, and complexity (patchiness of burned and unburned areas within the fire boundary).

Severity in forestry terms is defined as the percent mortality on the overstory vegetation layer and does not consider the influence on understory vegetation layers, soils, and other fauna. Fire regime is a function of the mean and range of the interval (usually in years) between fire events for a defined area. The fire regime for a specific area is influenced by climate, regional location, fuel characteristics (biomass and structure), and recovery time following disturbance, topography, season and frequency of ignition, and vegetation composition.

Relative frequency of fire in sagebrush systems has been estimated using composite fire intervals (CFI) and natural fire rotation (NFR) (Heinselman 1973; Baker, this volume, chapter 11). Both estimation methods are critically dependent on the spatial and temporal period over which they are computed. Each provides a perspective on the role of fire in sagebrush systems that must be interpreted from the appropriate scale.

Composite fire intervals are derived by documenting burns occurring at individual sites that are typically several hectares or less in size. Composite fire intervals can reveal fine-scale variation in fire frequency in both time and space within a specific landscape. Heterogeneity in fire occurrence at fine scales can have important ecological consequences related to seed dissemination, succession, rate of recovery, diversity, landscape complexity, and habitat suitability. Each of these small areas is often within a single plant association or ecological site, allowing evaluation of fire occurrence within these ecological units. Historical fire frequencies can vary at relatively fine scales (tens to hundreds to thousands of hectares) across a landscape in at least some sagebrush systems, affecting the spatial distribution of vegetation (Heyerdahl et al. 2006, Miller and Heyerdahl 2008). Small fires or fires with moderate to high complexity can have important ecological consequences for vegetation. In addition, site-specific estimates can place local fire intervals within a historic range of variation specific to a location. Computer simulations indicate that CFI can be an accurate estimator of mean fire interval (Van Horne and Fulé 2006, Parsons et al. 2007).

The probability of fire occurrence estimated from NFR substitutes space for time by incorporating multiple burns within landscapes or regions. Natural fire rotation does not directly consider variation across space or time (Heinselman 1973, Agee 1993, Reed 2006) but instead estimates

the probability that, on average, a location will burn within a year (e.g., a fire rotation of 100 years means that fire will burn the entire landscape over a 100-year period and that each point in the landscape will burn, on average, once during that period; Baker 2006b). Large fires dominate the computation of NFR and are best computed for an area that exceeds the largest expected fire in one rotation. Broad estimates at regional scales present differences in dominant disturbances (e.g., fire is a dominant and frequent disturbance in the Snake River Plain compared to the Wyoming Basin) or relative changes within regions between periods used for estimating NFR (Knick and Rotenberry 2000; Baker, this volume, chapter 11).

Pre-Euro-American Dynamics

A clear picture of the complex spatial and temporal patterns of historic fire regimes in many sagebrush communities prior to Euro-American settlement is unlikely. The vast variation in fuel composition and structure, landscape heterogeneity, ignition from aboriginal and lightning sources, weather, and topography contained within the SGCA makes this challenge even more daunting. We can only estimate the potential of different vegetation phases, plant associations, and ecological sites, each on unique landscapes to burn based on proxy data, which include variables that drive fire regimes. Even then, pre-Euro-American settlement fire regimes reconstructed from cross-dated fire scars across the SGCA are few and spatially limited.

The magnitude of aboriginal burning and its impacts on western landscapes prior to the effects of Euro-American diseases and settlement is highly controversial; literature accounts range from limited (Baker 2002, Vale 2002) to significant (Keeley 2002, Kay 2007). Lightning, a primary ignition source, varied greatly in the Intermountain West and is influenced by regional location, topographic variability, and moisture availability (Knapp 1997). Ignitions from lightning increased moving from east to west toward the eastern slopes of the Sierra and Cascade Mountains, with increased variation in elevation (Knapp 1997) and increasing elevation (Gruell 1985). Early explorers observed fires in higher elevations but seldom reported fires in the sagebrush valleys at lower elevations (Gruell 1985).

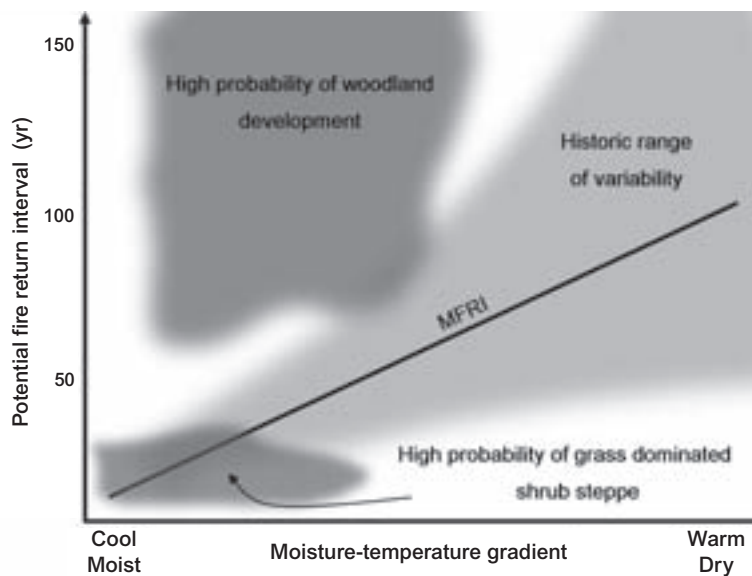
The temporal dynamics of regional fire occurrences and shifts in vegetation over long time

periods (5,000 years) in the SGCA have been based on abundance of charcoal and ash collected in pond and lake sediment cores in the Great Basin (Mehring 1985, Mehring and Wigand 1987, Wigand 1987). These data imply that fire occurrence in the surrounding area increased during relatively wet periods, which increased fuel abundance and resulted in distinct long-term patterns. Recent macroscopic charcoal work in central Nevada also correlated an increase in fire occurrences in Wyoming big sagebrush during periods of wetter climate (Mensing et al. 2006).

Most estimates of fire-return intervals in drier Wyoming big sagebrush communities were based on opinion and circumstantial evidence (e.g., estimated time for sagebrush to reestablish), rather than on experimental data. Fire-return intervals of up to a century were recorded in Wyoming big sagebrush communities located in valley bottoms in central Nevada and varied with climate and fuel accumulation (Mensing et al. 2006). More arid sites in the Wyoming big sagebrush alliances characterized by long fire-return intervals also may have had an extremely wide historic range of variation in years between fires, possibly resulting in a range of several burns in a single century to not burning at all in more than 100 years (Fig. 10.8). High variability in fire occurrence in both time and space illustrates the difficulty in describing fire regimes as well as the complexity of this disturbance as a mechanism in structuring sagebrush communities.

Fire generally was more common in mountain big sagebrush at higher elevation sites. Mean composite fire intervals in southwestern Montana were estimated to be 32 years in mountain big sagebrush communities (Lesica et al. 2007). Composite fire intervals for ecological sites supporting mountain big sagebrush and an Idaho fescue understory that were adjacent to large stands or surrounding small patches of ponderosa pine (*Pinus ponderosa*) or Douglas fir were as frequent as 10–35 years (estimated from 1- to 10-ha plots) (Miller and Rose 1999, Heyerdahl et al. 2006, Miller and Heyerdahl 2008). Fire-return intervals within small sample plots (1 to several hectares) systematically located across a 1,030-ha study area experienced fires prior to livestock grazing every 2–84 years (Heyerdahl et al. 2006). Communities with relatively short intervals (<20 years) likely would have been predominantly grasslands with scattered patches of shrubs (Fig. 10.8). These areas are typically in moist (35- to 40-cm precipitation zone) habitats on deep to moderately deep soils with strong, well-developed mollic horizons (Heyerdahl et al. 2006, Miller and Heyerdahl 2008). Fine-scale variation in CFI in northeastern California ranged from 10 to more than 100 years among plots located systematically within plant associations containing mountain big sagebrush in at least one successional state but differing in soil depth and texture, aspect, slope, and dominant diagnostic grass species (Miller and Heyerdahl 2008).

Figure 10.8. Conceptual model illustrating the historic potential mean fire return interval (MFRI) and historic range of variation (light gray area) in sagebrush steppe as it relates to temperature and moisture, resulting in a change in structure, composition, and abundance of fuels. Persistent vegetation that occupies the light gray area would likely be a sagebrush herbaceous mix, although herbaceous vegetation would occupy the site immediately following fire until the sagebrush stand redeveloped (derived by S. C. Bunting and R. F. Miller).



Post-Euro-American Settlement Changes at Low Elevations—Exotic Grasses

Wyoming big sagebrush communities in the Intermountain West prior to Euro-American settlement consisted of sagebrush and perennial grasses that were spatially discontinuous and generally did not carry fires except under extreme weather conditions (e.g., low humidity and high winds) or several wet years that allowed fine fuels to accumulate (Hull and Hull 1974, Vale 1975, Mensing et al. 2006). Invasion by exotic annuals into this sagebrush alliance has resulted in dramatic increases in number and frequency of fire occurrences (Young and Evans 1973, West 2000). Cheatgrass readily invades sagebrush communities, especially sites where native perennials have been depleted. Cheatgrass fills the interspaces between the shrubs and grasses, providing a continuous fuel source that facilitates fire spread. Fires kill sagebrush, which needs to regenerate from seed, while cheatgrass recovers within the first 2 years after fire (Young and Evans 1978). Observations of increased fire frequencies were reported in the early to mid-1900s after annual grasses had invaded much of the Intermountain West (Pickford 1932, Piemeisel 1951, Robertson 1954). Recent fire-return intervals in Wyoming big sagebrush were as low as 5 years in portions of the Snake River Plain where cheatgrass now dominates (Whisenant 1990), although how these estimates were derived was not documented.

Post-Euro-American Settlement Changes at High Elevations—Woodland Expansion

The initial increase in postsettlement conifer expansion beginning in the mid- to late 1800s influenced large portions of today's sagebrush habitats at higher elevations. Tree establishment that created lasting woodlands in sagebrush steppe communities was not synchronous but varied spatially and temporally within six different mountain ranges in Utah, Nevada, Idaho, and Oregon, and it continues to be an ongoing process today (Fig. 10.9) (Miller et al. 2008). Current stages of woodland development at sites that historically were sagebrush steppe communities range from initial phases of encroachment to tree-dominated stands that have been closed for >50 years. The rate of woodland expansion across these six mountain ranges has declined since the

1960s because few sagebrush steppe communities that do not contain juniper or pinyon pine remain within the woodland belt.

Woodlands can encroach into sagebrush communities when the interval between fires becomes long enough for seedlings to establish and trees to mature and dominate a site. Juniper and pinyon trees are killed by fire. However, the probability of western juniper being killed by fire decreases for trees >50 years of age on productive sites and >90 years on low productive sites (Burkhardt and Tisdale 1976, Bunting 1984, Miller and Rose 1999). Trees that have escaped fire can attain >100–200 years of age. Old trees in the Great Basin most frequently occupy sites characterized by shallow rocky soils supporting limited fuels that often are intermixed with ecological sites or plant associations on deeper soils.

The probability of woodlands displacing sagebrush communities increases where seed sources are nearby and fire intervals in more productive sagebrush steppe sites increase to >50 years (Fig. 10.10). Pinyon or juniper trees can reestablish within the first decade following fire if seed is available. Most seeds are dispersed by birds; the majority of seeds are deposited <100 m from the seed tree (Schupp 1993; Chavez-Ramirez and Slack 1994; Schupp et al. 1997; Chambers et al. 1999a,b).

The initial increase in rates of establishment and expansion of conifers into sagebrush steppe communities in the 1800s did not have an immediate effect until density and size allowed trees to dominate the vegetation. Stands that established with sufficient densities in the 1860s and 1870s in southeastern Idaho began to close in the 1950s (Johnson and Miller 2006). The majority of these trees would not have reached maturity under the historic fire regime.

An immediate widespread decline in fires coincided with the introduction of large numbers of livestock in the late 1800s (Miller and Rose 1999; Heyerdahl et al. 2006; Swetnam et al. 2001). Fire declines in the 17th–19th centuries in some areas in the Southwest coincided with the early introduction of sheep, goats, and cattle by Navajos and Hispanic settlers (Savage and Swetnam 1990, Touchan et al. 1995, Baisan and Swetnam 1997). The majority of these sudden declines in fire occurred prior to fire suppression efforts that began in the early 1900s (Miller and Rose 1999; Swetnam et al. 2001). Livestock grazing can

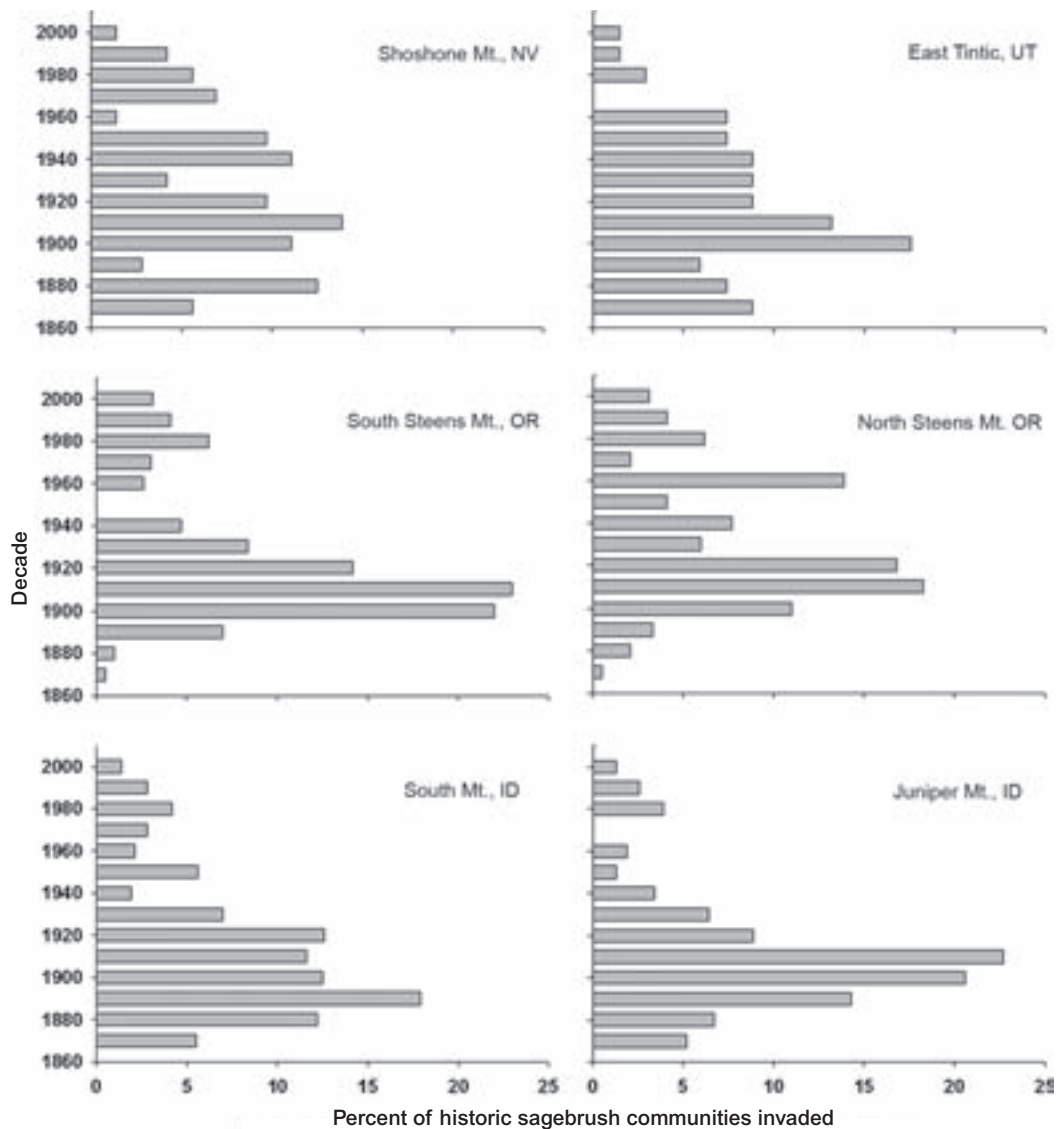


Figure 10.9. The proportion of decadal encroachment across six mountain ranges of pinyon and/or juniper between 1860 and 2000 into historic sagebrush steppe stands with no evidence of pre-settlement trees. Data were collected from transects established along an elevation gradient (14–24 km in length), which extended from the lower to upper boundaries of each woodland (adapted from Miller et al. 2008).

influence fire by reducing the current year's fuel loads and changing the structure, composition, and abundance of vegetation. A reduction in fine fuels through livestock's heavy use of herbaceous plants during the late 1800s then could have reduced fire occurrence across western landscapes and helped drive postsettlement increase and expansion of pinyon and juniper woodlands. The establishment of tree seedlings in mountain big sagebrush plant associations can occur within a decade following fire where a seed source is

available (Miller and Rose 1999; Johnson and Miller 2006, 2008; Miller and Heyerdahl 2008). However, the reduction of fire in the late 1800s allowed these trees to reach maturity in historic sagebrush steppe communities, forming woodlands with different fuel characteristics and resulting in a different fire regime.

Evidence for a direct relationship between livestock grazing and woodland encroachment through a reduction of competition from native grasses, forbs, and shrubs is difficult to document.

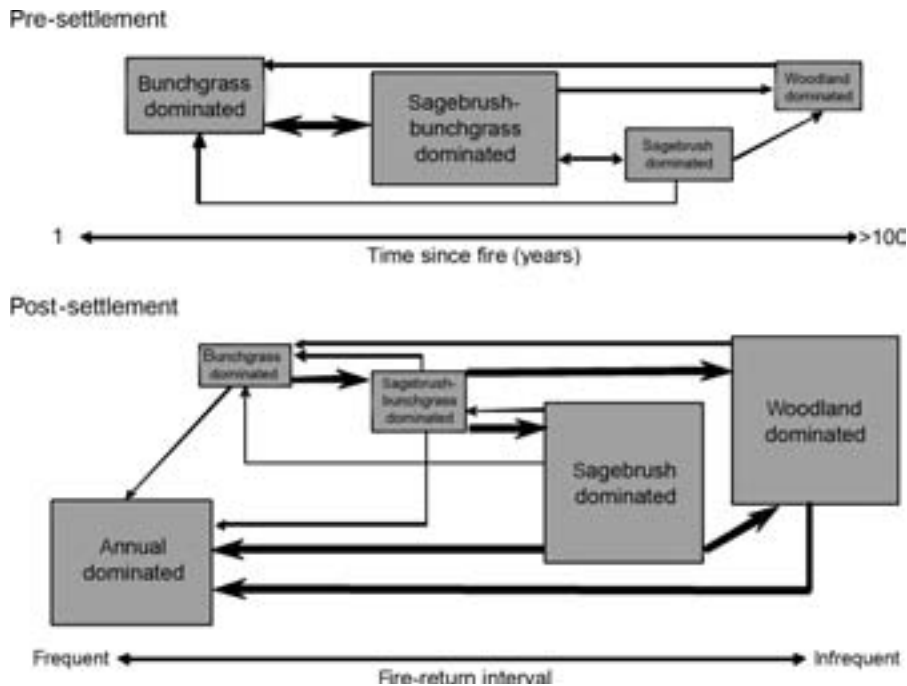


Figure 10.10. Conceptual model of pre- and post-settlement dynamics for plant communities in the Sage-Grouse Conservation Area. Differences in box and arrow sizes imply a difference in proportion of phases and transition from one phase to another within and across steady states.

Both the extent (size) and complexity of fires prior to the introduction of livestock are unknown. Conifer encroachment patterns in sagebrush steppe have not been related to fence line contrasts, distance to water, or ecological conditions that are often observed with other vegetation layers. Western juniper also has increased in the absence of grazing on a relict area (Soulé and Knapp 1999, 2000). Elevated carbon dioxide levels also may have accelerated woodland expansion (Knapp and Soulé 1999, Knapp et al. 2001) but were unlikely to be a factor during the late 1800s and early 1900s.

Change in Fire Regimes

We developed a database of fire statistics from records assembled across the SGCA. Although the fire records include forested areas, we attempted to eliminate those portions of fire polygons that did not burn sagebrush habitats by masking forested areas identified in the vegetation coverage (LANDFIRE 2006). We plotted frequency of fires for all years since 1900 for which fires were documented (Figs. 10.11–10.15); records of fires in some regions were present

from 1870. We recognize, however, that analyses of these data are confounded by: (1) an increase in reporting effort by the agencies, (2) differential reporting across regions, (3) lack of recordkeeping in some districts until the 1980s, (4) lack of records of geographic information system–based polygons of fire data until the 1990s, and (5) the fact that the most recent source for fire polygons, the United States Geological Survey Geospatial Multi-Agency Coordination Group, began maintaining fire records in 2004. Therefore, we mapped previous fires only from 1960 through 2007 (Fig. 10.16) for descriptive purposes and conducted statistical analysis on fire size, number of fires, total area burned, and within-year variation in fire sizes recorded from 1980 through 2007 (Table 10.5).

Number of fires and total area burned across the SGCA increased in each of the geographic subdivisions except the Snake River Plain from 1980 through 2007 (Table 10.5). Average fire size increased during this period only in the Southern Great Basin. Within-year variation in fire size decreased in all geographic regions except the Snake River Plain (Table 10.5). The decrease in variation within years is probably because of

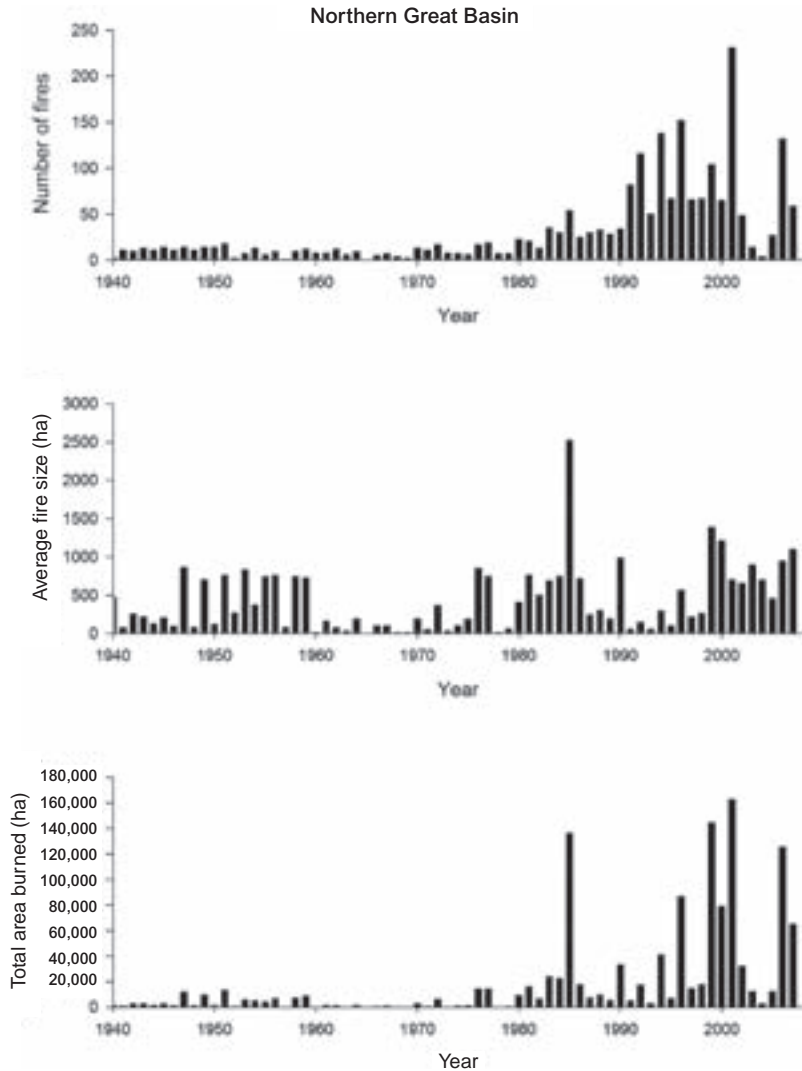


Figure 10.11. Number of fires, average fire size (ha), and total area burned (ha) within the Northern Great Basin (Fig. 10.3). Regression models of changes in fire statistics from 1980 through 2007 are presented in Table 10.5.

greater suppression capabilities. The increased number of fires resulted in a significant increase in total area burned since 1980.

Location of fires mapped since 1960 was related to the distribution of cheatgrass, particularly within the Snake River Plain and Northern Great Basin (Figs. 10.6, 10.16). Cheatgrass was established throughout the area by the 1920s and 1930s (Klemmedson and Smith 1964, Mack 1981, Billings 1990). Consequently, much of the Snake River Plain occupied by cheatgrass in southern Idaho has been well-defined by fires. Fires in northern Nevada and eastern Oregon,

also within the cheatgrass region, were more pronounced since 1980. Fires in the eastern section of the SGCA have been recorded only in more recent years.

The total area burned each year on or adjacent to lands managed by the Bureau of Land Management (BLM) was highly variable from 1997 through 2005, illustrating the difficulty in planning for an average year. Area burned per year varied almost sixfold, from 1,455 km² (sum of force and contract accounts, BLM and non-BLM lands) in 1998 to 8,142 km² in 1999, and tenfold to 14,365 km² in 2006 (Tables 10.6, 10.7). The

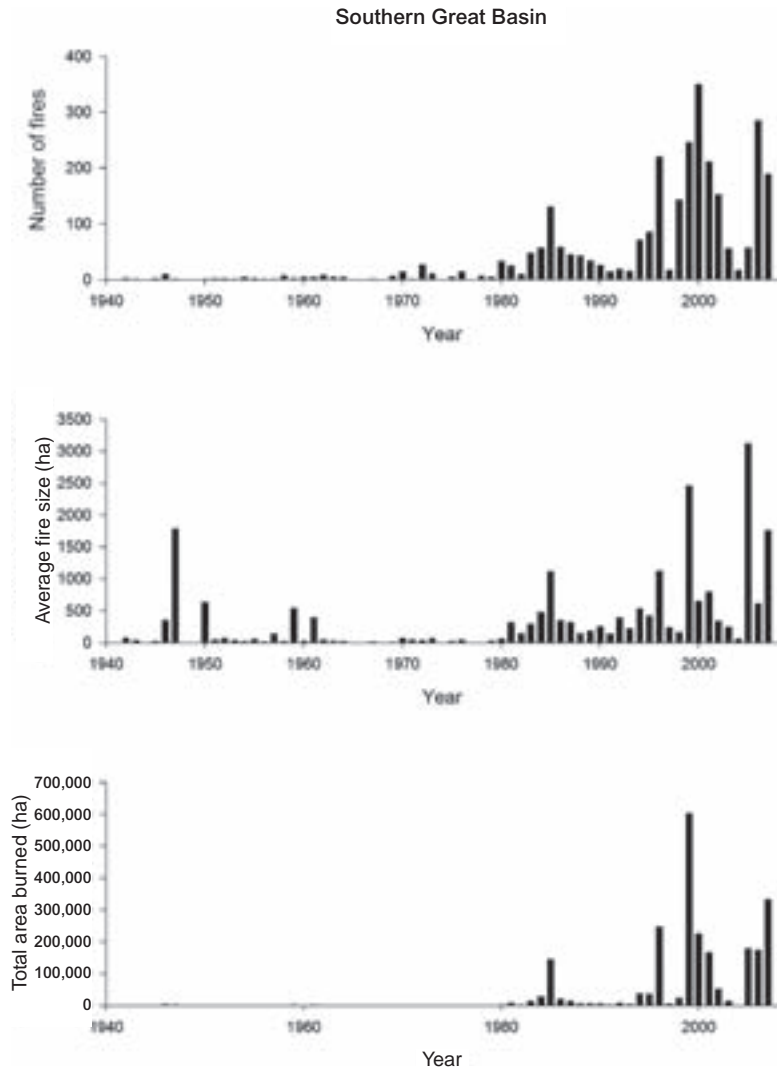


Figure 10.12. Number of fires, average fire size (ha), and total area burned (ha) within the Southern Great Basin (Fig. 10.3). Regression models of changes in fire statistics from 1980 through 2007 are presented in Table 10.5.

most total area burned was consistently found in cheatgrass regions throughout Idaho, Nevada, and Oregon.

Human-caused fires within the SGCA were related to the network of roads (Fig. 10.17). Fire ignitions are an additional consequence of roads and access by humans, in addition to road influences on habitat fragmentation and spread of exotic plant species (Trombulak and Frissell 2000). Of 3,465 fires ignited during 2006 on or adjacent to lands managed by the BLM in Colorado, Idaho, Montana, Oregon, Utah, Washington, and Wyoming, 822 (24%) were caused by

humans (United States Department of Interior 2006a:Public Land Statistics Table 6-1).

Global Climate Change

Seasonal and Annual Patterns

Areas dominated by sagebrush habitats are characterized as well-vegetated semideserts in semiarid climates (West and Young 2000, Miller and Eddleman 2001). Timing and abundance of water availability, which varies seasonally and annually, are the major factors that affect the structure, composition,

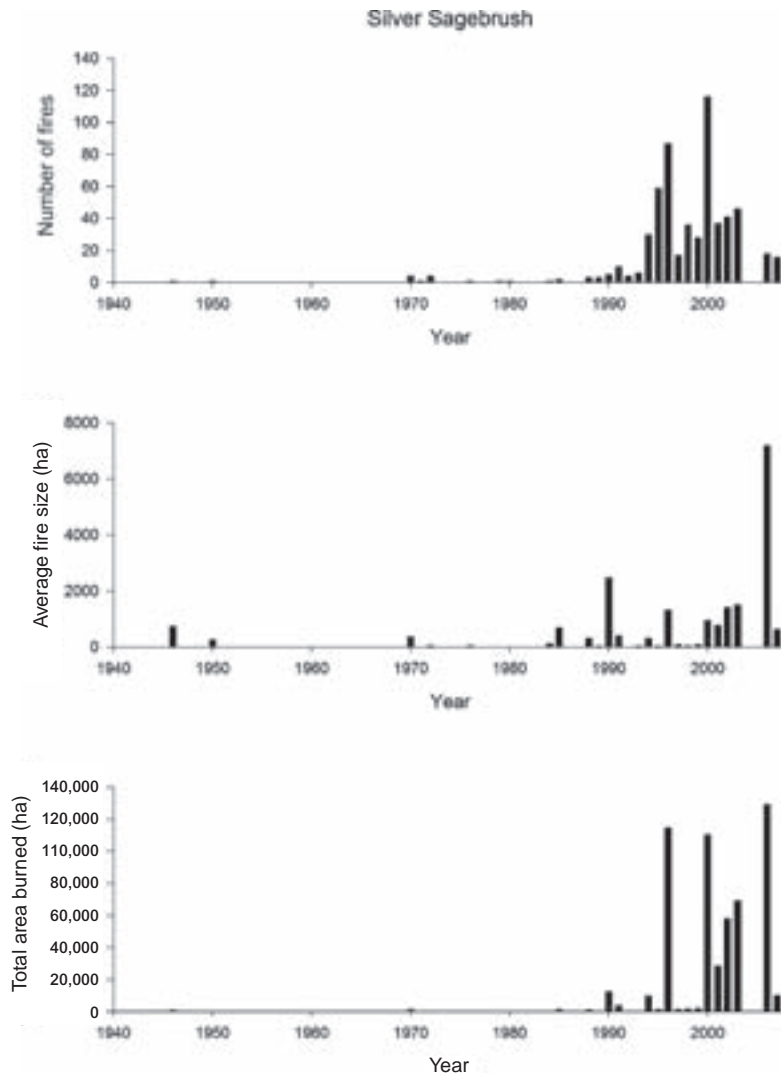


Figure 10.13. Number of fires, average fire size (ha), and total area burned (ha) within the Silver Sagebrush (Fig. 10.3). Regression models of changes in fire statistics from 1980 through 2007 are presented in Table 10.5.

and abundance of vegetation (Toft et al. 1989, West 1996, Anderson and Inouye 2001). Sagebrush systems maximize net annual production in late spring and early summer, when available soil moisture and warm temperatures coincide (West 1983b). Most of the water available to plants in the surface soil layers is depleted by midsummer (Anderson et al. 1987, Jensen 1989b, Obrist et al. 2004). Plants in sagebrush communities have evolved mechanisms, such as deep rooting systems, shedding of leaves (Miller and Shultz 1987), or becoming dormant, to survive periods of high water stress.

Most moisture available to sagebrush communities occurs seasonally as precipitation during

winter and spring. The short-term, high-intensity rainfall typical of isolated convective storms during summer generally is of minor importance, because little moisture infiltrates the fine-textured soils characteristic of sagebrush habitats and annual occurrence of such rainfall in any one area is undependable (West and Young 2000). Almost all water recharge in the soils was through roots by hydraulic redistribution (Ryel et al. 2003). Loss of deep-rooted sagebrush plants and conversion to systems dominated by annual grasses thus reduce the potential for deep soil water recharge.

The amount of summer precipitation increases across the SGCA from west to east and north to

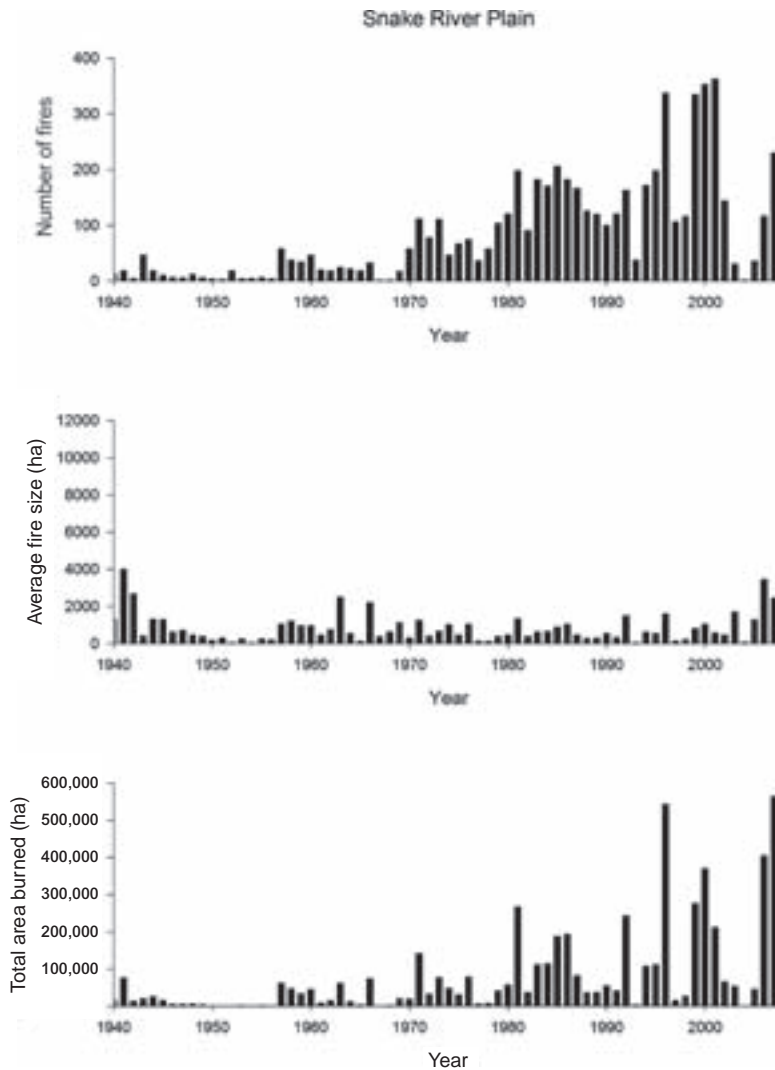


Figure 10.14. Number of fires, average fire size (ha), and total area burned (ha) within the Snake River Plain (Fig. 10.3). Regression models of changes in fire statistics from 1980 through 2007 are presented in Table 10.5.

south due to mixing of air from the Pacific Ocean with summer monsoons from the Gulf of Mexico (Houghton 1969, West 1983a). Elevation, aspect, and soil depth, structure, and texture are also important factors influencing the amount of available moisture (Jensen 1989a,b) and the potential sagebrush community it can support (Jensen et al. 1990).

Interannual variation of precipitation varies greatly across the SGCA, which is subject to periodic drought (Patterson 1952, Thurow and Taylor 1999, Seager et al. 2005). An operational definition of drought is based on the departure from an average amount of precipitation or other climate

variables that have been derived from a historical (usually 30-year) average (United States Department of Commerce 2004). Thus, drought defined relative to an average set of conditions has occurred periodically but not regularly in sagebrush habitats. Drought affected sagebrush landscapes during the periods approximated by the late 1890s to 1905, mid-1920s to 1940, early 1950s to mid-1960s, mid-1970s, mid-1980s to mid-1990s, and 1999–2004 (Fig. 10.18). Local scale patterns of drought also have occurred with varying length and severity; water-year precipitation was above average in only 4 years between 1933 and 1956 in the upper Snake River Plain (Anderson and Inouye

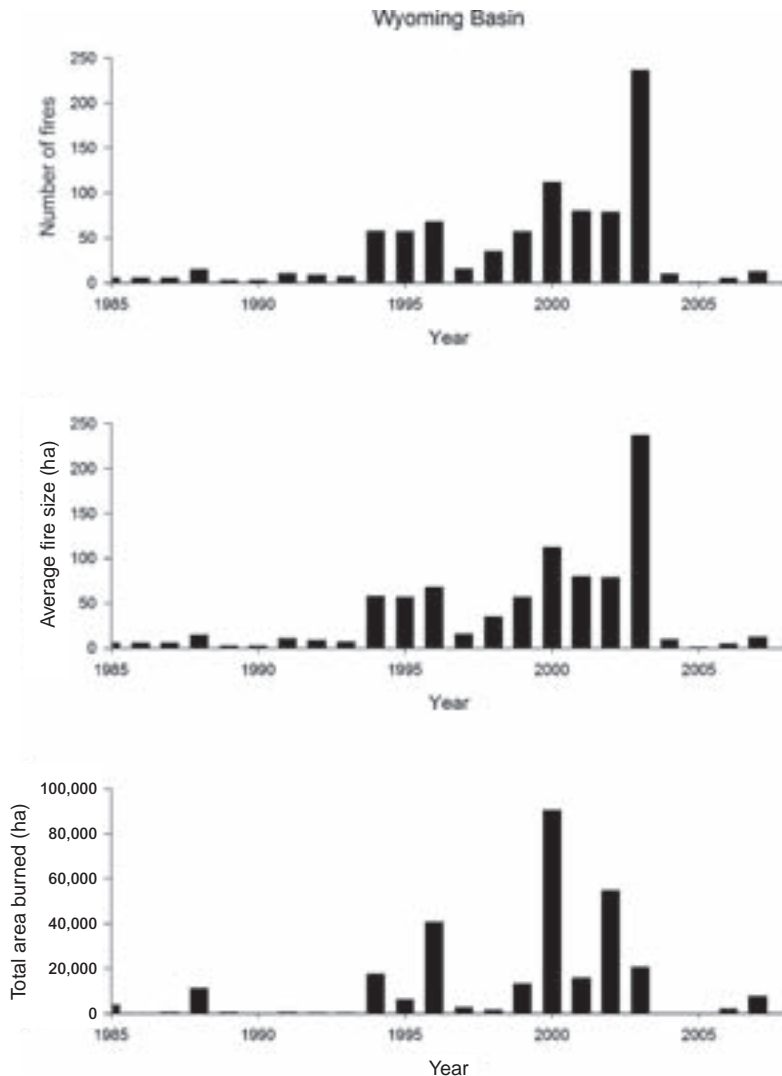


Figure 10.15. Number of fires, average fire size (ha), and total area burned (ha) within the Wyoming Basin (Fig. 10.3). Regression models of changes in fire statistics from 1980 through 2007 are presented in Table 10.5.

2001). In contrast to much of the western United States, drought duration and severity have increased in interior regions and across much of the SGCA during the 20th century (Andreadis and Lettenmaier 2006; Seager et al. 2005, 2007).

Soil erosion is considered the greatest threat to long-term sustainability of shrublands (Society for Range Management 1995). Semiarid shrublands are subject to soil erosion during drought because precipitation is insufficient to maintain vegetative cover (Morrison 1964, Thurow and Taylor 1999). Reduced vegetation cover and increased soil erosion result in long-term changes characterized by

reduced soil depth, decreased water infiltration, and reduced water storage capacity (Milton et al. 1994, Thurow and Taylor 1999) possibly resulting in a shift of vegetation to a new steady state.

Climate Change

Climate change is a complex process in which interactions among natural and anthropogenic sources affect long-term trends in temperature, precipitation, and atmospheric characteristics (Notaro et al. 2006). The Intergovernmental Panel on Climate Change defined climate change as “a change in the

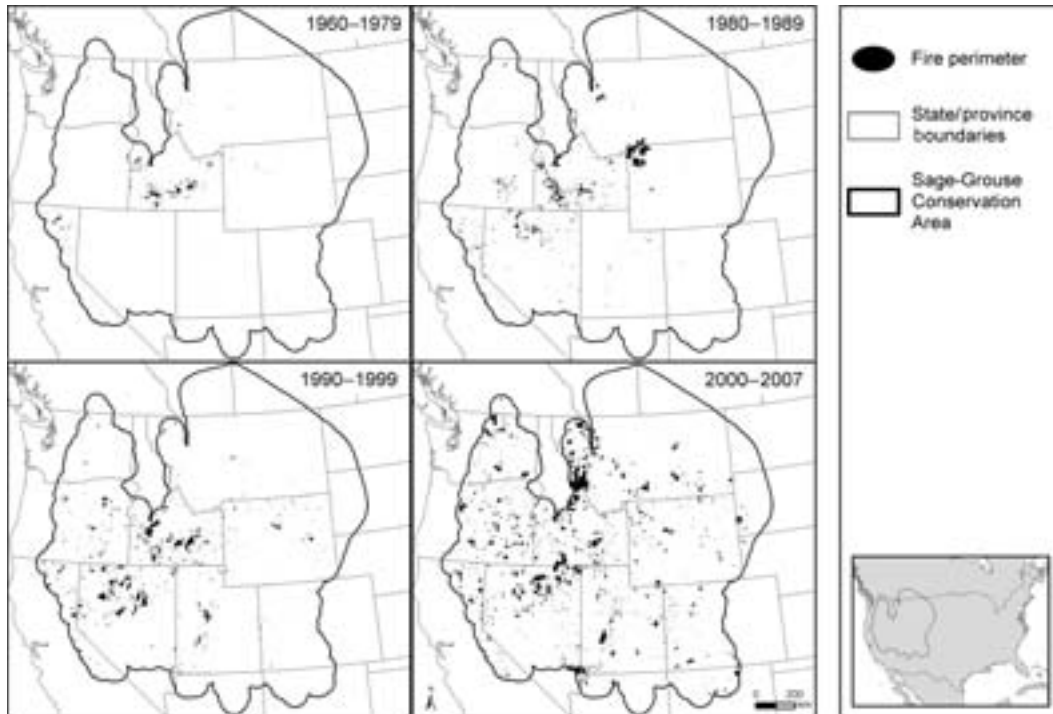


Figure 10.16. Fires mapped in the western United States from 1960 to 2007. Fire information was obtained from >500 source data sets obtained from the U.S. Bureau of Land Management, U.S. Forest Service, U.S. National Park Service, and other state and federal agencies.

state of the climate that can be identified, using statistical tests, by changes in the mean and/or the variability of its properties, and that persists for an extended period, typically decades or longer” (Intergovernmental Panel on Climate Change 2007:30). The definition includes any change and does not distinguish between human-induced or natural causes. Analysis of historical patterns presents dominant trends that may (or may not) continue into the future because of feedbacks among drivers and changes in their dynamics at different equilibria (Intergovernmental Panel on Climate Change 2007).

Global climate-change models predict that more variable and severe weather events (drought, storms), 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, Christensen et al. 2007, Intergovernmental Panel on Climate Change 2007). Potential changes for locations or vegetation communities across arid and semiarid landscapes in specific regions are difficult to predict, because most climate change models project scenarios over extremely broad geographic or continental regions (Reynolds

et al. 1997). Average temperatures may warm by as much as 2.8°C to >6°C for the Great Basin and much of the adjacent areas that support sagebrush (Bachelet et al. 2001, Neilson et al. 2005, Christensen et al. 2007). Atmospheric carbon dioxide, methane, nitrous oxide, and halocarbons including fluorine, chlorine, and bromine also are predicted to increase under most global climate change scenarios (Christensen et al. 2007, Intergovernmental Panel on Climate Change 2007).

Long-term changes in global climate and atmospheric conditions, particularly increased temperature and levels of carbon dioxide, will shift competitive advantage among individual plant species. Increased temperatures could exert additional stresses on sagebrush if increased transpiration rates during winter months coupled with changes in precipitation patterns result in increased soil moisture loss throughout the year (Shafer et al. 2001). In addition, the trend for decreased snowpack, earlier onset and warmer spring periods, and reduced summer water flows in the western United States further extends the period of summer stress (Cayan et al. 2001, Intergovernmental Panel on Climate Change 2007). A substantial increase in

TABLE 10.5

Changes in number and average size of fires, total area burned, and variation in fire size by year within floristic divisions from 1980–2007.

We used a time-series linear model regressing year by fire variables with dependent variables log-transformed.

We conducted our analysis only for fires recorded after 1980 because of more consistent reporting, even though fire statistics were available prior to 1980 (Figs. 10.11–10.15). Data were incomplete for 2004 and 2005, and were not included in the analysis.

Floristic Provinces (no. of fire years 1980–2007)	Dependent variable	F	P	r ²	Coefficient ^a
Northern Great Basin (25)	Fires (N)	8.95	<0.01	0.25	0.05
	Mean size (ha)	0.29	0.59	0.01	0.01
	Total area (ha)	5.31	0.03	0.19	0.07
	CV ^b	8.28	<0.01	0.26	–0.01
Southern Great Basin (25)	Fires (N)	13.71	<0.01	0.37	0.08
	Mean size (ha)	4.57	0.04	0.17	0.04
	Total area (ha)	10.71	<0.01	0.29	0.13
	CV	9.20	<0.01	0.25	–0.01
Silver Sagebrush (19)	Fires (N)	14.87	<0.01	0.44	0.14
	Mean size (ha)	2.77	0.11	0.09	0.11
	Total area (ha)	10.70	<0.01	0.35	0.24
	CV	5.29	0.03	0.19	–0.04
Snake River Plain (25)	Fires (N)	0.02	0.90	0.00	0.00
	Mean size (ha)	1.64	0.21	0.03	0.03
	Total area (ha)	0.92	0.35	0.00	0.03
	CV	1.91	0.18	0.04	0.00
Wyoming Basin (21)	Fires (N)	8.31	0.01	0.27	0.11
	Mean size (ha)	2.59	0.12	0.07	0.06
	Total area (ha)	8.66	0.01	0.28	0.17
	CV	5.79	0.03	0.19	–0.02

^a Log-transformed coefficient.

^b Coefficient of variation (CV) used to standardize estimates of standard deviations across years.

temperature also confers a competitive advantage to frost-sensitive woodland vegetation that currently dominates the Chihuahuan and Sonoran Deserts but is temperature limited in its northern distribution. These woodlands may expand substantially northward into and displace large areas of sagebrush and other shrublands in the western portions of the sagebrush ecosystem, based on variety of projected conditions of climate change (Neilson et al. 2005). However, a large portion of these woody species also may be replaced by exotic grasses and forbs.

Atmospheric carbon dioxide has increased from preindustrial levels of 280 $\mu\text{mol mol}^{-1}$ to current levels of 360 $\mu\text{mol mol}^{-1}$ and are projected to be 420 $\mu\text{mol mol}^{-1}$ in 2020 and may exceed 600 $\mu\text{mol mol}^{-1}$ by 2100 (Bazzaz et al. 1996, Ziska et al. 2005). Over the last century, atmospheric carbon dioxide has increased by >20% (Polley 1997). Ecosystem responses to enhanced carbon dioxide involve interactions of biogeochemical cycles, water and energy fluxes, and vegetation dynamics dependent on the temporal scale over which

TABLE 10.6
Fire area (km²) on or adjacent to lands managed by the Bureau of Land Management from 1998–2006.
 Fire protection given public land administered by the BLM using protection forces and
 facilities supervised and operated by the BLM.

State	2006		2005		2004		2003		2002		2001		2000		1999		1998	
	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM
Arizona	185	36	1,158	132	29	12	20	0	33	0	8	2	4	0	92	6	8	0
California	38	43	95	95	25	97	45	8	88	97	30	7	34	11	81	96	17	30
Colorado	31	25	12	11	15	20	28	1	26	24	1	0	1	4	34	7	9	1
Idaho	1,844	433	1,170	346	21	19	104	69	127	85	325	178	1,146	460	1,265	127	87	70
Montana	221	720	10	22	7	27	281	312	3	6	0	0	0	24	3	0	4	0
Nevada	3,597	806	5,682	290	86	4	29	6	84	30	1,456	168	1,951	229	3,995	1,254	160	69
New Mexico	25	55	2	6	1	0	4	0	82	82	10	5	250	33	16	20	1	0
Oregon	1,242	375	61	10	4	2	24	0	415	23	600	50	600	143	46	6	130	107
Utah	728	281	526	291	65	20	233	96	16	16	57	62	315	99	249	119	136	79
Washington	10	29	2	30	0	0	38	16	4	16	0	0	3	25	4	12	0	0
Wyoming	95	520	9	18	4	5	20	6	1	1	14	33	101	200	3	3	2	1
Total km ²	8,017	3,324	8,726	1,249	256	206	824	513	878	380	2,502	504	4,405	1,228	5,787	1,650	554	356

SOURCE: United States Department of the Interior 1998, 1999, 2000d, 2001b, 2002b, 2003f, 2004e, 2005d, 2006c.

TABLE 10.7
Fire area (km²) on or adjacent to lands managed by the BLM from 1998–2006.

Fire protection given public land administered by the BLM using contracted protection forces and facilities.

State	2006		2005		2004		2003		2002		2001		2000		1999		1998	
	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM	BLM	Non-BLM
Arizona	0	0	0	313	0	0	0	0	0	0	0	0	0	0	0	2	0	1
California	179	651	71	20	206	1,603	21	122	68	106	4	50	99	181	37	73	29	71
Colorado	3	9	0	0	0	3	0	1	0	0	0	0	0	0	0	2	0	0
Idaho	1	17	0	5	0	7	4	24	9	7	7	13	61	233	20	45	2	227
Montana	35	1,099	19	64	0	8	0	36	4	21	0	0	2	2	2	2	4	2
Nevada	277	478	194	89	0	2	8	9	0	10	1	190	24	84	146	202	7	56
New Mexico	9	26	0	2	0	0	0	1	0	1	1	2	7	28	0	2	4	7
Oregon	2	76	13	9	8	13	1	1	1	1	0	1	3	8	0	0	29	100
Utah	2	60	1	26	0	0	1	0	5	15	0	0	43	26	47	124	5	1
Washington	45	6	1	10	7	80	3	6	8	3	3	5	0	0	0	0	0	0
Wyoming	9	42	1	4	1	1	10	1	1	2	1	22	1	48	0	0	0	0
Total km ²	565	2,463	299	540	222	1,718	49	201	95	165	16	283	239	609	253	453	80	465

SOURCE: United States Department of the Interior 1998, 1999, 2000d, 2001b, 2002b, 2003f, 2004e, 2005d, 2006c.

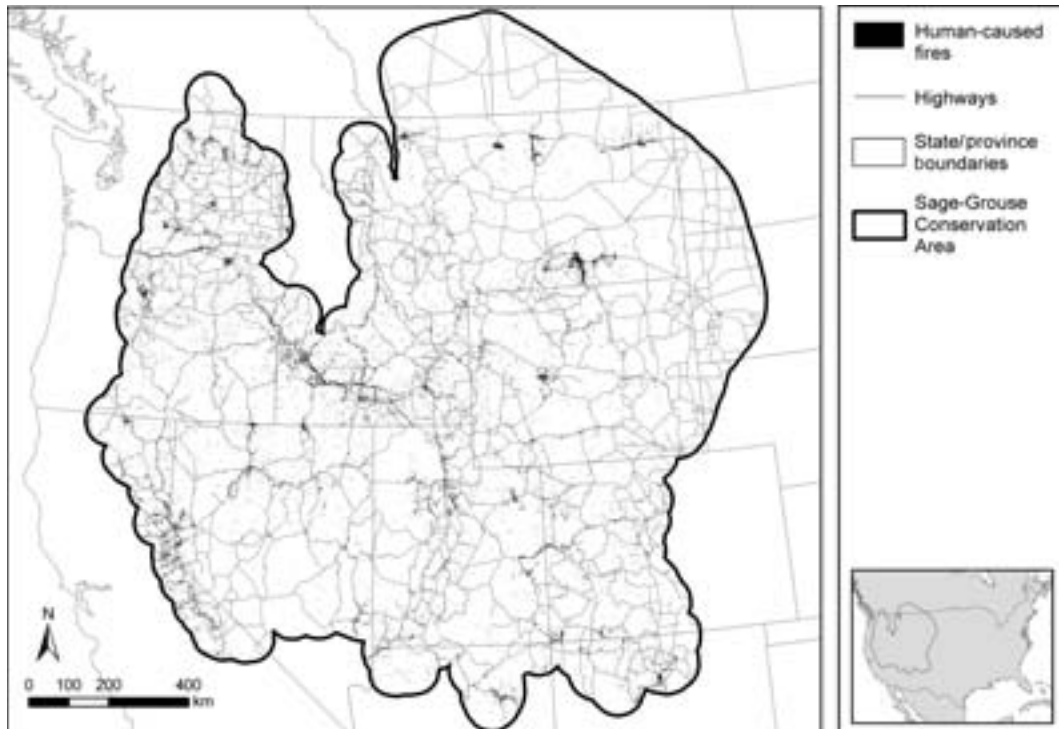


Figure 10.17. Distribution of human-caused fires relative to roads from 1960 to 2007. Fire information was obtained from the National Fire Occurrence database.

carbon dioxide increases (Körner 1996, Walker and Steffen 1996). The trend of increasing atmospheric carbon dioxide is likely to continue, and natural feedbacks or anthropogenic changes on the landscape that influence ecosystem processes may either ameliorate or intensify those effects (Noble 1996).

Increased levels of carbon dioxide favor exotic annual grasses at the expense of native vegetation in arid shrubland ecosystems (Mooney and Hobbs 2000, Smith et al. 2000). Cheatgrass responded positively to elevated carbon dioxide when compared to native grasses (Smith et al. 1987). In controlled laboratory tests, reproductive biomass of cheatgrass doubled and time to maturation decreased by 10 days at the current atmospheric carbon dioxide levels compared to lower levels recorded in preindustrial time (Ziska et al. 2005). Cheatgrass already competes successfully against native grasses because of earlier maturation, shallow root systems to collect water in soils, greater seed production, and the ability to respond quickly to resources released during disturbance (Klemmedson and Smith 1964). Thus, the ability of cheatgrass to compete in sagebrush ecosystems created by enhanced

carbon dioxide or changes in annual precipitation, temperature, or severe storms will facilitate its spread and exacerbate the cycle of fire and cheatgrass dominance (d'Antonio and Vitousek 1992, d'Antonio 2000, Ziska et al. 2005).

Each 1°C increase in temperature was predicted to result in a loss of 87,000 km² of existing sagebrush habitat, primarily to increasing distribution of other woody vegetation (Neilson et al. 2005). Only 20% of the current sagebrush distribution would remain under the most extreme scenario of an increase of 6.6°C (Neilson et al. 2005). Moreover, these scenarios have largely overlooked the potential response of exotic plant species and changes in fire cycles to further reduce sagebrush from its current distribution. These models predict that future remaining sagebrush habitats will primarily be in the more northerly latitudes and higher elevations (Fig. 10.19) (Shafer et al. 2001, Neilson et al. 2005).

CONSERVATION IMPLICATIONS

Altered disturbance regimes in many regions of the SGCA are shifting the potential natural community outside of the range of historic variation

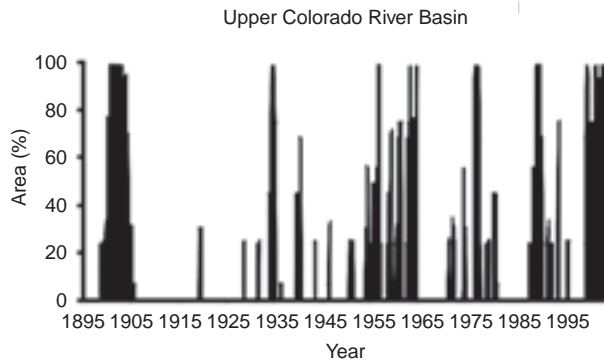
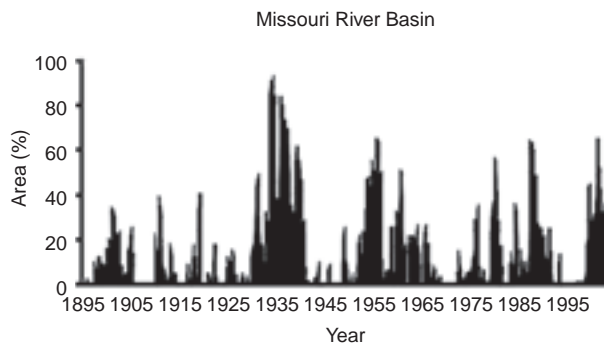
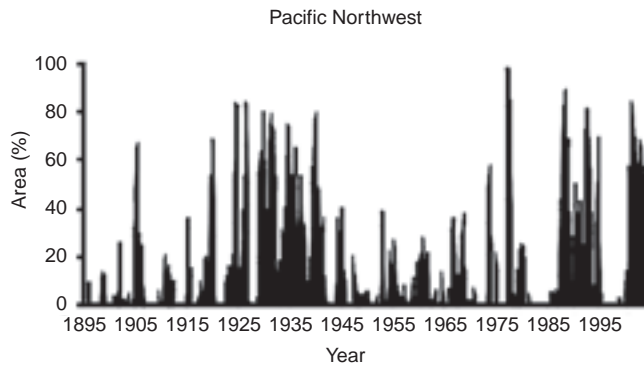
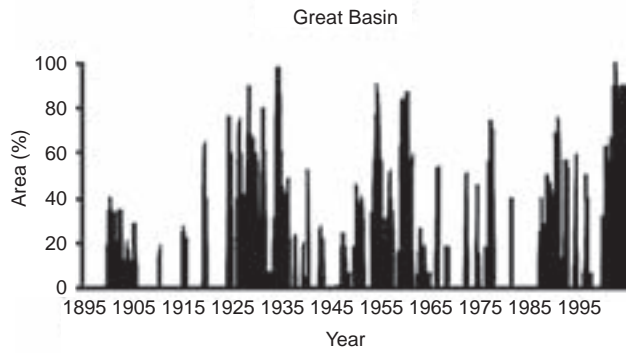


Figure 10.18. Percent of major river basins experiencing drought conditions from 1895 to 2004 (United States Department of Commerce 2004). The graphs represent the Palmer (1965) Drought Severity Index, which measures the extent of departure from the long-term average based on precipitation, temperature, and available water capacity.

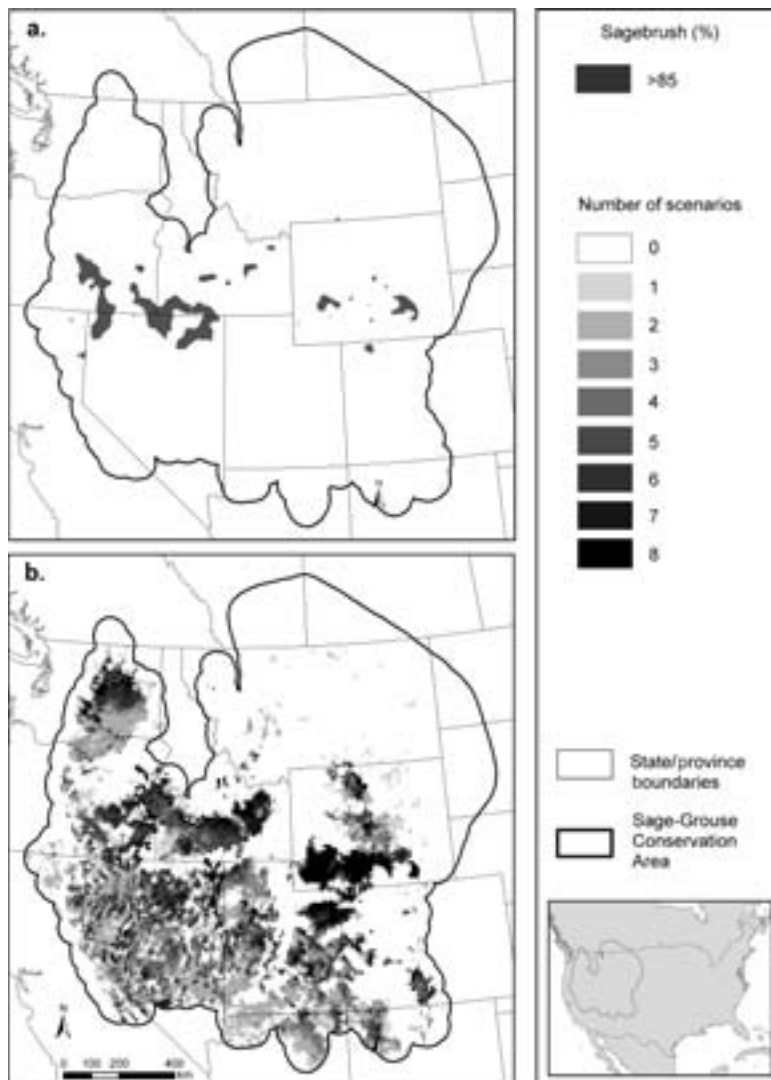


Figure 10.19. (a) Core areas of the current sagebrush distribution were displayed only in those regions in which >85% of the landscape (Fig. 10.1) was dominated by sagebrush. (b) Distribution of sagebrush predicted under current climate and seven models of future scenarios (Neilson et al. 2005). Each cell in the map is the sum of alternate models for future climate scenarios predicting that sagebrush will remain in that location.

to new steady states. Changes in plant composition and structure of sagebrush habitats as a result of disturbance events or successional trends may occur over a decade, over multiple decades, or within a single unique combination of weather and disturbance events over a few weeks. Resistance and resilience to change resulting from disturbance generally increase with increasing moisture and decreasing temperatures, and vary widely across different sagebrush plant associations and ecological sites. Sagebrush communities that

have persisted may change if disturbances become chronic or severe, replaced by exotic plants that are better adapted to more frequent disturbance or changing climate, and/or are better able to exploit new resources.

Cheatgrass currently is present throughout much of the western United States (Wisdom et al. 2005b, Meinke et al. 2009). Once established in a sagebrush community, the effects cascade in synergistic feedbacks toward increasing cheatgrass dominance resulting from increased fire disturbance, loss of

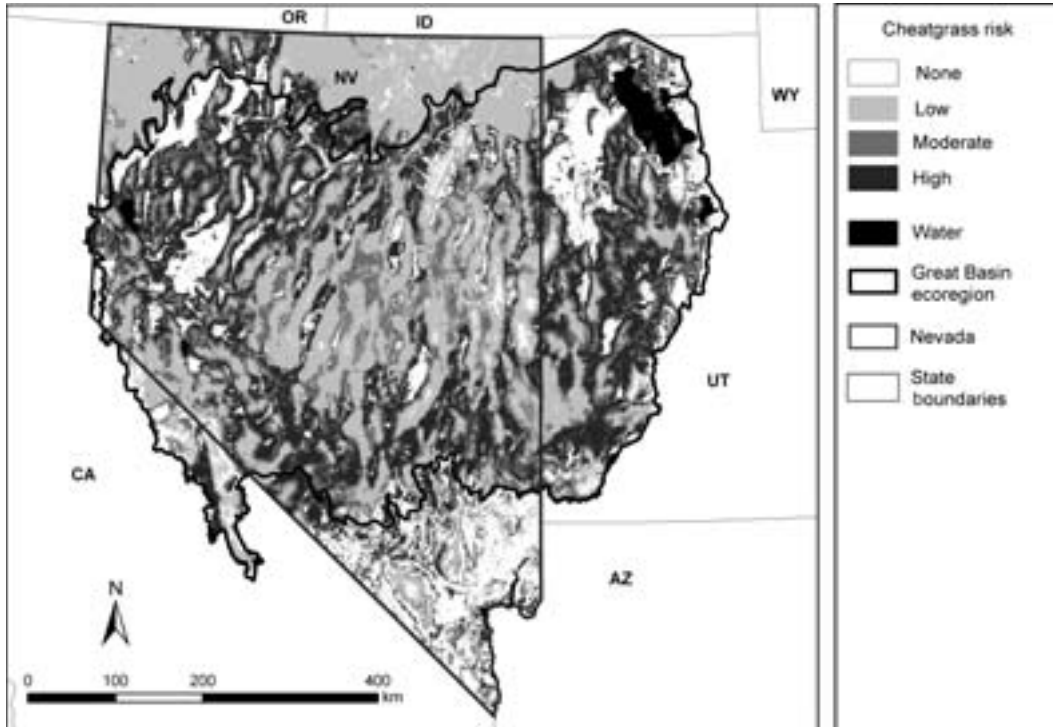


Figure 10.20. Estimated risk of cheatgrass displacement of sagebrush and other susceptible land cover types in the Great Basin ecoregion during the next 30 years (from Suring et al. 2005b, with permission from Alliance Communications).

perennial species and their seed banks, and decreased stability and resilience to changes in interannual weather and long-term climate patterns (d'Antonio and Vitousek 1992, d'Antonio 2000, Brooks et al. 2004a, Chambers et al. 2007).

The extensive distribution of cheatgrass, combined with its aggressiveness in replacing sagebrush, poses substantial risk of increased fire and loss of existing sagebrush communities. The risk of fire was <60% when ground cover of cheatgrass was <20%; areas with >45% ground cover of cheatgrass had a 100% risk of fire (Link et al. 2006). Of the 83,000 km² of existing sagebrush cover types in the Great Basin ecoregion of Nevada, Utah, and California, nearly 58% (48,000 km²) is estimated to be at moderate or high risk of elimination from continued invasion of cheatgrass during the next 30 years (Fig. 10.20) (Suring et al. 2005b). Similar estimates of risk posed by cheatgrass are projected for a vast area of 200,000 km² of existing sagebrush that encompasses southeast Oregon, southern Idaho, northeast California, Nevada, and western Utah (Suring et al. 2005b). Approximately 56% (113,000 km²) of existing sagebrush among these provinces was estimated at moderate or high risk

of loss from cheatgrass during the next 30 years (M. J. Wisdom, unpubl. data). Sagebrush sites considered to be at higher risk were those at lower elevations and on south- and west-facing slopes; these conditions generally represent the warmer, drier sagebrush sites on which cheatgrass has most successfully displaced big sagebrush communities in the past (Suring et al. 2005b). Sites considered at lower risk were those at higher elevations and on north- and east-facing slopes, which represent colder, wetter sagebrush communities. Cheatgrass is a self-pollinator and evolves slowly, but the species is comprised of many different populations that have different adaptive strategies (Kinter and Mack 2004). This results in uncertainty of its ability to invade plant associations currently thought to be resistant to encroachment. Elevated temperatures due to climate change may also increase the competitive ability of cheatgrass at higher elevations, thus expanding the range of cheatgrass into regions where it currently is limited.

The past and present role of wildfire in the SGCA is temporally and spatially variable. The increase in fine fuel continuity as a result of cheatgrass invasion (Whisenant 1990, Knapp 1995) and encroachment

of juniper and pinyon into some sagebrush plant associations and ecological sites (Tausch et al. 1981, Miller et al. 2008) suggest that fire return intervals have changed significantly across portions of the SGCA. Present-day fire regimes across the SGCA also have shifted spatially; much of the increased fire has occurred in the more arid Wyoming big sagebrush communities and decreased across many mountain big sagebrush communities. Both scenarios of altered fire regimes, resulting in conifer expansion at high-elevation interfaces and exotic weed encroachment at lower elevations, have caused a significant loss in Greater Sage-Grouse habitat (Knick et al. 2005, Wisdom et al. 2005b).

Shifts in fire regimes have significant conservation implications through changing the proportion of phases and new steady states in both time and space (Fig. 10.10). Repeat fires that eliminate or reduce shrubs, native grasses, and forbs; disturb soils and biological crusts; and release nutrients have allowed cheatgrass and other introduced annuals to replace the native shrub and herb layers. The converted landscape, which is largely composed of introduced annuals, is more susceptible to annual weather patterns and varies greatly from year to year, depending on moisture availability (Knapp 1995). In addition, shrub cover is absent from these landscapes, the season of available green plant material is shortened, high-quality perennial forbs are scarce, food for wildlife is limited or absent in late summer through winter, and the fire season is lengthened (Miller and Eddleman 2001).

Fires are an increasingly significant disturbance throughout much of the SGCA. Part of our recorded increases may be a function of differences in reporting fires and better technology to map fire polygons. However, the increases also suggest a significant spatial shift of more frequent fires occurring in the drier sagebrush plant associations than occurred prior to Euro-American settlement. The increase in areas burned each year coupled with decreases in total area of sagebrush habitats will further accelerate the trajectory of habitat loss for Greater Sage-Grouse.

Future effects of global climate change for sagebrush-dominated ecosystems 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 large-scale conversion of shrublands into

exotic annual grassland (Brooks et al. 2004a). Long-term changes in climate that facilitate or enhance invasion and establishment by invasive annual grasses further exacerbate the fire regime and accelerate loss of sagebrush habitats (d'Antonio and Vitousek 1992). Similarly, increases in woody vegetation will also increase fire potential (Neilson et al. 2005). Thus, effects of disturbances will be amplified by greater susceptibility for habitats to burn as well as the decreased likelihood for recovery of shrublands. The increasing amount of land-use activities also will have a significant influence on the soils, biological crusts, and vegetation of these systems and their ability to recover from the cascading effects created by invasive plants, fire, and climate change (Foley et al. 2005, Belnap et al. 2006).

The effects of climate change also must be factored into long-term conservation actions. The U.S. Department of the Interior, in considering potential listing for Greater Sage-Grouse under the Endangered Species Act, projected scenarios for extinction for 100–200 years into the future (United States Department of the Interior 2005b). Over these periods, the predicted changes in atmospheric carbon dioxide and temperature become increasingly important and are sufficiently large to overwhelm any current trajectories of habitat loss and alteration. We caution that projections of the effects of global warming become less reliable as they extend further into the future because of the complexity of interactions among the primary variables driving changes.

Land management agencies have placed major emphases on restoration programs, which are difficult and expensive, and may take centuries for complete restoration of a functioning system of sagebrush habitats within a landscape mosaic (United States Department of the Interior 1996, Hemstrom et al. 2002). The primary disturbance variables presented in this chapter—introduction of exotic plants, changes in wildfire, and climate change—will further complicate our ability to conserve and manage sagebrush communities as they now exist. They also call into question our ability to restore these ecosystems in the future. For example, sagebrush may be particularly susceptible to climatic influences on seedling recruitment (Maier et al. 2001, Perryman et al. 2001). The addition of new disturbance factors since Euro-American settlement of the region has created new steady states, including plant associations that did not exist in the past. Sagebrush communities that currently are the dominant

land cover across large areas are at risk of shifting to new states from which returns to previous states following land use or disturbance are unlikely (Laycock 1991, West and Young 2000).

In summary, sagebrush habitats are severely stressed across much of the SGCA, and their total area likely will decline in the relatively near future as a result of invasive species, fire, and climate change. Restoration programs are in relative infancy, and, together with the long-term dynamics of sagebrush systems, will require many years to have a substantial impact in slowing or stabilizing this loss. The end result is likely to be continuing and perhaps accelerated declines of many sagebrush-dependent species, including Greater Sage-Grouse.

ACKNOWLEDGMENTS

This chapter benefited from comments by nine anonymous reviewers from the Ecological Society of America, who reviewed previous material in the Conservation Assessment for Greater Sage-Grouse and Sagebrush Habitats (Connelly et al. 2004); and J. W. Connelly; and two reviewers of the revised manuscript. The U.S. Geological Survey Forest and Rangeland Ecosystem Science Center supported STK during development and writing. K. A. Fesenmyer, S. E. Hanser, and K. M. Ross updated the GIS figures and tables. We thank R. P. Neilson and R. J. Drapek for the global climate change data used in Figure 10.19.