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June 4–7 2001, Boise, ID

Abstract

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Declines in habitat of greater sage-grouse and Gunnison sage-grouse across the western United States are related to degradation, loss, and fragmentation of sagebrush ecosystems resulting from development of agricultural lands, grazing practices, changes in wildfire regimes, increased spread of invasive species, gas and oil development, and other human impacts. These losses are focusing management efforts on passive and active approaches to maintaining and restoring sagebrush rangelands. This series of 14 papers summarizes current knowledge and research gaps in sagebrush taxonomy and ecology, seasonal sage-grouse habitat requirements, approaches to community and landscape restoration, and currently available plant materials and revegetation technology to provide a basis for designing and implementing effective management prescriptions.

Keywords: *Centrocercus urophasianus*, *Artemisia tridentata*, big sagebrush, native species, biodiversity, ecology, revegetation, rehabilitation, shrub steppe

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Sage-Grouse Habitat Restoration Symposium Proceedings

June 4–7 2001, Boise, ID

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Nancy Shaw
Mike Pellant

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Contents

Sage-Grouse Habitat Restoration Symposium

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Sage-grouse (greater sage-grouse [*Centrocercus urophasianus*] and Gunnison sage-grouse [*C. minimus*]) were once abundant over a range that approximated that of sagebrush (*Artemisia* spp.) in 16 Western States and three Canadian Provinces (Aldrich 1963; Connelly and others 2000; Johnsgard 1973). Although their specific requirements vary seasonally and over their life cycle, sage-grouse are almost completely reliant upon sagebrush habitats (Connelly and others 2000; Crawford and others 2004). Some populations are migratory and require ranges exceeding 1,300 km^2 (Wambolt and others 2002).

Sage-grouse are now among the 338 or more species whose populations are considered at risk for persistence (Wisdom and others 2003) and are dependent on sagebrush ecosystems. Connelly and Braun (1997) and Braun (1998) estimated that since European settlement, the distribution of sage-grouse has been reduced by 50 percent, and breeding populations have declined by 17 to 45 percent since 1985. Four petitions for subspecies or populations and three rangewide petitions have been filed to list the greater sage-grouse under the U.S. Endangered Species Act (Kritz 2004). The Gunnison sage-grouse is currently listed as a candidate species (U.S. Fish and Wildlife Service 2000).

Sagebrush communities earlier occupied about 63 million ha in Western North America (West 1983; West and Young 2000). Degradation, loss, and fragmentation of sagebrush habitat has occurred as a result of excessive livestock grazing, conversion to agricultural lands or seedings of introduced grasses, spread of invasive exotic plants and native conifers, alterations of fire regimes, oil and gas development, and other human-caused disturbances (Crawford and others 2004; Hann and others 1997; Knick 1999; Knick and others 2003; Noss and others 1995). Many areas have been degraded beyond the threshold where recovery is likely to occur naturally (Laycock 1991; West and Young 2000). As a result, some sagebrush ecosystems are among the most imperiled in North America (Noss and Peters 1995; Noss and others 1995). Conserving and protecting extant portions of sagebrush communities, altering management to encourage passive restoration of at-risk areas, and actively restoring degraded lands incapable of recovering without intervention presents a major challenge for Western land managers. This symposium was organized to provide an overview of science and technology addressing this issue.

Invited papers discussed sagebrush systematics, communities, ecology, and distribution (Goodrich, this proceedings; Rosentreter; this proceedings). Habitat requirements and movements of sage-grouse were described to indicate specific seasonal requirements and to demonstrate the need for planning restoration at the landscape level (Braun and Connelly, this proceedings; Wisdom and others, this proceedings). Other papers examined the principles of ecological restoration (Roundy, this proceedings) and native plant materials available for use on degraded sagebrush rangelands (Jones and Larson, this proceedings; Walker and Shaw, this proceedings). Additional papers described techniques for reestablishing sagebrush and understory species and managing woody vegetation (Shaw and others; Fairchild and others; Lambert; Lysne; Pellant; all in this proceedings).

Sixteen posters added depth to the range of topics discussed during the meeting. Two of these, included here, address wildlife-sagebrush relationships (Hampton, this proceedings; Wambolt, this proceedings). A 2-day field tour focused on successes and failures of local revegetation efforts and concluded with a demonstration of restoration equipment. Although the challenge of restoring millions of acres of sage-grouse habitat is formidable, the science and practical approaches presented during the symposium provided attendees with an overview of the status of the sagebrush ecosystem, sage-grouse habitat requirements, and the potential for restoring those habitats.

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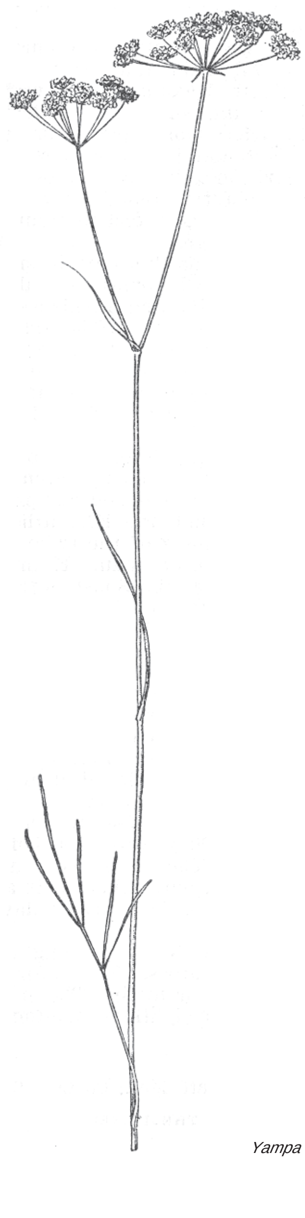

Yampa

Sagebrush Identification, Ecology, and Palatability Relative to Sage-Grouse

Roger Rosentreter

Abstract—Basic identification keys and comparison tables for 23 low and big sagebrush (*Artemisia*) taxa are presented. Differences in sagebrush ecology, soil temperature regimes, geographic range, palatability, mineralogy, and chemistry are discussed. Coumarin, a chemical produced in the glands of some *Artemisia* species, causes UV-light fluorescence of the leaves. Coumarin-containing taxa, such as mountain, xeric, subalpine big, subalpine early, black, and low sagebrush, each fluoresce a bright bluish-white color. These taxa are also the most palatable. A table of UV-light fluorescence of 20 sagebrush taxa in water solution is provided. How plant chemicals, such as coumarin and methacrolein and their seasonal variation, relate to palatability and animal preference is discussed in terms of sage-grouse. Restoration guidelines for some sagebrush taxa are also presented.

Keywords: Sagebrush, *Artemisia*, sage-grouse, palatability, preference, UV-light fluorescence

Introduction

The woody sagebrushes (*Artemisia*) are a major food source of and provide critical habitat for the declining sage-grouse (*Centrocercus urophasiamus*), icon of Western rangelands (Braun and others1977; Connelly and others 2000; Drut and others 1994). Improved identification of the types of sagebrush this species eats and uses for nesting and cover will help in its management. To the biologist and general public who are unfamiliar with the many different species and subspecies of sagebrush, this ecosystem may appear to be a bewildering array of variability. However, sagebrush communities are actually repetitive and easily identifiable (Beetle 1960; West 1988). Recognizing them is important because they are indicators of a given local ecosystem composed of specific vegetation types, soil depth, climate, topography, and wildlife species. Each type of sagebrush has different palatability and structural characteristics, which influences its particular values for wildlife (Sheehy and Winward 1981).

Woody sagebrush species have been of major interest and concern to land managers. Additional taxonomic research on western North America's woody *Artemisias* is needed. Collections made in fall when sagebrush taxa flower and are most distinctive are particularly valuable. This genus could include more genetic and morphological groups than are currently described. As more studies are conducted on the taxonomy of *Artemisia*, many of the subspecies and variety-level taxa will likely be raised to that of the species; new subspecies and varieties can be expected as well. The sagebrushes have been successful, in large part, due to their ability to exchange genetic material by hybridization and introgression (Hanks and others 1973; McArthur and others 1988), thus maintaining genotypic variation with sufficient plasticity to allow the development of ecotypes. This genetic variability may have also helped minimize disease and herbivory, which weaken and limit less genetically diverse species.

Why bother determining sagebrush and other vegetation to the species or even subspecies level? As former, and now deceased, University of Montana Professor Mel Morris used to say, "The better the plant is at indicating ecological condition or palatability, the more one should learn to identify that plant." Winward and Tisdale (1977) state that separation of big sagebrush into subspecies assists in the recognition of (1) habitat types (fig. 1), (2) production potential, (3) chemical content, and (4) palatability preference. When Nuttall described *Artemisia tridentata* in 1841, more than 20 present-day taxa were included. This broad species concept would not help us today in managing the 23 named sagebrush taxa that comprise sage-grouse habitat.

Palatability is defined as "plant characteristics or conditions that stimulate a selective response by animals" (Heady 1964). Webster defines the word "palatable" as pleasing to the taste. The term "preference" is reserved for selection by the animal and is essentially behavioral. Relative preference or relative palatability is a proportional choice among two or more foods. Items positively correlated with preference include (Heady 1964) (1) high protein content, (2) linolenic and butyric acids, (3) fat content, (4) sugar, and (5) phosphate and potash. Food items negatively correlated with preference include (1) high lignin content, (2) crude fiber, (3) tannins, and (4) nitrates (Heady 1964). In general, sagebrush species and populations that are more palatable to mule deer are also more palatable to sheep, cows, insects, and sage-grouse (Kelsey and Shafizadeh 1978; Sheehy and Winward 1981; Wambolt 2001; Wambolt and others 1991; Welch and Davis 1984; Welch and others 1983).

It is well documented that some sagebrush species are more palatable due to their chemical content (Morris and

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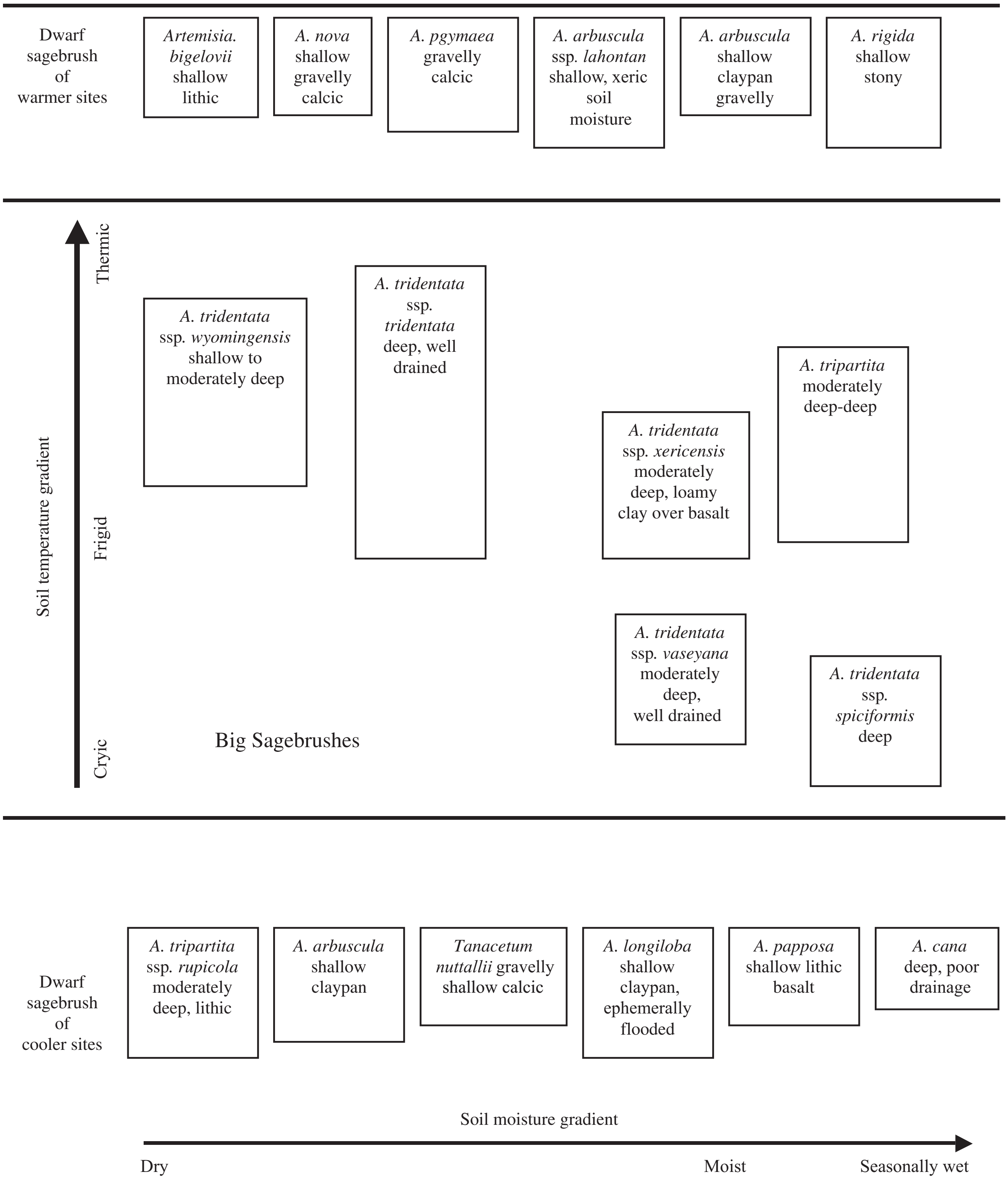


Figure 1—Environmental distribution of woody *Artemisia* taxa and the related *Tanacetum nuttallii* arranged by soil moisture, depth, texture, mineralogy, and soil temperature (modified from West 1988).

others 1976; Sheehy and Winward 1981; Wambolt 2001; Welch and others 1983). The difference in palatability is based on plant chemistry and the amount of volatile chemicals present in sagebrush leaf glands (Kelsey and others 1984; Striby and others 1987). Glands vary seasonally in the amount and concentration of chemicals they contain, with concentrations highest in spring and lowest in winter (Cedarleaf and others 1983; Kelsey and others 1984). This is due to the semievergreen nature of sagebrush and the presence of persistent leaves, produced in the spring, with glands full of volatile chemicals to discourage herbivory. In fall and early winter, gland cell walls and neck cells age and break open. These volatile chemicals are the distinctive fragrance one smells after a rain in the sagebrush desert. After releasing these chemicals, the sagebrush leaf becomes more digestible. This difference has been shown through "in vitro" digestibility of sagebrush leaves and alfalfa with the addition of sagebrush-specific volatile compounds (Striby and others 1987; Wambolt and others 1991). So, while some sagebrush species' high crude protein content encourages herbivory, others contain chemicals, such as volatiles, methacrolein, acetone, and 1-8 cineole, that discourage feeding (Kelsey and others 1982; Wambolt 1996; Wambolt and others 1991).

The amount of methacrolein in the three common subspecies of big sagebrush is consistent with their order of food preference (Wambolt and others 1991). It might be reasonable to assume that the chemicals that mule deer, sheep, and insects avoid will also be avoided by sage-grouse. The seasonal change in volatile oils also supports the premise for greater wildlife use of sagebrush foliage in the winter, as compared with spring and summer. However, in spite of what is known, palatability information for the different *Artemisia* species and subspecies is incomplete and somewhat based on assumption. This information gap is in part due to the difficulty of distinguishing common sagebrush species, as well as a lack of awareness of less common species that may have seasonal importance. I hope this paper helps improve this situation and stimulates research and discussion about the seasonal vegetative needs and preferences of the declining sage-grouse.

Methods

This paper summarizes current literature and the author's personal field and laboratory knowledge of woody sagebrush. Unfortunately, most of the sagebrush identification and ecological literature has been treated on a State-by-State rather than regional basis (Beetle 1960; Beetle and Johnson 1982; Morris and others 1976; Winward and Tisdale 1977). Broader treatments using detailed flower characteristics for species divisions have been developed (Hall and Clements 1923; McArthur 1979; Ward 1953); however, because they rely on the presence of the tiny sagebrush flowers, they are impractical for most of the calendar year, or for the biologist with no dissecting scope or herbarium reference material. Most plant characteristics referred to in this paper are visible with the naked eye or a 10x hand lens during any season. All woody shrub and subshrub sagebrush utilized by sage-grouse for food and habitat are included. The geographic scope includes the Great Basin sagebrush steppe and adjacent portions of the Great Plains and Colorado Plateau that have currently or historically supported sage-grouse (Connelly and others 2000). The 23 sagebrush species and subspecies treated are listed in table 1, arranged by their common and scientific names. The table includes one non-*Artemisia* taxon, *Tanacetum nuttallii* (chicken sage), a low-growing woody species that vegetatively resembles *Artemisia* and is utilized by sage-grouse.

Most palatability information does not come from sage-grouse use observations, since they are difficult to raise in captivity, but are based on observations of other wildlife species and on digestibility experiments such as those by Barnett and Crawford (1994), Kelsey and others (1982), Schwartz and others (1980), Sheehy and Winward (1981), Wambolt (2001), Wambolt and others (1991), and Yabann and others (1987). Much of the sagebrush chemistry literature is reported in highly technical chemistry-oriented journals and is in need of synthesis and interpretation for sage-grouse biologists and managers. Palatability of sagebrush and other plants depends on the individual animal or population of animals feeding on it. In addition to the chemical

Table 1—Twenty-three sagebrush taxa (species and subspecies) are listed in the order they are discussed. Nomenclature follows McArthur (1983), with additional, newly described subspecies following Goodrich and others (1985), Rosentreter and Kelsey (1991), and Winward and McArthur (1995). The author chose to exclude taxa that are either beyond the geographic scope of this paper or that can be accounted for at a higher taxonomic level.

Scientific name	Common name
Dwarf sagebrush	
Artemisia rigida	Stiff sagebrush
A. spinescens	Budsage
A. papposa	Fuzzy sage
A. tripartita ssp. *rupicola*	Wyoming threetip sagebrush
A. bigelovii	Bigelow sagebrush
A. pygmaea	Pygmy sagebrush
Tanacetum nuttallii	Chicken sage
Artemisia longiloba	Early sagebrush
A. arbuscula ssp. *longicaulis*	Lahontan sagebrush
A. nova	Black sagebrush
A. arbuscula	Low sagebrush
Tall sagebrush	
A. cana ssp. *cana*	Plains silver sagebrush
A. cana ssp. *bolanderi*	Bolander's silver sagebrush
A. cana ssp. *viscidula*	Mountain silver sagebrush
A. tripartita ssp. *tripartita*	Threetip sagebrush
A. tridentata ssp. *spiciformis*	Subalpine big sagebrush
A. tridentata ssp. *vaseyana*	Mountain big sagebrush
A. tridentata ssp. *vaseyana* var. *pauciflora*	Few-flowered mountain big sagebrush
A. tridentata ssp. *wyomingensis*	Wyoming big sagebrush
A. tridentata ssp. *tridentata*	Basin big sagebrush
A. *tridentata* ssp. *xericensis*	Xeric big sagebrush
Subshrub sagebrush	
Artemisia frigida	Fringed sage
A. pedatifida	Birdsfoot sage

content of food, learned behaviors may also dictate the food choices animals make. Availability of the plant is also a factor since hoofed animals may avoid, for example, a low sagebrush site that is sloped and rocky, while sage-grouse can readily use this type of terrain and the low sagebrush it supports.

Results and Discussion

Taxonomy and the UV-Light Test

Several keys and comparison tables for field and lab identification of woody *Artemisia* species are presented. The environmental distribution of these species is displayed by soil moisture, depth, texture, mineralogy, and soil temperature (fig. 1). Field identification can be done year round; however, sagebrush specimens collected in the fall are much easier to identify to species and subspecies. Ecological site knowledge and preferred soil mineralogy also help narrow down the possible taxa that might occur at a given location (fig. 1).

It is easier to distinguish the different species and subspecies of sagebrush using both morphological and chemical characteristics. Chemical analysis is a good tool to verify field determinations and can help eliminate identification problems due to morphological variation (Brunner 1972; Scholl and others1977; Stevens and McArthur 1974). A water extract of fresh or dried leaves of sagebrush can be viewed under a long-wave ultraviolet (UV) light. Prior to applying UV-light, several whole leaves are placed in a glass vial with 10 ml or more of water and shaken. Leaves must be from the same shrub rather than a composite sample since one leaf with positive fluorescence will yield a false positive result (Stevens and McArthur 1974). Table 2 contains the UV-light response for each taxon.

Glass vials must be thoroughly cleaned between samples to avoid contamination from previous tests. A voucher specimen of mountain big sagebrush should be the standard for sample comparison. A positive test produces a bluish-white fluorescence or glow, with the light held several inches from the vial of leaf/water solution. Testing is best done in a dark room or closet. Taxa cannot be distinguished solely by water extract color differences, but the test is useful for taxa likely to be confused based on morphology. This method can also be applied to digested sagebrush from sage-grouse scats in the field, using a portable UV-light and a dark-pigmented bag, or the sample can be returned to the lab.

Palatability and the UV-Light Test

A positive test with blue fluorescence indicates the presence of coumarin, a chemical compound in certain sagebrush species (Heywood and others 1977; McArthur and others 1988). These compounds, principally isocopoletin, scopoletin, and esculentin, are water soluble and fluoresce under ultraviolet light. The higher the compound concentration in a plant, the brighter the leaf/water fluorescence will be (Stevens and McArthur 1974). Coumarin appears to correlate with increased palatability in most sagebrush taxa. Palatability differences of individuals of the same taxa have even been shown to correlate with UV-light fluorescence intensity (Wambolt and others 1987, 1991; Welsh and others 1983).

Table 2—UV-light fluorescence of sagebrush taxa in water. Fluorescence intensity is indicated as: (1) intense—very bright bluish white that can be seen in a lighted room indoors; (2) strong—bright bluish white that can serve as a good standard for comparison in a dark location; (3) moderate—bluish white in a dark location; (4) light—very light blue and must be tested in complete darkness; and (5) colorless—no fluorescence.

Bluish white	Colorless
Early (intense)	Basin big sagebrush
Subalpine big (intense)	Wyoming big
Mountain big (strong)	Bud
Xeric big (strong)	Fuzzy
Low (strong)	Stiff
Bigelow (moderate)	Chicken sage
Lahontan (moderate)	Black “type b”
Black “type a” (moderate)	
Pygmy (moderate)	
Silver (light)	
Three-tip (light)	
Wyoming three-tip (light)	

Species	Palatability
Early	
Low	
Subalpine big	
Mountain big	
Wyoming big	
Xeric big	High
Lahontan	
Black “type a”	
Mountain silver	
Plains silver	
Three tip	
Chicken	
Fringed	
Bigelow	Moderate
Bud	
Pygmy	
Bolander silver	
Black “type b”	
Basin	
Wyoming three-tip	Low
Bird foot	
Fuzzy	
Stiff	

Figure 2—Relative palatability gradient of sagebrush for sage-grouse, from most to least palatable. Palatability is defined as “plant characteristics or conditions that stimulate a selective response by animals” (Heady 1964). Webster’s New World Dictionary (Guralnik 1972) defines palatable as “pleasing to the taste.” The term “preference” is reserved for selection by the animal and is essentially behavioral. Relative preference or relative palatability is a proportional choice among two or more foods.

Coumarin presence is a taxonomic indicator, separating several of the sagebrush taxa (Kelsey and others 1982); however, there are two exceptions to this. Wyoming big sagebrush has little to no fluorescence, but is still highly palatable. Bigelow sagebrush, which has a light-colored fluorescence, contains volatile monoterpenes that discourage herbivory (fig. 2). Hybrids of taxa that brightly fluoresce are intermediate in their response.

The UV-light test is an essential tool for sagebrush identification and palatability testing and should be used by sagebrush botanists and sage-grouse wildlife biologists. Palatability of sagebrush stands and individual plants can be ranked based on their fluorescence intensity, without even knowing the species.

Chemical Ecology

As previously mentioned, sagebrush chemicals are produced in glandular trichomes (Diettert 1938). These glands cover 21 to 35 percent of both sides of a leaf's surface and are hidden beneath a dense mat of hairs (Slone and Kelsey 1985). Glands contain coumarin as well as monoterpenes and sesquiterpene lactones, all of which influence a plant's palatability. Presence of these volatile monoterpenes contributes to the characteristic smell of sagebrush. Because these compounds are volatile, their concentration changes seasonally, with lower concentrations in fall and winter.

The sesquiterpene lactones are the pasty, black material found in sage-grouse scat, indicating that even sage-grouse cannot digest these tar-like lactones. These chemicals are probably deterrents to herbivory (Kelsey and others 1984; Welch and others 1983). In laboratory experiments, a 10-percent solution of lactones, extracted from big sagebrush leaves and placed in potato dextrose agar (PDA), completely inhibited growth of the common fungal mold, *Alternaria* sp. A 5-percent solution of these lactones inhibited the growth of *Alternaria* to as little as 25 to 61 percent of the control (Rosentreter 1984).

Sagebrush Identification Guidelines

In order to identify sagebrush with a key, a few simple rules must be followed. First, there are three types of leaves on most sagebrush species (Miller and Shultz 1987; Winward and Tisdale 1977). The "persistent" overwintering leaf is the representative leaf shape and size used in the keys (Diettert 1938). The "ephemeral" leaf is generally larger and often irregularly lobed. Ephemeral leaves are produced in spring and shed in the summer when there is drought stress. These odd-shaped leaves should be ignored, because they are fast growing and atypical. The third leaf type is on the flowering stalk. These leaves are often entire and lack the typical lobes and shape of persistent leaves.

Comparison tables (tables 2, 4, and 5) and a dichotomous key to all woody sagebrush species and subspecies are provided. Leaf characteristics are based on overwintering persistent leaves. Bell-shaped leaves have curved margins, strap-shaped leaves have straight margins, and cleft-shaped leaves are three parted. An "even crown" refers to flat-topped shrubs with seedstalks originating at the same height across a plant's crown. A 10-power (10x) hand lens can be used to examine leaf glands and hairs.

Individual Species Descriptions

Descriptions of each taxa are provided, including the preferred mineralogy, palatability, ecology, distribution, and management recommendations (figs. 1, 2). Dwarf sagebrush are discussed below as a group (also see table 3), followed by tall sagebrush and subshrub taxa (see table 4).

Dwarf Sagebrush

A. Stiff Sagebrush (*A. rigida*)—Stiff sagebrush occurs on very shallow skeletal basalt soils (Daubenmire 1982). Stiff sagebrush has also been called scabland sage due to the scabby, skeletal sites it prefers. Geographically, it grows in the Pacific Northwest portion of the United States and evades drought by being deciduous. Stiff sagebrush has brittle or stiff branches and grows from 12 to16 inches tall. Leaves are not reported to be palatable to any wildlife, but sheep will eat the flowering stalks in late summer and fall (Rosentreter 1992). Flower stalks are full of seeds that are relatively high in protein. Stiff sagebrush has a large seed (0.3 inch) that germinates quickly in 2 to 5 days (Rosentreter, unpublished data). Sites are ephemerally saturated, and contain a large diversity and cover of forbs when the sites are not degraded (Rosentreter 1992; Rosentreter and McCune 1992). Sandberg bluegrass (*Poa secunda*) is the most common grass in these habitats due to the shallow soils. Stiff sagebrush is not a resprouter as some authors have reported. It provides good spring and summer brood-rearing habitat for sage-grouse. Suitable sites of stiff sagebrush should be maintained and restored. The large seeds make restoration of stiff sagebrush feasible and easier than many other sagebrush species.

B. Budsage (*A. spinescens*)—Budsage grows on shallow, often saline soils at lower elevations, and is frequently mixed with salt desert shrub vegetation. It flowers in the spring (April to May). Budsage is geographically widespread, occurring from Montana to Arizona. It has palmately divided leaves that are deciduous. The leaves are fragrant and smell different than the other species. Budsage is considered to have low palatability, yet on degraded sites it will be heavily used in the early spring by antelope, sheep, and cattle. Budsage has a relatively large seed similar to stiff and fuzzy sage, two other spring-flowering, deciduous species. Budsage has not been used in restoration projects, but with its large seeds, it would appear to be feasible.

C. Fuzzy Sage (*A. papposa*)—Fuzzy sage occurs at mid elevations (>5,000 ft) on shallow soils similar to low sagebrush sites (Rosentreter 1992). However, fuzzy sage is always on basalt bedrock, often with very shallow to almost no soil over the skeletal basalt. Fuzzy sage is generally found on large, flat basalt tables that ephemerally flood at the landscape level. It occurs in Idaho and Oregon (Rosentreter 1992). It is deciduous and has relatively large red or yellow flowers in late spring. By late summer, plants are dried up and domestic sheep, horses, and many wildlife species will

Key to the Woody Sagebrush of the Great Basin and Adjacent Areas

1. Short or tall shrubs with woody twigs 2
1. Short subshrubs with nonwoody twigs .... Key A
   2. Dwarf shrubs, mature plants generally <24 inches tall .... Key B
   2. Tall to medium-sized shrubs, mature plants generally ≥24 inches tall .... Key C

Key A. Subshrubs with nonwoody twigs, woody at the base only

1. Leaf surface silvery, canescent .... Fringed sage, *Artemisia frigida*
1. Leaf greenish gray, pubescent 2
   2. Old flowering branches reduced to long spines, leaves dehiscent after spring, occurs at low elevations (also keyed as a dwarf shrub in Key B) .... Budsage, *A. spinescens*
   2. Plants without spines, leaves persistent, occurs at higher elevations in Wyoming and Montana .... Birdsfoot sage, *A. pedatifida*

Key B. Dwarf shrubs generally <24 inches tall

1. Plants deciduous, losing all their leaves in winter 2
1. Plants semievergreen, retaining some leaves through winter 4
   2. Leaves three lobed and linear .... Stiff sagebrush, *A. rigida*
   2. Leaves multilobed 3
3. Woody stems spiny, leaves light green .... Budsage, *A. spinescens*
3. Woody stems lacking spines, leaves palmately lobed, gray green and fuzzy with many hairs on the surface .... Fuzzy sage, *A. papposa*
   4. Persistent leaves deeply cleft up to 1.5 inches, grows on shallow soils on ridges at high elevations (7,500 to 9,000 ft) .... Wyoming threetip sagebrush, *A. tripartita* ssp. *rupicola*
   4. Persistent leaves shallow lobed 5
5. Pointed lobe tips, shallow lobes, and sharply three-toothed leaves .... Bigelow sagebrush, *A. bigelovii*
5. Rounded lobe tips 6
   6. Persistent leaves multilobed (>3 lobes), restricted to calcareous gravelly soil in Utah .... Pygmy sagebrush, *A. pygmaea*
   6. Persistent leaves three lobed 7
7. Mature plants <4 inches tall, large flowered, growing only on windswept calcareous gravel ridges in Idaho and Montana Chicken sage, *Tanacetum nuttallii*
7. Mature plants >4 inches tall, smaller flowers, growing on various habitats 8
   8. Early-maturing plants, flower in early summer; seed set by late August, often layering, leaves broadly cuneate with relatively well-developed lobes, large seeds (4 times the size of low sagebrush seeds) .... Early sagebrush, *A. longiloba*
   8. Late-maturing plants, flower in late summer or fall with seed set in October or November 9
9. Plant crown flat topped (even), flower stalks long and prominent, mostly above the plant, middle leaf lobe barely fits between the outside lobes .... Lahontan sagebrush, *A. arbuscula* ssp. *longicaulis*
9. Plant crown irregular not flat topped (uneven), flower stalks shorter and irregularly oriented, scattered throughout the crown, leaves smaller with lobes of similar size 10
   10. Leaves with numerous glands on the surface best visible at 10x, leaves sticky to touch, when mashed between fingers, yields a glossy green to a black color, old flower stalks brown and persistent, flower heads with 3–5 florets .... Black sagebrush, *A. nova*
   10. Leaves lack glands visible at 10x, gray green to silver-colored leaves, old flower stalks tan and nonpersistent, flower heads with 5–11 florets .... Low sagebrush, *A. arbuscula*

Key C. Tall to medium sized shrubs (generally ≥24 inches tall)

1. Persistent leaves linear, layering .... Silver sagebrush, *A. cana*
1. Persistent leaves three lobed, layering absent or rare 2
   2. Persistent leaves deeply cleft .... Three-tipped sagebrush, *A. tripartita* ssp. *tripartita*
   2. Persistent leaves shallowly lobed 3
3. Leaves large, 1.5 to 2.5 inches long and up to 0.75 inch wide, leaves dark green, broadly cuneate, often layering, only found at high elevations .... Subalpine sagebrush, *A. tridentata* ssp. *spiciformis*

Key to the Woody Sagebrush of the Great Basin and Adjacent Areas (con.)

3. Leaves smaller, gray green, not layering 4
    4. Plant crown flat topped (even), flower stalks long and prominent, mostly above the plant, leaf margins tapered, leaves widest just below the lobes, leaves in water fluoresce bright bluish white under UV light .... Mountain big sagebrush, *A. tridentata* ssp. *vaseyana*
    A. Persistent leaves widest at base of lobes. Inflorescence a spike or raceme with relatively few heads, plants occasionally layering .... *A. t.* var. *vaseyana*
    B. Persistent leaves widest slightly below the base of the lobes. Inflorescence a panicle with numerous heads, plants do not layer .... *A. t.* var. *pauciflora*
    4. Plant crown irregular (uneven), not flat topped, flower stalks smaller and irregularly oriented, scattered throughout the crown 5
5. Mature shrubs short, <3 ft tall, leaves bell shaped, grows at lower elevations, plant is U shaped Wyoming big sagebrush, *A. tridentata* ssp. *wyomingensis*
5. Mature shrubs taller, generally >3 ft tall, leaves not bell-shaped, grows at low to high elevations, plant is Yshaped with a central stalk, rather than U shaped or with cupped growth form of many other species 6
    6. Leaf margins straight, leaves long and strap shaped, leaves in water do not fluoresce, prefers deep well-drained soils, widespread geographically and ecologically .... Basin big sagebrush, *A. tridentata* ssp. *tridentata*
    6. Leaf margins tapered, leaves widest just below the lobes, leaves fluoresce bright blue under UV light, grows only in loamy clay soils in western Idaho .... Xeric big sagebrush, *A. tridentata* ssp. *xericensis*

eat the mature flower stalks (Rosentreter 1992). Its leaves are generally not palatable and are avoided except by horses (personal observation by the author in Idaho and Oregon).

D. Wyoming Threetip Sagebrush (*A. tripartita* ssp. *rupicola*)—Wyoming threetip sagebrush is a dwarf shrub rarely more than 7 inches tall, with relatively long (1–1.2 inch), deeply cleft leaves with narrow (1-mm wide) linear lobes. It occurs only in cold sites at high elevations greater than 7,200 ft, east of the Continental Divide in Wyoming and Montana. It is chemically similar to tall threetip sagebrush (*A. tripartita* ssp. *tripartita*) and is not very palatable. It will resprout weakly following physical disturbance or fire, unlike the tall and more common threetip sagebrush that readily resprouts. Because of their high elevation, most Wyoming threetip sagebrush sites are not heavily impacted by livestock, but the shallow rocky soils along ridgelines can be impacted by off-highway trail proliferation.

E. Bigelow Sagebrush (*A. bigelovii*)—Bigelow sagebrush can be confused with both low and Wyoming big sagebrush; however, Bigelow sagebrush leaves are more shallowly lobed and sharply pointed. The pointed leaf tips make identification of this species easy, as long as biologists and managers are aware of its potential presence. It occurs on arid and mesic calcareous soils and on highly decomposed granite. It grows throughout the Southwest from California to west Texas and north to northwest Colorado. Bigelow sagebrush is one of the sagebrush taxa that fluoresces, but is not considered highly palatable (silver sagebrush is another). Increased awareness of Bigelow sagebrush by the wildlife community, particularly in Colorado, may provide additional information on its palatability in the future.

F. Pygmy Sagebrush (*A. pygmaea*)—Pygmy sagebrush grows on dry alluvial fans at elevations from 5,000-7,000 ft. It occurs in shallow soils less than 20 inches deep with a gravelly soil surface texture and is often mixed with black sagebrush. It is found only in Utah, Nevada, and northern Arizona. Pygmy sagebrush has moderate palatability and may be utilized by wildlife in the winter, due to its availability at moderately low-elevation sites.

G. Chicken Sage (*Tanacetum nuttallii*)—Taxonomically, chicken sage has been treated as either *Tanacetum nuttallii* or *Sphaeromeria nuttalli.* Both of these genera are closely related to the genus *Artemisia*. Chicken sage grows on windswept benches and large flat areas on very shallow, calcareous gravels in Idaho, Montana, and Wyoming. Chicken sage looks like a diminutive low sagebrush, but it has smaller leaves and flowers in early summer. Flower heads are relatively large with a small pappus forming around the achene. Due to the presence of the small scalelike pappus, it has not been classified as an *Artemisia*, despite its other similar morphological characteristics. Chicken sage is woody and has three-lobed leaves like many *Artemisia*s. Palatability of this species is moderate, based on its use by antelope (Brent 1976; Thomas and Rosentreter 1992). Sage-grouse are found where this species is common, but it is unknown if they utilize it for food. Its growth form is low spreading to almost creeping, and its branches are less than 4 inches tall; thus, it does not provide structural or hiding cover for sage-grouse. Brent (1976) recorded antelope spending large amounts of time in windswept, normally snow-free chicken sage sites. This suggests their availability for sage-grouse in winter as well.

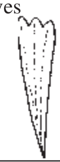
H. Early Sagebrush (*A. longiloba*)—Early sagebrush grows on shallow, ephemerally flooded soils, often with a claypan or skeletal rock layer near the surface (Robertson and others 1966). It is frequently found in low-drainage

Table 3—Morphological and chemical characteristics that distinguish the low sagebrushes.

Species or subspecies	Crown	Leaf shape and color	Leaf margin[a]	Plant architecture[b]	Layering	Preferred soil mineralogy and soil temperature	Color of sage/water solution under UV light[c]	Flowering begins
Low sagebrush	Uneven-even 	Gray 	Slightly bell shaped	Dwarf spreading, U shaped	No	Shallow claypan, granitic or basalt; aridic-cryic	Moderately bluish white	August
Lahontan sagebrush	Even, due to the long flower stalks 	Gray, larger leaf than low sage 	Elongated bell shape, ephemeral leaves often very large	U shaped, often taller than the often low sages	No	Shallow clay soil; aridic-mesic	Strongly bluish white	August
Black sagebrush	Uneven-even 	Dark green 	Bell shaped	Dwarf, spreading, U shaped	No	Calcareous; shallow, aridic-frigid	Moderately bluish white; some colorless populations exist	August
Early sagebrush	Even 	Dark gray green 	Broadly bell shaped with an obviously cleft central lobe	Dwarf, spreading, U shaped	Yes	Shallow claypan, often in ephemerally flooded but not always alkaline areas; frigid-cryic	Intense bluish white	June-July

[a] Leaf margin refers to the overall shape of the leaf, the curvature of its margin, and how wide it is at its apex.
[b] Plant architecture refers to a species' basal branching pattern—multiple stems bestow an overall "U shape," and one to few stems produce a wider appearance at the top or a "Y shape."
[c] Consider all taxonomic characteristics in combination when making a determination. A long-wave UV light should be used to test the sagebrush leaf/water solution in a dark location.

Table 4—Morphological and chemical characteristics that distinguish the big sagebrushes.

Species or subspecies	Crown	Leaf shape	Leaf margin[a]	Plant architecture[b]	Layering	Preferred soil mineralogy and soil temperature	Color of sage/water solution under UV light[c]	Flowering begins
Basin big sagebrush	Uneven 	Long and narrow 	Straight	Y shaped, erect	No	Deep, well drained; aridic-mesic	Colorless	Late August
Mountain big sagebrush	Even 	Broadly cuneate 	Tapered	U shaped, basal branching	No	Well drained, frigid-cryic	Strong bluish white	July, (<6 flowers/ inflorescense)
Wyoming big sagebrush	Uneven 	Belled 	Bell shaped	U shaped	No	Shallow to moderately deep soil, aridic	Colorless	August
Xeric big sagebrush	Uneven 	Broadly cuneate 	Tapered	Y shaped, radiate	No	Loamy clay over basalt, mesic-xeric	Strong bluish white	Late August
Subalpine big sagebrush	Even 	Large leaves 	Tapered to elongated bell shape	Spreading, multiple stemmed	Yes	Deep, noncalcareous, often a clay layer; frigid	Intense bluish white	July, (>6 flowers/ inflorescense)

[a] Leaf margin refers to the overall shape of the leaf, the curvature of its margin, and how wide it is at its apex.
[b] Plant architecture refers to a species' basal branching pattern—multiple stems bestow an overall "U shape," and one to few stems produce a wider appearance at the top or a "Y shape."
[c] Consider all taxonomic characteristics in combination when making a determination. A long-wave UV light should be used to test the sagebrush leaf/water solution in a dark location.

areas of flats, plateaus, or tables. Early sagebrush is a prolific seed producer and could be used for restoration in appropriate, shallow soil sites (Beetle and Johnson 1982; Monsen and Shaw 1986). It layers and can resprout after cool fires. Early sagebrush is one of the most valuable taxa for sage-grouse, and many of the largest leks in Idaho are in areas dominated by this species (Camas Prairie, south of Fairfield, ID) (fig. 1) (Robertson and others 1966). It flowers very early in the summer, in contrast to other low-statured species. Early sagebrush has been confused with low-growing Wyoming big sagebrush because of its broadly cuneate 3-lobed leaves, and with low sage because of its dwarf size. Early sagebrush is palatable to sheep and, historically, stands were commonly used as lambing areas (Beetle and Johnson 1982). These areas should be monitored to prevent heavy spring grazing by domestic livestock. Early sagebrush has also been referred to as "alkali sagebrush," although sites may or may not be alkaline (Robertson and others 1966).

I. Lahontan Sagebrush (*A. arbuscula* ssp. *longicaulis*)—Lahontan sagebrush is a type of low sagebrush that grows on shallow clay soils formed on the shore of Pleistocene Lake Lahontan. It grows in northwest Nevada and adjacent California and Oregon at elevations from 3,400-6,600 ft. It differs from low sagebrush chemically and by its longer floral stalks and larger leaves. Lahontan sagebrush occurs on soils similar to low sagebrush, but in areas that receive less precipitation (5-12 inches). It is moderately to highly palatable (Winward and McArthur 1995).

J. Black Sagebrush (*A. nova*)—There appears to be at least two chemical races of black sagebrush in the West (Kelsey 2002, personal communication; McArthur and Plummer 1978). One race with gray leaves is highly palatable, while the green-leafed race has low palatability (fig. 2) (McArthur and Plummer 1978). This latter form does not fluoresce under UV light. Additional studies are needed to determine the geographic ranges and correlation with physical characteristics for these two races. Black sagebrush has been greatly reduced or eliminated on some ranges where sheep graze in winter (Clary 1986). The best feature to identify this species is its flower stalks. The stiff, erect stalks dry to brown and persist through the following year. Most populations have leaf glands visible with a 10x hand lens (Kelsey and Shafizadeh 1980). Black sagebrush grows well on very shallow stony soils, often on windswept slopes and ridges at mid- to high elevations where annual precipitation is more than 10 inches (Behan and Welch 1985). It prefers calcareous or well-decomposed granitic soils that seem to mimic calcareous sites due to weathering of calcium feldspars. Black sagebrush is a widespread species, second only in its geographical distribution to basin big sagebrush.

K. Low sagebrush (*A. arbuscula*)—Low sagebrush grows on shallow soils with a restrictive layer of bedrock or clay pan. This species is usually found where annual precipitation is greater than 12 inches. Parent material is noncalcareous. Low sagebrush is one of the most palatable sagebrushes for sage-grouse. It is a wide-ranging species, found throughout the Great Basin. Black, early, Bigelow and Lahontan sagebrush, and chicken sage are often misidentified as low sagebrush.

Tall Sagebrush

L. Silver Sagebrush (*A. cana*)—Silver sagebrush is a tall shrub with three subspecies that grow in distinctly different habitats. All three subspecies are root-sprouters and layer vegetatively. The three subspecies are distinguished as:

1. Mature plants 3 to 5 ft tall, leaves mostly >0.8 inch long and strongly pubescent, a plant of arid riparian drainages
   .... Plains silver sagebrush, *A. cana* ssp. *cana*
1. Mature plants <40 inches tall, leaves mostly <0.8 inch long 2
   2. Leaves pubescent and silver gray, plant of playas (internally drained basins)
      .... Bolander silver sagebrush, *A. cana* ssp. *bolanderi*
   2. Leaves sparsely pubescent and dark green, plant of high elevations
      .... Mountain silver sagebrush, A. *cana* ssp. *viscidula*

Mountain and plains silver sagebrush are considered highly palatable (Wambolt 2001), while Bolander silver sagebrush is only moderately palatable. The former two species generally grow where they receive additional moisture from the surrounding vegetation. All three subspecies are within the range of sage-grouse. Plains silver sagebrush is often the only *Artemisia* used by grouse on the flat plains of central and eastern Montana.

M. Threetip Sagebrush (*A. tripartita* ssp. *tripartita*)—Threetip sagebrush is a fairly tall, erect shrub (4 to 6 ft). It grows on deep, well-drained soils, often mixed with basin or mountain big sagebrush. It will seldom layer without disturbance, but will vigorously stump sprout and layer after burning. It is considered highly palatable to wildlife (Wambolt 2001); however, there is high seasonal variation in its utilization. Livestock, including sheep, appear to avoid utilization of this species. Beware of control or prescribed burning in threetip sagebrush habitat, as it can increase well beyond the site's preburn density. It is common in Washington, Idaho, Montana, Wyoming, Utah, and Colorado.

N. Subalpine Big Sagebrush (*A. tridentata* ssp. *spiciformis*)—Subalpine big sagebrush grows on deep, cryic soils and is highly palatable. Sage-grouse reportedly use this species; however, it probably becomes unavailable in late winter due to snow cover. It frequently grows where large snowdrifts form, unlike dwarf sagebrush types that grow in windswept areas. Subalpine big sagebrush can occur on ridgelines, similar to some of the dwarf and low-stature sagebrushes. These ridgelines are frequently used by sage-grouse. Chemically, subalpine big sagebrush appears to be a choice food for sage-grouse and other wildlife species. It layers vegetatively and resprouts following defoliation from heavy snow. It occurs in Idaho, Wyoming, Montana, Utah, and Colorado (Goodrich and others 1985; McArthur and Goodrich 1986).

O. Mountain Big Sagebrush (*A. tridentata* ssp. *vaseyana*)—This includes varieties *pauciflora* and *vaseyana*. Mountain big sagebrush is a flat-topped shrub that grows to 3 ft tall (Tisdale and Hironaka 1981). It has a U-shaped crown and is found on moderate to deep, well-drained, frigid soils, generally above 5,000 ft. Mountain big sagebrush can grow as low as 3,000 ft, and when it does, soils are typically very well drained. It is highly palatable to most wildlife; however, limited access in the winter and the chemical content in spring and summer may discourage herbivory (Kelsey and Shafizadeh 1978; Kelsey and others 1984). Mountain big sagebrush is a major food source for sage-grouse in the winter months. Sage-grouse scats collected from Wyoming big sagebrush-dominated sites in Idaho and Colorado tested positive for UV light, indicating that grouse were feeding nearby on UV positive sagebrush (mountain big sagebrush or low sagebrush) (Vasquez 2002). In the Gunnison Basin of Colorado, sage-grouse utilize a hybrid of *A. t.* ssp. *vaseyana* and *A. t.* ssp. *wyomingensis* (Vasquez 2002).

Compared to other sagebrush taxa, mountain big sagebrush has a greater potential to increase its density due to its general ecology and the higher moisture its habitat receives. Stands can become so dense they are difficult for humans to walk through. In much of the West, heavy livestock use, both historic and current, has reduced forb, perennial grass, and biological soil crust components, allowing sagebrush and exotic annual grasses to become dense (Billings 1994; Rosentreter and Eldridge 2002). Mechanical control, burning, or seeding followed by rest from grazing, is necessary in many areas to restore the vegetative and structural diversity needed for optimal wildlife habitat. "Hobble Creek" mountain big sagebrush, a highly palatable cultivar of *A. t.* ssp. *vaseyana*, is recommended for restoration projects with the goal of improving wildlife winter range (Welch and others 1990).

P. Wyoming Big Sagebrush (*A. tridentata* ssp. *wyomingensis*)—Wyoming big sagebrush is a medium sized shrub from 1 to 3 ft tall. It branches from the base, giving it a U-shaped architecture. Wyoming big sagebrush grows at warmer, lower elevations and is more available as forage in winter and early spring (Wambolt 1998). It occurs at sites receiving from 8 to 12 inches of precipitation. This species is generally palatable, though its palatability is highly variable. Many Wyoming big sagebrush sites have been severely degraded and converted to exotic annual grasslands; thus, in harsh winters, they are no longer available for sage-grouse use (Hilty and others 2003).

Nondegraded, lightly grazed Wyoming big sagebrush sites have a high percent cover of biological soil crusts and low percentage of cheatgrass cover (Kaltenecker and others 1999; Rosentreter 1986; Rosentreter and Eldridge 2002). Due to their susceptibility to invasion and domination by cheatgrass and other exotic annuals, use of fire to manage them must be approached with caution. Wyoming big sagebrush sites should be managed for retention of the biological soil crust component. Late fall, winter, and early spring is the most appropriate season of use for this low-elevation vegetation type. Four to 6 weeks of moist soil conditions in late spring facilitates regrowth of biological soil crusts disturbed by trampling (Memmott and others 1998; Rosentreter and Eldridge 2002). "Gordon Creek" Wyoming big sagebrush, a highly palatable cultivar, is recommended for restoration projects aimed at improving wildlife winter range (Welch and others 1992).

Q. Basin Big Sagebrush (*A. tridentata* ssp. *tridentata*)—Basin big sagebrush is the least palatable of the big sagebrushes (Wambolt 1998), though it is chemically and genetically (2n = 18 or 36) highly variable. It is considered of low palatability relative to other sagebrush taxa (fig. 2), and it is also the tallest. Its architecture is somewhat single trunked (tree like) or Y shaped, with lateral branches diverging from the main stem at a different angle than those of either Wyoming or mountain big sagebrush. This prolific seed producer grows on deep, well-drained soil (Daubenmire 1975). The extra moisture runoff from roads can create artificial sites for this subspecies, even in soils normally occupied by Wyoming big sagebrush. Large areas dominated by Wyoming big sagebrush will frequently have basin big sagebrush adjacent to the road ditch. Seed of the more prolific, larger statured basin big sagebrush is often harvested along with seed of Wyoming big sagebrush. Dalzell (2004) found that 8 percent of shrubs in Wyoming big sagebrush seedings in southern Idaho were basin big sagebrush. Basin big sagebrush leaves have rarely been identified in sage-grouse scats (Rosentreter 2001, unpublished data; Vasquez 2002). However, even small dense stands of this shrub can provide good nesting habitat for sage-grouse in Colorado's Gunnison Basin.

R. Xeric Big Sagebrush (*Artemisia tridentata* ssp. *xericensis*)—Xeric big sagebrush is a tall shrub (>3 ft) with Y-shaped architecture similar to basin big sagebrush. However, its chemistry, leaf shape, and palatability are most similar to mountain big sagebrush. At lower elevations (2,500–4,500 ft), this Idaho subspecies is restricted to heavy clay-loam and drier, xeric soils than mountain big sagebrush. In Idaho, mountain big sagebrush grows between 4,000 and 9,500 ft, in moister Udic soils. Xeric big sagebrush is heavily utilized in winter by mule deer and, based on its chemistry (high crude protein) (Rosentreter and Kelsey 1991), is likely preferred by sage-grouse. It can increase in density similar to mountain big sagebrush, with heavy spring, summer, or early fall cattle grazing.

Subshrub Sagebrush

S. Fringed Sage (*A. frigida*)—Fringed sage is a small subshrub, woody only at the base. It is the most widespread species treated in this paper, extending into other North American and Asian biomes, such as alpine meadows, the Great Plains, and mountain meadows. It was described from Siberia before being identified in North America. Fringed sage occurs in a variety of soil types and depths, but prefers shallow soils with frigid soil temperatures (Morris and others 1976). Some sites are windswept and are readily available to wildlife in the winter. Fringed sage is moderately palatable. In the Gunnison Basin, sage-grouse have been observed eating fringed sage seedlings in early spring (Young 2001).

T. Birdsfoot Sage (*A. pedatifida*)—Birdsfoot sage is a small subshrub, weakly woody at the base. It occurs in dry shallow soils at high elevations with frigid soil temperatures in Wyoming, Montana, and Idaho. This subshrub is most commonly found in montane grasslands and on windswept sites. It is 1 to 5 inches tall with finely canescent, basal leaves. Leaves are once or twice ternately divided into linear divisions with finely white-tomentulose hairs. Flowers are brownish. Birdsfoot sage has low palatability.

Conclusion

Coumarin-containing taxa such as mountain, xeric, subalpine big, subalpine early, black, and low sagebrush all fluoresce a bright bluish-white color. These taxa are also the most palatable. Plant chemicals such as coumarin and methacrolein and their seasonal variation affect shrub palatability and animal, including sage-grouse, preference. Because sagebrush species also differ vastly in their structural characteristics and habitat requirements, knowledge of *Artemisia* ecology will enhance our ability to improve and manage habitat for sage-grouse.

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Arrowleaf balsamroot

Classification and Capabilities of Woody Sagebrush Communities of Western North America With Emphasis on Sage-Grouse Habitat

Sherel Goodrich

Abstract—This paper deals with diversity, classification, and capabilities of different sagebrush (*Artemisia* spp.) communities. Capabilities of sagebrush communities in terms of production, plant diversity, potential for ground cover and sage-grouse (*Centrocercus urophasianus*) habitat are discussed. Reaction to fire and relationships with introduced annuals are also discussed for some communities.

Introduction

There are about 30 taxa of woody and semiwoody sagebrush (*Artemisia* spp.) in western North America. Many of these sagebrush taxa are community dominants that are highly specialized. They are high-resolution indicators of climate, geomorphic settings, soils, elevations, and other features of the environment. Ecotones between stands of different species of sagebrush are often narrow and even sharply abrupt. Ecotones between taxa of the same species are often wider, but these are also sometimes quite narrow.

Sagebrush taxa are often more site specific than many associated indicator species. Bluebunch wheatgrass (*Elymus spicatus*) and Idaho fescue (*Festuca ovina* var. *ingrata*) provide vivid examples. These are useful indicator plants in the sagebrush ecosystem. However, they are generalists compared to some of the sagebrush taxa. Respectively, these grasses are listed as indicator species in 9 and 11 sagebrush types, and they extend beyond the amplitude of sagebrush into grassland (Mueggler and Stewart 1980; Tisdale 1986), mountain brush, and some forested communities.

Hybrids and hybrid zones of sagebrush often indicate intermediate environments (Freeman and others 1999; Graham and others 1999; McArthur and Sanderson 1999b). Hybrid zones coupled with a propensity for development of polyploidy further increase the complexity of the sagebrush group (McArthur and Sanderson 1999a; McArthur and others 1981). With many highly specialized taxa, hybrid zones, and polyploidy, the sagebrush complex is highly diverse.

Classification of vegetation types inherently includes information useful in understanding capabilities of the land, and many classification works include management implications based on estimates or measurements of capabilities. Most classification of sagebrush vegetation types has been at local scales and based on different methods of data collection and ordination. Although a comprehensive and uniform classification would be useful, the work of the past provides considerable information about the sagebrush ecosystem. Classification works were relied upon heavily in preparing this paper. Units of sagebrush classification found in the literature include community types, plant associations, habitat types, and ecological units. Distinction of classification types used by different workers is not made in this paper. All methods of classification seem to be useful in helping to define the variability and capability of sagebrush systems. Recognizing differences in capabilities of sagebrush communities is important for restoration projects, management for desired condition, and other aspects of land management.

Evaluations of values for sage-grouse (*Centrocercus urophasianus*) habitat made in this paper are based on indicated capabilities of various sagebrush types in comparison with tables 1, 2, and 3 in Connelly and others (2000). In general Connelly and others (2000) indicated sage-grouse nest sites are associated with sagebrush with heights of 29 to 80 cm and crown cover of 15 to 38 percent. Characteristics needed for productive sage-grouse winter habitat include sagebrush with heights of 25 to 35 cm and crown cover of 10 to 30 percent. Characteristics needed for productive breeding and brood-rearing habitat include sagebrush with heights of 30 to 80 cm and crown cover of 15 to 25 percent for breeding habitat and 10 to 25 percent for brood-rearing habitat.

The majority of this paper is focused on Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis*), Vasey big sagebrush, and mountain big sagebrush (*A. t.* ssp. *vaseyana*) communities. These taxa are selected for greater discussion because of their comparatively wide distribution, the large area they cover, and their greater importance to sage-grouse than many of the other taxa.

In this paper common names of plants are not used in reference to classification. However, they are used in the text, and a list of scientific names including authors and corresponding common names is provided in tables 1 and 2. Symbols of plant taxa as listed by athe U.S. Department of

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Agriculture (2001) are also included in tables 1 and 2. Only indicator species of vegetation types are listed. Expanded lists are found in some of the works listed in the references section. For the most part, plant nomenclature in this paper follows that of Shultz (1986) for *Artemisia* and Welsh and others (1993) for other plant species.

References to production are generally based on standing crop (a measurement at one time of the year). Thus, plants that grow in spring and early summer then dry and wither away are often not well represented in production values. As used in this paper, ground cover includes the percentage of the ground surface covered by live plants, litter or plant residue, and rock. Bare soil and pavement of less than about 2 cm diameter are generally not considered ground cover. For the cited references, however, definitions of bare ground may vary among authors.

Artemisia arbuscula

This plant of western North America has been treated as a subspecies and as a variety of *A. tridentata*. However, it has been accepted at the species level by many Intermountain authors. Shultz (1986) recognized three subspecies in this complex and provided distribution maps for each. Winward and McArthur (1995) described a fourth subspecies. These are listed below. All plants of the complex are low, nonsprouting, wintergreen shrubs.

Artemisia arbuscula ssp. *arbuscula* Low Sagebrush

This plant is the most widespread taxon of the complex. It ranges from Washington to Wyoming and south to California and Utah.

Classification

Artemisia arbuscula ssp. *arbuscula*/Bunchgrass (*Elymus-Festuca*) (Hall 1973)

Artemisia arbuscula ssp. *arbuscula*/*Elymus spicatus* (Hironaka and others 1983: Jensen 1989; Mueggler and Stewart 1980; Nelson and Jensen 1987; Tew 1988; Zamora and Tueller 1973)

Artemisia arbuscula ssp. *arbuscula*/*Festuca idahoensis* (Hironaka and others 1983; Jensen 1989; Mueggler and

Table 1—Scientific name, codes, and common names of sagebrush type indicator plants: woody plants.

Scientific name	Code	Common name
Artemisia arbuscula Nutt. ssp. *arbuscula*	ARARA	Low sagebrush
Artemisia arbuscula ssp. *longicaulis* Winward & McArthur	ARARL3	Lahontan sagebrush
Artemisia arbuscula ssp. *thermopola* Beetle	ARART	Cleftleaf sagebrush, hotsprings sagebrush
Artemisia bigelovii Gray	ARBI3	Bigelow sagebrush
Artemisia cana Pursh ssp. *cana*	ARCAC5	Plains silver sagebrush
Artemisia cana ssp. *bolanderi* (A. Gray) Ward	ARCAB3	Sierra silver sagebrush
Artemisia cana ssp. *viscidula* (Osterh.) Beetle	ARCAV2	Mountain silver sagebrush
Artemisia filifolia Torrey	ARFI2	Sand sagebrush
Artemisia frigida Willd.	ARFR4	Fringed sagebrush
Artemisia longiloba (Osterh.) Beetle	ARLO9	Alkali sagebrush, early sagebrush
Artemisia nova A. Nels. var. *nova*	ARNO4	Black sagebrush
Artemisia nova var. *duchesnicola* Welsh & Goodrich	ARNOD	Red clay sagebrush
Artemisia papposa S. F. Blake & Cronquist	ARPA16	Owyhee sagebrush
Artemisia pedatifida Nutt.	ARPE6	Birdsfoot sagebrush
Artemisia porteri Cronq.	ARPO5	Porter sagebrush
Artemisia pygmaea A. Gray	ARPY2	Pygmy sagebrush
Artemisia rigida (Nutt.) A. Gray	ARRI2	Scabland sagebrush; stiff sagebrush
Artemisia rothrockii A. Gray	ARRO4	Rothrock sagebrush
Artemisia spinescens D. C. Eaton	ARSP5	Bud sagebrush, budsage
Artemisia tridentata ssp. *parishii* (A. Gray) Hall & Clements	ARTRP2	Parish big sagebrush
Artemisia tridentata ssp. *spiciformis* (Osterh.) Kartesz & Gandhi	ARTRS2	Spiked big sagebrush, subalpine big sagebrush
Artemisia tridentata ssp. *tridentata*	ARTRT	Basin big sagebrush
Artemisia tridentata ssp. *vaseyana* var. *pauciflora* Winward & Goodrich	ARTRP4	Mountain big sagebrush
Artemisia tridentata ssp. *vaseyana* (Rydb.) Beetle var. *vaseyana*	ARTRV	Vasey big sagebrush
Artemisia tridentata ssp. *wyomingensis* Beetle & Young	ARTRW8	Wyoming big sagebrush
Artemisia tridentata ssp. *xericensis* Rosentreter and Kelsey	ARTRX	Xeric big sagebrush
Artemisia tripartita Rydb. ssp. *tripartita*	ARTRT2	Threetip sagebrush
Artemisia tripartita ssp. *rupicola* Beetle	ARTRR4	Wyoming threetip sagebrush
Atriplex confertifolia (Torr. & Frem) Wats.	ATCO	Shadscale
Chrysothamnus nauseosus (Pallas) Britt.	CHNA2	Rubber rabbitbrush
Grayia spinosa (Hook.) Moq.	GRSP	Spiny hopsage
Purshia tridentata (Pursh) DC.	PUTR2	Bitterbrush
Sarcobatus vermiculatus (Hook.) Torr.	SAVE4	Greasewood
Symphoricarpos occidentalis Hook.	SYOC	Wolfberry
Symphoricarpos oreophilus Gray	SYOR	Mountain snowberry

Table 2—Scientific name, symbols, and common names of sagebrush type indicator plants: graminoids.

Scientific name	Symbol	Common name
Bouteloua gracilis (H.B.K.) Lag. ex Steudel	BOGR	Blue gramma
Bromus carinatus H. & A.	BRCA5	Mountain brome
Carex geyeri F. Boott	CAGE2	Elk sedge
Deschampsia cespitosa (L.) Beauv.	DECE	Tufted hairgrass
Elymus cinereus Scribn. & Merr.	ELCI2	Basin wildrye
Elymus elymoides (Raf.) Swezey	ELEL5	Bottlebrush squirreltail
Elymus lanceolatus (Scribn. & Sm.) Gould	ELLA3	Thickspike wheatgrass
Elymus salinus Jones	ELSA	Salina wildrye
Elymus smithii (Rydb.) Gould	ELWM3	Western wheatgrass
Elymus spicatus (Pursh) Gould	ELSP3	Bluebunch wheatgrass
Elymus trachycaulus (Link.) Gould ex Shinn.	ELTR7	Slender wheatgrass
Festuca ovina L. var. *ingrata* Hackel ex Beal (*Festuca idahoensis* Elmer)	FEOVI	Idaho fescue
Festuca ovina var. *rydbergii* St. Yves	FEOVR	Sheep fescue
Festuca scabrella Torr.	FESC	Rough fescue
Festuca thurberi Vasey	FETH	Thurber fescue
Koeleria macrantha (Ledeb.) Schultes	KOMA	June grass
Leucopoa kingii (Wats.) W. A. Weber	LEKI2	Spike fescue
Poa fendleriana (Stuedel) Vasey	POFE	Muttongrass
Poa pratensis L.	POPR	Kentucky bluegrass
Poa secunda Presl	POSE	Sandberg bluegrass
Sporobolus cryptandrus (Torr.) Gray	SPCR	Sand dropseed
Stipa comata Trin. & Rupr. var. *comata*	STCOC2	Needle-and-thread grass
Stipa comata var. *intermedia* Scribn. & Tweedy	STCOI	Mountain needle-and-thread grass
Stipa hymenoides R. & S.	STHY6	Indian ricegrass
Stipa lettermanii Vasey	STLE4	Letterman needlegrass
Stipa nelsonii Scribn.	STNE3	Columbia needlegrass
Stipa richardsonii Link	STRI2	Richardson needlegrass
Stipa thurberiana Piper	STTH2	Thurber needlegrass
Trisetum spicatum (L.) Richter	TRSP2	Spike trisetum

Stewart 1980; Nelson and Jensen 1987; Volland 1976; Zamora and Tueller 1973)

Artemisia arbuscula ssp. *arbuscula*/*Festuca idahoensis*/*Poa secunda* (Nelson and Jensen 1987)

Artemisia arbuscula ssp. *arbuscula*/*Stipa thurberiana* (Zamora and Tueller 1973)

Artemisia arbuscula ssp. *arbuscula*/*Purshia tridentata*/*Agropyron spicatum* (Zamora and Tueller 1973)

Of the plants recognized above as indicator species of classification units within the low sagebrush type, all likely have broader ecological amplitude than does low sagebrush. This taxon appears to be the most specialized of any of the associates listed above.

Habitat and Capabilities

Low sagebrush forms stands below the pinyon-juniper (*Pinus-Juniperus*) belt, within this belt, and well above it. Stands in the Toiyabe Range of Nevada extend up to 3,200 m elevation or higher (Goodrich 1981) and up to 3,900 m in the White Mountains of California (Mozingo 1987). Perhaps low sagebrush is less common below the pinyon-juniper belt than is black sagebrush (*A. nova*).

Low sagebrush is often an indicator of soils with clay subsurface horizons or bedrock within 8 to 33 cm of the surface (Fosberg 1963; Summerfield and Peterson 1971). This plant is sometimes found where restrictive soil horizons cause water to stand on the surface in late winter and spring, during which time the soils are poorly aerated (Hironaka 1963). The restrictive soil layers also tend to reduce water-holding capacity in summer. The soil features associated with low sagebrush indicate a habitat with potential for soil saturation in spring and drier conditions in summer than are typical for big sagebrush.

Sabinske and Knight (1978) found low sagebrush on gravelly soils without a restrictive horizon in Wyoming. In the mountain ranges of central Nevada, low sagebrush occupies shallow, rocky, well-drained soils as well as those with heavy clay horizons (Goodrich 1981).

Low sagebrush is highly competitive on sites where it is well adapted. It will reestablish from seed after disturbance, but it does not sprout. Comparatively low productivity and other site features indicate difficulty and low economic return for restoration and forage improvement projects in this type. Such projects should be planned and conducted carefully (Johnson 1987; Winward 1980). Low-elevation stands can be expected to be vulnerable to displacement by cheatgrass (*Bromus tectorum*). High-elevation stands are likely to be more resistant.

Annual production of between 392 to 728 kg/ha (350 to 650 lbs/acre) has been reported for low sagebrush communities (Hall 1973; Jensen 1989; Nelson and Jensen 1987; Tew 1988). Volland (1976) listed a mean of 200 kg/ha (179 lbs/acre) for sites in fair condition in the pumice zone of Oregon.

As implied by its common name, low sagebrush is short in stature. Height of low sagebrush is commonly 20 to 40 cm and less commonly from 10 to 50 cm (Cronquist 1994;

Harrington 1954; Welsh and others 1993). In four habitat types in Nevada, maximum height of low sagebrush is between 11 to 19 cm and crown cover is between 12 to 16 percent (Zamora and Tueller 1973).

Hall (1973) listed 13 percent surface rock, 10 percent erosion pavement, and 16 percent bare ground for a low sagebrush/bunchgrass type in Oregon and Washington. Tables in Tew (1988) indicate ground surface with an average of 21 percent bare soil, 42 percent vegetation and litter, and the remaining 37 percent rock and pavement. Volland (1976) listed 11 percent rock and 42 percent bare ground and pavement for a type in fair condition in the pumice zone of Oregon.

Low sagebrush sites of low and moderate productivity are indicated to lack the capability to meet the requirements of shrub height and percent crown cover for sage-grouse nesting habitat. Some of the more productive sites might provide moderate value nesting habitat. However, the crowns of low sagebrush are often in contact with the ground or close enough to the ground to obstruct sage-grouse nesting.

Low sagebrush sites dry rapidly in early summer. This indicates moderate-value brooding habitat in the spring and low value in summer and fall. Small stands of low productivity likely provide high-value strutting habitat especially where they are mixed with stands of big sagebrush (*Artemisia tridentata*). On windswept areas low sagebrush might provide high-value winter forage. However, snow can be expected to cover shrubs of this community on flat and concave areas before shrubs of big sagebrush are covered.

Artemisia arbuscula ssp. *longicaulis* Lahontan Sagebrush

This plant is known from western Nevada, southern Idaho, and eastern California where the old shorelines of Pleistocene Lake Lahontan are one of the centers of its current distribution (Winward and McArthur 1995).

Apparently this taxon has not been included in published vegetation type classifications. It is interspersed with salt desert shrub species including shadscale. It is also associated with Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*), black sagebrush, and low sagebrush. The most common understory species include bluebunch wheatgrass, Thurber needlegrass (*Stipa thurberiana*), desert needlegrass (*Stipa speciosa*), Indian ricegrass (*S. hymenoides*), bottlebrush squirreltail (*Elymus elymoides*), and Sandberg bluegrass (*Poa sandbergii*) (Winward and McArthur 1995). The value of Lahontan sagebrush to sage-grouse is likely similar to that of other taxa of the *A. arbuscula* complex.

Artemisia arbuscula ssp. *thermopola* Cutleaf Sagebrush

This plant ranges from central Idaho to western Wyoming and south to northern Utah. It is restricted to areas adjacent to those occupied by *A. tripartita*, which perhaps hybridized with *A. arbuscula* to form *A. a. thermopola* (Beetle 1960; Shultz 1984).

Classification

Artemisia arbuscula thermopola/Festuca idahoensis (Hironaka and others 1983)

Habitat and Capabilities

Habitat includes ridgetops and glacial outwash areas that are thinly mantled, poorly drained, and usually within forest openings. Soils are shallow and have a strongly developed clay horizon (Hironaka and others 1983). Low stature of shrubs and herbaceous species can be expected to be inherent to stands of cutleaf sagebrush. K. Johnson (1987) listed this as a spreading, lax shrub up to 30 cm tall.

Value for sage-grouse likely depends upon the size of forest openings in which stands are found. Smaller openings can be expected to have lower value than larger openings. Low site productivity indicates that stature and crown cover of cutleaf sagebrush will be of low value for nesting habitat. Soil and habitat features listed by Hironaka and others (1983) indicate early drying of herbaceous species and thus low value for brooding habitat. Windswept knolls might provide moderate- to high-value breeding and winter habitat where these are adjacent to stands of big sagebrush.

Shallow soils and poor drainage indicate low productivity. Restoration projects can be expected to be of low success due to the short growing season and water stress occurring during the seedling establishment stage (Hironaka and others 1983).

Artemisia bigelovii Bigelow Sagebrush

Bigelow sagebrush is known from California east to Colorado and Texas. Habitat includes rimrock, cliff faces, canyons, and ravines within the desert shrub and pinyon-juniper belts. Shrubs are commonly 10 to 30 cm tall with the flower stalks often equal to or greater than the length of the woody stems. The flower stalks die back each year.

The stature and growth form of Bigelow sagebrush is not conducive to sage-grouse nesting. Sage-grouse generally show low selectivity for the habitat of this plant. Where stands of Bigelow sagebrush are adjacent to big sagebrush stands, it might be used by sage-grouse as winter forage or sparingly at other times of the year.

Artemisia cana ssp. *bolanderi* Bolander Silver Sagebrush

Bolander silver sagebrush is primarily a plant of the Sierra Nevada Mountains of California. It extends into Oregon and barely into Nevada.

Habitat and Capabilities

This shrub occurs along streams and in snow catchment basins on granitic soils (Shultz 1986). Its value as sage-grouse habitat is likely similar to that of mountain silver sagebrush (*A. cana* ssp. *visicidula*). However, prolonged flooding of some sites likely limits the values of this type for sage-grouse habitat. The plant is known from the margin of the range of sage-grouse.

Artemisia cana ssp. *cana* Plains Silver Sagebrush

Plains silver sagebrush ranges from Montana and North Dakota south to northern Colorado and western Nebraska. As indicated by its common name, this is a plant of the plains and it is mostly found east of the Continental Divide. However, outlier populations have been found in Sweetwater County, Wyoming; Daggett County, Utah; and in Moffat County (Yampa River Valley) and Rio Blanco County, Colorado, on the west side of the Continental Divide.

Classification

Artemisia cana ssp. *cana/Elymus smithii* (Johnston [1987] listed several references for the Plains.)

Artemisia cana ssp. *cana / Symphoricarpos occidentalis / E. smithii* (Johnston [1987] listed three references for North Dakota.)

Artemisia cana ssp. *cana / Elymus cinereus* (Thatcher 1959)

Habitat and Capabilities

Plains silver sagebrush is most common on deep, well-drained soils on alluvial flats and terraces along water-courses (Walton and others 1986). This is well demonstrated in the distribution map by Beetle (1960) where the distribution of this plant is represented by linear lines following watercourses. It extends beyond watercourses especially on sandy soils (Thatcher 1959). Although not requiring a high water table, plains silver sagebrush can tolerate a water table within the rooting zone. Thatcher (1959) found it on soils with pH between 6.6 and 8.5 with most sites having a pH of above 7.4; he did not find it on heavy-textured soils regardless of topographic position. In Albany County, Wyoming, Thatcher (1959) found plains silver sagebrush from about 18 to 102 cm tall with shorter plants on upland sites. Within its range, it is capable of growing up to 1.5 m tall (Walton and others 1986) on better sites.

Stature of plains silver sagebrush indicates high value for sage-grouse for nesting and for winter habitat. This shrub is capable of forming dense stands (Walton and others 1986), which indicates high value for nesting habitat. However, the crown-form of plains silver sagebrush might not be of equal value to that of Vasey big sagebrush or Wyoming big sagebrush. The plains habitat indicates productive understory communities. Where stands occur with sufficient shrub crown cover, plains silver sagebrush might be expected to provide yearlong, high-value habitat for sage-grouse. However, linear stands of this plant might not have as high a value as do broad areas covered by Wyoming big sagebrush.

Artemisia cana Pursh ssp. *viscidula* Mountain Silver Sagebrush

Mountain silver sagebrush ranges from western Montana south to Nevada and Colorado.

Classification

Artemisia cana Pursh ssp. *viscidula / Deschampsia cespitosa* (Padgett and others 1989)

Artemisia cana Pursh ssp. *viscidula / Elymus trachycaulus* (Bramble-Brodahl 1978; Tew 1988)

Artemisia cana Pursh ssp. *viscidula / Festuca idahoensis* (Bramble-Brodahl 1978; Hironaka and others 1983; Youngblood and others 1985)

Artemisia cana Pursh ssp. *viscidula / Festuca ovina* (Padgett and others 1989; Tew 1988)

Artemisia cana Pursh ssp. *viscidula / Festuca thurberi* (Francis 1983; Johnston [1987] listed six references for Colorado; Tew 1988)

Artemisia cana Pursh ssp. *viscidula / Poa pratensis* (Padgett and others 1989; Youngblood and others 1985)

Artemisia cana Pursh ssp. *viscidula / Symphoricarpos oreophilus / Elymus trachycaulus* (Tew 1988)

Habitat and Capabilities

This is a plant of montane valleys where it is commonly associated with riparian communities. Soils of some communities show evidence of seasonally high water tables (Padgett and others 1989). Communities of mountain silver sagebrush often form between wetlands of sedges (*Carex* spp.) and willows (*Salix* spp.) and uplands of big sagebrush, aspen (*Populus tremuloides*), and coniferous forests. Occasionally, mountain silver sagebrush forms stands on upland sites well removed from riparian communities.

In the cool and moist setting of mountain silver sagebrush communities, cool season graminoids are highly productive. Tables from Tew (1988) from central Utah show total herbage production ranges from 858 kg/ha (855 lbs/acre) for the low end of an *A. c. viscidula / Festuca ovina* type to 1,919 kg/ha (1,713 lbs/acre) for an *A. c. viscidula / Festuca thurberi* type. With high production, potential for ground cover is also high. Tables from Tew (1988) indicate potential ground cover ranges from 84 to 97 percent with an average of 92 percent.

Mountain silver sagebrush is a multiple-stemmed shrub with capacity to sprout from the root crown. Although the leaves of mountain silver sagebrush persist though some or all of the winter, they dry upon freezing and many fall.

Common grasses include those listed above as vegetation type indicators. Succulent forbs including dandelion (*Taraxacum officinale*), longfoot clover (*Trifolium longipes*), yarrow (*Achillea millefolium*), and everywhere aster (*Aster chilensis*) are often present, and they likely increase with heavy, persistent livestock grazing.

Many stands of mountain silver sagebrush are likely too wet or snow covered to be of more than low-value nesting habitat for sage-grouse. Many are at elevations of deep snow accumulations in winter where low value for winter habitat is indicated. Drying and deciduous leaves indicate mountain silver sagebrush has low value for winter forage for sage-grouse. Crowns of mountain silver sagebrush are typically thin and ascending compared to the dense, spreading crowns of Vasey big sagebrush and Wyoming big sagebrush. This is likely a limiting factor in value for sage-grouse nesting habitat.

Presence of succulent forbs and proximity to water and meadows indicate high value for brood-rearing habitat for sage-grouse. This seems especially so where mountain silver sagebrush stands are adjacent to, or near to, stands of big sagebrush. Where mountain silver sagebrush communities are adjacent to stands of big sagebrush, value for sage-grouse is indicated to be higher than where they are surrounded by coniferous forests.

High elevation, cool climate, relatively high annual precipitation, and high productivity of aggressive perennials indicate high resistance to invasion of cheatgrass.

Artemisia filifolia Sand Sagebrush

This is a plant of the western Great Plains and Colorado Basin from eastern Wyoming and South Dakota, south to Arizona and New Mexico and extending into Texas and Mexico.

Classification

Artemisia filifolia / Andropogon hallii (Johnston [1987] listed two references for the Comanche and Cimarron National Grasslands.)

Artemisia filifolia / Sporobolus cryptandrus-Bouteloua gracilis (Johnston [1987] listed a reference for the Comanche National Grasslands.)

Habitat and Capabilities

Sand sagebrush is most common on sandy soil including dunes. The plants are from 50 to 150 cm tall, and they have a stature that would indicate favorable cover for sage-grouse. However, due to the sparse, filiform leaves, the crowns of sand sagebrush provide a thin cover. Shrub density in communities of sand sagebrush is also sometimes rather low.

Features of this shrub and its communities indicate low value for sage-grouse at any time of the year. The distribution of sand sagebrush is mostly at the margin and well beyond the current and historic range of sage-grouse. This could indicate sage-grouse did not find suitable habitat in sand sagebrush communities during their evolutionary history.

Artemisia frigida Fringed Sagebrush

This subshrub ranges from Alaska to the Atlantic and south to Arizona and Kansas. It is a characteristic species of the high plains of central North America, and it is disjunct in Siberia (Cronquist 1994; Welsh and others 1993).

Classification

Artemisia filifolia / Elymus spicatus (Francis 1983; Tew 1988)

Although Baker and Kennedy (1985) did not formally recognize a fringed sagebrush association, they did note the presence of this plant in a bluebunch wheatgrass/cushion plant community of rocky, wind-exposed summits.

Habitat and Capabilities

Fringed sagebrush is a subshrub that dies back to near the base in winter in most locations. Where it is a dominant or codominant, it often indicates windswept sites. It often grows as a minor component in many communities in more protected sites. Fringed sagebrush can readily pioneer disturbed sites, and it occurs through all successional stages including climax (Wambolt and Frisina 2002).

Stature and crown cover of fringed sagebrush rarely, if ever, meet the needs of sage-grouse for nesting, brood-rearing, or wintering. Where stands of fringed sagebrush are adjacent to those of big sagebrush, they might be used as strutting grounds. Windswept areas might provide winter forage. As fringed sagebrush dies back in winter, availability of active leaves can be expected to decrease. However, the lower leaves remain green throughout winter. Where these leaves are not covered by snow, they can be expected to be of some value to sage-grouse in winter.

Artemisia longiloba Early Sagebrush, Longleaf Sagebrush, Alkali Sagebrush

Early sagebrush ranges from extreme southeastern Oregon to western Wyoming and south to Nevada and Colorado. Early sagebrush has been treated as a part of the *A. arbuscula* complex. However, unlike taxa of the *A. arbuscula* complex, this plant flowers in late spring and early summer instead of fall. The difference in flowering period seems to indicate considerable phylogenetic distance from *A. arbuscula*. *Artemisia longiloba* also differs from the *A. arbuscula* complex in habitat and, to some extent, in geography. Recognition at the species level is consistent with treatments by Welsh and others (1993) and Wambolt and Frisina (2002).

Classification

Artemisia longiloba/Elymus spicatus (Bramble-Brodahl 1978; Robertson and others 1966)

Artemisia longiloba/Festuca idahoensis (Bramble-Brodahl 1978; Hironaka and others 1983; Nelson and Jensen 1987; Zamora and Tueller 1973)

Artemisia longiloba / Poa secunda (Francis 1983)

Habitat and Capabilities

Habitat includes fine-textured soils derived from alkaline shales in clay basins and in mountains (Schultz 1986; Wambolt and Frisina 2002). It was found only on claypan soils in Idaho (Hironaka and others 1983), on the Humboldt National Forest, Nevada (Nelson and Jensen 1987), and at North Park, Colorado (Robertson and others 1966). Soil features indicate curing of herbaceous plants in late spring or early summer.

Francis (1983) recorded only 24 plant species per stand in an early sagebrush type in Colorado compared to 30 to 40 species in Wyoming big sagebrush types and 55 to 67 species in Vasey big sagebrush types. Zamora and Tueller (1973) listed maximum shrub height at 24 cm and shrub crown cover at 19 percent for a habitat type in northern Nevada. Average annual production of 504 kg/ha (450 pounds/acre) (Nelson and Jensen (1987) and 572 kg/ha (510 pounds/acre) (Robertson and others 1966) have been reported for northeastern Nevada and northern Colorado, respectively.

Productivity, succulence of summer herbage, shrub height, and crown cover of most early sagebrush stands indicate relatively low value for sage-grouse nesting, brooding, and wintering habitat. Small stands might provide moderate to high-value strutting habitat, especially where they are adjacent to larger stands of big sagebrush. Windswept sites might provide some forage in winter. Heavy clay subsoil and shallow depth to bedrock limit the potential for management (Nelson and Jensen 1987).

Artemisia nova
Black Sagebrush

This nonsprouting, evergreen, low shrub ranges from Oregon to Montana and south to eastern California and north-central New Mexico.

Classification

Artemisia nova/Bouteloua gracilis (Tew 1988)
Artemisia nova/Elymus spicatus (Baker and Kennedy 1985; Hironaka and others 1983; Nelson and Jensen 1987; Tew 1988; Thatcher 1959; Zamora and Tueller 1973)
Artemisia nova/Festuca idahoensis (Hironaka and others 1983)
Artemisia nova/Stipa comata (Baker and Kennedy 1985; Zamora and Tueller 1973)
Artemisia nova/Stipa hymenoides (Nelson and Jensen 1987; Zamora and Tueller 1973)
Artemisia nova/Atriplex confertifolia/Elymus elymoides (Nelson and Jensen 1987)

Habitat and Capabilities

Black sagebrush has a rather wide elevational range that includes valleys within the cold desert shrub belt to slopes and ridges of mountains up into the lower end of the coniferous forest belt. It is noted to extend from 1,524 to 3,353 m in California (Munz 1959). Its extension upslope into montane areas is often facilitated by calcareous substrates.

Soils of black sagebrush communities are often basic and shallow or rocky or both. Compared to soils of big sagebrush taxa, soils of black sagebrush communities often have the lowest water-holding capacity, organic matter, and nitrogen content, and the thinnest mollic epipedons (Jensen 1990). In cold desert areas, soils can be expected to be without mollic epipedons.

Cold Desert Stands—Stands of black sagebrush in the cold desert shrub belt appear to be highly competitive with other native plants, and they appear to be long-persistent with little potential for displacement by other shrubs. However, in some areas black sagebrush is highly selected by ungulates, and heavily browsed stands are thinned and replaced by shrubs more resistant to grazing (Clary 1986; Holmgren and Hutchings 1972; Hutchings and Stewart 1953). With the introduction of cheatgrass, cold desert stands of black sagebrush are highly subject to conversion to cheatgrass systems. Replacement or displacement of black sagebrush stands in cold desert shrub systems can be expected to be long-persistent with low potential for recovery of sagebrush.

Crown cover of black sagebrush in stands of the cold desert belt is often limited to less than 15 percent by the low capacity of these sites to support vegetation. Height of black sagebrush in cold desert areas is mostly less than 25 cm. Cold desert shrub communities inherently have low potential for perennial forbs. Limitations are mostly a function of precipitation, which is commonly as low as 15 to 18 cm/year. Depending on amount and timing of precipitation, forbs might flourish in some years and be essentially absent in others.

Canopy cover and height of black sagebrush in cold desert areas and low presence of succulent forbs indicate low (or no) value for sage-grouse for nesting and brooding. Some stands might have moderate value in winter, or other times of the year, if they are adjacent to stands of Wyoming big sagebrush.

Artemisia nova var. *duchesnicola* (red clay sagebrush) is endemic to the Duchesne River Formation in the Uinta Basin in northeastern Utah (Welsh and Goodrich 1995). Plants of this taxon are restricted to the cold desert shrub environment. The discussion of values for the cold desert shrub belt of black sagebrush are applicable to communities of red clay sagebrush.

Disturbances that reduce black sagebrush in cold desert systems can be expected to be long-persistent, and recovery of black sagebrush can be expected to be slow. Risk of long-term loss of black sagebrush is high in cold desert areas compared to montane stands.

Stands Within the Pinyon-Juniper Belt—Stands of black sagebrush are common within the pinyon pine (*Pinus* spp.) and juniper (*Juniperus* spp.) belt where they are clearly seral to pinyon-juniper. Pinyon-juniper trees have advanced into many black sagebrush communities in the past several decades. Degree of displacement depends on time since disturbance and other factors. Stands within the pinyon-juniper belt are also highly vulnerable to conversion to cheatgrass-driven communities. Although black sagebrush is highly sensitive to fire, the long-term maintenance of black sagebrush within the pinyon-juniper belt requires periodic fires to reduce conifer encroachment. The potential for cheatgrass invasion greatly complicates management of black sagebrush communities within the pinyon-juniper belt in many areas.

Value of black sagebrush stands within the pinyon-juniper belt for sage-grouse is dependent on fire or other disturbance that reduces pinyon-juniper cover. Height and crown cover of black sagebrush on better sites in the pinyon-juniper belt are often within the limits indicated by Connelly and others (2000) for at least winter habitat. However, capability for height and crown cover decreases as site potential decreases. Sites of low to moderate potential for sagebrush cover have low value for sage-grouse nesting. These limitations also indicate low value for winter habitat except where stands are adjacent to more productive sagebrush sites. Early drying of herbaceous plants in summer indicates low value for brood rearing.

Montane Stands—Stands of black sagebrush above the pinyon-juniper belt are often quite stable but they are sometimes vulnerable to displacement by Douglas-fir (*Pseudotsuga menziesii*) and other conifers. These stands appear to be more resistant to displacement by cheatgrass than those of lower elevations.

Black sagebrush can be highly competitive in montane stands. At two sites on the Tavaputs Plateau (Ashley National Forest [ASNF] 67-26, 68-1) stands of black sagebrush were plowed and seeded with crested wheatgrass (*Agropyron cristatum*) and other aggressive introduced species. Where protected from livestock and lagomophs, black sagebrush returned to pretreatment status within 20 years, and essentially displaced the seeded grasses.

At another site (ASNF 45-6) on the Uinta Mountains, Ashley National Forest, black sagebrush was sprayed with 2,4-D. Percent sagebrush kill was high and grass production increased 3.2 times over what it was prior to treatment and in the control. At 14, 19, and 23 years post treatment, crown cover of black sagebrush was measured at 5, 12, and 17 percent, respectively. At 23 years post treatment, crown cover in the treated area equaled that of the control.

Black sagebrush does not sprout after fire. However it is capable of vigorous seedling establishment following fire. Montane stands of black sagebrush on the Tavaputs Plateau have returned to pretreatment crown cover within about 20 years after burning.

Crown cover of black sagebrush in stands at the upper edge of the pinyon-juniper belt and above have been measured at 27 percent (ASNF 68-1) and 28 percent (ASNF 68-69) at sites where use by wild ungulates has been minor. In the absence of livestock or at low levels of livestock use, but with mule deer (*Odocoileus hemionus*) concentrations in winter, crown cover has been measured at 15 to 18 percent (ASNF 4-4; 45-6U). At high concentrations of mule deer and elk (*Cervus elaphus*), crown cover was 11 percent (ASNF 32-64).

Shrubs of black sagebrush in the pinyon-juniper belt and above are 20 to 30 cm, and less commonly to 50 cm tall. Forbs are more common in montane stands than in cold desert stands. However, number of forbs and abundance of forbs are commonly much lower than in mountain big sagebrush communities.

The short stature of black sagebrush and lower abundance of forbs indicates that stands of this shrub have lower value than stands of mountain big sagebrush for sage-grouse nesting and winter habitat. The crowns are often in contact with the ground or close enough to the ground to obstruct sage-grouse nesting. However, some stands of black sagebrush do have height and crown cover features that meet the requirements indicated by Connelly and others (2000) for nesting and winter habitat for sage-grouse. Some windswept ridges supporting short and scattered plants of sagebrush have been found to be highly selected by sage-grouse as winter-feeding areas (Gullion 1964).

Artemisia papposa
Owyhee Sagebrush

This half-shrub ranges from the northern margin of the Snake River Plain in Idaho to adjacent Nevada and Oregon. It is limited in height to 5 to 15 cm, and the woody stems are commonly only half this height. Features of this plant indicate little, if any, value for sage-grouse nesting, brood rearing, or wintering. Where stands of Owyhee sagebrush are adjacent to stands of big sagebrush, they might provide strutting habitat and some forage. Herbaceous plants in Owyhee sagebrush communities can be expected to dry in early summer, and they likely have low value for sage-grouse in summer.

Artemisia pedatifida
Birdsfoot Sagebrush

This dwarf half-shrub ranges from Wyoming and southwestern Montana to Colorado. It is limited in height to 5 to 15 cm. Value for sage-grouse habitat is likely similar to that of Owyhee sagebrush discussed above.

Artemisia porteri
Porter Sagebrush

This is another dwarf half-shrub limited to less than 20 cm tall. It is a narrow endemic of barren clay hills in Fremont County, Wyoming (Dorn 1992). The habit and habitat of this plant indicate low value for sage-grouse except where stands are adjacent to stands of big sagebrush.

Artemisia pygmaea
Pygmy Sagebrush

Pygmy sagebrush is mostly limited to the Great Basin and the Uinta Basin, mostly in Nevada and Utah, with the margins of its range reaching into Arizona and Colorado. It is mostly confined to geologic strata that weather to semibarrens. It is an indicator of inherently semibarren, low productive sites. It is sometimes associated with narrowly endemic plants.

Stature of this plant is short enough that it has little, if any, value as cover for sage-grouse at any season of the year. It is indicative of sites of inherently low, scattered plants where rock, pavement, and bare ground dominate the soil surface. It is often associated with pinyon and juniper where the presence of trees also limits its value for sage-grouse. Forage value of pygmy sagebrush for sage-grouse is apparently unknown.

Sites occupied by pygmy sagebrush are not recommended for forage improvement projects. Reclamation of disturbed stands can be expected to be slow and difficult. The badland habitat indicates low potential for cheatgrass to form closed stands. However, cheatgrass can be expected to grow in the habitat of pygmy sagebrush.

Artemisia rigida
Scabland Sagebrush, Rigid Sagebrush, Stiff Sagebrush

This sprouting, deciduous shrub ranges from southeastern Washington and northeastern Oregon to western Idaho and northwestern Montana where it grows on basalt scablands of the Columbia Basin (Schultz 1986; Wambolt and Frisina 2002).

Classification

Artemisia rigida/Poa sandbergii (Hall 1973; Hironaka and others 1983; Tisdale 1986)

Habitat and Capabilities

Scabland sagebrush is the primary woody *Artemisia* of the Palouse grasslands (Hironaka and others 1983). It typically grows on a thin mantle of soil, underlain by basalt bedrock. The soil is commonly saturated in winter and spring, frequently enough to preclude establishment of other woody sagebrush species. The scabland habitat supports a relatively sparse cover of vegetation (Hironaka and others 1983). Hall (1973) listed 232 kg/ha (207 lbs/acre) herbage production, 5 percent surface rock, 18 percent erosion pavement, 20 percent bare ground, and soil depth of 8 to 25 cm with stoniness commonly of 25 to 60 percent for a community type in Oregon and Washington.

Cronquist (1994) described scabland sagebrush with a height of (20) 30 to 60 cm and with capability to sprout from the roots. This shrub is deciduous and drops all of its leaves in winter (Hironaka and others 1983).

Stature of the plant indicates moderate value for sage-grouse nesting. However, the scabland habitat indicates relatively low percent crown cover of sagebrush. Hall (1973) reported that cover for this sagebrush ranged from 5 to 20 percent. Tisdale (1986) found that foliar cover averaged 12 percent with no other shrubs in high-elevation communities of scabland sagebrush. He also found 11 species of perennial forbs occurring on more than one-half of the sites. On low-elevation sites with soils only 7 to 10 cm deep, Tisdale (1986) found sparse cover of scabland sagebrush and poor representation of forbs. The crowns of scabland sagebrush seem to be somewhat less spreading and not as dense as mountain big sagebrush or Wyoming big sagebrush. This also indicates lower value for nesting habitat for sage-grouse.

The scabland habitat indicates early summer drying of succulent plants. This feature suggests low value for nesting and brood rearing of sage-grouse. The deciduous feature of the plant makes this sagebrush of little value as winter forage for sage-grouse. In Idaho, scabland sagebrush forms a mosaic with other sagebrush types (Hironaka and others 1983). In this setting stands of this shrub likely provide strutting habitat.

The scabland habitat indicates difficulty and low returns for forage improvement and reclamation projects. Features of the type that indicate management limitations include severe moisture saturation during winter and severe frost heaving (Hall 1973; Hironaka and others 1983).

The sprouting capability of scabland sagebrush indicates high value of this shrub for reclamation where cheatgrass and high fire frequencies have altered structure and function of plant communities. However, high fidelity of this plant for basalt scablands indicates seedings beyond its rather narrow ecological amplitude are not likely to succeed.

Artemisia rothrockii Rothrock Sagebrush

As treated by Shultz (1986), Rothrock sagebrush is endemic to the southern Sierra Nevada and San Bernardino Mountains of California where it is found in high-elevation silt basins and on rocky slopes. This places Rothrock sagebrush, in a strict sense, beyond the range of sage-grouse. References to Rothrock sagebrush in the Intermountain West are likely based on specimens of spiked big sagebrush (*A. tridentata* ssp. *spiciformis*), because habit and habitat features of Rothrock sagebrush are somewhat similar to those of spiked big sagebrush.

Adaptation to moist conditions, a strong tendency to root-sprout, large heads, and small amounts of pubescence on the involucres suggest a close relationship to *A. cana* (Ward 1953). The origin of Rothrock sagebrush is perhaps similar to that of spiked big sagebrush, but perhaps with Bolander silver sagebrush, rather than mountain silver sagebrush, being one of the parent taxa.

Artemisia spinescens Bud Sagebrush

This is a plant of cold desert areas from Oregon to Montana and south to California and New Mexico. The shrubs are commonly 5 to 30 cm tall and rarely taller. Bud sagebrush is commonly associated with other desert shrubs, but occasionally it forms small stands where it is the dominant shrub. Cold desert shrub communities typically have relatively few forb species, and these are mostly annuals that flourish in some years and are essentially absent in others, depending on the amount and season of precipitation. These communities are commonly of low stature and plant density.

The height and form of bud sagebrush and the communities in which it is found indicate low value for sage-grouse. Where bud sagebrush is adjacent to stands of big sagebrush, it likely provides high-value forage for sage-grouse in spring and early summer. Stands of bud sagebrush might be used as strutting grounds, particularly where they are adjacent to stands of big sagebrush.

Artemisia tridentata ssp. *parishii* Parish Big Sagebrush

Parish big sagebrush is restricted to the costal ranges and cismontane region of California. Features of this taxon and its communities are quite similar to those of basin big sagebrush. Features ascribed to Parish big sagebrush might be found at random over much of the range of basin big sagebrush. Perhaps additional work will indicate *A. t.* ssp. *parishii* should be reduced to synonymy. If a distinct taxon, Parish big sagebrush is known from outside the range indicated for sage-grouse.

Artemisia tridentata ssp. *spiciformis* Spiked Big Sagebrush, Snowfield Big Sagebrush

Plants of this taxon are of apparent hybrid origin with mountain silver sagebrush and montane taxa of big sagebrush being the apparent parents. Some variation in populations of spiked big sagebrush could be a function of different parent taxa on the big sagebrush side. Features of silver sagebrush include numerous fine stems, large heads, and ability to sprout. Features of big sagebrush include lobed leaves and upland habitat.

Generally there are fewer flower heads per inflorescence in spiked big sagebrush than is common in the suggested parent taxa. Flower heads of spiked big sagebrush are often larger than those of either mountain silver sagebrush or other taxa of the big sagebrush complex, and in most populations there are more flowers per head. When transplanted at low elevations, plants of spiked big sagebrush have come to full flower by the end of May and first of June (McArthur and Goodrich 1984). This is a feature not shared by either silver sagebrush or other taxa of big sagebrush.

This taxon has been confused with Rothrock sagebrush that Shultz (1986) considered endemic to California. References to Rothrock sagebrush outside California, including Bramble-Brodahl (1978), are based on spiked big sagebrush. Vegetation types outside California listed under Rothrock sagebrush are included in the following list for spiked big sagebrush. Spiked big sagebrush has also been confused with Vasey big sagebrush. Osterhout's *Artemisia spiciformis*, based on his type specimen, clearly describes plants with a few large heasds per inflorescence. Beetle's concept of *Artemisia tridentata* ssp. *vaseyana* forma *spiciformis* is based, at least in part, on Vasey or subalpine big sagebrush. Vegetation types of Hironaka and others (1983) listed under spiked big sagebrush follow Beetle's concept and are treated in this paper under Vasey or subalpine big sagebrush.

Spiked big sagebrush is a sprouting, evergreen shrub that ranges from southern Idaho to western Wyoming and southwestern Montana and south to California, Nevada (Ruby Mountains), and Colorado. In Utah it is mostly confined to basic substrates.

Classification

Artemisia tridentata ssp. *spiciformis/Elymus trachycaulus* (Tart 1996)

Artemisia tridentata ssp. *spiciformis/Trisetum spicatum* (Tart 1996)

Artemisia tridentata ssp. *spiciformis*/mountain forb (Bramble-Brodahl 1978)

Habitat and Capabilities

Habitat of spiked big sagebrush typically includes open slopes and openings within the coniferous forest belt. Populations of this plant extend upward to treeline and approach alpine conditions. It is known from 2,680 to 3,350 m elevation in Utah (Welsh and others 1993). It is capable of withstanding longer duration of snow cover than are other taxa of big sagebrush. The common name of snowfield big sagebrush is comparatively well applied to many populations of this taxon. The capacity to sprout appears to be a factor in the ability to occupy areas of deeper snow cover.

Some associated species also indicate high elevation, snow-persistent sites. These include slender wheatgrass, mountain brome (*Bromus carinatus*), and yellowbrush (*Chrysothamnus viscidiflorus* var. *lanceolatus*). High elevation and cold climate indicate high resistance of these communities to invasion by cheatgrass.

Although height and crown cover of spiked big sagebrush and some features of spiked big sagebrush communities indicate high-value habitat for sage-grouse, the elevation of many populations of this taxon are such that they are not snow-free until after the nesting season of sage-grouse. Stands of this sagebrush are typically covered with too much snow in winter to provide winter habitat. Thus, their value for sage-grouse can be limited primarily to the summer season. The abundance of succulent vegetation during much of the summer suggests potential for high-value summer habitat. However, sage-grouse are not likely to reach some of the higher elevation populations of spiked big sagebrush at any season of the year.

Artemisia tridentata ssp. *tridentata* Basin Big Sagebrush

Basin big sagebrush ranges from southern British Columbia to southwestern North Dakota and south to Baja California and northern New Mexico. This taxon rivals, or perhaps equals, Wyoming big sagebrush in its geographic range of distribution. However, the extent of area covered is much less than that covered by Wyoming big sagebrush.

Classification

Artemisia tridentata ssp. *tridentata/Bouteloua gracilis* (Johnston 1987)

Artemisia tridentata ssp. *tridentata/Chrysothamnus nauseosus* (Johnston 1987)

Artemisia tridentata ssp. *tridentata/Elymus cinereus* (Baker and Kennedy 1985; Johnston [1987] listed six references for Colorado and Wyoming.)

Artemisia tridentata ssp. *tridentata/Elymus lanceolatus* (An apparent type unpublished, Ashley National Forest.)

Artemisia tridentata ssp. *tridentata/Elymus smithii* (Francis 1983)

Artemisia tridentata ssp. *tridentata/Elymus spicatus* (Hironaka and others 1983; Jensen 1989; Nelson and Jensen 1987)

Artemisia tridentata ssp. *tridentata/Festuca idahoensis* (Hironaka and others 1983; Nelson and Jensen 1987)

Artemisia tridentata ssp. *tridentata/Stipa comata* (Hironaka and others 1983)

Artemisia tridentata ssp. *tridentata/Purshia tridentata / Elymus lanceolatus* (Johnston 1987)

Artemisia tridentata ssp. *tridentata/Sarcobatus vermiculatus / Elymus smithii* (Francis 1983)

Habitat and Capabilities

Stands of this shrub are most highly developed on deep alluvial soils of canyon bottoms and drainage ways of valleys. Although of alluvial settings, basin big sagebrush is highly sensitive to flooding and high water tables, and it does better on well-drained areas, such as old terraces, than it does on current flood plains with wet-meadow vegetation. Soil texture is often sandy, gravelly, or loamy. Height of basin big sagebrush plants is commonly 1 to 3 m and occasional specimens exceed 3 m.

Jensen (1989) listed annual production in one basin big sagebrush type at 665 kg/ha (593 lbs/acre). Although this indicates lower production than for many mountain big sagebrush communities, potential production under irrigation is high. This tall sagebrush was used as an indicator of productive sites for agriculture during settlement of the West. Due to the deep alluvial, productive soils and valley locations, much of the original habitat of basin big sagebrush has been converted to agricultural land. Many populations are highly vulnerable to cheatgrass invasion and subsequent displacement by cheatgrass via high fire frequency.

Based on observations and studies on the Tavaputs Plateau of Utah, it appears that basin big sagebrush might take 4 or more decades to achieve 20 percent crown cover following fire where livestock grazing is limited to winter. In this respect it appears to function more like Wyoming big sagebrush than like Vasey big sagebrush. However, in other parts of its range, it might recover more rapidly, and especially under spring and summer use by livestock.

Tall stature indicates low value for strutting habitat. Although stature of this shrub meets the height needs for sage-grouse nesting, the crowns of the shrubs are often elevated above the ground as much as 60 to 100 cm or more. Sage-grouse are likely to avoid stands with this feature when selecting nest sites. Stands with succulent forbs might provide moderate value brood-rearing habitat.

Tall stature and valley bottom habitat indicate foliage of basin big sagebrush would be available to sage-grouse in most locations in most winters. However, in very tall stands, the leaves are difficult for sage-grouse to reach. Other features of some basin big sagebrush habitats appear to be of low preference by sage-grouse. This includes the bottoms of narrow stream canyons where the physical features of the canyon perhaps discourage sage-grouse. Stands in wide valleys are indicated to be of much higher value to sage-grouse than those confined to narrow canyons.

Artemisia tridentata ssp. *vaseyana* Vasey Big Sagebrush, Subalpine Big Sagebrush, Mountain Big Sagebrush

In addition to *Artemisia tridentata.* ssp. *vaseyana*, *A. t.* var. *pauciflora* has been recognized as part of the Vasey big sagebrush complex based on fewer flowers per head and on distribution and habitat (Goodrich and others 1985).

In general, populations of Vasey big sagebrush are more common from Washington to Montana and south to Oregon and Wyoming. Stands of mountain big sagebrush are more common and more highly developed in the Great Basin, Colorado Plateau, and central Rocky Mountains. In Wyoming, communities of Vasey big sagebrush are generally at higher elevation than those of mountain big sagebrush (Tart 1996). Even at high elevations in Utah, very few stands of sagebrush are clearly identifiable as typical Vasey big sagebrush. Critical review of numerous populations is needed to more fully understand the distribution and to clarify separation of these two taxa. Intermediate populations might be encountered throughout much of the range of these two taxa.

The Vasey/mountain big sagebrush complex is a landscape dominant in mountains of much of the West from British Columbia and western Montana south to California and Colorado. Within the big sagebrush group, the Vasey sagebrush complex is likely second only to Wyoming big sagebrush in extent of area covered.

Classification

Distinction between *Artemisia tridentata* ssp. *vaseyana* and *A. t.* var. *pauciflora* is rare in literature dealing with vegetation type classification. This is because publication of *A. t.* var. *pauciflora* (Goodrich and others 1985) followed much of the sagebrush classification work listed below, and likely because of reluctance to accept the distinction. Of the works listed below, only Tart (1996) recognized *A. t.* var. *pauciflora*. Since the distinction was not made in most classification works, *A. t.* var. *pauciflora* is not treated separately in the following list of vegetation types or in the discussion that follows. All references to the complex are lumped under *A. t.* ssp. *vaseyana* or Vasey big sagebrush.

Artemisia tridentata ssp. *vaseyana*/*Bouteloua gracilis* (Tew 1988)

Artemisia tridentata ssp. *vaseyana*/*Bromus carinatus* (Hironaka and others 1983; Tew 1988)

Artemisia tridentata ssp. *vaseyana*/*Carex geyeri* (Hironaka and others 1983)

Artemisia tridentata ssp. *vaseyana*/*Elymus cinereus* (Jensen 1989; Nelson and Jensen 1987)

Artemisia tridentata ssp. *vaseyana*/*Elymus cinereus*/*Bromus carinatus* (Mooney 1985; Tew 1988)

Artemisia tridentata ssp. *vaseyana* /*Elymus lanceolatus* (An apparent type unpublished, Ashley National Forest.)

Artemisia tridentata ssp. *vaseyana*/*Elymus smithii* (Tew 1988)

Artemisia tridentata ssp. *vaseyana*/*Elymus spicatus* (Baker and Kennedy 1985; Bramble-Brodahl 1978; Hironaka and others 1983; Jensen 1989; Mooney 1985; Mueggler and Stewart 1980; Nelson and Jensen 1987)

Artemisia tridentata ssp. *vaseyana*/*Elymus spicatus*/*Poa fendleriana* (Mooney 1985)

Artemisia tridentata ssp. *vaseyana*/*Elymus trachycaulus* (Tart 1996)

Artemisia tridentata ssp. *vaseyana*/*Festuca idahoensis* (Bramble-Brodahl 1978; Francis 1983; Hironaka and others 1983; Jensen 1989; Mooney 1985; Mueggler and Stewart 1980; Nelson and Jensen 1987)

Artemisia tridentata ssp. *vaseyana*/*Festuca idahoensis*/*E. spicatus* (Tart 1996)

Artemisia tridentata ssp. *vaseyana*/*Festuca thurberi* (Francis 1983; Johnston[1987] listed nine references for Colorado.)

Artemisia tridentata ssp. *vaseyana/Festuca scabrella* (Mueggler and Stewart 1980)

Artemisia tridentata ssp. *vaseyana/Leucopoa kingii* (Johnston [1987] listed two references for Colorado; this is also found in Utah.)

Artemisia tridentata ssp. *vaseyana/Leucopoa kingii / Koeleria macrantha* (Mooney 1985)

Artemisia tridentata ssp. *vaseyana/Stipa comata* (Bramble-Brodahl 1978; Hironaka and others 1983)

Artemisia tridentata ssp. *vaseyana/Stipa richardsonii* (Tart 1996)

Artemisia tridentata ssp. *vaseyana/Trisetum spicatum* (Tart 1996)

Artemisia tridentata ssp. *vaseyana/Purshia tridentata* (Tart 1996)

Artemisia tridentata ssp. *vaseyana/Purshia tridentata / E. smithii* (Johnston [1987] listed three references for this association in Colorado.)

Artemisia tridentata ssp. *vaseyana/Purshia tridentata / F. idahoensis* (Francis 1983)

Artemisia tridentata ssp. *vaseyana/Symphoricarpos oreophilus* (Tart 1996)

Artemisia tridentata ssp. *vaseyana/Symphoricarpos oreophilus / Bromus carinatus* (Jensen 1989; Nelson and Jensen 1987)

Artemisia tridentata ssp. *vaseyana/Symphoricarpos oreophilus / E. spicatus* (Bramble-Brodahl 1978; Hironaka and others 1983; Jensen 1989; Nelson and Jensen 1987; Tew 1988; Tueller and Eckert 1987)

Artemisia tridentata ssp. *vaseyana/Symphoricarpos oreophilus / E. trachycaulus* (Tew 1988; Tueller and Eckert 1987)

Artemisia tridentata ssp. *vaseyana/Symphoricarpos oreophilus / F. idahoensis* (Bramble-Brodahl 1978; Hironaka and others 1983; Tueller and Eckert 1987)

Artemisia tridentata ssp. *vaseyana/Symphoricarpos oreophilus / Carex geyeri* (Hironaka and others 1983)

Habitat and Capabilities

The lengthy list of vegetation types listed above for the Vasey big sagebrush complex is a reflection of the high diversity of habitats occupied by this complex. It also reflects the favorable climate for plant growth within these habitats. Paralleling the diversity of vegetation types is the number of associated species, as discussed below and shown in table 3.

Stands of this complex are most highly developed in areas with about 35 to 75 cm of annual precipitation. Stands on sites receiving more than 75 cm annual precipitation are mostly confined to southerly exposures within the aspen and coniferous forest belts. Within the coniferous belt, stands are found on sites receiving as much as 149 cm annual precipitation. On drier sites receiving less than 35 cm of annual precipitation, they often grade into Wyoming big sagebrush communities.

Florescence tests in Utah show a rather broad hybrid or mixing zone between Wyoming big sagebrush and Vasey big sagebrush in the 30 to 36 cm precipitation belt, which in Utah corresponds rather well with the pinyon-juniper belt (Goodrich and others 1999a). Where stands of Vasey big sagebrush occur within the distribution of pinyon-juniper, they are often found in and mostly above the pinyon-juniper thermal belt (Goodrich 1981; Goodrich and others 1999a; Hodgkinson 1989). Within the mixing zone, bitterbrush is sometimes an associate. Below the pinyon-juniper belt and within well-marked Wyoming big sagebrush communities, bitterbrush is rare or lacking.

Table 3—Comparison of features of Wyoming big sagebrush communities (ARTRW) from pediments associated with the Green River and mountain big sagebrush communities (ARTRP) on the Bishop Conglomerate Formation (from Goodrich and Huber 2001).

Community feature	ARTRW	ARTRP
	n = 9	*n = 6*[a]
Crown cover of sagebrush (percent)	0–22	0–38
Total number of vascular plant taxa	51	93
Total number of shrub taxa	7	7
Total number of graminoid taxa	13	20
Total number of forb taxa	26	66
Average number taxa/site (alpha diversity)	18	47
Average number of taxa/quadrat (beta diversity)	2.92	11.35
Average number of shrub taxa/site	4	4
Average number of graminoid taxa/site	6	12
Average number of forb taxa/site	7	31
Shrubs with 100 percent consistency	1	3
Shrubs with >49 percent consistency	1	4
Graminoids with 100 percent consistency	1	7
Graminoids with >49 percent consistency	7	12
Forbs with 100 percent site consistency	0	8
Forbs with >49 percent site consistency	4	31
Average ground cover (percent)	58	92

[a]n = 6 except for crown cover of sagebrush for which n = 42.

Big sagebrush of this complex is often indicative of Mollisols or mollic intergrades with mollic epipedons. In Oregon, Swanson and Buckhouse (1984) found it on Mollisols and Wyoming big sagebrush on Aridisols. Jensen (1989, 1990) listed five community types for Vasey big sagebrush in northeastern Nevada, all of which were found principally on Cryoboralls (Mollisols with cold temperature regime). Tart (1996) found communities of Vasey big sagebrush common on Cryoboralls on the west flank of the Wind River Mountains, Wyoming.

Plants of the Vasey big sagebrush complex frequently grow where depth and duration of snow pack is considerably greater than what is typical for Wyoming big sagebrush (Meyer and Monsen 1992). Sturges and Nelson (1986) found Vasey big sagebrush grew where snow depth was greater than 38 cm, and Wyoming big sagebrush was more common where snow depth was less than 40 cm.

Compared to those of Wyoming big sagebrush, the ecological settings of Vasey big sagebrush are more productive. Higher production is indicated as inherent over at least the past few hundred to thousands of years by the dominance of Mollisols instead of Aridisols in Vasey big sagebrush communities. Higher production of Vasey big sagebrush compared to Wyoming big sagebrush at present is verified by comparison of annual growth per unit area. It is also indicated by greater ground cover potential, higher diversity of understory species, and the potential for greater crown cover of sagebrush.

A range of annual production of 418 to 2,354 kg/ha (373 to 2,100 lbs/acre) is indicated for Vasey big sagebrush communities (Goodrich and Huber 2001; Harniss and Murray 1973; Jensen 1989; Tart 1996; Tew 1988). The low end of the range (418 kg/ha) was reported from near Dubois, Idaho, where sample sites were likely near the ecotone with Wyoming big sagebrush. Sites where bluebunch wheatgrass and blue gramma (*Bouteloua gracilis*) were indicators generally had lower production than sites with snowberry (*Symphoricarpos* spp.) and slender wheatgrass (*Agropyron trachycaulum*) as indicator species.

Ground cover is rarely less than 65 percent, and it is commonly above 85 percent even under moderate livestock grazing. At sites in the Uinta Mountains with a history of about 100 years of livestock grazing, ground cover has been found at 87 to 95 percent with an average of 92 percent (Goodrich and Huber 2001). For the Fishlake National Forest, Tew (1988) considered Vasey big sagebrush communities at potential to have between 7 and 22 percent bare soil. This indicates ground cover potential of between 78 and 93 percent for that National Forest. Tart (1996) listed percent bare soil at 6 to 15 percent for 8 plant associations at late seral condition on the West Flank of the Wind River Mountains. This indicates potential for ground cover of about 85 to 94 percent.

Potential for recovery of ground cover following fire is high. Ground cover was found at potential (90 percent or higher) within 5 to 7 years postfire on the Ashley National Forest. This recovery was concurrent with light to moderate livestock grazing. At a site without livestock (ASNF 5-29), ground cover increased from 55 to 97 percent at 2 and 6 years postfire, respectively. However, it is apparent that not all Vasey big sagebrush sites have potential for ground cover of over 90 percent. At two adjacent sites on Bare Top Mountain, Uinta Mountains (ASNF 5-13), ground cover was found at 62 and 85 percent in an area closed to livestock grazing for over 20 years. The site with 62 percent ground cover supported moderate frequency of rock goldenrod (*Petradoria pumila*) and low pussytoes (*Antennaria dimorpha*). These plants were lacking on the site with 85 percent ground cover. These and other plants might be used as indicators of lower potential for ground cover in Vasey big sagebrush communities.

On the west flank of the Wind River Mountains, Wyoming, Tart (1996) generally found 10 to 44 vascular plant taxa per 375 m^2 plot in associations of the Vasey big sagebrush complex. He also indicated potential for ground cover was greater than 80 percent in these associations.

Fisser (1962) found average percent basal area was 12.4 for graminoids and 6.4 for forbs. Although he found 43 forbs in his study area, only silver lupine (*Lupinus argenteus*) and 11 other forbs had greater than trace amounts of cover, and the lupine accounted for 70.1 percent of the forb cover.

Diversity of vascular plants is relatively high in Vasey big sagebrush communities compared to Wyoming big sagebrush communities. Table 3 compares Vasey big sagebrush communities on plateau lands of the Bishop Conglomerate Formation in the eastern Uinta Mountains with Wyoming big sagebrush communities on gravel pediments at the base of the Uinta Mountains. This comparison includes a contrast of an average of 47 and 18 vascular plant taxa per site for Vasey big sagebrush and Wyoming big sagebrush communities, respectively (Goodrich and Huber 2001).

Average crown cover of Wyoming big sagebrush rarely exceeds 25 percent in areas of over 0.5 ha, while values of greater than 25 percent are common in Vasey big sagebrush communities. Without fire or other crown-reducing disturbance for greater than 30 years, crown cover values between 25 and 40 percent are common for Vasey big sagebrush communities in the Uinta Mountains. On the west flank of the Wind River Mountains, Tart (1996) found that canopy cover of all shrubs in mid and late seral communities of Vasey big sagebrush was between 23 to 40 percent. Crown cover of Vasey big sagebrush of nearly 50 percent was recorded in recent studies in Strawberry Valley and West Tavaputs Plateau, Utah, where values of between 35 and 45 percent are not uncommon (Goodrich and Huber 2004, unpublished data).

Higher productivity, including greater crown closure of Vasey big sagebrush communities, supports a higher fire frequency than for Wyoming big sagebrush communities. Although Houston (1973) did not specify subspecies of sagebrush for his study area, the location and associated species indicate Vasey big sagebrush where he suggested a mean fire frequency of 20 to 25 years.

Miller and Rose (1999) found mean fire intervals of 12 to 15 years for a sagebrush steppe basin in Oregon. Citing four papers, Miller and others (1999) indicated fire return intervals usually ranged between 10 and 25 years. This is well within the range of 10 to 40 years suggested by Winward (1991).

Shorter recovery time for Vasey big sagebrush, compared to Wyoming big sagebrush, following fire is also indicative of an ecological history of higher fire frequency. Recovery from a burn to 20 percent canopy cover can range from 12 years in the Vasey big sagebrush type to over 40 years in the drier Wyoming big sagebrush type (Winward 1991). Annual herbage production of sagebrush has returned to preburn levels within 30 years or less following fire (Harniss and Murray 1973). Review of data from over 40 sites on the Ashley National Forest shows Vasey big sagebrush returning to over 20 percent crown cover within 20 years of burning (Goodrich and others 2004, unpublished data). History of some of these sites included previous burns or herbicide applications from which sagebrush recovered prior to the most recent fire.

Although some species are reduced in density and production for a few years, essentially all plant species of Vasey big sagebrush ecosystems return to burned areas by seedbanks, sprouting, or mobile seeds. Big sagebrush is among the least responsive species to fire, but it generally returns to burned areas.

The general adaptation of plant species to fire within Vasey big sagebrush ecosystems is additional evidence of an ecological history of fire. Also indicative of a high fire frequency for Vasey big sagebrush communities is the ease with which these communities are burned under prescribed conditions. On the Ashley National Forest, fire has spread though these communities when most other communities will not carry fire.

In general, Vasey big sagebrush communities demonstrate greater capacity to recover from fire and to resist cheatgrass dominance or turn into cheatgrass-driven communities than do Wyoming big sagebrush communities. In the Uinta Mountains, cheatgrass has made little entry into Vasey big sagebrush communities of plateau lands above 2,450 m elevation. It has made conspicuous advances into these communities on southerly facing aspects of these mountains, and particularly on gradients exceeding 40 percent. However, even on southerly aspects, native plants have demonstrated high capacity to recover in the presence of cheatgrass following fire.

Height and crown cover of Vasey big sagebrush in mid- and late-seral communities provide high-value nesting habitat for sage-grouse. Distance between the ground and lower level of the dense, spreading crowns of Vasey big sagebrush is often highly favorable for concealment of sage-grouse nests. The high capacity of these communities to produce an abundance of understory herbage is highly conducive to nest concealment and for brood-rearing habitat. Abundance and diversity of forbs and graminoids and succulence of herbaceous plants in summer indicate higher value for sage-grouse brooding habitat than for many Wyoming big sagebrush communities.

Deeper snow accumulations than in Wyoming big sagebrush communities indicate lower value for winter habitat. However, in the presence of tall shrubs or on windswept sites, high-value winter habitat is provided within some stands of Vasey big sagebrush. In some cases sage-grouse are known to summer in Vasey big sagebrush areas (Strawberry Valley, Utah), and migrate to Wyoming big sagebrush areas for winter (Current Creek near Fruitland, Utah) (Bambrough 2002).

The differences between communities of Wyoming big sagebrush and those of Vasey big sagebrush are major. These differences have implications for restoration projects and habitat values for sage-grouse and other wildlife. Where expectations exceed the capability of the land, restoration projects and other management activities are marked for disappointment.

Artemisia tridentata ssp. *wyomingensis* Wyoming Big Sagebrush

Wyoming big sagebrush is a landscape dominant of plains and basins from eastern Washington to western North Dakota and south to California and northern Arizona and New Mexico (McArthur and Sanderson 1999a). This range includes the Snake River Plain of Idaho, basins of the Great Basin, and the Intermountain basins and high plains of Wyoming that grade into the mixed grass prairie.

Classification

Artemisia tridentata ssp. *wyomingensis* / *Atriplex confertifolia* / *Grayia spinosa* / *Stipa comata* (Baker and Kennedy 1985)

Artemisia tridentata ssp. *wyomingensis*/*Elymus elymoides* (Hironaka and others 1983; Nelson and Jensen 1987)

Artemisia tridentata ssp. *wyomingensis*/*Elymus spicatus* (Baker and Kennedy 1985; Francis 1983; Hironaka and others 1983; Mueggler and Stewart 1980)

Artemisia tridentata ssp. *wyomingensis* / *Poa secunda* (Hironaka and others 1983)

Artemisia tridentata ssp. *wyomingensis*/*Elymus lanceolatus* (An apparent type unpublished, Ashley National Forest.)

Artemisia tridentata ssp. *wyomingensis* / *Elymus smithii* (Baker and Kennedy 1985; Johnston [1987] listed nine references from Colorado, Idaho, Montana, North Dakota, and Wyoming for this association. Although both *A. t. vaseyana* and *A. t. wyomingensis* were listed by Johnston [1987] for this association, elevation and soils strongly indicate *A. t. wyomingensis* is the sagebrush of this association in most areas.)

Artemisia tridentata ssp. *wyomingensis* / *Stipa comata* (Hironaka and others 1983)

Artemisia tridentata ssp. *wyomingensis*/*Stipa hymenoides* (Johnston [1987] listed three references for Colorado.)

Artemisia tridentata ssp. *wyomingensis* / *Stipa thurberiana* (Hironaka and others 1983)

Hilaria jamesii / *Artemisia tridentata* ssp. *wyomingensis* (Everett and others 1980)

Habitat and Capabilities

The list of vegetation types for Wyoming big sagebrush is only about one-third as long as that for the Vasey big sagebrush complex. This shorter list is an apparent reflection of the lower diversity of habitats occupied by Wyoming big sagebrush. It also reflects the more restrictive climate for plant growth within the habitat of this shrub. Parallel to the reduced diversity of vegetation types is the reduced number of associated species, as discussed below and as shown in table 3.

Soils under Wyoming big sagebrush communities are typically Aridisols (Barker and McKell 1986; Swanson and Buckhouse 1984) with restrictive horizons at about 30 to 56 cm (Goodrich 1981; Winward 1983). However, communities of this plant are also found on soils with alluvial features in which deposition exceeds horizon development where the restrictive horizon, if present, is much deeper than indicated above.

Stands of Wyoming big sagebrush are most highly developed in areas with about 22 to 30 cm of annual precipitation. They do develop with only 17 to 22 cm of annual precipitation, but in this case they are often confined to drainageways and other local conditions where they grade into cold desert shrub communities. With about 30 to 36 cm of annual precipitation they often grade into stands of the Vasey big sagebrush complex.

Florescence tests in Utah show a rather broad hybrid or mixing zone between Wyoming big sagebrush and Vasey big sagebrush within the 30 to 36 cm precipitation belt, which in Utah corresponds rather well with the pinyon-juniper belt (Goodrich and others 1999a). Where Wyoming big sagebrush is found within the distribution of pinyon-juniper, it is often found in and below the pinyon-juniper thermal belt (Goodrich 1981; Goodrich and others 1999a; Hodgkinson 1989).

Lower annual precipitation indicates less annual production for Wyoming big sagebrush communities than for the Vasey big sagebrush complex. Lower production potential is reflected in a lower potential for ground cover. Winward (1983) noted as much as 25 percent bare ground for Wyoming big sagebrush communities even under undisturbed conditions. Kindschy (1994) reported 38 percent bare soil and 21 percent rock for a pristine site with 5 to 7.5 percent crown cover of Wyoming big sagebrush in the Jordan Crater Kipukas of southeastern Oregon. In Daggett County, Utah, a mean of 55 percent ground cover and 45 percent bare soil was found for six sites that had been rested from livestock use for greater than 10 years (Goodrich and others 1999b). Rowlands and Brian (2001) indicated as high as 70 percent bare ground was found in a relic site in northern Arizona in what appeared to be a Wyoming big sagebrush community.

Lower potential for production compared to Vasey big sagebrush communities is also reflected in the lower potential for crown cover of Wyoming big sagebrush. Average crown cover of Wyoming big sagebrush rarely exceeds 25 percent in areas of over 0.5 ha, while values of greater than 25 percent are common in Vasey big sagebrush communities.

Fisser (1962) found average percent crown cover of big sagebrush ranged from 17.8 to 27 with an overall average of 22.4 on arid, low-elevation sites on the Owl Creek Range in Wyoming. In contrast, he found average percent crown cover of big sagebrush ranged from 13 to 60 with an overall average of 35.6 on high-elevation, moist sites. The work of Fisser (1962) preceded the published taxonomic distinction of Wyoming big sagebrush by Beetle and Young (1965); however, his description of sites and community features leaves little doubt that he was distinguishing between Wyoming big sagebrush and the Vasey big sagebrush complex.

On low-elevation, semiarid sites, average crown cover of big sagebrush of up to 23 percent has been reported by Anderson and Holte (1981), Goodrich and others (1999b), Rowlands and Brian (2001), Tuller and Blackburn (1974), and Winward (1991). Average values of higher than 23 percent were not found in the literature reviewed for this paper. The high of 27 percent reported by Fisser (1962) appears to represent the high end for Wyoming big sagebrush. Higher values might be found under some conditions including alluvial soils without restrictive horizons and where some moisture is funneled; however, these appear to be exceptions limited to localized areas.

Winward (1991) suggested Wyoming big sagebrush sites with the least disturbance had sagebrush crown cover values of between 8 and 11 percent while those with highest grazing impacts had sagebrush crown cover that exceeded 20 percent.

Sturges and Nelson (1986) found Wyoming big sagebrush grew in locations where the snow pack was less than 40 cm deep and Vasey big sagebrush grew where snow pack was greater than 38 cm. In this respect, Wyoming big sagebrush communities are indicted to provide a larger base of winter habitat for sage-grouse than communities of Vasey big sagebrush.

In addition to sagebrush-grass communities, Wyoming big sagebrush forms communities with sparse understory at lower elevations and especially at the lower end of its precipitation range. Such communities are particularly common in some geomorphic settings. They are common on the Green River Formation in Sweetwater County, Wyoming, where annual precipitation is about 18 to 20 cm, and where they intergrade into cold desert shrub communities as they do in other parts of the West. Striking contrasts in understory composition and abundance are common at the interface of materials of the Green River Formation and areas where this formation is covered with pediment materials deposited by the Green River. The pediment materials support grass-rich communities, and the Green River Formation is grass-poor. Wyoming big sagebrush dominates the overstory on both substrates.

Volland (1976) provided another example of low-producing sagebrush communities of specific geomorphic settings for the Central Oregon Pumice Zone. Rhyolite was recognized as a major abiotic feature of this community. Although he did not specify subspecies of big sagebrush, Volland (1976) recognized a big sagebrush/needlegrass plant community for which, under good condition, herbage production was approximately168 kg/ha (150 lbs/acre) and bare ground and pavement ranged from 60 to 85 percent. Sagebrush crown cover was limited to 10 to 20 percent.

Sparse understory is indicated for some areas of the Wyoming big sagebrush belt by early explorers of the West. One of the examples cited by Peters and Bunting (1994) is from John C. Fremont and illustrates this point. The report, made in 1843, includes a note made near the mouth of Goose Creek on the Snake River Plain in which Fremont reported "the country has a barren appearance, sandy, and densely covered with the artemisias from the banks of the river to the foot of the mountains." Several days later, near the present site of Twin Falls, Fremont added "there was no grass here, the soil among the sage being entirely naked."

Current relic sites also indicate low potential for diversity and production. In a study of a relic site that included big sagebrush communities on Fishtail Mesa in northern Arizona, Rowlands and Brian (2001) found crown cover of big sagebrush and muttongrass averaged 19.2 and 2.6 percent, respectively. All other species including forbs contributed less than 1 percent cover each, and comparison of relative cover with absolute cover in the Rowlands and Brian (2001) data indicates total absolute cover of all other species was only 2.4 percent. They listed only 14 forbs for their sagebrush sites. Eight of the 14 forbs were found in only 12 percent of their sample sites. Number of forbs/sample site varied from 1 to 6 with an average of 3.75 forbs/site. The sample size in this study was a line intercept 121.9 m long. Although they did not specify subspecies of big sagebrush, their site seems to fit within the climatic zone for Wyoming big sagebrush for northern Arizona. That their site was considered a relic without livestock grazing strongly suggests low inherent capability to produce forbs.

The Arizona location is toward the margin of the range of Wyoming big sagebrush, and this could be a factor in the sparse understory found in the relic site of Rowlands and Brian (2001). However, Marquiss and Lang (1959) reported similar conditions for two relic sites in Sweetwater County, Wyoming. On the two relic sites, they found only five forbs with greater than 2 percent frequency. Common forbs included Hoods phlox, cushion buckwheat (*Eriogonum ovalifolium*), and Hooker sandwort (*Arenaria hooker*). Hooker sandwort had 74 percent frequency for the two sites. This cushion plant is an indicator of harsh sites with inherently high percent bare soil and exposed gravel. Marquiss and Lang (1959) also noted forbs were more abundant in adjacent areas grazed by livestock than in the relic sites.

Winward (1983) noted relatively few perennial forbs for Wyoming big sagebrush communities even in undisturbed conditions. Bunting (1985) reported that Wyoming big sagebrush habitat types have low perennial forb components in any successional stage. Although Fisser (1962) found 38 forbs in a study area, the total combined average percent basal area was 2.6, and only Hoods phlox (*Phlox hoodii*) and six other forbs were found with greater than trace amounts. Hoods phlox alone accounted for 69.5 percent of forb cover. A contrast of Wyoming big sagebrush and Vasey big sagebrush communities in the Uinta Mountains indicated 7 and 31 forbs per site, respectively, for these communities (Goodrich and Huber 2001). This and other comparisons are made in table 3.

In a relic site in Oregon, two forbs were found at 5.9 percent and 1.7 percent crown cover. However, Wyoming big sagebrush crown cover was only 5.2 percent at this site (Kindschy 1994). Tueller and Blackburn (1974) defined an overstory/understory inverse relationship in which needle-and-thread grass decreased from 7.5 percent to 1.4 percent basal area as big sagebrush increased from 1.3 percent to 13.5 percent crown cover. Similar overstory/understory relationships have been discussed by a number of workers (Fisser 1986; Rittenhouse and Sneva 1976; Sturges 1983, 1986; Tanaka and Workman 1988; Wambolt and Payne 1986; West and Hassan 1985; Winward 1983). Some Wyoming big sagebrush sites might not have the capability to provide high percent crown cover of herbaceous plants concurrent with high percent crown cover of sagebrush.

A low perennial forb component is often characteristic for salt-desert shrub communities (Goodrich 1986). Proximity of many Wyoming big sagebrush communities to salt-desert shrub communities is also an indicator that a low forb component is inherent to Wyoming big sagebrush communities. As Wyoming big sagebrush communities intergrade into desert shrub communities, it is reasonable to expected fewer forbs, decreasing percent ground cover, and lower production. As they grade toward Vasey big sagebrush communities, it seems reasonable to expect more forbs, higher percent ground cover, and greater production.

The preceding discussion of overstory/understory relationships could be misleading if it is read to imply that a vigorous understory, including forbs, is necessary for Wyoming big sagebrush communities to function as sage-grouse habitat. Sage-grouse populations have been sustained in forb-poor Wyoming big sagebrush and mountain big sagebrush stands in Sweetwater County, Wyoming, Parker Mountain, Utah, and other locations. These populations strongly indicated that the structure and forage provided by big sagebrush is far more important than any other habitat component. Features of Wyoming big sagebrush, including spreading, dense crowns in close proximity to the ground, appear to be major compensating factors for low presence of herbaceous plants. This is not to imply that succulent forbs are not beneficial. However, an abundance of herbaceous plants will not compensate for lack of sagebrush structure and sagebrush forage.

The discussion of overstory/understory relationships is directed to the need for management to be based on inherent capabilities of the land. If expectations exceed the inherent capabilities of Wyoming big sagebrush communities, management decisions will be poorly founded, and habitat improvement or restoration projects will fail to achieve desired results.

Wyoming big sagebrush is commonly 40 to 70 cm tall and sometimes taller. The crowns are often dense and spreading with the lower part of the crown in close proximity to the ground, but not so close as to obstruct sage-grouse nesting. These features indicate high values for nesting and other habitat needs of sage-grouse. Potential for crown cover of many Wyoming big sagebrush communities is within the range indicated by Connelly and others (2000) to meet the needs of sage-grouse. Crown cover of stands of Wyoming big sagebrush often ranges from about 5 to 25 percent. However, the low percent crown cover (5.2 percent) of sagebrush found in a relic site by Kindschy (1994) indicates potential for sagebrush cover can be well below the range recommended by Connelly and others (2000). Other reports of inherently low potential for cover are listed above. Also there are many areas where browsing by wild ungulates, including pronghorn antelope (*Antilocapra americana*), keeps crown cover of Wyoming big sagebrush well below the levels most preferred by sage-grouse.

Due to its growth form, height, widespread distribution, and expansive areas covered, Wyoming big sagebrush is perhaps the most important plant in the ecology of sage-grouse. Perhaps no other factor is more important to declining populations of sage-grouse than are the changes that have taken place in the Wyoming big sagebrush system. Substantial areas have been converted to agriculture. Others have fallen to urban expansion. However, more extensive loss of sagebrush cover is associated with the introduction of annual grasses. In the past few decades, millions of hectares of Wyoming big sagebrush have been converted to cheatgrass and other annuals (Billings 1994; Monsen 1994).

Monsen (1994) reported over 1.3 million ha burned between 1979 and 1993 in southern Idaho with much of this in an ecological setting where cheatgrass is highly competitive. A table of acres burned from 1979 to 1993 in southern Idaho (Monsen 1994) shows area burned varying widely from year to year without an upward trend. In 1999 and 2000, over 121,410 hectares (300,000 acres) burned on lands managed by the Bureau of Land Management in Idaho. This indicates large acreages of annual burning will continue. In 1999, about 68,800 hectares (1.7 million acres) burned in Nevada. Much of this burning was in sagebrush types that are highly vulnerable to cheatgrass dominance. The 1999 fire season in Nevada represents about one-half of the burned acreage reported by Monsen (1994) for a 14-year period in southern Idaho. Perhaps the 1999 season in Nevada will prove to be a rare exception. However, it could also be an indication of occasional to common fire seasons of the future. Many of the acres included in annual reports also burned in previous fires of recent years. The influence of fire is not only size, it is also frequency. High fire frequency sustains cheatgrass communities and prevents the return of sagebrush.

Cheatgrass with its associated fire regime becomes the driver of the system (Billings 1994; Peters and Bunting 1994). This has reduced diversity and eliminated structure and forage required by sage-grouse.

The displacement of Wyoming big sagebrush by cheatgrass and its associated fire system over millions of hectares is likely the most important factor in the future of sage-grouse in many parts of the West. Reclamation of lands under a system driven by cheatgrass and fire will be important to the future of sage-grouse and other sagebrush associated species.

Due to low precipitation inherent in much of the Wyoming big sagebrush system, reclamation will be difficult with limited plant materials likely to succeed in reclamation projects. Hull (1974) evaluated performance of 90 plant taxa, including many natives, in seedings on rangelands of southern Idaho. Where annual precipitation was less than 25 cm, only 17 of the 90 taxa rated over 1 on a relative scale of 1 to 10. Of these 17, only 6 were natives, and none of the natives rated over 2.1. Many natives failed to show any establishment. Although the work of Hull (1974) is over 25 years old, it clearly indicates the difficulty in reclamation of Wyoming big sagebrush communities. The difficulty inherent to the low precipitation of many of these communities is compounded by the highly competitive capability of cheatgrass. Difficulty of reclamation with a diversity of native plants was acutely demonstrated by the work of Hull (1974). Later review of restoration of areas similar to those studied by Hull (1974) supports Hulls' work. Monsen (1994) indicated low potential for successful seedings in areas with less than 250 mm of annual rainfall. To be successful in breaking the high fire frequency associated with cheatgrass, plant materials used in restoration projects must be able to compete with and reduce cheatgrass, and they will be most useful if they have comparatively low propensity to carry fire (Monsen 1994).

"Hands-off" or passive management is sometimes recommended with an apparent concept that left alone, cheatgrass-dominated areas will revert to native sagebrush/grass steppe. However, this concept is not supported by studies in relic or natural areas. Kindschy (1994) reported the presence and increase of cheatgrass in southeastern Oregon's Jordan Crater Research Natural Area that has been protected from human activities including livestock grazing. In Red Canyon of the Green River, cheatgrass has been found as the most frequent species where livestock use and other disturbance have been minimal (Goodrich and Gale 1999). Young and Tipton (1990) cited two works from southeastern Washington that documented observations of cheatgrass successfully inserting itself into climax perennial grass/shrub communities that had been protected from fire and grazing for as long as 50 years. They proposed that the idea of cheatgrass spreading in a biological vacuum created by excessive grazing may be somewhat misleading or overstated.

Young and Allen (1997) have emphasized that site degradation is not necessary for cheatgrass invasion. In western Utah, Harper and others (1996) found cheatgrass able to establish in ungrazed areas in desert shrub communities where, although native perennials were able to greatly suppress the size of cheatgrass plants, cheatgrass was able to maintain a presence by which it could expand upon disturbance, including gopher mounds.

Austin and others (1986) found cheatgrass present in Red Butte Canyon of the Wasatch Mountains where livestock grazing was discontinued in 1905, likely before cheatgrass reached that area. They reported higher cover values for cheatgrass in 1983 than for 1935 in Red Butte Canyon. The Red Butte Canyon study demonstrates increase of cheatgrass in absence of livestock grazing.

Knight (1994) reported that the cheatgrass problem is not restricted to land managed for livestock, and he gave an example of an increase of cheatgrass following fire in Little Bighorn Battlefield National Monument in southern Montana. He suggested that managing vegetation of a National Monument to reflect presettlement conditions is a goal that may be impossible to attain once certain introduced species become established.

A hands-off approach does not seem to address the capability of cheatgrass. As suggested by Peters and Bunting (1994), the introduction of annuals, including cheatgrass, possibly has been the most important event in the natural history of the Snake River Plain of Idaho since the last glacial period. The system has changed. A hands-off approach that anticipates the return of presettlement ecosystems does not seem to include the reality of the change. Restoration of sagebrush-grass communities replaced or displaced by cheatgrass and other introduced plants is difficult. No satisfactory method for controlling weed competition on large-scale sagebrush restoration projects has been developed (Meyer 1994). Although current technology of dealing with cheatgrass probably has a long way to go, the future of much of the Wyoming big sagebrush system will likely be determined by development and application of restoration technology. This technology is a focus of these proceedings.

Restoration will include expectations of production, diversity, and potential cover of shrubs and other species. These expectations can be expected to be associated with costly disappointments and poorly based management decisions if inherent as well as induced features of specific sites are not considered.

Artemisia tridentata ssp. *xericensis* Foothills Big Sagebrush, Xeric Big Sagebrush

Foothills big sagebrush is known from west central Idaho where it grows on mesic temperature and xeric moisture rated soils derived from Columbia River basalt at elevations between 752 and 1,524 m (Rosentreter and Kelsey 1991). Morphology and ecology of foothills big sagebrush suggest a hybrid origin involving basin big sagebrush and Vasey big sagebrush (Rosentreter and Kelsey 1991).

Classification

Artemisia tridentata ssp. *xericensis/Elymus spicatus* (Hironaka and others 1983)
Artemisia tridentata ssp. *xericensis/Festuca idahoensis* (Hironaka and others 1983)

Habitat and Capabilities

The habitat and growth form, as indicated above, are intermediate between basin big sagebrush and Vasey big sagebrush. Moderate to high values for sage-grouse for nesting, breeding, and wintering are indicated. Due to drying of herbaceous plants, values for brood rearing in mid and late summer are likely lower than in higher montane stands of Vasey big sagebrush. However, brood rearing is common in big sagebrush areas equally as dry or drier than found in the range of this taxon.

Artemisia tripartita ssp. *rupicola* Wyoming Threetip Sagebrush

This shrub is known from east of the Continental Divide in Wyoming on the Owl Creek Range, Wind River Range, and the Laramie Range where its elevational range of dominance is between 1,768 and 2,560 m (Fisser 1962) and south-central Montana (Wambolt and Frisina 2002).

Classification

Artemisia tripartita ssp. *rupicola*/*Festuca idahoensis* (Johnston [1987] listed references under *Artemisia tripartita* for the Medicine Bow National Forest, Roosevelt National Forest, and Shoshone National Forest. These National Forests are within the range of *A. t. rupicola*.)

Artemisia tripartita ssp. *rupicola*/*Poa secunda* (Johnston [1987] listed two references for this type for the Shoshone National Forest under *Artemisia tripartita*. This National Forest is within the range of *A. t. rupicola*.)

Artemisia tripartita ssp. *rupicola*/*Stipa comata* (Johnston [1987] listed two references for the Medicine Bow National Forest under *A. tripartita*. This National Forest is within the range of *A. t. rupicola.*)

Thatcher (1959) found slimstem muhly (*Muhlenbergia filiculmis*) associated with *A. tripartita ssp. rupicola,* but not with any other sagebrush taxa in his study area in Albany County, Wyoming.

Habitat and Capabilities

This shrub is found on windswept, shallow, gravelly soils of foothills and mountains and flats with estimated average annual precipitation of between 38 and 48 cm (Thatcher 1959). It is associated with a minimum of winter snow accumulation and is generally not present on sites where snow tends to drift (Fisser 1962). In addition to the associated grasses listed above under classification, this shrub is found with blue gramma and fringed sagebrush (Knight 1994).

Height of this somewhat sprawling shrub, excluding the flowering stems, is seldom as much as 15 cm (Dorn 1992) and not over 18 cm (Thatcher 1959). Percent crown cover is generally between 4 and 12, and other shrubs are commonly of minor cover (Fisser 1962). Fisser (1962) found Hood's phlox and three other forbs provided most of the forb cover in Wyoming threetip sagebrush communities.

Height and crown cover common to this plant indicate little value for sage-grouse at any time of the year except when stands of this shrub are adjacent to taller taxa of sagebrush. Shultz (1986) indicated this shrub is usually on barren knolls surrounded by well-developed grasslands. Proximity to big sagebrush is comparatively uncommon. However, Fisser (1962) found some stands adjacent to big sagebrush stands. Where adjacent to stands of big sagebrush, stands of Wyoming threetip sagebrush might be selected for strutting grounds, and the windswept habitat might indicate some forage value in winter. However, since the leaves are deciduous, winter value of this shrub is very low. The sprouting feature indicates high value for reclamation projects.

Artemisia tripartita ssp. *tripartita* Idaho Threetip Sagebrush, Tall Threetip Sagebrush

This deciduous, sprouting shrub ranges from Washington and adjacent British Columbia south to Elko County, Nevada, Box Elder and Cache Counties, Utah, and to Teton County, Wyoming, and the southwestern portion of Montana (Wambolt and Frisina 2002). Centers of distribution are in western Washington and east-central Idaho. Much of the area once covered by threetip sagebrush has been converted to agricultural land (Schultz 1986).

Classification

Artemisia tripartita ssp. *tripartita*/*Elymus spicatus* (Hironaka and others 1983)

Artemisia tripartita ssp. *tripartita*/*Festuca idahoensis* (Hironaka and others 1983; Mueggler and Stewart 1980; Johnston [1987] listed six other references that are from within the range of *A. t. tripartita.*)

Artemisia tripartita ssp. *tripartita*/*Stipa comata* (Hironaka and others 1983)

Habitat and Capabilities

Idaho threetip sagebrush occurs primarily on fertile volcanic soils (Shultz 1986) that class in a variety of Mollisols or intergrade with Mollisols (Hironaka and others 1983) on dry plains and hills at 1,100 to 2,300 m (Cronquist 1994).

Height of Idaho threetip sagebrush is 20 to 100 cm, and the plant has high capacity to form closed stands with high percent crown cover. The herbaceous understory can be fairly productive, but it is sometimes inversely related to crown cover of the shrubs (Hironaka and others 1983).

The features of Idaho threetip sagebrush communities indicate high-value habitat for sage-grouse nesting, breeding, and brood rearing. However, due to the deciduous feature of the shrub, it can be expected to provide little, if any, forage in winter.

Hironaka and others (1983) noted cheatgrass is generally not a problem in this type even though the vegetation may be severely disturbed. The sprouting feature of the shrub indicates high value for reclamation projects where ability to recover from fire is important.

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Tapertip Hawksbeard

Seasonal Habitat Requirements for Sage-Grouse: Spring, Summer, Fall, and Winter

Clait E. Braun
John W. Connelly
Michael A. Schroeder

Abstract—Sage-grouse (*Centrocercus minimus, C. urophasianus*) are dependent upon live sagebrush (*Artemisia* spp.) for all life processes across their entire range. This paper describes habitats used by sage-grouse as documented in the scientific literature. The leaves of sagebrush are eaten by sage-grouse throughout the entire year and comprise 99 percent of their winter diets. Spring (late March through May) habitats are those with intermixed areas of taller (40 to 80 cm) sagebrush with canopy cover of 15 to 25 percent and taller (>18 cm) grass/forb cover of at least 15 percent. Sites used for display have shorter vegetation, frequently few or only short sagebrush plants, but with taller, more robust sagebrush within 100 to 200 m that is used for escape cover. Nesting cover mimics that used overall during spring but with clumps of tall (>50 cm), dense (about 25 percent) live sagebrush and abundant forbs (>10 to 12 percent cover). Early brood rearing areas are those within 200 m (initial 3 to 7 days posthatch) to 1 km (up to 3 to 4 weeks posthatch) of nest sites. Forbs and taller (>18 cm) grasses are important for broods; forbs provide succulent foods, grasses provide hiding cover, and the grass/forb mixture supports insects used by chicks. Summer use areas are those with abundant succulent forbs with live, taller (>40 cm), and robust (10 to 25 percent canopy cover) sagebrush useful for cover. These areas continue to be used into fall when sage-grouse move to higher benches/ridges where they forage on remaining succulent forbs such as buckwheat (*Eriogonum* spp.) and switch to more use of sagebrush leaves. Winter (early December to mid-March) use areas are often on windswept ridges, and south to southwest aspect slopes as well as draws with tall, robust live sagebrush. Height (25 to 35 cm) of sagebrush above the surface of the snow in areas used in winter is important, as is canopy cover (10 to 30 percent). Management of habitats used by sage-grouse should initially focus on maintaining all present use areas. Practices to enhance sagebrush habitats to benefit sage-grouse are reviewed, as is the need to annually monitor sage-grouse numbers along with systematic monitoring of the health of sagebrush ecosystems.

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Introduction

Sage-grouse (*Centrocercus minimus, C. urophasianus*) historically occurred in at least 16 States and three Canadian Provinces (Aldrich 1963; American Ornithologists' Union 1957; Johnsgard 1973). They have been extirpated in five States and one Canadian Province (Braun 1998; Connelly and Braun 1997) and their overall distribution has become discontinuous (fig. 1). The changes in sage-grouse distribution have been attributed to loss, fragmentation, and degradation of habitats (Braun 1995, 1998; Connelly and Braun 1997), and it is probable that at least one-half of the original occupied area can no longer support sage-grouse (Braun 1998). Because of the reduced amount of available habitat, sage-grouse abundance has also markedly decreased with reported declines of 10 to 51 percent (Connelly and Braun 1997) and as much as 45 to 82 percent since 1980 (Braun 1998). The known decreases in distribution and abundance have led to concern about stability of sage-grouse populations and the health of sagebrush ecosystems upon which they depend. Petitions to list sage-grouse under the Federal Endangered Species Act have been filed for northern sage-grouse (*C. urophasianus*) and for Gunnison sage-grouse (*C. minimus*).

Sage-grouse are dependent upon ecosystems with vast and relatively continuous expanses of live, robust, taller sagebrushes (*Artemisia* spp.) with a strong grass and forb component. This dependency upon sagebrush, especially the subspecies of big sagebrush (*A. tridentata vaseyana, A. t. wyomingensis, A. t. tridentata*), low sagebrush (*A. arbuscula*), black sagebrush (A. *nova*), silver sagebrush (*A. cana*), and three-tip sagebrush (*A. tripartita*), as well as a variety of less apparent and abundant species, has been well documented (Patterson 1952; reviews by Braun and others 1977 and Connelly and others 2000a). Since the early 1960s, the sage-grouse/sagebrush relationship has focused attention by Western States and Provinces on the need to maintain healthy sagebrush-steppe communities over large expanses. Guidelines for maintenance of sage-grouse habitats were developed from the scientific literature (Braun and others 1977, completely revised by Connelly and others 2000a) and promoted by the Western States Sage-Grouse Technical Committee. The purpose of this paper is to present an overview of the habitat needs of sage-grouse based on the scientific literature, identify the issues that affect maintainance of useful habitats for sage-grouse, and discuss management strategies to maintain, enhance, and restore habitats

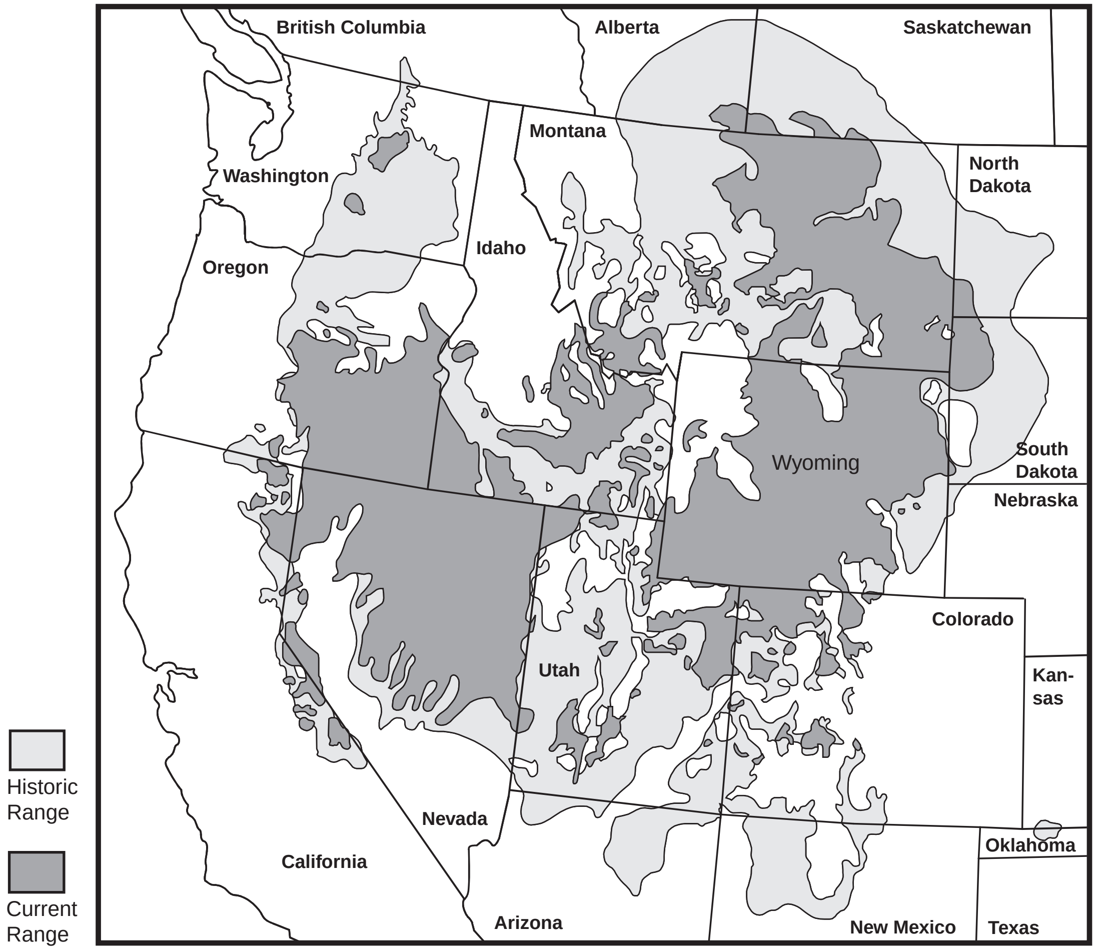

Figure 1—Historic and current distribution of sage-grouse (map prepared by M. A. Schroeder).

for sage-grouse. This paper draws extensively on the published *Guidelines to Manage Sage Grouse Populations and Their Habitats* (Connelly and others 2000a).

Habitat Overview

Spring

Timing of spring breeding activities of sage-grouse is dependent on elevation and amount of persistent snow cover. Attendance at leks may start in early to mid-March or, at higher elevations, in early April. Males may attend and display at leks until late May but most display and mating activities are greatly reduced by mid-May. Amount and depth of snow cover greatly influence sage-grouse breeding activities; thus, snow-free areas are important components of spring habitat. Habitats used by sage-grouse during the breeding period are those associated with foraging, leks, escape, and nesting. Depending upon moisture regimes, height of sagebrush in used habitats varies from 30 to 80 cm with canopy cover from 15 to 25 percent (Connelly and others 2000a). Lek sites typically have low amounts of sagebrush and appear relatively bare, but they may have extensive cover of low grasses and forbs. Taller, robust live sagebrush used as escape cover is normally within 100 to 200 m of active leks. The average distance from a nest to the nearest lek varies from 1.1 to 6.2 km, and the actual size of the breeding habitat appears largely dependent on the migratory characteristics of the sage-grouse population as well as distribution of sagebrush cover with respect to lek location (Connelly and others 2000a). Habitats selected for nesting are those with abundant (15 to 30 percent canopy cover) live, taller (30 to 80 cm) sagebrush plants within a community with >15 percent ground cover of taller (40 to 80 cm) grasses and forbs (Connelly and others 2000a). Early brood-rearing habitats (fig. 2) are normally those within 100 m to 1 km of nesting sites, especially areas with high plant species richness, moisture, and taller grasses and forbs (Connelly and others 2000a). Adult sage-grouse, while still foraging extensively on leaves of live sagebrush, eat leaves and flower parts of forbs during spring, as do chicks (Apa 1998; Drut and others 1994; Dunn and Braun 1986; Klott and Lindzey 1990).

Summer

Habitats used by sage-grouse in summer (early to mid-June to mid to late September) are those that provide

Figure 2—Sage-grouse brood hen in good quality Wyoming big sagebrush habitat, North Park, Colorado (photograph by C. E. Braun).

adequate forage, especially succulent forbs, and cover useful for escape. These habitats may include those used for agriculture, especially for native and cultivated hay production, edges of bean and potato fields, as well as more typical sagebrush uplands and moist drainages. Taller (>40 cm) and robust (10 to 25 percent canopy cover) sagebrush is needed for loafing and escape cover as well as a source of food. Grass and forb ground cover can exceed 60 percent (hayfields). Provided moisture is available through water catchments or from succulent foliage, sage-grouse may be widely dispersed over a variety of habitats during this period (Connelly and others 2000a). As late summer approaches, there is movement from lower sites to benches and ridges (fig. 3) where sage-grouse forage extensively on leaves of sagebrush.

Fall

Fall (late September into early December) is a time of change for sage-grouse from being in groups of hens with chicks or males and unsuccessful brood hens to separation into larger flocks frequently segregated by gender. Some birds may continue to use lower riparian or hayfield habitats, but there is movement onto higher, frequently north-aspect slopes where succulent native forbs, such as buckwheats, provide green forage. Use of sagebrush leaves for food becomes more common as does use of extensive stands (>20 percent canopy cover) of taller (>25 cm), live sagebrush (Connelly and others 2000a). Movements can be slow but there is a general shift toward traditional winter use areas (Connelly and others 1988).

Winter

Flocks of sage-grouse are somewhat nomadic in early winter but may remain within chosen areas for periods of several weeks or more depending upon extent of snow cover and depth (Beck 1977; Hupp and Braun 1989b). Sagebrush height (>20 cm, but usually >30 cm, above the surface of the snow) is important as is the robust (>10 to 30 percent canopy cover) structure of live sagebrush (Connelly and others 2000a). Sage-grouse use a variety of sites in winter including windswept ridges with open (10 to 20 percent canopy cover) (fig. 4) stands of sagebrush to draws with dense (>25 percent canopy cover) stands. Quality of the snow can be important because sage-grouse are known to use snow roosts and burrows (Back and others 1987). Aspect is also important with south and southwest slopes most used in hilly terrain (Hupp and Braun 1989b). Leaves of live, vigorous sagebrush plants provide >99 percent of the foods eaten during the winter period (early December until early to mid-March) (Patterson 1952; Remington and Braun 1985; Wallestad and others 1975). Generally, winter is a time of body mass gain (Beck and Braun 1978), although severe winter conditions over prolonged intervals can reduce the amount of area available for foraging and cover (Beck 1977) and thus affect body condition (Hupp and Braun 1989a). Overall movement during winter may be extensive and home ranges can be large (Connelly and others 2000a). As winter wanes, flocks of sage-grouse move toward breeding areas that may be immediately adjacent to or far distant from winter use areas (Connelly and others 2000a).

Figure 3—Radio-tracking sage-grouse in high-elevation summer range with a stand of mountain big sagebrush in the background (photograph by J. W. Connelly).

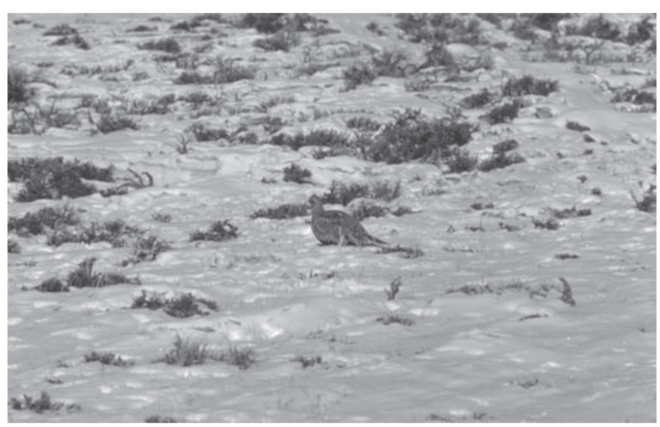

Figure 4—Sage-grouse winter range in Wyoming big sagebrush habitat in North Park, Colorado (photograph by C. E. Braun).

Issues

Decreases in distribution and abundance of sage-grouse have been ascribed to a complexity of factors (Braun 1987, 1998; Connelly and Braun 1997). The three major causes, (1) habitat loss (mostly permanent), (2) fragmentation (frequently permanent but reversible at times), and (3) degradation (usually can be corrected), are generally accepted but the latter two are poorly recognized and understood. Examples of permanent habitat loss include conversion of sagebrush rangelands to agricultural crops, town and subdivision developments, placement of power plants or surface mines, and reservoir construction. Fragmentation of habitats occurs with power lines, paved and other high-speed road development (including maintenance and improvement of farm roads), habitat-type conversion projects, fire, or any permanent development that reduces the size of existing habitat patches. Less understood are the impacts of fences, seasonal use trails, oil and gas wells with surface pipelines, noise, and so on. Some of these impacts can be resolved and sage-grouse will reoccupy some formerly disturbed areas (Braun 1987).

Distribution of habitat types useful to sage-grouse is also important, as these species are habitat specialists using a variety of areas within a larger landscape mosaic. Thus, not only is the quantity of sagebrush habitats important, but also the juxtaposition and quality of those habitats. All sagebrush habitats are not equal in their acceptability to sage-grouse, and location of areas used may affect sage-grouse distribution. Size of habitat patches is important and larger (>30 km^2) is better than smaller, although the spatial relationships of habitats for sage-grouse are not well understood. Sage-grouse use a mosaic of habitats that is normally present in sagebrush-steppe because of differences in soils, moisture, topography, aspect, insect defoliation, wildfires, and other factors. Sagebrush naturally regenerates as overmature plants die and seedlings become established. Use of the term "decadent" for sagebrush is generally inappropriate because it implies that sagebrush communities are not dynamic with a variety of age classes from seedlings to overmature. Since most sagebrush communities are resilient and represent a continuum of age classes within a mosaic of habitats, creation of "edge" to benefit sage-grouse is rarely needed. Because of human activities, the presence of too much edge (especially in straight lines) is more common than too little edge and results in degradation of sage-grouse habitats.

Sagebrush ecosystems have been managed through a variety of treatments from domestic livestock grazing, mechanical and chemical clearing or thinning, to use of prescribed fire (Braun 1998). Fire was a natural event in more mesic sagebrush communities but was infrequent as demonstrated by the lack of resprouting of big sagebrush, black sagebrush, and low sagebrush. Fire was more common in areas with three-tip sagebrush and silver sagebrush because both species resprout. Recent research suggests there is little gain in forage production of grasses and forbs after fire, because it can take longer than 30 years to return to preburn conditions (Wambolt and others 2001).

Treatments of sagebrush communities have primarily been conducted to benefit another treatment (livestock grazing). Use of some treatments has led to plantings of exotic grasses, invasion of areas by exotic plants, conifer invasion of sagebrush habitats, and increased fire frequency. Many, if not most, of these treatments have been applied to improve rangelands for domestic livestock but have had negative impacts on sagebrush communities and animals dependent on them (Braun and others 1976). Further, successive treatments have been applied to landscapes with little understanding of the cumulative effects that may impact both sagebrush-dependent animals, such as sage-grouse, and the overall health of the plant community. The impacts of natural events such as periodic drought are further exacerbated by human treatments of sagebrush communities. All of these issues emphasize the need for active protection of habitats presently used by sage-grouse as well as restoration of habitats that formerly supported sage-grouse populations.

Sage-Grouse Habitat Management Strategies

The objectives of habitat management to benefit sage-grouse, in order of importance, should be (1) to protect and maintain existing occupied habitats, (2) enhance existing occupied habitats, (3) restore degraded habitats that still receive some sage-grouse use, and (4) rehabilitate significantly altered habitats that no longer support sage-grouse. Strategies to accomplish these objectives should include:

- Vigorous suppression of wildfire.
- Reconsideration of any use of prescribed fire.
- Proper livestock management (including reconsideration of time of grazing, stocking rates, season of use, and frequency of use).
- Use of nitrogen fertilizer, except in areas infested by annual weeds.
- Mechanical chopping of sagebrush.
- Fence type and placement.
- Water management.
- Rehabilitation and restoration techniques discussed in these proceedings.

At times, manipulation of some occupied sage-grouse habitat may be necessary to enhance the overall quality of a seasonal range. An example would be removing or reducing some sagebrush canopy cover in known breeding habitat to enhance a depleted understory. Removal of 57 percent of sagebrush cover resulted in a significant decline in a sage-grouse breeding population (Connelly and others 2000b) and degradation of early brood-rearing habitat (Fischer and others 1996). More recently, a wildfire that removed about 30 percent of the sagebrush cover in a breeding habitat resulted in a 60 percent decline in sage-grouse nest success (Connelly, unpublished data, 1998). Because of this information and the fact that wildfires, drought, and insect infestations cannot be predicted, any sagebrush removal efforts should affect a relatively small portion of the occupied habitat. Connelly and others (2000a) suggested that >80 percent of breeding and winter habitat with vegetative characteristics necessary for productive sage-grouse habitat should remain intact to adequately provide for the needs of sage-grouse. However, an even greater percentage should be protected if sage-grouse populations are declining or the population status is unknown. All proposed habitat manipulations should carefully consider the current condition of habitat, status of the sage-grouse population, and likely outcome of the vegetation treatment, including recovery time necessary for the area to again provide adequate habitat for sage-grouse nesting and early brood rearing.

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Fernleaf biscuitroot

Plant Succession and Approaches to Community Restoration

Bruce A. Roundy

Abstract—The processes of vegetation change over time, or plant succession, are also the processes involved in plant community restoration. Restoration efforts attempt to use designed disturbance, seedbed preparation and sowing methods, and selection of adapted and compatible native plant materials to enhance ecological function. The large scale of wildfires and weed invasion requires large-scale approaches to restoration. Practices and equipment from traditional rangeland revegetation are being adapted to establish diverse, native communities. The challenge is to meet the establishment requirements of different species and to create weed-resistant plant communities.

Introduction

In the past, range scientists developed range improvement techniques directed mainly at controlling unpalatable woody species and establishing forage grasses for livestock and erosion control, but also to establish plants critical for big game habitat (Roundy 1996). Our goals now are to restore functional, diverse native plant communities. The processes of restoration are the same processes that operate in plant succession. Understanding these processes can help us develop realistic techniques and goals for large-scale restoration. I will briefly review concepts and processes of plant succession and discuss associated aspects of community restoration.

Models of Succession

Plant succession is the change in vegetation that occurs over time after fire, heavy grazing, flooding, or other natural or human-related disturbances. Secondary succession occurs when the land retains some residual soil and biological components from the plant and animal communities that existed before the disturbance (Barbour and others 1998). Primary succession occurs on new substrates, such as on a newly formed volcanic island. Two major views of this process were taught by Clements (1916) and Gleason (1926). In Clement's model, vegetation changes from pioneer species through a series of predictable communities or seres, which replace each other in order until a final stable or climax community dominates the site. This model is said to be linear (always follows the same order) and deterministic (is predictable). On the other hand, Gleason suggested that vegetation change after disturbance was a function of the kinds of plants involved and their characteristics relative to the disturbance. More recently, ecologists have recognized that features of both models may describe what actually happens. State and transition models that allow for multiple steady states of vegetation, with different probabilities of transition or change from one state to another (Westoby and others 1989), have been proposed. These models work better with the current recognition that some disturbances, such as fire, have a natural frequency and play a major role in shaping many upland plant communities. Similarly, seasonal flooding shapes riparian communities (Middleton 1999).

Clements (1928) identified the processes of succession as nudation (disturbance), migration (movement of new seeds or other plant propagules to the site), ecesis (plant establishment), interaction (sorting out of species that establish), reaction (the effects of the successful species on the environment), and stability. These processes correspond to revegetation and restoration principles and practices (table 1). Although all of these successional processes may be active in most systems, some are more controlling for some systems

Table 1—Plant successional processes that correspond to restoration and revegetation principles and practices.

Process	Principle or practice
Disturbance	Site potential after disturbance, designed disturbance to control undesirable plants
Dispersal, migration, residuals	Sowing sufficient germinable seed, preempting resources from or controlling residual propagules of undesirable species, renovation to maintain or stimulate residual desirable species
Establishment	Seedbed preparation and sowing to maximize germination and seedling establishment; selecting adapted plant materials
Interaction/reaction	Selecting ecologically functional, compatible plant materials for mixed communities that are weed resistant
Stabilization	Restoring disturbance regimes and management strategies that favor ecological function

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and biomes, while others are more controlling for other biomes (Chambers and others 1992). For example, historically, fire is a major controlling disturbance in grasslands, but not arid deserts, where drought is most operative. Interaction (especially competition) and reaction are particularly operative in forest systems, but not as operative in tundra or desert systems where the harsh environment may result in fewer highly adapted species.

The state and transition model has been used to describe successional processes in the sagebrush system (Westoby and others 1989). Continued heavy spring grazing moves a sagebrush/bunchgrass system to one dominated mainly by big sagebrush (*Artemisia tridentata*). Introduction of annual weeds like cheatgrass (*Bromus tectorum*), and the attendant increased fire frequency holds this system in cheatgrass dominance unless major inputs in weed control and revegetation move it to another state. The cheatgrass-dominated system has little likelihood of transitioning to another state on its own because the rapidly maturing cheatgrass provides a fine and well-distributed fuel over a long fire season. It is adapted to establishment after fire, while other species cannot survive the 3- to 5-year frequency that can occur; so the system is stuck in a stable, but not highly desirable state. Reasons that systems can be held in a stable steady state include (1) frequent or severe disturbances, such as fire, or heavy, continuous grazing; (2) establishment inertia, or lack of establishment associated with harsh environmental conditions, such as in arid deserts where establishment occurs only in unusually wet years; or (3) competitive exclusion where shrubs or trees that are highly competitive, such as pinyon and juniper (*Pinus edulis, P. monophylla* and *Juniperus osteosperma, J. occidentalis*), eventually dominate in the absence of disturbance such as fire. Another example of the latter reason is the replacement of aspen (*Populus_tremuloides*) by conifers in the absence of fire.

Changes in the disturbance regime interact with the environmental context of a plant community to result in transitions or the lack of transitions to other states. For example, Harper (1985) provides evidence that could be used to suggest that lack of fire on acidic soils, such as those in the Uinta Mountains, may result in conifer replacement of aspen much sooner than on the calcareous limestone soils of the Wasatch Mountains, Utah. Damming of rivers and streams controls spring flooding, a process that is necessary for dispersal and establishment of cottonwood (*Populus*) and other riparian species (Middleton 1999; Stromberg and others 1991). This flooding is also essential for the erosion, deposition, and sediment transport functions of the stream that result in the natural geomorphologic features that are necessary to support riparian plant communities.

Lack of fire in communities once dominated by sagebrush and bunchgrass can result in invasion and dominance by pinyon and juniper (Tausch 1999). On deeper alluvial or more fertile soils, tree canopies expand until they touch while the understory vegetation and seed bank die out. These communities are then susceptible to catastrophic crown wildfires and subsequent weed invasion. On shallower soils invaded by pinyon and juniper, resources are too limited for tree canopies to touch, but loss of understory vegetation and subsequent wind and water erosion of interspaces may still degrade the site (Roundy and Vernon 1999).

Transitions to states of much reduced biotic and physical function are said to have passed a biotic or physical threshold, after which a transition back to the previous state is highly unlikely without major intervention (Whisenant 1999). Such thresholds are called irreversible because natural processes alone are insufficient to move them back to the prethreshold state. Susceptibility to such thresholds depends on the environmental context and past management of the site and plant community. For example, sagebrush communities that lack a good understory of perennial grasses pass a biotic threshold into cheatgrass dominance after fire more easily than those with a good perennial grass understory that survives the fire. Invasion and dominance of pinyon and juniper on a site of high erosion potential (higher slopes, finer textured soils, and more frequent occurrence of intense summer thundershowers) may result in major erosion and passing of a physical threshold, while such invasion on sites of low erosion potential may not (Davenport and others 1998).

Management to avoid passing biotic and physical thresholds is much preferred and less costly than attempting restoration after passing these thresholds. For example, use of fire or mechanical treatments to control pinyon and juniper before the understory vegetation or soil is lost, or grazing management to maintain a good perennial understory in sagebrush communities is less costly and risky than attempts to restore these communities after they have passed degrading thresholds. Nevertheless, many of our landscapes have already passed such thresholds and now require major intervention. Restoration after crossing a biotic threshold requires control of the dominating vegetation and revegetation with more ecologically functional and desirable species. Restoration to some historic, natural plant community after passing a physical threshold may not be possible at all, requiring that we set our goals as restoration of ecological function, rather than historic composition. For example, our goals on an eroded site might be to establish a persistent perennial plant cover to hold the remaining soil in place, rather than risk additional soil erosion by attempting to establish a diverse, native community that may no longer be adapted to the degraded conditions of the site. On riparian areas, biotic restoration may be very difficult without restoring the physical disturbance regime or seasonal flooding that drives the biotic responses.

Environmental Context of the Sagebrush Systems

The environmental context of both succession and restoration efforts has an overriding influence on the outcome. Every restoration project requires characterization of the site in order to determine potential for success, species selection, and seeding methods. Two major sagebrush systems are recognized across the Western United States (West 1983a,b). Sagebrush steppe is north of the drier Great Basin and Colorado Plateau sagebrush systems and has more potential for sagebrush renovation and revegetation success than those drier systems to the south. Mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) communities with higher precipitation have more potential for revegetation than the lower Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) communities. Black sagebrush (*Artemisia nova*) communities on shallow soils are more difficult to revegetate than many Wyoming big sagebrush sites. Higher elevation pinyon-juniper communities have higher precipitation than lower elevation Wyoming big sagebrush communities, and are therefore not only easier to successfully revegetate with desirable species, but may also be more susceptible to invasion of more mesic weeds such as the knapweeds (*Centaurea* spp.). Salt-desert shrub communities are especially hard to revegetate due to low precipitation and fine-textured, saline-alkaline soils that can flow and crust after disturbance and wetting. In general, direct-seeding revegetation is risky with annual precipitation <250 mm, and much less risky at >400 mm. Between these ranges, soils, species selection, seedbed preparation and sowing methods, but especially precipitation during and following the year of seeding greatly affect success.

Designed Disturbance: Residuals

Just as natural disturbances, such as fire and flooding, free up resources for new colonization, revegetation or restoration requires a designed disturbance to reduce undesirable plant populations prior to planting (Sheley and others 1996). Methods of undesirable plant control include biological, mechanical, chemical, and fire. Biological control works best when used in a program of integrated weed control that employs other methods to greatly reduce weed populations. Concurrent or subsequent biological control can then work well to contain weeds. Herbicides have the potential for greatest control of specific undesirables. Mechanical methods have the versatility of configuring the control across the landscape in patterns to maximize wildlife benefits by providing cover, edge, and vegetation linkages where desired. Mechanical methods have less risk of treating nontarget areas than do fire, which can get away, or herbicides, which may drift or move with soil in wind or water erosion. Smooth, Ely (railroad rails attached perpendicular to the chain), or Sagar (rails attached parallel to the chain) anchor chains that produce less to more soil disturbance can be chosen to control large areas of nonsprouting pinyon and juniper trees. Chaining after fire and broadcast seeding helps cover seeds. Broadcast seeding without chaining can result in weed dominance (Ott and others 2003).

The goal of designed disturbance may be to retain components of the original community. For example, sagebrush communities may be treated to rejuvenate older shrubs and control enough of them to establish a more diverse and productive herbaceous understory. Chaining and one-way harrowing may kill about half of the sagebrush in a stand treated for renovation, while discing or two-way harrowing will kill 70 to 90 percent of the sagebrush (Summers and others 2003). This designed disturbance for renovation is equivalent to the successional importance of residuals after a disturbance. Designing plant control to remove undesirables and leave some or most of the desirable plants requires knowledge of plant characteristics such as regeneration potential in relation to the kind of disturbance and location of growing points or seed survival.

Dispersal, Migration, Establishment: Sowing Sufficient Pure Live Seed in Seedbeds Prepared to Maximize Establishment

The restoration equivalent of the successional process of dispersal is direct seeding or transplanting. Plant communities not dominated by desirable plants will require sowing of desirable plants after designed disturbance or wildfire to restore vegetation, ecological function, and value. The large-scales of our weed-dominated landscapes and burned areas require direct seeding for restoration or fire rehabilitation. Since weed seeds are often in the seed bank or are adapted to wind or mechanical dispersal to a site, sowing of desirable plants to reduce weed invasion or reestablishment is necessary. Methods of sowing also employ methods of seedbed preparation to place seeds in the seedbed where their requirements for germination and seedling establishment will be met. This can be a challenge when seeding species of different seed sizes and shapes into the highly variable seedbeds and soils of wildlands. However, if seeds are not placed where their establishment requirements are met, plants will not establish.

Seedbed Preparation and Sowing Methods

Common methods of large-scale sowing of rangelands include drilling and broadcasting seeds. With both methods, the goal is to bury seeds, but to place them at the best depth for their size. Species with smaller seeds like sagebrush or kochia (*Kochia prostrata*) may establish best when broadcast, then firmed into the surface by a rubber-tired cultipacker. The challenge for these seeds is to firm them in, but to not bury them deeper than a few millimeters. Sagebrush seeds can also be seeded through a rangeland drill using a trashy seed box with a pick-wheel inside to force the seed into the seed tubes. To avoid competition with sown grasses, sagebrush is commonly seeded separately in its own rows, and the seed tubes are pulled to let seed fall on top of the ground and avoid excessive burial. A concern for seeding sagebrush this way is that seeds may not be anchored to the surface and may blow away.

Grasses generally establish best when drilled in the fall. Larger seeds such as those of Indian rice grass (*Achnatherum hymenoides*) can be sown deep (2 to 5 cm) in sandy soils, while most grass species should be sown 1 to 2 cm deep. Depth bands on the disks of a rangeland drill are used to prevent excessive seed burial on sandy soils. Newer drills are now available that provide better control of seed placement than the standard rangeland drill. These drills should be tested for success with a range of species on different sites and soils.

Where topography or surface debris makes it impossible to pull a drill across the landscape, or where small seeds can be firmed into the soil, broadcasting is used. The best example of large-scale broadcasting is for fire rehabilitation in burned pinyon-juniper woodlands. Typically seeds are broadcast from a whirlybird seeder suspended from a helicopter or broadcast from a venturi-type seeder on a fixed wing aircraft. Sites are oneway chained to cover seed after broadcasting. Broadcasting seeds without chaining or some other form of seed coverage results in lack of revegetation success and weed invasion (Ott and others 2003). Larger seeds, such as those of four-wing saltbush (*Atriplex canescens*) or bitterbrush (*Purshia tridentata*), can be sown while chaining. These seeds are sown from a dribbler box attached above the tracks of the two caterpillar tractors pulling the chain. The seeds fall out of the box onto the top of the track and are buried in the track imprints. Smaller seeds of sagebrush or rabbitbrush (*Chrysothamnus* spp.) should not be seeded this way because they will be buried too deep.

It is a challenge to broadcast seed mixtures of grass, forb, and shrub seeds with different seed shapes and sizes. Continuous mechanical stirring of seeds mixed with trashy seeds such as those of sagebrush is necessary to provide adequate seed flow from broadcast seeders. The different seed burial requirements of seed mixtures make it difficult to maximize establishment for any one species. Chaining or other post-broadcast seed coverage techniques, such as railing, cabling harrowing, or imprinting, will probably best help establish diverse mixtures when done soon after broadcasting on soils where a wide range of micro topographically diverse safe sites will be created. Determining the advantages of different methods requires experimental comparisons for different sites, species, and methods.

Various methods of improving the seedbed environment have been developed over the years. These include furrowing, imprinting, aerating, or otherwise configuring the seedbed to create safe sites or locations for seeds that favor their germination and establishment. The idea is to bury seeds at the proper depth for emergence, but to increase the time of available water, reduce salinity, moderate temperature, or otherwise maximize favorable environmental conditions for establishment. The success of these methods depends on the soil, seeded species, and precipitation after seeding. Some methods such as drilling and imprinting can result in excessive seed burial on sandy soils, or lack of sufficient burial on heavy-textured or compacted soils. Various methods of seedbed enhancement and sowing should be compared experimentally across a range of sites and with a variety of species in order to make best recommendations for specific sites. Seedbed enhancement may increase seedling establishment on average to moderately wet years, but does not ensure establishment on dry years (Winkel and Roundy 1991).

Seeding Rates

Seeding sufficient germinable seed of adapted species requires an understanding of germination characteristics as well as adaptability of candidate species. Traditional rangeland revegetation guidelines recommend sowing 5.4 to 8.9 lbs/acre (6 to 10 kg/ha) of pure live seed of grass species known to have a fairly wide range of adaptability. These recommendations have proven successful for introduced grasses, but additional considerations are needed to successfully sow native species. Pure live seed is the amount of viable seed in a bag of seeds. It can be expressed as a percentage of the total weight of viable seeds, plus other matter such as seed parts, weed seeds, and nonviable seeds. Viability, or whether the seed is dead or alive, can be determined by a tetrazolium chloride solution or TZ test, where the active dehydrogenase enzyme in live seeds results in a red staining. This test does not determine germinability. Dormant seeds are viable but not germinable until dormancy is broken by artificial means or by specific environmental conditions. State seed testing laboratories determine germination percentages at temperature, light, and other incubation specifications generally known to maximize germination for a particular species. Some species may also be subjected to pretreatments such as seed coat scarification or chilling prior to incubation to maximize germination. When both seed viability and germination are tested, seed tags may bear germination and hard seed (viable but dormant) percentages.

Bulk seeding rates are calculated by dividing the recommended pure live seeding rate by the pure live seed percentage. For large-scale fire rehabilitation projects, the Bureau of Land Management (BLM) typically contracts for lots of seed specifying at least a 80 percent germination for grasses, or lesser percentages for some species, such as sagebrush, that are hard to clean. The BLM sends samples of their seed purchases to a State seed testing laboratory to verify the specifications. Because fire rehabilitation seeding is rushed in the late summer and early fall, seed labs may only have time to do a TZ test. If seed lots are found to have lower pure live seed percentages than was specified in the contract, the BLM may return the seed or adjust their price downward. The BLM often seeds using bulk rates for introduced grasses and legumes known to have high germination percentages (>80 percent). These rates typically run from 1.8 to 3.6 lbs/acre (2 to 4 kg/ha) of each species in a mixture of three or more species. For fire rehabilitation in the past, mainly introduced grasses have been seeded with some native grasses and a few introduced legumes such as sainfoin (*Onobrychis vicifolia*), small burnet (*Sanguisorba minor*), or alfalfa (*Medicago sativa*) (Richards and others 1998).

Successful establishment of native grasses, forbs, and shrubs may require higher seeding rates than those for simple introduced species mixtures. Thompson (2002) found successful large-scale establishment of native seed mixtures at 17.8 lbs/acre (20 kg/ha) bulk total seed drilled on burned sagebrush sites and 16 to 26.8 lbs/acre (18 to 30 kg/ha) total seed broadcast and chained on burned pinyon-juniper sites. Bulk rates required to get similar establishment from standard BLM seed mixes were generally lower and cost much less, but did not result in comparable establishment of native plants. Pyke and others (2003) found native plants in BLM fire rehabilitation projects, but they were unable to determine if those plants were residual to the sites or established by seeding. Native mixtures may require higher seeding rates than introduced species, and more careful species selection for specific sites. In Thompson's (2002) study, seeding predominately Indian ricegrass, known for its ability to emerge from deeper sandy soils, could have saved the extra expense and failure of other native grasses that were probably drilled too deep on the sandy sagebrush site tested.

Interaction/Reaction: Establishing and Facilitating Diverse, Native Communities

Plant ecologists have identified numerous combinations of plant-plant interactions (Barbour and others 1998). Clements (1928) stressed the importance of competition as a driving force in succession and what eventually dominates a site. This makes sense for classic forest succession where the dominant climax tree species are the ones that eventually develop large enough root and canopy structures to compete best for resources. Disturbance plays a vital role in opening up resources for a more diverse suite of species. Although competition evidently is a major driving force for the plants that eventually dominate a site after disturbance, other interactions may be more important in providing for long-term compatibility and diversity in a community. Plants may partition resources among themselves in time by growing during different seasons, or in space by accessing different soil depths. Rabbitbrush is evidently less competitive with grasses than Wyoming big sagebrush because its taproot uses deeper soils and avoids major competition with shallower grass roots (Frischknecht 1963). On the other hand, the two-layered surface and taproot system of Wyoming big sagebrush makes it a strong competitor with perennial bunch grasses. Scientists in Turkmenistan developed range improvement practices to improve forage quality and quantity for livestock. They selected woody species to use deeper soil moisture than the extant herbaceous communities (Nechaeva 1985). Agroforestry and intercropping practices are dependent on finding crops and trees that yield more when grown together than when grown separately. The best known example is growing nitrogen-fixing legumes with grasses. However, legume enhancement of grass growth requires long periods of available soil moisture to work best.

Although shrubs are generally considered competitive with herbaceous species, they also offer a suite of services to a diverse community, such as

- Enhance soil fertility
- Catch seeds, spores, soil, and snow
- Moderate the temperature environment
- Improve soil aggregate stability and infiltration rates
- Harbor beneficial insects

(Call and Roundy 1991; West 1989). Because there are many ecological and management benefits to mixed communities, we would like to restore them or establish them in fire rehabilitation seedings. Such a goal is much more ambitious than the single species or simple introduced species mixes of past rangeland revegetation. Use of native species in this effort requires understanding about which ecotypes are best adapted to specific regions or sites. The large-scale requirements of fire rehabilitation suggest that we should use native plant materials with a wide range of adaptation if possible.

Plant Materials Selection and Improvement

Plant adaptation and plant materials trials in the past have taken an agronomic approach. Numerous collections are planted in separate rows or blocks in common gardens and evaluated over many years. When a particular collection appears to be more vigorous than the others, it is selected for release. This approach takes a very long time to release a given plant material, and fails to address some important ecological aspects of mixed community restoration. It limits genetic diversity for out-crossing species by keeping the collections separate. Mass selection and other crossing techniques could be used to maximize genetic diversity. Very few examples of such approaches have been tested. An emerging approach is to certify seeds as “source-identified” (Young 1995). These collections are certified as originally collected from a particular site, representing a specific environment. Managers could choose “source-identified” plant materials from sites with similar regional environmental conditions as the sites they need to restore or rehabilitate. Once a large native seed industry is developed, managers could even choose physical mixes of a number of source-identified plant materials to best cover their estimated environmental conditions. Such an industry will need establishment of large seed warehouses and seed storage guidelines to allow stockpiling and a consistent market for these plant materials. Commitment of government to large restoration efforts such as the Great Basin Restoration Initiative will also support a more consistent demand for specific native plant materials. In the past, the demand for native seeds has been highly variable and subject to the severity and extent of the current fire season.

Establishing Diverse, Weed-Resistant Communities

Another aspect of plant materials evaluation, not generally tested much in the past, is that of how well plant materials work together rather than separately. The larger concern, of course, is that of establishing “stable, diverse” plant communities that are resistant to weed invasion. That is a major challenge. Not only are we not really sure how mixtures of plants will persist together, but it is a major challenge to seed diverse mixtures and have all the seeded species establish. When we have seeded aggressive, more weed-resistant introduced grasses in mixtures with native species, the introduced species eventually dominate (Pyke 1996). Approaches to more successfully establishing diverse communities include (1) seeding more aggressive species at a much lower rate than less aggressive ones, (2) seeding certain species or mixes in separate rows, strips, or patches, or (3) interseeding slower growing species such as some shrubs after scalping out established grasses. Because sowing equipment and environmental conditions favor establishment of many grass species over that of shrub and forb species, you cannot expect that just mixing species will produce a community in the same proportion as the seed mix (Newman and Redente 2001).

Weed-control strategies include designed disturbance to reduce weed populations and controlled establishment of desirable plants to preempt resources from weeds and prevent their invasion in the future (Sheley and others 1996). In that regard, we really do not know enough about what constitutes a community resistant to specific weeds on different sites. Our goal is to establish a suite of desirable plants that allows for their own coexistence, but excludes weeds. Goldberg (1990) has suggested that plant-plant interactions are often indirect through intermediate resources. To guide seeding mixture recommendations of the future, we must look at resource needs and use in time and space by desirable and weedy species. This type of research can require many years to develop recommendations, given the great range of species, weeds, and environmental conditions. In the meantime, it is very important that diverse seedings be monitored for response of both desirable and weedy species. Every fire rehabilitation or restoration project is an experiment from which something can be learned to guide future efforts.

It may be unrealistic to expect weed control and successful revegetation of native plants on sites where precipitation is low and the proximity of weed populations threaten reinvasion. On such sites, use of bridging species such as crested wheatgrass (*Agropyron desertorum*), which are more easily established and resistant to weed invasion, may be necessary. The bridging species could later be controlled and may be much easier to replace with native species than weedy species (Cox and Anderson 2004). Restoration will require innovative approaches to meet the requirements of native plants.

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Status and Use of Important Native Grasses Adapted to Sagebrush Communities

Thomas A. Jones
Steven R. Larson

Abstract—Due to the emphasis on restoration, native cool-season grass species are increasing in importance in the commercial seed trade in the Western U.S. Cultivated seed production of these native grasses has often been hampered by seed dormancy, seed shattering, and pernicious awns that are advantageous outside of cultivation. Relatively low seed yields and poor seedling establishment have also restricted their cultivation. Most are members of the Triticeae tribe. Bunchgrasses include Snake River wheatgrass (*Elymus wawawaiensis*), bluebunch wheatgrass (*Pseudoroegneria spicata*), basin wildrye (*Leymus cinereus*), and squirreltail (*E. elymoides* and *E. multisetus*). Rhizomatous grasses include western wheatgrass (*Pascopyrum smithii*), thickspike wheatgrass, and streambank wheatgrass (both *E. lanceolatus*), and beardless wildrye (*L. triticoides*). Important non-Triticeae native bunchgrasses include native bluegrasses (*Poa* spp.) and Indian ricegrass (*Achnatherum hymenoides*). These grasses may be either self-pollinated, cross-pollinated, or apomictic. These mating systems are reflected in the patterns of genetic variation characteristic of these species. At the USDA Agricultural Research Service (ARS) Forage and Range Research Laboratory at Utah State University, our goals are to understand distribution-wide patterns of genetic variation and to develop native cool-season grasses that are adapted to rangeland environments, are reflective of natural patterns of genetic variation, and are amenable to commercial seed production. To best accomplish these goals, we are attempting to develop a better understanding of the correlation between *genetic variation* and *ecological adaptation* at a variety of levels ranging from the whole plant to the DNA molecule. Besides ourselves, plant materials of these species have been released by USDA Natural Resources Conservation Service (NRCS) Plant Materials Centers (Bridger, MT; Aberdeen, ID; Pullman, WA; Lockeford, CA; Los Lunas, NM), Forest Service Shrub Sciences Laboratory (FS SSL) (Provo, UT), and Upper Colorado Environmental Plant Center (UCEPC) (Meeker, CO).

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Squirreltail (*Elymus elymoides* [Raf.] Swezey and *E. multisetus* [J.G. Smith] M.E. Jones)

Squirreltail is a short-lived perennial grass that is a prominent understory species in the sagebrush steppe community (Jones 1998). Squirreltail is a complex of five taxa, all of which are found in southwestern Idaho (Wilson 1963). Each taxon can be easily identified by spike morphology using a dichotomous key. *E. elymoides* ssp. *elymoides* is the most common and widespread taxon and is probably most closely related to *E. elymoides* ssp. *californicus*, which is prominent on the eastern slope of the Sierra Nevadas. *E. elymoides* ssp. *brevifolius* is especially common in the central and southern Rockies where plants and seeds are exceptionally large; however, these populations are conspicuously different from *E. elymoides* ssp. *brevifolius* plants originating in the Northwest. *E. elymoides* ssp. *hordeoides* is the most diminutive of the group, and probably the least common overall. *E. multisetus*, commonly called big squirreltail, is often considered a distinct species from the others, a position supported by our molecular data (Larson and others 2003). It is most common in the Northwest.

Squirreltail is a self-pollinated tetraploid ($2n = 28$) that genetically consists of the **St** and **H** genomes, which include 14 chromosomes (7 pairs) each. These two genomes are characteristic of *Elymus* worldwide and are indicative of the evolutionary history of this polyploid genus. The **St** genome originated from the bluebunch wheatgrasses (*Pseudoroegneria* spp.) and the **H** genome from the barleys (*Hordeum* spp.). Most *Elymus* species are predominately self-pollinating, including blue wildrye (*E. glaucus*), slender wheatgrass (*E. trachycaulus*), and Canada wildrye (*E. canadensis*). Prominent cross-pollinated exceptions include thickspike wheatgrass, Snake River wheatgrass, and the introduced species, quackgrass (*E. repens*) (Jensen and Asay 1996; Jensen and others 1990).

Squirreltail germplasms released to date are Sand Hollow (*E. multisetus*; Emmett, ID) (Jones and others 1998), Toe Jam Creek (*E. elymoides ssp. californicus*; Tuscarora, NV), Fish Creek (*E. elymoides* ssp. *elymoides*; Carey, ID), and Tusas (*E. elymoides* ssp. *brevifolius*; multiple locations in New Mexico). In addition, several seed growers are producing local proprietary seed sources.

Table 1—Seed production acreages of all classes of certified seed in the United States from 1996 to 2000 for native cool-season grasses commonly seeded on rangelands in the Intermountain Region[a].

Species/Cultivar	1996	1997	1998	1999	2000	Mean
Western wheatgrass	**934**	**959**	**1,626**	**3,029**	**3,371**	**1,984**
Rosana	366	355	992	1,593	1,662	994
Arriba	209	255	286	919	1,055	545
Barton	273	272	272	432	548	359
Rodan	46	37	36	45	66	46
Flintlock	40	40	40	40	40	40
Thickspike wheatgrass	**931**	**1,196**	**1,135**	**2,183**	**3,666**	**1,822**
Critana	459	430	501	1,086	1,840	863
Sodar	345	505	299	596	905	530
Bannock	127	229	312	453	505	325
Schwendimar	0	0	23	48	416	97
Elbee	0	32	0	0	0	6
Native bluegrasses	**469**	**560**	**956**	**1,267**	**1,721**	**995**
Sherman	394	494	836	1,239	1,623	917
Canbar	75	66	120	28	98	77
Snake River wheatgrass/Secar	**313**	**292**	**600**	**949**	**2,054**	**842**
Basin wildrye	**512**	**416**	**572**	**851**	**941**	**658**
Magnar	371	369	445	526	448	432
Trailhead	141	47	127	325	493	227
Bluebunch wheatgrass	**126**	**537**	**603**	**586**	**1,260**	**622**
Goldar	49	422	473	401	965	462
Whitmar	77	115	130	185	295	160
Indian ricegrass	**190**	**126**	**196**	**307**	**944**	**353**
Nezpar	129	107	115	133	499	197
Rimrock	26	3	65	55	251	80
Paloma	35	16	16	119	194	76
Beardless wildrye	**59**	**74**	**93**	**0**	**65**	**58**
Shoshone[b]	59	74	91	0	65	58
Rio	0	0	2	0	0	0
Squirreltail/Sand Hollow	**0**	**0**	**0**	**8**	**8**	**3**
Proprietary (all species)	**30**	**419**	**0**	**289**	**2**	**148**
Total	**3,564**	**4,579**	**5,781**	**9,469**	**14,032**	**7,485**

[a]Figures compiled from AOSCA (1996–2000).
[b]Actually the introduced *Leymus multicaulis*.

Squirreltail exhibits a high degree of racial differentiation, as described for other species by Clausen and others (1947). We evaluated squirreltail accessions (27 in data set 1, 47 in data set 2) for a battery of ecological traits, which could be used to characterize the ecological relationships between and within taxa (Jones and others 2003). For data set 1, 13 traits were measured, including days to seedling emergence, length of the seedling's first leaf, total plant dry matter, root-to-shoot ratio, leaf area, specific leaf area, root length, specific root length, heading date, seed mass, emergence index (from 20 mm), emergence index (from 60 mm), activity of the nitrate reductase enzyme, plant height, and heading date. For data set 2, these same traits were measured except leaf area, specific leaf area, the emergence indices, and nitrate reductase activity. Seed mass was also measured for data set 2.

In data set 1, *E. multisetus* accessions showed greatest seedling vigor and root development. *E. elymoides* ssp. *elymoides* accessions had lowest seed mass and earliest phenological development. *E. elymoides* ssp. *brevifolius* had thickest leaves and slowest germination. Taken together, the 13 traits clearly demarcated the three groups of accessions. In data set 2, *E. elymoides* ssp. *elymoides*, *E. elymoides* ssp. *brevifolius*, and *E. multisetus* accessions again separated discretely, but *E. elymoides* ssp. *brevifolius* accessions separated into three subgroups. Early (subgroup B) and late-maturing (subgroup A) accessions from the Rocky Mountains separated apart from each other and also apart from intermediate-maturing accessions from southwestern Idaho (subgroup C).

Native Bluegrasses (*Poa secunda* Presl.)

Native bluegrasses also serve as important understory components of the sagebrush steppe vegetation. A large number of scientific names (*P. secunda, P. ampla, P. canbyi, P. gracillima, P. incurva, P. juncifolia, P. nevadensis, P. scabrella,* and *P. curtifolia*) have been given to various of the native bluegrasses, but Kellogg (1985, 1990) combined them all into *P. secunda*, except *P. curtifolia*, an endemic from central Washington. Kellogg argued that, because morphological variation among these entities is continuous,

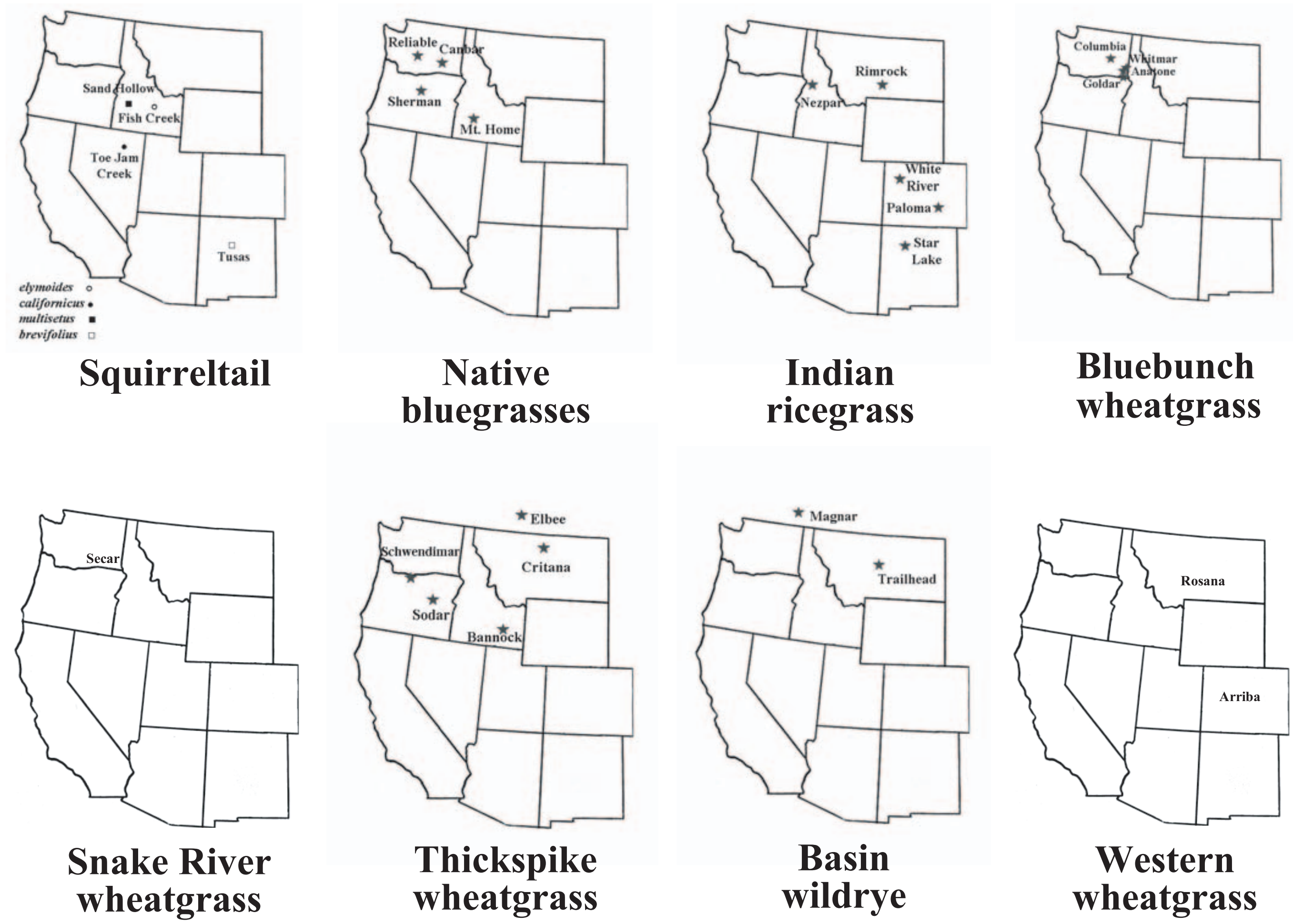

Figure 1—Points of origin of plant materials of squirreltail, native bluegrasses, Indian ricegrass, bluebunch wheatgrass, Snake River wheatgrass, thickspike wheatgrass, basin wildrye, and western wheatgrass.

any attempt to subdivide this group would be arbitrary. Nevertheless, she tolerated the retention of common names within the group.

Releases are 'Canbar' (WA), commonly known as "canby bluegrass," 'Sherman' (OR), commonly known as "big bluegrass" because of its larger stature and longer leaves, and Reliable germplasm (Yakima, WA), commonly known as "Sandberg bluegrass" (fig. 1). Certified seed production acreage of Sherman has greatly increased in recent years, while acreage of Canbar remains low (table 1). Sandberg bluegrass, a diminutive plant, mimics the phenology of cheatgrass. It flowers at least as early as cheatgrass and senesces upon seed ripening in late spring or early summer.

Taxonomic confusion in *Poa* is enhanced by facultative apomixis, an asexual form of reproduction by seed that may preempt sexual reproduction (Kellogg 1987). This means that new genetic variation generated by sexuality can be replicated in large quantities at or close to 84 chromosomes, making them dodecaploids (12*x*), but big bluegrass plants usually have about 63 chromosomes (9*x*) (Hartung 1946). Obviously, this odd number is something that could only be fixed asexually. The frequency with which apomixis occurs probably varies with genotype, but few data have been collected to determine the mean and range among genotypes. Do 63-chromosome plants arise from hybridization of 84-chromosome plants and 42-chromosome (6*x*) plants? Do 63-chromosome plants have a consistently higher level of apomixis than 84-chromosome plants or do they reproduce sexually, spinning off more odd chromosome-numbered plants? We believe a better understanding of these issues would provide a more complete understanding of the taxonomy of the native *Poa* complex.

We used amplified fragment length polymorphism (AFLP) analysis to characterize genetic diversity within Canbar, Sherman, Mountain Home (ID), and Reliable and genetic divergence between them (Larson and others 2001). AFLP methodology involves using sets of DNA primers to copy DNA fragments of different lengths that are then multiplied via the polymerase chain reaction. The resultant array serves as a "fingerprint" of the plant's genotype. Genetic diversities of the wildland collections, Mountain Home and Reliable, are similar to one another, but much greater than Canbar (fig. 2). Sherman shows no genetic diversity whatsoever, which is excellent evidence for apomixis and hints at

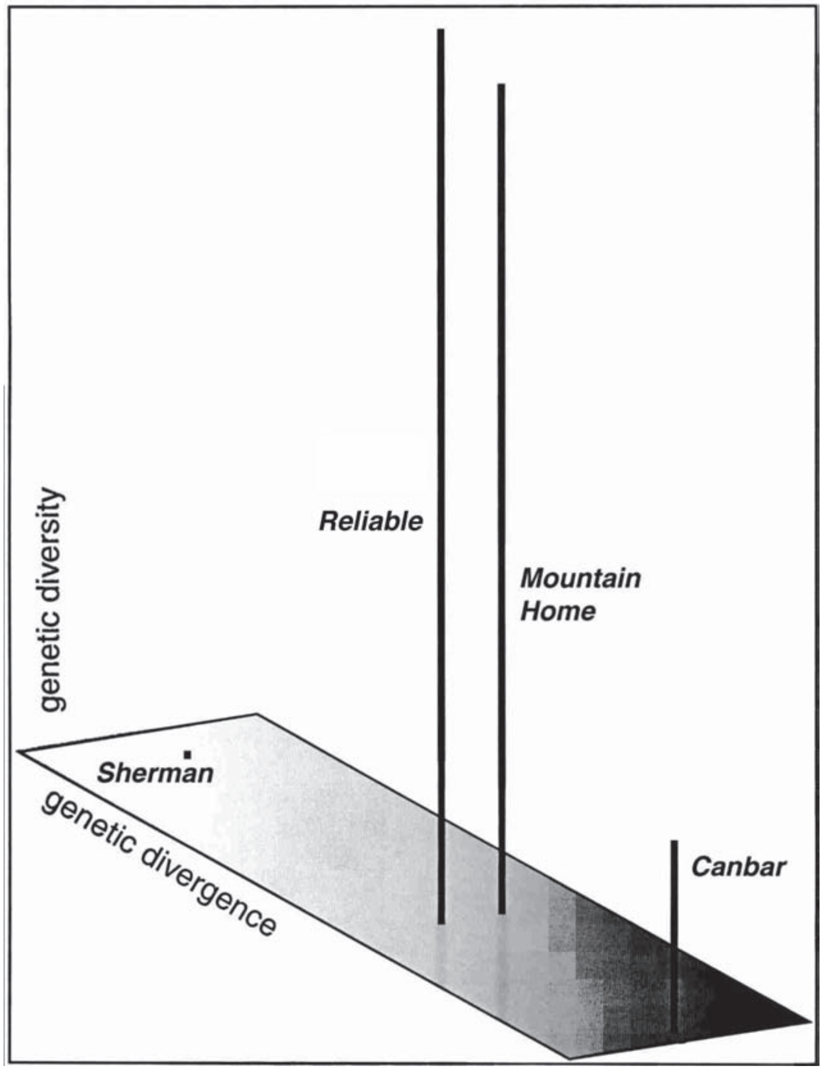

Figure 2—Genetic divergence (plane) and genetic diversity (vertical) of 'Sherman' big bluegrass, 'Canbar' canby bluegrass, and Reliable and Mountain Home Sandberg bluegrass germplasms.

the second question posed in the previous paragraph. Genetic divergence (separation on the plane) between Mountain Home and Reliable is small. These Sandberg bluegrasses are intermediate to Canbar (another 84-chromosome genotype) and Sherman ($2n = 63$), but more similar to the former.

Indian Ricegrass (*Achnatherum hymenoides* [Roem. & Schult.] Barkworth)

Indian ricegrass is a short-lived perennial bunchgrass that favors light-textured soils, especially fine sandy loams, sandy loams, loamy sands, and sands (Jones 1990). It is found in especially arid environments in the Intermountain Region as well as the Great Plains. Though it is a cool-season C_3 grass, it grows longer into the summer heat than native wheatgrasses and wildryes. This species exhibits great genetic diversity between and within populations. Seed polymorphism is common in Indian ricegrass (Jones and Nielson 1996). Larger seeds typically have greater seed dormancy than the smaller seeds (Young and Evans 1984). These two seed morphs are produced on different plants, which usually differ in other respects as well. Like squirreltail, Indian ricegrass is highly self-pollinating.

Releases are 'Paloma' (Pueblo, CO), 'Nezpar' (White Bird, ID), 'Rimrock' (Billings, MT), and Star Lake (McKinley County, NM), and White River (Rio Blanco County, CO) germplasms (fig. 1). Certified seed production acreage has increased in recent years (table 1).

Bluebunch Wheatgrass (*Pseudoroegneria spicata* [Pursh] A. Löve)

Bluebunch wheatgrass is widespread in the Intermountain Region, foothills, and open slopes of the Rocky Mountains and the northern Great Plains. It is probably most frequent in the region where Washington, Oregon, and Idaho converge. Many populations have been reduced or extirpated because this grass is both palatable and highly susceptible to overgrazing in the spring. Populations may be awned or awnless or are often mixed. Cultivars include the awnless Whitmar (Colton, WA) and the awned Goldar

(Umatilla National Forest, Asotin County, WA) (fig. 1). Certified seed production acreage of Goldar has greatly exceeded Whitmar in recent years. 'Secar' (Lewiston, ID) was released as a bluebunch wheatgrass, but was later discovered to be the newly recognized Snake River wheatgrass (see below). P-7, a multiple-origin polycross of 25 accessions, and Anatone (WA) are recent germplasm releases. Our work at ARS has emphasized selection for grazing tolerance and improved seed production.

Bluebunch wheatgrass may be either diploid ($2n = 14$) or tetraploid ($2n = 28$), with the diploid being far more common. Only the **St** genome is found in bluebunch wheatgrass; it is present in single or double dose depending on ploidy. Tetraploids appear to be most frequent in mesic regions such as southern interior British Columbia and parts of southeastern Washington. Accessions may be mixed for ploidy, but this is not visually obvious because the two ploidys are morphologically indistinguishable. In the past, chromosome number has been measured directly using root-tip cells, but today we typically measure it indirectly using flow cytometry, a much more rapid and convenient technique.

While bluebunch wheatgrass is the only New World species of *Pseudoroegneria*, other taxa are known in the Old World (Jensen and others 1995). These Old World taxa can be morphologically indistinguishable from bluebunch wheatgrass and can be successfully hybridized with it. Both the pollen and the seed of these hybrids are virtually sterile.

AFLP analysis was used to measure genetic diversity and divergence (see Sandberg bluegrass, above) in and among Goldar, Whitmar, and P-7 (Larson and others 2000). As expected, P-7, the multiple-origin polycross, had greater diversity than either Goldar or Whitmar, both of which originated from single sites in southeastern Washington (fig. 3). Divergence was greatest between Goldar and Whitmar, with P-7 being more similar to Goldar than to Whitmar.

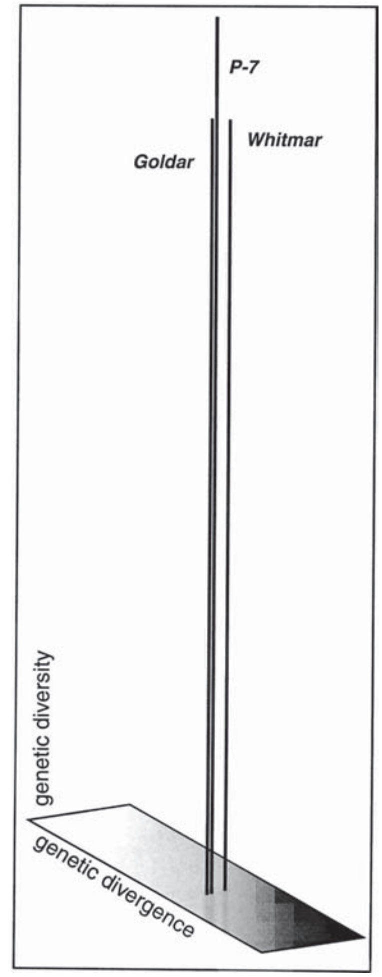

Figure 3—Genetic divergence (plane) and genetic diversity (vertical) of 'Goldar' and 'Whitmar' bluebunch wheatgrass and the P-7 multiple-origin polycross bluebunch wheatgrass germplasm.

Snake River Wheatgrass (*Elymus wawawaiensis* J. Carlson & Barkworth)

Relative to bluebunch wheatgrass, Snake River wheatgrass has a limited distribution within eastern Oregon and Washington and northern and central Idaho. Its frequency has likely been severely reduced because of the high degree of cultivation in that region. Snake River wheatgrass is a bunchgrass with excellent seed production and drought tolerance. It has performed very well in seedings in the Snake River Plain, despite the fact that it is not native to southern Idaho. The most southerly material of which we have knowledge originates from near Cambridge, ID. Snake River wheatgrass is less susceptible to overgrazing than bluebunch wheatgrass (Jones and Nielson 1997). At ARS we are selecting for clipping tolerance to improve its tolerance to grazing.

Until 1985, Snake River wheatgrass was confused with bluebunch wheatgrass (Carlson and Barkworth 1997). In 1980 'Secar' (fig. 1) was released as a bluebunch wheatgrass. Snake River wheatgrass is an **StH** species, thus its hybrids with bluebunch wheatgrass, an **St** species, are cytologically irregular (Carlson 1986). However, hybrids of Snake River wheatgrass with thickspike wheatgrass, another **StH** species, are cytologically regular (Jones and others 1995). It was these observations that initially justified the recognition of Snake River wheatgrass as a separate species apart from bluebunch wheatgrass and its placement in the genus *Elymus* rather than *Pseudoroegneria*. Traditionally, analysis of chromosome pairing in hybrids has been used to determine genomic composition, but we now utilize the FISH (fluorescent in situ hybridization) technique. FISH utilizes DNA-binding dyes that are specific to particular genomes. When used in combination, these dyes generate red or yellow fluorescence, depending on the genome. Intermediate genomes fluoresce orange.

While Snake River and bluebunch wheatgrass share a superficial resemblance, there are many features that distinguish the two from each other (Jones and others 1991). The spike of Snake River wheatgrass is more compact than bluebunch wheatgrass, primarily because the rachis internodes of Snake River wheatgrass are shorter. Snake River wheatgrass is always awned, whereas bluebunch wheatgrass may be awned or awnless. Snake River wheatgrass is always tetraploid ($2n = 28$), while bluebunch wheatgrass is primarily diploid and occasionally tetraploid ($2n = 14, 28$).

Snake River wheatgrass seed is considerably smaller than bluebunch wheatgrass seed. Snake River wheatgrass glumes are more lanceolate in shape than the nontapered bluebunch wheatgrass glumes, but size of the glumes is not diagnostic. Certified seed production acreage of Snake River wheatgrass has greatly increased in recent years and now greatly exceeds that of bluebunch wheatgrass (table 1).

Thickspike Wheatgrass (*Elymus lanceolatus* [Scribn. & J.G. Smith] Gould)

Thickspike wheatgrass is found throughout the Intermountain Region and the northwestern Great Plains, but is most common in Wyoming. Thickspike wheatgrass is a cross-pollinating rhizomatous grass closely related to Snake River wheatgrass. It is generally found on loamy to sandy soils. Like Snake River wheatgrass, thickspike wheatgrass is an **StH** tetraploid ($2n = 28$). Thickspike wheatgrass is much more tolerant of overgrazing than either Snake River wheatgrass or bluebunch wheatgrass. Cultivars appropriate for our region include Bannock (OR, ID, WA), Schwendimar (The Dalles, OR), Elbee (AB, SK), Critana (Havre, MT), and Sodar (Canyon City, OR) (fig. 1). Thickspike wheatgrass rivals western wheatgrass as the most important species for certified seed production acreage (table 1). In recent years Critana has been the leading cultivar, followed by Sodar and Bannock.

Sodar is known in the trade as "streambank wheatgrass," technically a botanical variety of thickspike wheatgrass (Dewey 1969). In Canada, thickspike wheatgrass is known as "northern wheatgrass."

Basin Wildrye (*Leymus cinereus* [Scribn. & Merr.] A. Löve)

Basin wildrye is a large-statured bunchgrass that is prominent in the Intermountain region. It is valued for winter grazing and for the shelter it provides during calving. In the Snake River Plain, basin wildrye is often found in locations with deep soils and high water-holding capacity. Basin wildrye has two chromosome races, tetraploid ($2n = 28$) and octoploid ($2n = 56$), that feature the **Ns** and **Xm** genomes in single or double dose, respectively. The **Ns** genome originated in *Psathyrostachys*, the Russian wildryes, but the origin of the **Xm** genome is uncertain.

The octoploid race, characterized by more robust plants that exhibit a glaucous blue color under drought conditions, is found in central and northeastern Oregon, northern Idaho, eastern Washington, and interior British Columbia. The tetraploid race, with somewhat smaller plants that remain green (rather than turning blue) in response to drought, is found in Central and southeastern Oregon, southern Idaho, Nevada, Utah, Colorado, Wyoming, Montana, Alberta, and Saskatchewan. Magnar is an octoploid cultivar (BC) and Trailhead is a tetraploid cutivar (Roundup, MT) (fig. 1). Magnar is the older cultivar, but its certified seed production acreage has remained stagnant in recent years (table 1). Trailhead's acreage has greatly increased in recent years and it now is equal to Magnar's. ARS is assembling several accessions into a multiple-origin polycross oriented towards northern Nevada, similar to the approach used in the development of P-7 bluebunch wheatgrass.

Beardless Wildrye (*Leymus triticoides* [Buckley] Pilger)

Beardless wildrye is a rhizomatous corollary to the bunchgrass basin wildrye in the genus *Leymus*, much as thickspike wheatgrass is to Snake River wheatgrass in the genus *Elymus*. Beardless wildrye is most common in northern Nevada, eastern Oregon, and northeastern California. It is an important riparian species, but it is also adapted to arid upland sites. Beardless wildrye and basin wildrye often occupy the same site and hybrids are common in such locations. Beardless wildrye has very high levels of seed dormancy, even higher than Indian ricegrass, a feature that is not prominent in basin wildrye. The only cultivar of beardless wildrye is Rio (Stratford, CA), which is propagated commercially by rhizomes because of its poor germination. 'Shoshone' is prominent in the seed trade where it is known as beardless wildrye, but it is now known to be *Leymus multicaulis*, an introduced species. *Leymus triticoides* and *L. multicaulis* are easily distinguished when the two species are grown side-by-side. Shoshone certified seed acreage has not increased in recent years (table 1).

Western Wheatgrass (*Pascopyrum smithii* [Rydb.] A. Löve)

Western wheatgrass is a cross-pollinating rhizomatous grass like thickspike wheatgrass. These two grasses are often confused, but they may be distinguished on the basis of glume shape. The glume of western wheatgrass is asymmetrical, while the thickspike wheatgrass glume is symmetrical. However, determination of ploidy is the most foolproof method to separate these grasses; thickspike wheatgrass is always tetraploid ($2n = 28$) and western wheatgrass is always octoploid ($2n = 56$) (Dewey 1975).

Western wheatgrass is believed to have originated from hybridization of thickspike wheatgrass and beardless wildrye (Dewey 1975). Western wheatgrass combines the **St** and **H** genomes of *Elymus* with the **Ns** and **Xm** genomes of *Leymus*. The frequency and distribution of western wheatgrass exceed either of its parents. Western wheatgrass is rhizomatous because both of its parents are rhizomatous. Western wheatgrass has an intermediate level of seed dormancy because beardless wildrye has high seed dormancy and thickspike wheatgrass does not exhibit seed dormancy. Compared to thickspike wheatgrass, western wheatgrass is more rhizomatous, better adapted to heavier soils, and less adapted to arid conditions.

All cultivars of western wheatgrass originate in the Great Plains. The two most widely used in the Intermountain Region are Rosana (Forsyth, MT) to the north and Arriba (Flagler, CO) to the south (fig. 1). Certified seed production acreage has increased greatly in recent years (table 1). Rosana is the leading cultivar (table 1). Western wheatgrass and thickspike wheatgrass acreages are similar and are greater than those of other species discussed herein.

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Nineleaf biscuitroot

Current and Potential Use of Broadleaf Herbs for Reestablishing Native Communities

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Abstract—Use of forbs for revegetation in the Intermountain West has been problematic due to the large number of species and lack of research data. Some forbs are found in numerous plant communities and distributed over wide geographic ranges while others are more narrowly adapted. Seed sources for revegetation use may be selected from species and ecotypes indigenous to the planting area. Management of local stands to improve seed production may be required to insure the availability of adequate quantities of seed. Alternatively, seed of an increasing number of commonly used species is being grown in agricultural settings with more reliable seed supplies resulting. Advances in wildland seeding methodology and forb seed production, harvesting, and conditioning technology have resulted from recent research and plant materials development programs.

Introduction

Shrublands of the Great Basin have undergone drastic changes since early settlement. Domestic livestock grazing and other human impacts, the spread of weedy annuals, and concurrent changes in fire frequency and intensity have reduced the diversity of sagebrush (*Artemisia* spp.) communities. Considerable effort has been directed toward revegetating disturbed rangelands. Seedings have generally been designed to increase forage production for livestock and, more recently, to re-establish native plant communities and wildlife habitat. Seed mixes have included various introduced and native grasses, forbs, and shrubs (Stevens 1983). Over the past 30 to 40 years most seed mixes were dominated by introduced grass species. These grasses were seeded because of their ease of establishment, forage production potential, ability to compete with invasive exotics, and their soil stabilization characteristics. In addition, the required quantities of seed were usually available. Plant materials development efforts over this period have gradually increased the number of native grass and forb species seeded and the amount of seed marketed. Research has also improved our understanding of secondary successional processes occurring when combinations of species are planted together (Walker and others 1995).

The use of native forbs in restoration has been limited due to a number of factors. These include the large number of forb species present in the Great Basin, our limited knowledge of seed production and seeding requirements for most species, the difficulty of harvesting forb seed from wildland stands, the highly unpredictable quality and quantity of wildland seed collections, and the frequently high cost of available seed. Overcoming these obstacles requires selection of accessions adapted to proposed planting areas, development of seed and seeding technology for each species, and establishment of seed production fields. In some situations, managing selected wildland stands to improve seed production may be a viable alternative when specific ecotypes are required for use over extensive areas. Both options require considerable time and effort for the development of reliable seed supplies.

Importance of Native Forbs

Native forbs are common and important components of most native plant communities of the Great Basin. Numerous species occur within this region, but most are not as widespread as the more common native grasses. Salt desert shrub and Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) communities support fewer native forbs than more mesic shrubland communities. Forbs occasionally grow in pure stands, but more frequently they are found as associated species growing with grasses and other forbs in grasslands, or as understory species in shrub or tree communities. Individual species may be distributed over wide geographic ranges and occur in a variety of plant communities, or they may be narrowly distributed edaphic endemics. Some species exhibit considerable variability within and among populations (Shaw and Monsen 1983).

The addition of native forbs to revegetation projects contributes to the establishment of a more complex community and enriches the food supply for sagebrush-associated wildlife, such as mule deer (*Odocoileus hemionus*), pronghorn (*Antilocapra americana*), elk (*Cervus elaphus*), sage-grouse (*Centrocercus urophasianus*), sharp-tailed grouse (*Tympanuchus*

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phasianellus), blue grouse (*Dendragapus obscurus*), and small mammals. Native forbs play a number of roles in wildland seedings:

1. They increase community diversity, health, and resilience.
2. Pioneer forbs provide ground cover and soil stabilization on disturbed and unstable sites.
3. Leguminous forbs improve nitrogen availability.
4. Forbs reduce the ability of exotic species to enter the community.
5. Vegetative plant parts as well as fruits and seeds of individual forb species are often valuable seasonal food sources for specific organisms.
6. The fire resistance of seedings may be improved by the addition of forbs that remain green well into the summer.
7. Forbs improve the aesthetics of seeded disturbances and low maintenance landscaping projects.

Sage-Grouse Use of Forbs

Forb-rich sagebrush communities are vital to the survival of sage-grouse and other species that rely on the complex of vegetation and associated faunal components of the community. The importance of forbs in the survival of upland bird species is becoming better understood and is proving to be more important than previously thought. In spring, sage-grouse diets change from sagebrush-dominated to forb-dominated. Breeding habitats used by prelaying hens should provide a diversity of forbs high in calcium, phosphorus, and protein; the condition of these areas may greatly affect nest initiation rate, clutch size, and subsequent reproductive success (Barnett and Crawford 1994; Connelly and others 2000). Juvenile sage-grouse are dependent upon insects and succulent forbs as critical food sources after hatching and until brood dispersal in fall. Chicks follow the habitat use of the hens with broods and broodless hens (Barnett and Crawford 1994; Beck and Mitchell 1997; Braun and others 1977; Wallstead 1971).

Reestablishment of Native Forbs

Selection of Seed Sources

For seedings on rangeland sites, local seed sources of native forbs should be added to seed mixes of compatible grasses and shrubs. Where reestablishment of native communities is the goal, land managers may need to contract collection and increase of local seed to produce adequate supplies for the project site. Released forb varieties or prevariety releases are available for a few species (table 1). Seed supplies of released materials are generally produced in agricultural seed fields and sufficient quantities for major seedings may be available or can be produced under contract. The area of adaptation for these has been determined to varying degrees, and is described in the release documentation. Species and seed sources known to be competitive with invasive species must be used if weeds are expected to be a problem. The availability and cost of native forb seed remains the greatest obstacle to inclusion of these species in seeding mixes.

Seeding Mixes

Although forb species may make up only a small portion of the plant composition and cover in sagebrush habitats, they are extremely important in the diets of sage-grouse broods and many other vertebrate and invertebrate species. Consequently, it is important to provide forb diversity in seedings, even when species must be seeded at low densities or spot seeded in favorable areas. Established seedings should be managed to allow forb growth to continue through spring and summer, particularly in sage-grouse breeding habitat (Paige and Ritter 1999).

Seeding Techniques

Even the commonly seeded forbs represent a number of plant families and genera and, consequently, an array of fruit and seed types and sizes is represented. Successful seeding requires that seedings be conducted at the appropriate season, usually late fall in the Great Basin, in order that moist prechilling requirements be met. This permits seeds to germinate in early spring when soil moisture conditions are most likely to be favorable. Seed must be planted at the appropriate depth. Another consideration is the condition of the fruit or seed structure planted. Although some will flow well and are easily seeded, others tend to mat together and must be seeded in drill boxes with agitators. Broadcasting can generally be successfully accomplished if seeds are diluted with seeds of grasses and covered as necessary following seeding.

Relative competitive abilities and affecting factors are largely unknown for native forbs. Nevertheless, substantial anecdotal information on common species strongly suggests that forbs should be sown separately or with other species that are sown in low densities to minimize competition. Likewise, the ability of seeded forbs to compete with invasive species that may be present on the site must be considered. If site preparation practices are not adequate to control weeds, the use of scarce and valuable seed, or the seeding itself, should be reconsidered.

Like all seedings, those including native forbs should be protected from grazing until plants are fully established and managed carefully thereafter to prevent degradation of the seeding or decline or loss of individual species. Monitoring data can be used to evaluate seeding techniques and may suggest improvements for future seedings.

The Future of Native Forb Development

Native forb seed grown in agricultural fields or collected from wildland stands is currently available for some species (table 1). Supplies, however, remain limited and erratic. In the last several years, threats to Great Basin ecosystems have fostered efforts to develop plant materials and seed and seeding technology for native forbs considered priority species by land management agencies. Personnel at State and Federal laboratories are developing selections of individual

Table 1—Forb characteristics and releases[a].

Species	Ecological status[b]	Origin[c]	Longevity[d]	Adaptation: Soil	Adaptation: Annual precipitation	Seeds per pound	Planting requirements[e]	Expected seed quality at time of purchase: Viability	Expected seed quality at time of purchase: Purity	Releases	Market availability of seed[f]
					inches	*number*[g]		*percent*	*percent*		
Achillea millefolium Western yarrow	E/M	N	P	Loam to clay loam	8–26	2,124,000–4,124,000	B/D	85–90	20–90+	None	5
Agoseris glauca Pale agoseris	M/L	N	P	Moist, well drained	>10	560,000	B/D	20–50; variable	20–60	None	2
Antennaria rosea Rose pussytoes	M	N	P	Coarse textured, well drained	>12	6,000,000	B	20–50	20–60	None	1
Artemisia ludoviciana Louisiana sage	E/M/L	N	P	Sandy loam to clay loam	>10	2,495,000–2,500,000, 3,800,000	B/D	75–85	20–90	'Summit'	2
Aster chilensis Pacific aster	M/L	N	P	Coarse to clay loam	>10	2,600,000	B/D	80–90	20–90	None	2
Aster glaucodes Blueleaf aster	M	N	P	Loam to clay loam	12–16	540,000	B/D	80–90	20–90	None	2
Astragalus cicer Cicer milkvetch	M	I	P	Sandy loam to clay loam	>15	114,000–122,000, 130,000		95	95	'Lutana', 'Monarch' 'Windsor'	5
Balsamorhiza sagittata Arrowleaf balsamroot	L	N	P	Deep, well drained, coarse to medium textured	8–25	55,000	D	80–90	95–99	None	3
Coronilla varia Crownvetch	M	I	P	Coarse to medium textured, well drained, low fertility, low pH	>26	109,760, 138,160	D	75	95	'Emerald'	5
Crepis acuminata Tapertip hawksbeard	M/L	N	P	Sandy to coarse, gravelly soil	8–20	800,000		80–90	90	None	1
Eriogonum ovalifolium Oval-leaved eriogonum	M/L	N	P	Gravelly to sandy clay, well drained	10	120,000	D/B	90	95–98	None	1
Eriogonum umbellatum Sulfur eriogonum	M/L	N	P	Well drained, medium textured, slightly basic to neutral pH	8–18	120,000, 209,000, 300,000	D	90	95–98	'Sierra'	2
Hedysarum boreale Northern or Utah sweetvetch	M/L	N	P	Dry, often rocky soil; well drained sandy loam to basic, silty clay loam	12–18	33,585, 70,000, 86,000–96,700, 125,000	D	95	90	'Timp'	3
Helianthella uniflora Oneflower helianthella	M	N	P	Coarse, shallow, well drained	10–35	52,560–53,560, 267,000	B/D	60	75	None	1

(con.)

Table 1—(Con.)

Species	Ecological status[b]	Origin[c]	Longevity[d]	Adaptation: Soil	Adaptation: Annual precipitation	Seeds per pound	Planting requirements[c]	Expected seed quality at time of purchase: Viability	Expected seed quality at time of purchase: Purity	Releases	Market availability of seed[f]
					inches	*number*[g]		*percent*	*percent*		
Linum lewisii Lewis flax	M/L	N	P	Fine to coarse textured, well drained	10–24	278,000–422,325	B/D	90	95–99	'Appar'	5
Linum perenne Perennial flax	M	I	P	Fine to coarse textured, well drained	10–24	278,280	B/D	90	95–99	None	2
Lomatium dissectum Fern-leaved lomatium	M/L	N	P	Dry, rocky	>14	42,225, 134,000	D	60–85	85	None	1
Lomatium nudicaule Barestem lomatium	M	N	P	Sandy			B/D	60–85	85	None	1
Lomatium nuttallii Nuttall lomatium	M	N	P	Light to heavy textured, well drained, basic	13–16	42,200–42,225	D	60–80	90	None	1
Lomatium triternatum Nineleaf lomatium	M/L	N	P	Well drained or dry, rocky, acidic to basic	8–20	42,000, 100,000	D	70	75	None	2
Lupinus argenteus Silvery lupine	E/M	N	P	Fine to medium, coarse	>10	18,300, 126,000	D	95	90	None	2
Onobrychis viciifolia Sainfoin	M	I	P	Sandy loam to clay-loam	12–16	22,000–26,000	D	90–95	95–98	'Eski', 'Remont'	5
Penstemon acuminatus Sharpleaf penstemon	E/M	N	P	Dry, open, sandy areas, often in sand dunes	12–30	400,000, 592,379	B/D	84	95	None	1
Penstemon deustus Scabland penstemon	E/M	N	P	Basalt scabland, volcanic soils	10–18	400,000	B/D	90	95	None	1
Penstemon eatonii Eaton Penstemon	M	N	P	Well drained, rocky to sandy loam	8–16	350,000–400,000	B/D	90–95	95	Richfield Selected	4
Penstemon pachyphyllus Thickleaf beardtongue	E	N	P	Sandy or gravelly	12–16	336,000	B/D	70	95	None	1
Penstemon palmeri Palmer penstemon	E	N	P	Well drained gravelly to sandy loam	10–16	609,675–610,000, 294,000	B/D	85–90	95	'Cedar'	4
Penstemon speciosus Sagebrush penstemon	E/M	N	P	Coarse-textured soils	8–24	400,000, 481,544	B/D	66–97	95	None	1
Penstemon strictus Rocky Mountain penstemon	M	N	P	Silty clay loam to sandy loam	14–26	262,000, 400,000	B/D	85–90	95	'Bandera'	3
Phlox hoodii Spiny phlox	M/L	N	P	Heavy clay, silty, loamy, or coarse deep soils			B		60–80	None	1

(con.)

Table 1—(Con.)

Species	Ecological status[b]	Origin[c]	Longevity[d]	Adaptation: Soil	Adaptation: Annual precipitation	Seeds per pound	Planting requirements[c]	Expected seed quality at time of purchase: Viability	Expected seed quality at time of purchase: Purity	Releases	Market availability of seed[f]
					inches	*number*[g]		- - - - - *percent* - - - -			
Sanguisorba minor Small burnet	E/M	I	P	Well drained sandy loam to silty clay, basic loams	12-25	34,000–55,115	B/D	90	80–95+	'Delar'	5
Sphaeralcea coccinea Scarlet globemallow	M	N	P	Sandy loam to clay	6–35	500,000–500,660, 800,000	D	90	80–90 (with gravity table)	ARS-2936	3
Sphaeralcea grossulariaefolia Gooseberryleaf globemallow	E	N	P	Sandy loam to clay	8–14	500,000–500,660, 850,000	D	90	80–90 (with gravity table)	ARS-2892	3
Trifolium repens White clover	M	I	P	Moist, deep soils	>20	672,000–880,000	B/D	80	95	None	5
Vicia americana American vetch	L	N	P	Sandy, clayey, medium textured soils	>9	24,500–41,314	D	85–95	90–98	None	1
Viguiera multiflora Showy goldeneye	M	N	P	Sandy loam to clay loam	15–20	800,000, 1,054,885–1,055,000	B/D	40–60	90–95	None	2

[a] References: Dayton (1960); Davis and others (2002); Goodrich and Neese (1986); Hermann (1966); Hitchcock and Cronquist (1973); Johnson (1993); Jorgensen and Stevens (2004); Link (1993); Monsen (2000); Nold (1999); Plants for a Future (2004); Plummer and others (1968); Redente and others (1982); Rumbaugh (1983); Rumbaugh and others (1993); Shaw and Monsen (1983); Stevens and others (1996); Stevens and Monsen (1988); Stevens and others (1985); U.S. Department of Agriculture, Forest Service (1937, 2004); U.S. Department of Agriculture, Natural Resources Conservation Service (2004).

[b] Ecological status: E = Early seral, M = Mid seral, L = Late seral, I = Introduced, no status.

[c] Origin: N = Native, I = Introduced.

[d] Longevity: P = perennial, B = biennial.

[e] Planting: B = broadcast or aerially seed, D = drill seed.

[f] Market availability: 5 = Excellent, large quanties, 4 = Good, generally sufficient, 3 = Fair, some seed available, 2 = Limited, small quantities may be available, 1 = Very limited, seed not usually available except by contract harvesting.

[g] Literature values vary widely, possibly resulting from differences in seed cleaning practices and the structure considered the seed.

species. Additional research conducted by scientists at these agencies and at universities provides seed and seeding technology and seeding guidelines for selected species. Plant materials developed through these efforts are made available to private growers for increase and for commercial seed production. In some cases, USDA Forest Service, USDI Bureau of Land Management, and other State and Federal agencies have provided wildland seed collections directly to growers for contract production of local seed sources for specific projects. Private growers have also made source-identified seed collections from wildland sites and planted them in seed fields. The challenge now for public and private agencies is to set priorities for the seed needs by species and population to direct research efforts and provide some stability for seed producers attempting to contend with the wide annual fluctuations in postfire seedings as well as the funding available for revegetation projects.

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Landscape Restoration for Greater Sage-Grouse: Implications for Multiscale Planning and Monitoring

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Abstract—Habitats and populations of greater sage-grouse (*Centrocercus urophasianus*) have declined throughout western North America in response to a myriad of detrimental land uses. Successful restoration of this species' habitat, therefore, is of keen interest to Federal land agencies who oversee management of most remaining habitat. To illustrate the challenges and potential for landscape restoration, we summarized recent findings of restoration modeling for sage-grouse in the Interior Northwest. Changes in amount and quality of habitat were evaluated under proposed Federal management and under two restoration scenarios. Under the two scenarios, the rate of habitat loss was reduced and the quality of habitat was substantially improved compared to proposed management. These results have direct implications for restoration planning and monitoring. First, a strategic, multiscale approach is needed that links the scale of the stand with scales of the seasonal, year-round, and multipopulation ranges of sage-grouse. Second, consideration of connectivity across scales is essential. Third, extensive and sustained use of a holistic suite of passive and active restoration treatments is needed. And finally, monitoring of both habitat and population responses across scales is critical. We offer suggestions on these and related points for effective restoration planning and monitoring of sage-grouse habitat.

Introduction

Habitats and populations of greater sage-grouse (*Centrocercus urophasianus*) have declined substantially across the species' range in response to a variety of detrimental land uses (Connelly and Braun 1997; Schroeder and others 1999). New guidelines were developed recently (Connelly and others 2000a) to help managers conserve and restore habitats for the species at the stand scale, but similar guidelines do not exist for landscape scales that encompass all or major portions of the species' range. The cumulative effects of management at these larger scales can greatly influence regional extirpation of sage-grouse (Raphael and others 2001), and recent landscape evaluations (Hemstrom and others 2002; Raphael and others 2001; Wisdom and others 2000, 2002a,b) offer new insights for effective restoration planning across the species' range.

The prospect of continued and widespread habitat declines for sage-grouse and other sagebrush (*Artemisia* spp.) obligates (Raphael and others 2001; Wisdom and others 2000, 2002a) points to the urgent need for development of restoration efforts across large landscapes. Without such restoration efforts, continued management of Federal lands under current land use plans will likely result in further loss and degradation of sagebrush steppe, with an increasingly high risk of population extirpation for sagebrush-dependent species (Raphael and others 2001).

In this paper, we summarize results of recent landscape evaluations to restore habitats for sage-grouse on lands administered by the U.S. Department of Agriculture, Forest Service, and U.S. Department of the Interior, Bureau of Land Management (FS-BLM) in the Interior Columbia Basin and adjacent portions of the Great Basin (Basin) (fig. 1). The 58 million-ha Basin encompasses a major portion of current and historical range of greater sage-grouse (fig. 1) (Wisdom and others 2002a). Proposed management of the Basin's sagebrush steppe will therefore substantially affect sage-grouse and other sagebrush obligates. That as context, our goals were to summarize the conditions projected for greater sage-grouse from a previous study within the Basin (Raphael and others 2001) in relation to two restoration scenarios recently developed and evaluated by Hemstrom and others (2002) and Wisdom and others (2002a) and to place the results in appropriate biological context for management of sage-grouse, particularly in terms of multiscale land use planning and monitoring.

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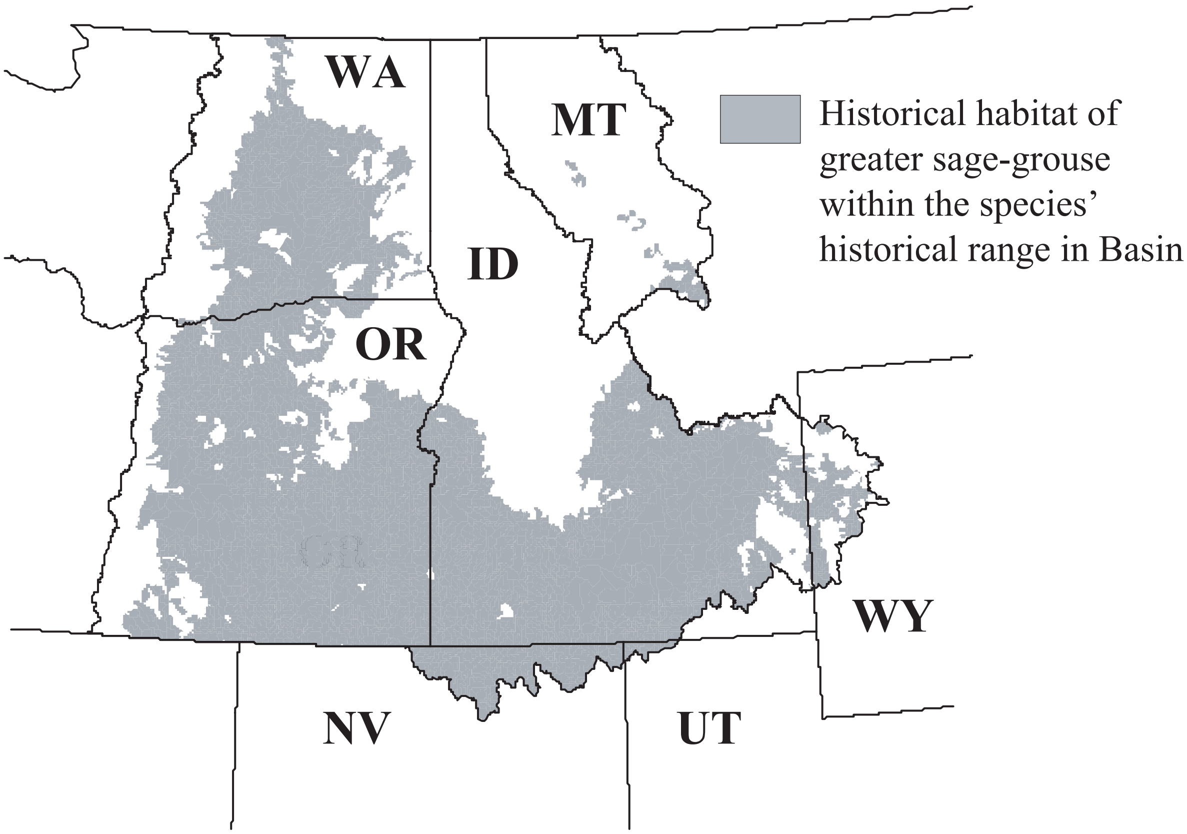

Figure 1—The Interior Columbia Basin assessment area in the Western United States, encompassing eastern Washington (WA), eastern Oregon (OR), most of Idaho (ID), northwestern Montana (MT), and adjacent areas of northwestern Wyoming (WY), northwestern Utah (UT), and northern Nevada (NV), and the historical habitats of greater sage-grouse within the species' historical range in the Basin (from Wisdom and others 2000).

Restoration Scenarios

Raphael and others (2001) evaluated the effects of proposed management of FS-BLM lands, projected 100 years in the future, on sage-grouse and other vertebrates that depend on sagebrush steppe in the Basin. These effects were associated with three management alternatives proposed in the Supplemental Environmental Impact Statement (SEIS) (USDA-FS and USDI-BLM 2000) of the Interior Columbia Basin Ecosystem Management Project (ICBEMP). Under proposed management, as well as current land management, Raphael and others (2001) found that most species that depend on sagebrush steppe, including sage-grouse, had a high probability of local or regional extirpation.

Hemstrom and others (2002) and Wisdom and others (2002a) evaluated the benefits of dramatically increasing the extent and intensity of restoration in sagebrush steppe to gain insight into the potential for improving environmental conditions for sage-grouse and thereby reducing the risk of extirpation compared to that under proposed management. Evaluations were based on Hemstrom and others' (2002) modeling of two restoration scenarios that substantially increased the combination of passive and active restoration of sagebrush steppe within the historical range of sage-grouse in the Basin.

As the basis for the two scenarios, Hemstrom and others (2002) defined passive restoration as "the process of modifying or eliminating existing management activities (for example, livestock grazing, roads, or recreation) that contribute to environmental degradation of desired resources." In contrast, Hemstrom and others (2002) defined active restoration as "the application of treatments that contribute to recovery of targeted resources (for example, appropriate use of wildfire suppression, prescribed fire, or seeding with native plants)." These definitions are similar to those summarized for rangeland restoration by McIver and Starr (2001).

The two scenarios substantially increased the levels of both passive and active restoration in relation to proposed management because of managers' desire to understand the magnitude by which sagebrush habitats could be improved relative to what was originally proposed. Scenario 1 assumed a 50 percent reduction in detrimental grazing effects by livestock as the main form of passive restoration. Detrimental grazing effects were defined as the probability, associated with grazing, of moving from a desired vegetation state, which provides habitat for sage-grouse (for example, gray boxes, fig. 2), to an undesired state (for example, white boxes, or nonhabitat, fig. 2). Accordingly, 50 percent and 100 percent reductions in detrimental grazing effects represented like reductions in the probability of transitioning from desired to undesired states for sage-grouse in relation to livestock grazing. Detailed rationale and supporting literature regarding these grazing effects on sagebrush habitats, and on sage-grouse, are described in Hemstrom and others (2002).

To achieve reductions of 50 percent and 100 percent in detrimental grazing effects, like reductions in stocking rate of livestock were assumed in combination with additional, positive changes in grazing systems (for example, increasing rest periods in rest-rotation systems) (Hemstrom and others 2002). This form of passive restoration under scenario 1 was applied to 6.4 million ha of FS-BLM lands in the Basin that have potential to be sage-grouse habitat or that currently serve as habitat (referred to as potential sage-grouse habitat). Two points are important here. First, not all grazing effects were assumed by Hemstrom and others (2002) to be detrimental to sage-grouse habitat. However, there is growing awareness that the herbaceous component of sagebrush stands, which can be reduced substantially in occurrence and percent cover with intensive livestock grazing (Anderson and Inouye 2001), is a primary requirement for successful nesting and brood rearing by sage-grouse (Barnett and Crawford 1994; Connelly and others 2000a; Crawford 1997). Consequently, there is a need to mitigate the detrimental effects of livestock grazing on native grasses and forbs important to sage-grouse productivity. And second, reduction in stocking rate of livestock can effectively restore native, herbaceous components in sagebrush steppe (Anderson and Inouye 2001). In defense of this point, Hemstrom and others (2002) stated

> Our assumed reductions in stocking rate needed to achieve a desired reduction in detrimental grazing were based on empirical data demonstrating that herbage production on rangelands is affected mostly by variation in stocking rate, and less so by changes in grazing system (Holechek and others 1998; Van Poolen and Lacey 1979). On arid rangelands such as those dominated by sagebrush, a positive response in herbage production must include a reduction in stocking rate in combination with active restoration treatments (see empirical synthesis by Holechek and others 1998).

Active restoration under scenario 1 was integrated with passive restoration on the same 6.4 million ha of potential sage-grouse habitat (Hemstrom and others 2002). By contrast, active restoration under proposed management targeted approximately 1.1 million ha of potential sage-grouse habitat. Thus, scenario 1 represented a sixfold increase in areas treated with active restoration beyond that identified in proposed management.

Key forms of active restoration included seedings and plantings of desired vegetation, particularly after fire events; wildfire suppression in vegetation types where such fires would facilitate invasion of exotic plants; prescribed fire in vegetation types where such fires would reduce woodland encroachment; and use of a variety of other chemical and mechanical treatments to control invading conifers and enhance composition of native grasses and forbs (Hemstrom and others 2002). The specific combination of active restoration treatments was tailored to the unique, desired response of each sagebrush community to the treatments. For example, use of prescribed fire to suppress juniper (*Juniperus* spp.) invasion, and enhance growth of herbaceous vegetation, was applied to many areas dominated by mountain big sagebrush (*A. tridentata* ssp. *vaseyana*), where fire effects are largely beneficial (Miller and Eddleman 2000). By contrast, suppression of wildfire, in combination with chemical treatments and native seedings to control spread of exotic grasses, was applied to many areas dominated by Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis*), where cheatgrass (*Bromus tectorum*) and other annuals often supplant native vegetation following fire events (Miller and Eddleman 2000).

Restoration scenario 2 was based on a 100 percent reduction in detrimental grazing effects by livestock, with a like reduction in stocking rate (Hemstrom and others 2002). This high level of passive restoration was integrated with the same level of active restoration assumed for scenario 1, with the same 6.4 million ha of FS-BLM lands targeted for treatment. Detailed methods, assumptions, and rationale associated with the scenarios are described in Hemstrom and others (2002) and Wisdom and others (2002a).

Conditions under the two restoration scenarios were projected 100 years into the future, as was done for proposed management. Restoration activities for each scenario were sustained throughout the 100-year period, with the frequency, intensity, and type of each activity designed to substantially recover or maintain desired conditions (Hemstrom and others 2002). Three landscape variables for sage-grouse were targeted for improvement as part of the restoration scenarios: (1) habitat amount, and two indices of habitat quality, (2) HRV departure (an acronym for historical range of variability departure, as defined by Hann and others 1997), and (3) uncharacteristic grazing (Hemstrom and others 2002; Wisdom and others 2002a).

Habitat amount is the area of sage-grouse habitat within the Basin, as defined by Wisdom and others (2000). Sage-grouse habitats in the study area primarily include low- to medium-height shrublands in basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*), Wyoming big sagebrush, mountain big sagebrush, and low sagebrush (*A. arbuscula*) communities, as well as herbaceous wetlands.

HRV departure was used to index the degree to which exotic plants have invaded and displaced components of native sagebrush steppe, particularly native grasses and forbs that are required by sage-grouse for successful nesting and brood rearing (Connelly and others 2000a; Crawford 1997; Drut and others 1994; Sveum and others 1998). Uncharacteristic grazing, (UG) was used to index changes in species richness, height, and cover of native understory grasses and forbs in response to livestock grazing, and to subsequent effects on quality of nesting and brood-rearing habitat for sage-grouse (Wisdom and others 2002a).

Restoration activities were designed to enhance habitat quantity (through increased habitat amount) and quality (through reductions in HRV departure and uncharacteristic grazing). Restoration was particularly designed to retard the cheatgrass (*Bromus tectorum*)-wildfire cycle (fig. 2), a pervasive problem in the Wyoming big sagebrush communities that compose >60 percent of sage-grouse habitat in the Basin (Hemstrom and others 2002).

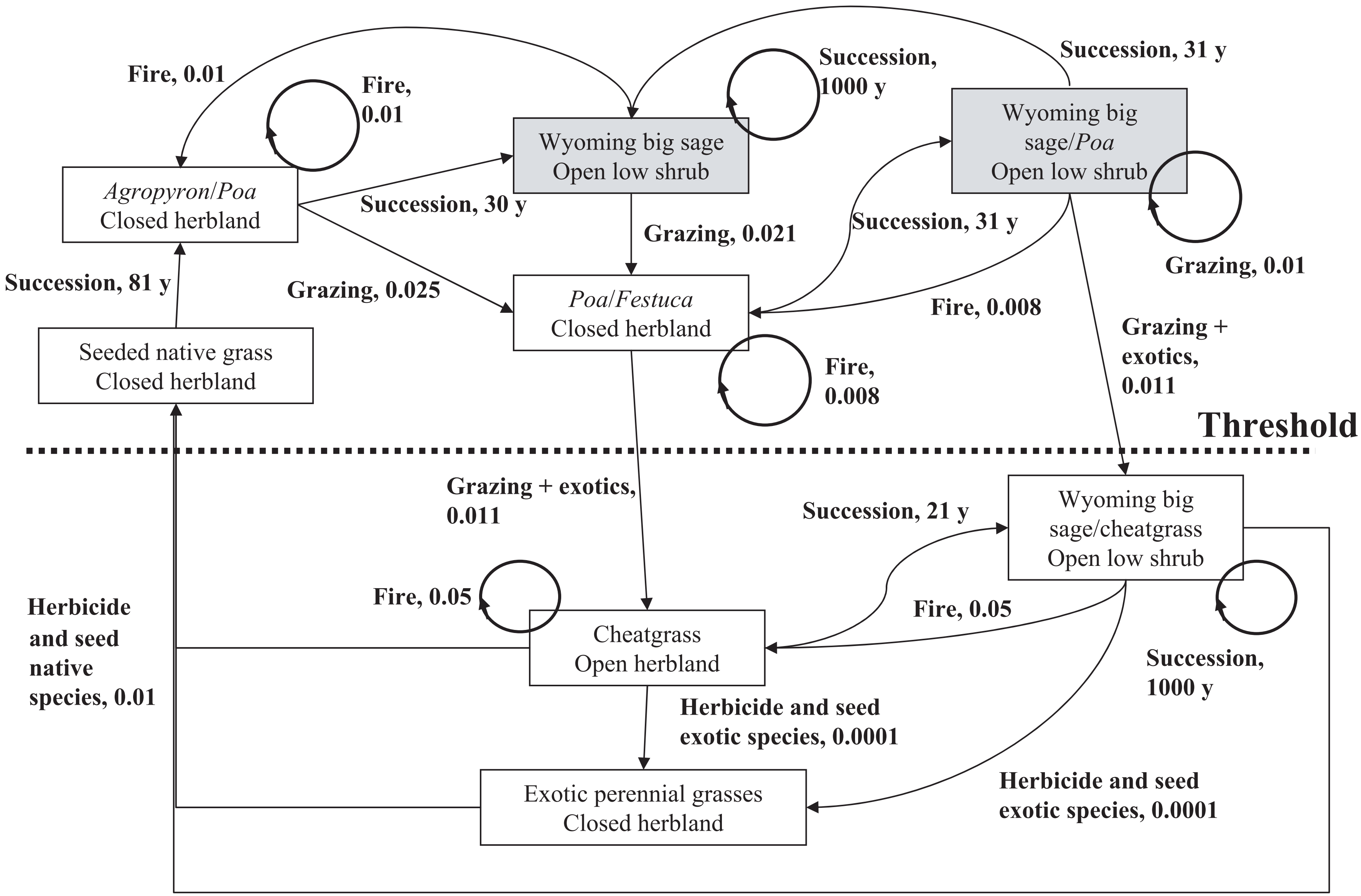

Figure 2—Example state-and-transition model (based on concepts of Westoby and others 1989 and Laycock 1991) for the potential vegetation type of warm, dry Wyoming big sagebrush (defined by Hann and others 1997) under an active restoration prescription in the Interior Columbia Basin (from Hemstrom and others 2002). Boxes are vegetation states and arrows are transitions from one state to another. In this example, transitions are brought about by succession and disturbance agents of fire, grazing, chemical treatments, and seedings. When disturbance is the primary cause for transition, the annual probability of moving from one state to another is shown in association with the disturbance that causes the transition (for example, fire, 0.01). Successional transitions of 1,000 years (y) indicate stable states. The dotted line indicates a transition threshold, below which restoration is difficult and requires a sustained, extensive combination of active and passive treatments.

Methods used to model these improvements under the restoration scenarios were deliberately conservative in terms of the assumed enhancements that such activities could produce. A conservative modeling approach was adopted because of the high uncertainty of restoration outcomes in sagebrush steppe (West 1999). This high uncertainty is related to incomplete knowledge of appropriate restoration methods and technologies, and the logistical challenges posed by sustained and integrated application of restoration treatments across vast areas of sagebrush steppe, which to date has not been attempted (Knick 1999).

Results from the restoration modeling were evaluated in terms of risk of regional extirpation of sage-grouse, as expressed in five outcome classes (see population outcome model described by Wisdom and others 2002b). Outcome A was defined as a very low risk of regional extirpation, followed by low (outcome B), moderate (outcome C), high (outcome D), and very high (outcome E) degrees of risk. These levels of risk corresponded to empirical findings of Wisdom and others (2002b), showing that areas of the Basin historically occupied by sage-grouse were associated with outcome A, whereas areas of current extirpation were associated with outcome E. Moreover, areas of the Basin currently occupied by sage-grouse have undergone an intermediate level of habitat loss and degradation between that estimated for historically occupied areas versus currently extirpated areas, resulting in an intermediate outcome of class C (Wisdom and others 2002b).

Risk of extirpation was assessed for FS-BLM lands and for all lands. The five outcome classes that indexed risk on FS-BLM lands were referred to as environmental outcomes (Raphael and others 2001).

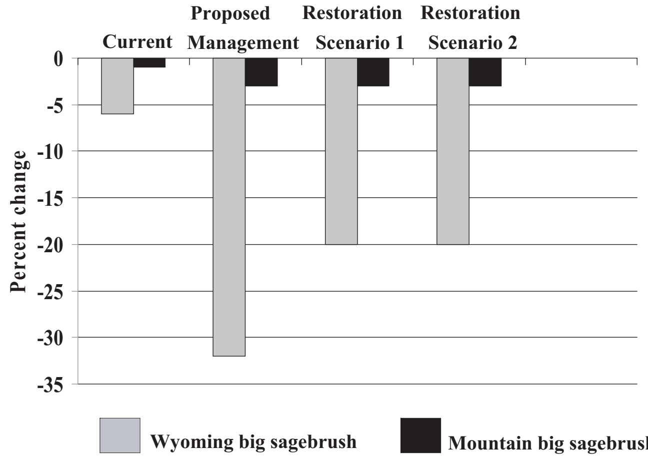

Figure 3—Percent change in habitat amount for sage-grouse, by major sagebrush communities, in 1,831 FS- and BLM-dominated subwatersheds within the historical range of sage-grouse in the Interior Columbia Basin (adapted from Hemstrom and others 2002). Results for proposed management (PM) and the restoration scenarios are for 100 years in the future. Decline is relative to amount of habitat estimated for historical conditions (circa 1850–1900) (Hann and others 1997).

Restoration Effects

Results from Hemstrom and others (2002) showed that under proposed management, sage-grouse habitat on FS-BLM lands would decline by 27 percent compared with the current amount (weighted average of percent declines across the sagebrush communities shown in fig. 3). However, habitat declined more slowly under restoration scenarios 1 and 2 (by about 19 percent and 17 percent, respectively), but neither scenario halted the long-term downward trend. Most future habitat loss was associated with sagebrush transitions to herblands and grasslands dominated by cheatgrass and other exotic plants in large areas of the Wyoming big sagebrush communities. Substantially smaller habitat losses were projected in the future in mountain big sagebrush communities, with losses due mostly to encroachment by juniper. In mountain big sagebrush communities, however, some loss to exotic plant invasion was projected at lower elevation, drier sites, while loss to woodland and forest encroachment was projected at higher elevation, mesic sites. Additionally, small declines in habitat amount were projected for other sagebrush communities, such as low sagebrush (Hemstrom and others 2002).

Restoration scenarios 1 and 2 increased habitat amount, relative to proposed management, by about 0.6 million ha and 0.8 million ha, respectively. The model projections indicated that a substantial increase in habitat from passive and active restoration would be offset by large (>1 million ha) losses associated mostly with wildfire and the subsequent invasion of cheatgrass in the Wyoming big sagebrush communities.

In contrast to results for habitat amount, the quality of habitat improved substantially under the restoration scenarios compared with proposed management, as indexed by substantial reductions in UG and HRV departure (fig. 4). Only 22 percent and 12 percent of subwatersheds were characterized by high UG under scenarios 1 and 2, whereas high UG occurred in 68 percent and 53 percent of subwatersheds during the current period and under proposed management, respectively. Percentage of subwatersheds with high HRV departure under the restoration scenarios (2 percent) also was substantially lower than the percentage with high HRV departure currently (6 percent) and under proposed management (7 percent). The restoration scenarios also were associated with a higher percentage of subwatersheds in the low and none classes of UG and HRV departure compared to current conditions and proposed management (fig. 4).

Risk of sage-grouse extirpation on FS-BLM lands was reduced to a moderate level under the two restoration scenarios compared to a high risk under proposed management (fig. 5). The moderate risk of extirpation under the restoration scenarios was the same as that estimated for the current period (fig. 5). The difference between a moderate versus a high risk of extirpation, as evaluated under the outcome classes for sage-grouse, was found by Wisdom and others (2002b) to represent a substantial difference in the probability of regional extirpation for the species.

Three landscape variables contributed to the increased risk of sage-grouse extirpation under proposed management (Wisdom and others 2002a): (1) reduced habitat quantity and quality, as reflected in an overall reduction in habitat capacity; (2) increased contraction of the species' range,

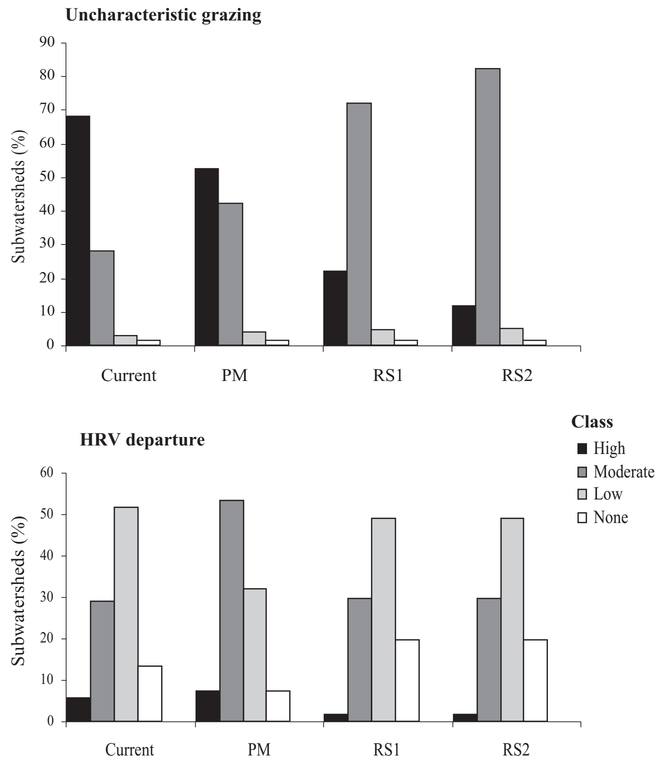

Figure 4—Changes in habitat quality for sage-grouse across time, as indexed by classes of uncharacteristic grazing and HRV departure in 1,831 FS- and BLM-dominated subwatersheds within the historical range of sage-grouse in the Interior Columbia Basin (from Hemstrom and others 2002). Results for proposed management (PM) and the two restoration scenarios (RS1 and RS2) are for 100 years in the future.

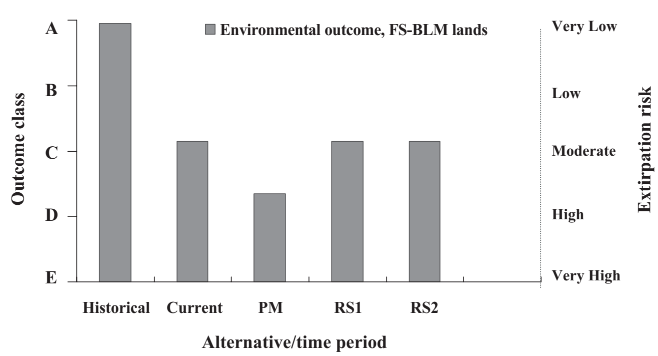

Figure 5—Risk of regional population extirpation for sage-grouse in the Interior Columbia Basin (adapted from Wisdom and others 2002a), as indexed by classes of environmental outcome (A = very low risk, B = low, C = moderate, D = high, and E = very high) projected for historical and current periods and 100 years in the future under proposed management (PM) and two restoration scenarios (RS1 and RS2). Results for environmental outcomes are for Forest Service (FS) and Bureau of Land Management (BLM) lands. Historical, current, and PM results are from Raphael and others (2001).

owing to continued habitat loss; and (3) decreased connectivity of habitats that remained within the contracted range. Wisdom and others (2002a) provide details.

Implications for Multiscale Planning and Monitoring

Results of restoration modeling by Hemstrom and others (2002) and Wisdom and others (2002a) have substantial and direct implications for management of sage-grouse habitats. We offer the following suggestions for planning and monitoring restoration activities that follow directly from results of the restoration scenarios:

1. **A strategic, multiscale approach is needed that links the scale of individual sagebrush stands with scales of the seasonal, year-round, and multi-population ranges of sage-grouse.** Identification of nesting and brood-rearing areas is critical for effective management of sage-grouse summer range. Similarly, identification of wintering areas is important for maintaining conditions adequate for winter survival. Information about conditions of individual sagebrush stands within seasonal ranges, as well as information about overall conditions on seasonal ranges, can be synthesized at larger scales to evaluate composite conditions for the species on a year-round basis. In turn, this information can be further synthesized to scales of local or multiple populations of sage-grouse, with patterns identified and summarized at larger, regional scales, such as the Snake River Plain or Columbia River Plateau. At each scale, relevant information is available for identifying management threats, setting restoration priorities, and implementing and monitoring a desired suite of restoration activities. While local areas are the traditional focus for restoration planning and implementation, the larger, regional scales are a critical and effective complement to local work. Information at regional scales, for example, can be used to target large areas that may deserve high priority for restoration and monitoring. By contrast, information about conditions on seasonal ranges, or of individual stands within seasonal ranges, is important for effective implementation of local restoration priorities.

2. **Consideration of connectivity across scales is essential.** Connectivity of summer range with winter range, and of local populations with multiple populations, is critical for maintaining viable populations of sage-grouse across the species' range. Raphael and others (2001) and Wisdom and others (2002a,b) used a landscape method to evaluate connectivity of sage-grouse habitats in the Basin as part of their population outcome model of extirpation risk. The method assessed the degree to which subwatersheds containing sage-grouse habitat fell within the median dispersal distance of juvenile grouse. This measure of connectivity was later validated as an important landscape measure of extirpation risk (Wisdom and others 2002b). Specifically, the connectivity of subwatersheds in areas currently occupied by sage-grouse was 61 percent (on a scale of 0 to 100 percent, where 100 percent represents habitats that are fully connected across the range of the species). By contrast, connectivity in areas where sage-grouse have been extirpated was only 23 percent (Wisdom and others 2002b). Similar

measures of connectivity need development and validation at a variety of scales to allow managers to understand how well restoration plans might improve the connectivity of habitat for sage-grouse, and to monitor the population response of sage-grouse to presumed improvements in connectivity. Development and validation of such connectivity measures will be most successful if conducted as a partnership between land managers and scientists, owing to the absence of research on this topic and the challenges of management application at multiple scales.

3. **Sustained use of a comprehensive suite of passive and active restoration treatments over extensive areas is needed.** Hemstrom and others (2002) and Wisdom and others (2002a) found that restoration of sagebrush habitats will require monumental spatial and temporal scales of application if downward trends are to be slowed or reversed. Expansive and sustained habitat restoration can maintain desired conditions and reduce the future risk of sage-grouse extirpation on FS-BLM lands. Local restoration efforts, without coordination and implementation across large areas as an adaptive management experiment, appear to have a low probability of reducing extirpation risk for sage-grouse in the Basin. This is due to the vast areas over which restoration must occur and the comprehensive, integrated manner in which a suite of restoration treatments must be implemented (Knick 1999). Knowledge voids about effective methods of restoration pose a major challenge. For example, few methods exist for effective restoration of native forbs in sagebrush habitats, and these forbs are critical for successful nesting and brood rearing by sage-grouse (Barnett and Crawford 1994; Drut and others 1994). Continued spread of exotic plants presents a formidable challenge to successful restoration, and warrants substantial research and management attention. In particular, the lower elevation, Wyoming big sagebrush communities are most susceptible to future loss from wildfire and subsequent invasion by cheatgrass. These areas warrant special attention for restoration activities. Moreover, results from Hemstrom and others (2002) suggest that suppression of wildfire, combined with improvements in grazing management, are critical for preventing expansive conversions of sagebrush to cheatgrass in Wyoming big sagebrush communities (fig. 2).

4. **Monitoring of both habitat and population responses across scales is critical.** Three types of monitoring have been defined and used by Federal land management agencies: (1) implementation, (2) effectiveness, and (3) validation monitoring. Implementation monitoring is the assessment of whether restoration and other management actions are implemented in the manner specified. By contrast, effectiveness monitoring evaluates whether the desired results from implementation were achieved, while validation monitoring determines the scientific validity of the concepts, methods, and predictions associated with the expected benefits of the management actions. For sage-grouse, all three types of monitoring are needed. For example, goals may be set for improving the amount, quality, and distribution of sage-grouse habitat under a restoration plan. The primary goal may be to improve habitat attributes, but invariably, the ultimate goal of such plans is to increase population growth of sage-grouse and associated species. Measuring such population responses will require regional scales of monitoring, and are best accomplished as part of research. Nonetheless, implementation monitoring is needed to determine whether the treatments are applied in the manner specified. Moreover, effectiveness monitoring is needed for two purposes: (1) to assess whether the desired habitat improvements were achieved with successful implementation (for example, were the desired improvements in habitat amount, quality, and distribution actually accomplished?); and (2) to determine whether the associated population of sage-grouse responded positively from the habitat improvements (for example, did the improvements in composition of understory bunchgrasses and forbs increase nest success and brood survival?). Finally, validation monitoring may be needed to understand why certain habitat restoration efforts might have failed, or how such restoration efforts worked successfully. Unfortunately, nearly all management and research of sage-grouse has focused on habitats or populations, but not both. Consequently, few monitoring efforts have considered effects on both habitats and populations, and considered all three types of monitoring. A notable exception is Connelly and others' (2000b) monitoring of sage-grouse response to prescribed burning in southeastern Idaho; this integrated type of habitat and population monitoring is needed to guide future restoration work. The model of Edelmann and others (1998), which evaluates the quality of sage-grouse habitat and predicts effects on growth rate of sage-grouse populations, provides a comprehensive framework for conducting all three types of monitoring at stand and landscape scales.

5. **A comprehensive set of species that depend on the sagebrush ecosystem needs to be targeted for restoration planning and monitoring.** A common management assumption about restoration efforts for sage-grouse is that such efforts will confer like benefits to a larger set of plants and animals that depend on sagebrush steppe. This assumption, however, has not been evaluated with empirical research and needs testing at multiple scales (Rich and Altman 2001). Moreover, new approaches that explicitly consider the needs of a comprehensive set of species need to be developed for effective restoration planning for all key attributes of the sagebrush ecosystem. Recently, Wisdom and others (2002c) developed a habitat network for a large set of vertebrates of conservation concern that depend on sagebrush steppe in the Basin. Watersheds for these species were characterized as one of three habitat conditions: Condition 1—habitats of high resiliency, abundance, and quality; Condition 2—habitats of high abundance but moderate resiliency and quality; and Condition 3—habitats that are highly degraded, fragmented, and isolated, or that have been extirpated. This type of characterization of a comprehensive, multispecies habitat network could be used to maintain habitats in a relatively unchanged state from historical conditions (Condition 1), to improve habitats where quality and resiliency have declined (Conditions 2 and 3), to restore habitats in areas of extirpation or low abundance and quality (Condition 3), and to improve connectivity where spatial gaps have developed (Condition 3).

Conclusions

Restoration of sage-grouse habitat represents a daunting task. Without consideration of multiple scales of planning and monitoring, chances for success may be substantially reduced. New approaches that integrate a holistic suite of restoration treatments, including changes in management of livestock, are essential. Results of recent restoration modeling for sage-grouse provide a starting point for development of multiscale strategies that could facilitate effective recovery of key habitats across large areas of sage-grouse range. In addition to restoration planning and monitoring for sage-grouse, efforts that consider a comprehensive set of sagebrush-dependent species are needed. An example is the development of a habitat network and related multispecies approaches for restoration planning (Wisdom and others 2002c), which could facilitate a more holistic recovery of the sagebrush ecosystem. Without such efforts, managers will be faced with a high likelihood of continued habitat loss and increasing extirpation risk for species that depend on sagebrush habitats.

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Big Sagebrush Response to One-Way and Two-Way Chaining in Southeastern Utah

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Abstract—A decadent, mixed stand of Wyoming big sagebrush, *Artemisia tridentata wyomingensis*, and mountain big sagebrush, *Artemisia tridentata vaseyana*, located north of Cisco, Utah, was subjected to one-way and two-way chaining treatments in November 1987. The effect of the treatments on plant community characteristics and shrub vigor was documented over a 3-year period. Stand density was reduced 60 percent on sites chained two ways and 43 percent on sites chained over once. Shrubs on one-way chained sites produced more leader growth in 1989 and 1990 than those on untreated sites or sites chained two ways. Browse production on one-way chained sites surpassed that of untreated sites and two-way chained sites by 140 percent and 350 percent, respectively. Over the short term, a one-way chaining was shown to be an effective method for improving sagebrush vigor and production on a critical mule deer winter range.

Introduction

The importance of big sagebrush as winter forage for mule deer has received considerable attention as researchers have attempted to determine the species' role in mule deer nutrition. There is widespread agreement that it is frequently a dominant constituent of the winter diet of mule deer (Kufeld and others 1973; Leach 1956; Pederson and Welch 1982). Research shows it to be a highly digestible food during the winter (Urness and others 1977; Wallmo and others 1977; Welch and Pederson 1981), and its nutritional value is enhanced when taken with other forages (Nagy and others 1964; Smith 1959). Although its importance in the diet increases in late winter (Carpenter and others 1979; Wallmo and others 1977), it is not a starvation food, and has been shown to contribute substantially to the diet of mule deer in the fall and early winter when other forages are available.

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Historically, mule deer carrying capacity has increased on winter ranges that have experienced an increase in shrub density (Urness 1979). However, on many sagebrush ranges in Utah, shrub density has peaked, and plant communities have become dominated by old, decadent shrubs. Browse production has decreased dramatically in these areas, especially during recent drought periods (Davis and others 2000).

In the 1960s, Plummer and others (1968) suggested chaining as a viable method for thinning stands that had become extremely dense and decadent. This method was suggested with or without seeding on depleted game ranges to retain native grasses and forbs. Similarly, Cain (1971) reported on the effectiveness of the Ely chain in restoring depleted rangelands dominated by pinyon-juniper and big sagebrush. Mechanical treatment, as opposed to spraying or burning, allows for controlled or partial treatments where an adjustment in relative shrub density is desired (Urness 1979). Such treatments retain shrubs in the treated area and avoid the problems associated with establishing shrubs from seed on semiarid sites.

This study compared one-way and two-way chaining as tools for restoring decadent stands of big sagebrush on important big game winter ranges. Treatment and control sites were identified during the planning phase of a 134-ha chaining project. Two sites were used to evaluate each management alternative (no chaining, one-way chaining, and two-way chaining). The unmodified anchor chain that was used for the project measured 76 m in length and weighed 60 kg/m.

Location

The study site was located on an important mule deer winter range at the base of the Book Cliff Range, just east of Nash Wash in southeastern Utah. The elevation of the study area ranged from 1,945 to 1,980 m. The deer winter range was characterized by a narrow band of pinyon-juniper woodland, interspersed with sagebrush parks. The parks were dominated by Wyoming big sagebrush (*Artemisia tridentata wyomingensis*), but mountain big sagebrush (*Artemisia tridentata vaseyana*) was also present, a common situation on sagebrush winter ranges in the region (McArthur, personal communication).

Methods

Plant Community Assessment

A 30-m baseline was located at each of the six study sites following treatment. Three 30-m transects were run perpendicular to the baseline and centered on the 15-m mark. The transects were randomly located at 3-m intervals along the line. A 1-m^2 quadrat was positioned at 3-m intervals on alternating sides of the transect to yield 10 sampling quadrats per transect and 30 quadrats per site.

Cover was estimated for all plants, individual species, bare ground, litter, rock, and cryptogamic crust within each quadrat using a modified Daubenmire (1959) method described by Davis and others (2000). Shrub density was estimated along a 0.005-ha strip plot centered on the 30-m survey tape. Each shrub rooted within the strip was counted and placed in one of the following age classes: seedling, young, mature, decadent, or dead (USDI-BLM 1996). Mature shrubs were classified as decadent when 25 percent or more of the branches were dead. A one-way analysis of variance test was used to determine if mean plant cover and shrub density values differed among treatment and control sites. Sagebrush production was estimated at each site using a modified method for predicting annual production based on shrub crown measurements (Dean and others 1981). The modification involved estimating crown denseness as percent cover in two dimensions, rather than percent volume in three-dimensional space. Regression equations were developed for each site to predict annual biomass from crown measurements. Shrubs selected for analysis were aged according to the method described by Ferguson (1964). Stem diameter and annual ring count data for each shrub were subjected to regression analysis to allow prediction of ages from stem diameter measurements.

Shrub Vigor Assessment

At 2-m intervals along each baseline, the nearest mature sagebrush plant was selected for the shrub vigor analyses. Subspecies determinations were made for each shrub according to the method described by Stevens and McArthur (1974). Baseline shrubs were sampled in fall to obtain estimates of annual leader growth, number of seed stalks per shrub, and seed stalk frequency, weight, and length. A one-way analysis of variance test was used to determine if mean shrub vigor measurements differed among treatment and control sites.

One of each of the replicated treatment and control sites was selected to evaluate browse utilization by mule deer. The first 10 shrubs selected for the shrub vigor assessment along the baseline of each of the three sites were included in the utilization study following the method described by Smith and Urness (1962). The "use-index" (leader length times percent use) described by Smith and Urness (1962) was used to evaluate treatment effects on browse use by mule deer.

Results

Precipitation

A rain gauge was located within 1.6 km of the study area and monitored monthly by the USDI-BLM. A summary of moisture received from November to May and from June to October is given for the years 1987 to 1990 in table 1. Winter precipitation, which greatly influences annual vegetative growth, decreased steadily from 203 cm (1987 to 1988) to 91 cm (1989 to 1990). The severity of the drought in the Nash Wash area was verified by observations of pinyon pine (*Pinus edulis*) trees at lower elevations that died in 1989 and 1990. The dead trees were scattered across the landscape, and many exceeded 100 years in age (Davis, personal communication).

Plant Community Responses

Shrub Density and Production—Pretreatment shrub density averaged 18,500 plants/ha on the six sites. Density estimates in the final year (1990) showed one-way chained sites supporting 10,500 plants/ha and two-way chained sites supporting 7,500 plants/ha, for reductions of 43 percent and 60 percent, respectively. Sagebrush seedling establishment was negligible on treated and control sites (<1 percent). Annual production (kg/ha) of shrubs in the one-way chained treatment was consistently greater than in the two-way chained treatment or the control from 1988 to 1990 (table 2). Over the 3 years, browse production on one-way chained sites surpassed that on untreated sites and two-way chained sites by 140 percent and 350 percent, respectively.

Table 1—A summary of moisture received (mm) from November to May and June to October, 1987 to 1990, at a BLM precipitation gauge 1.6 km from the Nash Wash study site.

	1987 to 1988	1988 to 1989	1989 to 1990
	- - - - - -	*mm*	- - - - - -
June to October	236	91	99
November to May	203	155	91
Total	439	246	190

Table 2—Browse production on chained one-way, two-way, and control treatments[a].

Treatment	Year 1988	1989	1990	Average
	- - - - - -	*kg ha^{-1}*	- - - - - -	
One-way	430	570	480	490[a1]
Two-way	130	130	160	140[b]
Control	370	380	300	350[c]

[a]Values not sharing the same letter superscript are significantly different at the 0.05 level.

Age Structure—Seedling establishment did not contribute significantly to shrub density of the treatment or control sites at Nash Wash. In the 3 years following the chaining, seedlings averaged about 1 percent of the stand density on treated sites and were almost nonexistent on untreated sites.

The untreated sites supported a shrub population that had a higher proportion of older shrubs than was found on either treatment. The regression equation generated to predict age (y) based on stem diameter (x) was $y = 0.83x + 12.26$, with a coefficient of simple determination of 0.52. Even though only 52 percent of the variation in age was accounted for by measuring stem diameters, some trends were apparent. Shrub age on control sites ranged from 32 to 97 years and averaged 54 years. Shrubs averaged 43 years of age (range: 19 to 62 years) on the one-way chained sites and 39 years (range: 19 to 62 years) on the two-way chained sites. Older shrubs with wide stem diameters lacked flexibility, offered resistance to the force of the chain, and were more effectively uprooted than were the younger plants.

Although age-class determinations (seedling, young, mature, and decadent) can vary among observers, their value in characterizing shrub populations has been recognized by many researchers and land management agencies. There was no significant difference in the contribution made by each age-class on one-way and two-way chained sites. However, a higher proportion of decadent shrubs was consistently found in the untreated sites (table 3). The widest margin of difference was noted in 1990 following the dry winter of 1989 to 1990. The proportion of older shrubs classified as mature rose to a high in 1989 on all sites.

Plant Cover—Cover class estimates for sagebrush did not reveal cover differences between treatments, but did show a difference between treatments and controls. The 3-year average for sagebrush cover on treated sites and control sites was 6 percent and 14 percent, respectively. Total herbaceous plant cover was highly variable and limited on all sites. The 3-year average for herbaceous plant cover on one-way chained, two-way chained, and control sites was 7 percent, 14 percent, and 5 percent, respectively. Cheatgrass made up about 50 percent of the herbaceous plant cover on all sites (table 4). The extremely low cover values for all sites in 1990 reflected the severity of the drought.

Shrub Vigor Responses

Leader Growth—The most striking impact of the thinning treatments was an increase in the length of vegetative leaders on remnant shrubs. Average leader length for sagebrush in treated areas was almost double that of control sites in 1988 (table 5). No significant difference in leader growth between the two treatments was observed the first year after chaining, but sagebrush vegetative growth on one-way chained sites surpassed that of two-way chained sites in 1989 and 1990. Essentially no growth was observed on control sites in 1990 due to the drought. In addition, leader growth in 1990 on shrubs from treated sites was well below the average observed on control sites during the previous two growing seasons.

Seed Production—The reproductive growth response to thinning was more gradual than the vegetative growth response. During the first growing season, seed stalk production was similar on treated and untreated sites. No significant differences were detected in the average number of seed stalks per shrub or the average length of seed stalks among treatments and controls (table 6). In 1989, the number of seed stalks per shrub increased slightly (although not significantly), and seed stalk length on treated sites

Table 4—Total herbaceous cover and percent contribution from cheatgrass (in parentheses) on chained one-way, two-way, and control treatments[a].

Year	One-way	Two-way	Control
		percent	
1988	11[a] (45)	18[b] (61)	10[a] (90)
1989	9[a] (67)	18[b] (50)	4[a] (25)
1990	2[a] (50)	5[a] (20)	2[a] (50)

[a]Values not sharing the same letter superscript are significantly different at the 0.05 level.

Table 5—Average sagebrush leader length in chained one-way, two-way, and control treatments[a].

	Year		
Treatment	1988	1989	1990
		cm	
One-way	7.5[a]	9.4[a]	1.4[a]
Two-way	7.1[a]	7.3[b]	1.2[b]
Control	4.0[b]	3.9[c]	1.0[c]

[a]Values not sharing the same letter superscripts in each column are significantly different at the 0.05 level.

Table 3—Proportion of decadent shrubs in the stand on chained one-way, two-way, and control treatments[a].

Shrub	Year								
	1988			1989			1990		
condition	1-way	2-way	Control	1-way	2-way	Control	1-way	2-way	Control
					percent				
Decadent	86	90	99*	59	58	64	76	74	98*
Not decadent	14	10	1*	41	42	46	24	26	2*

[a]Proportions followed by asterisks are significantly different (<0.05 level) than those of the other treatments based on categorical analysis using the chi-square statistic.

Table 6—Seed stalk length and number per shrub in chained one-way, two-way, and control treatments[a].

	Year					
	1988		1989		1990	
Treatment	Length	Number	Length	Number	Length	Number
	cm		*cm*		*cm*	
One-way	16.9[a]	0.5[a]	17.0[a]	3.3[a]	5.1[a]	14.7[a]
Two-way	17.2[a]	0.4[a]	15.1[a]	1.3[a]	4.9[a]	10.2[a]
Control	10.1[a]	1.0[a]	9.4[b]	1.3[a]	2.4[a]	0.3[b]

[a]Values not sharing the same letter superscripts in each column are significantly different at the 0.05 level.

exceeded that on control sites by 170 percent. In addition to almost a doubling in length, seed stalk weight per cm was 6 times greater on shrubs from treated versus control sites, averaging 0.097 gm/cm and 0.016 gm/cm, respectively. In 1990, shrubs from treated sites produced far more seed stalks than shrubs from control sites; however, stalk length did not vary significantly (table 5).

Browse Use by Mule Deer—Treatment effects on browse utilization by wintering mule deer are summarized in table 7. No significant differences were detected in browse use between the two treatments, but there were differences between treatments and controls. Deer utilized more forage from shrubs in treated areas than untreated areas even though utilization percentages were comparable. Average utilization for each treatment and control ranged from 67 to 77 percent during winter 1988 to 1989 and from 51 to 67 percent in 1989 to 1990. Significantly higher use-index values (leader length times percent use) were associated with shrubs from treated versus untreated areas during both winters.

Discussion

Utah Division of Wildlife Resources (UDWR) and BLM biologists have recognized the depleted condition of the deer winter range in the Nash Wash area since the mid-1960s. The close proximity of this critical winter range to the adjacent salt desert shrub type at lower elevations, traditionally used by sheep during the winter, has led to excessive use of sagebrush and preferred herbaceous species. Coles and Pederson (1967) were concerned that the combined winter use by deer and sheep on sagebrush in the Nash Wash area was excessive, and that if allowed to continue, would result in a widespread sagebrush die-off. In an effort to avoid such a loss, the Grand Resource Area (USDI BLM) modified the grazing plan for the Nash Wash area and limited winter sheep grazing to the salt desert shrub community below the critical deer winter range.

The increase in forage production per shrub on chained sites revealed the high degree of intraspecific competition that existed among sagebrush plants in the study area. Shrub density on one-way chained sites was sufficient to out-produce untreated control sites the following year and in each succeeding year of the study. Sites chained two ways did not respond with sufficient browse production to compensate for the additional shrub removal. The second pass reduced sagebrush production to 50 percent of that found on control sites.

The reproductive growth response to sagebrush thinning was not as striking as the vegetative response. A gradual increase in reproductive growth response to sagebrush thinning (both treatments combined) was observed over the 3-year study period. No significant differences were observed in stalk length, weight, or number the first year following treatment. A reproductive response was observed the second year in increased seed stalk lengths and stalk weights. Seed stalk production (number per plant) did not increase until the third flowering period following treatment.

Several factors appeared to be working against sagebrush productivity in the study area. In general, the sagebrush plants were old and decadent. Shrub age averaged 54 years, and the oldest shrubs were close to 100 years in age. Seasonal precipitation patterns, typical of the Colorado Plateau, may have limited the potential for sagebrush production at Nash Wash. Weather data collected at Thompson, UT (16 km from the study site), over the past 30 years, showed that only 55 percent of the annual precipitation was received during the winter (November to May) period. Stations at the Salt Lake City Airport and Fillmore reflected the trend in the Great Basin where approximately 70 percent of the annual precipitation was received during the November to May period. Since winter precipitation is known to influence big sagebrush production (Daubenmire 1975; Elderkin and others 1986), and only 55 percent of the annual precipitation at Nash Wash is received during the winter, shrub health may be vulnerable, especially during drought periods. Drought conditions were exacerbated at Nash Wash, where competition among sagebrush plants was evident. Although not evaluated in this study, the prevalence of cheatgrass, a

Table 7—Browse utilization by mule deer on chained one-way, two-way, and control sites[a].

	Year					
	1989			1990		
Treatment	Leader length	Use	Use index	Leader length	Use	Use index
	cm	*percent*		*cm*	*percent*	
One-way	5.1[a]	67[a]	3.2[ab]	4.7[a]	64[a]	3.2[a]
Two-way	5.4[a]	76[a]	4.1[a]	4.4[a]	67[a]	3.0[a]
Control	2.7[b]	77[a]	2.1[b]	2.2[b]	51[b]	1.1[b]

[a]Values not sharing the same letter superscripts in each column are significantly different at the 0.05 level.

winter annual, could have had a significant impact on sagebrush production and seedling establishment.

Chaining in one direction with a light, unmodified anchor chain, was shown to be an effective treatment for thinning a decadent stand of sagebrush without reducing the carrying capacity for a wintering deer herd. The treatment should be considered with or without artificial seeding, depending on the condition of the herbaceous understory, as a practical method to improve sagebrush health and wildlife habitat on critically important sagebrush ranges. Different results would be expected from chaining one-way with heavier chains, chains modified with rails welded crosswise to the links, or chains pulled in a more aggressive "J" configuration as opposed to the "U" shape maintained during the Nash Wash project.

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Seeding Considerations in Restoring Big Sagebrush Habitat

Scott M. Lambert

Abstract—This paper describes methods of managing or seeding to restore big sagebrush communities for wildlife habitat. The focus is on three big sagebrush subspecies, Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*), basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*), and mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*). Natural colonization of the native plant community may be the preferred management action on sites where native seed sources are available to successfully reestablish the desired wildlife habitat. On highly disturbed or otherwise damaged sites and where competition from weeds is excessive, seeding will be utilized to restore big sagebrush for wildlife habitat. Big sagebrush seed is never seeded alone in site rehabilitation and restoration projects. The best time to seed or interseed big sagebrush seed mixes, including grasses and forbs, is in late fall or early winter. The overall best method to reestablish big sagebrush is to use a rangeland drill at a shallow setting following site preparation, including tillage and weed control. When big sagebrush is drill seeded with other seed types, it is recommended that it be seeded through a separate drill box to permit very shallow seeding and proper seed placement for plant establishment. Seedings of native plants, including big sagebrush, should be protected from grazing for at least 3 to 5 years to allow time for the shrubs and forbs to become established.

Introduction

Big sagebrush dominated plant communities occupy over 25 percent (67 million acres) of the landscape in the Great Basin region of Idaho, Utah, Nevada, Oregon, and California. A total of about 96 million acres in the Western States has historically been big sagebrush habitat (Blaisdell 1953). In some areas, over one-half of the big sagebrush areas have been severely disturbed, altered, or even removed by wildfire, grazing, prolonged drought, and other natural events or human activities. This paper will concentrate on seeding considerations for habitat restoration, including big sagebrush (*Artemisia tridentata*) subspecies Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*), basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*), and mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*).

Beginning in the 1930s and 1940s, big sagebrush-dominated plant communities were reseeded to introduced forage grasses and forbs such as crested wheatgrass (*Agropyron cristatum*), Siberian wheatgrass (*Agropyron fragile* ssp. *sibericum*), alfalfa (*Medicago sativa*), and pubescent wheatgrass (*Elytrigia intermedia*). Intermediate wheatgrass (*Elytrigia intermedia*) was one of the preferred introduced species seeded on mountain big sagebrush sites, which usually have higher annual precipitation. These plant species were often used to improve forage production and reduce soil erosion after wildfires and other site disturbances.

Since the 1990s, the emphasis has been to restore wildlife habitat with a diversity of native plant species on public lands in the Western States. Seeds of many native and introduced species are commercially available, including certified cultivars and source-identified native germplasms.

Basin big sagebrush and Wyoming big sagebrush seeds are light brown to grayish brown or black and very similar in size and shape (Parkinson 2004). Mountain big sagebrush seeds are slightly larger and darker than basin big sagebrush or Wyoming big sagebrush seeds.

One proven method to obtain seed of the desired big sagebrush subspecies is to procure Certified Source Identified (SI) seed of the big sagebrush subspecies appropriate, or adapted, to the site. SI seed has been verified as to the subspecies. The SI seed is collected from mature plants growing on an identified natural site.

Identification of big sagebrush seed to subspecies may be difficult to determine just by looking at the seed. In the past, Bureau of Land Management and others that seeded big sagebrush seed were not always provided with seed of the subspecies that was specified in seeding contracts. Wyoming, basin, and mountain big sagebrush establish and thrive on sites with different environmental conditions. Using seed of the inappropriate subspecies of big sagebrush may be a reason for stand failure on some sites (Lysne and Pellant 2004).

The habitat management option of "do not seed" has been the action taken on about 50 percent of the potential rehabilitation or restoration sites on public land in the Great Basin States (Lambert and Hamby 2003). Natural colonization of native or introduced species is often allowed to occur on sites where the seeds of desired plants exist in the soil seed bank or on adjacent lands. Natural plant recovery will only be successful on those sites with adequate soil and moisture conditions and where competition from weeds is not a problem.

Some big sagebrush site restoration projects, especially those in Wyoming big sagebrush habitat, may initially require seeding for soil stabilization and weed control. These seedings may include adapted introduced plant species. At a later time, other desirable native plants can be seeded (or

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interseeded) to provide a greater diversity of native species for wildlife habitat.

Descriptions of Big Sagebrush Subspecies

Basin Big Sagebrush (*Artemisia tridentata* ssp. *tridentata* Nutt.)

Basin big sagebrush is an evergreen shrub commonly 3 to 6 feet, but sometimes as much as 14 feet tall and 5 to 8 feet wide. It is erect, spreading, and heavily branched, with an uneven top. This subspecies of big sagebrush is native to the Interior Western United States, including the Great Basin. It is found on well-drained, moderately deep to deep loam or silt loam soils, and at elevations up to 9,500 feet. The moisture regime is semidry; 8 to 18 inches of mean annual precipitation is required. Basin big sagebrush is the most abundant shrub in the Western United States aridlands (Passey and others 1982). It is sometimes interspersed with Wyoming big sagebrush, and may also occur in riparian sites. It is generally fire intolerant.

Basin big sagebrush averages 2,500,000 cleaned seeds per pound (McArthur and others 1979); seed is harvested in the late fall. The drilled seeding rate is 0.1 to 0.2 lb pure live seed (PLS) per acre in a seed mixture. Big sagebrush seed is planted very shallow (approximately 1⁄16 inch deep). Caution: Basin big sagebrush seed is very short-lived and may be fragile when improperly handled. The seed must be kept in cold storage (<38 °F and <25 percent relative humidity [RH]). It is recommended the seed be used within 1to 2 years of harvest.

Mountain Big Sagebrush (*Artemisia tridentata* ssp. *vaseyana* [Rydb.] Beetle)

Mountain big sagebrush is an evergreen shrub that is usually less than 3 feet tall with a spreading and even topped crown. It is native to the Interior Western Unites States, including the Great Basin region. It occurs on soils that are deep and well drained with pH usually about 7.0. It is naturally found in association with mountain shrub plant communities at moderate to high elevations, up to 10,000 feet. Its moisture regime is semidry; at least 13 inches of mean annual precipitation is required to maintain a population of this subspecies. After fire, mountain big sagebrush often recovers from seed remaining in the soil. On many restoration sites, mountain big sagebrush sites do not require reseeding unless there is an overwhelming population of weedy plants. In such situations, weed control would be required prior to seeding.

Mountain big sagebrush averages 2,250,000 cleaned seeds per pound (McArthur and others 1979); the seed is harvested earlier in the fall than the other two subspecies. The drilled seeding rate for mountain big sagebrush is 0.1 to 0.2 lb PLS per acre in a seed mixture. The seed should planted about 1⁄16 inch deep in the soil (Jacobson and Welch 1987). 'Hobble Creek' is a native cultivar that originated in Wasatch County, Utah, and was released by the USDA Forest Service, Shrub Sciences Laboratory, Provo, Utah (Welch and others 1986). Caution: mountain big sagebrush seed is very short-lived and may be fragile when improperly handled. The seed must be stored in cold storage (<38 ° F) with humidity maintained at <25 percent R.H. It is recommended the seed be used within 1 to 2 years of harvest.

Wyoming Big Sagebrush (*Artemisia tridentata* ssp. *wyomingensis* Beetle & Young)

Wyoming big sagebrush is an evergreen shrub usually up to 3 feet tall and usually up to 3 feet wide at maturity. It is basally branched and rounded in form with an uneven top. It is native to the Interior Western United States, including the Great Basin region. This subspecies grows on shallow, gravelly, or sandy to silt clay loams at elevations from 2,000 to 7,000 feet. Its moisture regime is dry; 7 to 14 inches of mean annual precipitation is required for Wyoming big sagebrush. It is considered to be fire-intolerant and does not resprout after wildfire.

Wyoming big sagebrush averages 2,500,000 cleaned seeds per pound (McArthur and others 1979); the seed is harvested in late fall. The drilled seeding rate for Wyoming big sagebrush is 0.1 to 0.2 lb PLS per acre in a seed mixture. The seed is planted shallow in soil (1⁄16 inch in depth). Cultivar: 'Gordon Creek' Wyoming big sagebrush originated in Carbon County, Utah, and was released by the USDA Forest Service, Shrub Sciences Laboratory (Welch and others 1992). Currently there is no commercial seed production of Gordon Creek. Caution: Wyoming big sagebrush seed is very short-lived and may be fragile when improperly handled. The seed must be stored in cold storage (<38 °F) with humidity maintained at <25 percent RH. It is recommended the seed be used within 1 to 2 years of harvest.

Seeding Specifications

Seeding Dates for Big Sagebrush

Late fall is the overall best time to seed big sagebrush, and rangeland drill seeding is the recommended method. Early winter is the preferred time for aerial seeding. Whichever seeding method is used, it is best to seed prior to any significant snowfall.

Seeding Requirements

A firm, packed, weed-free seedbed should be prepared on restoration sites. The best soils for establishing big sagebrush from seed are silt loam to sandy loams, deep to shallow and well drained. One of the best methods for successful establishment of big sagebrush by seed is to broadcast the seed on the soil surface and then lightly rake or harrow to barely cover the small seed. Another method of big sagebrush establishment that has been successfully used is to apply the seed with a rangeland seed drill (Truax, Tye, and so forth). The seed drill is set to place the seed very shallow, about 1⁄16 inch deep. The packer wheels of the seed drill should lightly cover the seed with soil. The best chance for a successful seeding is if the seed drill has a separate seed box for the big sagebrush seed.

Aerial or broadcast seeding has been successful on some sites in late fall or just prior to snowfall in early winter. Aerial seeding may not be as good a seeding method as a properly installed drilled seeding or broadcasting seeding followed by harrowing to lightly cover seed with soil (Lysne and Pellant 2004).

Seeding Rate for Big Sagebrush Subspecies

The typical drill seed rate, pure live seed (PLS), for all three big sagebrush subspecies is 1⁄10 lb per acre in a seed mix. Basin big sagebrush has, on average, 2,500,000 cleaned seeds per pound (1⁄10 lb = 57 seeds per square foot). Mountain big sagebrush has 2,250,000 cleaned seeds per pound (1⁄10 lb = 51 seeds per square foot). Wyoming big sagebrush has 2,500,000 cleaned seeds per pound, according to McArthur (1979) (1⁄10 lb = 57 seed per square foot).

Seed Mixes

Challenges to Seeding Small Seeds—Big sagebrush seed is much smaller than most other species that are typically drill seeded. Big sagebrush seedlings are slow growing and often less competitive than most other species that occur in the same habitat. The best options to establish big sagebrush are to use a rangeland drill with a separate seed box for big sagebrush or broadcast seed and very lightly cover with soil. An additional option is to grow the seed in a greenhouse to produce container seedlings and then transplant to the field location, or grow the seed in a field to produce bare-root stock and transplant to the desired site.

Establishing Big Sagebrush With Understory Species—Big sagebrush subspecies are not seeded alone. Sagebrush seed is included in seed mixtures with other species, usually grasses and forbs. If big sagebrush is drill seeded, it is recommended that it be seeded through a separate drill box to permit very shallow seeding. The drilled seed mix should be installed at the PLS rate of 60 to 80 seeds per square foot. To determine a seed mix for your specific ecological situation, utilize information available through the PLANTS database (http://plants.usda.gov/). The PLANTS Web site includes links to *VegSpec*, a database to help with seeding prescriptions, and the Ecological Site Information System (ESIS).

The following seed mixes are examples for each of the three big sagebrush subspecies and include understory grasses and native forbs for habitat restoration projects (tables 1, 2, and 3). Examples of two seed mixes for soil stabilization and erosion control are also provided (table 4).

Checklist of Potential Causes for Seeding Problems or Failure

The seed doesn't grow when initially planted.

1. The seed selected was not appropriate for the given environment. This problem may be due to inaccurate seeding specifications or inadequate seed availability. For many native species, a local seed source may be preferable to those sources that originate further away from the seeding

Table 1—Example of generic basin big sagebrush steppe seed mix, drilled rate (PLS).

Native species	Pounds per acre
Basin big sagebrush	0.1
Rubber rabbitbrush	0.1
Bluebunch wheatgrass	2.0
Indian ricegrass, on sandy soil	2.0
Thurber's needlegrass	1.0
Sandberg's bluegrass	1.0
Bottlebrush squirreltail	2.0
Six-weeks fescue	1.0
Pale agoseris	0.2
Threadstalk milkvetch	0.2
Western yarrow	0.1
Arrowleaf balsamroot	0.2
Tapertip hawksbeard	0.2
Nineleaf biscuitroot	0.2
Fleabane spp.	0.1
Total	8.4

Table 2—Example of generic mountain big sagebrush community seed mix, drilled rate (PLS).

Native species	Pounds per acre
Mountain big sagebrush	0.1
Woods rose/mountain snowberry	0.5
Sulfur-flowered buckwheat	0.2
Idaho fescue	2.0
Bluebunch wheatgrass	2.0
Bottlebrush squirreltail	1.0
Slender wheatgrass	1.0
Pale agoseris	0.2
Woollypod milkvetch	0.2
Arrowleaf balsamroot	0.2
Tapertip hawksbeard	0.2
Sagebrush mariposa-lily	0.2
Total	7.8

Table 3—Example of generic Wyoming big sagebrush steppe seed mix, drilled rate (PLS).

Native species	Pounds per acre
Wyoming big sagebrush	0.1
Rubber rabbitbrush	0.1
Indian ricegrass (on sandy site)	2.0
Thurber's needlegrass	1.0
Sandberg's bluegrass	1.0
Bottlebrush squirreltail	2.0
Six-weeks fescue	1.0
Pale agoseris	0.2
Threadstalk milkvetch	0.2
Western yarrow	0.1
Tapertip hawksbeard	0.2
Nineleaf biscuitroot	0.2
Fleabane spp.	0.1
Total	8.25

Table 4—Samples of soil stabilization seed mixes for sites with less than 12 inches mean annual precipitation.

Species and variety	Pounds per acre
	PLS, drilled rate
A. Species for a sandy soil site	
Siberian wheatgrass, 'Vavilov' or 'P27'	2
Indian ricegrass, 'Nezpar'	1
Thickspike wheatgrass, 'Critana', 'Bannock', or 'Schwendimar'	1
Sand dropseed	1
Alfalfa, 'Ladak'	2
Total	7
B. Species for a silt-clay loam soil site	
Crested wheatgrass, 'Nordan' or 'Hycrest'	2
Western wheatgrass, 'Arriba'	1
Snake River wheatgrass, 'Secar'	2
Pubescent wheatgrass, 'Luna'	1
Sainfoin, 'Eski'	1
Total	7

location in terms of distance, elevation, or other site factors. The seed from nonlocal collections or seed production fields may be less well adapted to the restoration site. Seed not adapted to the site may have no or a low percentage of seeds that germinate, establish successfully, and persist over time.

2. The seed may remain dormant in the soil for varying periods of time. Most seed planted in late fall or early winter will germinate the following spring. Some seed, such as the hard-coated seeds of Indian ricegrass, may take several years under normal conditions before initiation of germination.

3. There may have beeen poor seed storage conditions prior to seeding. In general, seed stored for more than 2 weeks must be held in a climate-controlled warehouse with temperatures not to exceed 80 °F and 30 percent RH.

4. The seed delivered was not viable. This results when viability has declined following testing or when seed is damaged during transport or handling. To overcome this potential problem, arrange for a certified seed sampler to confirm or deny the original seed tag information by taking seed samples of the seed lots prior to mixing. Then have the seed samples sent to a certified seed lab for purity and germination or TZ tests and noxious weed seed analysis. Do not accept seed lots with low germination rates or unacceptable PLS. Acceptable germination rates or PLS percent should be determined by the agency or by the Seed Certification standards for the seed type or species. An example of the Certified seed standard minimums set for bluebunch wheatgrass are 85 percent pure seed, 80 percent germination, and 68 percent PLS (AOSCA 2001).

5. The seed was planted at too great a depth. Be sure that all seed drills and other equipment are set to install the seed in the soil correctly before you start the seeding operation.

6. The seed was damaged during application. Some damage to seed may occur with rough handling, transportation of seed to the field site, or during hydroseeding applications.

7. If seed is applied in mulch, the medium may not be capable of sustaining seed germination on dry sites. Some temporary mulches, especially those made of recycled newsprint, can contain inks and metals that are toxic to the newly germinated seedlings.

8. The time of seeding was past the normal germination season for that species or population. This problem can vary among species. Some species will actually germinate under snow cover during the winter, while others will break seed dormancy and germinate in the following spring or summer.

The seed has germinated and later dies off.

1. Soil nutrients and moisture content are insufficient to sustain seedling growth.

2. The soil type, such as silt loam, clay loam, or sandy loam, is not capable of sustaining seed germination and seedling development of plants that were seeded.

3. Diseases, such as damping off (a fungal disease), may kill seedlings soon after germination.

Environmental conditions may cause additional stress and prevent seedling establishment.

1. Conditions that may increase seedling mortality include drought, excessive heat or cold spells, wind, flood, early frost, or late frost.

2. Competing vegetation, especially weeds such as cheatgrass (*Bromus tectorum*), will cause seedling mortality in less competitive plant species (West and Hassan 1985). Control of weeds may be essential to seedling establishment.

3. After seeding it was found that the seeding mix contained some weed seeds that germinated and took over the site. To overcome this problem, arrange for all seed lots to be tested by a certified seed lab for noxious and other weed seeds prior to mixing. Do not accept seed lots with unacceptable weed seeds on the seed lab analysis.

4. Animals and insects may have eaten the seed/seedlings. You may need to provide some manner of protection to emerging seedlings. Seedings of native plants, including big sagebrush, should be protected from grazing for at least 3 to 5 years to allow time for the shrubs and forbs to become established.

5. The soil may lack the microorganisms and fungal mycorrhizae necessary for seedling establishment. Inoculation of seed or soil with microorganisms may be necessary for plant health.

Monitoring Seedings

Use monitoring protocols from the BLM/NRCS Monitoring Handbook (Elzinga and others 1998) or other monitoring protocols as *determined by* the administering agency to evaluate the success of seedings. On arid sites, a seeding is often considered to be successful if at least 0.5 plants per square foot are established.

Planting Big Sagebrush Using Seedlings

On some sites in the Western United States, good establishment of big sagebrush has resulted when seedlings are planted in early spring on locations with the "best" soils and aspect (Everett 1980). Arrangements should be made with a nursery or grower to purchase or produce big sagebrush seedlings when restoration plantings are the preferred plant establishment option. Field-grown bare-root stock or greenhouse-grown containerized plants may be used. The time of sowing seed in a production field to lifting of conservation grade bare-root stock ready for planting could be as much as 2 years. Greenhouse grown seedlings may be ready to plant within a 6 to 8 month time period.

To establish the big sagebrush seedlings it is recommended that they be randomly placed in clumps or blocks on the best sites for restoring big sagebrush. Seedlings planted in natural blocks or clumps become fertile islands of big sagebrush as they mature. These may establish additional plants throughout the adjacent areas.

Sources for Big Sagebrush Seed

Wildland Collected Seed (Source-Identified)—Big sagebrush seed is collected from natural big sagebrush stands. Most seeds of big sagebrush subspecies are sold as Source-Identified seed from wildland collection sites. Certified Source Identified seed of native plants is recommended for all rehabilitation and restoration seedings. The Source Identified Certification tag, usually yellow, verifies the species, County, State, elevation, seed lot number, and sometimes other geographic location information for the collected seed.

Commercial Seed Field Production—Certified seed, cultivars or source-identified, of native or introduced plants is recommended for all rehabilitation and restoration seedings. The Certified tag on a seed bag (blue tag) verifies the cultivar, species, location where the seed was grown, no noxious weed seed for the States specified, and the minimum percent pure seed and percent germination.

Hobble Creek is the only big sagebrush cultivar currently available in limited quantities from seed vendors. Native and introduced cultivars of other plant species are available in limited to good quantities from seed vendors (table 5).

Table 5—Partial list of commercial big sagebrush seed vendors (these seed vendors may have seed of other species also).

Vendor	Vendor
Barton Seed Co. 222 E. Union Street Manti, UT 84642 Phone: (435) 835-9200	Maughan Seed Co. PO Box 72 700 W. 2100 S. Manti, UT 84642 Phone: (435) 835-0401
Comstock Seed Co. 917 Hwy 88 Gardnerville, NV 89410 Phone: (775) 746-3681	NP Seed Co. 206 E. 300 S. Manti, UT 84642 Phone: (435) 835-8301
Fremont Trading Co. 450 S. 50 E. Ephraim, UT 84627 Phone: (435) 283-4701	Rainier Seed Co. PO Box 70 Port Orchard, WA 98367 Phone: (800) 828-8873
Geertson Seed Farms 1665 Burroughs Road Adrian, OR 97901 Phone: (541) 339-3768	Native-Seed Co. 7361 Pineridge Drive Park City, UT 84098 Phone: (435) 640-0557
Granite Seed Co. 1697 W. 2100 N. Lehi, UT 84043 Phone: (801) 768-4422	Plummer Seed Co. PO Box 70 Ephraim, UT 84627 Phone: (435) 283-4844
Harvest Moon Seed Co. PO Box 532 Richfield, UT 84701 Phone: (435) 979-8549	Stevenson Intermountain Seed Co. PO Box 2 Ephraim, UT 84627 Phone: (435) 283-6639
Intermountain Seed Co. Box 62 370 W. 300 N. Ephraim, UT 84627 Phone: (435) 283-4703	Southern Utah Seed Co. PO Box 124 192 W. 100 S. Junction, UT 84740 Phone: (435) 577-2142
Landmark Seed Co. N. 120 Wall St., Suite 400 Spokane, WA 99201 Phone: (509) 835-4967	Wagstaff Seed 1900 E. Oakhill Lane Wallsburg, UT 84082 Phone: (435) 654-3439

Summary

Natural colonization of the native plant community may be the preferred management action on sites where native seed sources are available to successfully reestablish the desired wildlife habitat. On highly disturbed or otherwise damaged sites or where excessive competition of weeds occurs, seeding will be used to restore the big sagebrush for wildlife habitat.

Big sagebrush seed is never seeded alone in site rehabilitation and restoration projects. The best time to seed or interseed big sagebrush seed mixes, including grasses and forbs, is in late fall or early winter. The best method to seed big sagebrush is by using a rangeland seed drill. The seed is drilled shallowly with the seed drill set for the appropriate planting depth and PLS seeding rate. Wyoming big sagebrush sites are usually the most arid and shallowest soil sites where big sagebrush naturally occurs. Mountain big sagebrush sites are those sagebrush sites, in general, with the coolest average temperature and annual rainfall greater than 12 inches. Basin big sagebrush sites are often transitional between the foothills and mountains where the mountain big sagebrush community is found and the arid lowlands are dominated by the Wyoming big sagebrush community. Basin big sagebrush often naturally occurs on microsites with deeper soils in Wyoming big sagebrush habitat and may be found along Interior Western riparian zones. After seeding, the restored sites should be rested, or protected, from grazing for at least 3 to 5 years to allow time for the shrubs, forbs, and grasses to become fully established for wildlife habitat.

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Big sagebrush

Strategies to Enhance Plant Structure and Diversity in Crested Wheatgrass Seedings

Mike Pellant
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Abstract—Crested wheatgrass (*Agropyron cristatum* sensu amplo [L.] Gaertn.) is an introduced, caespitose grass that has been seeded on millions of acres of Western rangelands. In some areas, crested wheatgrass seedings overlap with critical sage-grouse (*Centrocercus urophasianus*; *C. minimus*) habitat, raising the question of how plant diversity might be restored in these closed plant communities. A three-step process is described to reduce crested wheatgrass competition, introduce desired species, and manage to maintain desired species for use long term. Crested wheatgrass is a strong competitor with other species and a prolific seed producer, which hinders treatments to reduce its influence and improve conditions for establishment of desirable seeded species. Herbicides, burning, mechanical treatments, livestock grazing, droughts, and combinations of these are effective to varying degrees in reducing crested wheatgrass competition. Once crested wheatgrass competition is reduced, either seed or seedlings can be used to increase diversity in these seedings. Post-establishment management and monitoring are essential components of the strategy to maintain plant diversity into the future.

Introduction

Use of introduced species invokes a range of emotions from unequivocal support to outright opposition and has even been linked to fascism and racism (Simberloff 2003). In the Western United States, the planting of introduced perennial wheatgrasses for rangeland rehabilitation has and continues to be practiced after disturbances such as wildfires, on cropland taken out of production, and to increase forage production for livestock. Given the focus of this symposium on the restoration of habitat for sage-grouse, using crested wheatgrass (*Agropyron cristatum* sensu amplo [L.] Gaertn.) to meet certain land-use objectives must be evaluated relative to the millions of acres already planted to these grasses and their continued use in rangeland rehabilitation projects.

This paper will focus on a review of the characteristics, use, and control techniques for crested wheatgrass and other closely related introduced, caespitose bunchgrasses (Siberian wheatgrass [*Agropyron fragile*], and Russian wildrye [*Psathyrostachys juncea*]), prior to reintroducing plant diversity. Crested wheatgrass and its close relatives were introduced from Eurasia and selected for land rehabilitation in the Central and Western United States. The areas where crested wheatgrass has been extensively used in the Western United States overlap closely with historic sage-grouse distribution in the low elevation rangelands where annual precipitation ranges from 8 to 12 inches annually (USDA NRCS 2004). The first part of this paper will include a review of the historical use, competitive characteristics, and concerns regarding the use of crested wheatgrass, especially with regard to sage-grouse habitat. The remainder of this paper will emphasize potential treatments to reduce crested wheatgrass competition, where acceptable functional or structural vegetation components required by sage-grouse are not present, prior to increasing the diversity of desirable herbs and shrubs.

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Crested Wheatgrass: Introduction, Uses, and Issues in Western Ecosystems

The first collections of crested wheatgrass were made in 1897 to 1989 and again in 1906 from the dry steppes of Eastern Russia (Dillman 1946; Rogler and Lorenz 1983). These collections were classified as crested wheatgrass and desert wheatgrass (*Agropyron desertorum*) and were distributed to 15 experiment stations throughout the West. Minimal use of these introduced grasses occurred until the 1930s when a combination of cheap labor (for example, Civilian Conservation Corps), the "dust bowl" in the Midwest, and the need to reestablish perennial vegetation on abandoned farmlands prompted their increased use. Crested wheatgrass, primarily the caespitose bunchgrasses, were used extensively to revegetate abandoned croplands that were subject to wind erosion in the Northern Great Plains (Holechek 1981; Young and Evans 1986). The first planting of crested wheatgrass in the Intermountain area occurred in eastern Idaho in 1932 (Hull and Klomp 1966). As the need for plants to reclaim abandoned cropland increased, the production of seed of crested wheatgrass also increased (Sharp 1986). With greater demands for red meat production from Western rangelands during World War II, Congress allocated funds to convert unproductive sagebrush rangelands to more productive introduced grasslands (Young and McKenzie 1982). Researchers for the Forest Service developed a series of bulletins on rangeland seeding, emphasizing the use of crested wheatgrass and other introduced wheatgrasses in Idaho (Hull and Pearse 1943), Nevada (Robertson and Pearse 1943), and Utah (Plummer and others 1943).

Following World War II, the pace of rehabilitation accelerated again during the 1950s and early 1960s when millions of acres of Central and Western rangelands were seeded to crested wheatgrass. The objectives of these seedings included increasing forage for livestock, weed control, watershed stabilization, and reducing wildfire hazards. The use of crested wheatgrass to biologically suppress halogeton (*Halogeton glomeratus*), a nonnative poisonous forb, was funded by Congress in 1952 and ultimately paid for a major portion of crested wheatgrass seedings in Nevada and other Great Basin States (Young and Evans 1986).

The rehabilitation of degraded rangelands with crested wheatgrass was also accelerated by the development of equipment to control competitive plants and distribute seed effectively across a wide range of edaphic conditions. In particular, the development of the rangeland drill in the early 1950s hastened the ability of land managers to seed large acreages to crested wheatgrass, while the brushland plow provided managers with a tool for sagebrush removal prior to seeding (Young and McKenzie 1982). By the early 1970s, referred to by Young and Evans (1986) as the "golden age" of seeding crested wheatgrass, an estimated 12.4 million acres were seeded to this species (Dewey and Asay 1975). A recent report (USDI 2001) on the condition of public lands managed by the Bureau of Land Management in the Western United States indicates that approximately 5 million acres of rangelands have been seeded (USDI 2001), the majority of which we estimate included crested wheatgrass in the seed mixture. However, since this information is based in large part on inventory information collected in the mid-1970s, the acreage of public lands seeded in part with crested wheatgrass is expected to exceed this figure.

The use of crested wheatgrass has come under increasing scrutiny since the 1970s. Legislation, such as the Surface Mining Control and Reclamation Act (PL 95-87, 1977), required the use of native species for mine reclamation. The National Environmental Policy Act of 1969 (Pub. L. 91-190, 42 U.S.C. 4321-4347) required the preparation of an impact analysis on activities funded by the government; this included the use of introduced species in seedings. Federal agency guidance on this subject has also changed. Prior to 1984, the Bureau of Land Management's guidance on postwildfire seeding encouraged the use of introduced grass species given their cost, ease of establishment, and erosion prevention capability (USDI BLM 1981). More recently, Presidential Executive Order 13112 on Invasive Species (Clinton 1999) directs Federal agencies to use native species when feasible to restore ecosystems where invasive species are a problem. Finally, the BLM's Great Basin Restoration Initiative (GBRI) gives preference to the use of native species in seeding projects, "pending seed availability, cost and chance for success." (USDI BLM 2000).

Competitive Characteristics of Crested Wheatgrass

An understanding of the competitive characteristics of crested wheatgrass is essential in order to develop strategies to increase plant diversity in seedings dominated by this species. The same features that make crested wheatgrass appealing to land managers (for example, provide soil stability and compete with and control invasive species) can also result in community dominance of this species, displacement of native species, and reduced plant diversity (Broersma and others 2000; D'Antonio and Vitousek 1992; Marlette and Anderson 1986; Roundy and others 1997). Although some studies reported that crested wheatgrass is not "mobile" and does not deter the reestablishment of native species (Broersma and others 2000; Krzic and others 2000), several other studies have shown that established stands have spread beyond the original seeded area (Hull and Klomp 1966; Marlette and Anderson 1986). Other studies have shown that crested wheatgrass seedings resulted in near monospecific stands (Hull and Klomp 1966; Looman and Heinrichs 1973; Schuman and others 1982).

There are several characteristics of crested wheatgrass that contribute to its competitiveness with both invasive species and native vegetation. At the seedling stage, crested wheatgrass has an advantage over some native plants, due in part to its ability to efficiently capture nutrients and water (Bakker and Wilson 2001; Schuman and others 1982). Established crested wheatgrass plants were more efficient at securing phosphorus than native bluebunch wheatgrass (*Pseudoroegneria spicatum*) when both species were grown in association with big sagebrush (*Artemisia tridentata*) (Caldwell and others 1985). Big sagebrush was also negatively affected by the ability of crested wheatgrass to rapidly extract soil water during the same period that sagebrush requires this resource (Cook and Lewis 1963; Eissenstat and Caldwell 1988; Sturges 1977). Other studies have shown the competitive advantage of crested wheatgrass during the initial stages of plant establishment for Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) and antelope bitterbrush (*Purshia tridentata*) (Blaisdell 1949; Fortier 2000;Hall and others 1999; Schuman and others 1998). Native grasses may also reduce shrub seedling establishment; however, the effect is less than that exhibited by crested wheatgrass (Eissenstat and Caldwell 1988; Hubbard 1957). For example, Frischknecht and Bleak (1957) reported that seeded stands of bluebunch wheatgrass were more likely to permit sagebrush seedling recruitment than seeded stands of crested wheatgrass.

Another attribute that favors crested wheatgrass establishment is its prolific seed production. Marlette and Anderson (1986) germinated seed from a crested wheatgrass monoculture planting and reported around 500 crested wheatgrass seedlings per m^2. Pyke (1990) compared the demography of crested wheatgrass and bluebunch wheatgrass and found crested wheatgrass to have a decided advantage over the native grass in seed production, seed bank carryover, seed dispersal, and seedling survival. Seed production for crested wheatgrass ranged from 1,772 seeds per m^2 in a wet year to 1,037 seeds per m^2 in a dry year. Bluebunch wheatgrass seed production during this same period ranged from 26 seeds per m^2 in the wet year to no seed production in the dry year. In a recent study, Romo (2005) found an average return of 2 percent of crested wheatgrass seeds sown while Heidinga and Wilson (2002) reported a 4 percent return. For example, seed production rates of 1,000 per m^2 with a 3 percent rate of return would result in an initial count of 30 crested wheatgrass seedlings per m^2.

Carryover of germinable seed for more than 1 year is another competitive characteristic of crested wheatgrass. Under ideal seed storage conditions in a lab, crested wheatgrass seeds remained germinable for over 20 years (Ackigoz and Knowles 1983). Seed life is much shorter in the natural environment; however, crested wheatgrass has been observed by the authors to germinate in the second and, in a few cases, the third growing season after the seed was planted.

Sage-Grouse and Crested Wheatgrass

Crested wheatgrass has been planted or has the potential for establishment (Rogler and Lorenz 1983) over a large portion of the historic and current range of sage-grouse (Connelly and others 2004). The question of the quality of crested wheatgrass seedings as habitat for sage-grouse is difficult to answer since it depends on both spatial and temporal scales that are rarely monitored (Connelly and others 2004). The size of the seeding, juxtaposition on the landscape in relation to suitable habitat, species composition of the seeding, and sagebrush cover are other factors that affect the utility of crested wheatgrass seedings as sage-grouse habitat. A recent review of sage-grouse habitat needs and associated habitat threats does not directly identify introduced seedings as a factor in the decline of sage-grouse (Wambolt and others 2002). They indicated that the extreme reduction in canopy cover of sagebrush, and associated loss in understory plant diversity typical of some introduced species seedings, can significantly reduce sage-grouse habitat quality throughout the year.

Understory plant diversity is important to sage-grouse in the spring and summer, whereas sagebrush provides essential cover and forage in the winter and late fall (Connelly and others 2000; Wambolt and others 2002). Brood-rearing habitats with an array of plant species provide a diversity of insects important to sage-grouse, especially during brood rearing (Connelly and others 2000; Drut and others 1994). The recently published "Guidelines for Sage-Grouse Habitat" (Connelly and others 2000) identified grass height (over 18 cm) and canopy cover (greater than 15 percent for breeding and brood-rearing habitats) as important habitat requirements for nesting sage-grouse. Therefore, crested wheatgrass, in the appropriate proportions, could provide similar habitat structure compared to native bunchgrasses. Connelly and others (2000) recommend that nonnative species that are functionally equivalent to natives be used in restoration projects if native forbs and grasses are unavailable.

Another situation where the use of crested wheatgrass may be necessary is the restoration of habitat that is severely degraded and dominated by aggressive invasive species (Pyke 1994). Beginning in the 1930s numerous studies were conducted that showed the superior competitive ability of crested wheatgrass with cheatgrass in the Intermountain area (Hull 1974; Hull and Holmgren 1964; Hull and Pehanec 1947; Hull and Stewart 1948). Recently, the establishment of crested wheatgrass as a "bridge" plant community to replace cheatgrass-dominated lands for future restoration to a more diverse plant community has been proposed as an alternative to seeding a full complement of native species in one treatment. This strategy, also referred to as "assisted succession" (Cox and Anderson 2004), will be discussed later in this document.

It is known that sage-grouse will not thrive in large homogenous stands of a single plant species (Crawford and others 2004). The modification of composition and structure of existing crested wheatgrass seedings to increase plant diversity for sage-grouse should be implemented only after considering social impacts, economic considerations, and land-use objectives. This synthesis paper was not produced to support the replacement of crested wheatgrass with native species. It is intended to provide a review of existing science and knowledge that can be used to increase diversity and structure of crested wheatgrass seedings to benefit sage-grouse, other wildlife species, and livestock. Local land-use plans, science, and public input should be incorporated into the decisionmaking process when selecting crested wheatgrass seedings to implement the strategies described in this paper.

Steps Required to Enhance Structure and Diversity in Crested Wheatgrass Seedings

The competitive characteristics of crested wheatgrass, discussed in the previous section, illustrate the difficulty in designing treatments to increase plant diversity in crested wheatgrass seedings. This grass is extremely resistant to grazing by herbivores. It has a variable response to mechanical or chemical treatments and a large viable seed reserve in the soil that must be considered in any control treatment. Increasing plant diversity in established stands of crested wheatgrass is presented as a three-step process.

- Step 1. Reduce competition of crested wheatgrass to facilitate the establishment and persistence of the desired species.
- Step 2. Introduce the desired plant(s) as seed or seedlings.
- Step 3. Implement appropriate management and monitoring to maintain plant diversity of the seeding.

The discussion that follows focuses on step 1, because the knowledge and literature on seed or plant application (step 2) and managing restored seedings (step 3) are generally available.

Step 1: Reduce Crested Wheatgrass Competition

Grazing by domestic livestock, mechanical or chemical treatments, and fire are potential treatments that can be implemented singly or in combination to reduce crested wheatgrass competition prior to introducing desired plants to a seeding. Periodic droughts also offer opportunities, again singly or in combination with the above treatments, to reduce crested wheatgrass competition.

Livestock Grazing—The design of livestock grazing systems to maintain crested wheatgrass and reduce encroachment of sagebrush into these seedings has been studied for almost as long as crested wheatgrass has been used.

Much of this early research focused on grazing systems and utilization levels to maintain the productivity of seedings and minimize the return of sagebrush in seeded areas (Cook and others 1958; Hull and Klomp 1974) Sagebrush was considered an "invader" in crested wheatgrass seedings because it reduced the productive capability and economic returns from seedings in direct proportion to the ratio of shrub to grass. Rittenhouse and Sneva (1976) determined that each 1 percent increase in Wyoming big sagebrush canopy cover was associated with a 3.3 to 5.2 percent decline in crested wheatgrass production in eastern Oregon. This early research can now be used "in reverse" to develop grazing systems to reduce crested wheatgrass competition in order to increase plant diversity, especially shrubs, in seedings.

Since crested wheatgrass can withstand heavy grazing (Caldwell and others 1981; Cook and others 1958; Hull 1974; Laycock and others 1981), reducing competition using only livestock may be insufficient to permit establishment of desirable seeded species. In general, high levels of utilization by livestock during the growing season reduces the vigor of crested wheatgrass and may lead to mortality of some, but not all, plants (Cook 1973; Wilson and Partel 2003). Early summer grazing may be detrimental to crested wheatgrass due to lower carbohydrate (Trlica and Cook 1972) and nitrogen reserves (de Kroon and Bobbink 1997) of grazed plants. Other studies have demonstrated that heavy use alone (up to 70 percent) did not significantly affect stands of crested wheatgrass (Frischknecht and Harris 1968; Lodge and others 1972; Springfield 1963). Heavy utilization by livestock was also cited by some of these authors as necessary to reduce development of crested wheatgrass "wolf plants." Wolf plants are crested wheatgrass plants that contain a higher proportion of dead stems than consistently grazed plants and, as a result, are not preferentially selected by livestock.

Olson and others (1988a,b) found that grazing to reduce vigor, cause mortality, or reduce establishment of new crested wheatgrass seedlings is most effective if the treatment is done during or immediately after tiller elongation (internode elongation) and results in removal of the apical meristem. These same studies showed that grazing crested wheatgrass before internode elongation had little effect on reducing tiller replacement and could increase tiller density if grazing intensity and timing were not closely monitored. Olson and others (1988b) indicated that short-duration grazing at a conventional stocking rate in eastern Oregon increased tiller density of crested wheatgrass. This study also demonstrated that most crested wheatgrass tillers are produced in the fall, overwinter, and flower the following growing season. If an adjacent crested wheatgrass plant was removed, the tiller production and resource uptake of its neighbor was increased. Thus, attempts to reduce crested wheatgrass by livestock grazing are influenced by growing season conditions, level of utilization in relation to plant phenology, degree of use of neighboring plant, and the dynamics of tiller production.

It is well established that crested wheatgrass is adapted to withstand heavy livestock use with minimal mortality. How does livestock grazing affect the recruitment and establishment of crested wheatgrass? Most crested wheatgrass recruitment occurs between the rows of established plants in a seeding (Salihi and Norton 1987). The success of recruitment in the interspaces is reduced by the impacts of livestock trampling since cattle (*Bos* sp.) generally avoid stepping on plant tussocks (Balph and Malechek 1985). As crested wheatgrass plants age, elevated tussocks develop because of the plant's caespitose growth form, further increasing cattle avoidance of stepping on the mature plants. This results in increased mortality of seedlings that are trampled (Salihi and Norton 1987), and accelerates soil erosion and compaction (Balph and others 1985) in the interspace areas. In another study, Krzic and others (2000) stated that long-term grazing of crested wheatgrass did not result in degradation of soil properties, with one exception: soil compaction was greater in seedings grazed in spring compared to native rangeland. Salihi and Norton (1987) measured less than 1 percent crested wheatgrass seedling survival in grazed plots compared to 12 percent survival in ungrazed plots.

The combination of properly timed livestock use to reduce vigor and survival of mature crested wheatgrass plants along with the trampling of new recruits in the interspace areas should result, over time, in a reduction in both numbers and vigor of mature crested wheatgrass plants and their recruitment potential. The decision to use these intensive grazing treatments must be weighed against the detrimental effects of heavy grazing on soil properties, weed entry and/or expansion, and erosion potential as well as the management objectives for the seeding.

Another benefit of livestock use at the appropriate time and intensity in crested wheatgrass seedings is to facilitate the return of sagebrush. As mentioned earlier, control of "reinvading" sagebrush in crested wheatgrass seedings was the focus of past research on treatments to physically remove the sagebrush or livestock management systems to maintain the crested wheatgrass productivity and minimize the reinvasion of sagebrush. It is well established that sagebrush encroachment in seedings is less under light to moderate spring livestock use, but increases under high crested wheatgrass utilization levels for this same period (subject to climatic, grazing management system, and initial treatment variables) (Frischknecht and Harris 1968; Hull and Klomp 1974; Laycock and Conrad 1981; Robertson and others 1970). For example, crested wheatgrass utilization levels of 80 percent on a Utah seeding resulted in loss in vigor of crested wheatgrass and an increase in sagebrush (Frischknecht and Harris 1968). By comparison, fall grazing by cattle resulted in less sagebrush encroachment in seedings when compared to heavy spring livestock use (Laycock and Conrad 1981). However, grazing by sheep (*Ovis aries*) in fall often resulted in reduced production or mortality of sagebrush (Frischknecht 1978).

Therefore, assuming that there is a sagebrush seed source in or near the target crested wheatgrass seeding, a grazing system that promotes heavy spring livestock use over a period of years could promote an increase of sagebrush in crested wheatgrass seedings. Angell (1997) found that this same grazing management system would also promote the survival of juvenile sagebrush plants due to decreased soil water depletion by crested wheatgrass. He found that only the short duration, double stocking rate treatment in spring resulted in an increase in juvenile sagebrush plants when compared to the continuous grazing and moderate short duration grazing treatments. In a similar study, Owens and Norton (1990) found that juvenile sagebrush survival was greater in a pasture that received high intensity use for repeated short durations (short duration grazing system) during the growing season when compared to a traditional continuous growing season treatment.

Thus, once juvenile sagebrush plants are established in a seeding, continued heavy livestock use will accelerate sagebrush growth and potentially increase additional sagebrush recruitment. This strategy is predicated on concentrated heavy use of crested wheatgrass and may require temporary fencing to concentrate livestock in a smaller portion of a larger seeded pasture. The temporary fence could then be moved to another portion of the seeding to increase sagebrush establishment over a larger area, if desired. Other considerations in applying this strategy to increase sagebrush in crested wheatgrass seedings are the effects of soil compaction, potential for weed entry, increased soil erosion, and effects on residual native grasses and forbs in the heavy use areas. Introduction of sagebrush seed may be required if a seed source is not already present in or immediately adjacent to the treatment area.

Drought and Livestock Grazing—Periodic droughts provide another window of opportunity to reduce crested wheatgrass density, especially when combined with properly timed, heavy levels of livestock use. Tiller regrowth of crested wheatgrass was limited by clipping and drought over a 2-year period (Busso and Richards 1995). They cautioned that repeated late spring grazing under droughts lasting 2 or more years could reduce the persistence of crested wheatgrass in a stand. Conversely, light or moderate grazing (around 40 percent) of crested wheatgrass in a drought was found to enhance production and survival because of a decrease in the leaf area and associated respiration (Mohammad and others 1982). In this same study, no plant recovery occurred when water stress was severe and crested wheatgrass defoliation was 80 percent.

Crested wheatgrass has the potential to recover rapidly after a drought due to the high accumulation of total nonstructural carbohydrate reserves accumulated in the plant organs during times of stress (Busso and others 1990). Thus, any benefits in reduction in competition of crested wheatgrass achieved by livestock grazing during droughts may be lost quickly if treatments to increase diversity are not implemented in a timely manner. Another concern with using drought and livestock to reduce competition of crested wheatgrass is the opportunity for an increase in invasive species during periods between droughts and the average or above average precipitation periods following droughts (Svejcar 2003). Heavy livestock use may also increase the potential for loss of biological soil crusts and residual native plants in the seeding (Anderson and others 1982; Kimball and Schiffman 2003). Even with these concerns, livestock grazing during multi-year droughts may reduce crested wheatgrass competition sufficiently to allow successful reintroduction of desired species.

Herbicide Application—The application of an appropriate herbicide at the proper time can reduce perennial grass density (Nelson and others 1970; Whisenant 1999). A number of different herbicides are effective in reducing vigor or causing mortality of crested wheatgrass. Glyphosate (*N-[phosphonomethyl]glycine*) is a contact herbicide that stunts or kills the entire plant upon application. Application of glyphosate (trade name Roundup™) reduced crested wheatgrass cover from 12 to 4 percent in 1 year and had no effect in the second year of a 2-year study in Utah that looked at the utility of several treatments to reduce competition prior to seeding native species (Cox and Anderson 2004). This difference in effect between years, probably due to timing of application, illustrates the importance of applying contact herbicides at the appropriate phenological stage. In Canada, a spring application of glyphosphate reduced crested wheatgrass by 50 percent, which was adequate control to establish a native warm season grass seeded at a high application rate (Bakker and others 1997).

Wilson and Partel (2003) applied multiple herbicide treatments to maximize the mortality of crested wheatgrass in Canadian grasslands. A total of 13 glyphosphate applications over 6 years significantly reduced cover of crested wheatgrass; however, the surviving plants in the herbicide treatment area produced 42 seedheads per m^2 compared to 12 seedheads per m^2 in the control. Crested wheatgrass seedlings emerging from the seedbank were not significantly different between the herbicide treatment and control (average density of 284.4 seedlings per m^2). Even though crested wheatgrass was not eliminated with the herbicide treatments in this study, Bakker and others (2003) reported that native species diversity and abundance were enhanced on these study sites. They reported that the careful application of glyphosate by wicking or spraying prior to the active growth of warm season native species can suppress crested wheatgrass and promote native species establishment. They recommend considering cultivation prior to seeding and applying multiple control treatments (herbicide and intensive grazing) in hot dry years to further reduce crested wheatgrass competition if herbicide application alone is not adequate.

Another Canadian project evaluated multi-year application of glyphosphate to reduce crested wheatgrass competition before seeding native species (Ambrose and Wilson 2003). Glyphosphate was applied as a spray in the spring of the first year and applied with a wick applicator in the 3 subsequent years. Surprisingly, emergence of crested wheatgrass seedlings from the seedbank was not decreased by 4 years of glyphosphate treatments, due primarily to the tripling in number of seed heads on surviving plants in the herbicide plots compared to the control plots. The impacts of releasing crested wheatgrass from intraspecific competition with glyphosphate and thereby increasing seed production on remaining plants must be considered when selecting treatments to reduce crested wheatgrass competition.

When Romo and others (1994) investigated the effects of a combination of fall burning followed by a spring application of glyphosate on crested wheatgrass mortality in Canada, they found that burning had little effect on crested wheatgrass survival, while glyphosphate applied early in the growing season on the burned crested wheatgrass reduced cover from 78 to 35 percent on one site and from 81 to 55 percent on another site. In another study, Romo and others (1994) applied mowing in the fall to reduce crested wheatgrass vigor followed by an application of glyphosate on individual plants and recorded 100 percent crested wheatgrass mortality. They also observed total elimination of crested wheatgrass with an application of 25 percent glyphosphate in early spring when two to four leaves per tiller were present.

Another consideration in using glyphosphate to reduce crested wheatgrass is the differential effect that this herbicide appears to have on different species of crested wheatgrass. Lym and Kirby (1991) found that 'Fairway' crested wheatgrass was less susceptible to glyphosphate in terms of yield than was 'Nordan' crested wheatgrass. Also glyphosate generally does not interfere with the establishment of seeded species (Bakker and others 1997; Masters and Sheley 2001) since it is bound to the soil once applied and is not available for uptake by plants. On the negative side, since glyphosphate is a contact herbicide, it has no residual effect on crested wheatgrass regrowth and may need to be applied multiple times in the same growing season or over multiple years, depending on climatic conditions and plant phenology and growth patterns.

Paraquat (1,1´-dimethyl-4,4´-bipyridinium ion) is another herbicide that has been used to treat crested wheatgrass. Sneva (1970) found that paraquat applied for 3 consecutive years did not significantly reduce crested wheatgrass yield in the fourth year. In this study, clipping crested wheatgrass to ground level in May of each year was more effective in reducing the percent of apical meristems than was the herbicide application. Atrazine and simazine were evaluated as tools to rejuvenate weed infested seedings in Nevada (Eckert 1979). The reduction of weedy competition in the stand by these herbicides resulted in slightly more crested wheatgrass seed production and minimal mortality on treated compared to control sites. Crested wheatgrass seedling production was significantly greater in the atrazine treated plots compared to the control, indicating that the reduction in weedy competition not only favored seed production but greatly enhanced seedling establishment.

In summary, herbicides can be very effective in controlling crested wheatgrass, especially when combined with other treatments such as burning or mowing. Label restrictions on their use should be closely followed in order to minimize adverse effects. If complete crested wheatgrass mortality is not obtained (usually the case), the seed production on surviving plants increases significantly and provides significant competition with desirable plants introduced on the treated areas. As Whisenant (1999) points out, effective herbicide use requires knowledge of individual site characteristics and knowledge of herbicide effects on the individual species and the environment.

Mechanical Treatments—Mechanical treatments can be used to either physically remove crested wheatgrass biomass (for example, mowing) to reduce plant vigor or cause mortality, or uproot plants and cause direct mortality (for example, plowing). Mechanical removal of live crested wheatgrass foliage will be discussed first, followed by an overview of equipment that can be used to cause direct mortality. Clipping studies to simulate grazing have been previously discussed in the **Livestock Grazing** section of this paper, and the reader is encouraged to review that information as it applies to the effects of mowing described in this section.

Lodge (1960) compared mowing in the fall, burning in spring and fall, and double disking in the fall. He found that mowing had little effect on floristic composition or in reducing basal area of crested wheatgrass in Canada. Double disking was the only treatment that significantly reduced crested wheatgrass basal area (from 6.6 percent on the control to 2.7 percent in the treatment areas); this treatment effect disappeared within 2 years. A clipping study to reduce crested wheatgrass competition was conducted in northern Utah by Cook and others (1958). They hand-clipped crested wheatgrass plants at 1- and 3-inch stubble heights throughout the growing season over a 5-year period. The 1-inch clipping height and more frequent clipping treatments reduced yield, vigor, and seed production of crested wheatgrass more than did the 3-inch clipping height and less frequent clipping treatments. Seed production, as expressed by number of spikes per plant, was not significantly affected by clipping height; however, increasing the frequency of harvesting decreased the number of spikes produced. Finally, this study documented that frequency and season of clipping were the most influential factors affecting viable seed production. At the end of the 5-year study period, control plants produced 1,834 viable seeds per plant, while clipping once in mid-June or early July for 5 years reduced the number of seeds per plant to nearly zero.

Lorenz and Rogler (1962) compared several mechanical techniques to "renovate" stands of crested wheatgrass in North Dakota. Plowing in spring eliminated crested wheatgrass production for 2 years, while a spring scarification treatment (heavy field cultivation that uprooted about one-third of the plants) significantly reduced yields in only the first year following treatment. In subsequent years the scarified treatment produced more herbage than the control plot in one year and similar yields in the remaining years of the study. The authors urged caution with the plowing treatment due to the potential for increased wind erosion.

Bakker and others (1997) rototilled crested wheatgrass plots in May on a sandy site in Canada, reducing cover of crested wheatgrass from around 40 percent on control plots to 20 percent on the treatment plots in August of the same year. Finally, Cox and Anderson (2004) investigated the effectiveness of two tillage treatments and a herbicide treatment in reducing crested wheatgrass competition prior to seeding native species. Tillage treatments were done in February in 2 consecutive years on a crested wheatgrass seeding in an arid (average annual precipitation of 7 inches) portion of Utah. The tilling treatment was done with a cultivator that removed all vegetation and mixed the soil to a 7-inch depth, while the harrowing treatment was done with a field harrow that uprooted some, but not all plants. Tillage was more effective than harrowing in reducing crested wheatgrass cover in this study. The control plots averaged 12 to 4 percent crested wheatgrass cover during the 2-year study compared to 1 to 2 percent cover on tilled and 4 to 7 percent cover on harrowed plots.

Another category of mechanical equipment, the interseeders and transplanters, remove plant competition in narrow bands and seed (interseeder) or plant seedlings (transplanter) in a one-pass operation (Giunta and others 1975; Stevens 1994; Stevens and others 1981; Wiedemann 2005). Scalping to reduce plant competition is generally done with either modified disks or a plow pulled behind a tractor. The seeder or transplanter is mounted immediately behind the disk or plow. Recommended widths for scalping crested wheatgrass prior to seeding shrub seeds are 40 to 60 inches (Van Epps and McKell 1978). This width should be adjusted according to density, vigor, and growth form of existing vegetation, the species to be interseeded, and local site conditions (Stevens 1994). A side benefit of scalping is that the scalp captures and holds additional moisture from snow and rain, which enhances seedling establishment and growth (Stevens 1994). An indepth description of interseeders and transplanters can be found in Chapter 28 of Restoring Western Ranges and Wildlands (Monsen and others 2004b).

Other equipment not specifically addressed in studies cited above that could be used to reduce crested wheatgrass competition includes pipe harrows, anchor chains with welded railroad rails (for example, Ely and Dixie Sager chains), and the disk chain (Monsen and others 2004a). Effectiveness of these types of equipment in providing crested wheatgrass control is expected to be moderate to excellent, although published studies to support their use for crested wheatgrass control are few.

In summary, the use of mechanical equipment to reduce crested wheatgrass competition will vary in effectiveness, dependent upon a wide array of factors. Some cautions on the use of plows or disks include increased chance of soil erosion and weed entry, loss of residual native plants and biological crusts, and treatment costs.

Step 2: Introduce Desired Species

The challenges in controlling crested wheatgrass competition, described in Step 1, must be resolved prior to implementing the seeding or planting treatments outlined in Step 2. The benefits of increasing plant diversity in grass monocultures include improved habitat, greater species richness and community diversity, improved aesthetics, more soil cover (Stevens 1994), and increased diversity of birds, mammals, reptiles, and insects (Reynolds 1980).

Most of the treatments implemented in the past to increase diversity in crested wheatgrass stands have involved interseeding or transplanting single species or a few species such as big sagebrush, rubber rabbitbrush (*Chrysothamnus nauseosus*), fourwing saltbush (*Atriplex canescens*), antelope bitterbrush, Lewis flax (*Linum perenne*), Palmer penstemon (*Penstemon palmeri*), western yarrow (*Achillea millefolium*), and globemallow (*Sphaeralcea* spp.) (Monsen and Shaw 1983; Pendery and Provenza 1987; Stevens 1994). However, single rows of shrubs or forbs in monocultures of crested wheatgrass may not meet all of the resource (for example, sage-grouse habitat) or management objectives for a particular area.

Step 2 involves the selection of adapted species to plant and appropriate equipment to implement the planting. It is essential to select the species and seed mixtures that meet resource objectives and are adapted to the ecological site(s) that will be seeded. Nonadapted seeds may respond differently to germination cues and germination may occur at an inappropriate time, resulting in seeds that fail to germinate or persist (Meyer 1994). Additional considerations for seed mixture development include the potential for interspecific interactions among the species in the seed mixture during the establishment phase, the ability of plants to coexist, and the ability of the species to regenerate itself on the site (Archer and Pyke 1991; Pyke 1994; Pyke and Archer 1991).

If the objective of the crested wheatgrass treatment(s) is to restore ecosystem functioning and biological diversity to a site, this will often require the use of native species (Lesica and Allendorf 1999). Native species introduced into a crested wheatgrass seeding may facilitate recruitment of additional native species. For instance, Frischknecht and Bleak (1957) found that seeded stands of bluebunch wheatgrass were more likely to permit sagebrush seedling recruitment than seeded stands of crested wheatgrass. Introduced species may also increase the diversity of a crested wheatgrass seeding, improving it as habitat for sage-grouse. Dryland alfalfa (*Medicago sativa*) and small burnet (*Sanguisorba minor*) are introduced forbs that are preferred by sage-grouse that can be successfully reintroduced into crested wheatgrass seedings. It is important to select site-adapted species (native or introduced) that are competitive in the posttreatment environment and that will be maintained over the long term with livestock management systems.

The selection of a seed mixture should not be done without consideration of how seed will be distributed during the planting process. Rangeland drills vary considerably in their ability to seed native species. If suitable equipment is not available to properly seed a species in the proposed seed mix, the mix should be changed or the proper equipment secured. Another factor to consider is that some site preparation treatments, such as plowing or disking, may create an unfavorable planting seedbed that requires additional treatments. Harrowing or cultipacking after these surface disturbing treatments may be required to mitigate these unsatisfactory seedbed surfaces (Whisenant 1999).

Direct seeding by drilling or aerially broadcasting seed is relatively inexpensive, widely applicable, and under appropriate seedbed conditions, provides good plant establishment (Whisenant 1999). Applying seed with a rangeland drill is considered the best method for establishing species with large, hard seeds because the seed is placed in contact with the soil and at an appropriate depth (Hull 1948; Pyke 1994). However, seeding many native species with the standard rangeland drill is problematic given the lack of control of seeding depth, variable seed coverage with soil, and absence of a mechanism to improve soil to seed contact. Surface obstructions such as rocks, steep slopes, and soddy vegetation also limit the effectiveness of rangeland drills in establishing any seed mixture, especially native forbs and grasses. One unknown in the use of rangeland drills to seed diverse seed mixtures into crested wheatgrass seedings is the effectiveness of these drills in cutting through the dead plant crowns and the shallow root mass of the seeding. If this is a problem, the deep furrow rangeland drill (Hull and Stewart 1948), which has a double furrow opener, may be more effective in soddy conditions than the rangeland drill, which has a single furrow opener. The single disk or double disk opener on the rangeland drill does create a furrow that can capture and store water for seedlings. However, the soil disturbance created by this drill also opens the plant community for the entry of other invasive species.

Another option for ground application seeding into treated crested wheatgrass stands is the use of a minimum till drill that creates less soil disturbance than the rangeland drill. The Truax and Amazon drills are minimum till drills that can place seed at different depths, and their press wheels improve soil to seed contact. An overview of rangeland drills, manufacturer's specifications, and contact information is the Revegetation Equipment Catalog available online at http://reveg-catalog.tamu.edu (Wiedemann 2005). This catalog also contains similar information on most of the equipment discussed in this paper. The reader is encouraged to utilize this Web site for all treatments requiring the use of equipment.

Aerial broadcasting is often easier and less expensive than ground application methods because large areas can be seeded quickly and topography or slopes are generally not a limiting factor (Monsen 2000). Aerially broadcasting seed followed by cultipacking, harrowing, or dragging a chain over the surface, where slope or surface rock is not limiting, places the seed in contact with the soil; however, seeding depth is not uniform (Pyke 1994; Stevens 2004). Livestock trampling has been suggested as another alternative for covering seed that has been aerially applied. Eckert and others (1986) found that heavy livestock trampling appeared to favor the emergence of sagebrush and weedy annual forbs, but was detrimental to the emergence of perennial grasses and forbs. Aerial seeding native species mixes into treated crested wheatgrass stands without some sort of incorporation into the soil is not advised. Given the high cost of seed and the different seedbed requirements of native species (seeding rates are generally doubled on aerial seedings), seeding with rangeland drills is recommended over aerial seeding with or without seed coverage.

Alternatives to ground or aerial application of seed include transplanting individual plants from existing populations ("wildings") or planting container stock or bare-root seedlings grown from seed. In arid and semiarid environments, transplanting young plants may be a more reliable, albeit a considerably more expensive method for establishing native species in crested wheatgrass seedings. Transplanting young plants bypasses the high-risk germination and seedling stage. In addition, transplanting may enhance the success of species that do not establish rapidly from seed and provide larger plants that are more capable of coping with competition and herbivory (Archer and Pyke 1991; Van Epps and McKell 1980; Whisenant 1999). If small islands of native species are desired in a crested wheatgrass seeding or if greater native plant diversity is desired in an existing native species seeding, transplanting wildings, bare-root, or containerized stock of desired forbs or shrubs may be a good option. However, costs of treating larger acreages this way will generally be prohibitive.

The use of livestock to disseminate seed of desired species via dung (Auman and others 1998; Doucette and others 2001; Ocumpaugh and others 1996; Welch 1985) into crested wheatgrass seedings is another option since livestock preferentially graze these areas. Seeds ingested by cattle are deposited in a moist, nutrient-rich medium that may facilitate germination and establishment of ingested seeds and may result in patches of desirable species (Archer and Pyke 1991). Fecal-seeding offers a nonintrusive, relatively low cost method of seeding small areas (Archer and Pyke 1991; Shinderman and Call 2001). Seeding response can be slow and sporadic, and there is the potential for the introduction and spread of exotic species by livestock (Auman and others 1998; De Clerck-Floate 1997; Lyon and others 1992; Pleasant and Schlather 1994). A study by Auman and others (1998) found that cattle dung provided favorable conditions for the germination of crested wheatgrass as well as cheatgrass. Also, livestock grazing would need to be closely monitored to ensure that livestock did not overutilize and eliminate the very plants they were dispersing (Archer and Pyke 1991).

Step 3: Posttreatment Management

The long-term success of any project implemented to increase plant diversity in crested wheatgrass seedings is dependent on applying appropriate management during the establishment and postestablishment period. Documentation of implementation practices and the effectiveness of treatments must be conducted via a well designed monitoring program in order to adjust management now and design more effective projects in the future (for example, implement an adaptive management program). An adaptive management program is not possible without good implementation information combined with sound effectiveness monitoring.

Livestock Management—It is essential that livestock grazing and rest intervals are matched with the phenology and life history attributes of desired plant species (Archer and Pyke 1991; Holechek 1983). Grazing should be restricted until plants are adequately established and sexually reproducing (Pyke 1994). Many plants require at least 2 years, and as many as 5 years, to become established with adequate root systems to endure grazing (Pyke 1994; Stevens 1994; Vallentine 1989; Vallentine and others 1963; West and Hassan 1985). Areas seeded to shrubs must be protected from grazing during the establishment period (Ganskopp and others 1999; Richardson and others 1984). Plant seedlings are particularly sensitive to herbivory because they have low nutrient and energy reserves and shallow, low-density root systems relative to adult plants (Archer and Pyke 1991; Holechek 1983).

Once the plant establishment period (period of time when livestock were excluded from the project area) has passed, an appropriate livestock management plan must be followed to maintain the diversity restored in the crested wheatgrass seeding (Archer and Pyke 1991). If plant diversity is increased in a crested wheatgrass seeding to benefit sage-grouse, additional livestock or recreation management changes may be necessary to maintain structure, composition, and forage quality to meet seasonal habitat requirements. Impacts of livestock grazing can be positive, negative, or neutral to sage-grouse, depending on the timing and intensity of livestock grazing and which seasonal habitat is being considered (Crawford and others 2004). Heavy livestock grazing can reduce grass competition and increase sagebrush density (Crawford and others 2004; Vallentine 1989) or it can decrease big sagebrush seedling survival under certain management systems (Owens and Norton 1990). In general, the season and duration of livestock use and the stocking rate should be managed to promote optimum growth of forbs, grasses, and sagebrush to maximize habitat values for sage-grouse (Beck and Mitchell 2000).

Monitoring—Monitoring involves the orderly collection of data, analysis, and evaluation of data. Combined with experience, monitoring is a powerful tool to improve the effectiveness of restoration efforts now and into the future. Implementation monitoring includes summarizing how, what, where, and when treatments were actually implemented. The timing of treatments, conditions during application of treatments (for example, was the soil dry or frozen when seeding occurred), and posttreatment events (for example, Mormon cricket density was high the first year following seeding) are all important factors in evaluating treatment effectiveness. The origin and percent of pure live seed of each species in the seed mixture should also be documented in the project file to improve the accuracy of seeding establishment interpretations.

Effectiveness monitoring measures the success of the treatments that were implemented relative to the project objectives. Implementation monitoring provides the context to evaluate the effectiveness of the treatments. It is important that project objectives be developed before selecting monitoring protocols. The sage-grouse guidelines developed by Connelly and others (2000) provide a good starting point to develop sage-grouse habitat objectives in crested wheatgrass seedings proposed for treatments to increase their diversity.

Monitoring information, if collected appropriately, provides the framework to implement an adaptive management program to improve restoration practices in the future. Adaptive management acknowledges uncertainty and imperfect knowledge in implementing projects (Walters 1986), and encourages research and management to be conducted simultaneously (Smallwood and others 1999; Walters and Holling 1990). An adaptive management approach would be especially helpful in identifying treatments that are effective in reducing crested wheatgrass competition (Step 1). To maximize the utility of this approach, different treatments would be implemented and evaluated on the same project, promoting a better understanding of treatment effectiveness. At a minimum, adequate monitoring data should be collected to determine if short- and long-term management objectives are met when restoring diversity of crested wheatgrass seedings.

Summary

This review identifies some of the actions that can be taken to increase plant diversity in crested wheatgrass seedings for sage-grouse and other uses. The importance of proper planning and posttreatment management has been stressed as an essential component of a three-step process to convert parts of existing crested wheatgrass seedings into more diverse plant communities. This three-step process and the treatments associated with it could be used as part of a more ambitious strategy to first convert cheatgrass monocultures (Allen 1995; Pellant 1990; Tausch and others 1995) into perennial grasslands followed by the steps described above to increase plant diversity in these crested wheatgrass grasslands, for example, assisted succession as described by Cox and Anderson (2004). This strategy provides a bridge between the difficult conversion of exotic annual grasslands into native plant communities.

It is important to remember that crested wheatgrass seedings have been an important management tool used to increase livestock production, reduce weed problems and wildfires, and mitigate soil erosion potential following disturbances since they were first established in the late 1930s. Regardless of whether our objectives now are to increase plant diversity in selected crested wheatgrass seedings or as part of a larger strategy to reduce cheatgrass domination in the Intermountain region, the application of good science and professional experience tempered with results from monitoring studies should guide our actions. Sage-grouse, other wildlife species, and all resource uses will benefit from an objective-based approach (both at the site and landscape levels) to restoring plant diversity to selected crested wheatgrass seedings. As always, social, economic, and political values will provide the context for these important restoration decisions.

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Rose pusseytoes

Restoring Wyoming Big Sagebrush

Cindy R. Lysne

Abstract—The widespread occurrence of big sagebrush can be attributed to many adaptive features. Big sagebrush plays an essential role in its communities by providing wildlife habitat, modifying local environmental conditions, and facilitating the reestablishment of native herbs. Currently, however, many sagebrush steppe communities are highly fragmented. As a result, restoring big sagebrush is considered a priority in the conservation and rehabilitation of sagebrush steppe ecosystems. Wyoming big sagebrush can often be difficult to establish, because many environmental factors act to restrict its emergence and persistence. On fire rehabilitation projects in Idaho, Wyoming big sagebrush seed is typically aerially broadcast over the soil surface. This method has had some success; however, several alternative seeding treatments, such as cultipacking, have resulted in the establishment and persistence of Wyoming big sagebrush. In addition, transplanting bareroot and containerized stock may be useful for restoring shrub stands in critical areas.

Keywords: *Artemisia tridentata*, revegetation, rehabilitation, seeding, shrub-steppe

In the Western United States, big sagebrush (*A. tridentata* Nutt.) steppe communities dominate over 60 million ha (Wambolt and Hoffman 2001) and provide essential habitat and forage for many species (West 2000). Fragmentation of sagebrush steppe communities has occurred through excessive livestock grazing, conversion to agricultural cropland, invasion of exotic plants, and increasing frequency of large fires (Anderson and Inouye 2001; Knick 1999; Knick and Rotenberry 1997; Noss and others 1995). More than 350 species of plants and animals associated with sagebrush ecosystems have been identified as species of conservation concern due to declining habitats or populations (Wisdom and others 2003).

Big sagebrush is important because of its wide distribution and the extent of disturbance within its range. It provides both food and cover for sage-grouse (*Centrocercus urophasianus* Bonaparte) year round (Connelly and others 2004). This paper presents a review of the literature on big sagebrush taxonomy and characteristics, germination requirements, relevance in rehabilitation projects, and methods for improving its establishment in seedings and transplantings. It will focus primarily on Wyoming big sagebrush (*A. t.* Nutt. ssp. *wyomingensis* Beetle and Young).

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Big Sagebrush Taxonomy and Characteristics

There are five subspecies of big sagebrush. These include basin big sagebrush (*A. t.* Nutt. ssp. *tridentata*), Wyoming big sagebrush, mountain big sagebrush (*A. t.* Nutt. ssp. *vaseyana* [Rydb.] Beetle), xeric big sagebrush (*A. t.* Nutt. ssp. *xericensis* Winward ex R. Rosentreter & R. Kelsey), and subalpine big sagebrush (*A. t.* Nutt. ssp. *spiciformis* [Osterhout] Kartesz and Gandhi) (Wambolt and Frisina 2002; West and Young 2000). The distribution of the subspecies is regulated by seasonal precipitation patterns, elevation, and soil conditions (McArthur 2000; McArthur and others 1979, 1995; Monsen and Shaw 2000).

The dominance and ubiquitous occurrence of big sagebrush can be attributed to many factors. One factor is the production of seasonally dimorphic leaves. Ephemeral leaves, larger and often irregularly lobed, develop in spring and are shed in summer following moisture stress (West and Young 2000). Persistent leaves are typically three-lobed, smaller, develop in late spring, and remain on the shrubs through winter (West and Young 2000). A second major factor contributing to the widespread occurrence of big sagebrush is an efficient two-component root system (West and Young 2000). Its fibrous root system captures water and nutrients near the soil surface, permitting plants to take advantage of summer precipitation (West and Young 2000). The taproot, in turn, allows for utilization of water and nutrients deep within the soil profile and below the principal rooting zone of associated herbaceous species (West and Young 2000).

Several additional adaptive features influence the distribution and persistence of big sagebrush subspecies. These include, but are not limited to, variable growth forms, response to fire, the production of allelopathic substances in roots and leaves, the ability to conduct photosynthesis at low temperatures, temperature requirements for seed germination, seed dispersal strategies, seed size, and structure and timing of seed maturation (Blaisdell and others 1982; Kelsey 1986; Meyer and Monsen 1992; Peterson 1995).

Big sagebrush plants are capable of producing seed in their second year and will continue to produce some seed annually, except during years of severe moisture stress (Meyer and Monsen 1992). Plants flower in fall, following the summer drought period, and the fruits (achenes) mature from midfall to early winter (Meyer 2003). Achenes are small (about 1 by 1.5 mm) and shiny with a deciduous pappus (Meyer 2003). They are dispersed by gravity and wind, but do not possess any special adaptations for wind dispersal (Meyer 1994). Seeds may be blown by wind across crusty snow surfaces and dispersed by animals and water (Tisdale and Hironaka 1981; Young and Evans 1989a). Maximum dispersal distance of seeds can be up to 30 m; however, most seeds (85 to 90 percent) fall within 1 m of the shrub canopy (Meyer 1994; Young and Evans 1989a).

Big sagebrush seeds are surface or near-surface emerging and are sensitive to microsite conditions (Meyer 1994). Germination occurs in late winter to early spring, soon after snowmelt, in areas where snow accumulates (Meyer and Monsen 1992; West and Young 2000). A semi-gelatinous pericarp and hypocotyl hairs aid in the adhesion of the achene to the soil surface and permit the radicle to penetrate the soil (Young and Martens 1991). The achene's small size reduces the surface area for moisture loss (Young and Martens 1991). Achenes typically exhibit high seed viability and germination capacity at maturity (Meyer 2003).

Wyoming Big Sagebrush

Wyoming big sagebrush is the most xeric subspecies of big sagebrush. It generally occurs on shallow soil in areas receiving 200 to 300 mm of annual precipitation (Cronquist 1994; Monsen and Shaw 2000). Wyoming big sagebrush plants exhibit a ragged, irregular growth form, and most plants grow to less than 1 meter in height. The main stem is often branched into two or three twisted portions at or near ground level (Winward and Tisdale 1977). Persistent leaves are narrowly cuneate to cuneate with the margins curved outward, and exhibit a strong, pungent odor when crushed (McArthur and others 1979). The plants flower from late July to September, and seed maturation occurs in October and November (Monsen and Shaw 2000).

Germination and Establishment Ecology

Many environmental factors act to reduce sagebrush establishment and persistence. Seed germination is substantially limited by water stress, and a principal cause of seed mortality is early or prolonged drought (Meyer 1994). The successful establishment of large cohorts of big sagebrush shrubs can result from recruitment pulses that are associated with rare events of highly favorable precipitation (Watson and others 1997; West and others 1979; Williams and Hobbs 1989). High seed densities and synchronous germination can result in intense competition between big sagebrush seedlings. Intraspecific competition or self-thinning probably accounts for much of the initial mortality (Meyer 1994). Competition between sagebrush plants within a stand may also affect flowering and seed set, particularly in dry years (Meyer 1994).

Competition with herbaceous species may also impact the success of sagebrush seedings. However, reports on sagebrush seedling competitiveness with seeded wheatgrasses are contradictory. During the time period when sagebrush was being controlled on rangelands, managers often remarked on the ability of sagebrush to reestablish in perennial grass seedings (Meyer 2003). Conversely, researchers have demonstrated that competition with introduced and native grasses seeded before or with big sagebrush can reduce Wyoming big sagebrush establishment (Blaisdell 1949; Fortier 2000; Schuman and others 1998). Similarly, sagebrush seedings in areas with exotic annual grass competition have had little success (Meyer 2003). Competitive effects are probably related to the inability of sagebrush seedlings to compete for soil moisture during establishment (Cook and Lewis 1963; Sturges 1977). Blaisdell (1949) found higher grass yields on plots that were seeded with grass prior to or 1 year after sagebrush, and that prior grass establishment often prevented the establishment of sagebrush seedlings. However, when grasses were seeded 2 or 3 years following sagebrush seeding, grass yields were reduced and grass competition did not have an effect on sagebrush (Blaisdell 1949).

Sagebrush seedlings have high first-year survival rates, even through summer drought periods, on mine sites where there is little competition (Meyer 1994). Schuman and others (1998) found that grass competition reduced sagebrush seedling densities in a mined-land reclamation study using direct-placed topsoil. They concluded that successful establishment of big sagebrush may require seeding big sagebrush without grasses or with very low grass seeding rates (Schuman and others 2000).

Commercially available sagebrush seed is often not from locally or regionally adapted seed sources. Nonadapted seeds may respond differently to normal germination cues and germination may occur at an inappropriate time, resulting in seeds that fail to germinate or persist (Monsen and Meyer 1990). Using seedlots with the source or geographic origin of the seed verified (Source Identified) and matched to the site may be a key factor for achieving successful shrub establishment (McArthur and others 1995; Meyer and Monsen 1992). Commercially available seed often contains a mixture of sagebrush subspecies (Dalzell 2004). Currently, the Association of Seed Analysts does not provide guidelines or testing methods for differentiating sagebrush subspecies in purchased seed (AOSA 2003). Applying the big sagebrush subspecies matched to the restoration site is essential because big sagebrush subspecies exhibit differences in seedling establishment traits (McArthur and others 1995), growth rates (Welch and McArthur 1984), habitats (Winward and Tisdale 1977), and moisture (Barker and McKell 1983; Kolb and Sperry 1999), temperature (Harniss and McDonough 1976), and germination requirements (Meyer 1994).

Seed bank studies of big sagebrush indicate seed banks are transient, with very little seed carryover from one year to the next (Meyer and Monsen 1992). Most of the big sagebrush seeds produced in autumn are absent from the soil seed bank by late spring of the following year (Young and Evans 1989a). Wyoming big sagebrush seeds are, in general, short lived and do not survive fires (Young and Evans 1989a). Young and Evans (1989a) found that no mountain big sagebrush or basin big sagebrush seedlings emerged from germination tests of 1,000 soil surface samples taken from a burned area. In contrast, however, some Wyoming big sagebrush seeds applied with mulch cover on mined lands in Wyoming remained viable in the seed bank for up to 4 years (Schuman and others 1998).

Sagebrush seeds are highly viable with little or no dormancy at dispersal, but may have strong light requirements for germination (Meyer 2003; Young and Evans 1989b). The light requirement is removed through stratification (moist chilling), and most seeds are germinable by late winter or early spring (Meyer 2003).

Use of Big Sagebrush in Rehabilitation Projects

Reestablishing big sagebrush is considered a priority in the conservation and rehabilitation of sagebrush steppe ecosystems (USDI BLM 2002a). In addition to providing habitat for sage-grouse and other sagebrush obligate species, big sagebrush also plays an essential role in these communities by directly modifying local environmental conditions, thus providing a more favorable environment for seed germination and seedling survival (Schlesinger and Pilmanis 1998). Shrubs also help to retain soil nitrogen, increase organic matter, and create favorable environments for microorganisms, resulting in fertile islands or patches that develop over time (Cross and Schlesinger 1999; West 2000). By trapping blowing snow and moderating temperatures, big sagebrush facilitates the establishment of native herbs, and their canopy protects native herbs from overutilization (West 2000). Wyoming big sagebrush also develops mycorrhizal fungi associations, which aid in nutrient extraction and cycling (West 2000).

The establishment of big sagebrush is often difficult due to poor seed quality (Harniss and McDonough 1976; Young and Evans 1989a), low seedling vigor, exposure to unfavorable seedbed conditions (McDonough and Harniss 1974; Meyer and Monsen 1992), competition with herbaceous species (Blaisdell 1949; Sturges 1977), and inadequate moisture (Cook and Lewis 1963; Sturges 1977). Improved seed cleaning, handling, and purchasing requirements have made higher quality seed easier to obtain (Meyer and Monsen 1992; Olson and others 2000). Also, seedbed conditions can be manipulated to reduce competition and facilitate seed germination (McArthur and others 1995; Welch and others 1992). Ultimately, however, environmental factors still play a central role in determining the success of big sagebrush restoration projects.

Seeding treatments can have a strong influence on the emergence and survival of big sagebrush seedlings. On Bureau of Land Management (BLM) fire-rehabilitation projects in Idaho, Wyoming big sagebrush seed is typically aerially broadcast over the soil surface by helicopter (USDI 2002b). Aerial broadcasting is often desirable over other methods, because large areas can be seeded quickly and the seed can be placed on the soil surface (Monsen 2000). This seeding method has had some success (Monsen 2000); however, results from a recent study in southern Idaho indicate that aerially seeding Wyoming big sagebrush had limited effect on shrub establishment (Dalzell 2004). In this study, seeding did not increase the density or cover of Wyoming big sagebrush on seeded plots compared to adjacent unseeded plots (Dalzell 2004). In fact, shrubs failed to establish on 23 of the 35 (66 percent) study sites sampled (Dalzell 2004).

Another key factor in the establishment and persistence of sagebrush seedlings is the timing and amount of winter snowfall. The recommended time for planting big sagebrush is in fall, just before the first winter snowfall. This is the time when big sagebrush would naturally be dispersing seed onsite (Meyer 1994). Snow cover can facilitate the establishment of big sagebrush—particularly in areas with reliable, long-term snow cover—by compacting or firming the soil surface and assisting in keeping the seed in contact with the soil. However on drier and warmer sites, winter snowfall may be inadequate to facilitate these physical processes to ensure successful big sagebrush emergence and establishment (Meyer 2003). Wyoming big sagebrush sites are typically windswept and relatively dry in both autumn and winter (Meyer and Monsen 1992). These environmental conditions are not favorable for sagebrush emergence or establishment.

Increasing Shrub Establishment

There are several alternatives to aerial seeding that have been shown to increase big sagebrush establishment and persistence. For example, seeding equipment that compacts the soil surface, such as cultipacking, chaining, and imprinting, can increase big sagebrush seedling establishment. Monsen and Meyer (1990) obtained significantly greater initial seedling emergence, compared to broadcasting, by seeding with the Oyer compact row seeder. This device compacts the soil and then presses the seed into the surface. Intermediate seedling emergence results were achieved by using the Brillion cultipacker seeder (Monsen and Meyer 1990). Using this device, the seed is broadcast over the surface and pressed into the soil (Pyke 1994). The cultipacker is a circular cylinder or set of wheels that are rolled over the soil surface to place the seed in contact with the soil near the soil surface (Pyke 1994).

The U.S. Department of the Interior, Bureau of Land Management's Lower Snake River District in Idaho achieved successful sagebrush establishment using a seeder that incorporates a fertilizer spreader, anchor chain or tire drags, and a vine-roller cultipacker (Boltz 1994). This sagebrush seeder covered the seed and firmed the soil surface on silt loams, but it was less effective on gravelly and stony areas (Boltz 1994).

Another option for establishing big sagebrush is to transplant bareroot or containerized stock. Stock that is 12 to 20 cm tall is transplanted in early spring (McArthur and others 1995). First year survival rates for transplanted stock are often 80 percent or higher (Welch and others 1992). Seedlings are typically transplanted only in small, critical areas due to the cost of using planting stock. Transplant stock can be grown from small amounts of seed from specific areas similar to the planting sites. Transplant stock is available locally or regionally from private contracted nurseries and from USDA Forest Service nurseries.

A similar method, the "mother plant" technique, combines transplanting and natural seed dispersal. The mother plants are planted as bareroot or containerized stock on key locations throughout the rehabilitation site. Within 3 to 5 years, established mother plants mature, disperse seed, and provide an established seed source for unseeded areas (Welch and others 1992). However, successful sagebrush establishment and subsequent dispersal also depends on the species composition in the unseeded areas.

Big sagebrush is considered an obligate vesicular-arbuscular mycorrhizal (VAM) plant (Wicklow-Howard

1994). Arbuscular mycorrhizae can improve the ability of plants to extract nutrients and water from the soil, thereby improving the host species' survival and growth on severely disturbed lands (Wicklow-Howard 1994). In a greenhouse study, Stahl and others (1998) found that sagebrush seedlings grown in topsoil with mycorrhizal inoculum exhibited significantly greater tolerance to drought stress than non-mycorrhizal seedlings. Arid land disturbances such as fire, mining, overgrazing, off-highway vehicle use, and cultivation significantly reduce the mycorrhizal inoculum potential (MIP) of the soil (Wicklow-Howard 1989). Efforts to add commercially available VAM fungal inoculum to the soil or to use VAM-inoculated plants on disturbed areas have met with limited success (Wicklow-Howard 1994).

To increase big sagebrush establishment, it is imperative that alternative seeding methods are considered in lieu of aerially seeding Wyoming big sagebrush, particularly if the seed is not adequately covered. Although transplanting bareroot and containerized stock is regarded as costly, this expense may be acceptable when considering the current failure to establish sagebrush using aerial seeding (Dalzell 2004). Because areas that have been depleted of sagebrush for several years may lack the proper mycorrhizal fungi in the soil, containerized stock should be inoculated with compatible fungi. Bareroot and containerized stock could be transplanted in small, critical areas and in areas currently dominated by introduced seeded grasses. Planting big sagebrush can also facilitate the restoration of highly palatable selections of sagebrush, such as Gordon Creek Wyoming big sagebrush, or local germplasms that are best suited for the site conditions (Welch and others 1992).

Another alternative method is to transplant big sagebrush shrubs on areas where fertile islands existed prior to burning, focusing on areas where native vegetation and shrub skeletons remain. The transplanted shrubs will assist in the formation of "islands of fertility" (Cross and Schlesinger 1999). These shrub islands will serve as habitat islands for animal species by providing shrub cover to reduce the risk of predation (Longland and Price 1991), providing temporary refuges to facilitate animal dispersal and the maintenance of a metapopulation, a group of spatially separated subpopulations that are interlinked and maintained by occasional dispersal (Longland and Bateman 2002). Areas that are positioned adjacent to the shrub islands and have remaining native vegetation could be left unseeded; thus reducing the mechanical disturbance of the soil surface by some seeding equipment and reducing the likelihood of exotic plant invasion, biological soil crust destruction, and subsequent wind erosion. In addition, the islands would serve as a seed source for the replenishment of native species with unavailable or limited commercial seed supplies. The shrub islands would not only provide a seed source for animals to harvest, consume, and disperse, but also provide a refuge for seed dispersers (Longland and Bateman 2002). In addition, these islands are sites with high vesicular arbuscular mycorrhizal activity. Ultimately, the development of fertile shrub islands would serve as inoculum focal points from which shrubs, VAM, and other species could spread (Allen 1987).

As previously mentioned, establishment of big sagebrush seedlings is impacted by competition. Seeding introduced grasses such as crested wheatgrass (*Agropyron cristatum*) and intermediate wheatgrass (*Elymus hispidus*) with big sagebrush has prevented shrub seedlings from establishing (Richardson and others 1986). Direct competition for available soil moisture and nutrients exists between seeded grasses and big sagebrush because of similar root distributions and growth periods (Cook and Lewis 1963; Sturges 1977). In contrast, studies have shown that stands of native bunchgrasses permitted big sagebrush recruitment (Booth and others 2003; Frischknecht and Bleak 1957). Frischknecht and Bleak (1957) reported that seeded stands of bluebunch wheatgrass (*Pseudoroegneria spicata*) were more likely to permit sagebrush seedling recruitment than seeded stands of crested wheatgrass. In addition, Booth and others (2003) found that the native perennial bunchgrass squirreltail (*Elymus elymoides*) permitted big sagebrush recruitment and also suppressed cheatgrass.

Seeds of several important native bunchgrasses are available. However, sources of other bunchgrasses adapted to the Interior Western United States, such as Great Basin wildrye (*Leymus cinereus*), bluebunch wheatgrass, Sandburg bluegrass (*Poa sandbergii*), bottlebrush squirreltail, and the needlegrasses (*Hesperostipa*) are just beginning to be marketed. Additional research is needed to develop appropriate seedbed preparation methods, planting techniques, and equipment for the establishment of individual native species and populations. Also, further research could focus on seeding big sagebrush in mixed seedings of native species and investigating the ability of these seedings to permit sagebrush establishment and compete with invasive species.

Successful rehabilitation following wildland fire is essential to mitigate the effects exotic, invasive plants have on ecosystems, decrease the frequency of large fires, provide suitable wildlife habitat, and halt the conversion of diverse sagebrush steppe communities to communities dominated by exotic, invasive plants. To increase big sagebrush establishment, it is imperative that other seeding methods be considered, utilized, monitored, and evaluated instead of the commonly used aerial seeding technique. If current sagebrush restoration efforts do not result in more consistent establishment and persistence of this important shrub, large areas of sagebrush-steppe may be lost, and rehabilitation may no longer be a viable option (West 2000).

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Engelmann Aster

Reseeding Big Sagebrush: Techniques and Issues

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Abstract—Reestablishing big sagebrush on rangelands now dominated by native perennial grasses, introduced perennial grasses, or exotic annual grasses, particularly cheatgrass (*Bromus tectorum*), serves to stabilize soil, improve moisture availability and nutrient recyling, increase biological diversity, and foster community stability and resiliency. A first priority in reseeding is identifying the subspecies of big sagebrush native to the site and procuring adapted, high-quality seed of that subspecies from a similar site. Seed should be planted on firm seedbeds and pressed into the soil to provide good seed-to-soil contact. Competition from invasive species and other seeded species must be minimized by site preparation practices and use of appropriate seeding strategies and equipment. Precipitation is often a major factor in determining seeding success on drier sites. Postseeding monitoring and careful management are necessary to maintain stands and provide feedback for improving future seeding efforts. Additional research and technological developments are required to better estimate and maintain big sagebrush seed quality, provide required seedbed conditions, and reestablish mixed seedings of big sagebrush and associated natives.

The sagebrush (*Artemisia* spp.) biome encompasses approximately 63 million ha of the Western United States, but little of this area has remained unaltered since Euro-American settlement. Vast tracts have been lost to agriculture, urbanization, and other human activities. Of the remaining area, it has been estimated that 50 to 60 percent has been converted to nonnative annual grasslands or contains exotic annual grasses in the understory (West 2000). Even though more than 70 percent of the sagebrush-steppe is publicly owned, less than 3 percent is protected in National Parks or other Federal reserves (Knick and others 2003). The increasingly rapid and widespread degradation, fragmentation, and, in some areas, near total loss of sagebrush has resulted in its being rated one of the most imperiled ecosystems in North America (Noss and Peters 1995). Some have advocated that a regional objective of no net loss of sagebrush be adopted to prevent further declines in biodiversity (Paige and Ritter 1999; West 2000).

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More than 20 sagebrush species and subspecies occur within the sagebrush biome (Goodrich, this proceedings; Rosentreter, this proceedings). It is spatially complex, with variable soils, topography, parent materials, climates, landscape patterns, and disturbance histories (Miller and Eddleman 2001). Sagebrush populations display a strong alliance to certain habitats, with morphological specializations and adaptations evolving along environmental gradients (Schultz 1986). Prior to Euro-American settlement, fire regimes were equally complex across this region and contributed significantly to landscape heterogeneity. With the shift in fire regimes that has occurred over the past 100 years, largely due to the spread of nonnative plant introductions into voids created by postsettlement livestock grazing, this once complex landscape has become increasingly homogeneous. All of these factors contribute to the enormous difficulty that land managers experience in attempts to restore native plant communities where natural recruitment is often limited by a lack of propagules, drought, a competitive exotic understory, disruption of hydrologic functioning, and changes in soil structure and biota as a result of past disturbances.

Early seeding success with introduced grasses contributed to their widespread use for soil stabilization and to type conversion of sagebrush landscapes for increased forage production; the latter is an objective that dominated our use of this biome for much of the twentieth century (Holechek and others 1998). From the 1930s into the 1970s, an estimated 2 to 6 million ha of sagebrush habitat was burned, sprayed, or treated mechanically to reduce sagebrush (Braun 1998; Vale 1974). Due to health concerns, use of 2,4-D and 2,4,5-T was curtailed in the 1980s, but other treatments continued through that decade. The total acreage impacted is unknown, but it has been estimated to exceed 20 to 25 percent of the total remaining sagebrush-dominated landscape (Braun 1998).

Concern for big game habitat loss increased as these large treatments continued. Monocultures of any one species do not constitute healthy or desirable rangelands (Stevens and others 1981), and generalist animals such as grasshoppers (*Orthoptera*), deer mice (*Peromyscus* spp.), horned larks (*Eremophila alpestris*), and introduced chukars (*Alectoris chukar*) occur in seedings of introduced grasses (Maser and others 1984). Public concern led to the increased use of browse species in wildlife habitat treatments. Blaisdell (1972) reported that research in shrub ecology had contributed to the identification of about 75 shrubs as promising for improving big game habitat (see Plummer and others 1968). Four shrubs, big sagebrush (*A. tridentata*), fourwing saltbush (*Atriplex canescens*), antelope bitterbrush (*Purshia tridentata*), and rubber rabbitbrush (*Chrysothamnus nauseosus*), were considered primary species to be promoted. The use of big sagegrush and other shrubs in rangeland rehabilitation treatments on Federal lands has gradually increased since the mid to late 1980s. More recently, the decline of sage-grouse and other sagebrush obligate species has given additional impetus to restoration of big sagebrush habitats. Although an additional two decades have passed, we still have much to learn about restoring this landscape dominant and its associated species to disturbed lands. Here we provide a review of recent big sagebrush restoration literature and recommendations for reestablishment and management of this species and its communities.

Natural Regeneration of Big Sagebrush

Most *Artemisia* species, subspecies, and ecotypes are easily killed by fire. They do not resprout and therefore must regenerate from seed. Of the five subspecies in the big sagebrush complex (table 1), only subalpine big sagebrush (*A. t.* spp. *spiciformis*) can resprout from root crowns or lower stem bases after being top-killed by burning (Winward 1985). Fire passing through a Wyoming big sagebrush (*A. t.* ssp. *wyomingensis*) plant will usually kill it (Britton and Clark 1985).

A big sagebrush plant may produce 500,000 seeds in a typical year (Welch and others 1990), but annual production varies greatly (Young and Evans 1975). Big sagebrush seeds are small and exceedingly light; those of basin big sagebrush (*A. tridentata* ssp. *tridentata*) are generally lighter (0.018 g/100 seeds) than those of mountain and Wyoming big sagebrush (0.025 g/100 seeds) (Meyer and others 1987). Big sagebrush seeds are dispersed primarily by gravity. Maximum dispersal distances are only about 30 m from the parent plant; 85 to 90 percent of all seeds fall within 1 m of the edge of the mother plant (Wagstaff and Welch 1990; Young and Evans 1989). Consequently, long-distance dispersal by wind is ineffective in recolonizing large burns or other disturbances (Meyer 1994).

Artemisia seeds rarely survive in the soil for more than a year (Caldwell 1978; McDonough and Harniss 1974; Walton and others 1986). However, some seed may carry over if buried and not exposed to light (Hassan and West 1986; Meyer 1994; Meyer and Monsen 1990; Richardson and others 1986). Schuman and others (1998) found that Wyoming big sagebrush seed survived up to 4 years when applied with mulch on mine spoils in Wyoming.

Table 1—Big sagebrush complex.

Common name	Scientific name
Subalpine big sagebrush	*Artemisia tridentata* ssp. *spiciformis*
Basin big sagebrush	*A. t.* ssp. *tridentata*
Mountain big sagebrush	*A. t.* ssp. *vaseyana*
Few-flowered mountain big sagebrush	*A. t.* ssp. *vaseyana* f. *pauciflora*
Wyoming big sagebrush	*A. t.* ssp. *wyomingensis*
Xeric big sagebrush	*A. t.* ssp. *xericensis*

Rapid reestablishment of most big sagebrush subspecies is more likely on sandy or gravelly soils that are well suited for supporting the species. Big sagebrush returns more slowly on fine-textured soils that have a greater potential for production of herbaceous species (Blaisdell and others1982; Hironaka and others1983). Xeric big sagebrush (*A. t.* ssp. *xericensis*) is the only taxon in the big sagebrush complex adapted to fine-textured clay soils.

Natural postfire reestablishment of big sagebrush has not been widely documented. During years of low precipitation, few Wyoming big sagebrush plants may establish, and it may take many years before recolonization takes place. Even under favorable conditions, site recovery may require 60 to 100 years. On dry Wyoming big sagebrush sites, several years may pass before conditions favoring establishment of new seedlings occur (Clifton 1981; Lowe-Dalzell and others 2003; Wambolt and Payne 1986; West and Hassan 1985; Young and Evans 1978). Because of these factors, big sagebrush must be artificially reseeded on sites where seed sources have been lost.

Postfire, Pretreatment Site Evaluation

Prior to treatment, it is imperative that a site evaluation be conducted to assure that artificial restoration measures are needed and that natural recovery will not occur within an acceptable time frame (fig. 1). If recovery is not anticipated without seeding, the preburn density of exotic annuals and the postburn seed bank of these species must be estimated to determine the potential for restoring the site, the overall objectives must be established, and the approach for accomplishing the seeding or planting must be selected.

The characteristics of various ecological sites and their distribution within a given management area should be thoroughly understood. Site characteristics vary according to the potential natural community, species present, soil depth and texture, effective precipitation, erosion potential, elevation, aspect, and other factors (National Research Council 1994). Burned big sagebrush sites that receive less than 250 mm of annual precipitation, particularly where the understory is cheatgrass (*Bromus tectorum*) dominated, have a low probability of regenerating naturally and providing preburn cover and structure in a reasonable length of time. This is due to inadequate seed supplies on surviving plants or in the soil seed bank and the combined effects of low and erratic precipitation and herbaceous competition from exotic annuals (Boltz 1994). It is these lands that are in the most urgent need of restoration, but are the most risky to treat. For such sites, a greater investment of time and money will be required, and priorities, objectives, and resource availability are particularly important considerations. Adapted species and subspecies must be planted using procedures that remove competition and create suitable seedbeds. Developing measures to remove or diminish competition is difficult, but failure to implement all proven site improvement measures significantly reduces the chance of success (Monsen and McArthur 1995; Stevens and Monsen 2004). For example, herbicide application or the use of container or bareroot transplant stock may be necessary

Figure 1—Site evaluation form.

Site Evaluation Form

Name ______________________ Date ______________________

Fire name ______________________ Fire No. ______________________

Date fire started ______________________ Date fire controlled ______________________

District or Forest ______________________ Elevation ______________________

Acres burned and ownership: ________ total acres ________ public ________ State ________ private

Preburn vegetation types and estimated acres of each: ______________________

____

____

____

Preburn ecological site(s) and estimated acres of each:

Range/ecological condition: ______________________

Precipitation zone(s) ______________________

Fire severity: ________ acres low ________ acres moderate ________ acres high

Soil series/name ______________________ Soil depth according to survey ______________________

Soil description and texture ______________________

____

Current land use(s) ______________________

Grazing season of use/type of system (specific dates) ______________________

____

Acres/AUM ______________________ No. pastures ______________________

Range condition ______________________

____

Is use pattern map available? ______________________ (if so, please attach copy)

Key wildlife seasonal habitat? ______________________

Noxious weeds? Species and occupied acres (attach map) ______________________

____

Resource objectives: ______________________

____

Fencing (describe preburn and identify additional needs) ______________________

____

____

Recommended treatments (include for noxious weeds) ______________________

____

____

Seed mix(es), rates, method of application, PLS cost:

Drill seeded—Seed mix 1	Drill seeded—Seed mix 2	Aerial seeded—Seed mix 3
species/subspecies/rate/cost	species/subspecies/rate/cost	species/subspecies/rate/cost
Totals:	Totals:	Totals:

under such conditions. Evaluation forms such as figure 1 may be used to facilitate the decisionmaking process.

Stevens (2002) recommends the following four steps be followed in selecting taxa for a seeding: (1) develop a list of species and ecotypes that would occur on the proposed planting site; (2) from this list, determine which species have a significant amount of high quality seed available for planting; (3) of these available species, determine those that are compatible as young developing plants and that will ensure ecological development of a desired plant community; and (4) evaluate the final species list to determine if project objectives can be achieved or whether the initial objectives require revision.

To successfully reestablish big sagebrush, the subspecies present preburn must be determined and utilized in the seeding effort. Remnant plants may be identified using the descriptions provided by Goodrich (this proceedings) and Rosentreter (this proceedings). Subspecies and populations of big sagebrush have evolved in distinct environments. Common garden studies have revealed differences in adaptive characteristics such as drought or frost tolerance (McArthur and Welch 1982; Meyer and Monsen 1990; Welch and others 1992); movement of populations to different climatic or edaphic conditions is not advised (Mahalovich and McArthur 2004; Monsen 2000). Specific ecotypes may be especially important on droughty sites or mineral soils. Matching treatment site characteristics, such as soil type and elevation, with the seed source is critical, but this has frustrated land managers and in some cases been impossible during large fire years when seed is in high demand and production low. Consequently, it is all the more imperative that the correct big sagebrush subspecies be used.

Seed Biology and Technology

Seed Harvesting and Conditioning

Big sagebrush flowers in summer and is wind pollinated. Large numbers of tiny flowers develop on spikes, racemes, or panicles, with individual plants producing hundreds of thousands of achenes (Welch and others 1990) in years with favorable weather conditions. Seeds (achenes) ripen in late fall and are usually dispersed within a few weeks of reaching maturity, depending upon weather conditions and subspecies. Seed of mountain big sagebrush generally ripens earlier than seed of basin big sagebrush or Wyoming big sagebrush, and considerable variability in date of ripening will be found within individual plants and populations. In addition, seed production varies widely from year to year based on weather conditions, herbivory, seed predation, and other factors (Wagstaff and Welch 1991; Young and others 1989). Because of these factors, seeds should be checked carefully with a hand lens before harvesting to ensure that adequate quantities of sound seeds are present to justify harvesting. Seed harvested too early will be immature and not viable. Delaying the issuing of permits until seed has matured has been suggested as one means of discouraging early harvest on public lands (AOSA 2003). Seed harvested too late, after dispersal of most sound seeds, will include large quantities of poorly developed seeds and fruit and flower parts. An additional complication is the frequent occurrence of basin big sagebrush and Wyoming big sagebrush mosaics with the basin big sagebrush growing in deeper soils or along roadways or along riparian areas. It is essential that care be taken to collect only the target subspecies.

Seed is hand harvested by beating or stripping the inflorescences into seed hoppers, boxes, bags, or other containers. Harvesting should be done when the humidity is low because the fruits separate more easily from the inflorescences when dry. Average moisture content of fully ripened seeds of big sagebrush has not been examined carefully. Moisture content of seed and debris is often high when seed is harvested from plants that are covered with snow or frost in late fall. Seed is initially dried to a moisture content of 18 to 20 percent before cleaning to protect seed viability and to reduce the volume of material to be conditioned (AOSA 2003). Appropriate drying techniques and rates and their effects on seed quality require further investigation; rapid seed deterioration with improper handling is considered a major obstacle to maintaining big sagebrush seed viability beyond the first year (AOSA 2003).

Purity of harvested seed lots is extremely low due to the presence of inflorescence branches, leaves, bracts, poorly developed fruits, and other debris. Seed is cleaned using a barley debearder or hammermill to break up the inflorescences and other debris. Screening and fanning then removes trashy material. Big sagebrush seed is generally cleaned to 10 to 15 percent purity (Stevens and others 1996), but purities of 80 percent or more can be obtained by further cleaning with an air screen separator. Cleaning to a purity of 35 percent has been suggested as a means of reducing bulk and cost for shipping and storage, increasing the consistency and accuracy of seed sampling and seed quality testing, improving the regulation of seed moisture content in storage, and facilitating seed metering through seeding devices (AOSA 2003; Welch 1995).

For current seeding practices, purity of 10 to 12 percent and viability of 85 to 95 percent is recommended by Meyer (2005). Lambert (2005) recommended 14 percent purity and 80 percent viability as minimum standards for USDI Bureau of Land Management purchases of big sagebrush. If all large debris is removed, seed cleaned to this level can be seeded through broadcast seeders, rangeland drills, Hansen browse seeders, and other standard seeding devices (Shaw and Monsen 1990). Maximum allowable moisture content can also be listed in purchase specifications.

Seed Storage and Longevity

Following late fall harvest, big sagebrush seed must be dried, cleaned, and tested before it can be sold. Consequently, a considerable amount of newly harvested seed is not marketed before the late fall seeding period, but must be held over in storage for at least 1 year. Storing seed at moisture contents of 6 to 8 percent and a temperature below 10 °C (Meyer 2005) may lengthen viability to as much as 5 years. Storage of seed under adverse conditions, even for short periods, can negatively affect seed quality and translate into a rapid decline in viability and vigor. Thus it is advised that seed be tested for viability before purchasing or seeding in order for prices and seeding rates to be determined accurately.

Seed Testing

Testing of big sagebrush seed lots is plagued by a number of problems stemming from the small seed size, low purity levels, and the large size of marketed seed lots. Additional research is urgently needed to provide guidelines that will aid users in maintaining and more accurately measuring seed quality. Such guidelines would reduce problems related to marketing, handling, and seeding big sagebrush.

Problems in assessing purity arise from sampling procedures at the warehouse and in the laboratory. Seed lots are often large and heterogeneous. Big sagebrush seed does not flow and samples must be drawn from bags by hand, a technique that introduces more variability than use of a trier or probe; thus purity of samples drawn from a single seedlot for submission to the seed laboratory can differ substantially. Initial samples drawn from large seed lots may be too large to submit to the laboratory and will require further subsampling, thus introducing additional variability. When the submitted sample reaches the seed laboratory, the working sample is obtained by dividing the sample by hand, a less reliable technique than use of mechanical dividers used for seeds that flow readily. The AOSA (2003) suggested variability in samples might be reduced by limiting seed lot size and by marketing big sagebrush seed at purities in the 35 percent range. Seeds might then be classified as flowable, and triers and mechanical dividers could be used for sampling, thus improving sampling consistency.

Purity testing for big sagebrush is slow and costly. In addition to sampling problems, the small seed size, large amounts of debris present, and problems related to selection of pure seed increase the time required for completion of tests and reduce the accuracy of results (AOSA 2003). Again, increasing purity levels to the 35 percent range would reduce the bulk of seed and debris that must be examined, remove many of the seeds that are small, nonviable, or poorly developed; speed the testing process considerably; and reduce the variability of results.

Procedures for testing viability of members of the genus *Artemisia* are provided by AOSA (2000). Results can be obtained quickly, depending primarily on the laboratory's backlog. AOSA germination tests are available for big sagebrush, black sagebrush (*A. nova*), and Louisiana sagebrush (*A. ludoviciana*) (AOSA 2000). The germination test for Louisiana sagebrush requires 14 days. Germination tests for big sagebrush and black sagebrush require 21 days; dormant seedlots require a 14-day prechill. Meyer (2005) recommends testing nongerminating seeds for viability as not all dormant seed will respond to the short prechill. Sampling problems and identification of pure seed reduces consistency of results.

Seed shipped for purity and germination or viability testing may be packaged in paper bags or containers. Seeds shipped for moisture testing should be packed in plastic bags to maintain the water content at the same level as the seed lot. Use of the International Seed Testing Association rule for testing moisture content (drying at 105 ^{o}C for 16 hours) should be specified for determining seed water content (AOSA 2003).

Germination and Seedling Establishment

The level of seed dormancy and the light requirement for germination vary widely among big sagebrush seed sources and tend to decline with afterripening in dry storage or with a moist prechill. Compared with seed from lower elevations, seed from high elevations generally requires a longer field or laboratory stratification to release dormancy and reduce the light requirement (Meyer 2005). Seeds that have lost their dormancy germinate rapidly under favorable moisture conditions. Likewise, germination under snow occurs slowly at high elevations, while only a short period of snow cover may facilitate rapid germination of low elevation seed sources (Meyer and Monsen 1990; Young and others 1990). Germination is highly erratic on dry and windy sites where snow cover is less reliable; seeds from such locations may be capable of germinating rapidly, even at low temperatures, when moisture conditions are favorable (Meyer 1994; Meyer and Monsen 1992). Favorable microsites for germination are provided if seeds are placed at or near the soil surface and pressed into a firm, but not compacted seedbed. This provides the exposure to light required for germination and good seed to soil contact for improving water uptake. Imbibed seed produces a layer of mucilaginous material that improves adhesion to the soil. In addition, the hairs that develop on emerging hypocotyls also aid in water uptake and soil contact (Walton and others 1986; Young and Martens 1991).

Although seedlings sometimes establish in large numbers due to high seed production, favorable weather, and appropriate microsite conditions, most seedlings are generally lost to late frosts or drought, disease, inter- or intraspecific competition, herbivory, or other factors. Seeding methods or techniques that provide favorable microsites or improve snow or water catchment, as well as the presence of mature shrubs that can function as nurse plants improve establishment (Monsen and others 1992).

Seeding Considerations

Artificial seeding should only be pursued when the objective is to reestablish shrubs more rapidly than would occur by natural recovery (Shaw and Monsen 1990). However, some circumstances such as severe site conditions or degradation, complex ownership patterns, absence of crucial habitat, small size of a treatment area relative to others in need of seeding, budgetary constraints, or some combination of these factors may render seeding impractical. Decisions are best made following a field site evaluation (fig. 1).

Site preparation and seeding techniques that reduce early competition from both annual grasses and seeded species and provide suitable microsite conditions for germination and early growth (Meyer 1994) must be selected. On degraded sites, extensive site preparation and weed control will be necessary when dense stands of annuals are present preburn or are expected to develop. Not only will preparation of a firm seedbed be required, but also control of exotic annual grasses, primarily cheatgrass (*Bromus tectorum* L.) and medusahead wildrye (*Taeniatherum caput-medusae* L.), will be necessary.

Recommended big sagebrush seeding rates range from 0.11 to 0.22 kg per ha pure live seed (PLS) (Meyer 1994;

Monsen 2000); increases of up to 50 percent are recommended for broadcast seeding (Welch and others 1986). Seeding rates should be calculated on a PLS basis using results of a recent viability or germination test (Meyer 1994). Because of the extreme variation in microsites, moisture availability, and temperature conditions encountered by seeds and seedlings as well as a lack of research and monitoring data, more definitive recommendations are not possible.

Due to its small size, big sagebrush seed is usually mixed with a carrier, thus cleaning to a higher purity than the commercial lot average of 10 to 20 percent PLS may not be cost effective or necessary (Meyer 1994). However, newer drills may be able to seed lots with higher purities at acceptable rates, thus reducing the bulk of seed lots required for individual projects; additional research is required to examine this possibility.

Seeding in late fall or early winter is recommended, as this is when big sagebrush naturally disperses and soil surfaces are more likely to be moist and firm; it also permits the stratification required to attain vigorous germination if adequate moisture is present. Spring seeding should be avoided (Meyer 1994).

Seeding Techniques

Big sagebrush seed should be planted on a firm seedbed with only a light covering of soil. Smooth, compacted seedbeds do not offer good seed to soil contact. Rough seedbeds may slough and bury seeds too deeply. Big sagebrush can be seeded with other species to increase diversity; however, seeding requirements and the relative seedling growth rates of each species must be considered when writing a seeding plan. Due to their earlier maturity, seeded grasses establishing with big sagebrush have an initial advantage and suppress big sagebrush seedlings. Dense stands of seeded grasses may entirely suppress big sagebrush seedlings or prevent big sagebrush reestablishment for an indefinite period (Blaisdell 1949). Due to these concerns and as a general rule of thumb, grass should be seeded at low rates (3.6 to 5.4 kg/ha) if big sagebrush establishment is one of the treatment objectives.

Seeding has frequently been accomplished by aerial broadcasting to keep seeds near the soil surface and to plant large, rough areas rapidly. Ground broadcasting using mechanical seeders or hand seeding is also commonly used. Coverage of broadcast seed using chains, harrows, rails, or other implements is recommended (Stevens and Monsen 2005). Lysne (this proceedings) found that in southern Idaho, big sagebrush seeded aerially and not covered failed to establish on 23 of 35 fire rehabilitation projects examined, while natural regeneration occurred on about one-fourth of the projects. Elevation on her sites ranged from 810 to 1,640 m and annual precipitation from 150 to 305 mm. Overall big sagebrush density did not differ between seeded and nonseeded portions of these projects. As alternative treatments for this area, Lysne (this proceedings) and Lysne and Pellant (2004) recommended seeding methods that create a firm seedbed and press the seed into the soil, thus at least some sagebrush seed is placed near the soil surface. They suggested use of equipment such as the Oyer compact row seeder (Monsen and Meyer 1990), Brillion cultipacker seeder (Monsen and Meyer 1990), Jarbidge big sagebrush seeder (Boltz 1994), or land imprinter (Monsen 1988; Haferkamp and others 1987).

Big sagebrush can be seeded through drills if seed is dropped on or near the soil surface and covered lightly by press wheels (Lambert, this proceedings) or by pulling an implement such as a cultipacker behind the drill. For drill seedings, Richardson and others (1986) recommended that big sagebrush be planted in separate rows from grass and forb species. Otherwise, due to their rapid development, grasses and forbs will compete directly with the slower growing shrub seedlings (Richardson and others 1986) for water and other resources.

New drills equipped with multiple seedboxes, seeding depth regulators for each drop, and surface compaction attachments offer greater flexibility for planting different species in separate rows (Boltz 1994; Wiedemann 2005). Addition of a fluffy seed box to the rangeland drill has also increased options for seeding sagebrush. The Utah Division of Wildlife Resources has recently begun purchasing big sagebrush seed cleaned to 30 percent purity with a minimum of 80 percent germination giving a PLS of 24 percent. Seed harvested and purchased in late autumn is placed in cold storage by December or January and seeded the next autumn. Use of Truax drills or rangeland drills with a fluffy seed box permits use of this seed without addition of a carrier and has reduced problems associated with seed testing (Vernon 2005).

Interseeding is another approach to establishing big sagebrush and other slow growing shrubs. This technique involves disking, plowing, or spraying to remove strips of established vegetation such as introduced grass seedings or invasive species. Shrubs and other species that are slow to establish are then seeded using a Hanson seeder or thimble seeder. Interseeders have been constructed to accomplish mechanical removal of existing vegetation and seeding in one pass (Stevens and others 1981; Wiedemann 2005). The Hansen seeder has also been used to drop big sagebrush seed and other shrubs ahead of the wheels of a tractor or the tracks of a caterpillar. The wheels or tracks create a firm seedbed and press the seed into the soil. Seed of a variety of species can be planted using this method as seeds are placed over a range of depths. Grasses and larger seeded forbs are seeded through the drill.

Hydroseeding is generally impractical for large rangeland rehabilitation or restoration projects. This technique is labor intensive and expensive. In addition, many sites are difficult to access with hydroseeding equipment or water trucks. Moreover, good seed to soil contact is generally not provided by incorporating the seed into the mulch on dry sites.

Establishing early seral native grasses and shrubs such as rubber rabbitbrush on burned or otherwise disturbed sites may reduce annual weed density and permit establishment of big sagebrush seeded at a later time. Summer precipitation occurring following the senescence of native herbaceous species may enhance big sagebrush establishment. Meyer and Monsen (1990) found evidence that previously establishing rubber rabbitbrush may have facilitated colonization of big sagebrush on a mined site in Nevada. Naturally reestablishing rubber rabbitbrush (*Chrysothamnus nauseosus*) invaded mine spoils at the Beacon Pit Mine that had not been covered with topsoil (Meyer 1994). Ten years following disturbance, more than 60 percent of the rubber rabbitbrush plants were in the adult size class, while about 70 percent of the big sagebrush plants were less than 30 cm tall. This suggests that initial establishment of the rubber rabbitbrush may have ameliorated site conditions and facilitated big sagebrush establishment. Planting disturbed areas to rabbitbrush to enhance big sagebrush establishment is feasible and ecologically practical (Monsen 2000). Rabbitbrush can be established either by seeding or transplanting. It is capable of establishing and spreading to sites occupied by cheatgrass. Rabbitbrush plants aid in trapping snow, moderating temperature extremes, and accumulating litter, all beneficial for the big sagebrush seedling environment.

Nursery Stock

To enhance shrub and forb communities in the Intermountain West, container-grown and bareroot stock have proven effective for increasing diversity (Stevens 1994). Because of the expense, the usefulness of transplanting seedlings may be limited to small, critical areas, if high shrub densities are required. Bareroot planting stock should be from 12 to 20 cm tall and over-wintered in a nonheated nursery bed or lathhouse (Welch and others 1986). Long and Trimmer (2004) reported that at the Lone Peak Conservation Nursery, 1-0 big sagebrush seedlings are root pruned at a depth of 30 cm in August. Following February or March lifting, seedlings are graded to specifications of a minimum of 15 cm shoot length and 4 mm collar diameter.

Transplanting should generally be done in spring when soil moisture and the chance of storms are greatest, temperatures are low, and frost heaving has ceased (Deitschman 1974; Stevens 1981; Welch and others 1986).

While transplanting is more expensive than direct seeding, success is often much greater and more evident. The more edaphically or climatically severe the site, the greater is the need for transplanting (Stevens 1981).

Everett (1980) found that planting containerized shrubs, including mountain big sagebrush, in late winter (February) was a viable method of establishing vegetation in the harsh environment of roadside cutbanks in the Sierra Nevada foothills (Everett 1980). Initial establishment was highly dependent on quality of planting stock and weather conditions. In every instance where small or insufficiently hardened planting stock was used, survival rates declined drastically (Everett 1980). Tiedemann and others (1976) reported on the importance of adequate transplant size for survival of shrubs in eastern Washington.

Stevens and others (1981) found that bareroot stock of many native shrubs, including mountain big sagebrush, could be planted successfully with a hand-fed tree planter in scalps 0.6 m on a side and 0.2 m deep made in heavy grass sod. The transplanting rate varied between 10 and 18 plants per minute depending on plant species, size and condition of plants, soil type, and surface conditions. Shrubs were planted at spacings of 0.9 to 2.4 m. Establishment was greater for bareroot stock than for container-grown stock. Bareroot stock with roots 15 to 30 cm long and tops at least 8 cm long were most successful.

Seeding may also be accomplished by employing the "mother plant" concept—big sagebrush transplants are planted on a 15- by 15-foot grid. These "mother plants" mature and produce enough seed in 3 to 5 years to supply the seed for natural dispersal throughout site if native grasses are reestablished to reduce weedy competition (Welch and others 1986). Mechanical or chemical treatments may be necessary to reduce competition in strips or scalps at the time of planting.

Postseeding Management

On Bureau of Land Management lands, seedings are typically excluded from livestock grazing for two growing seasons to allow establishment. Stevens (1994) found that grazing pressure must be removed from newly planted or seeded areas for a minimum of 2 years. However, others have suggested that longer periods of rest from livestock and wildlife use are probably needed, particularly when attempting to reestablish shrubs such as big sagebrush (Fisser 1981). Protected plants develop more rapidly and natural spread from seed is hastened with longer protection. Shaw and Monsen (1990) stated that 2 to 3 years of protection from livestock grazing reduces seedling losses from grazing or trampling. Richardson and others (1986) compared grazed and ungrazed treatments 7 years after seeding mountain big sagebrush on a mid-elevation site in southeastern Idaho. They found significantly lower big sagebrush densities in the grazed treatment, an effect they attributed, in part, to trampling.

Stevens and others (1996) reported that 20 to 40 percent of big sagebrush transplants in seed orchards produce seed by the second year, and 80 to 90 percent by the third and fourth years. When seed orchards are established from seed, 10 percent of the plants can be expected to produce seed by the second year, 30 to 50 percent by the third year, and 80 to 90 percent by the fourth to fifth year. On wildland sites, longer periods may be required, particularly under drought conditions.

Plummer and others (1968) stated that planted areas must not be overgrazed. Until seeded stands have become established and suppressed natives have had an opportunity to recover and become reproductive, livestock grazing should be light if permitted at all. After range restoration has been accomplished, grazing should be conservative. Either fencing or management of animals is often necessary to give young plants adequate time to attain mature stature. Protection fences are installed to protect a new seeding from grazing and trampling during the establishment period and to manage established seedings (Interagency Burned Area Emergency Stabilization and Rehabilitation Handbook 2002).

Management is also required to restrict off-highway vehicles and other human activities that may impact seeded areas. Early weed control may be required to reduce the risk of seeding failure and spread or recovery of invasive species.

Monitoring Seeding Establishment

Appropriate monitoring protocols (for example, Elzinga and others 1998) should be selected to measure the extent to which seedings are successful in meeting management goals and to provide for adaptive management. Establishment and monitoring of unseeded controls and grazing exclusion plots on seeded and unseeded areas permit evaluation of seeded species establishment, natural recovery, and the impacts of livestock grazing (Lysne, this proceedings; Lysne and Pellant 2004). Regular monitoring during the first few years postseeding, records of seed lot history (origin, quality, storage conditions) and seeding techniques applied, a site description, site conditions during the time of seeding, and weather records are invaluable for evaluating monitoring results, establishing the time required for individual species to reach reproductive status, determining readiness for grazing, and suggesting modifications for future seeding or planting efforts.

Conclusions and Recommendations

Big sagebrush and associated native species can be seeded on sites where seed sources have been lost and natural recovery is not expected to occur. Careful planning; acquisition of adapted, high quality seed; selection of seeding techniques appropriate for individual species as well as the combination of species selected; and careful post-seeding monitoring and management are all required to maximize seeding success, permit recovery of remnant native species, and maintain established seedings. Additional research and improved technology are required to solve problems related to maintaining and evaluating seed quality, providing seeding techniques that place big sagebrush seed in appropriate microsites for germination, and protect them from herbaceous competition, whether from co-seeded or invasive species. Low and erratic precipitation on drier big sagebrush sites often limits seeding success.

A major obstacle to the increased use of big sagebrush is the problem of obtaining adequate seed supplies of the required subspecies from adapted sites when needed. The difficulty of identifying the subspecies in individual seed lots, seed lots containing mixtures of subspecies, limited shelf life of big sagebrush seed, and inadequate cold storage space contribute to this problem. Efforts to delineate seed transfer zones for *Artemisia* taxa (Mahalovich and McArthur 2004) and a recent research initiative to select and manage wildland stands of Wyoming big sagebrush for seed production seek to address this issue. In situ conservation and protection of selected big sagebrush stands in areas where reseeding is likely to be required could increase the availability and quality of adapted seed.

Current literature and knowledge on seeding and establishment of big sagebrush subspecies have been summarized by Stevens and others (2004), McArthur and Stevens (2004), Lysne and Pellant (2004), and others. Seasonal habitat requirements for sage-grouse and recommendations for restoring degraded sage-grouse habitats are described in a number of publications including Connelly and Braun (1997), Connelly and others (2000), Crawford and others (2004), and Wambolt and others (2002). The SAGEMAP Project Web site (http://sagemap.wr.usgs.gov/sage_grouse.htm) provides a library of texts and databases for all aspects of shrub steppe and sage-grouse management in the Intermountain West.

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Big sagebrush

Sagebrush-Ungulate Relationships on the Northern Yellowstone Winter Range

Carl L. Wambolt

Abstract—Sagebrush (*Artemisia*) taxa have historically been the landscape dominants over much of the Northern Yellowstone Winter Range (NYWR). Their importance to the unnaturally large ungulate populations on the NYWR throughout the twentieth century has been recognized since the 1920s. Sagebrush-herbivore ecology has been the focus of research on the NYWR for 2 decades. Specific research topics include general ungulate use of the NYWR, forage relationships involving sagebrush taxa, the effects of browsing and sagebrush taxa on forage production, the results of long-term rest from severe browsing on the NYWR, sagebrush plant and community characteristics related to herbivory, and the relationships of sagebrush taxa to fire on the NYWR. Sagebrush taxa have been very important on the NYWR but are generally in a state of decline.

The Northern Yellowstone Winter Range (NYWR) is dominated by big sagebrush (*Artemisia tridentata*) taxa. Big sagebrush occupies relatively snow-free portions of the NYWR, making it accessible to ungulates throughout the winter. Heavy foraging use of sagebrush by the large ungulate populations of the region has been discussed by naturalists for almost a century (Wright and Thompson 1935) (fig. 1). My research has confirmed that early naturalists were right to be concerned about sagebrush habitat on the NYWR. Those investigations have determined the mechanisms that influence sagebrush-herbivore interactions on the NYWR and their effects.

Area Description

The NYWR is an extensive area covering about 100,000 ha along the lower elevations in Northern Yellowstone National Park (YNP), and extends northward into Montana along the Yellowstone River drainage (fig. 2). The Lamar and Gardiner River drainages, important lowland areas within YNP, are relatively free of snow and provide reliable winter foraging for ungulates. During the winter, 80 percent of all ungulates are found on the NYWR within YNP. In addition, there have been as many as 2,544 Rocky Mountain mule deer and 8,626 Rocky Mountain elk wintering on the portion of the NYWR north of the YNP boundary in Montana during the last decade.

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The climate is favorable in this area for ungulate foraging during winter. The mean annual precipitation is approximately 280 mm (1,616 m) at Gardiner, MT, 400 mm (1,899 m)

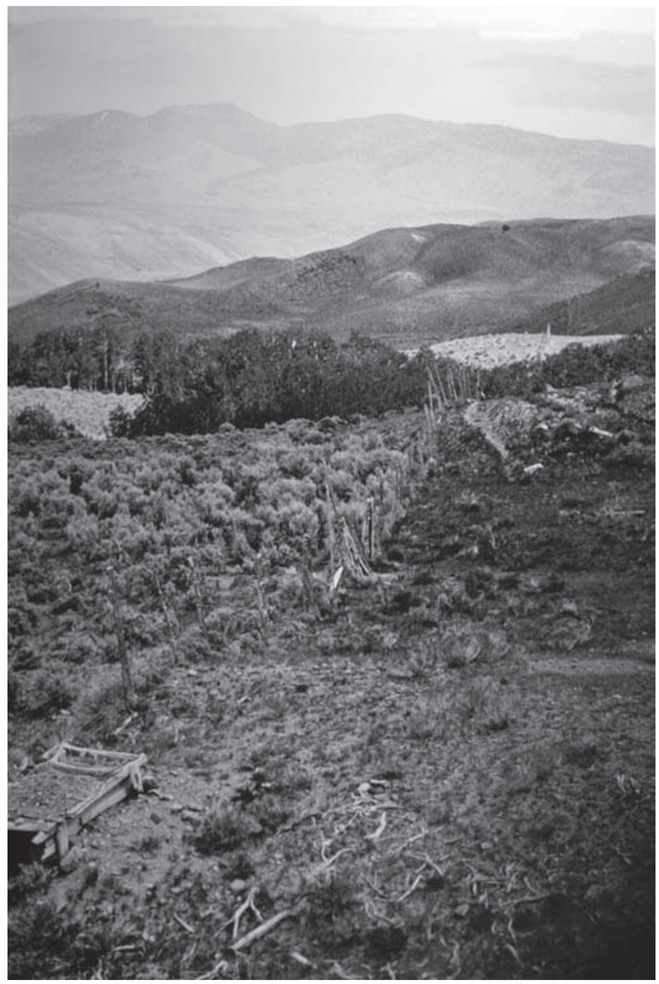

Figure 1—Photo from Wright and Thompson (1935). The original caption read: “End of the range. The right side of the picture shows elk winter range within Yellowstone National Park. The fence is the boundary. The left side of the picture shows the rank growth of sagebrush just outside the park. Here we can compare the original and the present state of the range. There can be no doubt of what has happened. (Photograph taken June 1, 1932, near Gardiner, Wildlife Division No. 2501.)”

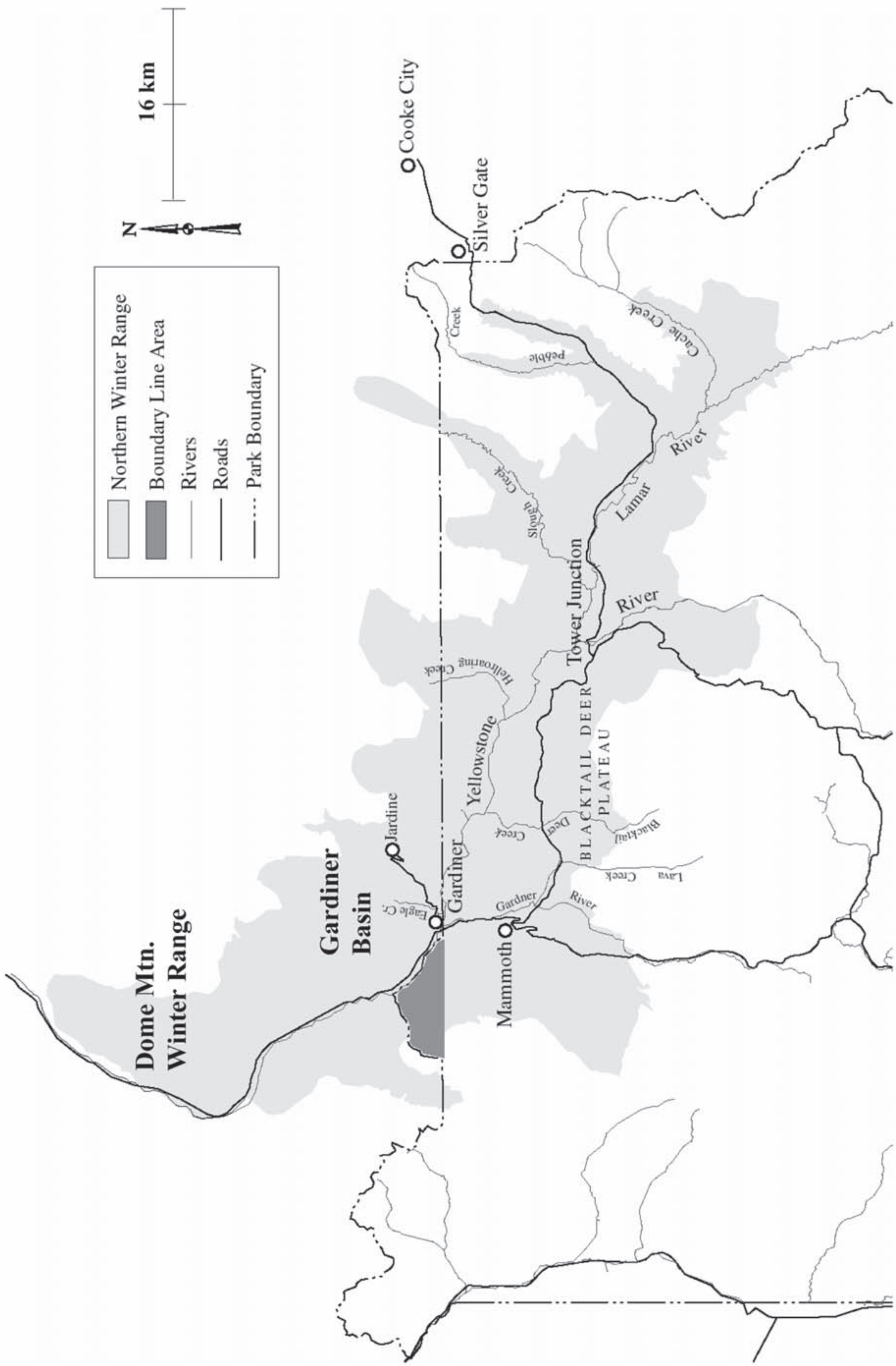

Figure 2—The Northern Yellowstone Winter Range, Montana and Wyoming (National Park Service Map).

at Mammoth, and 410 mm (1,912 m) at Tower Falls. About half of the moisture each year is received as snow, the peak occuring in spring and early summer.

The Gardiner Basin, an area near the town of Gardiner, MT, includes the boundary line area (BLA), and has less snow than the majority of the NWYR that lies to the east in YNP. Therefore, the Gardiner Basin is especially important for pronghorn and mule deer. Elk are usually able to negotiate deeper snow at higher elevations. Elk and bison tend to dominate the foraging over the remaining portions of the NYWR, but there is no strong evidence that bison have had an impact on the shrubs of sagebrush habitats. Therefore, elk are the only browsing ungulate found in significant numbers during winter on the portions of the NYWR outside the Gardiner Basin.

Not only is big sagebrush the largest vegetation type on the NYWR, but perhaps more importantly, sagebrush taxa are the dominant vegetative form on the portions of the NYWR that are most valuable as winter range for ungulates (DeSpain 1990). Sagebrush communities also furnish important security and thermal cover for ungulates and other animals (Wambolt and McNeal 1987).

The variety of sagebrush habitats within NYWR includes the Wyoming big sagebrush- (*Artemisia tridentata wyomingensis*) bluebunch wheatgrass (*Agropyron spicatum*) type that is found primarily at lower elevations in the Gardiner Basin. Mountain big sagebrush- (*A. t. vaseyana*) Idaho fescue (*Festuca idahoensis*) habitat type dominates the majority of the NYWR (Wambolt and Sherwood 1999). Other important sagebrush are basin big sagebrush (*A. t. tridentata*) and black sagebrush (*A. nova*). Important subdominant plants are the sprouting shrubs often found in sagebrush habitats including rubber rabbitbrush (*Chrysothamnus nauseosus*), green rabbitbrush (*C. viscidiflorus*), and gray horsebrush (*Tetradymia canescens*). These shrubs are found throughout the NYWR in various sagebrush habitat types.

Results and Discussion

Sagebrush Forage Relationships

The relationships of secondary compounds found in the foliage of sagebrush taxa to foraging by NYWR ungulates was studied. The relationships between crude terpenoid content of sagebrush and *in vitro* organic matter digestibility (IVOMD) as well as preference by mule deer was considered for the four dominant sagebrush taxa found on the NYWR. Basin big sagebrush was the most easily digested by mule deer followed in order of decreasing digestibility by Wyoming big sagebrush, mountain big sagebrush, and black sagebrush. The IVOMD for all four taxa generally increased from January to April as crude terpenoids decreased in the foliage. When the crude terpenoids were removed from the sagebrush foliage the IVOMD increased. However, terpenoid removal resulted in few differences in IVOMD among the taxa and collection dates during the winter (Striby and others 1987). Wild mule deer from the NYWR were compared to domestic sheep and steers for their ability to digest sagebrush. There were no significant differences found among the three species of animals; therefore, it appears that there should be no significant differences between the wild ungulates (mule deer, elk, and pronghorn) found on the NYWR.

Personius and others (1987) isolated 31 compounds from sagebrush foliage and found that seven of them were particularly important for preference indicators among the four sagebrush taxa. In order to separate the individual plants within a taxon into browse-form classes (use classes), seven other compounds were helpful. The compounds above were used by Bray and others (1991) to differentiate mule deer preference among sagebrush taxa during feeding trials. This study determined if the deer discriminated against forage treated with the individual compounds. During this feeding trial, mule deer selected significantly higher quantities of untreated forage compared to the same forage treated with the suspect compounds (table 1). These studies found that volatility of compounds had little influence on preference at the concentrations found in nature. Both volatile and nonvolatile compounds were found to be responsible for deterring foraging on the four taxa. The combination of nonvolatile sesquiterpene lactones with certain volatiles explained why one sagebrush taxon is often preferred over another.

Site characteristics that might be responsible for elk and mule deer foraging on the NYWR were studied (Wambolt and McNeal 1987). Most portions of the Gardiner Basin — even the timbered areas—were used by elk. However, elk fed mostly in mountain big sagebrush habitat types. The mule deer in the same area preferred to forage in the Wyoming big sagebrush type when it was found adjacent to steep and dry topography at lower elevations. These areas tended to furnish good security and thermal cover while also meeting foraging needs. Elk distribution and concentration on the NYWR vary with wind, snow depth, temperature, and snow crusting that might expose or conceal forage. In addition, hunting north of YNP affects animal distribution on the winter range.

Mule deer and elk preferences for the four sagebrush taxa were studied over 10 winters of varying severity in the Gardiner Basin (Wambolt 1996). This study was purposely long term in order to avoid anomalies over shorter periods that might lead to erroneous conclusions. Approximately 2,500 leaders were tagged on 244 plants each winter over the 10-year period to determine if browsing had occurred.

Table 1—Mule deer preference for seven compounds found in four sagebrush taxa. All seven significantly influence intake and are listed in decreasing order as a foraging deterant.

Compound
1,8-cineole [a]
Black sage NVCTF [b]
Wyoming big sage NVCTF
p-cymene [a]
Basin big sage NVCTF
Methacrolein [a]
Mountain big sage NVCTF

[a] Volatile.
[b] Nonvolatile crude terpenoid fraction.

Overall, the two ungulates browsed large amounts of the four sagebrush taxa, particularly considering that the 10-year period was below average for winter severity. In addition, mule deer diets averaged 52 percent big sagebrush over the 10-year study. The diets were determined by microhistological techniques on composite samples of feces collected early each spring. Both ungulates displayed a distinct preference among the four taxa. The amount of sagebrush use varied with winter and taxon, as expected. Use reached 91 percent for the preferred taxon, mountain big sagebrush, which averaged 56 percent over the 10-year study (fig. 3). Average use of Wyoming big sagebrush was 39 percent and basin big sagebrush was 30 percent. The least preferred taxon was black sagebrush at 17 percent. The results of our study of actual use by wild ungulates supported the findings of Personius and others (1987) and Bray and others (1991) that sagebrush terpenoids affect herbivory.

Regression models were developed to determine the production of winter forage for the three big sagebrush subspecies on the NYWR (Wambolt and others 1994). Models available in previous literature did not consider the variation among sagebrush subspecies nor use classes (developmental differences from past browsing) that might be present. During this study (Wambolt and others 1994), the consideration of taxon and form class increased the R_a^2 an average of 10 percent with values to 0.90. These models help with the determination of carrying capacity, detecting trends in forage production, and measuring plant responses to different management options. In addition, they clearly show the importance of taxon recognition and the role of past browsing history on present forage production of sagebrush taxa (table 2).

Ecology of Big Sagebrush on the NYWR

The question of whether Wyoming big sagebrush plants that had been protected from browsing for 35 years would exhibit similar growth characteristics to browsed plants was considered (Hoffman and Wambolt 1996). To test this hypothesis, paired comparisons in and out of a 2-ha National Park Service (NPS) exclosure constructed in 1957 were made. Because heavy browsing had occurred outside the exclosure for many years, plants there had no terminal leader growth. Plants inside the exclosure were dominated by terminal growth, and axial long shoots were rare. Therefore, to further investigate differences between protected and unprotected plants it was necessary to compare terminal

Table 2—Variables selected to model big sagebrush winter forage production (g).

Variables
Average seedhead weight (g)
Height (cm)
Average cover (cm)
Crown depth (cm)
Circular area (cm^2)
Major axis (cm)
Minor axis (cm)

Figure 3—Elk browsing mountain big sagebrush on the Northern Yellowstone winter range.

leaders of protected plants inside the exclosure to axial long shoots on browsed plants. The unbrowsed plants had a consistently higher production level than browsed plants (table 3). The average production per plant was 10 g with browsing and 45 g with protection. The browsed plants had large amounts of dead crown as well, although no measurements were taken of this parameter. Production of seedheads by big sagebrush plants was the parameter with the greatest difference between browsed and unbrowsed plants. The browsed plants averaged 0.08 seedheads per plant, where the unbrowsed plants averaged 60.3 per plant. Almost all terminal leaders on browsed plants were removed so flowering stems would have to be initiated from elsewhere (Hoffman and Wambolt 1996). This loss of seedhead production from browsing has undoubtedly resulted in reproductive declines for NYWR sagebrush, as these taxa do not reproduce asexually.

The importance of average seedhead production as expressed by their weight was discovered to be important for the construction of regression models to predict sagebrush production of winter forage from the three big sagebrush subspecies found on the NYWR (Wambolt and others 1994). Models improved for all three taxa when average seedhead weight was included for the lightly used (browse form class) plants. Because heavily used plants did not produce many inflorescences, the addition of average seedhead weight to the model for those plants was not beneficial. Separation of browse-form classes and inclusion of average seedhead weight in the models both recognized the impact browsing has had on the annual production and reproduction of NYWR sagebrush.

Snow cover is naturally light on the NYWR, but some snow falls each winter and may protect small sagebrush plants for several years before ungulates find them available for foraging. Among the mountain big sagebrush plants that established between the years 1978—1992, nearly half (47 percent) germinated in 1988. That was the year of the Yellowstone fires. That was also a year that offered relatively good seed production due to good spring moisture followed by a winter with considerably more snow than other winters during the 15-year period. The snow protected the seedlings from herbivores and provided an insulated environment for their welfare during this usually precarious establishment period. These conditions also coincided with a 35 to 40 percent reduction in elk that occurred during that most severe winter (1988—1989) of the 15-year period. This also benefited the seedlings that were allowed to establish during the next several years of reduced elk numbers.

Table 3—Average differences between browsed and unbrowsed Wyoming big sagebrush plants.

	Browsed	Unbrowsed
Production (g/plant)	10.0[a]	44.7[b]
Seedheads per plant	0.08[a]	60.3[b]
Leader length (mm)	22.9[a]	22.3[a]
Leader dry weight (g)	0.02[a]	0.02[a]

Values followed by different letters are significantly different (*P* <0.01).

Fire Relationships

The NYWR sagebrush taxa have no mechanisms to cope well with fire. Therefore, when fire has occurred on NYWR sagebrush habitats, they have been an additive injury, along with intense herbivory, to sagebrush plants. A wildfire in the Gardiner Basin was studied (Wambolt and others 1999) to determine, among other things, the rate of reestablishment of sagebrush and rabbitbrush taxa under continuous browsing following fire. Nineteen years after the fire, recovery of the three subspecies of big sagebrush was at levels between 1 and 20 percent (canopy coverage) of that in adjacent unburned sagebrush stands. This was found despite the fact that the adjacent stands were already in decline from historically heavy browsing (Wambolt 1996; Wambolt and Sherwood 1999). The recovery of Wyoming big sagebrush was less than that of mountain or basin big sagebrush in this burn. Similar relationships were also found for big sagebrush density and production when comparing burned areas to unburned portions.

Wambolt and others (1999) also studied seven prescribed burns conducted by the USDA Forest Service on mountain big sagebrush sites within the Gardiner Basin. These burns had been conducted 10 to 14 years before they were studied. The burns were compared with 33 unburned sites in the Basin to determine recovery following burning. The canopy coverage and densities of the sagebrush on the unburned sites averaged 12 and 15 times greater than on burned sites. This confirms that burning accelerated the browsing-induced decline (Wambolt 1996; Wambolt and Sherwood 1999) of NYWR sagebrush. Sagebrush not directly eliminated by fire were also subjected to unusually heavy browsing on surviving or reestablishing shrubs due to the loss of normally available forage from the fires.

Rens (2001), when studying the effects of fire on the NYWR, considered the effect of elk browsing on the recovery from fires on the Black-tailed Deer Plateau. This investigation also found significant differences in the development of protected and browsed shrubs. One decade after burning mountain big sagebrush, canopy cover averaged 20 percent with protection from elk browsing and 9.7 percent where browsed. Similar differences were found in density and production of protected and browsed mountain big sagebrush following fire on the Black-tailed Deer Plateau.

Browsing Impacts

Wambolt (1996) found that 35 percent of the heavily browsed mountain big sagebrush plants died between 1982 and 1992. Most of the plants that survived the decade of heavy browsing developed a form that clearly showed the effects of their past heavy use, and contained a high percentage of dead crown. The amount of dead crown found in the three big sagebrush subspecies was in direct proportion to the amount of browsing received by each taxon. Among the plants that survived the heavy browsing, the percentage of dead crown from mountain big sagebrush, Wyoming big sagebrush, and basin big sagebrush, was 58.7, 45.4, and 30.1 percent, respectively (Wambolt 1996).

The impact of herbivory on sagebrush across the NYWR was studied by Wambolt and Sherwood (1999) using NPS exclosures that were constructed in 1957 and 1962. The exclosures were originally constructed to learn what would happen under protection from the heavy herbivory that was occurring throughout the northern part of Yellowstone Park. Wambolt and Sherwood (1999) compared shrub parameters between browsed and unbrowsed plants with the aid of these exclosures. They discovered a significant difference between the development of protected and browsed big sagebrush communities across the NYWR.

> Since the period of exclosure construction in 1957 and 1962, there has been a significant difference in the development of protected and browsed big sagebrush communities. Average big sagebrush canopy cover on protected sites was 202% greater ($P \leq 0.0027$) than on browsed sites over the 19-paired sites. The average big sagebrush cover for all 19 sites was 19.7% inside and 6.5% outside the exclosures. This relationship was universal on sites with Wyoming big sagebrush or mountain big sagebrush, flat to very steep topographies, and all aspects and precipitation levels (Wambolt and Sherwood 1999).

Wambolt and Sherwood (1999) found that Wyoming big sagebrush sites that were protected by the exclosures averaged almost 10 times more sagebrush cover than where browsing had continued unabated (fig. 4). Where mountain big sagebrush was the dominant cover type, the same figure was almost three times as much sagebrush cover where protected.

The density of big sagebrush plants was affected in a similar manner by ungulate browsing. Across the NYWR, big sagebrush plants were twice as numerous where they had not been browsed in exclosures as where browsing had occurred throughout the interval since exclosure construction. This amounted to an average density of 30.5 plants per 60 m^2 inside and 15.3 per 60 m^2 outside the exclosures. In addition, individual mountain big sagebrush plants produced 88 percent more winter forage where protected. A similar production figure was not practical to obtain from Wyoming big sagebrush plants even though the differential between protected and browsed plants was even greater than it was for mountain big sagebrush. The Wyoming big sagebrush plants could not be quantified for this character because browsed plants were so reduced in size that their parameters were not suitable for inclusion in the models.

Summary

Both the historical evidence and our recent studies clearly show a significant decline of sagebrush on the NYWR. This decline has already impacted ungulates that rely on sagebrush habitats for nutritional and other requirements (fig. 5). Among the ungulates, it is likely that elk will be the least impacted from the decline in the sagebrush habitat types

Figure 4—National Park Service exclosure near Gardiner, MT, erected in 1957 after Wyoming big sagebrush was nearly eliminated by browsing. Shrubs inside show some recovery (photo taken September 2001).

Figure 5—Even large ungulates use sagebrush for cover. This is one of a dozen elk standing in a patch of basin big sagebrush on the NYWR. Only a portion of the elk's head is visible in the center of photo.

because of their abilities to use other types when available. However, it is clear that elk browsing has been responsible for declines of sagebrush over the majority of the NYWR (Wambolt and Sherwood 1999). It is logical to assume that numerous fauna will be affected by the decline of any native vegetative type. This is especially true with one as extensive as the sagebrush type in the region of the NYWR. The decline of an important habitat type should be given serious consideration when land management opportunities present themselves.

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Insects of the Idaho National Laboratory: A Compilation and Review

Nancy Hampton

Abstract—Large tracts of important sagebrush (*Artemisia* L.) habitat in southeastern Idaho, including thousands of acres at the Idaho National Laboratory (INL), continue to be lost and degraded through wildland fire and other disturbances. The roles of most insects in sagebrush ecosystems are not well understood, and the effects of habitat loss and alteration on their populations and communities have not been well studied. Although a comprehensive survey of insects at the INL has not been performed, smaller scale studies have been concentrated in sagebrush and associated communities at the site. Here, I compile a taxonomic inventory of insects identified in these studies. The baseline inventory of more than 1,240 species, representing 747 genera in 212 families, can be used to build models of insect diversity in natural and restored sagebrush habitats.

Introduction

The Idaho National Laboratory (INL), formerly the Idaho National Engineering and Environmental Laboratory, is located in a cool desert ecosystem characterized by shrub-steppe vegetation communities typical of the northern Great Basin and Columbia Plateau region. Established in 1949 to carry out nuclear energy research and related activities, public access to the 570-thousand-acre INL facility has been restricted for over 50 years. As a consequence, large remnants of relatively undisturbed sagebrush-steppe are still preserved in the interior portion of the site (Anderson 1999). In recognition of the ecological importance of INL lands, the facility was designated as a National Environmental Research Park in 1975 (DOE 1985).

As these important habitats continue to be lost and degraded, interest in the status and condition of remaining sagebrush communities has grown (Entwistle and others 2000; Knick 1999; Knick and Rotenberry 1997). Much is unknown about these ecosystems, and there is an immediate need to establish baselines, fill information voids, and focus research on critical issues, including restoration alternatives.

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Major portions of the INL have been burned by wildfires over the past several years, and restoration and recovery of sagebrush habitat are current topics of investigation (Anderson and Patrick 2000; Blew 2000). Most restoration projects, including those at the INL, are focused on the reestablishment of vegetation communities (Anderson and Shumar 1989; Williams 1997). Insects also have important roles in restored communities (Williams 1997) and show promise as indicators of restoration success in shrub-steppe (Karr and Kimberling 2003; Kimberling and others 2001) and other habitats (Jansen 1997; Williams 1997).

The purpose of this paper is to present a taxonomic list of insects identified by researchers studying cold desert communities at the INL. Insects act as herbivores, decomposers, pollinators, and predators, and they are major prey for reptiles, mammals, and birds inhabiting sagebrush communities, including sage-grouse chicks [*Centrocercus urophasianus* (Bonaparte)]. However, the function of most insects in sagebrush ecosystems is not well understood (West 1983). As natural sagebrush communities disappear, remaining habitats such as those at the INL represent important resources for establishing baseline attributes against which restoration success or indicator measures can be evaluated. A taxonomic inventory is useful for developing reference models of natural insect diversity and community composition in sagebrush habitats at the INL.

Methods

I constructed a baseline taxonomic list of insect species documented in six major investigations at the INL (Allred 1968a; Allred and Cole 1971; Bohart and Knowlton 1977; Karr and Kimberling 2003; Stafford 1983, 1987; Stafford and Johnson 1986; Stafford and others 1986). The initial list was expanded to include species from 16 smaller studies in which insects were identified or collected (Allred 1970; Blom and others 1991; Bromenshenk 1987; Cieminski and Flake 1995; Clark and Blom 1988, 1991, 1992, 1999; Johnson and Stafford 1986; Merickel and Clark 1994; Stafford and Johnson 1986; Vieth 1983; Wenninger 2001; Winter 1984; Youtie 1986; Youtie and others 1987). A resident collection of INL voucher specimens for 466 species, primarily collected and identified by M. P. Stafford, W. H. Clark, and P.E. Blom, was also incorporated into the list. All insects identified to family were included. Although over 100 species of other arthropods, including ticks, mites, spiders, solpugids, and scorpions, have been collected and identified at the INL (Allred 1968b, 1969, 1970, 1973; Allred and Muma 1971; Karr and Kimberling 2003; Wenninger 2001), only insect species were compiled for this list. Ordinal names and placement of families and genera are according to Borror and others (1992).

To minimize the potential for double counting, I included only the maximum number of unidentified species within the same family or genus cited in any single reference. The number of unidentified species was then reduced for each unique species within the same family or genus cited in additional references. Thus, hundreds of unidentified specimens were excluded from this inventory and remain to be examined further.

Taxonomic authority was compiled from original documentation, voucher specimens, and other local and regional species lists (Haws and others 1988; Horning and Barr 1970). The INL list was not reviewed for synonymy or misidentifications. Discrepancies in spelling between INL studies and other authorities were reconciled to Horning and Barr (1970) where possible. Otherwise, spellings of Haws and others (1988) or Arnett (2000) were adopted.

Results and Discussion

A list of over 1,240 insect species from 17 orders, representing 747 genera in 212 families, was compiled from 22 studies conducted at the INL (table 1). Insect sampling has been widely distributed across the INL (fig.1), but most studies were of short duration and focused on associations

Table 1—List of documented insect species at the Idaho National Engineering and Environmental Laboratory.

ORDER[a] / **Family[a]** / Scientific name[b]
COLLEMBOLA
Onychiuridae
spp. undetermined
Entomobryidae
spp. undetermined
Isotomidae
unident. sp. #1
Sminthuridae
unident. sp. #1
EPHEMEROPTERA
Baetidae
spp. undetermined
Caenidae
spp. undetermined
ISOPTERA
Kalotermitidae
unident. sp. #1
ODONATA
Aeshnidae
unident. sp. #1
Libellulidae
unident. sp. #1
Coenagrionidae
spp. undetermined
PSOCOPTERA
Liposcellidae
unident. sp. #1
PHTHIRAPTERA
Enderleinellidae
Enderleinellus sp.
Hoplopleuridae
Hoplopleura acanthopus
Hoplopleura arboricola
Hoplopleura erratica
Hoplopleura hesperomydis (Osborn)
Hoplopleura minimus
Polyplacidae
Fahrenholzia pinnata
Fahrenholzia sp. #2
Haemodipus setoni
Neohaematopinus inornatus Kellogg
Neohaematopinus laeviusculus
Neohaematopinus marmota
Neohaematopinus pacificus
Neohaematopinus sp. #5
Polyplax auricularis Kellogg and Ferris
Polyplax spinulosa Burmeister
Polyplax sp. #3
Trichodectidae
Geomydoecus sp.
Neotrichodectes interruptofasciatus
Mallophaga
spp. undetermined
ORTHOPTERA
Acrididae
Arphia pseudonictana Thomas
Aulocara elliotti Thomas
Cratypedes lateritius
Hesperotettix viridis Thomas
Melanoplus sanguinipes (Fabricius)
Trimerotropis gracilis
unident. spp. #1, #2
Tettigoniidae
unident. sp. #1
Gryllacrididae
Ceuthophilus maculatus
Gryllidae
Oecanthus sp.
unident. spp. #1, #2
HEMIPTERA
Enicocephalidae
unident. sp. #1
Corixidae
Cenocorixa wileyae
Sigara alternata
Notonectidae
spp. undetermined
Tingidae
Aclypta cooleyi
unident. sp. #2
Miridae
Atomoscelis modestus VanDuzee
Chlamydatus artemisiae
Chlamydatus associatus (Uhler)

(con.)

Table 1—(Cont.)

ORDER **Family** Scientific name
Chlamydatus obliquus (Uhler)
Chlamydatus sp. #4
Coquilletia insignis Uhler
Coquilletia sp. #2
Deraeocoris bakeri Knight
Deraeocoris brevis Uhler
Deraecoris schwartzi (Uhler)
Hadronema simplex Knight
Hesperocapsus davisi
Illnacorella argentata Knight
Irbisia pacificus (Uhler)
Labopidea sericata (Uhler)
Labops utahensis Slater
Litomeris debilis (Uhler)
Lopidea sp.
Lygus desertinus Knight
Lygus elisus VanDuzee
Lygus hesperus Knight
Lygus sp. #4
Melanotrichus albocostatus
Orectoderus arcuatus
Orthotylus coagulatus (Uhler)
Parthenicus sp.
Phyllopidea hirta
Phyllopidea picta Uhler
Phytocoris sp.
Pilophorus sp.
Plagiognathus sp.
Polymerus diffusus Knight
Psallus pilosulus Uhler
Psallus sp.
Slaterocoris pilosipes
Slaterocoris utahensis
Slaterocoris sp. #3
Stenodema laevigatum (L.)
Stenodema pilosipes
Stenodema virens (L.)
Stenodema vicinum
Thyrillus pacificus (Uhler)
Trigonotylus ruficornis (Geoffroy)
Nabidae
Nabis alternatus (Parshley)
Reduviolus alternatus
Anthocoridae
Orius tristicolor White
Reduviidae
Sinea diadema (F.)
Zelus tetracanthus
unident. sp. #1
Piesmatidae
Piesma sp.
Phymatidae
unident. sp. #1
Corimelaenidae
Corimelaena virilis M. and Mc.
Lygaeidae
Blissus sp.
Emblethis vicarius
Geocoris pallens Stal
Leptoterna sp.
Lygaeus kalmii Stal
Malezonotus sp.
Megalonotus sabicula
Nysius ericae (Schilling) Auctorum
Nysius niger
Nysius raphanus Horvath
Ortholomus scolopax (Say)
Sisamnes claviger Coreidae
Chelinidae vittiger
Rhopalidae
Arhyussus sp.
Corizus punctiventris Dallas
Corizus scutatus (Stal)
Harmostes reflexulus (Say)
Leptocoris trivittatus (Say)
Liorhyssus hyalinus (F.)
Cydnidae
unident. sp. #1
Scutelleridae
Vanduzeina balli
unident. sp. #1
Pentatomidae
Aelia americana Dallas
Chlorochroa sayi Stal
Codophila remota Horvath
Prionosoma podopioides
Rhytidilomia uhleri Stal
HOMOPTERA
Cicadidae
Okanagana annulata Davis
Okanagana bella Davis[c]
Okanagana luteobasalis Davis
Platypedia putnami lutea Davis
Membracidae
Campylenchia latipes (Say)
Tortistillus wickhami VanDuzee
Aetalionidae
Aetalion sp.
Cercopidae
Clastoptera brunnea Bale
Clastoptera delicata Uhler
Neophilaenus lineatus (L.)
Philaronia sp.
Cicadellidae
Aceratagallia poudris (Stal)
Aceratagallia sp. #2
Athysanella spp. #1, #2
Balclutha sp.
Ballana hebea
Ballana sp. #2
Calladonus montanus VanDuzee
Carsonus aridus
Ceratagallia artemisia Oman
Chlorotettix unicolor Fitch
Circulifer tenellus Baker
Comellus sp.
Dikraneura carneola (Stal)
Empoasca alboneura
Empoasca aspersa G. & B.
Empoasca nigra G. & B.
Exitianus exitiosus
Exitianus sp.
Hecalus viridis (Uhler)

(con.)

Table 1—(Cont.)

ORDER **Family** Scientific name
Idiocerus sp.
Idiodonus geminatus
Norvellina vermiculata
Parabolocratus viridis Uhler
Texananus sp.
Xerophloea viridis (Uhler)
Delphacidae
Eurysa obesa Beamer
unident. spp. #1 to #3
Fulgoridae
Delphacodes pellucida (F.)
Delphacodes campestris VanDuzee
unident. sp. #1
Psillidae
Aphalara artemesiae Forster
Aphalara calthae (L.)
Aphalara loca Calderwood
Aphalara minutissima Crawford
Calophya triozoma Schwarz
Aphididae
Aphis gregalis Knowlton
Aphis ornata (Gillette & Palmer)
Brachycaudus helichrysi (Kaltenbach)
Durocapillata utahensis Knowlton
Forda marginata (Koch)
Microsiphoniella acophorum (Knowlton & Smith)
Myzus persicae (Sulzer)
Obtusicauda artemisicola
Pleotrichophorus pycnorhysus (Knowlton & Smith)
Pleotrichophorus utensus (Pack & Knowlton)
Uroleucon escalanti Knowlton
Zyxaphis canae (Williams)
Zyxaphis filifoliae Gillette & Palmer
Phylloxeridae
unident. sp. #1
Margarodidae
unident. sp. #1
Coccidae
unident sp. #1
Pseudococcidae
Phenacoccus sp.
THYSANOPTERA
Aeolothripidae
Aeolothrips auricestus Treherne
Aeolothrips fasciatus
Thripidae
Aptinothrips rufus (Gmelin)
Frankliniella occidentalis (Pergande)
Sericothrips sp.
Thrips tabaci
Oelolothripidae
Oelolothrips sp.
Phlaeothripidae
Leptothrips mali (Fitch)
unident. sp. #1
NEUROPTERA
Coniopterygidae
unident. sp. #1
Hemerobiidae
Hemerobius sp.
Kimminsia coloradensis (Banks)
Micromus variolus (Hagen)
Chrysopidae
Chrysopa coloradensis Banks
Chrysopa nigricornis
Eremochrysis punctinervis MacLachlan
Myrmeleontidae
Myrmeleon sp.
COLEOPTERA
Cicindelidae
Cicindela decemnotata Say
Cicindela purpurea
Carabidae
Agonum balesi Gray
Agonum placidum (Say)
Amara apricaria Paykull
Amara farcata LeConte
Amara impuncticollis Say
Amara laticollis LeConte
Amara littoralis
Amara musculus Say
Amara quenseli Schönherr
Apristus sp.
Axinopalpus biplagiatus (Dejean)
Bembidion immaculosum Hatch
Bembidion nebraskense LeConte
Bembidion obscurellum Motschulsky
Bembidion rupicola Kirby
Bembidion timidum LeConte
Bradycellus congener
Calleida viridis
Calosoma luxatum Say
Clivina fossor
Cymindis planipennis LeConte
Dicheirus piceus Menetries
Harpalus amputatus Say
Harpalus basilaris Kirby
Harpalus fraternus LeConte
Harpalus sp.
Lebia vittata (Fabricius)
Microlestes nigrinus (Mannerheim)
Philophuga viridis Dejean
Piosoma setosa LeConte
Pseudomorpha behrensi Horn
Pterostichus sp.
Haliplidae
spp. undetermined
Dytiscidae
Laccophilus decipiens LeConte
Gyrinidae
spp. undetermined
Ptiliidae
spp. undetermined
Leiodidae
Hydnobius sp.
Letodes grassa
Ptomophagus californicus (LeConte)
Silphidae
Nicrophorus hecate Bland
Nicrophorus guttulus Motschulsky
Staphylinidae
Acratona sp
Aleochara sp.

(con.)

Table 1—(Cont.)

ORDER Family Scientific name
Aleocharinae spp. #1, #2
Anotylus sp.
Astenus longiusculus
Bledius strenuus Casey
Bryoporus testaceus
Oxypoda sp.
Philonthus concinnus (Gravenhorst)
Philonthus cruentatus (Gmelin)
Platystethus americanus (Erichson)
Quedius sp.
Tachinus angustatus Horn
Tachyporus canadensis
Pselaphidae
Pilopius sp.
Eucnemidae
Analestes sp.
Hydrophilidae
Berosus fraternus LeConte
Berosus styliferus Horn
Cercyon quisquilius (L.)
Helophorus sp.
Sphaeridium scarabaeoides (L.)
Histeridae
Psiloscelis corrosa Casey
Saprinus lanei
Saprinus lugens Erichson
Saprinus oregonensis LeConte
Xerosaprinus acilinea (Marseul)
Xerosaprinus lubricus (LeConte)
Eucinetidae
unident. sp. #1
Scarabaeidae
Aphodius denticulatus Haldeman
Aphodius distinctus Muller
Aphodius fossor (L.)
Aphodius granarius (L.)
Aphodius hirsutus Brown
Aphodius militaris (?) LeConte
Aphodius vittatus Say
Bolbocerus obesus
Boreocanthon simplex (LeConte)
Cremastocheilus crinitus bifoviatus Van Dyke
Dichelonyx truncata LeConte
Dichelonyx sp. #2
Diplotaxis brevicollis
Diplotaxis obscura LeConte
Diplotaxis subangulata LeConte
Diplotaxis tenebrosa Fall
Glaresis canadensis Brown
Glaresis clypeata
Ochodaeus simplex LeConte
Paracotalpa granicollis (Haldeman)
Phyllophaga sociata
Serica anthracina LeConte
Serica barri Dawson
Trox sp.
Buprestidae
Acmaeodera immaculata Horn
Agrilus politus Say
Agrilus pubifrons Fisher
Agrilus walsinghami Crotch
Anthaxia retifer LeConte
Chrysobothris deleta LeConte
Chrysobothris horningi Barr
Chrysobothris idahoensis Barr
Chrysobothris texana LeConte
Heteroceridae
Lanternarius brunneus (Melsheimer)
Nannularia brunneata (Knull)
Elmidae
spp. undetermined
Elateridae
Aeolus dorsalis
Aeolus mellilus
Agriotella fusca Lane
Ampedus ursinus (Van Dyke)
Anchastus cinereipennis (Eschscholtz)
Cardiophorus spp. #1 to #5
Cardiophorus tumidicollis
Ctenicera noxia (Hyslop)
Ctenicera pruinina Horn
Ctenicera semivittata (Say)
Horistonotus pilosus Lanchester
Hypolithus bicolor (Eschscholtz)
Limonius (?) sp.
Cantharidae
Malthodes spp.
Dermestidae
Dermestes marmoratus Say
Bostrichidae
unident. sp. #1
Cleridae
Enoclerus acerbus Wolcott
Enoclerus barri Knull
Phyllobanus sp.
Trichodes ornatus Say
Melyridae
Amecocerus spp.
Attalus glabrellus Fall
Attalus morulus smithi Hopping
Attalus oregonensis Horn
Collops bipunctatus (Say)
Collops bridgeri Tanner
Collops hirtellus LeConte
Collops punctulatus LeConte
Dasytellus sp.
Hoppingiana nitida Hatch
Neodasytes testaceous
Trichochrous paisleyi
Dasytes cruralis
Dasytellus nigricornis (?) Bland
Nitidulidae
Brachypterolous pulicarius (L.)
Carpophilus pallipennis (Say)
Cryptophagidae
Caenoscelis ferruginea
Cryptophagus cellaris
Phalacridae
Olibrus rufipes LeConte
Phalacrus penicilatus Say
Coccinellidae
Brachyacantha dentipes socialis Casey
Brachyacantha ursina uteella Casey

(con.)

Table 1—(Cont.)

ORDER **Family** Scientific name
Brumus serptentrionis Weise
Coccinella difficilis Crotch
Coccinella guttata Bennett
Coccinella septempunctata
Coccinella novemnotata degener Casey
Coccinella prolongata Crotch
Coccinella transversoguttata richardsoni Brown
Hippodamia apicalis Casey
Hippodamia convergens Guerin
Hippodamia glacialis lecontei Mulsant
Hippodamia quinquesignata Kirby
Hippodamia tredecimpunctata tibialis (Say)
Hyperaspidius hercules Belicek
Hyperaspidius vittigera LeConte
Hyperaspis lateralis montanica Casey
Hyperaspis postica LeConte
Nephus ornatus
Nephus sordidus (Horn)
Psyllophora vigintimaculata (Say)
Selvadius nunemacheri
Selvadius sp. #2
Scymnus ardelio Horn
Scymnus caurinus Horn
Scymnus marginicollis Mannerheim
Scymnus postpictus Casey
Latridiidae
Melanopthalma americana Mannerheim
Melanopthalma sp. #2
Melandryidae
Physicus sp.
unident. sp. #1
Mordellidae
Mordella atrata Melsheimer
Mordellistena aspersa (Melsheimer)
Mordellistena idahoensis Ray
Mordellistena sericans Fall
Tenebrionidae
Alaudes singularis Horn
Araeoschizus airmeti Tanner
Blapstinus barri Boddy
Blapstinus discolor
Blapstinus substriatus Champion
Coelocnemis punctatus LeConte
Coniontis obesa LeConte
Coniontis ovalis (Say)
Coniontis setosa Casey
Eleodes cordata Eschscholtz
Eleodes elongata
Eleodes extricata cognata Haldeman
Eleodes granulata
Eleodes hispilabris connexa LeConte
Eleodes humeralis
Eleodes nigrina LeConte
Eleodes novoverrucula Boddy
Eleodes obscura Say
Eleodes pilosa Horn
Eleodes rileyi Casey
Embaphion elongatum Horn
Eusattus muricatus
Helops californicus Mannerheim
Helops convexulus LeConte
Helops opacus LeConte
Melanstrus ater (LeConte)
Oxygonodera hispidula (Horn)
Sphaeriontis muricata (Casey)
Alleculidae
Mycetochara procera Casey
Meloidae
Epicauta immerita Walker
Epicauta normalis Werner
Epicauta piceiventris Maydell
Gnathium eremicola Macswain
Lytta vulnerata cooperi LeConte
Nemognatha lutea LeConte
Nemognatha scutellaris
Linsleya sphaericollis (Say)
Oedemeridae
Oxacis bicolor (LeConte)
Anthicidae
Anthicus cervinus LaFerte
Anthicus formicarius LaFerte
Anthicus hastatus Casey
Anthicus nanus
Notoxus serratus LeConte
Notoxus robustus
Notoxus sp. #2
Cerambycidae
Centrodera nevadica nevadica LeConte
Cortodera barri Linsley & Chemsak
Crossidius ater LeConte
Crossidius coralinus LeConte
Crossidius hirtipes allgewahri LeConte
Crossidius punctatus LeConte
Judolia gaurotoides Casey
Mecas bicallosa Martin
Megacheuma brevipennis (LeConte)
Megasemum asperum (LeConte)
Micas bicallosa
Prionus californicus Motschulsky
Bruchidae
Acanthoscelides pauperculus (LeConte)
Acanthoscelides pudis Fall
Acanthoscelides sp. #3
Chrysomelidae
Altica plicipennis
Anisostena californica Van Dyke
Brachycoryna montana (Horn)
Chaetocnema sp.
Crepidodera nana (Say)
Cryptocephalus spurcus LeConte
Dibolia borealis Chevrolat
Disonycha latifrons Shaeffer
Exema conspersa (Mannerheim)
Glyptina atriventris Horn
Glyptoscelis artemisiae Blake
Longistarsis oregonensis
Monoxia consputa LeConte
Monoxia pallida Blake
Monoxia puberula Blake
Pachybrachis caelatus LeConte
Pachybrachis jacobyi Bowditch
Phyllotreta albionica LeConte
Phyllotreta oregonensis

(con.)

Table 1—(Cont.)

ORDER
Family
Scientific name

Pseudoluporus longulus LeConte
Psylliodes punctulata
Pyrrhalta luteola Muller
Pyrrhalta nymphaeae (L.)
Saxinis saucia LeConte
Scelolyperus nigrovirescens (Fall)
Stenopodius flavidus
Stenopodius vanduzeei Blaisdell
Systena blanda Melsheimer
Trirhabda nitidicollis LeConte
Trirhabda sp. #2
Curculionidae
Acmaegenius granicollis Van Dyke
Anthonomus tenuis Fall
Anthonomus squamosus
Apion sordidum Smith
Brachyrhinus ovatus (L.)
Cercopedeis artemisiae (Pierce)
Ceutorhinchus adductus Dietz
Ceutorhinchus bakeri Hatch
Ceutorhinchus disturbans Dietz
Ceutorhinchus sp. #4
Cleonidius poricollis
Cleonus kirbyi Casey
Cleonus quadrilineatus (Chevrolat)
Cosmobarius americana Casey
Dinocleus denticollis Casey
Dyslobus alternatus Horn
Epimechus mimicus Dietz
Gyrotus sinuatus Hatch
Miloderoides maculatus Van Dyke
Myrmex vittatus (Horn)
Ophryastes latirostris LeConte
Sitona hispidula (Fabricius)
Sitona lineata
Smicronyx abnormis
Sphenophorus gentilis LeConte
Tostates cinerascens
Trachyphloeni sp.
Tychius tectus LeConte
Tychius mixtus (?) Hatch
Anthribidae
Trigonorhinus sp.
Anobiidae
Ptinis villiger
Xyletynus fucatus LeConte

SIPHONAPTERA

Ctenophthalmidae
Rectofrontia fraterna
Hystrichopsyllidae
Anomiopsyllus amphibolus
Callistopsyllus terinus
Catallagia decipiens
Epiledia stanfordi
Epiledia wenmanni
Hystrichopsylla occidentalis
Megarthroglossus divisus
Meringis hubbardi
Meringis parkeri
Phalacropsylla allos
Phalacropsylla paradisea
Rhadinopsylla sectilis
Stenistomera alpina
Stenistomera macrodactyla
Ceratophyllidae
Foxella ignota
Malaraeus bitterrootensis
Malaraeus euphorbi
Malaraeus telchinum
Megabothris abantis
Megabothris obscurus
Monopsyllus eumolpi
Monosyllus exilis
Monopsyllus wagneri
Opisocrostis labis
Opisocrostis tuberculatus
Opisodasys keeni
Orchopeas leucopus
Orchopeas sexdentatus
Thrassis bacchi
Thrassis francisi
Thrassis howelli
Thrassis pandorae
Leptopsyllidae
Amphipsylla siberica
Odontopsyllus dentatus
Peromyscopsylla hesperomys
Pulicidae
Cediopsylla inaequalis
Pulex irritans

DIPTERA

Tipulidae
spp. undetermined
Bibionidae
Bibio albipennis
Bibiodes sp.
Mycetophilidae
Bodetina (?) sp.
Fungivora sp.
Megalopelmna (?) sp.
unident. spp. #1, #2
Sciaridae
unident. spp. #1 to #4
Cecidomyidae
Rhopalomyia sp.
unident. spp. #1 to #4
Psychodoidea
spp. undetermined
Scatopsidae
Scatopse fusipes Meigan
Dixidae
unident. sp. #1
Cuculidae
Aedes dorsalis Meigan
Simuliidae
Cnephia munus D. & S.
Simulium bivittatum Malloch
Simulium venator D. & S.
Simulium venustum (?) Say
Simulium vittatum Zetterstedt
Simulium spp. #5, #6
Ceratopogonidae
Culicoides sp.

(con.)

Table 1—(Cont.)

ORDER
Family
Scientific name

Dasyhelea spp. #1, #2
Forcipomyia brevipennis Macquaert
Forcipomyia sp. #2
Chironomidae
Chironomus spp. #1 to #3
Stratiomyiidae
Nemotelus canadensis Loew
Therevidae
Ozodiceromya sp.
Psilocephala spp. #1 to #3
Thereva pseudoculata Cole
Thereva semitaria
Scenopinidae
Scenopinus sp.
Asilidae
Asilus occidentalis Hine
Asilus mesae
Efferia benedicta (Bromley)
Efferia subcuprea (Schaffer)
Heteropogon senilis Bigot
Holopogon seniculus Loew
Lasiopogon sp.
Leptogaster fornicata Martin
Megaphorus martinorum
Ospriocerus abdominalis (Say)
Stenopogon inquinatus Loew
Stenopogon neglectus
Bombyliidae
Aphoebantus mormon Melander
Apolysis arenicola
Exoprosopa caliptera (Say)
Geron sp.
Lepidanthrax inauratus (Coquillet)
Lordotus apicalis (Coquillet)
Mythicomyia armata
Mythicomyia atra Cresson
Mythicomyia rileyi Coquillet
Mythicomyia spp. #4 to #9
Oligodranes acrostichalis Melander
Oligodranes quinquenotatus
Phthirla sulphurea
Prorates arctos
Prorates claripennis Melander
Thyridantrax andrewsi
Toxophora virgata Osten Sacken
Toxophora sp.
Villa molitor Loew
Empididae
Drapetis spp. #1 to #3
Platypalpus sp.
Phoridae
Megaselia spp. #1 to #4
Syrphidae
Eupeodes volucris Osten Sacken
Scaeva pyrasti (L.)
Sphaerophoria philanthus
Pipunculidae
Tomosvaryella sp.
Conopidae
Physocephala texana (Willinston)
Thecophora propinqua (Adams)
Zodion fulvifrons Say

ORDER
Family
Scientific name

Micropezidae
unident. spp. #1, #2
Psilidae
Psila dimidiata Loew
Otitidae
Euxesta (?) sp.
Oedopa capito Loew
Physiphora demandata (F.)
Tritoxa cuneata
Tritoxa pollinosa Cole
Tephritidae
Aciurina ferruginea Doane
Aciurina luteana Cresson
Aciurina trixa Cresson
Aciurina spp. #4, #5
Euaresta tapetis Coquillet
Eutreta diana
Eutreta oregona Curran
Neaspilota sp.
Neotephrites finalis (Loew)
Oxyna palpalis Coquillet
Paroxyna clathrata (Loew)
Paroxyna corpulenta Cresson
Paroxyna minima Doane
Paroxyna steyskali Novatny
Procecidochares minuta Snow
Procecidochares sp. #2
Tephritis araneosa (Coquillet)
Trupanea bisetosa Coquillet
Trupanea jonesi Curran
Trupanea nigricornis Coquillet
Milichiidae
Leptometopa halteralis Coquillet
Madiza glabra (F.)
Neophyllomyza quadricornis Melander
Pholeomyia indecora Loew
Dryomyzidae
unident. sp. #1
Sciomyzidae
unident. spp. #1, #2
Sepsidae
Saltella scutellaris Fallen
Saltella sp. #2
Sepsis biflexuosa
Sepsis neocynipsea Melander & Spuler
Sepsis punctum F.
Lauxaniidae
Camptoprosopella borealis Shewell
Chamaemyiidae
Leucopis americana Malloch
Leucopis flavicornis Aldrich
Pseudodinea nitens Melander & Spuler
unident. spp. #1 to #3
Heleomyzidae
Heleomyza sp.
Pseudoleria sp.
Trixoscelidae
Trixoscelis sp.
Sphaeroceridae
Leptocera sp.

(con.)

Table 1—(Cont.)

ORDER Family Scientific name
Drosophilidae
unident. sp. #1
Ephydridae
Hydrellia griseola (Fallen)
Leptopsilota sp.
Mosillus bidentatus Cresson
Phylligria debilis Loew
Psilopa sp.
Chloropidae
Chlorops rubicundus Adams
Chlorops sordidellus
Conioscinella sp.
Goniopsita oophaga
Hippelates particeps Becker
Hippelates pusio Loew
Incertella sp.
Meromyza pratorum Meigan
Meromyza saltatrix (L.)
Neoneura flavifacies Collin
Neoneura polita Sabrosky
Olcella punctifrons Becker
Oscinella frit (L.)
Oscinella inserta Becker
Siphonella sp.
Thaumatomyia annulata (Walker)
Thaumatomyia appropinqua (Adams)
Thaumatomyia glabra (Meigan)
Anthomyiidae
Calythea micropteryx Thomson
Hydrophoria brunneifrons (Zetterstedt)
Hydrophoria divisa (Meigan)
Hylemyia spp. #1 to #3
Leucophora sp.
Pegomya sp.
Scatophaga stercoraria (L.)
Scatophaga sp. #2
Agromyzidae
Agromyza pusilla Meigan
Ceradontha dorsalis Loew
Melanagromyza sp.
Phytobia sp.
unident sp. #5
Scatophagidae
unident. sp. #1
Muscidae
Fannia sp.
Helina duplicata (Meigan)
Helina multisetosa
Helina troene Walker
Helina spp. #3, #4
Lasipcs septentrionalis Stein
Musca domestica L.
Muscina stabulans Fallen
Orthellia caesarion (Meigan)
Quadrolaria laetifica Robineau-Desvoidy
Schoenomyza dorsalis Loew
Calliphoridae
Calliphora lilae Walker
Phormia regina (Meigan)
Protophormia terrenovae (Macquaert)
Sarcophagidae
Hilarella hilarella (Zetterstedt)
Robineauella sp.
Sarcophaga spp. #1 to #3
Senotainia trilineata (Wulp)
Senotainia vigilans Allen
Sphenometopa tergata Coquillet
Taxigramma heteroneura (Meigan)
Cuterebridae
Cuterebra jellisoni
Tachinidae
Acemya tibialis Coquillet
Anthrycia cineria (Coquillet)
Anthrycia sp.
Bennettia compta (Fallen)
Blondelia (?) sp.
Catagoniops facialis (Coquillet)
Dinera grisescens (Fallen)
Euphorcera sp.
Exorista mella Walker
Gonia albagenae Morrison
Hyalomyia aldrichii Townsend
Lespesia archippivora (Riley)
Leucostoma simplex Fallen
Lydella radicis Townsend
Microchaetina valida Townsend
Norwickia latifacies Tothill
Norwickia latigena Tothill
Norwickia robinsoni
Paradidyma simulans Townsend
Paradidyma singularia Townsend
Patellea sp.
Peleteria malleola Bigot
Periscopsia cinerosa Coquillet
Periscepsia helymus (Walker)
Promasiphia (?) sp.
Siphosturmia maltana Reinhardt
Sitophaga sp.
Spallanzania sp.
Spathidexia reinhardti Arnaud
Stomatomyia parvipalpis Wulp
Voria ruralis Fallen
TRICHOPTERA
Leptoceridae
spp. undetermined
LEPIDOPTERA
Micropterigidae
unident. spp. #1, #2
Lyonetiidae
Bucculatrix seorsa
Bucculatrix tridenticola Braun
Coleophoridae
Coleophora sp.
Gelichiidae
Aroga websteri Clarke
Chionodes sp.
Tortricidae
Eucosma sp.
Phaneta salmicolorana
Phaneta setonana
Synnoma lynosyrana Walsingham
Plutellidae
Plutella maculipennis Curtis

(con.)

Table 1—(Cont.)

ORDER **Family** Scientific name
Pterophoridae
unident. sp. #1
Microlepidoptera spp. #1 to #9
Pyralidae
Loxostege commixtalis
Omnaptopteryx occellea Hanson
Hesperiidae
Hesperia harpalus (Edwards)
Hesperia juba (Scuder)
Papilionidae
Papilio sp.
Pieridae
Colias interior
Euchloe ausonides Lucas
Pieris beckerii
Pieris protodice Biusdyvak
Lycaenidae
Lycaena helloides Boisduval
Lycaena rubidus
Plebejus melissa (Edwards)
Nymphalidae
Euphydryas anicia (Doubleday)
Speyeria callippe nevadensis (Edwards)
Vanessa cardui (L.)
Satyridae
Cericyonis oetus oetus (Boisduval)
Saturniidae
Hemileuca hera hera Harris
Sphingidae
Hyles lineata (Fabricius)
Proserpinus clarkiae Boisduval
Arctiidae
Apantesis retilineata
Arctia caja (L.)
Leparctia sp.
Geometridae
Chlorosea sp.
Glaucina nephos
Pero modesto
Plataea linearia
Procherodes amplicineraria Pearson
Semiothisa nubiculata
Synaxis formosa
Noctuidae
Abagrotis nefascia Smith
Apamea occidens (Grote)
Aseptis characta Grote
Autographa californica Speyer
Copablepharon canariana McDonald
Cucullia arizona
Dicestra crotchi
Drasteria mirifica Hy. Edwards
Euxoa auxiliaris Grote
Euxoa costata idahoensis Grote
Euxoa pluralis Grote
Faronta diffusa (Walker)
Heliothis belladonna
Lacinipolia sp.
Rhynchagrotis exertistigma Morrison
Synedoida sp.
HYMENOPTERA
Cephidae
Cephus cinctus Norton
Ceraphronidae
Ceraphron sp.
unident. spp. #1, #2
Braconidae
Agathis californica (?) Rhower
Agathis gibbosa Muesebeck
Apanteles sp. #2
Apanteles yakutatensis (Say)
Aphidius sp.
Bracon hyslopi (Viereck)
Bracon nuperus (?) Cresson
Bracon spp. #3 to #5
Chelonus spp. #1, #2
Chorebus sp.
Diaeretiella spp. #1, #2
Hormius sp.
Macrocentrus ancylivorus Rohwer
Macrocentrus sp. #2
Meteorus leviventris Wesmail
Microplitis plutellae Muesebeck
Microctonus pusillae Muesebeck
Opius (?) sp.
Orgilus sp.
Rogas sp.
Tetrasphaeropyx sp.
Vipio (?) sp.
unident. spp. #1 to #8
Ichneumonidae
Amblyteles spp. #1, #2
Anomalon ejuncidum Say
Anomalon sp.
Banchus nubilus Townes
Campoplex sp.
Chorineaus sp.
Conoblasta sp.
Cryptus asymmetricus Pratt
Cryptus sp. #2
Diadegma plutellae (Viereck)
Diadegma sp. #1
Diplazon laetitorius (F.)
Exetastes sp.
Gelis sp.
Gnypetonoipla (?) sp.
Mesostenus gracilis Cresson
Ophion abnormalis Felt
Ophion purgatus Say
Pseudamblyteles superbus (Provander)
Pseudamblyteles kocheli Swezey
unident. spp. #1 to #8
Mymaridae
Gonatocerus sp.
unident. sp. #1
Trichogrammatidae
unident. sp. #1
Eulophidae
Achrysocharella sp.
Euderus sp.
Chrysocharis sp.

(con.)

Table 1—(Cont.)

ORDER **Family** Scientific name
Cirrospilus sp.
Diglyphus spp. #1, #2
Elachertus sp.
Entedon sp.
Necremnus duplicatus Gahan
Necremnus sp. #2
Pediobius utahensis (Crawford)
Tetrastichus coerulescens Ashmead
Tetrastichus spp. #2 to #10
Zagrammosoma nigrolineatum Crawford
Zagrammosoma sp. #2
unident. spp. #1 to #13
Elasmidae
Elasmus sp.
Aphelinidae
unident. sp. #1
Encyrtidae
Homalotylus (?) sp.
Oencyrtus sp.
unident. spp. #1 to #15
Eupelmidae
Eupelmus allynii French
Eupelmus sp. #2
Calosota sp.
unident. spp. #1 to #3
Torymidae
Torymus coloradensis
unident. spp. #1 to #11
Pteromalidae
Asaphes sp.
unident. spp. #1 to #37
Eutrichosomatidae
Eutrichosoma mirabile Ashmead
Perilampidae
Perilampus chrysopae Crawford
Perilampus hyalinus Say
Perilampus similis Crawford
Perilampus sp. #4
Eurytomidae
Eurytoma spp. #1 to #5
Harmolita spp. #1, #2
Rileya cecidomyiae Ashmead
Tetramesa elymophaga (Phillips)
Tetramesa sp. #2
unident. spp. #1 to #4
Chalcididae
Haltichella sp.
Spilochalcis albifrons Welsh
Spilochalcis ignoides (?) Kirby
Spilochalcis leptis Burks
Spilochalcis side Walker
unident. sp. #1
Eucoilidae
unident. sp. #1
Figitidae
Melanips coxalis
Trischiza sp.
Cynipidae
Periclistus sp.
Proctotrupidae
Proctotrupes florissantensis Kiefer
Diapriidae
unident. spp. #1, #2
Scelionidae
Gaeus sp.
Gryon sp.
Telenomus spp. #1 to #3
Trissolcus utahensis
Platygasteridae
Platygaster rohweri
Platygaster utahensis
Platygaster spp. #2, #3
Synopeas spp. #1 to #3
Inostemma sp.
unident. sp. #1
Chrysididae
Ceratochrysis perpulchra (Cresson)
Ceratochrysis trachyplenia R. Bohart
Chrysis canadensis Buysson
Chrysis coerulans (F.)
Chrysis coloradica
Chrysis dorsalis (?) Aaron
Chrysis vagabunda
Chrysis sp. #6
Chrysura densa (Cresson)
Cleptes purpuratus
Hedychridium carrilloi R. Bohart & Brumley
Omalus aeneus
Bethylidae
unident. spp. #1, #2
Dryinidae
unident. sp. #1
Sphecidae
Ammophila spp. #1 to #3
Ammoplanops (?) sp. #1
Ancistromma sp.
Astata bakeri Parker
Belomicrus sp.
Bembix amoena Handlirsch
Bembix spinolae Lepeletier
Cerceris minax Mickel
Cerceris nigrescans Simth
Dienoplus sp.
Diodontus spp. #1, #2
Dryudella immigrans William
Dryudella sp. #2
Ectemniis dilectus Cresson
Ectemniis sp. #2
Eucerceris sp.
Glenostictia megacera J. Parker
Gorytes sp.
Mimesa sp.
Miscophus (*Nitelopterus*) sp.
Nyson sp.
Oxybelus sp.
Philanthus multimaculatus Cameron
Podalonia spp. #1, #2
Prionyx canadensis (Provancher)
Solierella spp. #1 to #3
Sphex ichneumoneus (L.)
Steniolia elegans J. Parker
Stictella megacera Parker
Tachysphex irregularis

(con.)

Table 1—(Cont.)

<u>**ORDER**</u> **Family** Scientific name
Tachysphex tarsatus (Say)
Tachysphex williamsi
Melittidae
unident. sp. #1
Colletidae
Colletes dissoptus Timberlake
Colletes fulgidus Swank
Colletes lutzi Timberlake
Halictidae
Agapostemon texanus Cresson
Dialictus spp. #1, #2
Evylaeus sp.
Halictus farinosus Smith
Halictus ligatus
Halictus tripartitus Cockerell
Sphecodes arvensiformis (?) Cockerell
Sphecodes sp.
Andrenidae
Andrena prunorum Cockerell
Descurainia richardsoni
Perdita spp. #1, #2
Megachilidae
Anthidium emarginatum (Say)
Anthidium placitum Cresson
Anthidium utahense Swenk
Ashmeadiella gillettei Titus
Ashmeadiella opuntiae Cockerell
Dianthidium pudicum decorum Timberlake
Dioxys pomonae Cockerell
Hoplitis producta Michner
Megachile laurita Mitchell
Megachile onobrychidis Cockerell
Megachile parallela Smith
Osmia integra
Stelis sp.
Anthophoridae
Anthophora exigua Cresson
Anthophora ursina Cresson
Ceratina pacifica Cresson
Didadasia enavata Cresson
Epeolus minimus Robertson
Melissodes bimatris (?) LaBerge
Nomada articulata Smith
Nomada suavis Cresson
Synhalonia spp. #1, #2
Tetralonia fulvitarsis Cresson
Triepeolus helianthi (?) Robertson
Apidae
Apis mellifera (L.)
Bombus fervidus (F.)
Bombus huntii Greene
Tiphiidae
Brachycistis spp. #1, #2
Sapygidae
Sapyga pumila Cresson
Tiphia sp.
Mutillidae
Chyphotes sp.
Sphaerophthalma unicolor
Sphaerophthalma spp. #2 to #4
Scoliidae
Campsoscolia alcione Banks
Formicidae[d]
Camponotus hyatti Emery
Camponotus vicinus May
Camponotus sp. #2
Ephebomyrmex sp.
Formica ciliata Mayr
Formica cinerea canadensis Santschi
Formica fusca L.
Formica gynocrates Snelling and Buren
Formica haemorrhoidalis Emery
Formica hewitti Wheeler
Formica lasioides Emery
Formica laviceps Creighton
Formica manni Wheeler
Formica montana
Formica neogagates Emery
Formica obscuriventris
Formica obtusopilosa Emery
Formica oreas comptula Wheeler
Formica rufa (L.)
Formica subnuda
Formica subpolita Mayr
Formica whymperi
Formicoxenus diversipilosus
Formicoxenus hirticornis
Lasius alienus
Lasius crypticus Wilson
Leptothorax andrei
Leptothorax nevadensis Wheeler
Liometopum luctuosum W. M. Wheeler
Manica mutica
Monomorium minimum
Myrmecocystus mojave
Myrmecocystus testaceus Emery
Myrmica americana Weber
Myrmica lobicornis Emery
Pheidole californica
Pogonomyrmex occidentalis (Cresson)
Pogonomyrmex owyheei Cole
Pogonomyrmex salinus Olsen
Solenopsis molesta Say
Stenamma sp.
Tapinoma sessile Say
Veromessor lobognathus (Andrews)
Pompilidae
Ageniella spp. #1, #2 (?)
Anoplius insolens
Anoplius tenebrosus (Cresson)
Anoplius sp. #3
Aporinellus completus Banks
Aporinellus fasciatus (Smith)
Ceropales sp.
Episyron snowi (Viereck)
Evagetes padrinus (?) (Viereck)
Evagetes parvus
Evagetes sp. #3
Pompilus angularis (Banks)
Vespidae
Ancistrocerus spp. #1, #2
Euodynerus annulatus (Say)
Euodynerus sp. #2

(con.)

Table 1—(Cont.)

ORDER[a] / **Family**[a] / Scientific name[b]
Pterocheilus fasciatus Say
Pterocheilus pediculatus
Pterocheilus provancheri
Stenodynerus noticeps (?)
Stenodynerus sp. #2

[a] Order and Family names are according to Borror and others (1992).
[b] Spellings are according to Horning and Barr (1970), Haws and others (1988), and Arnett (2000). Authors are provided only in cases where they were available from literature cited in the reference list. Authors shown in parentheses indicate the generic name has changed since the species was originally identified (Borror and others 1992). Abbreviations: sp. = single species (Borror and others 1992), spp. = multiple species (Borror and others 1992), unident. = unidentified species or morphospecies, undetermined = specimens were not identified past family level, (?) = questionable identification noted by the researcher.
[c] Subsequently identified as *O. annulata*.
[d] Additional taxa and clarifications pending (Clark and Blom, in preparation).

with specific hostplants or particular insect groups. In most cases, only a few dozen species were collected and identified. The investigation by Bohart and Knowlton (1977), in which over 800 species were identified, constitutes the most extensive single inventory conducted at the INL, followed by the study by Karr and Kimberling (2003) and the multiple-year investigation conducted by Stafford (1983, 1987) and Stafford and Johnson (1986).

In comparison, a 3-year survey at nearby Craters of the Moon National Monument (CMNM) (Horning and Barr 1970) resulted in the identification of nearly 2,100 species, representing 248 families and 1,144 genera in 19 orders. Totals for the suborder Raphidoidea were reported separately in the summary of COM insect orders by Horning and Barr (1970); for the INL list, species Raphidoidea are included in totals reported for Neuroptera (see table 2). Although about 860 more insects were documented in the CMNM survey, only 157 of the 212 families, 396 of 747 genera, and 305 of 1,241 species identified at the INL have also been identified at CMNM. Insects in disturbed habitats at the INL have been investigated (Karr and Kimberling 2003; Wenninger 2001), but studies have been primarily in native sagebrush and grassland. While similar communities were included in the CMNM survey, the larger inventory likely reflects investigation of a wider variety of habitat types (Horning and Barr 1970) and greater nighttime collecting efforts.

Insect research at the INL has been focused on terrestrial species. However, a few aquatic insect families have been documented in and around industrial waste ponds (Cieminski and Flake 1995; Millard and others 1978). Additional aquatic insects have been collected from sections of the Big Lost River, but specimens have not been sorted and identified (R. C. Rope, personal communication).

In recent years, large wildfires at the INL have destroyed thousands of acres of sagebrush habitat, including several former research sites (fig. 1). The response of insects to fire has been studied at the INL (Stafford 1983, 1987; Winter 1994), but as the need for restoration of sagebrush habitats damaged by fire and other disturbance increases, a greater understanding of insect life cycles, population dynamics, and changes in species composition over time will be required. This baseline list of insects can be used to identify groups that have not been well characterized and will help focus further investigation of insect ecology and function in natural and restored sagebrush systems.

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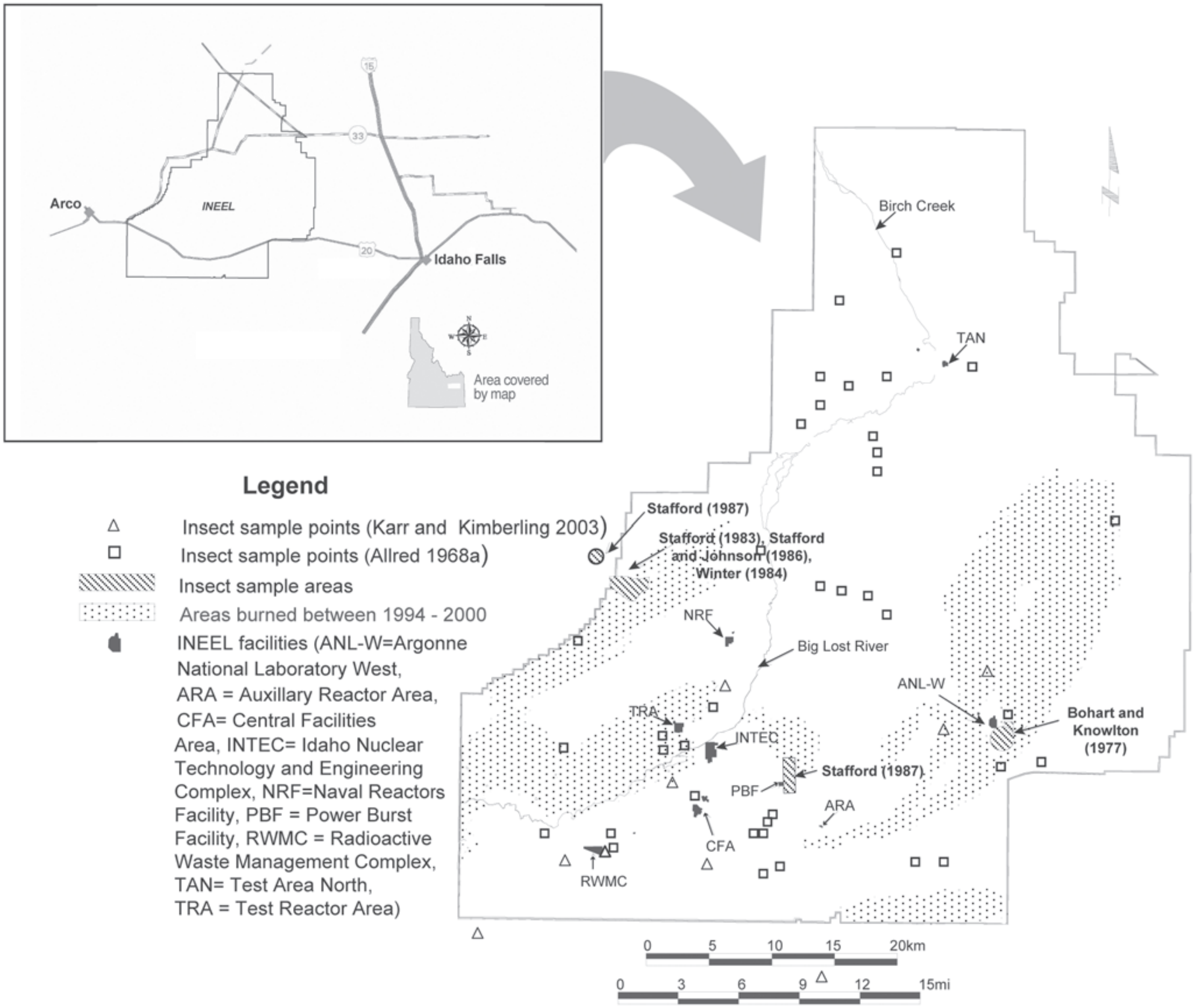

Figure 1—Sampling locations for major insect studies at the INL. NOTE: the INEEL is now the Idaho National Laboratory (INL); the TRA is now the Reactor Technology Complex (RTC); and the ANL-W is now the Materials and Fuels Complex (MFC).

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Table 2—Summary of insect inventories for the Idaho National Laboratory and Craters of the Moon National Monument.

Order	Families		Genera		Species	
	INL[a]	CMNM[b]	INL	CMNM	INL	CMNM
Collembola[c]	4	5	_	6	_	6
Thysanura	_	2	_	2	_	2
Ephemeroptera[c]	2	4	_	4	_	4
Odonata	3	4	3	6	3	10
Isoptera	1	1	1	1	1	1
Plecoptera	_	4	_	4	_	5
Dermaptera	_	1	_	1	_	1
Psocoptera	1	2	1	3	1	3
Phthiraptera	4	_	9	_	20	_
Orthoptera	4	4	8	14	11	23
Hemiptera	17	20	63	79	79	115
Homoptera	13	14	49	56	65	79
Thysanoptera	4	3	7	8	9	10
Neuroptera	5	5	9	14	10	24
Coleoptera	42	44	187	198	297	324
Strepsiptera	_	1	_	1	_	1
Siphonaptera	4	_	24	_	38	_
Diptera	46	50	159	286	238	521
Trichoptera[c]	1	6	_	10	_	12
Lepidoptera	18	35	49	140	66	218
Hymenoptera	43	43	178	311	403	705
Totals	212	248	747	1,144	1,241	2,064

[a] Idaho National Laboratory.
[b] Craters of the Moon National Monument (totals from Horning and Barr 1970).
[c] INL specimens were not identified past family level for these orders.

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