RESTORATION AND MANAGEMENT OF SAGEBRUSH/GRASS COMMUNITIES WORKSHOP

ELKO, NEVADA, NOVEMBER 4-8, 2002

SAGEBRUSH COMMUNITIES - DISTRIBUTION, IDENTIFICATION, AND RESOURCES

Gary Medlyn, "Soil/sagebrush correlation in Nevada"

Al Winward, "Identification of the principal sagebrush taxa within the Great Basin"

Sherel Goodrich, "Major sagebrush communities - distribution, areas of occurrence, and species composition"

Kent McAdoo, Sherman Swanson, Brad Schultz, "Habitat requirements of sagebrush-associated species and implications for management"

INFLUENCE OF MANAGEMENT PRACTICES, WEED INVASION, AND FIRE ON THE ECOLOGY OF SAGEBRUSH AND RELATED WOODLANDS

Rick Miller, "The role of fire across the sagebrush biome"

Robin Tausch, "Implications of previous and current management practices, fire, weed invasion, and climate on pinyon/juniper communities"

Steve Bunting, "Natural and prescribed fires in big sagebrush steppe - response of individual species and implication to burning practices"

Sherel Goodrich, "Comparative response of mountain and Wyoming big sagebrush communities to burning, post grazing management, seeding, weed invasion, and native restoration"

PRINCIPALS OF RESTORATION - SITE SELECTION

Brad Schultz, Sherm Swanson, "Using concepts behind state and transition models to improve the decision making process for restoration efforts on sagebrush rangelands"

Alan Sands, "Guidelines to manage sage grouse populations and habitats"

Gary Back, Berry Perryman, Kent McAdoo, "Application of sage grouse guidelines to sage grouse habitats in Nevada"

Michael Wisdom, Lowell Suring, Mary Rowland, Barbara Wales, Cara Wolff Meinke, Steve Knick, Linda Schueck, Richard Miller, Robin Tausch, "Regional assessment of threats to sagebrush habitats for species of conservation concern"

Mike Hess, "Historical use of Nevada rangelands and implications to mule deer habitats"

PLANT COMPETITION

Tom Monoco, "Biological and ecological features effecting plant competition and control of annual weeds, shrubs, and pinyon/juniper - methods to promote native recovery"

Jim Young, "Weed problems on Great Basin Rangelands"

Mike Pellant, "Cheatgrass: invasion, occurrence, biological/competitive features, and control measures"

Bob Wilson, "Chemical control measures to reduce understory weeds associated with sagebrush and pinyon/juniper woodlands"

Scott Walker, "Ecological response of sagebrush communities to different mechanical treatments"

INFLUENCES OF CLIMATE, PLANT BIOLOGY, AND ECOLOGICAL CONDITIONS ON SPECIES ESTABLISHMENT AND ADAPTATION

Stuart Hardegree, Gerald Flerchinger, Steven Van Vactor, "Variability in seedbed microclimate, prediction of seed-population response, and implications for emergency revegetation and restoration planning"

Stephen Monsen, "Ecotypic variability, seed features, and seedbed requirements of big sagebrush"

Stewart Sanderson, "Ecotypic variability in Fourwing saltbush (Atriplex canescens)

Stephen Monsen, "Ecological influence of planting introduced grasses and broadleaf herbs in sagebrush and related shrublands"

Richard Stevens, "Selecting adapted species, developing seeding combinations, seeding rates, and methods of treatment to facilitate establishment and recovery of native species"

Steve Knick, "What will we do with all the GIS maps? A spatial view of restoration in the Great Basin"

Biological and Ecological Features Effecting Plant Competition and Control of Annual Weeds, Shrubs, and Pinyon/Jjuniper – Methods to Promote Natural Recovery

Thomas A. Monaco

Competition is one of the most widely studied ecological interactions. However, it remains difficult to evaluate and quantify because of the numerous biological, edaphic, and environmental factors associated with plant community dynamics. Most frequently, competition is characterized by plants utilizing the same pool of limiting resources. Consequently, competitive ability is greater in species that can effectively compete for resources in a given environment. Because competitive ability depends on the environment, species differ greatly in functional traits associated with competitive ability. Competitive success may arise from one set of traits at an early stage of succession, but may depend on different traits at later stages of succession. Often, there are trade-offs associated with a species ability to compete successfully in a given environment. Identifying these trade-offs is critical to a complete understanding of competitive interactions associated with annual grass invasion and woody plant encroachment in semiarid rangelands.

Invasive annual grasses like cheatgrass and medusahead demonstrate traits that enable them to effectively compete with perennial plants. High growth rates (Arredondo et al. 1998, Lambers and Poorter 1992), even at low temperature allow these annual grasses to acquire nutrients while slower growing perennial species remain dormant. Numerous studies have also shown that annual grasses are more responsive than perennial species to transient pulses of soil nutrients which enable them to rapidly assimilate nutrients (i.e., Yoder and Caldwell 2002). However, if resources are not readily available to support high growth rates of the annual growth form, slower growing perennial species with slow leaf turnover rates and high nutrient retention in foliage may have a competitive advantage. My talk will discuss how shifts in nutrient availability modify competitive ability in annual and perennial species.

The co-existence of woody plants, i.e., shrubs and trees, with herbaceous vegetation creates an ideal situation to evaluate how trade-offs between competitive ability and stress tolerance determine dominance in shrublands. For example, juniper and piñon species are long-lived conifers characterized by slow growth rates, low nutrient demands, and low rates of leaf turnover. Consequently, leaf construction costs are higher and nutrients are retained longer in the leaves than in herbaceous vegetation (i.e., Archer 1995). The competitive ability of these woody species is predicted to be greater than herbaceous growth forms as resource availability decreases and sites become more xeric. I will discuss competitive interactions of herbaceous and woody plant species and how differences in functional traits may be associated with shifts in community dominance.

References:

Archer S (1995) Assessing and interpreting grass-woody plant dynamics, pp. 101-134 In: *The Ecology and Management of Grazing Systems* (Hodgson J, Illius AW, eds). CAB International, Wallingford, Oxon, UK.

Arredondo JT, TA Jones, DA Johnson (1998) Seedling growth of Intermountain perennial and weedy annual grasses. J Range Manage 51:584-589.

Lambers H, Poorter H (1992) Inherent variation in growth rate between higher plants: a search for physiological causes and ecological consequences. Adv Ecol Res 23:188-261.

Yoder C, Caldwell M (2002) Effects of perennial neighbors and nitrogen pulses on growth and nitrogen uptake by *Bromus tectorum*. Plant Ecol 158:77-84.

____

T. A. Monaco, Research Ecologist
USDA-ARS, Logan, UT
tmonaco@cc.usu.edu

NOTES:

Variability in Seedbed Microclimate, Prediction of Seed-Population Response, and Implications for Emergency Revegetation and Restoration Planning

Stuart P. Hardegree, Gerald N. Flerchinger and Steven S. Van Vactor

Millions of acres of rangeland in the western United States have been invaded by cheatgrass (*Bromus tectorum* L.), an introduced annual weed that proliferates after wildfire (Young et al., 1987; Young and Longland, 1996). A primary consideration in restoration of these rangelands is the selection of adapted plant materials that will establish and persist under weed competition (Roundy and Call, 1988; Call and Roundy, 1991). Rangeland seeding guides typically list a variety of plant species, ranked for general adaptation under different soil types and climatic regimes (Jensen et al., 2001). Unfortunately, the microclimatic conditions necessary for plant establishment are much more restrictive than those required for long-term maintenance of mature plant communities. Seedbed microclimate in the Intermountain west is highly variable over space and time (Pierson and Wight, 1991). In addition to variability in precipitation, soil moisture and temperature are affected by solar radiation, wind, air temperature, relative humidity and local vegetation and soil properties (Flerchinger and Pierson, 1997). Enhanced understanding of natural variability in seedbed microclimate, and it's impact on plant establishment and growth, may facilitate development of more effective revegetation strategies on disturbed, Intermountain rangelands. The focus of the current study was to estimate hydrothermal-germination response to simulated planting conditions for every day of a 38-year test period. Specific objectives were to quantify the impact of microclimatic variability on potential germination response in the field; to develop a probabilistic germination-response index for comparison of cheatgrass and native bunchgrass seedlots; and to provide perspective for setting establishment goals in arid and semi-arid rangeland ecosystems that have been invaded by annual weedy species.

In this study, two seedlots each of bluebunch wheatgrass [*Pseudoroegneria spicata* (Pursh) Löve], big squirreltail [*Elymus multisetus* (J.G. Smith) M.E. Jones], and cheatgrass (*Bromus tectorum* L.) were germinated under 12 constant-temperature regimes and 11 water potentials to parameterize a hydrothermal germination response model. The Simultaneous Heat and Water Model (SHAW; Flerchinger and Saxton 1989a,b) was calibrated with weather data and soil measurements and used to simulate hourly temperature and water potential at seeding depth at a field site in southern Idaho for the period between October 1, 1961 and September 30, 1999. These models were used together to estimate seed germination time for each seedlot had they been planted on any single day during the 38-year test period. These data and analyses were used for two purposes: to develop more ecologically relevant indices for seedlot comparison; and to evaluate potential modeling applications for improvement of rangeland revegetation and restoration efforts.

Seed germination under temperature and water stress has received a great deal of attention in the rangeland literature (Wester, 1991). Most of these studies evaluate only a limited number of environmental conditions with subsequent analyses constrained to simple treatment comparisons of germination-rate and total-germination indices (Scott et al., 1984; Brown and Mayer, 1988). Since Garcia-Huidobro et al. (1982) and Gummerson (1986), thermal and hydrothermal modeling have become the primary methods by which potential germination response to temperature and moisture have been assessed for agricultural plant species. These models generate coefficients that integrate germination response over a wide range of potential field conditions (Arnold, 1959; Garcia-Huidobro et al., 1982; Covell et al., 1986; Hardegree et al., 1999). Model coefficients can then be used as a basis for seedlot comparison and ranking (Covell et al., 1986; Ellis et al., 1986; Jordan and Haferkamp, 1989; Fidanza et al., 1996; Holshouser et al., 1996). Thermal and hydrothermal-time concepts have been applied to relatively few rangeland species, and mostly to develop indices for seedlot comparison (Jordan and Haferkamp, 1989; Allen et al., 2000; Meyer et al., 2000). These models, however, can also be used to simulate potential field response under alternative conditions of seedbed microclimate (Hardegree and Van Vactor, 2000; Hardegree et al., 2002). These simulations can provided a broader picture of potential environmental response than can be achieved by simple treatment comparisons, or analysis of model coefficients. In this study, two indices were developed to integrate hydrothermal and microenvironmental aspects of seed germination for purposes of seedlot comparison. The first index is based on estimation of mean-daily germination rates as a function of planting date. Summation of hourly or daily germination rates can be used as a second index to integrate potential germination response for comparison of different seedlots, seasons or years.

Previous studies have hypothesized that the ability to germinate rapidly at low temperature contributes the success of cheatgrass (Harris and Wilson, 1970; Wilson et al., 1974). This hypothesis has been supported by laboratory evidence demonstrating statistically significant differences in relative germination rate in a fixed, low-temperature test environment (Hardegree, 1994a, b). The ecological significance of these rate differences is more difficult to assess from standard germination indices at a few fixed test temperatures. This study showed that the relative germination advantage of cheatgrass persists across a wide number of historically simulated, field-variable conditions of seedbed microclimate. The ecological advantage of rapid germination rate *per se*, however, may not be as significant as the relative effect of seed numbers in the field. Cheatgrass seed densities after wildfire may be on the order of 10,000 seeds m^{-2} (Humphrey and Schupp, 2001).

The second purpose of this study was to determine whether microclimatic knowledge and information could be used to improve revegetation and restoration success rates. In order to take advantage of this knowledge, however, it may be necessary to separate short-term soil stabilization, and long-term biodiversity objectives. The highest priority for post-fire management of Intermountain rangelands has been the prevention of soil loss in areas that have been denuded of vegetation. The bulk of revegetation funds, therefore, have historically been expended in the year following wildfire. On average, shorter-term soil stabilization goals may be better met by selection of native or non-native plant materials based on ease of establishment. Incorporation of seedbed modeling and knowledge of weather variability my be of greater benefit to longer-term objectives for restoration of native plant communities in areas currently dominated by cheatgrass. In one scenario, restoration options could remain open until mid or late-winter. A high level of soil moisture in the winter may be of sufficient magnitude to increase the probability of favorable establishment conditions at seeding depth later in the spring. A second scenario would utilize medium and long-term weather forecasting to predict the probability of favorable seedbed conditions that would warrant expenditure of restoration funds (Allen and Meyer, 1998). Either of these scenarios would require re-examination of current strategies used to plan and fund rangeland revegetation and restoration efforts.

References:

Allen, P.S. and Meyer, S.E., 1998. Ecological aspects of seed dormancy loss. Seed Sci. Res., 8: 183-191.

Allen, P.S., Meyer, S.E., and Khan, M.A., 2000. Hydrothermal time as a tool in comparative germination studies. In: M. Black, K.J. Bradford and J. Vazquez-Ramos (editors), Seed Biology: Advances and Applications, CAB International, Wallingford, Oxon, pp. 401-410.

Arnold, C.Y., 1959. The determination and significance of the base temperature in a linear heat unit system. J. Am. Soc. Hort. Sci., 74: 430-445.

Brown, R.F. and Mayer, D.G., 1988. Representing cumulative germination. 1. Germination indices. Ann. Bot., 61: 117-125.

Call, C.A. and Roundy, B.A., 1991. Perspectives and processes in revegetation of arid and semiarid rangelands. J. Range Manage., 44: 543-549.

Covell, S., Ellis, R.H., Roberts, E.H. and Summerfield, R.J., 1986. The influence of temperature on seed germination rate in grain legumes. I. A comparison of chickpea, lentil, soyabean and cowpea at constant temperatures. J. Exp. Bot., 37: 705-715.

Ellis, R.H., Covell, S., Roberts, E.H. and Summerfield, R.J., 1986. The influence of temperature on seed germination rate in grain legumes. II. Interspecific variation in chickpea (*Cicer arietinum* L.) at constant temperature. J. Exp. Bot., 37: 1503-1515.

Fidanza, M.A., Dernoeden, P.H. and Zhang, M., 1996. Degree-days for predicting smooth crabgrass emergence in cool-season turfgrasses. Crop Sci., 36: 990-996.

Flerchinger, G.N. and Pierson, F.B., 1997. Modeling plant canopy effects on variability of soil temperature and water: model calibration and validation. J. Arid Environ., 35: 641-653.

Flerchinger, G.N. and Saxton, K.E., 1989a. Simultaneous heat and water model of a freezing snow-residue-soil system. I. Theory and development. Trans. ASAE, 32: 565-571.

Flerchinger, G.N. and Saxton, K.E., 1989b. Simultaneous heat and water model of a freezing snow-residue-soil system. II. Field verification. Trans. ASAE, 32: 573-578.

Garcia-Huidobro, J., Monteith, J.L. and Squire, G.R., 1982. Time, temperature and germination of pearl millet (*Pennisetum typhoides* S & H.). I. Constant temperature. J. Exp. Bot., 33: 288-296.

Gummerson, R.J., 1986. The effect of constant temperatures and osmotic potentials on the germination of sugar beet. J. Exp. Bot. 37: 729-741.

Hardegree, S.P., 1994a. Matric priming increases germination rate of Great Basin native perennial grasses. Agron. J., 86: 289-293.

Hardegree, S.P., 1994b. Drying and storage effects on germination of primed grass seeds. J. Range Manage., 47: 196-199.

Hardegree, S.P., Jones, T.A. and Van Vactor, S.S., 2002. Variability in thermal response of primed and non-primed seeds of squirreltail [*Elymus elymoides* (Raf.) Swezey and *Elymus multisetus* (J.G. Smith) M.E. Jones]. Ann. Bot., 89: 311-319.

Hardegree, S.P. and Van Vactor, S.S., 1999. Predicting germination response of four cool-season range grasses to field-variable temperature regimes. Environ. Exp. Bot., 41: 209-217.

Hardegree, S.P. and Van Vactor, S.S., 2000. Germination and emergence of primed grass seeds under field and simulated-field temperature regimes. Ann. Bot., 85: 379-390.

Hardegree, S.P., Van Vactor, S.S., Pierson, F.B. and Palmquist, D.E., 1999. Predicting variable-temperature response of non-dormant seeds from constant-temperature germination data. J. Range Manage., 52: 83-91.

Harris, G.A. and Wilson, A.M., 1970. Competition for moisture among seedlings of annual and perennial grasses as influenced by root elongation at low temperature. Ecol., 51: 530-534.

Holshouser, D.L., Chandler, J.M. and Wu, H.I., 1996. Temperature-dependent model for non-dormant seed germination and rhizome bud break of johnsongrass (*Sorghum halepense*). Weed Sci., 44: 257-265.

Humphrey, L.D. and Schupp, E.W., 2001. Seed banks of Bromus tectorum-dominated communities in the Great Basin. West. N. Am. Nat., 61: 85-92.

Jensen, K.B., Horton, W.H., Reed, R. and Whitesides, R.E., 2001. Intermountain Planting Guide. Utah State University Extension Publication. AG510. Logan, UT, pp. 104.

Jordan, G.L. and Haferkamp, M.R., 1989. Temperature response and calculated heat units for germination of several range grasses and shrubs. J. Range Manage., 42: 41-45.

Meyer, S.E., Debaene-Gill, S.B. and Allen, P.S., 2000. Using hydrothermal time concepts to model seed germination response to temperature, dormancy loss and priming effects in *Elymus elymoides*. Seed Sci. Res., 10: 213-223.

Pierson, F.B. and Wight, J.R., 1995. Variability of near-surface soil temperature on sagebrush rangeland. J. Range Manage., 44: 491-497.

Roundy, B.A. and Call, C.A., 1988. Revegetation of arid and semiarid rangelands. In: P.T. Tueller (editor), Vegetation Science Applications for Rangeland Analysis and Management, Kluwer Academic Publishing, Dordrecht, pp. 607-635.

Scott, S.J., Jones, R.A. and Williams, W.A., 1984. Review of data analysis methods for seed germination. Crop Sci., 24: 1192-1199.

Wester, D.B., 1991. A summary of range plant seed germination research. International Center for Arid and Semiarid Land Studies (ICASALS) # 91-2.

Wilson, A.M., Wonderchek, D.E. and Goebel, C.J., 1974. Responses of range grass seeds to winter environments. J. Range Manage., 27: 120-122.

Young, J.A., Evans, R.A., Eckert, R.E. and Kay, B.L., 1987. Cheatgrass. Rangelands, 9: 266-270.

Young, J.A. and Longland, W.S., 1996. Impact of alien plants on Grant Basin rangelands. Weed Tech., 10: 384-391.

S. Hardegree, Project Leader
USDA-ARS; Boise, ID
shardegr@nwrc.ars.usda.gov

NOTES:

Ecotypic Variability in Fourwing Saltbush (*Atriplex canescens*)

Stewart C. Sanderson

This work was done under the direction of Howard C. Stutz, now retired, whose enthusiasm led to discovery of many of the chromosomal races of fourwing saltbush and related species.

It has been found in recent years that numerous distinctive races exist within fourwing saltbush (Sanderson and Stutz 2001). In addition to visible differences between these, they are also characterized by different degrees of polyploidy. Polyploidy is when the DNA of all the cells of a plant is present in twice the normal amount, or in even higher multiples. In saltbushes, the normal ("diploid") number of chromosomes is 18, including 9 from each parent. Tetraploids have 36 total chromosomes, hexaploids 54, and octoploids 72. The highest known fourwing ploidy, from just over the border in Mexico, is about 20-ploid with 180 chromosomes (Sanderson & Stutz 1994).

The ancestral forms are the diploids, those with the lowest chromosome number. In fourwing saltbush these are all uncommon and localized. It is not known exactly why the polyploids are more successful. In woody species of plants, polyploidy often causes a decrease in size (Stutz, Melby, and Livingston 1975, Sanderson, et al. 1989), which might give more tolerance towards drought. In saltbushes, polyploids also seem more tolerant of salinity (Sanderson, et al 1989).

Nevada has a greater variety of races than any other western state except California. Several of these occur in Lincoln County. Nevada has diploid, tetraploid, hexaploid and octoploid chromosome races. The diploids are very narrow-leaved plants found in the vicinity of Pioche and at somewhat the same altitude as the town. Nearby Panaca, at a lower elevation has tetraploids exclusively. All plants at Pioche are not diploids however. Hybridization between diploids and tetraploids proceeds in a one-way manner so that tetraploids there resemble diploids to varying degrees, sometimes very closely, but the diploids are not changed in any way. The whole valley in that area is a hodgepodge of varying forms extending from one extreme to the other, but only plants at the narrow-leaved extreme are diploids. Diploids must have once existed at Valley of Fire as well, because the plants there, all tetraploid, show strong evidence of hybridization.

The tetraploids in Nevada belong to a widespread race that we call Occidentalis that extends from northern Mexico to a few plants as far north as Alberta, Canada. In Nevada, tetraploids are more or less divided into northern and southern parts with a hexaploid race, Nevadensis, taking up the space in the middle. These northern and southern Occidentalis plants seem to look alike, but are no doubt different ecologically.

The hexaploid race Nevadensis (six chromosome sets) is found in Nye Co., Esmeralda Co., most of Mineral Co., and in southern Churchill and Lander counties. Plants of race Nevadensis are usually shorter than Occidentalis but are most clearly distinguished by leaf width. The leaves within pollen-producing inflorescences of Nevadensis are approximately double their width in Occidentalis. A width of 7 mm for Nevadensis, or about ¼ inch, seems to be an appropriate cutoff in the case of medium-length leaves.

There are also a localized hexaploid and octoploid that resemble each other in Sand Springs Valley near Rachel, Lincoln Co., and a further hexaploid west of Pahrump, Nye Co. that may belong to the hexaploid Race Vallis of New Mexico and Arizona.

Races of apparent hybrid origin are Nana (hexaploid, Little Smokey Valley and the Black Rock area, US 6), *Atriplex nuttallii*/Hybrida (hexaploid, the Humboldt River Valley in the Battle Mountain area), and Bonnevillensis (tetraploid, mostly Utah, with a population of a few acres near Cherry Creek, Elko Co.). These are apparently crosses of fourwing saltbush, Race Nevadensis in the first case, and Occidentalis in the other two, with one or both of the subshrubs *Atriplex tridentata* and *A. falcata*. Some of the hybrid races are locally abundant.

While there are many distinctive forms of fourwing saltbush, reclamation has mainly made use of the tetraploid race Occidentalis. It is widespread, and varies in ecological characteristics from place to place. In areas where any of the other races occur, they must be used in reclamation to avoid genetic contamination with other material. Otherwise, plants of the Occidentalis race are usually preferable, and should be matched to the proper climatic zone.

References:

Fitzsimmons, K., C. Lovely, and E. Glenn. 1998. Growth differences among widely separated geographic accessions of fourwing saltbush (*Atriplex canescens*) in the Great Basin Desert, New Mexico, USA. Arid Soil Research & Rehabilitation 12: 87-94.

Glenn, E., D. Moore, S. Sanderson, J. J. Brown, D. Lash, M. Nelson, and J. Waugh. 1998. Comparison of growth and morphology of *Atriplex canescens* varieties *occidentalis* and *angustifolia*.

The Southwestern Naturalist 43: 176-182.
Sanderson, S. C., E. D. McArthur, and H. C. Stutz. 1989. A relationship between ploidy and habitat in western shrub species. A. Wallace, E. D. McArthur, and M. R. Haferkamp, compilers, proceedings- symposium on shrub ecophysiology and biotechnology. USDA Forest Service, Intermountain Research Station, Ogden, UT. INT-256: 23-30.
Sanderson, S. C. and H. C. Stutz. 1994. High chromosome numbers in Mojavean and Sonoran Desert *Atriplex canescens* (Chenopodiaceae). American Journal of Botany 81: 1045-1053.
Sanderson, S. C. and H. C. Stutz. 2001. Chromosome races of fourwing saltbush. E. D. McArthur and D. J. Fairbanks, eds. Shrubland ecosystem genetics and biodiversity: proceedings. USDA Forest Service, Rocky Mountain Research Station. Ogden, UT. RMRS-P-21: 75-88.
Stutz, H. C., and S. C. Sanderson. 1983. Evolutionary studies of *Atriplex*: chromosome races of *A. confertifolia* (shadscale). American Journal of Botany 70: 1536-1547.

S. Sanderson, Genetist
USDA-FS-RMRS; Provo, Utah
ssanderson@fs.fed.us

NOTES:

Ecological Influence of Planting Introduced Grasses and Broadleaf Herbs in Sagebrush and Related Shrublands

Stephen B. Monsen

Many plant communities throughout the West were seriously disrupted by livestock grazing which ultimately created considerable interest in developing remedial treatments to protect critical watershed, sustain livestock grazing, and control weed invasion. Studies were initiated prior to the turn of the century to select and cultivate plants for restoration and revegetation purposes. Studies initially established by USDA Forest Service scientists reveled that a number of introduced perennial grasses could be readily established amid different communities. These species furnished protective ground cover and were desirable forage plants. Scientists from other agencies also recognized the desirable traits of these species, and numerous publications were prepared to encourage land managers to accept and use various plant introductions. Introduced grasses have subsequently become widely available and are now the principal species used in many conservation and revegetation projects.

Only a few introductions are adapted and planted in the sagebrush communities. Desert or standard crested wheatgrass (Agropyron *desertorum*), Fairway crested wheatgrass (*A. cristatum*), Siberian wheatgrass (*A. fragile*), and Russian wildrye (*Psathyrostachys junea*) are the principal species seeded in the Wyoming big sagebrush communities. Intermediate wheatgrass (*Thinopyrum intermedium*), smooth brome (*Bromus inermis*), orchardgrass (*Dactylis glomerata*), and hard fescue (*Fescue ovina duriuscula*) are seeded in more mesic environments. Most commonly planted introductions share the ability to establish with minimal seedbed preparation and develop uniform stands on quite diverse sites. Most all produce seedlings capable of surviving extended periods of drought, and plants are able to successfully survive amid competition from other, often weedy, species.

Some of the first revegetation studies established throughout the Intermountain region were on high elevation watersheds and tall forb communities. Revegetation of sites occupied by cluster tarweed (*Media glomerata*), a native annual that had spread and dominated these sites was a primary objective. Disking or spraying were used to remove tarweed seedlings and a combination of introduced grasses were planted. From previous adaptation trials, smooth brome, timothy (*Phelum partense*), tall oat grass (*Arrhenatherum elatius*), orchardgrass, and intermediate wheatgrass were found to be best suited. These species established well and furnished excellent ground cover and herbage for 10 to 50 years, but in many situations the grasses have weakened and disappeared and tarweed has re-established. In contrast, these same grasses demonstrate extremely competitive and persistent stands amid aspen, open parks, mountain brush, and pinyon/juniper communities, but not in sites occupied by tarweed. These grasses are individually adapted to the mountainous communities, but are not ecologically adapted as mixed communities with the biotic and edaphic influences of tarweed sites. In contract, these same grasses, particularly intermediate wheatgrass and smooth brome, have established, gained dominance, and slowly eliminated the native understory and woody species in numerous aspen, mountain brush, and antelope bitterbrush communities. Suppression and elimination of native herbs and shrubs has taken 20 to 40 or 50 years. Plantings of intermediate wheatgrass, smooth brome, and crested wheatgrass have individually and collectively prevented natural recruitment of antelope bitterbrush (*Purshia tridentata*) from a number of study sites in Idaho and Utah.

A series of comparative adaptation trials including about 80 different herbaceous species were established in Utah, Idaho, and Nevada beginning in 1929. Additional species were added over a 30-year period. Plantings were established in mixed antelope bitterbrush/ mountain big sagebrush communities. At all locations, intermediate wheatgrass, smooth brome, hard fescue, and crested wheatgrass established and have completely dominated their original planting sites. These species have also spread at different rates to suppress nearly all other species in adjacent areas. During a 73-year period from 1929 to the present, each grass has prevented recovery and occurrence of any native species.

Crested wheatgrass has been widely planted throughout the pinyon/juniper and sagebrush ecosystems to reestablish a herbaceous understory. Both desert crested wheatgrass and Fairway crested wheatgrass have exhibited excellent establishment, but mature plants do not allow recovery of native herbs or shrubs. The aggressive establishment traits, early growth, and dominance expressed by these species and a few other introductions have prevented natural reestablishment of native herbs and shrubs. Most native herbs and shrubs must be seeded in rows separate from the introduced grasses to ensure successful establishment. Many shrubs common to the sagebrush and mountain brush communities cannot be maintained if introductions are planted in the understory.

Although mixed seedings of native and introduced species have been maintained for some time, introduced grasses seriously prevented natural increase and recovery of most semi-arid shrubs. Natural recruitment of Wyoming big sagebrush (*Artemisia tridentata ssp. wyomingensis*) and common winterfat (*Krascheninnikovia lanata*) have been documented and widely observed amid sites sustaining a combination of native perennials, including Sandburg bluegrass (*Poa secunda*), bluebunch wheatgrass *(Pseudrogneria spicata*), and Indian rice grasses (*Achnatherum hymenoides*). In contrast, crested wheatgrass seedings have prevent recruitment of big sagebrush and winterfat in many situations.

A primary concern of seeding introduced grasses is their inability to control the invasion of a number of recently introduced and extremely difficult perennial weeds. Established plantings of intermediate wheatgrass and crested wheatgrass have been able to control annual weeds, principally cheatgrass (*Bromus tectorum*). These grasses have not been successful in preventing the rapid spread of rush skeleton (*Chondrilla juncea*) in central and southern Idaho, but have actually favored and perpetuated spread of the perennial. Established stands of crested wheatgrass in central Utah have not been effective in controlling the spread of squarrose knapweed (*Centaurea virgata*), but have served as a conduit for the weed to spread to adjacent disturbances particularly in pinyon/juniper woodlands.

Introductions have been widely accepted to stabilize watershed disturbance, reduce annual weed invasions, reduce wildfires, seed following wildfires, and supply forage for livestock. Numerous cultivars of different species having specialized traits have been developed and are commonly planted. Some introductions establish more successfully and consistently than many natives species. Although these plants have utility, they are not compatible with most native communities, and they ultimately disrupt and prevent recovery of native species. Transition to the use of native species is advisable, but development of site adapted ecotypes and improvement of plant establishment traits in arid environments is essential.

References:

Giunta, B.C.; Christensen, D.R.; Monsen, S.B.1975. Interseeding shrubs in cheatgrass with a browse seeder-scalper. J. Range Manage. 28(5): 398-402.

Hubbard, R. L.; Zusman, P.; Sanderson, R.H. 1967. Bitterbrush stocking and minimum spacing with crested wheatgrass. Calif. Fish and Game 48(3); 203-208.

Monsen, S. B.; Anderson, V.J. 1993. A 52-year ecological history of selected introduced and native grasses planted in central Idaho. In: Proceedings of the XVII International Grassland Congress; 1993 Feb 13-16; Lincoln, New Zealand. Palmerston North, New Zealand: New Zealand Grassland Association & others: 1740-1.

Monsen, S.B.; Stevens, R.; Walker, S.W. 1996. The competitive influence of seeded smooth brome (*Bromus inermis*) and intermediate wheatgrass (*Thinopyron intermedium)* within aspen-mountain brush communities of central Utah. In: Fifth International Rangeland Congress. Rangelands in a Sustainable Biosphere. Society for Range Management, Denver, CO: 379-380.

Walker, S. C. 1999. Species compatibility and successional processes affecting seeding of Pinyon-juniper types. In: Monsen, S.B.; Stevens, R., comps. 1999. Proceedings: ecology and management of Pinyon-juniper communities within the Interior West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: USDA, FS, RMRS: 331-345.

Walker, S.C.; Stevens, R.; Monsen, S.B.; Jorgensen, K.R. 1994. Interaction between native and seeded introduced grasses for 23 years following chaining of juniper-pinyon woodlands. In: Roundy, B.; McArthur, E.D.; Haley, J.S.; Mann, D.K. comp. Proceedings: wildland shrub and arid land restoration symposium; 1993 October 19-21; Las Vegas, NV. Gen. Tech. Rep. INT-GTR-315. Ogden, UT. USDA, FS, Intermountain Research Station.

S.B. Monsen, Retired Ecologist
USDA FS,RMRS; Mapleton, UT
smonsen5@cs.com

NOTES:

Ecotypic Variability, Seed Features, and Seedbed Requirements of Big Sagebrush

Stephen B. Monsen

Natural recruitment and planting success of semi-arid shrublands is often poor due, in part, to irregular patterns of seasonal precipitation (Jordan 1982). Competitive weeds are widely distributed throughout big sagebrush (*Artemisia tridentata*) shrublands and restrict establishment of native or introduced species (Monsen 1994). Shultz (1986) reported that populations of big sagebrush display close alliance to certain habitats, and morphological specializations and adaptations have evolved along environmental gradients. This has produced the current distribution patterns of the principal subspecies of big sagebrush. Davis and Stevens (1986) reported that significant differences in growth occurred within and among subspecies of big sagebrush indicating adaptation to site of origin. Differences in photosynthetic characteristics among species described by Frank et al. (1986) also correlate with environmental conditions at their sites of origin. These studies suggest subspecies and ecotypes have evolved to survive in distinct environments, and movement of populations to different climatic or edaphic conditions is not advised.

Plants of big sagebrush flower in late summer and early fall. Drought, late winter storms, or persistent cold temperatures can prevent seed development. Because plants are partially self-fertile, isolated shrubs do set seed (McArthur et al. 1988). Seed production differs among big sagebrush subspecies and must be considered in their use. Basin big sagebrush plants are larger than those of Wyoming big sagebrush or mountain big sagebrush and produce grater numbers of flowers and seeds (McArthur and Welch 1982). Mountain big sagebrush plants usually produce some seed each year, but amounts may not justify harvesting. Wyoming big sagebrush plants are much less floriferous and produce little seed except in unusually wet years. Most commercial seed is harvested from wildland stands, and more favorable sites and subspecies are repeatedly harvested. Seed production is, in part, genetically is regulated, thus seeds collections must be matched with their origin.

Seeds of all subspecies are small, approximately 4.5 million per kilogram.

Ripe seeds may persist on the bush for over a month, and are normally harvested by hand stripping or flailing, which also removes considerable debris. Material is cleaned to about 12 to 15 % purity, and stored at about 15% moisture content. Seed is sold on a pure live seed (PLS) basis. Seed viability diminishes after 1 or 2 years depending on storage conditions.

Meyer and Monsen (1992) found seed dormancy and germination are habitat correlated for all subspecies, but each subspecies exhibits a different pattern of variation. Habitat-correlation in germination appears to be an important adaptive feature. Seeds of mountain big sagebrush from severe winter conditions contain a higher fraction of dormant seeds and germinate more slowly than seeds of other subspecies. Seeds collections of mountain big sagebrush vary in germination from 0 to 58%. Those from severe winter conditions require up to 113 days to reach 50% germination. Basin and mountain big sagebrush seeds are mostly nondormant and germinate quickly at warm temperatures. Maximum dormancy for either species exceeds 12%, and 50% of seeds germinate within 6 days at warm temperatures. Virtually all seeds from fall plantings germinate in spring.

Movement of seeds from cold winter environments to mild environments or the reverse can result in germination at less favorable times. Loss of entire seedings can occur if site-adapted sources are not planted. Poorly adapted sources have initially established in some situations, but natural recruitment has been seriously limited. Selecting adapted sources is more important for this species than for many other shrubs species.

Fall planted seeds emerge in early spring, and approximately 57 % of all seeds normally germinated under a snow cover by late February. Germination may occur from mid-winter to early spring, but once conditions are favorable, rapid and complete germination can be expected. Early spring germination favors establishment as soil moisture is more likely to be available to sustain small seedlings. A protective snow cover ensures sufficient moisture for germination and initial growth and furnishes a protective cover to small seedlings that are sensitive to spring frosts. In arid regions it is unlikely that big sagebrush seedlings will establish except in years when snow accumulates in late winter. Seedings may be delayed until mid-winter until a snow cover has developed to assure moisture conditions and protection is provided to the seedlings.

Planting nurse crops to intercept moisture and moderate seedbed conditions is practical in many situations. Seeding rubber rabbitbrush (*Chrysothamnus nauseosus*) on barren mine sites Meyer (1994) and extremely large and open burns has facilitated sagebrush seedling invasion on sites where attempts to seed the shrub have previously been unsuccessful. Mine wastes have naturally been converted from rabbitbrush to sagebrush over a 10-year period

Small sagebrush seeds should be planted near the soil surface at a depth not to exceed 0.6 cm. Seeding on a firm surface that contains some litter or materials that prevent soil crusting is advisable. Seeding with a compression or compact type seeder is advised. The "sagebrush seeder" developed by Mike Boltz has proven universally successful in planting sagebrush in the arid environments of southern Idaho. Somewhat variable but adequate number of seedlings (1,600 to 9,630 plants/ha.) have established on the most arid Wyoming big sagebrush sites, and between 15,860 to 72,500 seedlings/ha have established on more upland Wyoming big sagebrush sites (Boltz 1994).

Aerial seeding in late fall and mid-winter following wildfires is also a successful practice. Chaining to cover the seed has resulted in approximately 64,250 seedlings/ha on open south and west aspects compared with about 6,000 seedling/ha on similar sites that were similarly seeded but not chained.

Seeding big sagebrush at rates between 0.11 to 0.22 kg/ha PLS is normally sufficient for broadcasting and surface seedings. Weed control measures and weather conditions are more important than the amount of seed planted.

Controlling competition from annual and perennial weeds is necessary for establishment of sagebrush seedlings. Cheatgrass competition is a major problem that must be addressed prior to seeding this shrub. Seeding big sagebrush into or with crested wheatgrass (*Agropyron cristatum*) is not advised. Hironaka et al. (1983) reported that natural recruitment into stands of crested wheatgrass is controlled, in part, by climatic conditions and the presence of a seed source.

Interseeding big sagebrush into established stands of crested wheatgrass is possible (Stevens 1994). Removal of the perennial grass in strips that are 0.5 m wide by tillage or with herbicides is necessary (Guinta et al.1994).

Planting or maintaining native understory plants normally favors big sagebrush recruitment. Frischknecht and Bleak (1957) found that more big sagebrush seedlings encroached into established stands of bluebunch wheatgrass (*Pseudoroegneria spicata*) than crested wheatgrass. Big sagebrush seeds can be planted with seeds of other species, but partitioning the seed boxes and drills to allow the slower developing shrub seeds to be planted in separate rows from more rapid developing grasses and herbs is advised.

References:

Boltz, M.. 1994. Factors influencing postfire sagebrush regeneration in south-central Idaho. Pages 281-290. In Monsen and Kitchen (see below).

Davis, J.M. and R. Stevens. 1986. Comparison of production in twenty-seven accessions of four sagebrush taxa. Pages 336-341. In McArthur and Welch (see below).

Monsen, S.B. and S.G. Kitchen. 1994 Compilers. Proceedings- ecology and management of annual rangelands. USDA Forest Service Gen. Tech. Report INT-GTR-313, Ogden, UT.

McArthur, E.D. and B.L. Welch. 1986. Compilers. Proceedings – symposium on the biology of *Artemisia* and *Chrysothamnus*. USDA Forest Service Gen. Tech. Report INT-2000, Ogden, UT.

Meyer, S. and S.B. Monsen. 1991. Habitat-correlated variation in mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) seed germination patterns. Ecology 72:739-742.

____

S. B. Monsen, retired Ecologist
USDA FS RMRS; Provo, UT
smonsen5@cs.com

NOTES:

What will we do with all the GIS Maps? A Spatial View of Restoration in the Great Basin

Steven T. Knick

Restoration of sagebrush (*Artemisia* spp.) landscapes is a critical ecological and administrative challenge to researchers and managers in the Intermountain West. Driven primarily by the potential listing of Greater Sage-grouse (*Centrocercus urophasianus)* as a Threatened or Endangered Species, conservation and restoration of sagebrush lands are top priorities of federal and state land and wildlife management agencies. Management actions to benefit any of the >300 species associated with sagebrush ecosystems have major ramifications for land use of large areas of the western United States. Over half of the total land area covered by sagebrush is owned publicly and managed by state or federal agencies; the U.S. Bureau of Land Management is the principal management authority for these regions. Less than 2-3% of the land surface covered by sagebrush receives permanent legal protection from alteration or conversion of land cover. Therefore, the future of sagebrush ecosystems will be determined primarily by the use of public lands and management policies of public agencies.

Very little of the sagebrush range, which once covered an estimated 62.7 x 10^6 ha in western North America, now exists in an undisturbed condition. Qualitative changes in composition of the vegetation and wildlife community combined with quantitative changes resulting from fragmentation and loss of major structural or functional species in sagebrush landscapes has disrupted ecosystem processes. As much as 50-60% of the native sagebrush steppe has been converted to annual grasslands. An estimated 99% of the basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*) landscapes in the Snake River Plain have been converted to agricultural lands. Disturbance regimes, including fire, now are very different from historical conditions in sagebrush ecosystems. Consequently, sagebrush and sagebrush steppe habitats are among the most imperiled ecosystems in North America.

The ultimate objective of landscape restoration is to recreate both the form and function of the original landscape by integrating individual projects into interacting components of a larger mosaic. As such, landscape restoration encompasses objectives for smaller spatial and shorter temporal scales. Local site-specific (bottom-up) efforts must be done in the context of the large-scale landscape (top-down) processes. Conversely, restoration success in within the entire landscape can be improved by incorporating large-scale spatial and full range of temporal processes of vegetation dynamics in local restoration plans.

The application of spatial components in developing strategies for restoration is largely undeveloped relative to our understanding of site-specific characteristics. Yet, spatial analysis and modeling in a Geographic Information Systems (GIS) can be important for predicting suitable locations and assessing potential risk for restoration sites. By developing statistical models, knowledge obtained from site-specific studies can be extended into unsampled regions to predict probabilities of encountering a set of conditions necessary for restoration success. Similarly, spatial modeling can be used to predict potential distributions of animals, which often are critical components guiding restoration plans.

Habitat at a site is a function of the landscape within which it is embedded, its previous state, and the preceding trajectory of habitat change. Again, GIS analyses can be important tools into understanding the spatial and temporal dynamics within the landscape that strongly influence the success of restoration. Spread of disturbance, such as fire, is a function of the spatial distribution of vegetative biomass and geographic relief. Similarly, restoration plans to reduce fragmentation or increase connectivity among habitat fragments can be critical to long-term success of the project as well as benefit wildlife.

Finally, spatial modeling is necessary to prioritize regions when planning restoration objectives. Total restoration of large areas, such as the Great Basin, is not possible within short timeframes because personnel, financial, and logistical constraints permit only a collection of local efforts. Consequently, restoration planning must be based on an hierarchical approach in which regional prioritization guides individual site-specific projects. Regional prioritization again is based on spatial models of habitat or environmental variables, as well incorporate models currently being developed to assess risk of cheatgrass invasion or expansion of juniper woodlands.

Spatial analyses, particularly large-scale assessments, previously have been hindered because maps of habitats or other environmental variables have been unavailable, have minimal distribution, or have limited extent in their coverage. Therefore, we developed a website (SAGEMAP http://SAGEMAP.wr.usgs.gov) containing spatial data important for research and management of sagebrush ecosystems in western North America. The website currently has >1,100 spatial datasets available for download at no cost. Each data layer has an associated metadata reference. Recently, an updated map of sagebrush distribution, developed by stitching together existing vegetation maps and standardizing classifications, was made available on the website. This coverage will provide a critical bridge in large scale analyses until new maps of sagebrush distribution can be completed.

Spatial modeling and analysis using GIS is becoming standard in many research and management offices. As such, the potential to increase our understanding of spatial and temporal patterns and processes in sagebrush ecosystem is enormous. By integrating these analyses into restoration strategies, we can increase our success in meeting the incredible challenge of restoring sagebrush landscapes.

References:

Allen, E. B., Editor. 1988. The reconstruction of disturbed arid lands: an ecological approach. Westview Press, Boulder, Colorado, USA.

Baker, W. L. 1994. Restoration of landscape structure altered by fire suppression. Conservation Biology 8:763-769.

Cadwell, L. L., J. L. Downs, C. M. Phelps, J. J. Nugent, L. Marsh, and L. Fitzner. 1996. Sagebrush restoration in the shrub-steppe of south-central Washington. Pages 143-145 in J. R. Barrow, E. D. McArthur, R. E. Sosebee, and R. J. Tausch, compilers. Proceedings: shrubland ecosystem dynamics in a changing environment. USDA FS Gen. Tech. Rep. INT-GTR-336. Ogden, UT. 275pp.

Comer, P., J. Kagan, M. Heiner, and C. Tobalske. 2002. Current distribution of sagebrush and associated vegetation in the western United States. GIS coverage available online (http://SAGEMAP.wr.usgs.gov).

Knick, S. T. 1999. Requiem for a sagebrush ecosystem? Northwest Science 73:53-57.

Knick, S. T., and D. L. Dyer. 1997. Spatial distribution of black-tailed jackrabbit habitat determined by GIS in southwestern Idaho. Journal of Wildlife Management 61:75-85.

Knick, S. T., and J. T. Rotenberry. 1997. Landscape characteristics of disturbed shrubsteppe habitats in southwestern Idaho (U.S.A.). Landscape Ecology 12:287-297.

Knick, S. T., and J. T. Rotenberry. 1998. Limitations to mapping habitat use areas in changing landscapes using the Mahalanobis distance statistics. Journal of Agricultural, Biological, and Environmental Statistics 3:311-322.

Knick, S. T., and J. T. Rotenberry. 2000. Ghosts of habitats past: contribution of landscape change to current habitats used by shrubland birds. Ecology 81:220-227.

Knick, S. T., and J. T. Rotenberry. 2002. Effects of habitat fragmentation on passerine birds breeding in Intermountain shrubsteppe. Studies in Avian Biology 25:131-141.

Mladenoff, D. J., and W. L. Baker, editors. 1999. Spatial modeling of forest landscape change: approaches and applications. Cambridge University Press, Cambridge, UK. 352pp.

Morrison, M. L. 2002. Wildlife restoration: techniques for habitat analysis and animal monitoring. Island Press, Washington, DC. 209pp.

Noss, R. F., E. T. LaRoe, III, and J. M. Scott. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. National Biological Service Biological Report 28. Washington, D.C.

Roundy, B. A., E. D. McArthur, J. S. Haley, and D. K. Mann. Compilers. 1993. Proceedings: Wildland shrub and arid land restoration symposium. USDA Forest Service Gen. Tech. Report INT-GTR-315. Intermountain Research Station, Ogden, UT.

SAGEMAP. 2002. A GIS database for Sage Grouse and shrubsteppe management in the intermountain West. http://SAGEMAP.wr.usgs.gov

Shugart, H. H. 1998. Terrrestrial ecosystems in changing environments. Cambridge University Press, Cambridge, UK. 537pp.

West, N. E. 1996. Strategies for maintenance of and repair of biotic community diversity on rangelands. Pages 326-346 *in* R. C. Szaro and D. W. Johnston (editors). Biodiversity in managed landscapes: theory and practice. Oxford University Press, New York, NY.

West, N. E. 2000. Synecology and disturbance regimes of sagebrush steppe ecosystems. Pages 15-26 *in* P.G. Entwistle, A. M. DeBolt, J. H. Kaltenecker, and K. Steenhof, compilers. Proceedings: Sagebrush steppe ecosystems symposium. Bureau of Land Management Pub. No. BLM/ID/PT-001001+1150. Boise, Idaho, USA.

West, N. E., and J. A. Young. 2000. Intermountain valleys and lower mountain slopes. Pages 255-284 *in* M. G. Barbour and W. D. Billings (editors). North American terrestrial vegetation. 2nd Ed. Cambridge University Press, Cambridge, UK.

Whisenant, S. G. 1995. Landscape dynamics and arid land restoration. Pages 26-34 in B. A. Roundy, E. D. McArthur, J. S. Haley, and D. K. Mann, compilers. Proceedings: wildland shrub and arid land restoration symposium. USDA For. Serv. Gen. Tech. Rep. INT-GTR-315. Ogden, UT. 384pp.

Whisenant, S. G. 1999. Repairing damaged wildlands: a process-oriented, landscape-scale approach. Cambridge University Press, Cambridge, UK. 312pp.

S. Knick, Research Ecologist
USDI, USGS; Boise, ID
steve_knick@usgs.gov

NOTES:

Natural and Prescribed Fires in Big Sagebrush Steppe- Response of Individual Species and Implications to Burning Practices

Stephen C. Bunting

The response of individual species to fire may vary greatly within the big sagebrush steppe vegetation. Sources of this variation have been attributed to a number of factors. A primary consideration is the associated subspecies of big sagebrush. Currently, 6 taxa have been described (*Artemisia tridentata* subsp. *tridentata*, *A. t. wyomingensis, A. t. vaseyana, A. t. xericensis, A. t. pauciflora,* and *A. t. spiciformis*). Many subspecies of big sagebrush occur over wide environmental ranges and with widely varying understory species. For example, Wyoming big sagebrush generally occurs within the 7-12 inch precipitation zone and on sandy to loamy soils. Consequently, the associated understory species varies widely, as does the effects of fire on those species. Other factors that have been noted to affect the response of big sagebrush steppe communities to fire include pre- and post-fire environmental conditions, fire severity, post-fire management, and pre-fire ecological condition.

The role of fire in the more arid big sagebrush steppe has been greatly influenced by the introduction of annual grasses, particularly cheatgrass and medusahead. Annual grasses have altered the fine fuel characteristics of the site, increasing the probability of wildfire occurrence. Frequent widespread wildfires have reduced the abundance of many fire sensitive species in this portion of the big sagebrush steppe and have resulted in a more homogeneous landscape. In addition, the dominance of annual grasses and other introduced species has reduced the seedling recruitment potential of many native species for the affected big sagebrush steppe communities.

Many areas of big sagebrush steppe within the transition zone with forest and woodland vegetation have been encroached by a variety of conifer species including pinyon pines, junipers, ponderosa pine, and Douglas-fir during the recent 150 years. Conifer encroachment has resulted in the replacement of many sagebrush steppe species by those species more typical of woodland and dry forest vegetation. The loss of sagebrush steppe is extensive in some regions of the West. Landscapes in these regions have become more homogeneous and less diverse. Fires, when they occur, tend to be more severe and may initiate successional trajectories that are different than those of the pre-Euro-American period. Introduced annual grasses may thrive post-burn in some areas and create a more frequent fire cycle similar to those previously mentioned.

All taxa of big sagebrush are readily killed by fire and must re-establish on the site from seed. Recovery times vary and may be as short as 15 years for mountain big sagebrush or as long as 50-75 years for Wyoming big sagebrush. Initial post-fire recruitment results from seeds found in the seed bed or produced by nearby unburned stands. Recovery times typically increase as the burned area becomes larger and more uniformly burned. Once the initial recruits have achieved reproductive maturity, on–site seed production produces the propagules for the recruitment of remainder of the stand. Mountain big sagebrush may mature within 5 years resulting in a more rapid recovery process than in the slower growing Wyoming big sagebrush.

Great Basin wildrye, bottlebrush squirrel-tail and bluebunch wheatgrass are among the most fire tolerant perennial grasses commonly found in the big sagebrush steppe and can readily survive high severity fires. Post-burn recruitment of squirrel-tail is often high but recruitment rates for wildrye and bluebunch wheatgrass are less predictable. The needle-grasses, including western, Letterman, needle-and-thread and Thurber, are among the most fire sensitive perennial grasses associated with big sagebrush steppe and may suffer high mortality during severe fires. Some species such as western needlegrass have high recruitment rates following disturbance and may dominate the site during the initial portion of the post-fire recovery period. Sandberg bluegrass may also suffer high mortality rates but post-burn recruitment rates are normally high. Idaho fescue has moderate fire tolerance and mortality varies widely with varying fire severity and other factors. Rhizomatous grasses such as western wheatgrass are not common in big sagebrush steppe but most species survive fire well and rapidly expand vegetatively to occupy open microsites created by the fire.

The importance of the forb component varies across the big sagebrush steppe. Native forb richness is low in Wyoming big sagebrush steppe under most conditions. Following disturbance introduced forb species such as tumblemustard, prickly-lettuce and Russian-thistle may briefly dominate the site. Forb richness increases with increasing moisture, consequently, mountain big sagebrush steppe has a diverse array of associated forbs. Individual species responses vary but as a group tend to increase during the initial post-burn recovery period.

References:

Anderson, J E; Inouye, R S. 2001. Landscape-scale changes in plant species abundance and biodiversity of a sagebrush steppe over 45 years. Ecological Monographs. 71:531-556.

Akinsoji, A. 1988. Postfire vegetation dynamics in a sagebrush steppe in southeastern Idaho. Vegetatio 78:151-155.

Billings, W.D. 1994. Ecological impacts of cheatgrass and resultant fire on ecosystems in the western Great Basin. In: Monsen, S.B.; Kitchen, S.G., comps. Proceedings, ecology and management of annual rangelands. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, F.S., Intermountain Research Station: 22-30.

Blank, R.R.; Allen, F.; Young, J.A. 1994. Extractable anions in soils following wildfire in a sagebrush-grass community. Soil Science Society of America Journal. 58:564-570.

Blank, R.R.; Young, J.A. 1998. Heated substrate and smoke: influence on seed emergence and plant growth. Journal of Range Management. 51:577-583.

Brown, J.K.; Smith, J.K., eds. 2000. Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42-vol. 2. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 257 p. www.fs.fed.us/rm/main/pubs/electronic/rmrs_gtr.html

Bunting, S.C.; Kilgore, B.M.; Bushey, C.L. 1987. Guidelines for prescribed burning sagebrush steppe rangelands in the northern Great Basin. Gen. Tech. Rep. INT-231. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 33 p. [Note: This publication includes an annotated bibliography of sagebrush/fire literature published from 1980-1986.]

Bunting, S.C.; Kingery, J.L.; Strand, E. 1999. Effects of succession on species richness of the western juniper woodland/sagebrush steppe mosaic. In: Monsen, S.B.; Stevens, R., comps. Proceedings, ecology and management of pinyon-juniper communities within the interior West. Proc. RMRS-P-9. Ogden, UT: USDA, Forest Service, Rocky Mountain Research Station: 76-81.

Bunting, S.C.; Kingery, J.L.; Hemstrom, M.A.; Schroeder, M.A.; Gravenmier, R.A.; Hann, W.J. 2002. Altered Rangeland Ecosystems in the Interior Columbia Basin. Gen. Tech. Rep. PNW-GTR-553. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. [http://www.fs.fed.us/pnw/pubs.htm]

Cook, J.G.; Hershey, T.J.; Irwin, L.L. 1994. Vegetative response to burning on Wyoming mountain-shrub big game ranges. Journal of Range Management. 47:296-302.

Fischer, R.A.; Reese, K.P.; Connelly, J.W. 1996. An investigation on fire effects within xeric sage grouse brood habitat. J. Range Mgt. 49:194-198.

Fraas, W.W.; Wambolt, C.L.; Frisina, M.R. 1992. Prescribed fire effects on a bitterbrush-mountain big sagebrush-bluebunch wheatgrass community. In: Clary, W.P.; McArthur, E.D.; Bedunah, D.; Wambolt, C.L., comps. Ecology and management of riparian shrub communities. Gen. Tech. Rep. INT-289. Ogden, UT: USDA, Forest Service, Intermountain Research Station: 212-216.

Gruell, G.E. 1999. Historical and modern roles of fire in pinyon-juniper. In: Monsen, S.B.; Stevens, R., comps. Proceedings, ecology and management of pinyon-juniper communities within the interior West. Proc. RMRS-P-9. Ogden, UT; USDA, Forest Service, Rocky Mountain Research Station: 24-28.

Hassan, M.A.; West, N.E. 1986. Dynamics of soil seed pools in burned and unburned sagebrush semi-desert. Ecology 67:269-272.

Johansen, J.R.; Ashley, J.; Rayburn, W.R. 1993. Effects of rangefire on soil algal crusts in semiarid shrub-steppe of the lower Columbia basin and their subsequent recovery. Great Basin Nat. 53:73-88.

Knick, S.T. 1999. Requiem for a sagebrush ecosystem? Northwest Science. 73:53-57.

Knick, S.T.; Rotenberry, J.T. 1995. Landscape characteristics of fragmented shrub-steppe habitats and breeding passerine birds. Conservation Biology. 9:1059-1071.

Knick, S.T.; Rotenberry, J.T. 1997. Landscape characteristics of disturbed shrub-steppe habitats in southwestern Idaho (U.S.A.). Landscape Ecology. 12:287-297.

Melgoza, G.; Nowak, R.S. 1991. Competition between cheatgrass and two native species after fire: implications from observations and measurements of root distribution. J. Range Management. 44:27-33.

Miller, R.F.; Rose, J.A. 1999. Fire history and western juniper encroachment in sagebrush steppe. Journal of Range Management. 52:550-559.

Miller, R.; Rose, J. 1998. Pre- and post-settlement fire return intervals on intermountain Sagebrush steppe. Special Report No. 991. Eastern Oregon Agricultural Research Center annual report. Corvallis, OR: Agricultural Experiment Station, Oregon State University: 16-17.

Miller, R.; Svejcar, T.; Rose, J. 1999. Conversion of shrub steppe to juniper woodland. In: S.B. Monsen, S.B.; Stevens, R., comps. Proceedings, ecology and management of Pinyon-Juniper communities within the interior West. Proc. RMRS-P-9. Ogden, UT: USDA, FS, RMRS: 385-390.

Nelle, P.J.; Reese, K.P.; Conley, J.W. 2000. Long-term effects of fire on sage grouse habitat. Journal of Range Management. 53:586-591.

Patton, B.D.; Hironaka, M.; Bunting, S.C. 1988.

Effect of burning on seed production of bluebunch wheatgrass, Idaho fescue and Columbia needlegrass. Journal of Range Management. 41:232-234.

Peters, E.F.; Bunting, S.C. 1994. Changing fire frequencies on Idaho's Snake River plain. In: Kitchen, S.G.; Monsen, S.B., comps. Proceedings, ecology and management of annual rangelands. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 31-36.

Pierson, F.B.; Carlson, D.H.; Spaeth, K.E. 2002. Impacts of wildfire on soil hydrological properties of steep sagebrush-steppe rangeland. International Journal of Wildland Fire 11:145-151.

Ratzlaff, T.D.; Anderson, J.E. 1995. Vegetal recovery following wildfire in seeded and unseeded sagebrush steppe. J. Range Mgt. 48:386-391.

Sapsis, D.B.; Kauffman, J.B. 1991. Fuel consumption and fire behavior associated with prescribed fires in sagebrush ecosystems. Northwest Science. 65:173-179.

Simanton, J.R.; Wingate, G.D.; Weltz, M.A. 1990. Runoff and sediment from a burned sagebrush community. In: Krammes, J.S., ed. Effects of fire management of Southwestern natural resources. Gen. Tech. Rep. RM-191. Fort Collins, CO: US.DA, Forest Service, Rocky Mountain Forest and Range Experiment Station: 180-5.

Smith, J.K., ed. 2000. Wildland fire in ecosystems: effects of fire on fauna. Gen. Tech. Rep. RMRS-GTR-42-vol. 1. Ogden, UT: USDA, FS, Rocky Mountain Research Station. 83 p. www.fs.fed.us/rm/main/pubs/electronic/rmrs_gtr.html

Wambolt, C.L.; Hoffman, T.L.; Mehus, C.A. 1999. Response of shrubs in big sagebrush habitats to fire on the northern Yellowstone winter range. In: McArthur, E.D. ; Ostler, W.K. ; Wambolt, C.L., comps. Proceedings, shrubland ecotones. RMRS-P-11. Ogden, UT: USDA, Forest Service, Rocky Mountain Research Station: 238-242.

Wambolt, C.L.; Walhof, K.S.; Frisina, M.R. 2001. Recovery of big sagebrush communities after burning in southwestern Montana. Journal of Environmental Management. 61:243-252.

Whisenant, S.G. 1990. Changing fire frequencies on Idaho's Snake River plains: ecological and management implications. In: McArthur, E.D.; Romney, R.M.; Smith, S.D.; Tueller, P.T., comps. Proceedings, symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management. Gen. Tech. Rep. INT-276. Ogden, UT: USDA, FS, Intermountain Research Station: 4-10.

Wambolt, C.L.; Payne, G.F. 1986. An 18-year comparison of control methods for Wyoming big sagebrush in southwestern Montana. Journal of Range Management. 39:314-319.

Watts, M.; Wambolt, C. 1996. Long-term recovery of Wyoming big sagebrush after four treatments. J Environmental Management. 46:95-102.

West, N.E; Yorks, T.P. 2002. Vegetation responses following wildfire on grazed and ungrazed sagebrush semi-desert. J. Range Mgt. 55:171-81.

___

S. Bunting, Professor, College of Natural Resources
Univ. of Idaho; Moscow, ID
sbunting@uidaho.edu

NOTES:

Comparative Response of Mountain And Wyoming Big Sagebrush Communities to Burning, Post Grazing Management, Seeding, Weed Invasion, and Native Restoration

Sherel Goodrich

Monitoring studies in Mountain big sagebrush and Wyoming big sagebrush communities in northeastern Utah show considerable difference in crown cover of sagebrush, potential for ground cover, understory diversity, potential for weed dominance, production, ungulate relations, and other features of these communities.

Differences in capabilities and function of sagebrush communities have implications for restoration projects. Where expectations exceed the capability of the land, restoration projects and other management activities are indicated for disappointment. Management plans and guidelines that do not include the inherent differences in sagebrush communities will be limited by this exclusion. Recognition of different sagebrush communities can be valuable to planning and management of sagebrush systems.

S. Goodrich, Ecologist
USDA FS, Ashley NF; Vernal, UT
sgoodrich@fs.fed.us

NOTES:

Major Sagebrush Communities – Distribution, Areas of Occurrence, and Species Composition

Sherel Goodrich

There are nearly 30 taxa of woody and semiwoody sagebrush (*Artemisia*) in western North America. Many of these sagebrush taxa are community dominants that are highly specialized. They are high-resolution indicators of climate, geomorphology, geology, 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.

Plant community classification based on sagebrush taxa combined with consistent, dominant understory species provides additional resolution of ecological factors that determine capabilities and function of different sites. Descriptions of many sagebrush communities are available in literature. Classification of plant communities can facilitate management decisions and provide direction for management practices.

S. Goodrich, Ecologist
USDA FS, Ashley NF; Vernal, UT
sgoodrich@fs.fed.us

NOTES:

Ecological response of sagebrush communities to different mechanical treatments.

Scott Walker

Fire suppression and the gradual shift to older-aged, woody plant communities, has contributed to a decline in ecosystem health on many public and private rangelands throughout Utah. Sagebrush-steppe and pinyon-juniper woodland ecosystems have been especially impacted at lower elevations. The consequences have been observed in many areas, including impaired watersheds, damaged fish and wildlife habitats and loss of forage for livestock.

Over the past decade, the Utah Division of Wildlife Resources has monitored the effects of a variety of mechanical treatments on sagebrush-steppe plant communities, with special attention given to changes in shrub density and shrub cover. In the fall of 2001, in cooperation with the Deseret Land and Livestock Ranch and Brigham Young University, the Division initiated a study to compare several pieces of equipment; a rangeland disc/imprinter, Dixie harrow, pasture aerator, and the Ely-type anchor chain. Preliminary results show differences between pieces of equipment, number of passes, and season of treatment, ranging from 43% to 98% shrub mortality.

In an earlier study initiated in November 1987, 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. The effect of the treatments on plant community characteristics and shrub vigor was documented over a three-year period. Stand density was reduced 60 percent on sites chained two-ways and 43 percent on sites chained over once. Shrubs from one-way chained sites produced more leader growth in 1989 and 1990 than those from 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.

S. Walker, Great Basin Research Center Project Leader, Utah DWR; Ephraim, Utah
scottwalker@utah.gov

NOTES:

Implications of Previous and Current Management Practices, Fire, Weed Invasion, and Climate on Pinyon-Juniper Communities

Robin J. Tausch

Following European settlement there has been a pronounced increase in both the distribution and density of pinyon and juniper across some of the most productive and diverse *Artemisia* dominated communities of the Great Basin. Woodlands now cover a broad range of elevations and environments in the Great Basin totaling about 20 million acres. Prior to European settlement woodland species were generally confined to more fire safe sites but now occupy a wider range of sites including productive sites with deep well drained soils. Currently two-thirds, to as much as 90% of the woodlands, depending on location, are less than 130 years old. This expansion is largely attributed to the reduced occurrence of fire through the reduction of fine fuels by livestock grazing, particularly between 1870 and 1930. As tree dominance increases, shrubs and herbaceous vegetation declines, further contributing to a decline in the frequency of fire over most of the twentieth century. However, these changes are now driving an increase in fire size, intensity, and frequency. The introduction of exotic annuals, primarily grasses, has dramatically changed successional patterns following fire on many sites. Particularly where pinyon is dominant, tree dominated woodlands are also crossing a threshold to susceptibility to intense crown fires and these types of fires are on the increase. The intensity of these fires on tree dominated sites can also help push the site across another threshold to dominance by exotics, changing the successional dynamics of the site. During early to middle stages in development when woodlands contain adequate understories they generally respond well to various treatment methods, including fire. Once woodlands are tree dominated treatment becomes more difficult and expensive.

References:

Miller, R.F.; Rose, J.A. 1995. Historic expansion of *Juniperus occidentalis* (western juniper) in southeastern Oregon. Great Basin Naturalist.55:37-45

Nowak, C.L.; Nowak, R.S.; Tausch, R.J.;Wigand, P.E.. 1994a. Tree and shrub dynamics in northwestern Great Basin woodlands and shrub steppe during the late-Pleistocene and Holocene. American Journal of Botany. 81:265-277

Nowak, C.L.; Nowak, R.S.; Tausch, R.J.;Wigand, P.E.. 1994b. A 30,000 year record of vegetation dynamics at a semi-arid locale in the Great Basin. Journal of Vegetation Science. 5:579-590

Sprugel, D.G. 1991. Disturbance, equilibrium, and environment variability: What is ‘natural’ vegetation in a changing environment? Biological Conservation. 58:1-18

Tausch, R.J.; West, N.E.; Nabi, A.A. 1981. Tree age and dominance patterns in Great Basin pinyon-juniper woodlands. J. Range Mgt. 34: 259-264

Tausch, R.J.; Wigand, P.E.; Burkhardt, J.W. 1993. Viewpoint: plant community thresholds multiple steady states, and multiple successional pathways: legacy of the Quaternary? Journal of Range Management. 46: 439-447

Thompson, R.S. 1990. Lake Quaternary vegetation and climate in the Great Basin. In: Betancourt, J.L.; VanDevender, T.; Martin, P.S. eds. Packrat middens: the last 40,000 years of biotic change. Tucson, AZ: Univ. of Arizona Press: 200-39

West, N.E.; Tausch, R.J.; Tueller, P.T. 1998. A management-oriented classification of pinyon-juniper woodlands of the Great Basin. Gen Tech Rep. RMRS-GTR-12. Ogden, UT: USDA FS RMRS

Wigand, P.E.; Hemphill, M.L.; Sharpe, S.; Patra, S. 1995. Great Basin semi-arid woodland dynamics during the late Quaternary. In: Waugh, W.J. ed. Proceedings: climate change in the four corners and adjacent regions: implications for environmental restoration and land-use planning: 1994 Sept. 12-14; Mesa State College, Grand Jct, CO: US DE: 51-70

R. Tauch, Project Leader
USDA FS RMRS; Reno, Nevada
rtausch@fs.fed.us

NOTES:

The Role of Fire Across the Sagebrush Biome

Rick Miller

The continued decline of the sagebrush biome and concern over threatened sagebrush obligate species have increased the debate over the past role and use of prescribed fire in this ecosystem. The spatially and temporally complex nature of fire and limited presettlement information make it difficult to describe presettlement fire regimes and frequently lead to the over-simplification of this disturbance process across the region. In a first approximation of presettlement fire regimes Frost (1998) estimated mean fire return intervals (MFRI) of 13-25 years across the sagebrush biome. However, Brown (2000) estimated higher intensity and lower frequency burns with MFRI between 35-100 years across the same region. The sagebrush biome is spatially complex, represented by over 25 species and subspecies of *Artemisia*, variable soils, parent materials, topography, climate, landscape patterns, and disturbance histories. Fire regimes were equally complex across this region. Even within one subspecies of big sagebrush presettlement fire regimes varied from frequent (10-20 years) low intensity burns in the wetter mountain big sagebrush/Idaho fescue association to high intensity fires occurring at greater than 200 year intervals in the arid mountain big sagebrush/western needlegrass association occupying sand and pumice soils. The spatial and temporal variability of fire regimes within and across plant associations was an important factor in creating landscape heterogeneity. However, today the significant shift in fire regimes is creating a more homogenous landscape.

A significant portion of the sagebrush biome is being replaced by pinyon and juniper woodlands, which currently occupy 19 million hectares in the Intermountain West (Miller and Tausch 2001). Prior to settlement, MFRI in a large portion of sagebrush cover types being invaded by pinyon and juniper varied between 12-50 years. As trees gain dominance, fire-dependent communities are successionally replaced by fire-safe communities resulting in MFRI >100 years (Miller and Rose 1999, Miller and Tausch 2001). However, as tree canopies exceed 50% woodlands can support high intensity crown fires in drought conditions and sufficient wind velocities. Although pinyon and juniper woodlands are estimated to have increased 10 fold in the past 130 years they currently occupy far less land than they are capable of under current climatic conditions. In addition, many of these woodlands are in a transitional state where tree densities and cover are continuing to increase, causing declines in understory biomass, cover, species diversity, seed pools, structural complexity, water capture, and increases in erosion.

In the more arid Wyoming big sagebrush cover type relatively long MFRI have significantly decreased from 50-100 years to as much as <10 years. Fire occurrence, severity, size, and complexity prior to settlement were probably highly variable in space and time across this cover type. The majority of fires were most likely patchy due to limited and discontiguous fuels. However, the probability of large and more complete fires increased during a series of consecutive wet years, which allowed fine fuels to accumulate. Fire severity across this cover type probably ranged from moderate to high depending on weather conditions during the fire event. Recovery of plant communities within the Wyoming big sagebrush cover type following fire is generally slower than in the wetter mountain big sagebrush cover type. The dramatic decrease in MFRI is largely attributed to the invasion of exotic weeds (Whisenant 1990). This more arid big sagebrush series is considerably less resilient to disturbance and less resistant to invasion by alien species than the mountain big sagebrush cover type. Density and cover of sagebrush can increase and native forb and grass cover decrease with overgrazing. Increased frequency of disturbance also increases the abundance of rabbitbrush. Following fire, depleted plant communities in this cover type usually become dominated by cheatgrass, western tansymustard, tumblemustard and on sites with high clay content medusahead.

The application of prescribed fire for restoration across the sagebrush biome must be carefully evaluated. Factors to be considered include the response of both undesirable and desirable plants and on the health of the landscape unit taking into account the response of soil water, flora, and fauna.

References:

Brown, J. K. 2000. Introduction and fire regimes. Pages 1-7 *in*: J.K. Brown and J.K. Smith (eds.), Wildland fire in ecosystems: effects of fire on flora. USDA Forest Service RMRS-GTR-42-vol. 2.

Bunting, S.C. 1984. Fire in sagebrush grass ecosystems: successional changes. In: Sanders, K. and J. Durham eds. Rangeland fire effects. USDI Bureau of Land Management Idaho State Office. Boise, ID.

Burkhardt, J.W., and E.W. Tisdale. 1976. Causes of juniper invasion in southwestern Idaho.

Ecology 76:2-484.
Frost, C.C. 1998. Presettlement fire frequency regimes of the United States: A first approximation. Pages 70-81 *in* T.L. Pruden and L.A. Brennan (eds.) fire in ecosystem management: shifting the paradigm form suppression to prescription. Tall timbers Research Station, Tallahassee, FL.
Gruell, G.E. 1999. Historical and modern roles of fire in pinyon-juniper. Pages 24-28. *in* S. B. Monsen and R. Stevens (compilers), Proceedings: Ecology and management of pinyon-juniper communities within the interior West. USDA Forest Service Proceedings RMRS-P-9.
Gruel, G.E., L.E. Eddleman, and R. Jandl. 1994. Fire history of the pinyon-juniper woodlands of Great Basin National Park. USDI National Park Service Report NPS/PNROSU/NRTR-94/01
Houston, D.B. 1973. Wildfires in northern Yellowstone National Park. Ecology 54:1109-1117.
Knapp, P.A. 1995. Intermountain West lightning-caused fires: climatic predictors of area burned Journal of Range Management 48:85-91.
Martin, R.E., and A.H. Johnson. 1979. Fire management of Lava Beds National Monument. Pages 1209-1217 *in* R. E. Linn (ed) Proceedings of the First Conference of Science and Research in the National Parks. USDI National Parks Service Transactions Proceedings Series No. 5.
Miller, R.F. and L.E. Eddleman. 2000. Spatial and temporal changes of sage grouse habitat in the sagebrush biome. Oregon State University Agricultural Exper. Station Technical Bulletin 151.
Miller, R.F., and J.A. Rose. 1995. Historic expansion of *Juniperus occidentalis* (western juniper) in southeastern Oregon. Great Basin Nat. 55:37-45.
Miller, R.F. and J.A. Rose. 1999. Fire history and western juniper encroachment in sagebrush steppe. Journal of Range Management 52:520-559
Miller, R.F., T. Svejcar, and J.A. Rose. 2000. Western juniper succession in shrub steppe: impacts on community composition and structure. Journal of Range Management 53:574-585
Miller, R.F. and R.J. Tausch. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. Pages 15-30 *in* T.P. Wilson and KJ.E. M. Galley (eds.), Proc. Of the Invasive Species Workshop: the role of fire in the control and spread of invasive species. Fire Conference 2000, Tall Timbers Research Station, Tallahassee, FL.
Miller, R.F. and P. Wigand. 1994. Holocene changes in semiarid pinyon-juniper woodlands: response to climate, fire and human activities in the US Great Basin. BioScience 44:465-474.
Stewart, G. and A.C. Hull. 1949. Cheatgrass (Bromus tectorum L.) an ecologic intruder in southern Idaho. Ecology 30:58-74.
Tausch, R.J. 1999. Historic woodland development, p. 12-19. *In:* S.B.Monsen, R. Stevens, R.J. Tausch, R. Miller, S. Goodrich (editors). 1999. Proceedings: Ecology and management of pinyon-juniper communities within the interior west. USDA Forest Service. RMRS-P-9.
Tausch, R.J. 1999. Transitions and thresholds: influences and implications for management in pinyon and Utah juniper woodlands. Pages 61-65. i*n* S.B. Monsen, R. Stevens (compilers). Proceedings: ecology and management of pinyon-juniper communities within the interior west. USDA Forest Service. RMRS-P-9.
Tausch, R.J. N.E. West, and A.A. Nabi. 1981. Tree age and dominance patterns in Great Basin pinyon-juniper woodlands. Journal of Range Management. 34:259-264.
Wall, T.G., R.F. Miller, and T. Svejcar. 2001. Juniper encroachment into aspen in the Northwest Great Basin. J. Range Management 54:691-698.
Whisenant, S.G. 1990. Changing fire frequencies on Idaho's Snake River plains: ecological and management implications. Pages 4-10. *In*: Proc – Symp. On cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management. USDA Forest Service GTR- INT-276.
Young, J.A., and R.A. Evans. 1981. Demography and fire history of a western juniper stand. Journal of Range Management. 34:501-505.

R. Miller, Professor, Rangeland Resources
Oregon State University; Burns, OR
rmiller@gf.state.az.us

NOTES:

Selecting Adapted Species, Developing Seeding Combination, Seeding Rates, and Methods of Treatment to Facilitate Establishment and Recovery of Native Species.

Richard Stevens

NOTES:

Native species will establish or recover when provided with some or all of the following factors or conditions:

a. Removal and/or reduction of undesirable, competitive, or non-compatible species.
b. Proper short-term and long-term management practices including reduction or elimination of grazing and other related impacts.
c. Desigining seedings to ensure establishment of both target species and additional site-adapted companion species.
d. Modification of management practices to enhance development of the principal seeded species and the recovery of on-site species.

Four critical steps should be followed in selecting species that will be seeded. First, develop a list of species and ecotypes that are adapted and would occur on the proposed planting site. Second, from this list determine which species have significant amount of high quality seed available for planting. Third, of the species with available site-adapted seeds, determine those that are compatible as young developing plants and will assure ecological development of desired plant community. Fourth, evaluate the final species list to determine if project objectives can be achieved or if objective may need to be altered.

R Stevens, President
Freemont Trading Company; Ephraim, Utah
maplels@cut.net

Regional Assessment of Threats to Sagebrush Habitats for Species of Conservation Concern

Michael J. Wisdom, Lowell H. Suring, Mary M. Rowland, Linda Schueck, Cara Wolff Meinke, Barbara C. Wales, Steven T. Knick, Richard F. Miller, and Robin J. Tausch

We developed regional assessment procedures to evaluate pervasive threats to habitats for species of conservation concern (species with declining or rare habitats or populations) in the sagebrush (*Artemisia* spp.) ecosystem. We illustrated the utility of these procedures in a prototype application to sagebrush and associated habitats in the Great Basin Ecoregion, and adjacent portions of other Ecoregions, encompassing 14 ecological provinces within Nevada, Utah, California, Oregon, and Idaho (Figure 1). Our procedures for regional assessment are needed because of the prospect of continued and extensive habitat declines for many species associated with the sagebrush ecosystem and the resulting high risk of regional extirpation for many species.

Our prototype application focused on threats posed by cheatgrass (*Bromus tectorum*) and by pinyon-juniper (*Pinus* spp. and *Juniperus* spp.) woodland invasion into existing sagebrush habitats. These two threats have substantially reduced the amount and quality of sagebrush across large areas of the Great Basin and adjacent ecoregions, and continue to challenge land managers. Accordingly, we developed rule sets to estimate the risk that continued invasion by cheatgrass or pinyon-juniper, or both, would cause future displacement and loss of existing sagebrush. The rule set for estimating risks posed by cheatgrass was based on elevation zones and geographic location. The rule set for assessing risk from pinyon-juniper used a variety of factors, including phsyiography, precipitation, proximity to pinyon-juniper stands, and taxon of sagebrush affected.

To assess these risks, sagebrush and other habitats were mapped using cover types from the SAGEMAP Project (http://sagemap.wr.usgs.gov); sagebrush cover types were assigned levels of displacement risk from cheatgrass and pinyon-juniper, using our rule sets. Following are example results for the threat posed by cheatgrass in the Great Basin Ecoregion, summarized for all sagebrush habitats, and for specific habitats of Greater Sage-grouse (*Centrocercus urophasianus*) and sagebrush vole (*Lemmiscus curtatus*). Sagebrush composes 28% (8.3 million ha) of all cover types in the Great Basin Ecoregion. For displacement threat posed by cheatgrass, approximately 4% (313,845 ha) of sagebrush in the Ecoregion was estimated at high or very high risk; 39% at moderate risk, and the remaining 57% at low risk.

Results for sage grouse and sagebrush vole illustrated the variation in these levels of risk for displacement by cheatgrass in the context of species-specific analyses. Approximately 4% (235,477 ha) of sage-grouse habitat was at high or very risk, 29% (1,658,352 ha) at moderate risk, and 65% (3,753,719 ha) at low risk. Sagebrush vole had a somewhat higher percentage of habitat at high and very high risk (10%), a comparable percentage at moderate risk (26%), and a substantially smaller percentage at low risk (28%) compared to sage-grouse. This difference can be traced to the large percentage of sagebrush vole habitat at no risk to invasion by cheatgrass (36%), in contrast to only 3% for sage-grouse.

Our results illustrate the potential management utility of assessing regional threats to sagebrush habitats for species of concern. For example, mapping sagebrush habitats that are highly vulnerable to invasion by cheatgrass, versus areas highly vulnerable to encroachment by pinyon-juniper, provides spatially-explicit knowledge needed to target each threat with the appropriate management prescriptions, and to estimate the area, time, and resources required to apply the prescriptions. Alternatively, mapping areas where such threats are not imminent allows managers to target fewer resources to these areas. Knowledge of threats to habitats can be used for multi-species evaluations, as we describe in our presentation.

References:

Billings, W. D. 1994. Ecological impacts of cheatgrass and resultant fire on ecosystems in the western Great Basin. Pages 22-30 in S. B. Monsen and S. G. Kitchen, editors. Proceedings – ecology and management of annual rangelands. General technical report INT-GTR-313. U.S. Forest Service Intermountain Research Station, Ogden, Utah.

Burgman, M. A., S. Ferson, and H. R. Akcakaya. 1993. Risk assessment in conservation biology. Chapman & Hall, London.

Hemstrom, M. A., M. J. Wisdom, M. M. Rowland, B. Wales, W. J. Hann, and R. A. Gravenmier. 2002. Sagebrush-steppe vegetation dynamics and potential for restoration in the Interior Columbia Basin, USA. Conservation Biology 16:1243-1255.

Groves, C., L. Valutis, D. Vosick, B. Neely, K.

Wheaton, J. Touval, and B. Runnels. 2000. Designing a Geography of Hope: a Practitioner's Handbook for Ecoregional Conservation Planning. Arlington,VA: The Nature Conservancy. Available online: www.conserveonline.org.

Knick, S. T. 1999. Requiem for a sagebrush ecosystem? Northwest Science 73:53-57.

McIver, J., and L. Starr. 2001. Restoration of degraded lands in the interior Columbia River basin: passive versus active approaches. Forest Ecology and Management 153:29-42.

Miller, R., R. Tausch, and W. Waichler. 1999. Old-growth juniper and pinyon woodlands. Pages 375-384 in S.B. Monsen and R. Stevens, compilers, Proceedings, ecology and management of pinyon-juniper communities within the Interior West; 1997 September 15-18, Provo, UT. USDA Forest Service, Rocky Mountain Research Station RMRS-P-9.

Miller, R. F., and L. L. Eddleman. 2000. Spatial and temporal changes of sage grouse habitat in the sagebrush biome. Technical Bulletin 151. Agricultural Experiment Station, Oregon State University, Corvallis, Oregon.

Nachlinger, J., K. Sochi, P. Comer, G. Kittel, and D. Dorfman. 2001. Great Basin: an ecoregion-based conservation blueprint. The Nature Conservancy, Reno, NV.

Tausch, R. J., J. C. Chambers, R. R. Blank, and R. S. Nowak. 1995. Differential establishment of perennial grass and cheatgrass following fire on an ungrazed sagebrush-juniper site. Pages 252-257 in B. A. Roundy, E. D. McArthur, J. S. Haley, and D. K. Mann, compilers. Proceedings: wildland shrub and arid land restoration symposium. General technical report INT-GTR-313. U.S. Forest Service Intermountain Research Station, Ogden, Utah.

West, N. E., R. J. Tausch, and P.T. Tueller. 1998. A management-oriented classification of pinyon-juniper woodlands of the Great Basin. USDA Forest Service, Rocky Mountain Research Station General Technical Report RMRS-GTR-12, Ogden, UT. 42p.

West, N. E. 1999. Managing for biodiversity of rangelands. Pages 101-126 in W. W. Collins and C. O. Qualset, editors. Biodiversity in agroecosystems. CRC Press, Boca Raton, Florida.

Wisdom, M. J., M. M. Rowland, B. C. Wales, M. A. Hemstrom, W. J. Hann, M. G. Raphael, R. S. Holthausen, R. A. Gravenmier, and T. D. Rich. 2002. Modeled effects of sagebrush-steppe restoration on Greater Sage-Grouse in the interior Columbia Basin, USA. Conservation Biology 16:1223-1231.

Wisdom, M. J., B. C. Wales, M. M. Rowland, M. G. Raphael, R. S. Holthausen, T. D. Rich, and V. A. Saab. 2002. Performance of Greater Sage-Grouse models for conservation assessment in the interior Columbia Basin, USA. Conservation Biology 16:1232-1242.

____

M.J. Wisdom, Research Wildlife Biologist
USDA FS, Pacific Northwest Research Station; La Grande, OR
mwisdom@fs.fed.us

L.H. Suring, Wildlife Biologist
USDA FS, WO, Rocky Mountain Research Station; Boise, ID

M.M. Rowland, Wildlife Biologist
USDA FS, PNRS; La Grande, OR

L. Schueck, Computer Specialist
USGS Biological Resources Division, Snake River Field Office; Boise, ID

C.W. Meinke, Wildlife Biologist and GIS Specialist
USGS Biological Resources Division, Snake River Field Office; Boise, ID

B.C. Wales, Wildlife Biologist
USDA FS, PNRS; La Grande, OR

S.T. Knick, Research Wildlife Biologist
USGS Biological Resource Division, Snake River Field Office; Boise, ID

R.F. Miller, Range Scientist and Professor of Rangeland Resources, Oregon State University, Squaw Butte Experiment Station; Burns, OR

R.J. Tausch, Supervisory Range Scientist
Rocky Mountain Research Station; Reno NV

NOTES:

Cheatgrass: Invasion, Occurrence, Biological/Competitive Features and Control Measures

Mike Pellant

Invasion and Occurrence

Cheatgrass (*Bromus tectorum*) is an introduced annual grass that is widely distributed on rangelands in the western U.S. The origins of cheatgrass are probably southwestern Asia (Young and others 1987) via contaminated grain from Europe in the late 1890's (Mack and Pyke 1983). Cheatgrass was preadapted to the climate and soils in the Great Basin Desert and filled the void left vacant by the reduction of native herbaceous vegetation by livestock grazing at the turn of the century. Cheatgrass is found in most of the western states having reached its range of current distribution by 1930 (Mack 1981). In a more recent survey, Pellant and Hall (1994) found 3.3 million acres of public lands in the Great Basin Desert dominated by cheatgrass and another 76.1 million acres either infested with or susceptible to cheatgrass invasion.

Cheatgrass could increase in the future due to its ability to evolve to survive in new environments and presence of multiple genotypes contributing to the evolution of ecotypes adapted to different environments (Novak 1994). Since its introduction and expansion into the sagebrush biome in the late 1800's, cheatgrass has expanded into the salt desert shrub communities in the lower elevations (Sparks and others 1990) and the higher elevation ponderosa pine zone (Daubenmire 1952). Both global warming (Ryan 1991) and increased CO_2 (Smith et. al 1987) are also predicted to increase the success of annual plants such as cheatgrass in current and possibly new environments.

Biological/Competitive Features

Cheatgrass is a winter annual that germinates in the fall, if climatic conditions are favorable, or in the following spring insuring annual recruitment (Mack and Pyke 1983). Fall germination and rapid elongation of roots provides cheatgrass with a competitive advantage over native perennial species (Harris 1967). Cheatgrass is also an efficient user of soil water in the upper soil profile. Cline et al. (1977) found that a cheatgrass community was more efficient in using water to a soil depth of 0.5 m than a native bluebunch wheatgrass community. However, the bluebunch community extracted water deeper in the soil profile. The availability and uptake of nutrients is also impacted by cheatgrass. The high nitrogen-use efficiency of cheatgrass gives it a competitive advantage over associated annual forbs (McLendon and Redente 1992) and some perennial plants (Link and others 1990).

The prolific seed production of cheatgrass also contributes to the competitive advantage of this grass over native vegetation. Young and others (1969) and Young and Evans (1975) reported cheatgrass densities in degraded Nevada big sagebrush (*Artemisia tridentata*) communities of between 5,000 to 15,000 cheatgrass seeds per m^2 while even higher seed densities (17,717/m^2) were reported in Idaho (Stewart and Hull, 1949). Cheatgrass can also survive periodic drought because viable seeds survive in the soil for up to 5 years (Young and others 1969). Seed production of cheatgrass is supported by high plant densities of cheatgrass that range from 10-13,000 plants per m^2 in Nevada (Young and others 1969) to 6,500 plants per m^2 in Idaho (Hull and Pehanec 1947).

Cheatgrass supports wildfires and in turn is very well adapted to survive frequent wildfires. The short growth period of cheatgrass relative to native plants increases the likelihood of wildfire starts and spread (Pellant 1990, Whisenant 1990). Platt and Jackman (1946) reported that cheatgrass became flammable 4-6 weeks earlier and remained susceptible to wildfires 1-2 months later than native perennials. Cheatgrass is usually dry by mid-July whereas perennial plants can still contain 65 per cent moisture on the same date (Murray and others 1978). Historically, wildfires occurred at return intervals of 32-70 years in some sagebrush types in the Great Basin (Wright and others 1979) and are now less than 5 years on certain southern Idaho rangelands (Pellant 1990). As a result native plant diversity is reduced and recovery periods are longer on burned rangelands (Whisenant 1990).

Control

It is well established that cheatgrass must be effectively controlled prior to attempts to revegetate cheatgrass infested rangelands (Hull and Stewart 1948, Evans and Young 1977, Jordan 1983). However, the threshold above which cheatgrass is deemed a problem requiring control followed by reseeding perennial plants has not been established in the Great Basin. Laycock (1991) identified Southern Idaho cheatgrass communities as examples of a wildfire-maintained steady state where a threshold has been crossed requiring more than just changes in management for the original native vegetation to return. The decision as to whether cheatgrass has crossed a threshold requiring active control and reseeding or if it can be managed passively (e.g., changes in livestock management) to maintain it as a minor component of a plant community is difficult to make since very little research has addressed this issue. Thus, the type of cheatgrass control technique employed will depend in part on the degree of infestation and the kinds and proportions of remnant perennial plants and biological crust. The remainder of this document will describe and evaluate biological, mechanical, prescribed fire and herbicide strategies to control cheatgrass.

Biological Control

Biological control techniques for cheatgrass have traditionally emphasized livestock as a tool but more recently research on the use of pathogens as a control agent has been initiated. Meyer and others (2000) are exploring the potential for cheatgrass control with a naturally occurring pathogen (*Ustilago bullata*) that causes head smut in cheatgrass. Kennedy (1994) has found naturally occurring microbes (e.g., bacteria) that appear to target cheatgrass roots and can reduce seed production up to 64%. The use of bacteria for cheatgrass control is being commercially pursued at this time.

Livestock (sheep, cattle and horses) are another somewhat controversial tool for cheatgrass control. The scientific literature supports the contention that livestock can reduce cheatgrass dominance. Stewart and Hull (1949) found that heavy grass use by sheep in early spring greatly reduced cheatgrass density and height. More recently, Vallentine and Stevens (1994) reviewed the literature on the use of livestock to control cheatgrass and concluded that, with appropriate management considerations (season of use, careful livestock management, and appropriate livestock forage utilization levels), cheatgrass production could be reduced.

One of the biggest drawbacks to grazing cheatgrass for control is the large, climatically influenced, fluctuations in annual forage production of cheatgrass (Stewart and Young 1939; Klemmedson and Smith 1964). Hull and Pehanec (1947) found a tenfold difference in cheatgrass production between a wet and dry year (3,461 and 361 lbs/ac, respectively), compared to an introduced wheatgrass seeding that produced 2,472 and 1,285 lbs/ac during the same wet and dry years. Adjusting livestock numbers upwards to fully utilize cheatgrass in high precipitation years and totally destocking in drought years is not economically feasible for many livestock operators.

Mechanical Control

Mechanical techniques commonly employed in cheatgrass control include mowing and disking or plowing. Mowing is generally not very effective in reducing cheatgrass due to the difficulty in properly timing the treatment (before the cheatgrass seed ripens) and the inability of this technique to directly deplete the cheatgrass soil seed reserve. Multiple mowing treatments may be required in wet springs and the reduction in vigor of remnant perennial plants may increase cheatgrass and other weeds in subsequent years. Site conditions such as rocks and steep slopes also limit the application of this expensive and time-consuming technique.

Plowing or disking can reduce the reproduction of live cheatgrass plants as well as reduce the cheatgrass seed reserve. In order for plowing or disking to be effective, cheatgrass seed must be buried at least 6 cm to obtain effective control (Hulbert 1955). A moldboard plow provides the most effective cheatgrass control but is expensive and not feasible on many rangeland sites due to rocky conditions (Hull and Stewart 1948). Rangeland plows or disk plows are less effective than the moldboard plows in reducing cheatgrass competition but they can be used in moderately rocky rangelands. Plowing or disking treatments must be done prior to cheatgrass seedripe ("purple" stage) or after fall germination for adequate control. The soil disturbance caused by disking or plowing often creates a seedbed that is ideal for future weed germination and expansion, including cheatgrass, unless the mechanical treatment is followed by a successful seeding.

Prescribed Fire

Properly timed burning can greatly reduce cheatgrass densities the year following the fire (Pehanec and Hull 1945; Stewart and Hull 1949). Stark and others (1946) reported that cheatgrass was effectively controlled (around 90%) by burning in late spring before the seed matured. However, the cheatgrass seed reserve in the soil surface is not totally controlled by burning allowing recovery of the cheatgrass stand in a few years if reseeding with perennial grasses is not successful. Cheatgrass densities were reduced from 990 to 139 plants per m^2 the season after a June experimental burn near Boise, Idaho (Pellant 1990). The cheatgrass plants in the burned plots produced over twice as many seedstalks as did the cheatgrass plants in the unburned plots indicating a rapid recovery of the cheatgrass seed reserve. Burning, regardless of the timing, will reduce but not eliminate cheatgrass from the environment. Risks associated with using fire to reduce cheatgrass also include the danger of fire escape and the creation of a seedbed conducive to invasion by other weeds.

Herbicides

Herbicides are another option for controlling cheatgrass competition. Weed control systems utilizing herbicides were developed by Eckert and others (1974) to promote the establishment of perennial wheatgrasses in cheatgrass infested rangelands. These strategies were not applied to public lands until 1991 when the Final Environmental Impact Statement for Vegetation Treatment on BLM Lands in Thirteen Western States (USDI 1991) was approved allowing the use of 21 herbicides to control cheatgrass and other weeds. Herbicides are generally costly to apply yet provide better cheatgrass control than other control techniques. Ogg (1994) reviewed the herbicides effective in controlling cheatgrass. Economics, environmental impacts, selectivity, and effectiveness are among the factors that must be considered prior to selecting an herbicide to reduce plant densities in cheatgrass-infested rangelands

Recently, OUST® (sulfometuron methyl), a DuPont registered herbicide (DuPont 1996), has been used to reduce or eliminate cheatgrass prior to seeding perennial plants in fire rehabilitation or greenstripping projects (Pellant and others 1999). The use of this herbicide on public lands has recently been suspended in the wake of claims of damage to adjacent croplands in southern Idaho.

Summary

The control of cheatgrass is generally not the endpoint unless the plant community being treated still has the resilience and integrity to recover with a "passive" management intervention. For example, a "targeted" livestock grazing strategy may be useful in reducing cheatgrass in a native plant community if the plant phenology and livestock forage preference are closely monitored triggering appropriate management responses. If the threshold of plant community integrity has been crossed and cheatgrass is driving the community processes, cheatgrass control must be followed by restoration of perennial plants. Both introduced and native plants have been used to restore cheatgrass infested rangelands after control measures were implemented (Hull and Pehanec 1947, Hull and Stewart 1948, Hull and Holmgrem 1964, Hull 1974). Cost, availability, project objectives and the competiveness of the plants considered for seeding in the post-cheatgrass control environment must all be considered in developing the cheatgrass control and restoration plan.

References:

Cline, J.F.; Uresk, D.W.; Rickard, W.H. 1977. Comparison of soil water used by a sagebrush-bunchgrass community and a cheatgrass community. Journal of Range Manage. 30:199-201.

Daubenmire, R.F. 1952. Forest vegetation of northern Idaho and adjacent Washington, and its bearing on concepts of vegetation classification. Ecological Monographs. 22: 301-330.

DuPont Agricultural Products. 1996. OUST herbicide product label. E.I. DuPont de Nemours and Company, DuPont Agricultural Products, Wilmington, Delaware, USA.

Eckert, R.E. Jr.; Asher, J.E.; Christensen, M.D.; Evans, R.A. 1974. Evaluation of the atrazine-fallow technique for weed control and seeding establishment. Journal of Range Manage. 27: 288-292.

Evans, R. A.; Young, J. A. 1977. Weed control-revegetation systems for big sagebrush-downy brome rangelands. J. Range Manage. 30:331-336.

Evans, R. A.; Young, J. A. 1978. Effectiveness of rehabilitation practices following wildfire in a degraded big sagebrush-downy brome community. J. Range Manage. 31:184-188.

Harris, G.A. 1967. Some competitive relationships between *Agropyron spicatum* and *Bromus tectorum.* Ecological Monographs. 37:89-111

Hulbert, L. C. 1955. Ecological studies of *Bromus tectorum* and other annual bromegrasses. Ecological Monographs. 25:181-213.

Hull, A.C.1974. Species for seeding arid rangeland in southern Idaho. Journal of Range Manage. 27:216-218.

Hull, A.C., Jr., and Pehanec, J.F. 1947. Cheatgrass-a challenge to range research. Journal of Forestry. 45:555-564.

Hull, A. C., Jr.; Stewart, G. 1948. Replacing cheatgrass by reseeding with perennial grass on southern Idaho ranges. American Society of Agronomy Journal. 40: 694-703.

Hull, A. C., Jr.; Holmgren, R. C. 1964. Seeding southern Idaho rangelands. Res. Pap. INT-10. Ogden, UT: USDA, FS, Intermountain Forest and Range Experiment Station. 31 p.

Jordan, G. L. 1983. Planting limitations for arid, semiarid and salt-desert shrublands. In: Managing Intermountain rangelands--improvement of range and wildlife habitats: Proceedings; 1981 September 15-17; Twin Falls, ID; 1982 June 22-24; Elko, NV. Gen. Tech. Rep. INT-157. Ogden, UT USDA, FS, Intermountain Forest and Range Expt. Station. 11-18.

Kennedy, A. 1994. Biological control of annual grass weeds. In: Monsen, Stephen B.; Kitchen, Stanley G., comps. 1994. Proceedings--ecology and management of annual rangelands; 1992, May 18-22; Boise ID, Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 186-189.

Klemmedson, J.O.; Smith, J.G. 1964. Cheatgrass (*Bromus tectorum*). Botanical Review. 30:226-262.

Laycock, W.A. 1991. Stable states and threshold of range condition on North American rangelands-a viewpoint. Journal Range Manage. 44:427-433.

Link, S.O.; Gee, G.W.; Downs, J.L. 1990. The effect of water stress on phenological and ecophysiological characteristics of cheatgrass and Sandberg's bluegrass. J. Range Manage. 43:506-513.

Mack, R.N. 1981. Invasion of *Bromus tectorum* L. into Western North America: an ecological chronicle. Agro-Ecosystems. 7:145-165.

Mack, R.N.; Pyke, D.A. 1983. The demography of *Bromus tectorum:* variation in time and space. Journal of Ecology. 71:69-93.

Meyer, S.E., D.L. Nelson, S. Clement, A. Paulsen. 1999. Exploring the potential for biocontrol of cheatgrass with the head smut pathogen. In: Entwhistle, P.G., A.M. Debolt, J.H. Kaltenecker, and K. Steenhoff, compilers. 2000. Proceedings: Sagebrush steppe ecosystems symposium. Bureau of Land Management Publication No. BLM/ID/PT-001001+1150, Boise, ID. 70-72.

McLendon, T.; Redente, T. 1992. Effects of nitrogen limitation on species replacement dynamics during early secondary succession on a semiarid sagebrush site. Oecologia. 91:312-317.

Novak, S.J. 1994. Quantitative variation within and among cheatgrass populations: the roles of multiple introductions. In: Monsen, Stephen B.; Kitchen, Stanley G., comps. 1994. Proceedings--ecology and management of annual rangelands; 1992, May 18-22; Boise ID, Gen. Tech. Rep. INT-GTR-313. Ogden, UT: USDA, Forest Service, Intermountain Research Station. 103-108.

Ogg, A.J. Jr. 1994. A review of the chemical control of cheatgrass. In: Monsen, Stephen B.; Kitchen, Stanley G., comps. 1994. Proceedings--ecology and management of annual rangelands; 1992, May 18-22; Boise ID, Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, FS, Intermountain Research Station. 194-196.

Pechanec, J. F.; Hull, A. C. 1945. Spring Forage lost through cheatgrass fires. Nat. Wool Grower.

Pellant, M. 1990. Unpublished data on file at: U. S. Department of Interior, Bureau of Land Management, Idaho State Office, Boise, ID.

Pellant, Mike. 1990. The cheatgrass-wildfire cycle--are there any solutions? In: McArthur, E. Durant; Romney, Evan M.; Smith, Stanley D.; Tueller, Paul T., compilers. Proceedings--symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management; 1989 April 5-7; Las Vegas, NV. Gen. Tech. Rep. INT-276. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 11-17.

Pellant, Mike; Hall, Christine. 1994. Distribution of two exotic grasses on public lands in the Great Basin: Status in 1992. In: Monsen, Stephen B.; Kitchen, Stanley G., comps. 1994. Proceedings--ecology and management of annual rangelands; 1992, May 18-22; Boise ID, Gen. Tech. Rep. INT-GTR-313. Ogden, UT: USDA, Forest Service, Intermountain Research Station. 109-112.

Pellant, M, J. Kaltenecker, and S. Jirik. 1999. Pages 322-326 *in* S.B. Monsen and R. Stevens, compilers. Proceedings: ecology and management of pinyon-juniper communities within the Interior West. USDA Forest Service Proceedings RMRS-P-9.

Platt, K., and Jackman, E.R. 1946. The cheatgrass problem in Oregon. Oregon Agricultural Experiment Station Bulletin 668.

Ryan, Kevin C. 1991. Vegetation and wildland fire: implications of global climate change. Environmental International. 17:169-178.

Smith, S.D.; B.R. Strain; T.D. Sharkley. 1987. Effects of CO_2 enrichment on four Great Basin Grasses. Functional Ecology. 1:139-143.

Stark, R.H.; Toers, J.L.; Hafenrichter, A.L. 1946. Grasses and cultural methods of reseeding abandoned farmlands in southern Idaho. Idaho Agricultural Experiment Station Bulletin 267. 36 p.

Stewart, G.; Hull, A. C. 1949. Cheatgrass (Bromus tectorum L.) - an ecologic intruder in southern Idaho. Ecology. 30:58-74.

Sparks, S.R.; West, N.E.; Allen, E.B. 1990. Changes in vegetation and land use at two townships in Skull Valley, western Utah. In: McArthur, E. Durant; Romney, Evan M.; Smith, Stanley D.; Tueller, Paul T., compilers. Proceedings--symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management; 1989 April 5-7; Las Vegas, NV. Gen. Tech. Rep. INT-276. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 26-36.

Stewart, G.; Young, R.A. 1939. The hazard of basing permanent grazing capacity on *Bromus tectorum.* American Society of Agronomy Journal. 31: 1002-1015.

U.S. Department of Interior. 1991 Final Environmental Impact Statement for Vegetation Treatment on BLM Lands in Thirteen Western States. Bureau of Land Management, Casper, WY.

Vallentine, J.H., and A.R. Stevens. 1994. Use of livestock to control cheatgrass – a review. Pages 202-206 *in* SB. Monsen and S.G. Kitchen, compilers. Proceedings: ecology and management of annual rangelands. USDA Forest Service General Technical Report INT-GTR-313

Whisenant, Steven G. 1990. Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. In: McArthur, E. Durant; Romney, Evan M.; Smith, Stanley D.; Tueller, Paul T., compilers. Proceedings--symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management; 1989 April 5-7; Las Vegas, NV. Gen. Tech. Rep. INT-276. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 4-10.

Wright, H.A.; Neuenschwander, L.F.; Britton, C.M. 1979. The role and use of fire in sagebrush and pinyon juniper plant communities: a state-of-the-art review. Gen. Tech. Rep. INT-58. Ogden UT: USDA, Forest Service, Intermountain Forest and Range Experiment Station. 48 p.

Young, J. A.; Evans, R.A. 1975. Germinability of seed reserves in a big sagebrush community. Weed Science. 23:358-364.

Young, J. A.; Evans, R.A. 1978. Population dynamics after wildfires in sagebrush grasslands. J. Range Manage. 31:283-289.

Young, J. A.; Evans, R.A.; Kay, B. L. 1987. Cheatgrass. Rangelands. 9:266-270.

M. Pellant, Rangeland Ecologist
BLM, Idaho State Office; Boise, Idaho
Mike_Pellant@blm.gov

NOTES:

Historical Use of Nevada Rangelands and Iimplications to Mule Deer Habitats.

Mike Hess

The Nevada environment, relatively stable for thousands of years, changed radically in just a few decades because of new land uses by colonizing white settlers. Forests were the first major habitats to be altered by new uses. The mining boom, begun in 1859 with the discovery of the Comstock, extended throughout the region in the next decade exploiting the local forests for timbers, lumber and fuel wood on a massive scale. The correlation between wood consumption and bullion production for the Comstock Lode is strong. By 1900 about 196 stamp mills built on the Comstock supported by 131 sawmills and cordwood businesses operating in the adjoining Sierra Nevada. During the same period 124 mining camps with a total of 225 stamp mills operated in interior Nevada (outside the Comstock Lode). An extensive but poorly documented wood and lumber industry developed throughout Nevada prior to 1900, but most timber came from the Sierra Nevada to sites served by rail. Charcoal manufacturing for chlorination furnaces and smelters represented the largest segment of the interior wood industry. The Comstock consumed an estimated 15.9 million tons of lumber and cordwood, while collectively the interior camps consumed an estimated 18.4 million tons. Most consumption consisted of fuel wood, 74% for the Comstock and 90% for the interior camps. The present pattern of pinyon-juniper dominance and tree age distribution coincides with the pattern of exploitation predicted from local bullion production and ore processing methods. Corresponding increases in sagebrush habitat also occurred where trees were removed. Up to 90% of the modern pinyon-juniper woodland is less than 150 year old.

Local domestic livestock operations developed to serve the new markets the mines and mills provided. The completion of the transcontinental railroad in 1869 resulted in the expansion of the *laissez-faire* livestock industry by providing access to markets outside the state. Livestock populations erupted during the 1870's and 1880's, reaching estimated levels of 700,000 cattle and 400,000 sheep. The brutal winter of 1890 decimated range cattle and nomadic sheep operations erupted in response to decreased competition. Livestock numbers–especially sheep–peaked between 1910 and 1930. Three to 4 million sheep grazed Nevada ranges immediately following World War I. Severe overgrazing resulted in protective measures by the federal government, primarily the withdrawal of forest reserves between 1899 and 1911. Overgrazing increased on unregulated ranges until the 1930's when bank failures, drought and the new Grazing Service (BLM) ended unrestricted grazing on public lands. Many irrevocable changes occurred on Nevada ranges as a result of the severe overgrazing. Extreme erosion, lower water tables and the introduction of exotic plants were perhaps the most critical. Many brush species were given competitive advantage with the depletion of the herbaceous understory. Mule deer populations erupted in the late 1930's attaining peaks in the 1950's and again in the 1980's. This occurred as cattle numbers increased and remained high. Some deer populations lost critical winter ranges during a massive federal range conversion effort in the 1950's and 1960's, focused mainly on sagebrush conversion with some pinyon-juniper eradication. Some 65% of Nevada's deer winter ranges are associated with pinyon-juniper forests and deer wintering on these ranges maintain higher densities and higher winter fawn survival, indicating some advantage is available to deer in the mid-seral stages of the woodlands. Sagebrush is now recognized as critical forage and cover for deer, especially considering their cycle of appetite and subsequent behavior in winter.

At the turn of the 21st century, one critical deer range problem expanded explosively. Wildfire burned an unprecedented 1.8 million acres in 1999 and another 0.66 million acres in 2000. The majority of burned habitat was sagebrush range, and much was deer range. A massive rehabilitation effort was launched but poor precipitation years have yielded mixed results. Cheatgrass was critical in these wildfires and represents a worsening problem for sagebrush and deer ranges in the future. The range rehabilitation work at Dunphy Hills holds promise for mitigation of the cycle of cheatgrass range degradation. The Great Basin Restoration Initiative represents a refreshing approach, but it may never receive adequate funding for implementation.

References:

________. 1934. Grazing sheep on the public domain and in the national forests. Hearings before Special Committee on Conservation of Wild Life Resources, United States Senate. Washington D.C. U.S. Gov. Print. Office. 249pp.

Ansari, M.B. 1989. Mines and mills of the Comstock region western Nevada. Camp Nevada Monograph No. 8: Reno. 101pp.

Aldous, C. M. 1945. A winter study of mule deer in Nevada. J. of Wildl. Manage. 9(2):145-151.

Blackburn, W. H. and P. T. Tueller. 1970. Pinyon and juniper invasion in black sagebrush communities in east-central Nevada. Ecology 51(5):841-848.

Born, J.D., R.P. Tymcio, and O.E. Casey. 1992. Nevada forest resources. Resour. Bull. INT-76. Ogden, UT: U.S. Dept. of Agr., For. Serv., Inter. Res. St. 64pp.

Couch, B. F. and J. A. Carpenter. 1943. Nevada's metal and mineral production (1859-1940, inclusive). Geol. and Min. Series No. 37:4. Univ. of Nev., Reno. 159pp.

Georgetta, C. 1972. Golden fleece in Nevada. Venture Publ. Co.,Reno, Nev. 562pp.

Gill, R. 1990. Monitoring the status of European and North American cervids. GEMS Info. Series No. 8: Nairobi. 277 pp.

Gottfried, G.J. and K.E. Severson. 1994. Managing pinyon-juniper woodlands. Rangelands 16(6): 234-236.

Grayson, D. 1993. The desert’s past, a natural prehistory of the Great Basin. Smithsonian Inst. Press: Washington. 356plp.

Grazeola, F. 1969. The charcoal burners war of 1879: a study of the Italian immigrant in Nevada. M.S. thesis. Univ. of Nev.-Reno. 126pp.

Greever, W. S. 1963. The bonanza west: the story of the western mining rushes, 1848-1900. Univ. of Oklahoma Press: Norman. 430pp.

Griffiths, D. 1902. Forage conditions of the northern border of the Great Basin. Bureau of Plant Ind. Bull. No. 15.U.S.D.A.Gov. Print. Off. Washington, D.C. 60pp.

Gruell, G. E. 1986. Post-1900 mule deer irruptions in the Intermountain West: principle causes and influences. U.S.D.A. Inter. Mount. Res. St.: Ogden. General Tech. Rep. INT-206 37p.

Hall, E.R. 1946. Mammals of Nevada. Univ. Cal. Press. Reprinted 1995. Univ. Of Nev. Press: Reno, NV. 710pp.

Hanley, T.A. 1982. The nutritional basis for food selection by ungulates. J Range Mgmt.35:146-51

Hanley, T.A. and K.A. Hanley. 1982. Food resource partitioning by sympatric ungulates on Great Basin rangeland. J. Range Mgmt. 35:152-158.

Hardesty, D.L. 1985. Evolution on the industrial frontier. Pp. 213-229. In: S.W. Green and S.M. Perlman, eds. The archaeology of frontiers and boundaries. Academic Press, Orlando. 344pp.

Hattori, E. M., M. A. Thompson and A. R. McLane. 1984. Historic pinyon pine utilization in the Cortez mining district in central Nevada: the use of dendrochronology in historical archeology and historical reconstructions. Desert Res. Instit. Soc. Sci. Cent. Tech. Rep. No. 39: Reno. 35pp.

Hess, M.L. 1991. Nevada bonanza deforestation and mule deer. Presented at Western States Deer Workshop: Pacific Grove, CA. On file at Nev. Div. Of Wildlife, Reno, NV. 9pp.

Kay, R.N.B. 1987. The comparative anatomy and physiology of digestion in tragulids and cervids and its relation to food intake. In: C.M. Wemmer, ed. Biology and management of the Cervidae. Smithsonian Instit. Press:Washington, D.C. pp214-22.

Klebenow, D.A. 1984. Big game response to fire in sagebrush-grass rangelands. pp. 53-57. **In**: K. Sanders and J. Durham, eds. Rangeland fire effects, a symposium. Idaho St. B.L.M. Off., U.S.D.I.: Boise. 124 pp.

Koniak, S. 1985. Succession in pinyon-juniper woodlands following wildfire in the Great Basin. Great Basin Nat. 45:556-566.

Lane, R. H. 1974. The cultural ecology of sheep nomadism: northeastern Nevada, 1870-1972. Unpubl. Ph.D. thesis in anthropology. Yale Univ.: New Haven. 374pp.

Lanner, R. M. 1980. The pinon pine: a natural and cultural history. Univ. of Nev. Press, Reno. 208pp.

Leopold, A., L. K. Sowls and D. L. Spencer. 1947. A survey of over-populated deer ranges in the United States. J. of Wildl. Manag. 11(2):162-177.

Lincoln, F. C. 1923. Mining districts and mineral resources of Nevada. Nevada Newsletter Publishing Co. Reprinted in 1970 by D. McDonald, Verdi. 295pp.

Longhurst, W. M., A. L. Lesperance, M. Morse, R. J. Mackie, D. L. Neal, H. Salwasser, D. Swickard, P. T. Tueller, P. J. Urness and J. D. Yoakum. 1977. Livestock and wild ungulates. In: Proc. of the Workshop on Livestock and Wildl.-Fish. Relation. in the Great Basin. J. W. Menke, ed. Univ. of Calif.:Berkeley. 173pp.

Lord, E. 1883. Comstock mining and miners. Govern. Print.Office: Washington. Reprinted in 1959. Howell-North: Berkeley, Calif. 451pp.

McCullough, D.R. 1987. The theory and management of Odocoileus populations. pp. 535-549. In: C.M. Wemmer, ed. Biology and management of the Cervidae. Smithsonian Instit. Press: Washington, D.C. 577 pp.

Papez, N. J. 1976. The Ruby-Butte deer herd. Biol. Bull. No. 5. Nev. Dep. of Fish and Game: Reno. 61pp.

Paul, R.W. 1963. Mining frontiers of the far west, 1848-1880. Holt, Rinehart & Winston: New York. 236pp.

Putman, R. 1988. The natural history of deer.

Cornell Univ. Press. Ithaca, N.Y. 191pp.

Robertson, J.H. and P.B. Kennedy. 1954. Half-century changes in northern Nevada ranges. J. Range. Manage. 7(3)117-121.

Rogers, G.F. 1976. Then and now: a photographic history of vegetation changes in the central Great Basin desert. Salt Lake City, Univ. Utah Press. 152pp.

Sargent, C.S. 1879. The forests of central Nevada, with some remarks on those of adjacent regions. American Journal of Science and Arts (3rd Series), Vol. 17, No. 102:417-429.

Short, H.L. 1981. Nutrition and metabolism. p. 99-127. In: Mule and black-tailed deer of North America. O.C. Wallmo, ed. Univ. of Nebraska Press: Lincoln. 605pp.

Short, S. H. 1965. A history of the Nevada livestock industry prior to 1900. Unpubl. M.S. thesis. Univ. of Nev., Reno. 105pp.

Smith, G. H. 1943. The history of the Comstock Lode, 1850-1920. Univ. of Nev. Geol. and Mining Series No. 37. Eighth printing in 1975. Nev. State Print. Office: Carson City. 305pp.

Stager, D.W. and D.A. Klebenow. 1986. Mule deer response to wildfire in great basin pinyon-juniper woodland. Pinyon-Juniper Conference. Reno. pp. 572-579.

Tausch, R. J. 1973. Plant succession and mule deer utilization in pinyon-juniper chainings in Nevada. Unpubl. M.S. thesis. Univ. of Nevada: Reno. 150pp.

_________, N. E. West and A. A. Nabi. 1981. Tree age and dominance in Great Basin pinyon-juniper woodlands. J. Range Manage. 34(4):259-64

Tueller, P. T. and L. A. Monroe. 1975. Management guidelines for selected deer habitats in Nevada. Ag. Exp. Sta., Univ. of Nev.: Reno. 185pp.

_________, C. D. Beeson, R. J Tausch, N. E. West and K. H. Rea. 1979. Pinyon-juniper woodlands of the Great Basin: distribution, flora, vegetal cover. Res. Pap. INT-229. Inter. For. and Range Exp. Sta. U.S.D.A.: Ogden, Ut. 22pp.

_________ 1979. Food habits and nutrition of mule deer on Nevada ranges. Ag. Exp. Sta., Univ. of Nev.: Reno. 104pp.

Vale, T.R. 1975. Presettlement vegetation in the sagebrush-grass area of the intermountain west. J. Range Manage. 28(1):32-36.

Welch, B. L. and J. C. Pederson. 1981. In vitro digestibility among accessions of big sagebrush by wild mule deer and its relationship to monoterpenoid content. J. of Range Manag. 34(6):497-500.

Wentworth, E. N. 1948. America's sheep trails. The Iowa State College Press: Ames. 667pp.

Yorks, T.P., N.E. West and K.M. Capels. 1994. Changes in pinyon-juniper woodlands in western Utah's Pine Valley between 1933-1989. J. Range. Manage. 47:359-364.

Young, J.A. and R.A. Evans. 1973. Downy brome--intruder in the plant succession of big sagebrush communities in the Great Basin. J. of Range Manage. 26(6):410-415.

Young, J.A. and R.A. Evans. 1978. Population dynamics after wildfires in sagebrush grasslands. J. Range Manage. 31(4):283-289.

Young, J.A. and J.D. Budy. 1979. Historical use of Nevada's pinyon-juniper woodlands. J. of Forest Hist. 23(3):112-121.

Young, J.A. and B.A. Sparks. 1985. Cattle in the cold desert. Utah St. Univ. Press: Logan, UT 255pp.

Young, O. E. 1970. Western mining. Univ. of Oklahoma Press: Norman, Ok. 342pp.

Wilson, D. 1992. Sawdust trails in the Truckee Basin: a history of lumbering operations 1856-1936. Nev. Co. Historical Soc.: Nevada City, CA. 90pp.

Zeier, C. D. 1987. Historic charcoal production near Eureka, Nevada: an archaeological perspective. Historical Archeology 21:81-101.

M. Hess, Retired Big Game Biologist
NV Division of Wildlife; Sparks, NV
hessmr@charter.net

NOTES:

Habitat Requirements of Sagebrush-Associated Species and Implications for Management

J. Kent McAdoo, Sherman R. Swanson, Brad Schultz, and Peter F. Brussard

The sagebrush (*Artemisia* sp.) region of the West encompasses approximately 155.5 million acres (Paige and Ritter 1999). More specifically, there are two major regions of sagebrush dominance: (1) the "sagebrush steppe" that covers the northern portion of the Intermountain region from eastern Washington and northern Nevada to the western two-thirds of Wyoming and northwest Colorado, and (2) the drier Great Basin sagebrush region that includes most of Nevada, parts of Utah and northern Arizona, and some areas of southwest Colorado and northern New Mexico. Although sagebrush communities have undergone much change in modern history, the boundaries of sagebrush distribution have remained fairly constant. According to Vale (1974), after examining 29 historic journals and diaries from the early 19th century, presettlement vegetation in much of the Intermountain West was visually dominated by shrubs, with much of the area covered by thick brush stands. These journals contained frequent references to the lack of herbaceous vegetation, with stands of grass typically confined to wet valley bottoms, moist canyons, and valley slopes.

Sagebrush-grass communities within these sagebrush regions vary markedly. However, these communities to one degree or another provide food, thermal cover, escape routes, rearing sites, etc. for a variety of vertebrate wildlife species (McAdoo and Klebenow 1979). Some of these species inhabit sagebrush habitats year-round, while others use them only seasonally or occasionally. There are a few species that require sagebrush for some part of their life cycle and are therefore considered "sagebrush obligates;" other species with a wider amplitude of habitat adaptation occur not only in sagebrush but in other vegetation types as well. Sagebrush communities provide habitat for approximately 100 bird species and 70 mammal species (Braun et al. 1976). Several species of lizards and snakes also inhabit these sagebrush areas (Fautin 1946).

Grazers, browsers, and seed-eaters foraging within sagebrush-grass communities may utilize the grasses, forbs, sagebrush, and/or other shrub species found there. In turn, many of these species, including ungulates, rodents, hares and rabbits, small birds, reptiles, and insects are important as prey for predatory species living in/or near sagebrush-grass communities.

Habitat Requirements of Sagebrush Obligates

According to Paige and Ritter (1999), at least 8 vertebrate species are considered to be sagebrush obligates: the sage sparrow (*Amphispiza belli*), Brewer's sparrow (*Spizella breweri*), sage thrasher (*Oreoscoptes montanus*), sage grouse *(Centrocercus urophasianus)*, pygmy rabbit (*Sylvilagus idahoensis*), sagebrush vole (*Lagurus curtatus*), pronghorn antelope (*Antilocapra americana*), and sagebrush lizard (*Sceloporus graciosus*). However, the latter species is also found in greasewood (*Sarcobatus* sp.) habitats (McAdoo and Klebenow 1979) and may not be a true "obligate." In parts of their range, gray flycatchers (*Empidonax wrightii*) and least chipmunks (*Eutamias minimus*) may also be considered sagebrush obligates. Because of a west-wide decline in sage grouse populations and habitat, the entire Intermountain West is facing the possibility that sage grouse may be considered for listing as threatened or endangered, and much political attention has therefore been focused on this species.

Sage grouse were found historically throughout most of the western United States, including portions of 16 states and along the southern border of three western Canadian provinces. This distribution closely parallels the range of sagebrush communities. The current core of sage grouse populations includes areas of Colorado, Idaho, Montana, Nevada, Oregon, and Wyoming, with remnant populations in other states. Sage grouse require sagebrush for food and/or cover during each stage of their life cycle and are therefore "sagebrush obligates." Although sage grouse depend on sagebrush vegetation for survival, they thrive best in areas with a mosaic of sagebrush species, age, and cover classes. Optimal habitat is a diverse mosaic of sagebrush-grass with varying heights of sagebrush and a diverse understory of perennial grasses and forbs (broadleaf herbaceous plants). The proportion of sagebrush, perennial grasses, and forbs in an area varies with the species or subspecies of sagebrush, the ecological potential of the site, and condition of the habitat (Klebenow 2001). During the course of a year, sagebrush is quantitatively the most important component in the diet of sage grouse, comprising 60 to 80% of all food consumed. However, during spring and summer these birds shift from a sagebrush-dominated diet to one of forbs and insects (Klebenow and Gray 1968).

Habitat requirements of the other sagebrush obligates are variable. Sage sparrows, Brewer's sparrows, and sage thrashers all require sagebrush for nesting, with nests typically located in the sagebrush canopy. Sage thrashers typically nest in tall dense clumps of sagebrush within areas having some bare ground for foraging. Sage sparrows prefer large continuous stands of sagebrush habitat, and Brewer's sparrows are closely associated with sagebrush habitats having abundant scattered shrubs and short grass (Page and Ritter 1999). Pygmy rabbits are associated with clumps of tall sagebrush in friable soils, whereas pronghorn antelope are more typically associated with lower growing sagebrush presumably because of their keen eyesight adaptation for detecting danger at long distances. Like sage grouse, both pygmy rabbits and pronghorns may eat sagebrush almost exclusively during winter (Page and Ritter 1999). However, pronghorns depend primarily on forb species for much of the year.

Habitat Requirements of Other Sagebrush-Associated Species

A wide variety of other bird species are associated with sagebrush grass communities, and their habitat requirements are quite variable. Some species, like loggerhead shrikes (*Lanius ludovicianus*) nest primarily in the canopy of sagebrush and other shrubs. Others are primarily open ground and/or grass nesting species, requiring varying amounts of herbaceous cover. Such species include horned larks (*Eremophila alpestris*), vesper sparrows (*Pooecetes gramineus*), and western meadowlarks (*Sturnella neglecta*). Other species, like lark sparrows (*Chondestes grammacus*), are typically most abundant in areas with a diverse mixture of sagebrush and bunchgrass (McAdoo and Klebenow 1989). Horned larks and burrowing owls (*Athene cunicularia*) are adapted to more open areas, with both species often increasing after wildfire or other sagebrush-canopy reduction impacts.

In addition to pronghorn antelope, other ungulate big game species are dependent on sagebrush-grass communities to some extent. Mule deer (*Odocoileus hemionus*) are closely associated with sagebrush-grass communities in much of their range. Being primarily browers, any successional vegetation changes favoring shrubs may benefit mule deer populations. Many forbs and shrubs associated with sagebrush communities are important in mule deer diets, with grasses used primarily in spring. Forb use is highest in summer, and on many mule deer ranges, big sagebrush is the staple component in winter and early spring (Kufeld et al.1973). Elk (*Cervus elaphus*) use sagebrush-grass communities over portions of their range. In some locations, use of these areas occurs primarily during winter, but in parts of the Great Basin these areas are used in other seasons as well. Elk are generally dependent upon grasses for forage throughout much of their range, but they will also eat shrubs, including big sagebrush, especially during fall and winter (Kufeld 1973). Sagebrush grass communities are used by bighorn sheep (Ovis *canadensis* spp.) in some areas, especially as winter range. Although grasses are typically the major component in the bighorn sheep diet, shrubs are important, and big sagebrush is a preferred shrub (McQuivey 1978).

Five species of hares and rabbits may occur in sagebrush-grass communities. Of these, the most common in most areas is the black-tailed jackrabbit (*Lepus californicus*), an opportunistic feeder that selects for succulence. Blacktailed jacks eat primarily grasses and forbs until winter, when they feed on shrubs, including the leaves and bark of big sagebrush. During cyclic population highs, this species can cause considerable damage to rangeland vegetation and cultivated crops (McAdoo et al. 1987). Within sagebrush-grass habitats, blacktailed jackrabbits are typically associated with increasing shrub cover, whereas whitetailed jackrabbits (*L. townsendii*) are associated with increasing grass cover (Verts and Carraway 1998). Pygmy rabbits have already been identified (above) as sagebrush obligates. Two other rabbit species, desert cottontails (*Sylvilagus audubonii*) and mountain cottontails (*S. nuttallii*) are also found in some sagebrush-grass habitats.

Many rodents (at least 28 species) inhabit sagebrush-grass communities, with the deer mouse (*Peromyscus maniculatus*) being typically most common. Unlike the sagebrush vole that was mentioned above as a sagebrush obligate, deer mice occur in a wide variety of vegetation types. Great Basin pocket mice (*Perognathus parvus*) are restricted primarily to sagebrush habitats in some areas (McAdoo and Klebenow 1979). Most rodent species are herbivores and granivores, but specific habitat affinities for each species are quite variable. Rodent species in general have a reputation for negative impacts on rangelands, but some species can be quite beneficial in terms of seed dispersal and germination (McAdoo et al. 1983).

Management Implications

Habitat requirements for the many wildlife species in sagebrush-grass communities is obviously variable by species and even by season of use for many species, as described above. Before European settlement, "spotty and occasional wildfire probably created a patchwork of young and old sagebrush stands across the landscape, interspersed with grassland openings, wet meadows, and other shrub communities" (Paige and Ritter 1999). In drier regions, such as central Nevada, fire likely had less of an influence. The wildlife sightings of early explorers were a function of landscape ecology, season, and time interval since the last fire. Species like sage grouse seemed to be locally abundant, but regionally rare. According to Miller and Eddleman (2001), the range occupied by sage grouse is spatially diverse and temporally dynamic. By inference, since sage grouse distribution closely parallels the range of sagebrush communities in North America, the same principle holds for all sagebrush-associated vertebrate wildlife species. These spatial and temporal variables influence wildlife abundance, distribution, and diversity.

We can draw some inferences from the effects of sagebrush-grass community alteration on neotropical migrants (songbirds). Published research conducted in northern and central Nevada during the 1980s showed that sagebrush removal from large acreages had initially negative impacts on shrub-nesting birds, especially sagebrush obligates such as sage thrashers, sage sparrows, and Brewer's sparrows. In those areas where crested wheatgrass (*Agropyron desertorum*)) was planted after shrub removal, a corresponding increase was observed in ground and grass nesting species like horned larks, western meadowlarks, and lark sparrows. However, as successional establishment of sagebrush occurred in these areas over time, shrub-nesting bird species returned and grass-nesting species remained. Bird species diversity increased as the complexity of the plant community increased (McAdoo et al.1989).

What are the implications of these bird population responses for wildlife species in general as related to sagebrush habitat management? Much of the Intermountain West contains large expanses of sagebrush habitat where shrub cover is so dominant that herbaceous cover is almost absent, with only sparse populations of remnant native grasses and forbs. To improve the site productivity of these areas for seasonal use of such high profile species like sage grouse, proposals have been made to manage portions of these areas for reduction of mature sagebrush cover, regeneration of young sagebrush, and increased native herbaceous cover. We hypothesize that if such management strategies were carefully implemented in a mosaic fashion, a continuum of herbaceous, herbaceous-shrub, shrub-herbaceous, and shrub dominated habitats could be created. We hypothesize that most wildlife species on a landscape scale would be largely benefited. Because of the diverse habitat requirements of various wildlife species, habitat for all sagebrush-associated wildlife species would be present in varying amounts on a landscape scale. In other words, creating a mosaic of habitats with multiple-aged stands of sagebrush and varying degrees of herbaceous and shrub cover would provide both the vertical and horizontal vegetation diversity components required by diverse wildlife species.

The value of each landscape parcel for various wildlife species would change over time as the dynamics of natural or prescribed disturbance and secondary plant succession occur. State and transition models, imbedded into ecological site descriptions, offer the best tool for analyzing vegetation management options and priorities. The highest priorities for habitat treatments should be driven by the risk of crossing an ecological threshold (such as weed invasion) and the opportunity to apply an effective management tool (McAdoo et al. *Submitted*). Implementation of adaptive management strategies, including follow-up monitoring and adjustment of strategies if necessary, will ensure the perpetuation of a diverse and productive landscape. Success in establishing mosaics of native plant communities also complements sustainable rangeland management for multiple uses in addition to wildlife.

References:

Braun, C.E., M.F. Baker, R.L. Eng, J.S Gashwiler, and M.H. Schroeder. 1976. Conservation committee report on effects of alteration of sagebrush communities on the associated avifauna. Wilson Bull. 88:165-171.

Fautin, R.W. 1946. Biotic communities of the northern desert shrub biome in western Utah. Ecol. Monogr. 16(3):251-310.

Klebenow, D.A. 2001. Enhancing sage grouse habitat…a Nevada landowner's guide. Northwest Sage Grouse Working Group Publication.

Klebenow, D.A. , and G.M. Gray. 1968. Food habits of juvenile sage grouse. J. Range Manage. 21:80-83.

Kufeld, R.C. 1973. Foods eaten by Rocky Mountain Elk. J. Range Manage. 26:106-113.

Kufeld, R.C., O.C. Wallmo, and C.Feddema. 1973. Foods of the Rocky Mountain mule deer. USDA For. Serv. Res. Pap. RM-111. 31pp.

McAdoo, J.K., and D.A. Klebenow. 1979. Native faunal relationships in sagebrush ecosystems. Pages 50-61 in The Sagebrush Ecosystem Symposium. Utah State Univ., Logan, UT.

McAdoo, J.K., C.C. Evans, B.A. Roundy, J.A. Young, and R.A. Evans. 1983. Influence of heteromyid rodents on *Oryzopsis hymenoides* germination. J. Range Manage. 36:61-64.

McAdoo, J.K., W.S. Longland, G.J. Cluff and D.A. Klebenow. 1987. Use of new rangeland seedings by black-tailed jackrabbits. J. Range Manage. 40:520-524.

McAdoo, J.K., W.S. Longland and R.A. Evans. 1989. Nongame bird community responses to sagebrush invasion of crested wheatgrass seedings. J. Wildl. Manage. 53:494-502.

McAdoo, J.K., and S.R. Swanson. 2003. Impacts of Sagebrush Habitat Treatments on Wildlife. *In:* Proceedings of the 12th Wildland Shrub Symposium, Seed and Soil Dynamics in Shrubland Ecosystems. ***Submitted.***

McQuivey, R.P. 1978. The desert bighorn sheep of Nevada. Nev. Dep. Fish and Game. Biol. Bull. No. 6. 81pp

Miller, R.F., and L.L. Eddleman. 2001. Spatial and temporal changes of sage grouse habitat in the sage grouse biome. Oregon State Univ. Agric. Exp. Sta. Tech Bull. 151. 35pp.

Paige, C. and S.A. Ritter. 1999. Birds in a sagebrush sea: managing sagebrush habitats for bird communities. Partners in Western Flight working Group, Boise, ID.

Vale, T.R. 1975. Presettlement vegetation in the sagebrush-grass area of the Intermountain West. J. Range Manage. 28:32-36.

Verts, B.J., and L.N. Carraway. 1998. Land mammals of Oregon. Univ. of Calif. Press, Berkeley

J.K. McAdoo, Rangeland Resources Specialist
Univ. of Nevada Cooperative Extension; Elko, NV
mcadook@once.unr.edu

S.R. Swanson ,Professor of Range Science
Univ. of Nevada-Reno; Reno, NV

B. Schultz, Extension Educator
Univ. of Nevada Cooperative Extension;
Winnemucca, NV

P.F. Brussard, Biology Professor
Univ. of Nevada-Reno; Reno, NV

NOTES:

Weed Problems on Great Basin Rangelands

James A. Young and Charlie D. Clements

A common definition of what is a rangeland weed is a plant whose presence on a given range site interferes with the sustainable economic and ecological function of that site. Rangeland weeds can be both **exotic** and **native** species. They can be naturally **self invasive**, meaning they establish and spread without the conscious efforts of humans, or they may be specie obligate to the activities of humans. If a weed is particularly predaceous it may be designed by state or federal agencies as a legally **noxious** weed. Some governing bodies grade noxious weeds according to their perceived ecological and economic hazardous nature. The same plant species can change categories of weeds in different range sites and/or geographic locations. Gum weed (*Grindelia squarrosa*) is a common ruderal species. In Colorado it is considered a native species. In California, it is considered an exotic, invasive species. Utah juniper (*Juniperus osteosperma*) and single leaf-pinyon (*Pinus monophylla*) are co-dominant species of the conifer woodlands of the Great Basin. For much of the 20th century these two tree species expanded their range and gained dominance in formerly shrub/bunchgrass communities. This expansion of range may be a result of climatic changes or human interventions in the environment, but the invasion by these native tree species changes the sustainable ecological function of the former shrub/bunchgrass communities.

The weeds of the Great Basin that have attracted the most attention are exotic, self invasive species that change the aspect of rangelands. Cheatgrass (*Bromus tectorum*) is the most widely recognized of these species because it truncates succession, inhibiting assumption of dominance by native perennial species. In the terms ecologist use that study invasive species, cheatgrass is a **transformer** species in that it changes ecological processes. Although, it may appear a permanent dominant feature of Great Basin rangelands, cheatgrass is an extended dwelling point in a successional process among exotic species on Great Basin rangelands. Succession among exotic annuals on Great Basin rangelands begins with halogeton (*Halogeton glomeratus*), Russian thistle *(Salsola targus*), or barbwire Russian thistle (*S. paulsenii*). These are species that establish on severely disturbed, bare ground type seedbeds. The next level of succession is characterized by members of the mustard family such as tumble mustard (*Sisymbrium altissimum*) or shield-cress (*Lepidium perfoliatum*). Finally, cheatgrass emerges as the annual dominant after these exotic annual stages of succession. On different sites and on specific years, cheatgrass communities may contain some of the species from lower successional stages, but other exotic species are also represented. These include red stem filaree (*Erodium cicutarium*), the first exotic annual to be introduced to the Great Basin and the only exotic annual that was extensively intentionally spread by humans. Prickly lettuce (*Lactuca serriola*) is another late 19th century introduction that occasionally shares dominance with cheatgrass. Bur buttercup (*Ranunculus testiculatus*) is a much more recent introduction that has spread in many cheatgrass communities.

Cheatgrass communities are largely closed to the establishment of seedlings of native, but is notoriously open to invasion by new exotic species. Medusahead (*Taeniatherum caput-medusae*) is the prime example. Medusahead invasion of cheatgrass dominated sites is largely restricted to specific soils. If soil conditions are favorable, medusahead replacement of cheatgrass is surprisingly complete and on a landscape scale.

There are several biennial species that have successfully invaded cheatgrass dominated communities. These include skeleton weed (*Chondrilla juncea*), dyer's woad (*Isatis tinctoria*) (the species from which the word weed was derived), Scotch thistle (*Onopordium acanthium*) and Mediterranean sage (*Salvia aethiopsis*). The genus of weeds that probably will have there greatest impact on Great Basin rangeland is *Centaurea.* This group includes yellow starthistle (*C. solstitialis*), diffuse knapweed (*C. diffusa),* spotted knapweed (*C. maculosa*), tocalote (*C. melitensis*), and squarose knapweed (*C. squarrosa*). These species range in life from annuals such as yellow starthistle, to biennials such as spotted knapweed, to variable life form species such as diffuse knapweed, and squarose knapweed which is a tap rooted perennial.

Exotic perennial weeds that will become landscape scale herbaceous dominants on Great Basin rangelands are still in the probable stage. Farther north and in the Great Basin on specific, more mesic sites, leafy spurge fits the description as a landscape scale perennial weed. Two potentially high successional transformer species are Russian knapweed (*Acroptilon repens*) and perennial pepperweed *(Lepidium latifolium*). Both of these species are currently thought of as weeds of wetlands, riparian areas, and crop land. Colonies of these species are becoming increasingly frequent and larger in scale at even the more arid portions of the Great

Basin.

J.A. Young, Range Scientist
C.D. Clements
USDA ARS; Reno, NV
jayoung@scs.unr.edu

NOTES:

Soil/Sagebrush Correlations in Nevada

Gary Medlyn

Throughout the range of the sagebrush/ grasslands there are a series of plant communities delineated by the dominant shrub species and the understory6 grass species. To those unfamiliar with them, this network of species may appear to be a bewildering array of variability. In fact, however, the plant communities are repetitive and easily identifiable. Recognizing them is important because they are photometers, or living measurements, of a given local ecosystem compose of soil, topography, climate, animals and the plants themselves. (Young, 2002)

Soil is a natural body on the earth's surface in which plants grow. It is a mixture of varying proportions of rocks, minerals, organic matter, water, and air. The rocks and minerals are fragmented and partly or wholly weathered. Soils have distinctive layers, or horizons, that are the products of environment forces acting upon materials deposited or accumulated though geologic activity. Soil formation is a function of parent material, climate, time, relief and biological all working together differentially.

The characteristics of a soil at any given moment are determined by the interaction of the parent material. Parent material is the weathered rock of unconsolidated material from which soils form, the hardness, grain size, porosity, and weatherable mineral content of the parent material greatly influence soil formation.

The climate in which the soil material accumulated and has since existed; the biological forces that act upon the soil material; the relief which influences the local environment of the soil, its drainage, moisture content, aeration, stability, and exposure to sun and wind; and the length of time that climate, biological factors and relief have acted upon the parent material.

Land surfaces are not always stable long enough to permit the development of well-expressed indicators of soil formation. This is particularly evident in the cold desert region. Deep incisions in the earth reveal sequences of buried soil that have varying degrees of development and overlying geologic material. Each buried soil indicates a period of relative stability, although short lived, during which the soil forming processes had begun to leave their mark.

During this presentation, we will explore some of the soils characteristics and their corresponding general correlations to various sagebrush grasslands communities in Nevada.

Wyoming Big Sagebrush

Wyoming big sagebrush communities occur without respect to parent materials. They are expressed differently however due to precipitation, and depth to a soil root-limiting horizon. Where soils are deep, the Wyoming big sagebrush is well-expressed and about three feet in height. Where there are root-limiting layers Wyoming big sagebrush can be stunted to a height of 1 foot depending on the depth to the root-limiting layer. Understory species are responsive to climatic factors. As soil chemistry becomes more sodic or saline, codominant shrubs of greasewood are found in the community. In the central parts of Nevada in the 8 to 10 inches of precipitation zones Indian ricegrass generally is dominant, but in the 10 to 12 inch precipitation zone the understory is dominated by bluebunch wheatgrass. With increase in latitude, Bluebunch wheatgrass is also dominant in the 8 to 10 inch precipitation zone. Wyoming big sagebrush does not usually growth above the precipitation zones greater than 12 inches.

Black sagebrush

Black sagebrush grass communities occur generally on soils that are derived from parent materials formed from limestone or dolomite and that have some type of root-restrictive layer within 20 inches or less of the surface. There are, however, exception due to secondary carbonate deposition and enrichment on soils derived from noncalcareous parent material sources. Associated understory grasses follow a similar climatic precipitation and latitude response pattern to those of Wyoming big sagebrush. Black sagebrush grows on shallow or very shallow soils where precipitation exceeds 12 inches. This usually occurs on south facing slopes on mountain side slopes.

Basin Big Sagebrush

Basin big sagebrush is a found on soil that are deep or very deep (greater than five feet) and that have a high amount of water holding capacity. These soils are generally free of coarse fragments (gravel and cobbles) and frequently occur in drainage ways or along water courses where there is supplemental ephemeral water. The dominant understory grasses occurring in these communities in Great Basin wild rye. Soils in these communities are highly productive and have well-developed organic matter enriched surface horizons.

Mountain Big Sagebrush

Mountain big sagebrush occurs where precipitation is 11 inches or higher. Soils in this site are usually moderately deep to deep and well drained. Soils commonly are high in rock fragments with gravelly of cobbly surfaces. Soils in these communities are highly productive and have well-developed organic matter enriched surface horizons. Understory grasses are usually abundant and include bluebunch wheatgrass, and /or Great Basin wild rye. Shrubs may include antelope bitterbrush, snowberry, choke cherry or Mountain Mahogany. Where mountain big sagebrush occurs on shallow soils with root-limiting layers, it has a stunted appearance or growth form.

Low Sagebrush

Low sagebrush has a strong correlation to soils that are well drained and have a shallow effective rooting depth. A combination of impermeable bedrock at a shallow depth and the swelling of the heavy textured subsoil when wet, results in poor soil aeration and a perched water table within the root zone during the spring. This sagebrush usually occurs at precipitation zones of 12 inches or greater. Soil parent materials include quartzite, rhyolite, granite or andesite. Soils have a high percentage of gravels, cobbles , rocks or stones on the soil surface. The most common understory grass is bluebunch wheatgrass.

Lahonton Sagebrush

Lahontan Sagebrush occurs mostly in northwestern Nevada and in adjacent areas of California and Oregon at elevations of 3300 to 6700 feet. At lower elevation it is associated with salt desert shrub species such as shadscale, bailey greasewood, bud sagebrush. Precipitation ranges from 5 to 12 inches. Soils have low available water-holding capacities and a shallow depth to an argillic horizon and /or bedrock. These soil are similar to those of low sagebrush.

Pigmy Sagebrush

Pigmy sagebrush occurs on fan piedmonts. Elevations are 5000 to 7000 feet. Average annual precipitation is 8 to 12 inches. Soils have an affective rooting depth less than 20 inches. Soils ar4e derived from mixed alluvium, lake bed sediment oreroded sedimentary material. Soil surfaces are usually gravelly and the textures are gravelly sandy loams to loams. These sites are usually found in association with black sagebrush.

References:

Field Office Technical Guide, Section IIe, “Range Site Descriptions for Nevada”, USDA-NRCS, Reno, NV. 1991.

Soil Survey of Western White Pine County Area Nevada. parts of White Pine and Eureka Counties. USDA-NRCS. 1995.

Winward, Alan H., and E. Durant McArthur, Lahontan, Sagebrush (*Artemisia arbuscula ssp. Longicaulis*) a New Taxon. Great Basin Naturalist 55(2). pp. 151-157. 1995.

Young, James A. and B. Abbot Sparks, Cattle in the Cold Desert, University of Nevada Reno Press (Originally published 1985 at Utah State University Press). 2002

G. Medlyn, Soil Scientist
USDI, BLM; Ely, NV
Gary_Medlyn@nv.blm.gov

NOTES:

Application of Sage Grouse Guidelines to Sage Grouse Habitats in Nevada

Gary N. Back, Barry L. Perryman, and J. Kent McAdoo

The heightened awareness of the status of sage grouse (*Centrocercus urophasianus*) in the West has lead to regional, state, and local planning efforts to increase populations and improve habitat quality. The vast majority of the land in the Intermountain West, including the Great Basin, is administered by public land management agencies. These agencies have entered into a Memorandum of Understanding (MOU) with the Western Association of Fish and Wildlife Agencies (WAFWA), which includes application of the *Guidelines to manage sage grouse populations and their habitats* (Connelly et al. 2000). These guidelines were developed for a region that extends from eastern Colorado to western California and from southern British Columbia to southern Nevada. This area contains many different soil types, land forms, species and subspecies of sagebrush, and vegetation communities, as well as different climatic conditions such as the amount, form, and timing of precipitation, temperature, and growing seasons. As a result, the pre-European contact disturbance regimes, and therefore, the ecological succession processes, were likely to be different across this vast region. Consequently, we view the guidelines developed by Connelly et al. (2000) to be a starting point in management considerations, not the final basis for management decisions. The guidelines include provisions for incorporating local ecological expertise (page 975). This presentation is an example of how the local ecological knowledge can be combined with the guidelines to benefit sage grouse and sage grouse habitats within the multiple use management policy for public lands.

This presentation views the guidelines of Connelly et al. (2000) from a habitat and plant community succession perspective, incorporating the concepts of dynamic changes in plant communities on the landscape with thresholds, that when crossed, result in long-term habitat degradation. Land managers must be aware of these thresholds if they are to manage plant communities to avoid crossing these thresholds, and to be able to restore areas where thresholds have been crossed. In keeping with the theme of this workshop, we focus on the guidelines that address specific habitat protection and habitat restoration topics. Guidelines that address non-habitat issues, such as fences or power transmission lines, are not included in this review.

Although the linear successional model developed by Clemens (1916) has been the foundation for ecological theory and understanding, the Clemensian model of succession is being modified to recognize additional dynamics of plant communities (Laylock 1991, and 1995, West 1999). The unidirectional progression from bare mineral soil to a climax community can be redirected by climatic events, human intervention (planned or unplanned), natural events (e.g., fire), and introduction of new plants into the community (e.g., non-native, invasive plants). These forces can change the succession trajectory or transition process, resulting in a new plant community state. For example, a given range site that would "normally" respond to a wildfire with an initial flush of herbaceous plants that would be invaded by native shrubs over time and eventually become shrub-dominated (i.e., the unidirectional, linear successional model). However, this site could convert to an exotic annual grassland following fire if the seed source of annual plants is present and the fire intensity is sufficient to kill the perennial herbaceous plants and the soil seedbank. This conversion from a native plant community to annual grassland is the result of crossing a threshold. The annual grassland becomes a steady state with a different disturbance regime. The transition of the annual grassland state to perennial noxious weeds is possible, given the right combination of seed source, management, and natural events, all factors that represent crossing another threshold. Restoring lands that cross a threshold requires more energy and more intensive management practices than managing lands that have not crossed a threshold.

The guidelines include provisions for habitat protection of breeding habitat, summer-late brood habitat, and winter habitat. These guidelines do not specifically recognize the dynamic nature of plant community succession (or state and transition models) on the landscape. Successional stages that are unsuitable today (e.g., a recently burned area) but managed properly, can provide brood habitat in a decade or more after sagebrush re-establishes on the site. As the shrubs increase in size and density, the area will begin to provide nesting and winter habitat. As shrub dominance increases, the inter-specific competition will cause the herbaceous understory to decline in cover and plant density (Winward 2000), and nesting habitat is no longer provided. But the plant community will still provide winter habitat. Left "unmanaged," the inter-specific competition will eventually deplete the herbaceous understory, creating the opportunity for a threshold to be crossed. Eventually, intra-specific competition occurs as the sagebrush stands become decadent, and sagebrush seedlings in the understory become infrequent, possibly leading to altered fire regimes and potential invasion by undesirable plant species. Perpetual protection can lead to habitat degradation.

The appropriate time to interject a management treatment to move the community into a herbaceous dominated stage, initiating the process again, is before the herbaceous understory is sufficiently depleted to allow the threshold to be crossed. Therefore, an area that does not cross a threshold following fire (natural or prescribed), should serve as at least one of the three seasonal habitats for sage grouse for the next 20 to 30 years after shrubs are established.

Managers must be vigilant and not give in to the tendency of relegating habitat to a location instead of to a plant community successional or transitional condition. The guidelines imply that sage grouse will use areas forever and that these areas should be protected in perpetuity, a position contrary to basic ecological principles. Habitats occur on a continuum of successional stages, and as each community reaches a stage that provides certain cover and forage conditions, it will be used by sage grouse or other wildlife for a specific seasonal requirement. Nesting, brood-rearing, summer, and winter habitats will change spatially over any extended time frame whether or not the area undergoes disturbance. Protection, rather than management, will lead to a lack of brood-rearing and nesting habitat and an abundance of winter habitat (probably low quality) over an extended period of time. Consequently, the landscape should be managed to provide a variety of plant community stages, instead of being protected.

The guidelines recommend not treating more than 20 percent of the breeding habitat within a 30-year period, because the "30-year period represents the approximate recovery time for a stand of Wyoming big sagebrush." The management of a site should be based on the ecological site potential. Some sites have the capacity to respond more or less quickly after disturbance depending on the productivity of the site (i.e., a combination of soil, climate, and position on the landscape), the intensity of the disturbance (i.e., severity of fire in terms of impact to perennial plants and seed bank), the pre-disturbance condition of the plant community (i.e., the successional or transitional state), and the post-disturbance management. The guidelines need to be applied on a site- and situation-specific basis, rather than relying on a specific time interval. "Rule of thumb" management guidelines often limit good management applications.

The guidelines, while not ruling out the use of prescribed fire, certainly do not encourage the use of prescribed fire. Again, the site- and situation-specific conditions should be the basis for making management decisions. All range sites will not respond to fire in the same way. Sites in the more arid parts of Nevada or Utah may not support sufficient plant cover for fire to be used, or to have occurred, under all but extreme conditions. Alternatively, in northern Nevada, certain range sites are likely to respond well to fire that is conducted under the right set of conditions. However, dense stands of sagebrush with little or no understory burned extensively and intensively in northern Nevada in the past five years, crossing thresholds that have increased the acreage of annual grasslands in Nevada by hundreds of thousands of acres. Mature stands of sagebrush with high canopy cover have greater potential for high intensity fire than open stands of sagebrush with a healthy complement of herbaceous plants.

These same concepts apply to habitat restoration. Whether conducting burned area rehabilitation or restoration of degraded rangelands, the spatial and temporal component of plant communities must be considered in the land management plans. Recognition that plant community dynamics provide the seasonal habitat needs of sage grouse and other wildlife should be the cornerstone of these restoration activities. Reseeding a 15,000-acre burn or similarly sized annual grassland to sagebrush creates a large area that will generally provide only one seasonal habitat component at any given time. The initial establishment of shrubs should be conducted spatially to create "islands" of sagebrush that will expand under the proper conditions, creating the temporal and spatial aspects of the habitat mosaic.

Managers may need to take a long-term view of restoration of annual grasslands or lands dominated by noxious weeds. Converting directly from these degraded conditions to a sagebrush-bunchgrass community may not be possible, or at least may not be cost-effective. A multi-phased restoration that uses non-native species may be necessary. The first step is to treat the annual grass or noxious weed to a level that allows perennial species to establish. Aggressive species, such as crested wheatgrass, can be used to create a temporary, short-term (i.e., 30 years or less) perennial grass community that is open to overseeding with native perennial grasses and forbs, as well as sagebrush. The initial short-term plant community would not be considered sage grouse habitat, but examples exist of crested wheatgrass seedings that have been invaded by sagebrush and have had native perennial grasses and forbs reestablish over time. These areas are, being used by sage grouse for nesting and early brood habitat. Once the native grasses and forbs have been established, the area can be managed similar to other sagebrush-bunchgrass communities.

In summary, the following points need to be included in the site-specific and situation-specific use of the sage grouse guidelines:

- ∉ Knowledge of ecological status and trend are both necessary for making habitat management decisions;
- ∉ Successional stages that are unsuitable sage grouse habitat today, may become, under proper management, seasonal habitats for sage grouse in the future;
- ∉ Perpetual protection of sagebrush plant communities can lead to habitat degradation;
- ∉ Landscapes should be managed to provide a variety of plant community stages;
- ∉ Different sites have different capacities to respond to disturbance – know the site potential and current ecological site status before conducting vegetation treatments;
- ∉ Rehabilitation and restoration efforts should consider the temporal and spatial components of sage grouse habitat;
- ∉ Instant habitat creation is not always possible or cost effective; a longer-term perspective may be required to reach specific management goals; and
- ∉ If a given treatment (i.e., management tool) does not achieve the desired results, don't eliminate the tool; assess the way the tool was used, the conditions under which it was used, the time of year, etc. to learn why the desired result was not achieved.

References:

Clements, F.E. 1916. Nature and structure of the climax. J. Ecology 24:252-284.

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

Laycock, W.A. 1991. Stable states and thresholds of range condition on North American rangelands – a viewpoint. J. Range Mgt 44:427-433.

____. 1995. New perspectives on ecological condition of rangelands: can state and transition or other models better define condition and diversity. Pages 140-164 *in:* L. Montes and G.E. Olivia, eds. Proceedings – international workshop on plant genetic resources, desertification and sustainability. Centro Regional Patagonia Sur, INTA-EEA, Rio Gallegos, Santa Cruz, Argentina.

West, N.E. 1999. Synecology and disturbance regimes of sagebrush steppe ecosystems. Pages 15-26 *in:* P.G. Entwistle, A.M. DeBolt, J.H. Kaltenecker, and K. Steenhof, complilers. Proceedings: Sagebrush Steppe Ecosystems Symposium. Boise State University, Boise,Idaho.

Winward, A.H. 2000. Sagebrush Ecology. Presentation to the Northeastern Nevada Stewardship Group, Inc. Sagebrush Symposium. Elko, Nevada.

G. N. Back, Principal Ecologist
SRK Consulting; Elko, Nevada
gback@srk.com

B. L. Perryman, Assistant Professor
Dept. of Animal Biotechnology, Univ. of Nevada-Reno- bpsaluyi@mail.ag.unr.edu

J. Kent McAdoo, Rangeland Resources Specialist
Univ. of Nevada, Cooperative Ext. Service; Elko, NV
mcadook@unce.unr.edu

NOTES:

Using Concepts Behind State and Transition Models to Improve the Decision Making Process for Restoration Efforts on Sagebrush Rangelands

Brad Schultz and Sherm Swanson

Identifying sagebrush rangelands that require management treatments to initiate or sustain recovery requires that managers integrate their knowledge about existing species composition, the response of these species to various management treatments (based on their life history requirements and strategies), and the area's site potential and response potential. A landscapes existing plant composition (both species and lifeforms) tells us much about its past, and provides a glimpse about its future. The species on a site are the immediate capital available to a land management specialist. Each species and lifeform responds differentially (to one degree or another) to each potential management action; thus, a wide range of potential plant communities (defined by both presence and abundance) are possible. The current community's potential response, however, is modified by an area's site potential. That is, the site's ability to produce a specific suite of species, in specific proportions, and with in a specific range of total biomass. Differences in site potential result from a composite of many environmental and biological interactions. The response potential of a given site depends largely on the composition of the existing plant community at the time of the disturbance or biophysical event; how the individual species are likely to respond to the disturbance or event; the disturbance's frequency, intensity, and/or duration; the influence of weather and climatic patterns both before and immediately after the disturbance, and the inherent potential of the site's topography and soil to modify the effects of weather, particularly the availability of soil moisture.

The assessment of Range Condition and Ecological Status are two approaches commonly used to describe "rangeland condition". Range condition data (e.g., fair, good, mid-seral, late seral) are spatial data and are easily stored and displayed in a Geographic Information System as distinct polygons. Data from a number of adjacent polygons provide a pattern of "range condition" across a landscape. When "range condition" in an area of interest is less than desired (i.e., poor/early seral or fair/mid-seral) managers often propose range improvements and/or changes in management. Likewise, when condition class assessments indicate good or excellent range condition (i.e., late seral or Potential Natural Community, respectively), land managers seldom propose range improvements or changes in management. Numerous range management professionals have reviewed the use of "range condition" data to make management decisions and have identified numerous problems with the concept. A serious, but often overlooked problem, is that the same condition class (e.g., good or late-seral) can have very different species compositions. For example, most if not all sagebrush ecological sites in the sagebrush semi-desert can be dominated by either shrubs (largely sagebrush) or perennial grasses when classified as late-seral (i.e., good range condition). Sites dominated by shrubs have very different response potentials than do sites inhabited primarily by perennial grasses.

The transformation of quantitative composition data (e.g., percent composition by species) into qualitative condition class (e.g., late seral or good), with very different lifeform compositions (e.g., shrub dominated vs. grass dominated) within the same condition class, can result in missed management opportunities for future vegetation change. Likewise, potential hazards for undesired vegetation change are not apparent. For example, a large landscape with predominately sagebrush in the overstory, and relatively few desired grasses and forbs in the understory, can classify as late-seral (i.e., good condition). Many interpret this condition rating as desirable, and conclude there is no need for management intervention. This conclusion may miss a management opportunity to decrease shrubs and increase the desired herbaceous component, provided sufficient perennial grasses and forbs remain in the understory. This same "good condition" rating may mask management hazards if the desired herbaceous component has declined too far. The loss of too many of the desired grasses and forbs prevents their increase after a disturbance that removes the shrubs, and leaves the site open to occupation by invasive and/or noxious weeds. Some authors have developed and/or endorsed state and transition models as: 1) an alternative approach to assess the status of vegetation on rangelands, and 2) to develop decision support systems to apply management treatments and actions. State and transition models are useful tools that can aid the decision making process. The word "model", however, often creates unnecessary barriers that thwart the adoption of important concepts about vegetation states, change (transition) among states, and ecological thresholds for vegetation change. For land managers it is more important to understand the concepts behind state and transition models, not the mechanics or intricacies of model development. Among the important concepts are: 1) any ecological site on a landscape may produce numerous vegetative states (plant communities); 2) transitions are directions of vegetation change from one state to another; 3) a given vegetative state can change (transition) directly to another state, 4) some states are achieved only by a series of transitions (changes) through one or more states; 5) some states are resistant and/or resilient to change: others are not; 6) transitions may be reversible or irreversible; 7) irreversible transitions cross ecological thresholds (boundaries between states), and the prior state cannot be returned to without intensive (and often expensive) management inputs; 8) thresholds are points of irreversible change, due to the degradation, undesired alteration, and/or removal of critical ecological processes; 9) transitions across thresholds are often triggered by specific events or combinations of events that alter plant composition, vegetation structure, site potential, and/or ecological processes; 10) trigger events may be simple (e.g., fire); may occur under specific conditions (e.g., high intensity fire); or may be complex (e.g., fire followed by heavy spring grazing); and 11) the potential effect of a trigger event typically depends on the size, timing, intensity, frequency, and duration of the event, as well as post-event management.

Integrating knowledge about potential vegetation states, probable transitions, and the conditions under which each transition operates should allow managers to produce a description of potential management opportunities, as well as management hazards. Clearly identified management opportunities and hazards, combined with knowledge about the conditions required to transition between various states, should result in better management decisions. Finally, the acceptance of the concepts about alternative vegetation states and transitions between states fits well with the approach of adaptive management.

References:

Allen-Diaz, B. and J. W. Bartolome. 1998. Sagebrush-grass vegetation dynamics: comparing classical and state-transition models. Ecological Applications 8:795-804.

Ellis, J. E. and D. W. Swift/ 1988. Stability of African pastoral ecosystems: alternate paradigms and implications for development. J. Range Mg. 41:450-9

Friedel, M. H. 1991. Range Condition assessment and the concept of thresholds: a viewpoint. Journal of Range Management 44:422-26.

Jackson, R. D., J. W. Bartolome, and B. Allen-Diaz. 2002. State and Transition Models: Response to an ESA symposium. Bulleting of the Ecological Society of America 83:194-196.

Lauenroth, W. K. and W. A. Laycock (editors). 1989. Secondary Succession and the Evaluation of Rangeland Condition. Westview Press. Boulder, CO. 163 pp.

Laycock, W. A. 1991. Stable states and thresholds of range condition on North American rangelands. A viewpoint. Journal of Range Management 44:427-433.

Milton, S. J., W. R. J. Dean, M. A. de Plessis, and W. R. Siegfried. A conceptual model of Arid Rangeland Degradation. BioScience 44:70-76.

Rodriquez Iglesias, R. M. and M. M. Kothmann. 1997. Structure and causes of vegetation change in state and transition model applications. Journal of Range Management 50:399-408.

Stringham, T. K., W. C. Krueger, P. L. Shaver, and P. J. Mieman. 2001. States, transitions, and thresholds: Further refinement for rangeland applications. Agricultural Experiment Station, Oregon State University. Special Report 1024. 15 pp.

Walker, B. H. 1993. Rangeland Ecology: Understanding and managing change. Ambio 22:80-7

Westoby, M., B. Walker, and I. Noy-Meir. 1989. Opportunistic management for rangelands not at equilibrium. J. Range Management42:266-74.

____

B. Schultz, Extension Educator, Univ. of Nevada Reno, Cooperative Extension; Winnemucca, NV schultzb@unce.unr.edu

S. Swanson, State Extension Range Specialist, Univ. of Nevada Reno; Reno, NV swanson@cabnr.unr.edu

NOTES:

Chemical Control Measures to Reduce Understory Weeds Associated with Sagebrush and Pinyon/Juniper Woodlands

Bob Wilson

Maintenance of a diverse plant community is one of the major ecological efforts in the Great Basin. When native plants are displaced by alien invasive plant species, management efforts to reverse the situation are often called for. Herbicides are a tool that can be used to selectively control some understory vegetation types under some situations or conditions.

B. Wilson, Extension Educator
White Pine County; Ely, NV

NOTES:

Identification of the Principal Sagebrush Taxa within the Great Basin

Alma H. Winward

Although the genus Artemisia is found on most continents, woody *Artemisias* (Sagebrushes) found on the North American Continent are somewhat unique from those of the rest of the world. Many scientists believe the sagebrushes found here in the western United States have a common ancestry with the herbaceous or semi-herbaceous sages in Asia. Their belief is that this ancestry was disrupted when an ancestor (or ancestors) of the United States *Artemisias* arrived from Asia via the temporary Aleutian Land Bridge. Through genetic isolation and selection processes the American sagebrushes gradually developed into the unique sagebrushes we find here today.

Depending on ones interest and background there are at least 27 identified taxa of these woody *Artemisias*. Several others are still in various stages of taxonomic development as they evolve to fit the numerous combinations of climate/soil settings found in the western United States. At least three of these have not yet been named but are worthy of some type of official recognition.

Of the 30 named or to be named taxa, twelve are capable of root sprouting or stem layering, characteristics which provide them reproductive advantages over non-sprouting/layering taxa. Chemistry-wise, water extracts from at least 18 of these fluoresce a bright cream-blue color under long wave ultra-violet light, seven do not fluoresce and five fluoresce only a moderately blue color. Fourteen are considered low-statured forms (generally less than 18 inches in height), four are mid-statured (18 to 10 inches in height) and 12 are tall statured (greater than 30 inches in height). At least nine have flower bracts longer than the flower heads and the center leaf lobe on persistent leaves on eight are at least three times as long as wide. These characteristics, as well as others such as leaf morphology, flower stalk arrangement, and number of flowers per head, serve as markers for taxonomic identification.

Each of the taxa have moderate to vastly different palatability and structural characteristics which influence their particular values to mammals and birds. All likewise have their unique combinations of associated under story species. Examples of the 20 taxa found in the Great Basin will be discussed in this presentation.

References:

Beetle, A.A. 1960. A study of sagebrush. The section *Tridentatae* of *Artemesia*. Wyoming Agri. Exp. Sta.Bull, 368. 83p.

Beetle, A.A. and A. Young. 1967. A third subspecies in the *Artemisia tridentata* complex. Rhodora 67:405-406

Holbo, H.R. and H.N. Mozingo. 1965. The chromatographic characterization of *Artemisia,* Section *Tridentatae.* Amer. Jour. Bot. 52 (9): 970–8

McArthur, E.D.; Mudge, J.; Van Buren, R.; Anderson, W.R.; Sanderson, S.C.; Babbel, D.G. Randomly amplified polymorphic DNA analysis (RAPD) of *Artemisia* subgenus *Tridentatae* species and hybrids. Great Basin Naturalist; Vol 58, no1 (Jan 31, 1998): p. 12-27.

Ward, G.H. 1953. *Artemisia* section *Seriphidium* in North America. Contributions from the Dudley Herbarium. 4: 155-205.

Winward, A.H. and E. Tisdale. 1969. A simplified chemical method for sagebrush identification. Univ. of Idaho; For Wildlf. and Range Exp. Sta. Note No 11. 2 p.

Winward. A.H. and E. W. Tisdale: 1977. Taxonomy of the *Artemisia tridentata* complex in Idaho. For. Wildlf. and Range Exp. Sta Bull. 19. 15p.

A.H. Winward, Regional Ecologist
USDA FS; Ogden, Utah
awinward@fs.fed.us

NOTES:

Guidelines to manage sage grouse populations and habitats.

Alan Sands

Background

Sage grouse populations and habitats have been a concern to sportsman, wildlife enthusiasts, and biologists for > 80 yrs. Despite this concern populations and habitats have declined throughout much of its range. These decline has resulted in four petitions to list the species under the federal Endangered Species Act. Two petitions have resulted in candidate classification, one for Gunnison sage grouse and one for the Washington state population. Two other petitions, one for a Distinct Population Segment in California and parts of Western Nevada and the other for greater sage grouse rangewide, are pending.

Braun et al. (1977) provided guidelines for managing sage grouse habitats. New guidelines were published in 2000 (Connelly et al. 2000) based on the latest research. These new guidelines have created a number of questions and criticisms. This paper will review the updated guidelines and seek to clarify the more controversial elements.

Population Characteristics

Sage grouse have relatively low reproductive rates and high annual survival compared to most game birds. Annual production can also be affected significantly by weather events (e.g. Drought or cold, wet conditions during the peak of hatch).

Sage grouse display a variety of annual migratory patterns. Populations may have: 1) distinct winter, breeding and summer areas; 2) distinct summer areas and integrated winter and breeding areas; 3) distinct winter areas and integrated breeding and nesting areas, or 4) well integrated seasonal habitats (i.e. non migratory). Regardless of migratory pattern, large areas of suitable habitat are needed to support a population, especially for migratory populations. Migratory populations may occupy areas in excess of 2,700 km.

Habitat Characteristics

Habitats occupied by sage grouse vary with season. Spring breeding habitat includes leks, nests, and early brood rearing. Leks are usually located in small, open areas surrounded by sagebrush cover types. Nesting and early brood rearing habitat are usually in sagebrush cover and most nests are located under a sagebrush shrub. In general, sage grouse nests are placed under shrubs having larger canopies and more ground and lateral cover than at random sites. Moreover, successful sage grouse nests tend to be in these types of situations. Early brood rearing usually occurs near nest sites where a diversity of forbs and an abundance of insects are available. These sites are often more open (i.e. less sagebrush cover) than nesting sites/areas.

As sagebrush habitats desiccate during the summer, broods are moved to more mesic sites, including meadows, riparian areas, springs and seeps, ephemeral lakebeds, and irrigated farm fields. The common characteristic of summer brood rearing habitat is an abundance of forbs in or adjacent to sagebrush cover.

Sagebrush provides both food and cover during the winter. To meet winter survival needs, sufficient sagebrush cover protruding above snow is essential to sage grouse.

Productive sage grouse habitats have the following characteristics:

> Spring Breeding Habitat - 15-25% canopy cover of sagebrush, 15% perennial grass cover, 10% perennial forb cover, and late May, early June herbaceous cover $\geq$ 18 cm (7 in.). At least 80% of the potential habitat is within these parameters.
>
> Summer Brood Rearing Habitat – 10-25% canopy cover of sagebrush and >15% canopy cover of grasses and forbs. At least 40% of this habitat is within these parameters.
>
> Winter Habitat - 10-30% canopy cover of sagebrush that is at least 25-35 cm high regardless of snow conditions. At least 80% of the winter range is within these parameters.

Population management

Decisions regarding hunting should be based on careful assessments of population size and trends. Most populations appear to sustain hunting if carefully managed. Populations with less than 300 adults should not be hunted, however. For populations that are declining for three or more years, seasons and bag limits should be conservative or the season closed. Large populations that are stable or increasing can sustain a longer hunting season and a more liberal harvest. Harvest rates should not exceed 10%.

Predator management is more problematic than hunting. Predator control is costly, often ineffective, and sometimes has unintended consequences. Consequently, decisions regarding predator control should be based on sound data that clearly supports the need (e.g. nest success <25%, annual survival of adult hens <45%).

Habitat management

Sage grouse habitat management requires at least a rudimentary knowledge of both population distribution and the underlying habitat conditions. Knowing the location and extent of seasonal ranges, especially wintering and nesting areas, coupled with the habitat conditions (e.g. sagebrush coverage, herbaceous conditions) within those areas allows a manager to compare existing conditions relative to desired conditions as well as identify the nature and extent of habitat fragmentation. With this knowledge sound judgements can be made to maintain, enhance, or restore essential habitat characteristics. The importance of taking this landscape view cannot be overemphasized.

Habitat Protection/Restoration

Much historical sage grouse habitat has been lost or degraded significantly. Consequently, remaining areas that provide productive habitats should be protected from fire and other forms of alteration. Restoration efforts, potentially engaging a variety of tools (fire, grazing, mechanical, and/or chemical), should be targeted to key areas that have been lost or degraded and that offer the greatest opportunity to improve an existing population, link isolated populations, or reestablish an extirpated population.

Sage Grouse Conservation

Conservation plans, preferably developed at the local level involving all interested parties, that address sage grouse needs and identify actions to maintain, improve, and restore sage grouse habitats are needed to reverse the current trend in sage grouse populations and habitats. Natural resource agencies also need to increase their knowledge of populations and habitat conditions. The new sage grouse management guidelines are based on the best available information but clearly they do not cover every situation throughout the range of sage grouse. Site-specific ecological differences that affect sage grouse may occur and should be considered.

References:

Braun, C.E., T. Britt, and R. O. Wallestad. 1977. Guidelines for maintenance of sage grouse habitats. Wildlife Society Bulletin 5:99-106.

Connelly, J.W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Wildlife Society Bulletin 2002, 28(4): 967-985.

A. Sands, Stewardship Ecologist
The Nature Conservancy; Boise, ID
asands@tnc.org

NOTES: