

1 Ecology and Management of Sage-Grouse and Sage-Grouse Habitat

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Abstract

1
2 Sage-grouse (*Centrocercus urophasianus* and *C. minimus*) historically inhabited much of
3 the sagebrush-dominated habitat of North America. Today, sage-grouse populations are
4 declining throughout most of their range. Population dynamics of sage-grouse are marked by
5 strong cyclic behavior. Adult survival is high, but is offset by low juvenile survival, resulting in
6 low productivity. Habitat for sage-grouse varies strongly by life-history stage. Critical habitat
7 components include adequate canopy cover of tall grasses (≥ 18 cm) and medium height shrubs
8 (40-80 cm) for nesting, abundant forbs and insects for brood rearing, and availability of
9 herbaceous riparian species for late-growing season foraging. Fire ecology of sage-grouse
10 habitat changed dramatically with European settlement. In high elevation sagebrush habitat, fire
11 return intervals have increased (from 12 - 24 to >50 years) resulting in invasion of conifer
12 species and a consequent loss of understory herbaceous and shrub canopy cover. In lower
13 elevation sagebrush habitat, fire return intervals have decreased dramatically (from 50 - 100 to
14 <10 years) as a result of invasion by annual grasses, causing loss of perennial bunchgrasses and
15 shrubs. Livestock grazing can have negative or positive impacts on sage-grouse habitat
16 depending on the timing and intensity of grazing, and which habitat element is being considered.
17 Early season light to moderate grazing can promote forb abundance/availability in both upland
18 and riparian habitats. Heavier levels of utilization decrease herbaceous cover, and may promote
19 invasion by undesirable species. At rates intended to produce high sagebrush kill, herbicide-
20 based control of big sagebrush may result in decreased habitat quality for sage-grouse. Light
21 applications of tebuthiuron (N-[5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2-yl]-N,N'-dimethylurea)
22 can decrease canopy cover of sagebrush and result in increased grass and forb production which
23 may be locally important to nesting and foraging activities. The ability of resource managers to

1 address sage-grouse habitat concerns at large scales is aided greatly by geomatics technology and
2 advances in landscape ecology. These tools allow unprecedented linkage of habitat and
3 population dynamics data over space and time and can be used to retroactively assess such
4 relationships using archived imagery. The sage-grouse decline is a complex issue that is likely
5 associated with multiple causative factors. Solving management issues associated with the
6 decline will require unprecedented cooperation between wildlife biology, range science, and
7 other professional disciplines.

8 **Introduction**

9 Historically, greater (*Centrocercus urophasianus*) and Gunnison (*C. minimus*) sage-
10 grouse inhabited large portions of sagebrush-dominated North American rangelands [both
11 sagebrush steppe and sagebrush semi-desert plant assemblages (West 1983a and b, West and
12 Young 2000)]. The family Tetraoninae is reported to be of North American origin (Lucchini et
13 al. 2001), and at one time, the range of sage-grouse encompassed significant portions of the
14 western (US) states and extended north into the Canadian provinces of British Columbia,
15 Alberta, and Saskatchewan (Fig. 1). Many plant communities providing habitat to sage-grouse
16 have undergone significant, and in some cases, lasting changes in the 19th and 20th centuries.
17 Factors responsible for plant community change have included (but are not limited to) alterations
18 in fire regime; excessive livestock grazing; proliferation of non-native plant species; conversion
19 of rangeland to seeded pastures [e.g. Crested Wheatgrass (*Agropyron cristatum* L.)], cropland
20 and roads; and other land alterations. Concurrent with these habitat changes has been a
21 generalized decline in sage-grouse abundance. The reasons for this decline are difficult to
22 understand. Putting together the pieces of the puzzle involves integrating sage-grouse population
23 ecology and habitat requirements, as well as the ecology and management of plant communities

1 that comprise sage-grouse habitat. It is critical that the relationship between changes in habitat,
2 and changes in sage-grouse populations be defined at multiple scales, given the extended
3 temporal and spatial horizons that frame the ecology of these species.

4 This paper synthesizes current knowledge regarding pertinent topics in sage-grouse
5 ecology and management and suggests direction for future research and management. Others
6 (Braun et al. 1977, Beck and Mitchell 2000, Connelly et al. 2000, Rowland and Wisdom 2002)
7 have provided synthesis and review papers previously. Our effort is not comprehensive to all
8 factors affecting sage-grouse, but is meant to provide expanded coverage of topical management
9 concerns with an emphasis on habitat ecology.

10 **Population ecology**

11 Connelly and Braun (1997) estimate a 17 - 47% decline in sage-grouse breeding
12 populations since 1985 in states that have sufficient records for quantifying sage-grouse
13 numbers. For many years periodic fluctuations in abundance were attributed to cycles, often
14 over 8-12 year intervals (Batterson and Morse 1948, Rich 1985). Nevertheless, an explanation
15 for fluctuations in abundance has been difficult to support with documented causal relationships
16 (Braun 1998). Although the mechanisms underlying fluctuations in abundance are still debated,
17 the major concern now is that most (but not all) populations are showing long-term declines,
18 whether cyclic in the short-term or not (Braun 1998).

19 **Productivity, survival and recruitment**

20 The dynamics of a population are a reflection of productivity, survival, and recruitment.
21 Productivity can be further divided into stages, including clutch size, hatchability, nest
22 likelihood, re-nest likelihood, nest success, and annual reproductive success (Schroeder et al.
23 1999; Table 1). Sage-grouse productivity is low, despite their high reproductive potential.

1 Declines in productivity appear to be related to the substantial number of non-nesting females
2 (nest/re-nest likelihood in Table 1) and low rate of annual reproductive success. Nest success is
3 inversely correlated with density of predators, such as common ravens (*Corvus corax*, Batterson
4 and Morse 1948), however, rates of predation are tied to habitat quality, and it has been
5 suggested that the most efficient method for mitigating high rates of nest predation may be
6 through the effective management of habitat (Hamerstrom et al. 1957, Angelstam 1986, Andrén
7 and Angelstam 1988, Schroeder and Baydack 2001). The impact of re-nesting on productivity is
8 unclear; re-nesting has had limited impact on overall productivity in Oregon (Gregg et al. 1994,
9 Hanf et al. 1994) and Idaho (Connelly et al. 1993), however, Schroeder (1997) reported that 38%
10 of productivity in north-central Washington was due to re-nesting.

11 Productivity, and subsequently recruitment, is further impacted by low juvenile survival
12 rates between hatch and the following breeding season (Table 1). Juvenile survival has proven
13 difficult to document in the field but the available estimates for this parameter are very low,
14 suggesting that understanding juvenile survival may be critical in managing the population
15 dynamics of sage-grouse. Food availability (Pyle and Crawford 1996), habitat quality (Sveum et
16 al. 1998a), harvest (Crawford and Lutz 1985), predation (Batterson and Mores 1948), and
17 weather (Blake 1970, Rich 1985) may impact juvenile survival. Recruitment of young birds into
18 the breeding population may be further complicated by dispersal of juveniles outward from the
19 nesting location (Browsers and Flake 1985; Dunn and Braun 1985, 1986). The direct impact of
20 dispersal on population dynamics of sage-grouse remains largely unexplored.

21 In contrast to low nest success and low survival of juveniles, annual survival of breeding-
22 aged birds tends to be higher than 50% in most situations, and as high as 75% for breeding-aged
23 females in Idaho (Connelly et al. 1994). Although high adult survival rates may compensate for

1 low productivity, it has been insufficient to reverse their widespread declines in abundance
2 (Braun 1998).

3 **Population fluctuations and research needs**

4 The relatively high survival rates and low productivity of adult sage-grouse may help
5 explain the dramatic fluctuations in sage-grouse abundance that some have suggested resemble
6 "cycles" (Rich 1985). Although sage-grouse have a high reproductive potential, they may only
7 rarely have years where productivity is high. These infrequent "boom" years, in combination
8 with the high survival of breeding-aged birds, may produce multi-year fluctuations in abundance.

9 Dramatic fluctuations in abundance (Rich 1985) create tremendous problems for
10 evaluating population-level responses to management. For example, although habitat quality is
11 related to sage-grouse population dynamics (Edelmann et al. 1998), there are 4 basic reasons
12 why most management changes require years before a population change is detected. First,
13 changes in habitat management do not immediately change habitat characteristics. This is
14 particularly true where habitat has undergone gradual, long-term structural and/or compositional
15 changes. Second, sage-grouse population response may lag behind changes in sage-grouse
16 productivity. This lag effect occurs because yearling males may not display on leks (Jenni and
17 Hartzler 1978, Emmons and Braun 1984) and yearling females may not nest (Connelly et al.
18 1993, Gregg et al. 1994) during their first potential breeding season. Third, population responses
19 to short-term habitat management (< 10 years) may not be observed in sage-grouse populations,
20 because the typical fluctuations in a 10-year interval may dwarf any response to improved
21 management. Fourth, the lack of basic information on important stages in the life history of
22 sage-grouse, such as juvenile survival, may indicate that the appropriate habitat management
23 strategy for a given population is not yet known.

1 Habitat management is one of the few areas where research has shown that reproductive
2 parameters can be altered. For example, substantial data exists documenting significant
3 relationships between specific habitat characteristics and annual reproductive success (Bean
4 1941, Pyrah 1971, Wallestad and Pyrah 1974, Connelly et al. 1991, Gregg et al. 1994, Hanf et al.
5 1994, Young 1994, DeLong et al. 1995, Sveum, et al. 1998b). Adequate habitat provides the
6 cover necessary to conceal nests and to provide the foods necessary for hens to lay eggs and to
7 incubate clutches (Barnett and Crawford 1994). Manipulation of habitat also has potential to
8 influence other aspects of sage-grouse population dynamics including clutch size, nest and re-nest
9 likelihood, and survival of juveniles and breeding-aged birds. For example, adequate
10 vegetational canopy cover may provide critical escape cover, thus lowering the risk of predation.

11 Although many of the specific relationships between habitat quality and productivity and
12 survival are not clear, the overall relationship can be illustrated by the dramatic changes in
13 landscape throughout the historical North American distribution of sage-grouse. Most of the
14 remaining populations are associated with intact habitats in relatively northern latitudes, high
15 elevations, and/or mesic environments (Connelly and Braun 1997). In contrast, significantly
16 altered habitats and those in southern latitudes, low elevations, and/or xeric environments have
17 become uninhabitable. This is a trend that has been ongoing for the past 100 years and is likely
18 to continue unless there are widespread changes in management (Brown and Davis 1995). In
19 addition, the continued reduction in occupied habitat will result in increased fragmentation and
20 isolation of remaining sage-grouse populations.

21 Although a substantial quantity of data exists on some basic parameters associated with
22 population dynamics (e.g., clutch size, egg hatchability, nesting success, survival of breeding age
23 birds), information on juvenile survival, dispersal, and recruitment is inadequate. It is essential

1 that research be initiated as soon as possible, because of the dramatic declines in sage-grouse
2 distribution and abundance (Connelly and Braun 1997, Braun 1998) and because of the long time
3 periods (>10 years) needed to observe treatment effects in a species with low productivity and
4 high survival. It is also critical that other research continues, including the influence of nutrition
5 (Barnett and Crawford 1994, Pyle and Crawford 1996), weather (Gill 1966, Blake 1970, Hupp
6 and Braun 1989), predation (Batterson and Morse 1948, Schroeder and Baydack 2001), and
7 behavior (Scott 1942, Gibson and Bradbury 1986) on population dynamics.

8 **Sage-grouse habitat relationships**

9 While many factors likely influence productivity, the only factor that has been
10 consistently manageable is habitat (Connelly et al. 1991, Gregg et al. 1994, DeLong et al. 1995,
11 Sveum, et al. 1998b). The importance of sagebrush (woody *Artemisia*) as a source of cover and
12 food for sage-grouse is well known (Patterson 1952, Braun et al. 1977), however, sage-grouse
13 require a variety of plant community types for breeding, nesting, brood-rearing and wintering
14 (Table 2). Describing habitat relationships is complicated by the fact that sage-grouse
15 populations often display complex seasonal movement patterns. Populations may exhibit
16 different patterns of migration, with some populations remaining resident throughout the year,
17 some migrating between wintering and breeding habitat, and some with more complicated
18 movements (Connelly et al. 1988). Migratory birds in Idaho have been reported to range up to
19 125 km, with an annual home range size of 2,764 km² (Leonard et al. 2000). For management
20 purposes, spatial patterns of habitat use over time should be determined on a population-by-
21 population basis.

22 **Winter habitat**

23 During winter sage-grouse utilize medium to tall sagebrush communities (25 – 80 cm, or

1 25 – 35 cm above the snow) on south and west facing slopes (Ihli et al. 1973, Connelly et al.
2 2000; Table 2), and forage primarily on sagebrush leaves (Patterson 1952). Where available,
3 low sagebrush (*A. arbuscula* Nutt.) habitat (particularly on wind-swept ridges) is also utilized
4 (Hanf et al. 1994). Home range for wintering migratory and non-migratory populations has been
5 reported as > 140 km² (Robertson 1991) and 11 to 31 km² (Wallestad 1975), respectively.
6 Sagebrush canopy cover at sage-grouse winter use sites can range from 12% in Oregon (Hanf et
7 al. 1994) to 43% in Colorado (Schoenberg 1982), but adequate cover is typically available on a
8 landscape scale (Connelly et al. 2000). Unless snow completely covers sagebrush (Hupp and
9 Braun 1989), severe winter weather conditions have little effect on sage-grouse populations (Call
10 and Maser 1985) and sage-grouse may actually gain weight during the winter months (Beck and
11 Braun 1978).

12 **Lekking habitat**

13 Leks are typically located in sparsely vegetated areas (Call and Maser 1985) with few to
14 100 or more displaying males. Leks typically reflect the availability of nesting habitat in the
15 surrounding area. There is no evidence that lek habitat is limiting to sage-grouse populations
16 (Schroeder et al. 1999), and, if needed, lekking habitat can be created by management activity
17 (Eng et al. 1979, Tate et al. 1979).

18 **Pre-laying habitat**

19 The pre-laying period is defined as the 5-week period preceding incubation (Barnett
20 1992) when habitat use centers around low sagebrush (*A. arbuscula* Nutt., but also, *A. nova* A.
21 Nels. and *A. rigida* Nutt.) and Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis* Rydb.)
22 communities (Table 2). During this period hens consume 50 - 80% of their diet as sagebrush
23 leaves with the remainder composed primarily of forbs (Barnett and Crawford 1994). Although

1 sagebrush leaves contribute importantly to the dry mass of the diet of pre-laying hens, the
2 nutrient contribution of forbs overshadows that of sagebrush and may be associated with
3 increased reproductive success (Barnett and Crawford 1994). Some authors (e.g., Rogers 1964,
4 Patterson 1952, Wallestad et al. 1975) have reported that sagebrush comprises >85% of the diet
5 during the pre-laying period. However, these authors did not separate diets by sex.

6 **Nesting habitat**

7 Sage-grouse nests are typically located under sagebrush plants, often in mountain big
8 sagebrush (*A. t. ssp. vaseyana* Rydb) habitat (Wallestad and Pyrah 1974, Gregg et al. 1994). A
9 variety of other sagebrush-dominated community types as well as bitterbrush (*Purshia tridentata*
10 Pursh DC.) and rabbitbrush (*Chrysothamnus* spp. Nutt.) sites may also be utilized (Hulet et al.
11 1986, Crawford et al. 1992, Aldridge and Brigham 2002; Table 2). Nests are generally located
12 near leks, but hens may move long distances from leks to nest (Hanf et al. 1994). In Idaho, only
13 55% of nests were within 3 km of the lek of capture (Wakkinen et al. 1992). Poor reproductive
14 success may result from a lack of key structural habitat features necessary for nesting (Blake
15 1970, Autenrieth 1981). Nest site selection is largely a function of height and amount of shrub
16 canopy cover (Klebenow 1969, Roberson 1986, Gregg 1992), while tall (>18cm) residual
17 bunchgrasses provide cover for screening (Gregg 1992, Gregg et al. 1994). Successful nests in
18 Oregon had 41% canopy cover of medium height (40-80 cm) sagebrush and 18% tall bunchgrass
19 (residual) canopy cover in the 3-m² area surrounding the nest (Gregg et al. 1994). Other forms
20 of herbaceous vegetation (e.g., residual forbs) may provide nest screening cover (Sveum et al.
21 1998b), however, exotic invaders (e.g., cheatgrass *Bromus tectorum* L.) generally do not.
22 Sagebrush canopy cover in nesting habitat should range from 15 - 25% (Connelly et al. 2000).
23 Winward (1991) suggested that maximum understory herbaceous production would be realized

1 at 12% sagebrush cover in Wyoming big sagebrush types and 20% sagebrush cover in mountain
2 big sagebrush types.

3 The most common reason for nest failure is predation by coyotes (*Canis latrans*) and
4 avian and small mammal species (Batterson and Morse 1948, Nelson 1955, Autenrieth 1981,
5 DeLong 1994). However, adequate vegetation structure at the nest site provides visual, scent
6 and physical barriers between ground nesting birds and predators, and may ultimately determine
7 susceptibility to predation (Gregg 1992, Gregg et al. 1994). Canopy cover of tall grasses and
8 medium height sagebrush is inversely related to the probability of nest predation in big sagebrush
9 habitats (Connelly et al. 1991, DeLong et al. 1995, Sveum et al. 1998b).

10 **Brood-rearing habitat**

11 The pattern of habitat use during the brood-rearing period is related to changes in food
12 availability and hens with broods are typically found where forb abundance is greatest
13 (Klebenow 1969, Drut et al. 1994a). For example, Sveum et al. (1998a) reported > 20% canopy
14 cover of forbs at brood-rearing sites and decreased shrub canopy cover (14 vs. 20%) relative to
15 random locations in Wyoming big sagebrush habitat. Specific habitats used during brood-
16 rearing (Table 2) are more mesic as the growing season progresses, which is associated with forb
17 desiccation (Wallestad 1971). Home range size for broods has been reported to range from less
18 than 1 km² in Montana (Wallestad 1971) to 5 km² in Oregon (Drut et al. 1994a). Differences in
19 home range size among broods have been attributed to availability of forbs (Drut et al. 1994a).
20 Use of riparian habitat is dependent on desiccation of forbs in sagebrush uplands and may occur
21 earlier in drought years (Savage 1969, Oakleaf 1971, Danvir 2002).

22 Forbs and insects comprise the bulk of sage-grouse chick diets until they are approximately
23 12 weeks of age, at which time sagebrush becomes a common component (Dargen et al. 1942,

1 Nelson 1955, Klebenow and Gray 1968, Peterson 1970). Specific taxa consumed by chicks are
2 very diverse. In Oregon chicks consumed 41 families of invertebrates, 34 genera of forbs, 2
3 genera of shrubs, and 1 genus of grass (Pyle 1993, Drut et al. 1994b), however, only a few
4 specific taxa of forbs were preferentially selected (Drut et al. 1994b). The relationship between
5 chick survival/recruitment and dietary factors has not been elucidated for most wild galliformes,
6 especially the tetraonids (Potts 1986). However, data suggest that availability of forbs and
7 invertebrates is associated positively with survival/recruitment of sage-grouse chicks (Drutt et al.
8 1994b). This relationship may be of particular importance during drought years when forb
9 availability is low and sagebrush becomes a greater component of the chick diet at an earlier age
10 (Drut et al. 1994a).

11 **Broodless hens and male habitat (growing season)**

12 Because sage-grouse nesting and brood-rearing success is extremely low in some years, a
13 relatively large portion of the summer female sage-grouse population is composed of broodless
14 hens (Gregg et al. 1993). Survival of these hens may be important to population maintenance.
15 Broodless hens begin to form small flocks of 2-3 birds in mid-May which may increase in size to
16 25 hens by early June (Gregg et al. 1993). Habitat use is similar to that of hens with broods
17 (Table 2), however, broodless hens move to riparian habitat earlier than hens with broods (Dalke
18 et al. 1963, Martin 1976, Gregg et al. 1993). Males follow a similar pattern of habitat use, but
19 typically remain in flocks separate from females.

20 **Monitoring considerations and research needs**

21 Monitoring sage-grouse habitat is complicated by the migratory behavior of sage-grouse
22 populations, or segments of populations (Connelly et al. 2000). As such, monitoring efforts and
23 habitat assessments require knowledge of both the spatial and temporal dynamics of migratory

1 agriculture. In the nonarable regions, a large portion of sagebrush-dominated communities has
2 been altered by changes in the proportion of trees, shrubs, grasses, and forbs. Changes in
3 structure and composition in non-cultivated areas are primarily attributed to altered fire regimes,
4 improperly managed livestock grazing, introduction of exotic plants, and herbicide use (Miller et
5 al. 1994).

6 **Fire**

7 Management of both wild and prescribed fires is considered one of the key issues in
8 maintaining sage-grouse populations in sagebrush-dominated landscapes. Sage-grouse evolved
9 in ecosystems where fire was a primary disturbance process. However, the role of fire in the
10 sagebrush biome is often over generalized. Fire regimes are spatially complex and vary through
11 time across the sagebrush region, and, since the 1860s, the ecological role of fire has changed
12 dramatically (West 1983b, 2000; West and Young 2000; Miller and Tausch 2001).

13 ***Pre-European settlement***

14 Presettlement fire return interval varied greatly depending on plant community type and
15 moisture regime (Fig. 2). For example, mean fire return interval (MFRI = time between fires)
16 varied between 12 and 25 years on productive mountain big sagebrush sites (Houston 1973,
17 Burkhardt and Tisdale 1976, Gruell 1999, Miller and Rose 1999), but can exceed 200 years in
18 more xeric mountain big sagebrush/western needlegrass (*Stipa occidentalis* Thurber)
19 communities occupying sandy soils (Waichler et al. 2001, Miller unpublished data). Estimates
20 of MFRI reported for Wyoming big sagebrush communities (Wright and Bailey 1982) are
21 largely based on fuel loads and likely plant composition prior to settlement. However, MFRI
22 only partially describes the frequency of fire. The variability of fire-free periods within a fire
23 regime is very important in determining landscape plant community composition, structure, and

1 fire behavior. Information on the variability of presettlement fire-free periods is limited. Two
2 studies conducted in mountain big sagebrush communities where MFRI was relatively short (10
3 to 20 years) reported that presettlement fire-free periods varied between 8-29 years (Gruell
4 1999), and 3-28 years (Miller and Rose 1999). Fire size and complexity (patchiness) are also
5 important factors influencing seed source for plant re-establishment (particularly sagebrush) and
6 wildlife use patterns.

7 The response of presettlement communities following fire was largely determined by the
8 preburn plant composition and fire tolerances of those species. Many herbaceous species in
9 sage-grouse habitat are well adapted to fire (Blaisdell 1953, Wright and Klemmedson 1965,
10 Conrad and Poulton 1966, Wright and Bailey 1982, Young and Miller 1985). Forb species that
11 resprout belowground from a caudex, corm, bulb, rhizome, or rootstock, exhibit rapid recovery
12 following fire. Annual and biennial forbs usually increase following fire through seed dispersal
13 mechanisms. However, forbs that are suffrutescent, low growing, or mat forming such as
14 pussytoes (*Antennaria* spp. Gaertner) or several of the buckwheats (*Eriogonum* spp. Michx.) can
15 be severely damaged by fire (Table 3). Big and low sagebrush and young juniper are easily
16 killed by fire (Blaisdell 1953, Burkhardt and Tisdale 1976, Wright and Bailey 1982).

17 Reestablishment of sagebrush in burned sites is highly variable and dependent on nearby
18 seed sources or seed reservoirs produced during the previous growing seasons in addition to
19 weather conditions following the fire (West and Yorks 2002). Dispersal of sagebrush seed is
20 limited to several meters from the parent plant. Reestablishment generally occurs more rapidly
21 in the more mesic big sagebrush communities. Generally, shrub cover can reach or exceed
22 preburn levels in as little as 20 years but more typically within 25 - 45 years (Watts and Wambolt
23 1996, Wambolt et al. 2001). MFRI of less than 50 years in mountain big sagebrush communities

1 and 100 years in low sagebrush/Sandberg bluegrass (*Poa sandbergii* Vasey) communities were
2 sufficient to control the encroachment of pinyon or juniper (Burkhardt and Tisdale 1976, Miller
3 and Rose 1999, Miller and Tausch 2001).

4 ***Post-European settlement***

5 Two common scenarios of fire-related plant community change following European
6 settlement are: 1) a decline in fire frequency resulting in increased dominance of woody species
7 (shrubs or trees) and a decrease in perennial forbs and grasses; or 2) an increase in Eurasian
8 weeds (particularly ephemerals), a consequent increase in fire frequencies, and loss of native
9 perennial shrubs, forbs, and grasses (Fig. 3).

10 The first scenario represents one of the most significant losses in mountain big sagebrush
11 habitat. For instance, Miller and Tausch (2001) estimated juniper and pinyon woodlands have
12 increased 10-fold during the past 130 years from 2.9 - 29 million hectares in the Intermountain
13 West. Approximately 18.9 million hectares of these woodlands occur within the range of sage-
14 grouse and under current climatic conditions, and in the absence of fire, these woodlands will
15 continue to expand (Betancourt 1987, West and Van Pelt 1986, West 1999a, Miller et al. 2000).
16 Where juniper gains dominance in mountain big sagebrush communities, shrub cover declines to
17 <1% (Miller et al. 2000) and the season of available succulent forbs is shortened because of rapid
18 soil moisture depletion (Bates et al. 2000).

19 The second scenario, which has most extensively occurred in the Wyoming big sagebrush
20 cover type, is the invasion of annual grasses. Invasion by exotic annuals has resulted in dramatic
21 increases in both size and frequency of fire (Young and Evans 1973, Whisenant 1990, Swetnam
22 et al. 1999, Tausch 1999, West 2000). For example, Whisenant (1990) reported MFRI in
23 Wyoming big sagebrush communities has been reduced from 50-100 years to < 10 years. Repeat

1 fires have allowed cheatgrass and other introduced annuals to replace the native shrub and herb
2 layers. As early as the 1930s, range managers were aware of the rapid invasion of cheatgrass
3 following fire (Stewart and Hull 1949). Introduced annual plants from Eurasia now either
4 dominate or have a significant presence on 6.9 million ha of public land in the Great Basin
5 (Pellant 1994), and over much of this area, annual-dominated communities can be considered a
6 new steady state (Laycock 1991). These fine fuels shift fire seasonality to the more active
7 growing period of native perennials (Whisenant 1990). The end results are that herbaceous
8 cover varies greatly from year to year depending on moisture availability, shrub cover is absent,
9 the season of available green plant material is shortened, high quality perennial forbs are scarce,
10 and forage is absent in late summer through winter.

11 Risk of invasion by Mediterranean annuals in Wyoming and basin (*Artemisia tridentata*
12 ssp. *tridentata* Nutt.) big sagebrush communities increases below elevations of 1500 m and
13 becomes extreme below 1000 m. Exotic annual grasses such as cheatgrass will not usually
14 dominate more mesic and cooler sagebrush types characterized by mountain big sagebrush and
15 low sagebrush. Wyoming big sagebrush growing on old parent materials (low nutrient status,
16 e.g., West and Yorks 2002) and colder sites, such as the high deserts in central Nevada, southern
17 Utah, and eastern Wyoming, also appear to be more resistant to cheatgrass invasion. Colder
18 temperatures lower and delay germination and slow down phenological development of
19 cheatgrass. Competition from native herbaceous species may also be greater in these cooler and
20 more mesic plant communities. Cheatgrass usually fares poorly in black sagebrush communities.
21 Medusahead (*Taeniatherum asperum* Simonk.), however, can become abundant on some low
22 sagebrush sites below 1500 m, especially where clay content is high (Dahl and Tisdale 1975,
23 West and Young 2000).

1 Secondary weed species such as squarrose knapweed (*Centaurea squarrosa* Willd), rush
2 skeleton weed (*Chondrilla juncea* L.), and bur buttercup (*Ranunculus testiculatus* Crantz) are
3 rapidly invading cheatgrass and native plant communities in the Intermountain West, especially
4 on ecological sites where Wyoming big sagebrush once dominated. Squarrose knapweed, like
5 cheatgrass, produces an abundance of fine fuels. Continued spread of these secondary weeds
6 shortens fire return intervals, increases the homogeneity and size of fires across the landscape,
7 and threatens the integrity of Wyoming big sagebrush habitat.

8 ***Implications to sage-grouse and research needs***

9 A limited number of studies have produced mixed reports on the impact of fire on sage-
10 grouse populations (Rowland and Wisdom 2002). This is not entirely surprising given that the
11 impact of fire on sage-grouse habitat is contingent on a large number of factors, including site
12 potential, ecological condition, limiting functional plant groups, and the pattern, size, and season
13 of burning. Additionally, most studies investigating the effects of fire on sage-grouse have been
14 short-term (<10 years) (e.g., Bensen et al. 1991, Fischer et al. 1996). The lag response of sage-
15 grouse combined with the long time periods typically needed by sagebrush to reestablish after
16 fire, suggest that the ultimate evaluation of impacts of fire on sage-grouse habitat is to determine
17 long-term use of burned and unburned areas, as well as periodic evaluation of plant community
18 structural diversity over time, since burned areas will change in habitat suitability over time.

19 The impact of fire on the structure and composition of sage-grouse habitat may be
20 positive or negative. Fire can enhance native perennial forbs and grasses, particularly where
21 sagebrush is abundant, good populations of native herbs are present, and exotic species are
22 limited. This most often applies to mountain big sagebrush communities where shrub canopy
23 cover can exceed 35% and perennial forbs can increase 2 to 3 fold following fire (Blaisdell 1953;

1 Miller, unpublished data). However, the response of perennial forbs and grasses following fire
2 can be highly variable (Harniss and Murray 1973, Nelle et al. 2000). Fire can increase the
3 growing season length for forbs important to sage-grouse (Wroblewski 1999), enhance the nutrient
4 quality of forbs (McDowell 2000) and sage-grouse have been reported to be attracted to burned
5 areas during the brood-rearing period (Klebenow and Beall 1977, Martin 1990). Limited
6 research indicates that ants and beetles initially increase with fire in mountain big sagebrush
7 communities but are not affected long-term (Nelle et al. 2000). Periodic fires with intervals less
8 than 50 years will prevent negative habitat effects associated with pinyon and juniper
9 encroachment into shrub steppe communities (Miller and Tausch 2001). In areas where grasses
10 and shrubs have been drastically reduced or eliminated due to conifer dominance, mechanical
11 pre-treatment of conifers can be used to promote fine fuel production. In addition, prescribed
12 fires can break up fuel continuity, reducing the threat of future large and more complete burns.

13 Negative impacts of fire on sage-grouse habitat may include removal, at least
14 temporarily, of the sagebrush overstory, thus decreasing the value of affected communities as
15 winter and nesting habitat. In Wyoming big sagebrush dominated communities, there is little
16 evidence that fire will enhance sage-grouse habitat where there is already a balance of native
17 shrubs, perennial grasses, and forbs. Burning in these communities does not significantly
18 increase desirable forbs used as sage-grouse food (Fischer et al. 1996; Miller, unpublished data)
19 and abundance of beetles (Hymenoptera), an important chick food (Pyle and Crawford 1996),
20 may decrease (Fischer et al. 1996) or be unaffected (Pyle and Crawford 1996). Fire should not
21 be used where sagebrush cover is the limiting factor for sage-grouse or where introduced annuals
22 have replaced native perennial forbs and grasses. When deciding whether to burn on arid/low
23 elevation sites, or in the Wyoming big sagebrush cover type, managers must balance the desired

1 mix of plant communities with local assessments of the ability of shrubs to re-establish post-fire,
2 and the potential for fire-induced annual grass dominance. In addition, some herbaceous species
3 such as Idaho fescue (*Festuca idahoensis* Elmer) are sometimes decreased by fire and can
4 require long time intervals for recovery (Wambolt et al. 2001). The amount of less palatable
5 shrubs that resprout [rabbitbrush, horsebrush (*Tetradymia* spp. DC.), and broom snakeweed
6 (*Gutierrezia sarothrae* Lag.)] should also be considered. These species typically increase
7 following a burn but may be replaced by sagebrush in the absence of frequent disturbances
8 (Young and Evans 1974). The impact of fire on the ecology of other *Artemisia* species, such as
9 mid to high elevation silver sagebrush (*Artemisia cana* Pursh) communities, is not well
10 understood.

11 The goal of managing sage-grouse habitats for an optimal balance of shrubs, forbs, and
12 grasses at community and landscape scales should be analogous with restoring and or
13 maintaining form, function, and process in sagebrush-dominated habitats. However, many
14 questions remain regarding the impact of fire on sage-grouse habitat. For instance there is only
15 limited documentation on the rate, variability, and environmental factors affecting sagebrush re-
16 establishment in burns, and post-fire restoration of native herbaceous and shrub species in lower-
17 elevation sagebrush communities has met with only limited success. The importance of
18 successful restoration increases in proportion to the likelihood of post-fire annual grass invasion.
19 The spatial and temporal effects of fire at landscape scales has received only limited attention,
20 and should be addressed in concert with determining the landscape-scale mosaic of seral stages
21 that provides optimal habitat for sage-grouse.

22 **Livestock grazing**

23 Livestock grazing has been extant in sagebrush plant communities for more than a

1 century. However, only a few research studies have directly addressed the effects of livestock
2 grazing on habitat use by sage-grouse. Consequently, rangeland and wildlife managers must
3 rely, with caution, on indirect evidence for guidance. Livestock grazing may affect sage-grouse
4 habitat directly by altering structural habitat factors or plant community composition, or
5 indirectly by altering abiotic processes (e.g., MFRI) and invasibility of sagebrush plant
6 communities. While the impact of grazing on sagebrush plant communities varies with site
7 potential, ecological condition, and climate variables, the aspects of livestock grazing that are
8 controlled by management are, principally, the timing and intensity of defoliation.

9 ***Livestock grazing history***

10 Herbivory as a disturbance of sagebrush-dominated plant communities existed prior to
11 the arrival of domestic livestock in sage-grouse habitat (Burkhardt 1996). However, the
12 proliferation of domestic livestock in the latter 1800s represented a fundamental change in the
13 diversity of dominant herbivores, and the timing, and selection pressures associated with
14 herbivory (Miller et al. 1994). Historic grazing practices centered around season-long use with
15 stocking rates far exceeding carrying capacity (Young and Sparks 1985). The net impact of
16 these grazing practices on sagebrush-dominated plant communities was an increase in shrub
17 abundance, a decrease in perennial grasses, and the proliferation of non-native annual grasses
18 (Young et al. 1976, Young et al. 1972). By 1900, cattle and sheep on western rangelands totaled
19 over 30 million animals (Wagner 1978). Cattle and sheep AUM's on federal land declined since
20 the early 1900s (Council for Agricultural Science and Technology 1974, Laycock et al. 1996)
21 and decreased more than 25% in the last 40 years (USDI-BLM 1990). Concurrent with reduced
22 stocking of public rangelands has been measurable improvements in range condition during the
23 latter half of the 1900s (Box 1990, Laycock et al. 1996).

1 ***Timing and intensity of livestock grazing***

2 Research suggests that moderate livestock grazing or less in mid to late summer, fall, or
3 winter is generally compatible with the maintenance of perennial grasses and forbs in sagebrush
4 habitat (Pechanec and Stewart 1949; Mueggler 1950; Laycock and Conrad 1967, 1981; Gibbens
5 and Fisser 1975; Miller et al. 1994; Bork et al. 1998). Herbaceous species in sagebrush plant
6 communities are predominantly cool-season (C-3) plants that are vulnerable to defoliation during
7 late spring and early summer. Heavy grazing (approximately 60% or greater utilization by
8 weight) during this time has predictable results: 1) the vigor, yield, and cover of late-seral
9 grasses and forbs decrease; 2) early-seral species (including annual grasses) may increase; 3)
10 sagebrush density and canopy cover may increase (Craddock and Forsling 1938, Pechanec and
11 Stewart 1949, Mueggler 1950, Laycock 1967, Bork et al. 1998); and 4) transition of sagebrush
12 uplands to higher ecological status is inhibited (Mueggler 1950, Eckert and Spencer 1986,
13 Laycock 1987).

14 Moderate use has traditionally been defined as occurring within the range of 40 - 60%
15 utilization by weight, however, generalizing a specific level of utilization that represents "proper
16 use" can be difficult (Caldwell 1984). These difficulties arise in part due to lack of consistency in
17 measurement technique (Frost et al. 1994), and the variable impact of a given level of utilization
18 on plant communities in accordance with plant species present, site conditions, and climate
19 variables. Some perennial grasses, such as Indian ricegrass (*Oryzopsis hymenoides* (R. & S.)
20 Ricker), needle-and-thread (*Stipa comata* Trin. & Rupr.), Nevada bluegrass (*Poa nevadensis*
21 Vasey ex Scribn.), and Sandberg bluegrass, can withstand severe grazing (approximately 80% or
22 greater utilization) as long as defoliation does not occur during the plants' reproductive period
23 (Pearson 1964). Other grasses such as Idaho fescue, Thurber needlegrass (*Stipa thurberiana*

1 Piper), and bottlebrush squirreltail (*Sitanion hystrix* (Nutt.) J.G. Smith) decrease with heavy
2 grazing (Rickard et al. 1975, Eckert and Spencer 1987). Restoration of sites in poor ecological
3 condition may benefit from reduced utilization (Holechek et al. 1999) or rest (Fig. 4).
4 Additionally, grazing tolerance of sagebrush-dominated plant communities can decrease with
5 drought conditions and increase in periods of above average precipitation (Westoby et al. 1989).
6 When used in conjunction with other information sources (e.g., weather data, non-livestock
7 sources of herbivory) utilization data can be a valuable tool for helping to interpret the influence
8 of livestock herbivory on vegetation trend (Sanders 1998). However, utilization data are not a
9 substitute for long-term vegetation monitoring, and management objectives should be based on
10 desirable vegetation composition over time, not utilization guidelines (Sharp et al. 1994,
11 Burkhardt 1997, Sanders 1998).

12 Cattle, sheep, and horses (*Equus caballus*) in sagebrush habitat eat grass-dominated diets
13 in all seasons of the year (Severson et al. 1968, Harrison and Thatcher 1970, Mackie 1970, Uresk
14 and Rickard 1976, Olsen and Hansen 1977, Reiner and Urness 1982, Krysl et al. 1984, Ngugi et
15 al. 1992, Crane et al. 1997, Glidewell et al. 2001) although sheep may consume a higher
16 percentage of their diet as forbs. Livestock usually consume little to no sagebrush (< 10%)
17 unless snow depth exceeds 20 cm (Harrison and Thatcher 1970), but winter sheep use of low
18 elevation basin big sagebrush may be much greater (Cook et al. 1954). Sheep grazing in fall
19 favors production of perennial forbs, whereas spring grazing can decrease forb production (Bork
20 et al. 1998). Reduced sagebrush canopy cover in fall-grazed pastures (Mueggler 1950, Laycock
21 1967, Bork et al. 1998) is caused largely by competition from healthy grasses and forbs, rather
22 than fall livestock browsing of sagebrush (Wright 1970).

23 Sagebrush cover generally increases as utilization of the herbaceous understory increases

1 (Wright and Wright 1948, Pechanec and Stewart 1949, Mueggler 1950, Laycock 1967, Bork et
2 al. 1998). But, once sagebrush cover reaches an upper threshold, livestock exclusion may have
3 little effect on reversing the immediate trend (Johnson and Payne 1968, Rice and Westoby 1978,
4 Sanders and Voth 1983, Wambolt and Payne 1986). Over long time intervals (40 years or more),
5 sagebrush abundance may decline with a concomitant increase in understory herbaceous species
6 (Anderson and Inouye 2001). On Wyoming big sagebrush sites with dense sagebrush and an
7 understory of annual grasses, reductions in livestock grazing can hasten further habitat
8 degradation if ungrazed fuel loads promulgate wildfires that burn uniformly and kill sagebrush
9 on vast areas (Peters and Bunting 1994, West 1999b; Fig. 4).

10 Timing of grazing greatly influences the effects of livestock grazing in meadows and
11 riparian areas. These sites are particularly vulnerable in late summer when excessive grazing
12 and browsing may damage riparian shrubs, reduce the yield and availability of succulent herbs
13 (Kovalchik and Elmore 1992), and cause deterioration of riparian function over time (Klebenow
14 1985). However, moderate utilization by livestock in spring, early summer, or winter is
15 sustainable in non-degraded meadow and riparian areas within sagebrush habitat (Shaw 1992,
16 Clary et al. 1996, Mosley et al. 1997). Moderate use equates to a 10-cm residual stubble height
17 for most grasses and sedges and 5-cm for Kentucky bluegrass (Mosley et al. 1997, Clary and
18 Leininger 2000). Shrub utilization should not exceed 50-60% during the growing season, and at
19 least 50% protective ground cover (i.e., plant basal area + mulch + rocks + gravel) should remain
20 after grazing (Mosley et al. 1997). While hydrophytic shrubs may not directly serve as sage-
21 grouse habitat, they do impact the stability of riparian and meadow habitats important to sage-
22 grouse (Winward 2000). The length of time livestock have access to meadows may be more
23 important than the level of utilization; it has been suggested that livestock access be limited to <

1 3 weeks (Myers 1989, Mosley et al. 1997). In riparian and meadow habitat degraded by heavy
2 livestock utilization, rest from grazing may be necessary for recovery (Clary and Webster 1989).

3 *Implications to sage-grouse and research needs*

4 It is probably safe to assume that historic grazing practices had strong negative impacts
5 on sage-grouse habitat and perhaps populations (Patterson 1952, Wallestad 1975, Beck and
6 Mitchell 2000), although definitive historical population data do not exist. However, research
7 directly addressing the population-level impact of current livestock grazing practices on sage-
8 grouse is lacking (Connelly et al. 2000). As noted previously, livestock AUM's have decreased
9 and range condition has increased on federal lands since the mid 1900's, however, there has not
10 been a concomitant increase in sage-grouse populations during the same time interval. This does
11 not necessarily indicate a lack of association between grazing and sage-grouse populations, given
12 that 1.) “improved” range condition (mainly increases in perennial bunchgrass abundance)
13 associated with better livestock management practices may or may not equate to improvement in
14 all habitat needs of sage-grouse, 2.) those plant communities displaying steady state dynamics
15 may not change linearly with reduced stocking, 3.) it is unknown what portion of the areas with
16 reduced stocking represent critical sage-grouse habitat, and 4.) the complicated nature of sage-
17 grouse population dynamics may preclude their short-term response to management activities.
18 Additionally, there has also been continued habitat loss through other factors (e.g., annual grass
19 invasion, juniper encroachment, cultivation, road construction, powerline development, etc.).

20 A recent modeling exercise (Wisdom et al. 2002) incorporated 50 and 100% “reductions
21 in the detrimental effects of livestock grazing” into a population level model for sage-grouse in
22 the Interior Columbia Basin. The model predicted improved performance of sage-grouse
23 populations with a combination of active habitat restoration and reduced livestock stocking rate,

1 and equated reductions in livestock stocking rate to decreased detrimental effects of livestock on
2 sage-grouse habitat at a 1 to 1 ratio. While this approach may appear empirically appealing in
3 that it allows “what if” scenario modeling, caution is merited when assuming that reductions in
4 livestock stocking rate are in a constant 1 to 1 ratio with changes in sage-grouse habitat quality,
5 given that the exact slope of this relationship is unknown (it may be substantially greater or less
6 than 1) and is variable in accordance with timing and intensity of livestock grazing,
7 environmental factors, and specific type of sage-grouse habitat (e.g., nesting, brood-rearing,
8 etc.). Given the complexity of the successional dynamics of sagebrush plant communities,
9 combined with the multivariate nature of the effects of livestock grazing on these plant
10 communities, it remains difficult to draw large-scale (time and space) conclusions regarding the
11 impact of current livestock grazing practices on sage-grouse populations.

12 Livestock grazing may positively or negatively affect the structure and composition of
13 sage-grouse habitat. Brood-rearing habitat may be enhanced by grazing practices that favor
14 upland forb production (e.g., fall grazing) and prescribed light (<40%) to moderate spring
15 grazing can remove standing herbage and make forbs more accessible (Smith et al. 1979,
16 Fulgham et al. 1982). However, consumption of forbs by livestock may limit their availability to
17 sage-grouse (Call 1979). In riparian brood-rearing habitat, sage-grouse prefer the lower
18 vegetation (5-15 cm vs. 30-50 cm; Oakleaf 1971, Neel 1980, Klebenow 1982, Evans 1986) and
19 succulent forb growth stimulated by moderate livestock grazing (Neel 1980, Evans 1986).
20 Prescribed livestock grazing in spring and early summer, especially by sheep and goats (*Capra*
21 *hircus*), can help control invasive weeds (Mosley 1996, Olson and Wallander 2001, Merritt et al.
22 2001) and woody plant encroachment (Riggs and Urness 1989) in sage-grouse habitat and may
23 reduce wildfire risks to low elevation plant communities. However, the logistics of applying

1 such grazing treatments at large spatial scales remain difficult.

2 Excessive livestock grazing has negatively impacted sage-grouse habitat by creating seral
3 conditions that favor annual grass dominance and by reducing perennial grasses used as nesting
4 and escape cover (Beck and Mitchell 2000). However, the specific relationship between grazing
5 pressure and sage-grouse nest success has not been empirically evaluated. Heavy use of riparian
6 meadows by livestock reduces the availability of succulent plant species and may induce
7 avoidance of these habitats by sage-grouse (Neel 1980, Klebenow 1982, Klebenow 1985). Nest
8 destruction by livestock trampling is rare, however, the presence of livestock can cause sage-
9 grouse to abandon their nests (Rasmussen and Griner 1938, Patterson 1952, Call 1979).
10 Managers should consider delaying grazing of known nesting areas until after nesting (Beck and
11 Mitchell 2000).

12 Rotational grazing systems are one way to provide areas (i.e., pastures) free from
13 livestock disturbance during nesting. This benefit may be offset if heavy use occurs in the
14 grazed pastures (Holechek et al. 1982), especially since sage-grouse can display high site fidelity
15 (Fischer et al. 1993). One advantage of rest rotation grazing is that rested pastures can provide
16 emergency forage (Ratliff and Reppert 1974), which may prevent excessive grazing in the used
17 pastures during drought. This added residual cover may be important to sage-grouse, but light to
18 moderate utilization of grasses in well-managed continuously grazed systems may also provide
19 sufficient residual cover. Grazing systems in riparian areas have met with mixed results and
20 their influence on system recovery and vegetation response will vary based on site potential,
21 ecological condition, stream morphology, and climate (Elmore and Kauffman 1994). Compared
22 with no grazing, rest rotation grazing increased forb abundance on sage-grouse meadow habitat
23 in Nevada (Neel 1980).

1 Additional research is needed to address the direct effects of livestock grazing
2 management on sage-grouse. Given the limited research base, much of what needs to be done is
3 basic in nature. For example, research is needed to examine the effects of grazing variables such
4 as timing, intensity, frequency, and stock density on sage-grouse habitat use patterns, nest
5 success, and population dynamics. Additionally, research should continue to address the impacts
6 of livestock grazing on patterns of plant succession at multiple space and time scales. This
7 research should include both direct effects, as well as the interactive effects of grazing and
8 abiotic factors (e.g., fire frequency) on plant succession.

9 **Herbicide**

10 Control of sagebrush has impacted large portions of rangeland in the western U.S. By
11 1967, an estimated 12 - 15 million ha of sagebrush had been mechanically treated, sprayed, or
12 burned (Schneegas 1967), and by 1974, over 2 million ha had been treated with herbicide in 6
13 western states (Vale 1974). This practice has been widely associated with declines in sage-
14 grouse habitat quality (Connelly et al. 2000). Much of the research literature has focused on
15 maximum sagebrush kill in strips or blocks, but recent work has examined the impact of
16 selective thinning of sagebrush on wildlife habitat quality (Baxter 1998).

17 Where sagebrush density is high enough to limit understory expression of forbs and
18 grasses, some reduction of sagebrush may be desirable (Laycock 1991). Initial efforts to control
19 sagebrush used 2,4-D [(2,4-dichlorophenoxy) acetic acid], which was effective in suppressing
20 big sagebrush and typically resulted in dramatic increases in herbage production (Orpet and
21 Fisser 1979, Waltenberger et al. 1979, Kearn and Freeburn 1980). The impact of 2,4-D on forb
22 abundance varies by species (Table 3). Concerns over reduced plant diversity following 2,4-D
23 have severely limited its use. Tebuthiuron (N-[5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2-yl]-

1 N,N'-dimethylurea), a photosynthesis inhibitor with soil activity greater than 1 year, was
2 introduced in 1973 and can selectively control big sagebrush at low application rates. Recent
3 studies have demonstrated that big sagebrush canopy cover is reduced in proportion to
4 application rate, with simultaneous progressive increases in understory grass and forb abundance
5 (Whitson and Alley 1984, Whitson et al. 1988, Halstvedt 1994, Olson et al. 1994, Johnson et al.
6 1996, Olson and Whitson 1996, Olson et al. 1996, Olson and Whitson 2002). For example,
7 Halstvedt et al. (1996) reported a 59- 491% increase in native perennial grass production
8 following reduction of pre-treatment big sagebrush canopy cover (25- 35%) to 12- 15% after 10-
9 17 years following tebuthiuron thinning treatments. Forb production increased between 15-
10 127% on treated sites. Herbicides offer some advantages to mechanical manipulation of
11 sagebrush including cost effectiveness, longer treatment life, less damage to non-target shrub
12 species, decreased erosion risk, and better control of the extent of sagebrush kill (Blaisdell et al.
13 1982, Olson et al. 1994, Baxter 1998).

14 ***Implications to sage-grouse and research needs***

15 Block or strip applications of herbicide at rates that severely diminish sagebrush will
16 likely have negative impacts on sage-grouse habitat quality. In lekking habitat, some researchers
17 report decreases in sage-grouse males following sagebrush removal (Wallestad 1975, Connelly et
18 al. 1981), while others have found no clear effect (Gates 1985, Martin 1970, Benson et al. 1991,
19 Fischer 1994). Sage grouse may cease to use block treated areas as nesting habitat (Klebenow
20 1970) and winter habitat degradation is proportional to severity of sagebrush kill (Connelly et al.
21 2000). Increases in forb availability in strip or block-sprayed habitat may increase the value of
22 these areas as brood-rearing habitat (e.g., Autenrieth 1969) but use patterns often indicate
23 avoidance of treated areas (Klebenow 1970, Braun et al. 1977).

1 It has been suggested that thinning treatments can be used as a tool to manage sage-
2 grouse habitat (Beck and Mitchell 2000). Lekking habitat is rarely limited, but in areas where
3 dense, monotypic big sagebrush stands limit suitable lekking grounds, sagebrush thinning can
4 create small open areas for breeding activities. Sage-grouse have been reported to use newly
5 disturbed sites as leks (Connelly et al. 1981). In localized areas, reduced application rates of
6 tebuthiuron can be used to thin big sagebrush cover and increase understory perennial grasses
7 and forbs associated with nesting and brood-rearing habitat. Thinned Wyoming big sagebrush
8 stands may have maximum forb production at 11 to 17% sagebrush canopy cover (Johnson et al.
9 1996). Where sage-grouse winter cover is limited, thinning of big sagebrush should be avoided
10 (Klebenow 1985, Robertson 1991). Connelly et al. (2000) recommended that treatments be
11 limited to <20% of the breeding habitat (depending on sagebrush type) within a 20 to 30-year
12 period, primarily because of concerns over damage to winter habitat. Additional research is
13 needed to further identify the impact of sagebrush thinning on habitat use by sage-grouse.

14 Herbicides can also be used to control invasive annual plant species in sage-grouse
15 habitat. For instance, herbicides may be useful in controlling cheatgrass abundance (Mosley et
16 al. 1999, Pellant et al. 1999). Herbicides such as imazapic (Plateau) applied prior to cheatgrass
17 emergence can be used to release forbs and perennial grasses in the understory (Whitson 2003).
18 Herbicide control of cheatgrass has positive implications to the maintenance of communities in
19 danger of transitioning to annual-dominated states, and, when used in conjunction with native
20 grass seeding, the restoration of sites that have already realized that transition. This latter
21 practice has been suggested as a form of active restoration of sage-grouse habitat degraded by
22 annual grass dominance (Hemstrom et al. 2002).

23

Landscape issues in sage-grouse management and research

1 The foregoing outlines 1) the relatively complex life cycle of sage-grouse; 2) the
2 collective observations of the bird in the many different kinds of habitat it requires throughout
3 the year; and 3) different ways that individual birds and populations can be affected by abiotic,
4 biotic, and management factors. It is important to realize that this information has been
5 accumulated piecemeal (i.e., by many different people working in different places and at
6 different times). Thus, compositing this information to form our understanding and creating
7 guidelines for management (e.g., Connelly et al. 2000) is based largely on data taken from small
8 areas over short times. This process may give a deceptive picture, both generally and
9 specifically, for any given population. For instance, there may be differing causes of mortality in
10 different places, at different times, particularly between migratory and resident populations.
11 Some of the studies could involve shrinking populations, whereas others could involve stable to
12 growing populations. The type of population present in each study needs to be identified
13 because they require different types of management responses.

15 It is now apparent that rather than one, a few to many causes may be synergistically and
16 cumulatively operating to diminish sage-grouse. For instance, reducing predator control may
17 occur simultaneously with undesirable changes in vegetation structure triggered by other factors
18 (e.g., livestock grazing practices, fire control). While many believe that cause and effect
19 mechanisms need to be disentangled for declining sage-grouse populations, others judge that to
20 be neither feasible nor timely. Shrader-Frechette and McCoy (1993) emphasize that solving
21 complex issues, like sustaining sage-grouse populations, will be more tractable in a case study
22 mode.

1 ***Linking habitat and population changes***

2 Researchers and managers have long had a vague, qualitative notion that sage-grouse
3 respond to negative changes within entire landscapes (Connelly et al. 2000), portions of which
4 the birds use at various times of the year. Until recently, however, a ready means of quantifying
5 landscape patterns and change was lacking. Now, geomatics [combined remote sensing (RS),
6 global positioning systems (GPS), and geographic information systems (GIS)], can be employed
7 to give quantitative expressions and visualizations of habitat patterns over large areas of land for
8 the past several decades. Landscape ecology (Turner et al. 2001) provides a logical framework
9 and a new set of tools to examine how spatial arrangements of different kinds of habitat may
10 influence individuals and populations. Intermediate-sized landscapes of 250,000 to 2.5 million
11 ha and their macro-mosaics of ecological sites and stands in various seral stages seem to be the
12 most appropriate scale for management solutions to be successful, since improvements in only
13 part of the year-round habitat may be negated by degradation in other nearby habitat needed at
14 other times. Successful management at the scale of the entire geographical range of the species
15 is unlikely because all races of sage-grouse may not have the same habitat requirements or
16 respond to environmental changes and management in identical ways.

17 By combining landscape ecology and geomatics, it is now possible to characterize both
18 current spatial patterns and changes in these geographic patterns over about the past fifty years
19 by analyzing archived imagery. Data coverage of entire landscapes in the range of 100,000 to
20 250,000 ha is now feasible, as illustrated in the recent work of Washington-Allen (2003) at the
21 Deseret Ranch in northeastern Utah. Patches of land can be characterized as to cover dominance
22 by plant growth forms and bare ground. Fragmentation and coalescence, patch sizes and
23 boundary shapes and proximity to similar and dissimilar patches can be tracked over time.

1 Underlying GIS layers dealing with management and disturbance history, along with soils,
2 ecological site [as provided by Natural Resource Conservation Agency (NRCS) databases] and
3 seral status can be connected to indicators of sage-grouse abundance. A similar approach at large
4 scales (1km² pixels) was recently used to characterize habitat changes in the Interior Columbia
5 Basin (Hemstrom et al. 2002).

6 Sage-grouse is not a species that can thrive only where large homogeneous stands of any
7 single plant species occupy the bulk of the landscape. While recommendations exist for the
8 kinds of habitats that are preferred at different times in the life cycle of the bird (Connelly et al.
9 2000), the proportions of habitats that are optimum or even tolerable remains unknown. It is
10 likely that sage-grouse are responding to habitat attributes at multiple scales while other
11 sagebrush obligates may be responding at different scales. These questions can now be
12 addressed by applying concepts from landscape ecology linked through geomatics technology.
13 This will allow natural resource professionals to break away from their traditions of collecting
14 only short runs of point-based data focused on either livestock or wildlife species and the plant
15 communities in which they are found.

16 We suggest that areas where sage-grouse have recently diminished be collectively
17 identified. From archived aerial photos and satellite imagery, fundamental changes in the
18 landscape can be quantified via time series of landscape metrics (Turner et al. 2001) and other
19 RS/GIS based indicators (Washington-Allen 2003). Another useful effort would be to
20 characterize and compare the landscapes holding non-migratory populations to those that are
21 migratory. The existence of non-migratory populations implies that all habitat requirements of
22 sage-grouse are sometimes met in one relatively small area. In these areas, the mix of habitats
23 and their proportions needs to be defined. Similar studies involving lesser prairie chickens

1 diversity of management and research professionals working in concert to solve multifaceted
2 problems.

3 In a broader spatial and temporal context, the sage-grouse decline may be symptomatic of
4 long-term regional level problems. While sage-grouse are currently at the center of ecological
5 and political concern, other species, mainly sagebrush obligates [e.g., Brewer's sparrow (*Spizella*
6 *pallida*), sage sparrow (*Amphispiza belli*), sage thrasher (*Oreoscoptes montanus*), pygmy rabbit
7 (*Brachylagus idahoensis*), sagebrush vole (*Lagurus curtatus*)], are also declining, suggesting that
8 "fixing" the sage-grouse problem should be synonymous with improvements at the ecosystem
9 level. This point may be of particular importance given the burgeoning number of local and
10 state-level sage-grouse working groups in the western US. If system level problems are not
11 addressed now, the efforts of these working groups will have to be repeated for other sagebrush
12 obligates, as additional species take political front and center over time. Regional level dialogue
13 and planning should be facilitated by the Interagency Sage Grouse Conservation Framework
14 Team, which provides an effective linkage between state level conservation efforts. Access to
15 the data needed to make large-scale decisions has been aided by the SAGEMAP project
16 (<http://sagemap.wr.usgs.gov>), which serves as a storehouse for spatial data pertaining to
17 conservation of sage-grouse and sagebrush-dominated plant communities.

18 Natural resource professionals of all walks should strive to keep an open mind regarding
19 the potential structure of plant communities serving as sage-grouse habitat. A good example is
20 the amount of sagebrush cover a given community can be expected to produce. It is quite
21 probable that differences of opinion on this matter are due to differences in vegetation sampling
22 methodology. In such cases, published ecological site information can be a useful intermediary
23 for helping find common ground (e.g., NRCS data, Tisdale et al. 1965, Winward 1970, Mueggler

1 and Stewart 1980). Natural resources professionals should also consider that without purposeful
2 habitat management (e.g., prescribed fire-based juniper control) successional changes may
3 decrease the value of some plant communities as sage-grouse habitat. Active management will
4 likely be required to address the problem of annual grass invasion in sage-grouse habitat; a
5 dilemma for which there is not currently a definitive solution over large scales.

6 Bringing together groups of professionals (e.g., range and wildlife specialists) in an
7 effective manner involves coordinated planning. One potential avenue of cooperation would
8 involve re-visiting past sage-grouse research efforts. If the precise geographic locations of these
9 projects could be obtained, rangeland scientists could work in concert with wildlife scientists to
10 identify big sagebrush subspecies and serally interpret the vegetation structure preferred by sage-
11 grouse. This information could then be used as the basis of a succession-based model for
12 predicting management impacts, and planning habitat manipulations. Such efforts must have
13 active participation from both management and research entities; without management buy-in,
14 significant amounts of time and energy can be wasted developing models that will never be used.
15 As was previously pointed out, this is not a process that will produce quick results. Instead, time
16 is needed for management actions to produce changes in habitat and other environmental
17 variables before the impacts on sage-grouse populations can be manifested. Thus, it is
18 imperative that such efforts be initiated as soon as possible.

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