

Big and Black Sagebrush Landscapes

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Introduction

Perhaps no plant evokes a common vision of the semi-arid landscapes of western North America as do the sagebrushes. A collective term, sagebrush is applied to shrubby members of the mostly herbaceous genus, *Artemisia* L. More precisely, the moniker is usually restricted to members of subgenus *Tridentatae*, a collection of some 20 woody taxa endemic to North America (Beetle 1960; McArthur 1979; McArthur and Plummer 1978). As a group, the *Tridentatae* are distinguished from other members of the genus by a combination of traits including their woody habit, floral morphology, stem anatomy, plant chemistry, and chromosomal karyotype (McArthur 1979).

The genus *Artemisia* originated on the Eurasian landmass during the mid-Tertiary as the late-evolving

Asteraceae rapidly diversified in response to global expansion of drier and cooler habitats (Beetle 1979; Raven and Axelrod 1974). Toward the end of this period, herbaceous, mesic-adapted progenitors to contemporary *Tridentatae* migrated across Beringia and spread across western North America eventually developing a woody habit (Beetle 1979; McArthur 1999; McArthur and Plummer 1978; Stebbins 1972). Opportunities for continued diversification were plentiful in the spatially and temporally diverse environment provided by the interaction of a complex geography with the increasingly variable climate of the Quaternary (Beetle 1979; McArthur and Plummer 1978). Over time, one widely adapted species, big sagebrush (*Artemisia tridentata*), emerged as the “most widespread and common shrub of western North America” (fig. 1) (McArthur and Stevens 2004).



Figure 1—A big sagebrush (*Artemisia tridentata*) landscape.

Scientific and management interest given to sagebrush in general, and big sagebrush in particular, can be measured by the considerable volume of literature generated primarily during the last half-century. McArthur and others (1979), Blaisdell and others (1982), and McArthur and Stevens (2004) provide useful reviews of the ecology and management of sagebrush species and ecosystems. Various papers presented in thematic symposia (McArthur and Welch 1986; Utah State University 1979) were effective in synthesizing available knowledge, and although dated, the published proceedings remain valuable reference materials. Literature summaries are available online by species or subspecies from the USDA Forest Service, Fire Effects Information System data base (<http://www.fs.fed.us/database/feis/>) (Howard 1999; Johnson 2000; McMurray 1986; Tirmenstein 1999a). Welch's (2005) big sagebrush synthesis is the latest and most comprehensive review published for this species complex.

Our purpose is not to provide yet another general sagebrush review, but to explore relevant published work and current thought regarding structure and successional processes, particularly as they relate to fire in ecosystems dominated by big sagebrush, and to a lesser extent, black sagebrush (*A. nova*). Although we briefly consider paleo distribution to provide context, our temporal focus will be primarily the last two centuries with emphasis on systematic changes that resulted from Euro-American settlement in the mid- to late-1800s. The time periods before and after this major cultural/ecological shift will be referred to as pre- and post-settlement. The geographical area of interest is represented by a broad zone in central and southern Utah where the Great Basin and Colorado Plateau meet, referred to here as the Southern Utah study area, or simply, the study area. Black sagebrush is included in the discussion because of its broad ecological overlap with big sagebrush and because of its widespread distribution and importance in the study area. In certain cases, such as the assessment of paleo and historic distribution, information is lacking for clear distinctions among sagebrush taxa. However, we believe that inferences made for the group as a whole will largely hold true for its dominant member, big sagebrush, and the closely allied black sagebrush.

Sagebrush Biology and Reproduction

Big sagebrush is a medium- to long-lived (20 to 200+ years; McArthur and Stevens 2004; Perryman and others 2001) aromatic evergreen shrub with one to several main stems (McArthur and Stevens 2004). The gray to black bark on older branches is shredded and

shaggy (Beetle 1960; McArthur and others 1979). Typical persistent leaves are small, pale green to blue-green, narrowly wedge-shaped, with three blunt teeth on the broadened end (McArthur and Stevens 2004). Spring ephemeral leaves are larger and generally more variable in shape and size than persistent leaves (Miller and Shultz 1987). Leaves and young stems are covered by a mat of fine hairs that provide a silvery cast. Inconspicuous, wind-pollinated flowers are held above foliage on fine, more or less erect stems. The wind-dispersed seeds (achenes) are small (4,000 to 6,000 seeds per g) and lack specialized appendages (Meyer and others 1988a; Welch 2005). Dispersal distance varies with topography and local conditions. Maximum dispersal distance for sagebrush seeds has been estimated at 30 m (98 ft) (Johnson and Payne 1968; Walton and others 1986), although the majority of seeds generally disperse less than 3 m (10 ft) from the mother plant (Walton and others 1986; Wambolt and others 1989). Seeds are short-lived and do not form a persistent seed bank (McDonough and Harniss 1974; Meyer 1990; Young and Evans 1989).

Three widely recognized subspecies of big sagebrush differ in a number of morphological and physiological traits (McArthur and Stevens 2004). Basin big sagebrush (ssp. *tridentata*) is the tallest, typically 1 to 2.5 m (3.3 to 8.2 ft), and has an uneven-shaped crown. The crown shape of Wyoming big sagebrush (ssp. *wyomingensis*) is similar to that of basin big sagebrush, however, plants are generally less than 1 m (3.3 ft) tall. Mountain big sagebrush (ssp. *vaseyana*) is intermediate in height at 0.6 to 1.5 m (2 to 5 ft). Seed stalks and foliage are of even height above the crown and give it a more flattened appearance than the other subspecies. Seed maturation and dispersal vary among subspecies and stand elevations (McArthur and Stevens 2004; Welch 2005) and are generally latest for basin big sagebrush (November to early December) and earliest for higher elevation populations of mountain big sagebrush (late September to October). Seed production is greatest for basin big sagebrush and least for Wyoming big sagebrush. Significant variation in palatability and nutritive content to wildlife and livestock has been documented and related to differences in the chemistry of secondary compounds (Welch and McArthur 1981, 1986; Welch and others 1981, 1987; Sheehy and Winward 1981). Dormancy in recently dispersed seed correlates with climate (elevation) of the collection site (Meyer and Monsen 1991, 1992). Although similar in many ways, black sagebrush differs from big sagebrush in a number of traits including shorter stature (generally less than 60 cm [24 inches]), darker appearance of leaves and reproductive stems,

larger seed size (2,200 seeds per g), and fewer seeds produced (McArthur and Stevens 2004; Meyer and others 1988b).

Geographic Distribution

Vegetation reconstructions based upon sediment core pollen records and woodrat midden macrofossil assemblages reveal that sagebrush-dominated ecosystems have been widespread during both the warm and cold phases characteristic of the Quaternary (last 2 million years). Studies suggest that during the late Pleistocene and early Holocene (40,000 to 10,000 years b.p.), sagebrush was widespread throughout most of its modern range (Mensing 2001; Nowak and others 1994; Rhode and Madsen 1995; Thompson 1990) and sagebrush-steppe ecosystems likely formed an ecotone with montane and continental tundra (Anderson and others 1999; Betancourt 1990; Fall and others 1995; Vierling 1998). Although the specific sagebrush taxa are unknown, a dominant sagebrush steppe association extended eastward into what is now the Central and Southern High Plains (Fredlund 1995; Hall and Valastro 1995) and into the desert valleys and plains of the southwest (Spaulding 1990; Van Devender 1990) during this same period. Sagebrush largely retreated from these eastward and southern range extensions during the hot dry conditions of the mid-Holocene (8,000 to 5,000 years b.p.) to an area similar to its modern distribution.

Today, big sagebrush is found throughout western North America from southern Canada to Baja California (McArthur 1999; McArthur and Plummer 1978). Beetle (1960) estimated that big sagebrush-dominated communities occupy approximately 586,306 km² (226,374 mi²) in 11 western states. Although this is considered an overestimate (McArthur and Stevens 2004; Wright and others 1979), its widespread ecological dominance remains impressive. With an estimated area of dominance of 112,150 km² (43,301 mi²) (Beetle 1960), black sagebrush communities occupy a greater total area than any other member of the *Tridentatae* except for big sagebrush and silver sagebrush (*A. cana*; McArthur and Stevens 2004).

Historical Conditions

A long standing controversy persists regarding the relationship between modern and pre-settlement distribution and condition of big sagebrush communities (Johnson 1986; Peterson 1995; Young and others 1979).

One view holds that in response to livestock grazing practices and altered fire regimes, big sagebrush invaded large landscapes that were predominantly grasslands (Arno and Gruell 1983; Christensen and Johnson 1964; Cottam 1961; Cottam and Stewart 1940; Hull and Hull 1974, Stewart 1941). In this context, big sagebrush is considered an indicator, or even as an agent, of grassland degradation justifying eradication in the name of restoration (Blaisdell and others 1982; Britton and Ralphs 1979). This view became entrenched in early range ecology dogma (Cottam and Stewart 1940; Stewart 1941; Stoddard 1941) and has retained popularity to the present. The opposing view claims that, with the exception of lands converted to other uses, the distribution of big sagebrush landscapes is essentially unchanged from historic times (Hironaka 1979; Johnson 1986; Vale 1975; Welch 2005). This view admits to changes in shrub dominance in response to disturbance (for example, livestock grazing; Austin 2000), but denies the supposition that significant change in vegetation type has occurred. This view is supported by arguments that expansion rates for sagebrush are too slow to account for significant range advances in the suggested time frame of approximately 100 years (Welch 2005).

Early written accounts produced by trappers, explorers, immigrants, and settlers have been interpreted to support both mindsets. Young and others (1979) found support for grass-dominated systems in Stewart's (1941) review of historical records of Utah's rangelands. In his treatment of the same document, Welch (2005) argues that grassy areas were not characteristic. Vale (1975) found consistent references to expansive sagebrush-dominated landscapes from central Wyoming to western Nevada and central Oregon in early journal/diary descriptions of vegetation along major migration routes across the western United States. He concluded that grasslands were restricted to canyon and valley locations with favorable soil moisture conditions. His interpretations are strengthened by the decision to only use accounts of observations made before heavy use by immigrants and their livestock. Analysis of published series of photographs taken as early as the 1870s and recent re-takes provide additional insight but fail to resolve the issue. Arno and Gruell (1983) and Kay (2003) provide photographic evidence and arguments in support of sagebrush invasion of grasslands in southwestern Montana and central Utah. In contrast, Johnson (1986), after examination of 1870s photos from Wyoming, northern Utah, and southeast Idaho, concluded that grasslands and shrublands have been quite stable for 115+ years in spite of a wide range of disturbances imposed during the

interim. Interpretation of frozen-in-time descriptions or photos is confounded by a lack of knowledge regarding disturbance history and the relative proportions of the various seral stages on historic landscapes (Young and others 1979). Thus, a definitive answer to the debate may never be found due to lack of reliable information (Johnson 1986; Young and others 1979) and because of the likelihood that no single answer is correct across the full geography of big sagebrush. However, after a review of the available evidence and arguments, it is our opinion that, allowing for defensible exceptions, changes in the distribution of sagebrush-dominated landscapes over the last 150 years came primarily in the form of reductions rather than expansions. The principal causes for these reductions are land use conversion, woodland expansion, and more recently, increased fire frequencies associated with invasive annual grasses.

Ecological Distribution and Associated Species

Big and black sagebrush are largely restricted to semi-arid climate regimes where winter temperatures are cool to cold and winter-spring precipitation is sufficiently reliable to support spring growth (Beetle 1960; McArthur

and Stevens 2004). Summer precipitation varies regionally, but soils at rooting depth are typically dry for much of the growing season. Soils on big sagebrush-dominated landscapes are moderately shallow to deep, well drained, and low in salt content (West 1979). Soil pH may vary from slightly acidic to moderately alkaline. Soils associated with black sagebrush tend to be drier and are generally more shallow or of higher percent rock than soils supporting big sagebrush (McArthur and Stevens 2004; Welch 2005).

Big and black sagebrush ecosystems form a wide, mostly continuous band across gradients in elevation in the southern Utah study area with lower limits defined by an ecotone with salt-desert shrublands and an upper boundary somewhat restricted by dense stands of montane or subalpine forest (fig. 2). Imbedded at mid-elevations within this sagebrush-grass matrix is a broad zone prone to recurrent invasion by species of pinyon pine and juniper (fig. 3) (Miller and others 1999; Tausch 1999; Tausch and Hood, this volume). This invasion belt is somewhat centered on the Wyoming-mountain big sagebrush transition zone (Goodrich and others 1999). These woodland species have been more or less permanent occupants on fire-protected topographical



Figure 2—Upper end of the sagebrush zone, with mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana*), shrubs, and grass intermixed with conifers and aspen (*Populus tremuloides*).



Figure 3—Utah juniper (*Juniperus osteosperma*) moving into a stand of big sagebrush (*Artemisia tridentata*).

units since their arrival in the Holocene (Tausch 1999; West and others 1998). For millennia, the rate and extent of expansion into the sagebrush matrix and frequency of retreat have been regulated by climate, topography, and fire regime (Tausch 1999; Tausch and Nowak 2000). The close association of sagebrush and woodland cover types in this zone led Tausch and Hood (this volume) to suggest that they are best considered as different phases of a single system (West and others 1978, 1998). They characterize a system of pinyon-juniper woodlands superimposed over a sagebrush-grass matrix with variation in the relative importance of the two types as an expression of variation in successional status across complex topographic and disturbance landscapes. This is clearly a refreshing and useful way to consider these dominant shrubland-woodland mosaics. For a detailed consideration of sagebrush shrubland-pinyon-juniper woodland successional patterns and management implications, see Tausch and Hood (this volume). Our focus here will primarily be sagebrush-grassland ecological processes that function independent of a woodland component.

Before settlement, Wyoming big sagebrush was the most abundant of the three subspecies of big sagebrush across its geographic range (West 1979). Within the southern Utah study area, it is a dominant on deep to moderately deep soils at elevations between 1,500

and 2,000 m (4,920 and 6,560 ft). Typical landforms include broad alluvial fans, low foothills, plateaus, and valleys receiving 170 to 350 mm (6.7 to 13.8 inches) annual precipitation (Goodrich and others 1999). At lower and upper ends of its range, it is bounded by, and intermixed with, elements of salt-desert shrub, black sagebrush, and mountain big sagebrush-pinyon-juniper communities (Howard 1999). The kind and abundance of sub-dominant shrubs and perennial grasses varies with soil attributes and disturbance history. Common shrub associates include species of ephedra (*Ephedra* spp.), rabbitbrush (*Chrysothamnus* spp.), snakeweed (*Gutierrezia* spp.), horsebrush (*Tetradymia* spp.), saltbush (*Atriplex* spp.), antelope bitterbrush (*Purshia tridentata*), and winterfat (*Ceratoides lanata*). Historically, perennial grasses dominated herbaceous understory. Common native species in the study area are Sandberg bluegrass (*Poa secunda*), bluebunch wheatgrass (*Pseudoroegneria spicata*), western wheatgrass (*Pascopyron smithii*), bottlebrush squirreltail (*Elymus elymoides*), Indian ricegrass (*Stipa hymenoides*), needle-and-thread (*Stipa comata*), and galleta grass (*Hilaria jamesii*). Perennial forb diversity and cover is relatively low on Wyoming big sagebrush sites (Bunting 1985).

Mountain big sagebrush in the study area is found on foothills and dry mountain slopes and ridges at

elevations of 1,900 to 3,000 m (6,230 to 9,840 ft) in moderately deep loamy soils (fig. 4) (McArthur and Stevens 2004). Annual precipitation varies from 300 to 700 mm (11.8 to 27.6 inches) (Goodrich and others 1999). At higher elevations it occurs in forest openings of various sizes in association with quaking aspen (*Populus tremuloides*), Douglas-fir (*Pseudotsuga menziesii*), Englemann spruce (*Picea englemannii*), white fir (*Abies concolor*), and ponderosa pine (*Pinus ponderosa*) (Johnson 2000). At lower elevations, mountain big sagebrush dominates many treeless landscapes and co-exists in shrub — woodland mosaics with single-leaf pinyon (*Pinus monophylla*; Great Basin), two-needle pinyon (*P. edulis*; Colorado Plateau), Rocky Mountain juniper (*Juniperus scopulorum*), Utah Juniper (*J. osteosperma*), and Gambel oak (*Quercus gambelii*). The importance of co- and sub-dominant shrubs varies with topography, soils, and disturbance history. Common associates include mountain snowberry (*Symphoricarpos oreophilus*), common juniper (*J. communis*), currants (*Ribes* spp.), Oregon grape (*Mahonia repens*), mountain mahogany (*Cercocarpus* spp.), serviceberry (*Amelanchier* spp.), antelope bitterbrush, rabbitbrush, and green ephedra (*E. viridis*). Numerous species of perennial grasses and forbs combine to make a productive understory. Species composition and annual biomass production vary with site productivity potential and disturbance history.

Basin big sagebrush grows in deep, well-drained soils of plains, valleys, low foothills, and canyon bottoms at elevations of 1,500 to 2,100 m (4,920 to 6,890 ft) in the southern Utah study area (McArthur and Stevens 2004). Mean annual precipitation is approximately 300 mm (11.8 inches). It is most often found in association with species common to upper Wyoming big sagebrush and lower mountain big sagebrush although populations intermixed with salt tolerant black greasewood (*Sarcobatus vermiculatus*), shadscale (*Atriplex confertifolia*), and saltgrass (*Distichlis spicata*) also exist (McArthur and Stevens 2004).

Within the study area, black sagebrush is most abundant at elevations of 1,500 to 2,400 m (4,920 to 7,870 ft) (fig. 5) (McArthur and Stevens 2004) with an extended elevational range of 1,400 to 2,780 m (4,590 to 9,120 ft), thus the elevational range of black sagebrush is nearly equal to the combined range of the three sub species of big sagebrush (McMurray 1986). Black sagebrush segregation from big sagebrush is due to its ability to grow in shallow, rocky soils and mixing of the two species is generally limited to narrow ecotones. On lower elevations, it is common to find black sagebrush in nearly pure stands with only sparse herbaceous understory and associated shrubs. As with big sagebrush, the diversity and abundance of associated shrubs and understory species increases with elevation.



Figure 4—Dense stand of mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana*).



Figure 5—Black sage (*Artemisia nova*) with needle-and-thread grass (*Stipa comata*).

Fire Effects

Although extreme weather (Anderson and Inouye 2001; Nelson and others 1989; Nelson and Tiernan 1983), insects (Haws and others 1990; Nelson and others 1989), and disease (Nelson and others 1989, 1990; Sturges and Nelson 1986) apparently play a significant role in big sagebrush population dynamics, and do so at multiple spatial scales, fire is believed to be the dominant disturbance force in natural populations (Wright and Bailey 1982). Curiously, big sagebrush lacks morphological or physiological adaptations to survive fire or facilitate rapid recolonization (Welch and Criddle 2003). Plant stature is low to the ground and wood, bark, and foliage are highly flammable resulting in complete shoot mortality of burned plants (McArthur and Stevens 2004). Top-killed plants do not re-sprout from roots or crown (Blaisdell and others 1982; Britton and Ralphs 1979; Peterson 1995; Wright and others 1979). Seeds mature and disperse after the risk of fire has all but passed (Beetle 1960; Young and Evans 1989). Seeds have no mechanism for long-distance dispersal (Chambers 2000; Johnson and Payne 1968; Walton and others 1986; Wambolt and

others 1999; Young and Evans 1989). The soil seed bank is ephemeral or absent (Beetle 1960; McDonough and Harniss 1974; Meyer 1990, 1994; Meyer and Monsen 1992; Young and Evans 1989). It appears paradoxical that the widespread landscape dominant, big sagebrush, is so poorly adapted to flames when fire is considered the “keystone disturbance” of western North American landscapes (Frost 1998; Keane and others 2002). Black sagebrush is no better adapted to fire than is big sagebrush, but dominates on sites less prone to burn. In contrast, many co-occurring shrubs have at least some ability to tolerate burning or to rapidly recolonize after fire (table 1). Fire adaptation by herbaceous species associated with big and black sagebrush varies, but is generally superior to that of these dominant shrubs (Britton and Ralphs 1979; Wright and Bailey 1982). Hence, a resolution to the apparent fire-big sagebrush paradox is not to be found solely in a species by species description of fire effects and adaptations, but in an examination of the manner in which fire is manifest on the landscape through time and space, also known as the fire regime.

Table 1—Fire adaptations for big sagebrush and co-occurring shrub species. Each species is rated on a scale of 0 to 4 for each area of adaptation where a 0 indicates no adaptation and a 4 indicates the species is highly adapted. See discussion and references in (Alekssoff 1999; Anderson 2004; Howard 1997, 1999, 2003; Johnson 2000; Marshall 1995; Tirmenstein 1999a, b; Welch and Criddle 2003; Zlatnik 1999).

Scientific name	Common name	Resprouting capability	Seed maturation timing	Seed dispersal distance	Seed bank
<i>Artemisia tridentata</i>	Big sagebrush	0	0	1	1
<i>Chrysothamnus nauseosus</i>	Rubber rabbitbrush	4	0	4	0
<i>Purshia tridentata</i>	Antelope bitterbrush	2	2	2	2
<i>Amelanchier alnifolia</i>	Saskatoon serviceberry	4	1	3	1
<i>Cercocarpus montanus</i>	True mountain mahogany	3	1	3	1
<i>Atriplex canescens</i>	Fourwing saltbush	2	0	1	3
<i>Ephedra nevadensis</i>	Nevada ephedra	3	3	1	2
<i>Symphoricarpos oreophilus</i>	Mountain snowberry	3	0	1	4

Big and Black Sagebrush Fire Regimes

Fire regime is quantified using various temporal and spatial parameters including frequency, seasonality, predictability, extent, and pattern (Morgan and others 2001). Fire regime is also expressed in terms of intensity, a measure of heat production per unit of time, and severity, a measure of fire-induced ecosystem change (Romme and others 2003; Ryan and Noste 1985). Fire regimes vary through time and across the landscape. Temporal variation is generally climate driven (Brown and others 2001; Grissino-Mayer and others 2004; Grissino-Mayer and Swetnam 2000; Heyerdahl and others 2002; Swetnam and Betancourt 1998), while spatial variation is primarily a product of topographic variation through its effects on species composition, productivity, desiccation rates, fuel continuity, and wind speed (Brown and others 2001; Heyerdahl and others 2001; Swetnam and Baisan 1996; Taylor and Skinner 1998). Fire characteristics vary locally in response to recent fire history and adjacency to fire prone landscape units (temporal and spatial autocorrelation; Morgan and others 2001).

Fire frequency is the most common measure of fire regime. It is an expression of the mean number of years between fire events, or mean fire interval (MFI), for a defined geographic unit. Estimates and interpretations of MFI are dependent upon spatial scale (Baker and Ehle 2001); the larger the area the shorter the interval in which no fire occurred. Hence, the most interpretable estimates of fire frequency are those associated with relatively small geographic units (Xiaojun and Baker 2006). Because populations of fire intervals frequently are not normally distributed, other measures of central tendency may be more appropriate for predicting fire free intervals (Grissino-Mayer 1999). However,

differences among candidate statistics are generally not ecologically significant. Conversely, interval variability can be important and is often overlooked. For species that must regenerate from seed, such as big sagebrush, the length and frequency of short intervals is most important in determining the compatibility of the fire regime with species persistence (Crawford and others 2004). Conversely, the length and frequency of long intervals are also important for ecosystems prone to invasion by fire sensitive species. The susceptibility of many big sagebrush landscapes to invasion by pinyon pine, juniper, or other conifer species in the absence of fire illustrates this point (Heyerdahl and others 2006; Miller and others 1999; Tausch and Hood, this volume). Thus, big sagebrush-dominated ecosystems provide clear examples of how a fire-free window, defined by both short and long interval statistics, can be more useful in determining ecosystem structure than are estimates of central tendency alone.

Estimates of MFI for forested ecosystems are most often generated from two types of dendrochronological evidence. Years of low severity or surface fires are determined from tree ring series with datable fire scars (Arno and Sneek 1977). Once injured, fire scarred trees become more susceptible to injury from subsequent fires. Consequently, individual fire-recording trees may provide evidence of a large percentage of low severity fires that burned at one location for extended time periods. More complete fire chronologies are obtained by combining fire records from annually cross-dated trees growing in close proximity (Dieterich 1980). Fire dates from severe, stand-replacing fires can be estimated based upon synchronous patterns in stand establishment dates (Heyerdahl and others 2001).

These methods are difficult to apply to sagebrush ecosystems except where fire-recording trees grow in isolated pockets or forest-shrubland ecotones. Such conditions are limited primarily to the more mesic mountain big sagebrush sites. Houston (1973) used fire scarred Douglas-fir and lodgepole pine trees growing at the ecotone between forest and mountain big sagebrush-grass steppe in northern Yellowstone National Park to estimate MFI in sagebrush steppe for that area. After adjusting data to reflect only pre-1900 conditions, he calculated mean single-tree MFI values of 32 to 70 years for all study units and 44, 56, and 50 years for trees growing in the sample units most representative of the whole study area. He considered this an overestimate of true MFI so, using composite chronologies of questionable accuracy (not cross-dated), he estimated MFI values of 20 to 25 years. Using a similar approach, Arno and Gruell (1983) examined fire frequency at forest-mountain big sagebrush-grass steppe ecotones of southwest Montana. They calculated pre-1900 MFI values of 41, 45, and 74 years for moist, dry, and hot-dry habitat types. These investigators suggested that these estimates were likely overestimates of MFI due to possible missing fire evidence and adjusted their estimate of forest “grassland” ecotone MFI to 35 to 40 years. They concluded that sagebrush distribution and density has increased considerably in this region due to a reduction of fire frequency during the last 100 years. In a more recent study from the same region, Heyerdahl and others (2006) sampled fire scarred Douglas-fir trees from a 1,030 ha (2,544 acre) site topographically characterized as a mosaic of forest islands and mountain big sagebrush-grass elements. They estimated an average fire return interval of 37 years for the study period (1700 to 1860) with a range of fire-free intervals of 2 to 84 years. This is similar to the results observed by Arno and Gruell (1983). They also quantified the increase in tree distribution and density that occurred after 1860. Miller and Rose (1999) estimated MFI for mountain big sagebrush steppe in a south central Oregon study area using fire scars from four isolated clusters of ponderosa pine trees located in the mountain big sagebrush-grass matrix. Composite, pre-1900 MFI ranged from 12 to 15 years for three of the four clusters. Seven major fires (three or four clusters affected) occurred between 1650 and 1880, resulting in an approximate MFI for major fires of 38 years. Miller and Rose concluded that, “In the mountain big sagebrush community, mean fire intervals, prior to 1871, ranged from 12 to 15 years...” The estimates generated by these studies are cited extensively in the literature and provide the basis for

a well-developed core conventional wisdom regarding sagebrush and fire.

Developing estimates of big sagebrush fire frequency directly from fire chronologies found on proxy species (trees) has the advantage of temporal precision (when properly dated) over long time periods. Spatial ambiguity is lessened by sampling from multiple locations on the landscape (Brown and others 2001; Heyerdahl and others 2001, 2006; Morgan and others 2001; Taylor and Skinner 1998). There are, however, disadvantages to this approach. Scarred trees are often scarce and distributed disproportionately across the landscape leading to spatial gaps in the record. Also, difficult to test assumptions must be made regarding historic fire regime continuity across the shrubland-forest ecotone. Perhaps the greatest problem with this approach rests in the fact that no suitable proxy species exist for the great majority of big sagebrush habitat types. Even if fire frequency estimates derived from the above cited studies prove to be accurate, there is considerable risk of inappropriate extrapolation of values to other localities.

Using an alternative approach, historic fire frequency for big or black sagebrush-dominated communities can be estimated indirectly based upon post-fire succession rates (Welch 2005; Welch and Criddle 2003). We suggest that the recovery pattern of big sagebrush to pre-burn conditions serves as an adequate index of post-fire succession for these plant communities. Several studies have attempted to quantify big sagebrush recovery time following both wild and prescribed fires. Harniss and Murray (1973) determined that big sagebrush on an upper Snake River Plain site in Idaho required at least 30 years to recover to pre-burn conditions and shorter fire-free intervals would lead to shrub dominance by species of horsebrush or rabbitbrush. Humphrey (1984) examined community composition in eight areas of southeastern Idaho where time-since-burn ranged between 2 and 36 years. His data indicated that big sagebrush (probably mountain big sagebrush) was still in a recovery phase 30+ years after burning. In Wyoming big sagebrush steppe in southwestern Montana, Watts and Wambolt (1996) observed that big sagebrush canopy cover reached 10 percent 30 years after burn treatment compared to the 13.5 percent for the unburned control. Wambolt and others (1999) observed similar delays in recovery for all three major subspecies after a wildfire burned sagebrush steppe communities north of Yellowstone National Park. Stand density for Wyoming, mountain, and basin big sagebrush was 2, 12, and 16 percent respectively, of unburned reference areas 19 years after the fire. In a southwestern Montana study of 13 spring

and fall prescribed burn sites (2 to 32 years post-burn), mountain and Wyoming big sagebrush canopy cover or stand density was significantly less for burned areas than for unburned areas in 36 of 40 comparisons (Wambolt and others 2001). The authors suggested that 30 years might be inadequate for full recovery in many cases. We further anticipate that recovery periods will often be longer after the more intense, mid-summer wildfires. Twenty years after a wildfire burned a central Utah site, West and Yorks (2002) found that Wyoming big sagebrush recovery had barely started based on the low density of sagebrush plants. They concluded that recovery rates for “sagebrush semi-desert” communities are much slower than they are for sagebrush steppe communities. These studies suggest that big sagebrush requires from 20 to 35+ years for post-fire stand recovery under favorable conditions and much longer intervals when conditions dictate a slower pace of recovery. Correspondingly, longer intervals are expected on xeric sites where fine fuel production under average weather conditions is inadequate to carry fire except under severe conditions.

There is little information regarding historic fire regimes for black sagebrush-dominated landscapes. It is generally believed that fire was rare on these landscapes due to insufficient fine fuels to carry fire (Wright and others 1979). Miller and Rose (1999) found evidence of just two fires in 300 years of record for a low sagebrush (*Artemisia arbuscula*)-western juniper (*Juniperus occidentalis*) community. Low sagebrush is similar to black sagebrush and like black sagebrush, occupies low productivity sites (McArthur and Stevens 2004). Although this information is insufficient to estimate MFI, it is sufficient to support the notion of long fire-free intervals for sagebrush communities with low fine fuel production.

In cases where a significant number of big sagebrush seedlings establish from surviving residual seeds, stand recovery is relatively rapid (Bunting 1985), and the size, pattern, and continuity of the burn has little impact on recovery time. This phenomenon is most common when fires burn mature stands of mountain big sagebrush, but has also been observed with basin and Wyoming big sagebrush on mesic sites (Wambolt and others 1999, 2001). Frequently, big sagebrush seedlings do not establish within 1 year post-burn, either because of a lack of viable seeds on the landscape, or because of the failure of seeds to produce viable plants (Welch and Criddle 2003). When this occurs, big sagebrush recovery is dependent upon seed dispersal from unburned source areas and favorable weather patterns. Fire size, pattern, and continuity directly impact the distance that seeds

must disperse, and thus have considerable impact on recovery time (Crawford and others 2004). Welch and Criddle (2003) provide an example that illustrates this effect. They measured the down wind (west to east) advance of mountain big sagebrush 14 years post-burn on a southern Idaho site and determined a mean annual spread rate of 13 m (43 ft). Northward spread was half that of the eastward spread and big sagebrush encroachment from the north and east burn margins was negligible. Based on these data, they estimated that it would take 71 years for big sagebrush to reoccupy this 146+ ha (360+ acre) burn. In effect, large continuous fires without unburned islands result in long seed dispersal distances that translate into long recovery periods while recovery from small discontinuous fires with short seed dispersal distances is more rapid.

Although big sagebrush post-fire recovery time varies situationally, a general relationship between recovery time and minimum (significantly shorter than the mean) fire-free intervals should be expected. Therefore, estimates of MFI (or any other measure of central tendency) must be substantially longer than estimates of mean recovery time in order to capture the full variability in fire interval duration. Conversely, we recognize that on landscapes prone to conifer invasion, lengthy intervals would result in shrubland displacement by woodlands (Tausch and Hood, this volume). Taken together, these assumptions provide the theoretical basis for our estimates of MFI on sagebrush-dominated landscapes. We suggest that historic MFI values ranged from 40 to 80 years for mountain big sagebrush and some productive basin and Wyoming big sagebrush communities and were as long as 100 to 200 years or longer for big and black sagebrush sites with low productivity. We offer broad estimates here in order to capture the range in MFI length we believe existed across the full ecological and geographical distribution of big sagebrush. A range of intervals lengths must be expected in conjunction with a single MFI value. For example, natural variability corresponding to a MFI of 50 years might produce intervals as short as 10 to 15 years and as long as 100 to 120 years; however, most intervals would likely fall between 25 and 75 years. Our estimates are similar to pre-1900 MFI values calculated from tree-ring records at forest-shrubland ecotones by Heyerdahl and others (37 years; 2006), Houston (32 to 70 years; 1973), and Arno and Gruell (41 to 74 years; 1983) before data adjustment. Although the estimate for historic MFI of 12 to 15 years proffered by Miller and Rose (1999) is unrealistic, the approximate MFI of 38 years derived from their data for widespread landscape fires only

approaches our estimate and is likely a more realistic application of their data to the mountain big sagebrush matrix of their study area. In their North American synthesis of fire ecology, Wright and Bailey (1982) estimate MFI for mountain and Wyoming big sagebrush at 50 and 100 years – numbers that are in general agreement with what we suggest here.

So, what of the big sagebrush-fire paradox? It seems that in contrast to strategies employed by co-occurring shrubs (table 1), big sagebrush solved the fire problem by producing highly competitive, yet disposable plants. It does not invest resources in morphological or physiological adaptations to fire, as it never had to in its short evolutionary past. This was particularly true for the 2+ million years of the Pleistocene, during which time cooler climatic conditions would have rarely favored fire to the extent they do today. Sagebrush thrives on suitable landscapes as long as the fire-free intervals are sufficiently long to permit re-establishment of mature stands, and short enough to prevent displacement by forest or woodland (Miller and Tausch 2001). This leads us to the conclusion that past variation in big sagebrush distribution and dominance was to some degree influenced by climatically-driven changes in fire regime parameters. Where changes were large and persistent, fire regime-driven shifts among shrub-steppe, grassland, and woodland or forest ecosystems must have occurred. Consequently, caution is warranted when making comparisons of big sagebrush distribution or dominance from time periods experiencing distinct climatic patterns. Judgments made when comparing pre-settlement big sagebrush conditions corresponding to the end of the “Little Ice Age” with contemporary environmental conditions should be tempered by the context of the corresponding change in climate.

Variation in historic fire regimes might also be attributed to variation in human-caused ignitions. Fire was the most important tool available to aboriginal inhabitants for manipulating the natural environment (Griffin 2002; Williams 2004). It was used to promote the growth of desirable resource plants, enhance habitat for important animals, and drive game and insects (Griffin 2002). It might also have been important for warfare, clearing travel corridors, and providing fire-safe camp sites (Williams 2004). Anecdotal and ethnographic accounts describing the use of fire within the big sagebrush domain are reviewed by Baker (2002), Griffin (2002), and Whitlock and Knox (2002). These authors conclude that, in spite of known patterns of fire use, it is difficult to find available evidence sufficient to attribute landscape-scale variation in pre-settlement

fire regimes to human fire practices. Thus, we are left to consider the ecological consequences of historic fire regime variability with little ability to ascertain the human role in that variation.

Finally, we pose the question, “What does big sagebrush-grass climax look like in the absence of woodland invasion and fire?” Current theory suggests that, with time, competition from big sagebrush will eventually reduce perennial grasses and forbs to scattered remnants (Miller and Tausch 2001). Indeed, one does not need to look far when in sagebrush country to find dense stands of sagebrush with depleted herbaceous understories. However, the question can not be properly addressed without also considering the impact of domestic livestock on the competitive relationship between shrubs and perennial grasses (Austin 2000). Unfortunately, there are few reference areas not impacted, either historically or currently, by livestock grazing. The question is thus compounded by the additional variable and becomes, “What does big sagebrush-grass climax look like in the absence of fire, woodland invasion, and livestock grazing?” Although somewhat short term in nature, results from a 45-year study (Anderson and Inouye 2001) found that after livestock removal, a dynamic equilibrium was reached between shrubs and perennial grasses where both were well represented in the plant community. Other studies show significant, sustained increases in percent grass cover concurrent with increases in big sagebrush cover following livestock reduction or removal (Branson and Miller 1981; McLean and Tisdale 1972; Pearson 1965). We suggest that such a dynamic balance should be expected between big sagebrush and grasses in a variety of settings, although the actual nature of the balance is likely to differ substantially from place to place. This is not to suggest that fire is not a fundamental ecological process in sagebrush grass communities, nor that fire should not be used as a management tool. Rather, we suggest that losses of perennial grasses in big sagebrush-dominated communities may have more to do with the effects of intense, selective grazing than with the periodicity of fire.

Euro-American Settlement

Ecological processes associated with sagebrush/grass ecosystems in the southern Utah study area began to be altered soon after Euro-American settlement in the mid-1800s (Young and others 1979). First settlers were primarily small groups of Mormons sent by their leader, Brigham Young, to establish organized communities wherever water, timber, and forage conditions permitted. Numerous agricultural-based communities were started

where mountain streams met valley floors. Early settlers soon learned that the deep, loamy soils associated with tall (basin) big sagebrush were well suited for irrigated crops and significant acreages of this type were converted to cropland (Kearney and others 1914).

Originally, settlers pastured livestock anywhere good grass could be found close to their communities. Although few settlers owned many animals individually, livestock were commonly united in community herds (Young and others 1979). Cool-season grasses growing in association with big sagebrush at valley and foothill locations were used heavily and sometimes year-round when located at close proximity. The area impacted by grazing livestock gradually expanded as the local forage base depleted, herd numbers increased, and concern for potential losses to opportunistic native peoples decreased.

Livestock numbers, especially sheep, grew rapidly in the late 1870s and 1880s (Keck 1972; Murdock and Welsh 1971) as livestock production shifted from a subsistence to a market economy. In addition to expanding local herds, large numbers of sheep were trailed and eventually freighted by train to and from the area by parties with little or no interest in the grazing needs of local communities. Sheep herds often spent several months trailing between mountains in fall and summer and foothills and deserts in winter and spring. By 1890, livestock numbers far exceeded realistic estimates of carrying capacity and the degradation of plant/soil environments was widespread. Over-grazing persisted for several decades resulting in widespread degradation of big and black sagebrush landscapes similar to that imposed on other plant community types.

Current Conditions

Ungulate Impacts

To some degree, pasturing of domestic livestock on big and black sagebrush-dominated landscapes affected, and continues to affect, community structure and ecological processes at all grazing intensities. Minimum effects include the removal of fine fuels and subsequent reduction in capacity to carry fire. At higher grazing intensities, palatable herbaceous species were weakened or eliminated. The ecological changes caused by these alterations occurred rapidly on productive landscapes characterized historically by relatively short fire-free intervals and a propensity for invasion by woodland or forest conifers. In response to a weakened, and often depleted

herbaceous understory, big sagebrush density and canopy cover increased and the pace of woodland invasion was accelerated (Miller and Tausch 2001; Tausch 1999). The combination of reduced fire and accelerated woodland invasion resulted in loss of landscape level heterogeneity and a major shift in the sagebrush-woodland complex to increasingly more widespread and dense woodland dominance (Tausch and Hood, this volume). Similar degradation has occurred on drier landscapes, although woodland expansion is either absent or occurring at a slower pace.

Exotic Weed Impacts

Immigrating Europeans intentionally and accidentally brought the seeds of numerous plant species to North America. Some of these species are invasive in big sagebrush habitats. The winter annual, cheatgrass (*Bromus tectorum*), is particularly well adapted for big sagebrush habitats and has had considerable success invading the weakened and often depleted understories of basin and Wyoming big sagebrush, mid elevation black sagebrush, and drier mountain big sagebrush communities (Young 1994). During years with good spring moisture, cheatgrass produces large quantities of continuous fine fuels that cure earlier in the season than do native perennials and effectively lengthen the fire season. Where established, cheatgrass has resulted in shorter fire-free intervals, earlier fires, and larger, more continuous fires than existed historically (Peters and Bunting 1994; Whisenant 1990). Once established, cheatgrass out-competes seedlings of native perennials disrupting natural regeneration processes (Billings 1994; Young 1994; Young and Evans 1978). Where the perennial herbaceous understory is depleted, the cheatgrass-fire cycle eventually reduces what is left of sagebrush-dominated communities to cheatgrass-dominated annual grasslands (Pellant 1990; Young and Evans 1978).

Fire Regime Condition Classes

The classification of existing vegetative communities based upon the degree of departure from historic conditions and the risk of loss of one or more defining components provides a framework to guide restoration and management efforts. Three broad classes reflect increasingly greater departure from historic conditions implicating parallel increases in intervention needs.

Fire Regime Condition Class 1 (FRCC 1)

Herbaceous species diversity and relative cover for healthy (FRCC 1) big or black sagebrush-grass communities in the southern Utah study area vary across gradients in site productivity. In general, diversity and relative cover decrease with increasing aridity. The extreme of this condition can be observed in some low elevation (xeric) black sagebrush communities that essentially function as stable shrublands with extensive bare interspaces and few scattered herbs (McArthur and Stevens 2004). Conversely, mesic stands of mountain big sagebrush have a diverse assemblage of perennial grasses and forbs that resist weed invasion (Anderson and Inouye 2001; West and Yorks 2002). Depending on disturbance history, percent cover of mountain big sagebrush varies from <5 percent, 1 to 20 years post burning, to 20 to 40 percent at shrub/grass equilibrium 30 to 70 years post burn (Harniss and Murray 1973; Humphrey 1984; Wambolt and others 2001; Welch 2005; Welch and Criddle 2003). Mature sagebrush stands are generally multi-aged (Perryman and others 2001). On the landscape scale, multiple seral stages are represented in an ever-shifting mosaic reflecting periodic reoccurrence of fire or other disturbances (Crawford and others 2004). Although landscape-level complexity may appear to decrease with the longer fire-free intervals expected for basin and Wyoming big sagebrush sites, micro-scale compositional variation may in fact increase as the plant community has time to fine tune responses to small variations in the physical environment (Anderson and Inouye 2001). Typically, sagebrush cover for basin and Wyoming big sagebrush-dominated communities ranges from 15 to 35 percent (Welch and Criddle 2003). If present in FRCC 1, cheatgrass is scattered and forms an insignificant portion of the herbaceous biomass. At mid to lower elevations, cryptobiotic crusts may be present on bare inter-shrub openings. Woodland species are widely scattered, if present. Fires are generally patchy. If grazed by livestock, light to moderate stocking rates and periodic rest during the growing season are needed to maintain the herbaceous component of this condition class.

Fire Regime Condition Class 2 (FRCC 2)

FRCC 2 for sagebrush-grass communities occurs with and without elements of woodland invasion. The herbaceous perennial component in this condition class is moderately depleted in abundance or diversity relative to FRCC 1. A shift in species composition reflecting palatability is often noticeable. The primary causes are

chronic overstocking of livestock and periodic abusive grazing practices. Sagebrush cover often exceeds that of FRCC 1 due to competitive release by a weakened herbaceous understory. On drier sites, cheatgrass is generally present (some soil types excepted), but does not dominate except in scattered patches. Current fire-free intervals are often much longer than estimated historic intervals. However, the risks of weed invasion for this condition class are greater than the risks of woodland conversion. The probability of conversion (transition) from a shrub-grass community to weed-dominated grassland is moderate.

Tausch and Hood (this volume) give descriptions of big and black sagebrush landscapes susceptible to invasion by woodland trees. In FRCC 2, trees have reached 25 to 50 percent of their potential cover for the site and shrub, and herbaceous cover has been reduced by up to 75 percent. Total conversion to tree dominance (FRCC 3) may occur in 40 to 50 years on moderately productive sites (Tausch and Hood, this volume). Because cheatgrass invasion frequently occurs synchronously with woodland expansion, fire compounds risks. The risk of cheatgrass expansion with fire counters that of conversion to woodland (loss of shrubs and herbs) without fire. Alternative non-fire treatments or combinations of treatments are needed for sagebrush-grass restoration.

Fire Regime Condition Class 3 (FRCC 3)

Characteristically, perennial herbs are severely depleted in FRCC 3 big sagebrush communities of the study area. Weedy annuals, especially cheatgrass, dominate the herbaceous understory. Big sagebrush plants tend to be old-aged with little new recruitment. Shrub cover may be variable. There is a high probability that a single fire event will result in conversion to an annual-dominated community with corresponding short fire-free intervals and large fire size. Indeed, outside of the study area, extensive areas of big sagebrush plant communities have already experienced this type conversion (Billings 1994; Miller and others 1999; Whisenant 1990; Young and Evans 1978). The current fire-free interval may be much longer than estimated historic intervals; however, the greater risks for this condition class are associated with shortened fire-free intervals due to the loss of perennial herbs and probability of invasion by weeds such as cheatgrass. Where susceptible to woodland invasion, trees may have reached densities sufficient to fully occupy the site (Tausch and Hood, this volume). Shrubs and perennial herbs are weak and scattered in tree openings. Litter accumulates under trees and extensive bare soil may be exposed and eroding between

trees. Long-term overgrazing of stable black sagebrush communities by domestic sheep can result in conversion to a FRCC 3 characterized by nearly solid stands of low rabbitbrush (*Chrysothamnus greenii*) or broom snakeweed (*Gutierrezia sarothrae*).

Fire Regime Condition Class Assessment

We assessed condition class for big and black sagebrush sites in the southern Utah study area using data and photographs obtained from 148 sites between 1997 and 1999 (Davis and others 2004). Sites with boundaries roughly corresponding to the study area were selected within Utah Division of Wildlife Resources Wildlife Management Units 16, 19 through 25, and 27 through 30. Study sites in which neither big nor black sagebrush were listed as descriptive, that appeared to have been recently treated (chaining or seeding), or were protected from grazing were excluded. A second assessment was made for 117 of the sites from photos and data collected 5 years after the first field visit. Data from four additional sites were added for this later period. The number of sites per sagebrush taxon was 59, 15, and 57 for mountain, basin, and Wyoming big sagebrush, respectively, and 21 for black sagebrush. These data are the most comprehensive available and provide a sound basis for estimating big and black sagebrush condition class for the study area.

Four photographs per site/assessment period were used to make a qualitative assessment of condition class. An assessment of woodland encroachment (primarily juniper) was also made from photographs. As a quantitative indicator of condition for each site, we used the Desirable Components Index (DCI) developed by Davis and others (2004). The DCI is computed on a scale of 0 to 100 and is based on cover percentages for shrubs, perennial grasses, perennial forbs, and annual

grasses and the presence of noxious weeds. Shrub values are adjusted based upon size class (seedlings, young, and mature) and vigor (normal, decadent, and dead) distributions. Optimal values for the DCI are obtained by 20+ percent shrub cover, 15+ percent perennial grass cover, 5+ percent perennial forb cover, no annual grasses or noxious weeds, shrub decadence less than 20 percent, and percent young shrubs greater than 10 percent. For more details on DCI computation see Davis and others (2004). We derived condition class estimates from site DCI scores using two scales that correspond to mesic and xeric habitats (table 2).

Our qualitative and quantitative estimates of site condition class are in general agreement (table 3). Averaged across all sites, condition class ratings associated with the late 1990s assessment are somewhat higher than those for the 2002 to 2004 assessment. This difference reflects the severity of a regional drought from 1999 to 2004. Shrub and perennial grass mortality was particularly striking in 2003 at low elevations in the southern and eastern portions of the study area. These data suggest that from 10 to 20 percent of big sagebrush landscapes and 30 to 50 percent of black sagebrush landscapes in the study are currently in FRCC 1. Approximately 40 to 60 percent of the four sagebrush taxa are in FRCC 2. Relative area in FRCC 3 is highest for basin and Wyoming big sagebrush sites and lowest for mountain big sagebrush and black sagebrush. Moderate to advanced encroachment by juniper was not different for the three big sagebrush subspecies (40, 40, and 38 percent for mountain, basin, and Wyoming sites, respectively). We rated encroachment as moderate to advanced for 29 percent of black sagebrush sites. These estimates of woodland encroachment may be somewhat low due to a possible bias against woodland-dominated sites in the site selection criteria.

Table 2—Scales for deriving condition class from Desirable Components Index (DCI) scores (Davis and others 2004).

Scale	Condition class	DCI score
Scale 1 – mountain big sagebrush and upper elevation black sagebrush	1	70+
	2	45-69
	3	<45
Scale 2 – basin and Wyoming big sagebrush and lower elevation black sagebrush	1	55+
	2	30-54
	3	<30

Table 3—Big and black sagebrush condition class estimates for 152 sites in the southern Utah study area based upon Desirable Components Index (DCI) and qualitative assessment of photographs for each site (Davis and others 2004). Percentages are based upon 148 and 121 sites for the 1997 to 1999 and 2002 to 2004 assessments, respectively.

Sagebrush taxon	Condition class	Condition class estimate			
		1997 to 1999 DCI	Assessment photos	2002 to 2004 DCI	Assessment photos
----- <i>Percent of sites</i> -----					
Mountain big sagebrush	1	26	31	13	9
	2	64	64	58	74
	3	10	5	29	17
Basin big sagebrush	1	20	20	13	19
	2	53	40	47	37
	3	27	40	40	44
Wyoming big sagebrush	1	39	28	15	5
	2	47	53	48	65
	3	14	19	37	30
Black sagebrush	1	52	43	50	33
	2	39	57	36	58
	3	9	0	14	8
All taxa combined	1	34	31	18	11
	2	53	56	50	64
	3	13	13	32	25

Recommended Treatments

A variety of treatments has been developed to modify and restore big sagebrush communities. The appropriateness of each is dependent upon site condition class, existing uses, available resources, and management goals (Monsen 2004). The underlying principle is to repair structure and ecological processes in existing vegetative communities (Whisenant 1999). Treatment objectives for FRCC 1 and many FRCC 2 sites are maintenance in nature and include: reduction of big sagebrush density or cover; removal of young pinyon or juniper trees; increase density, productivity, or diversity of perennial herbaceous understory; increase productivity of associated shrub species; and create spatial heterogeneity among seral stages across the landscape. Potential treatments that are effective in achieving those objectives include prescribed fire; selective herbicide application; and low-impact mechanical treatments such as anchor chaining and riling. Objectives for treating lower end FRCC 2 and FRCC 3 sites are clearly remediation focused and include soil stabilization, water capture and retention, and reconstruction of resilient shrub-perennial grass

communities. Practices employed to achieve these objectives include removal of woodland trees, control of invasives (usually annuals but may include perennials), and restoration plantings of perennial herb and shrub elements. Successful restoration of big sagebrush-grass communities from FRCC 3 to FRCC 1 or 2 is generally an expensive multi-step process requiring combinations of treatments (Lancaster and others 1987) and fortuitous timing and quantities of precipitation after planting. Pre-emergent herbicides and tillage treatments have proven to be at least moderately effective in controlling invasive annuals. Bio-control methods for controlling cheatgrass have been investigated (Kennedy 1994; Meyer and others 2001); however, to date, are undeveloped or unproven. Prescribed fire and high impact mechanical treatments are employed to remove woodland trees. Restoration plantings require proper seedbed preparation and timely planting of appropriate seed mixes of adapted, compatible ecotypes (Monsen and Stevens 2004). Appropriate long-term management practices, including changes in livestock use, are essential after treatment. In the following, we discuss the advantages and limitations of these treatment options.

Prescribed Fire

Prescribed fire is an efficient, cost effective method for removing big sagebrush and woodland trees on portions of the landscape (Britton and Ralphs 1979; Bunting and others 1987; Wright and others 1979). Because it mimics natural fire, properly timed prescribed fire is supportive of natural ecological processes of nutrient cycling and plant succession. As previously discussed, time needed for big sagebrush recovery after burning can vary greatly and depends upon community composition before the burn, fire intensity (linked to season of burn), fire size and pattern, and weather conditions after the burn (Bunting and others 1987). Non-target species can be damaged, and increased herbaceous production is not always realized after burning (Britton and Ralphs 1979; Bunting and others 1987; Welch 2005; Wright and Bailey 1982; Wright and others 1979). Prescribed fire, or wildland fire use, should be limited to stands where perennial grasses and forbs are sufficiently abundant to preclude the risk of expansion by cheatgrass or other fire tolerant invasives. Typical sites are mountain big sagebrush communities in FRCC 1 and 2. Restoration fires should be small or patchy, facilitating the perpetuation of a mosaic of seral stages and minimizing seed dispersal distances for recovering big sagebrush. Large patches of mature sagebrush should be left unburned as critical wildlife habitat. Based on a 50 year MFI, mean area burned per year (natural and wildfires combined) for mountain big sagebrush should not exceed 2 percent. Prescribed fire should not be considered for dry basin and Wyoming big sagebrush and black sagebrush stands in the study area due to slow recovery time and the high risk of conversion to weeds. Deferral from livestock grazing for 1 or more years before burning may be necessary to allow for fine fuel accumulation and curing (Whisenant 2004; Wright and others 1979). A post-treatment rest from grazing of one to two growing seasons (Bunting and others 1987; Whisenant 2004; Wright and others 1979) should be considered a minimum requirement that is not always adequate. Burning restrictions often result in narrow windows of opportunity for treatment and may require rapid mobilization. Wright and Bailey (1982), Bunting and others (1987), and Whisenant (2004) outline guidelines for prescribed burning of big sagebrush communities.

Herbicides

Herbicide treatments are used effectively as substitutes for fire to reduce big sagebrush cover. The volume of literature dedicated to the development and testing

of various compounds for this purpose reveals the level of interest that existed during the mid 1900s in finding novel ways to control or eradicate this species (Crawford and others 2004; Welch 2005 and references therein). Herbicide selectivity and effectiveness varies with concentration, season of use, soil characteristics, and community composition (Vallentine 2004; Welch 2005). Here we discuss the use of the two herbicides most frequently used to control big sagebrush. Early work focused on the use of 2,4-D [(2,4-D-dichlorophenoxy) acetic acid] a synthetic auxin, or plant growth regulator (Welch 2005). Although effective in controlling big sagebrush short-term, long-term effects on the plant community were difficult to predict (Watts and Wambolt 1996; Welch 2005). This may be due to its effects on non-target species, especially broadleaf forbs. Consequently, 2,4-D is no longer the herbicide of choice for big sagebrush reduction (Crawford and others 2004). Tebuthiuron ([N-[5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2-yl]-N,N'-dimethylurea]; Spike[®]), a photosynthesis inhibitor, is applied to the soil where it moves into the rooting zone with water and remains active for several years (McDaniel and others 2005; Wachocki and others 2001). It is absorbed through the roots and is functionally selective against big sagebrush at low application rates (Baxter 1998; Crawford and others 2004; McDaniel and others 2005; Wachocki and others 2001). Post-treatment increases in productivity for herbaceous species can be substantial (Baxter 1998; McDaniel and others 2005; Olson and Whitson 2002). Herbicide treatments, particularly the use of tebuthiuron, pose certain advantages over prescribed fire. Spatial precision of treatment application is greater with herbicide application than with prescribed fire. Longer windows of opportunity for treatment are available, especially for tebuthiuron (Baxter 1998; Marion and others 1986), than for prescribed fire. Damage to non-target species is often less with tebuthiuron than with prescribed fire (Baxter 1998; McDaniel and others 2005). The level of "thinning" and associated treatment longevity are effectively regulated by altering application rates (Crawford and others 2004; McDaniel and others 2005; Olson and Whitson 2002; Wachocki and others 2001), although these must be calibrated for soil texture and precipitation (Baxter 1998). There are disadvantages to using tebuthiuron relative to prescribed fire including greater per-acre cost and ineffectiveness in controlling woodland trees. Tebuthiuron has greatest application where there is a need to reduce sagebrush density or cover to allow existing herbaceous understory to respond to the competitive release. It is particularly valuable

where prescribed fire is not practical or where complete removal of big sagebrush is undesirable. Tebuthiuron may have application with FRCC 2 basin and Wyoming big sagebrush communities where the ecological risks of prescribed fire would be excessive.

A second class of herbicides is used to control invasive annuals, such as cheatgrass, in big sagebrush communities. Broad spectrum contact herbicides are effective, but collateral damage to non-target species can complicate their use. Soil active, pre- or post-emergent herbicides have proven effective in killing annual weeds in early post-germination stages, thus effectively depleting the seed bank and releasing residual perennials from weedy competition. Applications of sulfometuron methyl (Oust[®]) have proven effective in providing a 1- to 2-year window of greatly reduced competition from cheatgrass (Pellant and others 1999). Questions remain regarding residual time in the soil and impacts on established perennials. In recent years, considerable interest has been generated for imazapic (Plateau[®]) as a soil-active herbicide. Early results suggest that this herbicide is quite selective in its effects and that it is particularly effective with annual bromes (Bekedam and Pyke 2004; Porath and others 2003; Smith and Anderson 2003; Whitson 2003). This kind of treatment has application on FRCC 2 sites to encourage release of weakened herbaceous plants and on FRCC 3 sites to deplete the weed seed bank in preparation for restoration plantings.

Mechanical Treatments

A wide variety of mechanical techniques has been devised to eliminate invading pinyon and juniper trees from sagebrush grass communities and to reduce cover and density of big sagebrush. These treatments are generally used as precursors or sometimes simultaneously with restoration/reclamation plantings (Monsen 2004; Stevens 1999). Treatment effectiveness and management considerations for the mechanical control of woodland trees are discussed in Stevens and Monsen (2004a) and Tausch and Hood (this volume). Parker (1979), Mattise and Scholten (1994), Welch (2005), and Wiedemann (2005) provide brief but adequate descriptions of major equipment developments and their applications. Various plow and disk type implements kill most big sagebrush plants, as well as associated, species necessitating follow-up plantings of adapted species. Mature woodland trees and a high percentage of non-sprouting shrubs can be removed by dragging a long section of anchor chain between two crawler tractors. Actual treatment outcome is affected by link weight and modifications, chain length, relative tractor positions, treatment passes, and tractor

speed (Stevens 1999). Low-impact treatments leave a majority of herbaceous species intact. A second pass of the chain (two-way chaining) improves juniper kill and is reasonably effective in burying seeds broadcast after the first pass as part of restoration plantings (Stevens 1999). Chaining is a preferred technique on rough terrain up to 20 to 30 percent grade. The disk chain is an implement that combines design features of the anchor chain and disk implements. The railer and pipe harrow are implements that are dragged behind tractors. They are designed to remove mature sagebrush and leave some smaller plants intact. Damage to herbaceous species is minimal; however, the pipe harrow creates enough soil disturbance to facilitate seed burial and establishment of desired species (Welch 2005). Equipment and practices should be selected to minimize risks to soil erosion. Archeological surveys are required prior to mechanical treatment in order to avoid cultural site disturbance. These treatments are generally used for FRCC 2 and 3.

Restoration Plantings

The concept of restoration planting can be defined in either broad or narrow terms. Narrowly, a restoration planting is seen as an attempt to re-establish a native plant community that is indistinguishable, or nearly so, in composition, structure, and ecological process from what is perceived as the natural state. This view dictates a careful selection of source germplasms for plant propagules, usually seeds, which are consistent with the goal. Although a worthy target to aim for, actualization of this kind of restoration is generally difficult to achieve for big or black sagebrush communities. Alternatively, a broader view of restoration plantings includes all attempts to establish complimentary assemblages of plant species that structurally and functionally resemble pre-disturbance conditions in so far as the level of site degradation will allow. Developed cultivars of native and introduced species may be planted in various combinations deemed most likely to achieve goals of site stabilization and other management objectives. Intermediate approaches with varying restrictions on plant material origin are common for big sagebrush-grass plantings (Roundy and others 1997; Stevens and Monsen 2004b). Although restoration practices vary, most are designed to either facilitate natural repair processes or supplant them (Whisenant 1999). Examples of facilitative actions associated with the repair of big sagebrush-grass communities might include protection of residual big sagebrush islands to allow natural seed dispersal into surrounding treated or disturbed landscapes (Longland and Bateman 2002) and delaying livestock grazing until after perennial grass seed shatter to allow maximum seed production and dispersal

on soil surfaces. Although facilitative restoration can be a slow process, costs are relatively low, allowing treatment of large areas.

Most efforts at developing restoration methodologies for big sagebrush-grass communities have taken a more direct, essentially agronomic approach. Over time, equipment and methods developed for the efficient establishment of crop monocultures on uniform, submissive environments were adapted and modified for planting a wide variety of seed types on highly variable and sometimes harsh environments (Keller 1979; Monsen and Stevens 2004; Young and Evans 1987). The greatest innovations have been associated with the collection, cleaning, and planting of native shrubs and forbs (Jorgensen and Stevens 2004). General principles and guidelines for big and black sagebrush restoration plantings that have stood the test of time and experience are discussed by Monsen and Stevens (2004) and Stevens and Monsen (2004b). First, competition from weedy species must be controlled. The seedbed should be firm, but not overly compacted. Effects of litter or mulching vary by species. Larger seeds should be sown at a depth of 1 to 2 cm (0.4 to 0.8 inches). Seeds of small seeded species, such as sagebrush, must be placed at or near the soil surface. It is generally best to plant seeds of slower growing forbs and shrubs separate from those of grasses. The optimal time for planting is from late fall to early winter, allowing for maximum use of winter and spring soil moisture and removal of seed dormancy. For seed collected from wild populations, climate and soils of the collection site should match those of the treatment site. Published seed transfer zones similar to those produced for trees species have been developed for a few key species (Mahalovich and McArthur 2004). More are needed. Although much progress has been made in recent decades, seeds of many species desirable for restoration plantings are either not available, or are available only in small quantities (McArthur and others 1987; McArthur and Young 1999; Roundy and others 1997). Too little is known of the biology of many of these species to plan for their efficient use. Additional research is needed to ascertain relationships between soil water and temperature and seed germination and seedling growth (Roundy 1994). The potential effects of present and future changes in biological and physical environments on community stability are not well understood and are in need of thoughtful attention. Finally, a commitment must be forged to manage preserved and restored big sagebrush landscapes for long-term sustainability or the degradation–restoration cycle will become a permanent feature of the big sagebrush landscape.

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