

Ecological site group R023XY918NV

Mountain Slope 10-12 PZ Utah Juniper with Thurber's Needlegrass and Sagebrush.

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Key Characteristics

- Site does not pond or flood
- Landform other than dunes
- Surface soils are not clayey
- Sites are tree dominated
- Elevations < 7000'
- Soils loamy or ashy
- Frost free days per year \geq 100

Provisional. A provisional ecological site description has undergone quality control and quality assurance review. It contains a working state and transition model and enough information to identify the ecological site.

Physiography

This group is on mountain slopes at elevations from 4,500 to 6,500 feet. Slopes are 10 to 75 percent.

Climate

The climate is classified as Cold Semi-Arid in the Koppen Classification System.

The area receives 8 to 16 inches of annual precipitation as snow in the winter and rain in the spring and fall. Summers are generally dry.

The frost-free period is 90 to 105 days.

Soil features

The soils in this group are very shallow to fractured bedrock, with a clayey-skeletal texture.

Taxonomically, the soils are Aridisols or Entisols, and the soil temperature regime is mesic to frigid.

The common soil series in this group are Coppereid, Burrita, and Sumya.

Vegetation dynamics

Ecological Dynamics and Disturbance Response:

An ecological site is the product of all the environmental factors responsible for its development. Each site has a set of key characteristics that influence its resilience to disturbance and resistance to invasives. According to Caudle et al. (2013), key characteristics include:

1. Climate factors such as precipitation and temperature.
2. Topographic characteristics such as aspect, slope, elevation, and landform.
3. Hydrologic processes such as infiltration and runoff.
4. Soil characteristics such as depth, texture, structure, and organic matter.

5. Plant communities and their associated functional groups and productivity.
6. Natural disturbance (fire, herbivory, etc.) regime.

Biotic factors that influence resilience include site productivity, species composition and structure, and population regulation and regeneration (Chambers et al., 2013).

Pinyon- and juniper-dominated plant communities in the cold desert of the Intermountain West occupy over 18 million hectares (44,600,000 acres) (Miller & Tausch, 2001). In the mid- to late 1900s, the number of pinyon and juniper trees establishing per decade began to increase compared to the previous several hundred years. The substantial increase in conifer establishment is attributed to a number of factors. These factors include:

1. Cessation of aboriginal burning (Tausch, 1999).
2. Change in climate due to rising temperatures (Heyerdahl et al., 2008).
3. Reduced fire frequency, likely driven by the introduction of domestic livestock.
4. Decreased wildfire frequency due to improved wildfire suppression efforts.
5. Potentially increased CO₂ levels that favor woody plant establishment (Tausch, 1999; Bunting, 1994).

Miller et al. (2008) found pre-settlement tree densities averaged 2 to 11 trees per acre in six woodlands studied across the Intermountain West. Current stand densities are 80 to 358 trees per acre. In Utah, Nevada, and Oregon, trees that established prior to 1860 account for only two percent or less of the total population of pinyon and juniper (R. Miller et al., 1999; Miller & Tausch, 2001; Miller et al., 2008). The research strongly suggest that for over 200 years prior to settlement, woodlands in the Great Basin were relatively low density with limited rates of establishment (Miller & Tausch, 2001; Miller et al., 2008). Tree canopy cover of 10 to 20 percent may be more representative of these sites in pristine condition. Increases in pinyon and juniper densities post-settlement were the result of both infill in mixed-age tree communities and expansion into shrub-steppe communities. However, the proportion of old growth varies depending on disturbance regimes, soils, and climate. Some ecological sites can support persistent woodlands, likely due to specific soils and climate that results in infrequent stand-replacing disturbances. In the Great Basin, old-growth trees typically grow on rocky, shallow, or sandy soils that support little understory vegetation to carry a fire (Burkhardt & Tisdale, 1976; Holmes et al., 1986; West et al., 1998; Miller & Rose, 1995).

Utah juniper (*Juniperus osteosperma*) is a long-lived tree species with wide ecological amplitudes (Tausch et al., 1981; West et al., 1998; Weisberg & Ko, 2012). Maximum ages of pinyon and juniper exceed 1,000 years and stands with maximum age classes are only found on steep, rocky slopes with no evidence of fire (West et al., 1975).

Juniper growth depends mostly upon soil moisture stored from winter precipitation, mainly snow. Much of the summer precipitation is ineffective because it is lost either through runoff after summer convection storms or through evaporation and interception (Tueller & Clark, 1975). Juniper is highly resistant to drought, which is common in the Great Basin. Taproots of juniper have a relatively rapid rate of root elongation and are thus able to persist until precipitation conditions are more favorable (Emerson, 1932).

Infilling by younger trees increases canopy cover, which causes understory perennial vegetation to decrease because of increased competition for water and sunlight. Additionally, there is evidence that phenolic compounds in juniper litter may have allelopathic effects on grass (Jameson, 1970). Furthermore, infilling shifts stand level biomass from ground fuels to canopy fuels, which has the potential to significantly impact fire behavior. The more tree-dominated pinyon and juniper woodlands become, the less likely they are to burn under moderate conditions, resulting in infrequent, high-intensity fires (Gruell, 1999; Miller et al., 2008). Additionally, as the vigor of understory vegetation declines, the ability of native perennial plants to recover after fire decreases (Urza et al., 2017). The increase in bare ground allows for the invasion of non-native annual species such as cheatgrass (*Bromus tectorum*), and with intense wildfire there is a serious threat that the site could potentially convert to a site dominated by annual exotics (Tausch, 1999; Miller et al., 2008).

Specific successional pathways after disturbance in juniper stands depend on a number of variables such as plant species present at the time of disturbance, individual species response to disturbance, past management, type and size of disturbance, available seed sources in the soil or adjacent areas, and site and climatic conditions throughout the successional process.

Insects and diseases of western juniper (*Juniperus occidentalis*) are not well understood or studied (Eddleman et al., 1994). A fungus called Juniper Pocket Rot (*Pyrofomes demidoffi*), also known as white trunk rot (Eddleman et al., 1994; Durham, 2014), can kill Utah juniper. Pocket rot enters the tree through any wound or opening that exposes the heartwood. In an advanced stage, this fungus can cause high mortality (Durham, 2014). Dwarf mistletoe (*Arceuthobium* spp.) is a parasitic plant that may also affect Utah juniper. Without treatment or pruning, it may kill the tree 10 to 15 years after infection. Seedlings and saplings are most susceptible to the parasite (Christopherson, 2014). Other diseases and pests that affect juniper include:

1. Witches' broom (*Gymnosporangium* sp.): Girdles and kills branches.
2. Leaf rust (*Gymnosporangium* sp.): Affects leaves and young branches.
3. Juniper blight (*Phomopsis* sp.).
4. Flat-head borers (*Chrysobothris* sp.): Attack the wood (Tueller & Clark, 1975).
5. Long-horned beetles (*Methia juniperi*, *Styloxus bicolor*) and round-head borers (*Callidium* spp.): Girdle branches and can kill branches or entire trees (Tueller & Clark, 1975).

Low sagebrush (*Artemisia arbuscula*) is fairly drought-tolerant. It also tolerates periodic wetness during some portion of the growing season (Fosberg & Hironaka, 1964; Blackburn et al., 1968a, 1968b, 1969). It grows on soils that have a strongly structured B2t (argillic) horizon close to the soil surface (Winward, 1980; Fosberg & Hironaka, 1964; Zamora & Tueller, 1973). Low sagebrush is also susceptible to the sagebrush defoliator known as the Aroga moth. Aroga moth can partially or entirely kill individual plants or entire stands of big sagebrush (*Artemisia tridentata*) (Furniss & Barr, 1975), but research regarding the damage sustained by low sagebrush populations is inconclusive.

The ecological sites in this group are dominated by deep-rooted, cool-season, perennial bunchgrasses and long-lived shrubs (at least 50 years old) with high root to shoot ratios. The dominant shrubs usually root to the full depth of the winter-spring soil moisture recharge, which is 1.0 to over 3.0 meters (Dobrowolski et al., 1990). Root length of mature sagebrush plants was measured to a depth of 2 meters in alluvial soils in Utah (Richards & Caldwell, 1987). However, community types with low sagebrush as the dominant shrub have soil depths—and thus available rooting depths—of 71 to 81 centimeters in a study in northeast Nevada (Jensen, 1990). These shrubs have a flexible generalized root system with development of both deep taproots and laterals near the surface (Comstock & Ehleringer, 1992).

In the Great Basin, most of the annual precipitation is received during the winter and early spring. This continental semiarid climate regime favors the growth and development of deep-rooted shrubs and herbaceous, cool-season plants using the C3 photosynthetic pathway (Comstock & Ehleringer, 1992). Winter precipitation and slow melting of snow results in deeper percolation of moisture into the soil profile. Herbaceous plants, more shallow-rooted than shrubs, grow earlier in the growing season and thrive on spring rains, while the deeper-rooted shrubs lag in phenological development because they draw from deeply infiltrating moisture from snowmelt the previous winter. Periodic drought regularly influences sagebrush ecosystems and drought duration and severity have increased throughout the 20th century in much of the Intermountain West. Major shifts away from historical precipitation patterns have the greatest potential to alter ecosystem function and productivity. Species composition and productivity can be altered by the timing of precipitation and water availability within the soil profile (Bates et al., 2006).

The ecological sites in this group have low to moderate resilience to disturbance and resistance to invasion. Resilience increases with elevation, northerly aspect, precipitation, and nutrient availability. Four possible states have been identified for this group.

Annual Invasive Grasses:

The species most likely to invade these sites are cheatgrass and medusahead (*Taeniatherum*). Both species are cool-season annual grasses that maintain an advantage over native plants in part because they are prolific seed producers, able to germinate in the autumn or spring, tolerant of grazing, and increase when fire is frequent (Klemmedson & Smith, 1964; H. Miller et al., 1999). Medusahead and cheatgrass originated from Eurasia and both were first reported in North America in the late 1800s (Mack & Pyke, 1983; Furbush, 1953). Pellant and Hall (1994) found 3.3 million acres of public lands dominated by cheatgrass and suggested that another 76 million acres were susceptible to invasion by winter annuals including cheatgrass and medusahead. By 2003, medusahead occupied approximately 2.3 million acres in 17 western states (Rice, 2005). In the Intermountain West, the exponential

increase in dominance by medusahead has largely been at the expense of cheatgrass (Harris, 1967; Hironaka, 1994). Medusahead matures 2 to 3 weeks later than cheatgrass (Harris, 1967). Recently, James et al. (2008) measured leaf biomass over the growing season and found that medusahead maintained vegetative growth later in the growing season than cheatgrass. Mangla et al. (2011) also found medusahead had a longer period of growth and more total biomass than cheatgrass and hypothesized this difference in relative growth rate may be due to the ability of medusahead to maintain water uptake as upper soils dry compared to co-occurring species, especially cheatgrass. Medusahead litter has a slow decomposition rate because of its high silica content, allowing it to accumulate over time and suppress competing vegetation (Bovey et al., 1961; Davies & Johnson, 2008). Harris (1967) reported medusahead roots have thicker cell walls compared to those of cheatgrass, allowing it to more effectively conduct water, even in very dry conditions.

Recent modeling and empirical work by Bradford and Lauenroth (2006) suggest that seasonal patterns of precipitation input and temperature are also key factors that influence regional variation in the growth, seed production, and spread of invasive annual grasses. Collectively, the body of research suggests that the invasion and dominance of medusahead onto native grasslands and cheatgrass-infested grasslands will continue to increase in severity because conditions that favor native bunchgrasses or cheatgrass over medusahead are rare (Mangla et al., 2011). Medusahead replaces native vegetation and cheatgrass directly by competition and suppression; it replaces native vegetation indirectly by increasing fire frequency.

Methods to control medusahead and cheatgrass include herbicide, fire, grazing, and seeding of primarily non-native wheatgrasses. Mapping potential or current invasion vectors is a management method designed to increase the cost effectiveness of control methods. A study by Davies et al. (2013) found an increase in medusahead cover near roads. Cover was higher near animal trails than random transects but the difference was less evident. This implies that vehicles and animals aid the spread of the weed; however, vehicles are the major vector of movement. Spraying with herbicide (Imazapic or Imazapic + glyphosate) and seeding with crested wheatgrass (*Agropyron cristatum*) and Sandberg bluegrass (*Poa secunda*) have been more successful at combating medusahead and cheatgrass than spraying alone (Sheley et al., 2012). Where native bunchgrasses are missing from the site, revegetation of medusahead- or cheatgrass-invaded rangelands has shown a higher likelihood of success when using introduced perennial bunchgrasses such as crested wheatgrass (Davies et al., 2015). Butler et al. (2011) tested four herbicides (Imazapic, Imazapic + glyphosate, rimsulfuron, and sulfometuron + Chlorsulfuron), using herbicide-only treatments, for suppression of cheatgrass, medusahead, and ventenata (*Ventenata dubia*) within residual stands of native bunchgrass. Additionally, they tested the same four herbicides followed by seeding of six bunchgrasses (native and non-native) with varying success. Herbicide-only treatments appeared to remove competition for established bluebunch wheatgrass (*Pseudoroegneria spicata*) by providing 100 percent control of ventenata and medusahead and greater than 95 percent control of cheatgrass. However, caution in using these results is advised, as only one year of data was reported.

Prescribed fire has also been utilized in combination with the application of pre-emergent herbicide to control medusahead and cheatgrass (J. L. Vollmer & J. G. Vollmer, 2008). Mature medusahead or cheatgrass is very flammable and fire can be used to remove the thatch layer, consume standing vegetation, and even reduce seed levels. Furbush (1953) reported that timing a burn while the seeds were in the milk stage effectively reduced medusahead the following year. He further reported that adjacent unburned areas became a seed source for reinvasion the following year.

When considering the combination of pre-emergent herbicide and prescribed fire for invasive annual grass control, it is important to assess the tolerance of desirable brush species to the herbicide being applied. J. L. Vollmer and J. G. Vollmer (2008) tested the tolerance of mountain mahogany (*Cercocarpus montanus*), antelope bitterbrush (*Purshia tridentata*), and multiple sagebrush species to three rates of Imazapic and the same rates with methylated seed oil as a surfactant. They found a cheatgrass control program in an antelope bitterbrush community should not exceed Imazapic at 8 ounces per acre with or without surfactant. Sagebrush, regardless of species or rate of application, was not affected. However, many environmental variables were not reported in this study and managers should install test plots before broad scale herbicide application is initiated.

Fire Ecology:

Large fires were rare on these sites. Lightning-ignited fires were common but typically did not affect more than a few individual trees. Replacement fires were uncommon to rare, returning every 100 to 600 years, and occurred primarily during extreme fire behavior conditions. Creeping, low-intensity surface fires had a very limited role in

molding stand structure and dynamics. Surface spread of fire was more likely to occur in higher-density woodlands on more productive sites (Romme et al., 2009). Pre-settlement fire return intervals in the Great Basin National Park, Nevada averaged 50 to 100 years; north-facing slopes burned every 15 to 20 years, and rocky landscapes with sparse understories burned very infrequently (Gruell, 1999). Woodland dynamics are largely attributed to fire size, fire return intervals, and long-term climatic shifts in temperature and the amount and distribution of precipitation (Miller & Tausch, 2001). Data that describes fire histories across woodlands in the Great Basin are limited. The infilling of younger trees into the old-growth stands and the expansion of trees into the surrounding sagebrush steppe ecological sites increase the abundance and landscape level continuity of fuels, which, as a result, increase fire severity and size; this, in turn, increases the risk of loss of pre-settlement trees (Miller et al., 2008).

Low sagebrush is killed by fire and does not sprout (Tisdale & Hironaka, 1981). Fire risk is greatest following a wet, productive year when there are more fine fuels (Beardall & Sylvester, 1976). Fire return intervals are not well understood because these ecosystems rarely coincide with fire-scarred conifers, but researchers estimate the return intervals are 20 to well over 100 years (Miller & Rose, 1995, 1999; Baker, 2006; Knick et al., 2005).

Historically, fires were probably patchy due to the low productivity of these sites (Beardall & Sylvester, 1976; Ralphs & Busby, 1979; Wright et al., 1979; Smith & Busby, 1981). Fine fuel loads generally average 100 to 400 pounds per acre (110 to 450 kilograms per hectare) but are occasionally as high as 600 pounds per acre (680 kilograms per hectare) in low sagebrush habitat types (Bradley et al., 1992). Reestablishment occurs from off-site wind-dispersed seed (Young, 1983). Recovery time of low sagebrush following fire is variable (Young, 1983). After fire, if regeneration conditions are favorable, low sagebrush recovers in 2 to 5 years. On harsh sites where cover is low to begin with and/or erosion occurs after fire, recovery may require more than 10 years (Young, 1983). Slow regeneration may subsequently worsen erosion (Blaisdell et al., 1982). We were unable to find any substantial research on success of seeding low sagebrush after fire. To date, we have not been able to find specific research on the fire response of Lahontan sagebrush.

Utah juniper is usually killed by fire and is most vulnerable to fire when it is under 4 feet tall (Bradley et al., 1992). Larger trees, because they have foliage farther from the ground and thicker bark, can survive low-severity fires but mortality occurs when 60 percent or more of the crown is scorched (Jameson, 1966). With the low production of the understory vegetation, high-severity fires on these sites are unlikely and rarely become crown fires (Bradley et al., 1992; Miller & Tausch, 2001). Tree density on these sites increases with management that focuses on fire suppression, and with grazing management that favors the removal of fine fuels. Fire severity is likely to increase when cheatgrass increases in the understory. Utah juniper reestablishes by seed from nearby seed sources or surviving seeds. Utah juniper begins to produce seed at about 30 years old (Bradley et al., 1992). Seeds establish best under a nurse plant such as sagebrush and rabbitbrush (Everett & Ward, 1984; Tausch & West, 1988; Bradley et al., 1992). Utah juniper woodlands reach maturity 85 to 150 years after fire (Barney & Frischknecht, 1974; Tausch & West, 1988).

The effect of fire on bunchgrasses relates to culm density, culm-leaf morphology, and the size of the plant. The initial condition of bunchgrasses on a site and the seasonality and intensity of the fire all factor into the individual species response. For most forbs and grasses, the growing points are located at or below the soil surface. This provides relative protection from disturbances that decrease aboveground biomass, such as grazing or fire. Thus, fire mortality is more correlated to duration and intensity of heat, which is related to culm density, culm-leaf morphology, size of plant, and abundance of old growth (Wright, 1971; Young, 1983).

Burning decreases the vegetative and reproductive vigor of Thurber's needlegrass (*Achnatherum thurberianum*) (Uresk et al., 1976). Fire can cause high mortality and reduces basal area and yield of Thurber's needlegrass (Britton et al., 1990). The fine leaves and densely tufted growth form make this grass susceptible to subsurface charring of the crowns (Wright & Klemmedson, 1965). Although timing of fire highly influences the response and mortality of Thurber's needlegrass, smaller bunch sizes are less likely to be damaged by fire (Wright & Klemmedson, 1965). Fall prescribed burns did not significantly affect cover of Thurber's needlegrass over the course of two years, which indicates that fall fire is not detrimental to this plant (Davies & Bates, 2008). Thurber's needlegrass often survives fire and continues growth or regenerates from tillers when conditions are favorable (Britton et al., 1990; Koniak, 1985). Reestablishment on burned sites is relatively slow due to low germination and competitive ability (Koniak, 1985). Cheatgrass is a highly successful competitor with seedlings of this needlegrass and may preclude its reestablishment (Evans & Young, 1978). Thurber's needlegrass decreased in density following a spring fire, but it produced more reproductive culms the year after a fall fire (Ellsworth & Kauffman, 2010).

Desert needlegrass (*Achnatherum speciosum*) is similar to Thurber's needlegrass in that both are easily killed by fire. Desert needlegrass does not germinate well in the presence of non-native annual species such as cheatgrass. Herbicidal treatment of cheatgrass prior to desert needlegrass seeding can help establishment (Rafferty, 2000).

Fire removes aboveground biomass from bluebunch wheatgrass but plant mortality is generally low (Robberecht & Defossé, 1995) because the buds are underground (Conrad & Poulton, 1966) or protected by foliage. Uresk et al. (1976) reported burning increased vegetative and reproductive vigor of bluebunch wheatgrass. Thus, bluebunch wheatgrass experiences slight damage from fire but is more susceptible to fire damage in drought years (Young, 1983). Plant response will vary depending on season, fire severity, fire intensity, and post-fire soil moisture availability.

The grasses likely to invade the sites of this group are cheatgrass and medusahead. These invasive grasses displace desirable perennial grasses, reduce livestock forage, and accumulate large fuel loads that foster frequent fires (Davies & Svejcar, 2008). Invasion by annual grasses can alter the fire cycle by increasing fire size, fire season length, rate of spread, numbers of individual fires, and likelihood of fires spreading into native or managed ecosystems (D'Antonio & Vitousek, 1992; Brooks et al., 2004). While historical fire return intervals are estimated at 15 to 100 years, fire return intervals in areas dominated by cheatgrass are estimated at 3 to 5 years (Whisenant, 1990). The mechanisms by which invasive annual grasses alter fire regimes likely interact with climate. For example, cheatgrass cover and biomass vary with climate (Chambers et al., 2007) and are promoted by wet and warm conditions during the fall and spring. Invasive annuals can take advantage of high nitrogen availability following fire because they have higher growth rates and increased seedling establishment relative to native perennial grasses (Monaco et al., 2003).

Livestock/Wildlife Grazing Interpretations:

This group of ecological sites is suitable for grazing. Grazing management considerations include timing, duration and intensity of grazing, and other disturbances that may change the resiliency and resistance of an ecological site. In addition, old-growth juniper stands provide habitats for a variety of plant and animal species. Bird surveys indicate that the highest abundance and diversity of songbirds occur in shrub steppe communities adjacent to old-growth stands (Reinkensmeyer et al., 2007) but may decline when canopy closure decreases understory complexity (Miller et al., 2005).

Domestic sheep and, to a much lesser degree, cattle consume low sagebrush particularly during the spring, fall, and winter (Sheehy & Winward, 1981). Heavy dormant season grazing by sheep reduces sagebrush cover and increases grass production (Laycock, 1967). Severe trampling damage to supersaturated soils could occur if sites are used in early spring when there is abundant snowmelt. Trampling damage, particularly from cattle or horses, in low sagebrush habitat types is greatest on wet soils that have high clay content. On drier areas with more gravelly soils, serious trampling damage does not occur, even when the soils are wet (Hironaka et al., 1983). Bunchgrasses, in general, best tolerate light grazing after seed formation. Britton et al. (1990) observed the effects of clipping date on basal area of five bunchgrasses in eastern Oregon and found grazing from August to October (after seed set) has the least impact. Heavy grazing during the growing season reduces perennial bunchgrasses and increases sagebrush. Abusive grazing by cattle or horses likely increases low sagebrush, rabbitbrush (*Chrysothamnus* spp.), and deep-rooted perennial forbs such as arrowleaf balsamroot (*Balsamorhiza sagittata*). Annual non-native weedy species such as cheatgrass, mustards, and potentially medusahead (*Taeniatherum caput-medusae*) may invade.

Throughout two years of site visits, Lahontan sagebrush was observed in a heavily-browsed state on several ecological sites in this group. This recently differentiated subspecies of low sagebrush (Winward & McArthur, 1995) is moderately to highly palatable to browse species (Rosentreter, 2005; McArthur, 2005).

The literature is unclear as to the palatability of Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*). Generally, Wyoming sagebrush is the least palatable of the big sagebrush subspecies (Bray et al., 1991; Sheehy & Winward, 1981). It may receive light or moderate use depending upon the amount of understory herbaceous cover (Tweit & Houston, 1980). Personius et al. (1987) found Wyoming big sagebrush and basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*) are intermediately palatable to mule deer when compared to mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*), most palatable, and black sagebrush (*Artemisia nova*), least palatable.

Antelope bitterbrush, although a minor component on these sites, but is a critical browse species for mule deer, antelope, and elk and is often utilized heavily by domestic livestock (Wood et al., 1995). Grazing tolerance depends

on site conditions (Garrison, 1953) and the shrub can be severely hedged during the dormant season for grasses and forbs.

Needlegrasses in general are valuable forage for both livestock and wildlife. They are grazed closely when the leaves are green in early spring but are usually avoided once seeds mature (Sampson et al., 1951). Thurber's needlegrass is an important forage source for livestock and wildlife in the arid regions of the West (Ganskopp, 1988). The seeds are apparently not injurious, but grazing animals avoid them when the seeds begin to mature. Sheep, however, have been observed grazing the leaves closely, leaving stems untouched (Eckert & Spencer, 1987). Heavy grazing during the growing season can reduce the basal area of Thurber's needlegrass (Eckert & Spencer, 1987). This suggests that both seasonality and utilization are important factors in the management of this plant. A single defoliation, particularly during the boot stage, can reduce herbage production and root mass, thus potentially lowering the competitive ability of this needlegrass (Ganskopp, 1988). Thurber's needlegrass may increase in crude protein content after grazing (Ganskopp et al., 2007).

Bluebunch wheatgrass is moderately grazing-tolerant and is very sensitive to defoliation during the active growth period (Blaisdell & Pechanec, 1949; Laycock, 1967; Anderson & Scherzinger, 1975). In studies, herbage and flower stalk production were reduced with clipping at all times during the growing season; clipping was most harmful, however, during the boot stage (Blaisdell & Pechanec, 1949; Britton et al., 1990) Tiller production and growth of bluebunch wheatgrass can be greatly reduced when clipping is coupled with drought (Busso & Richards, 1995). Mueggler (1975) estimated that low-vigor bluebunch wheatgrass may need up to 8 years rest to recover. Although an important forage species, it is not always the preferred species by livestock and wildlife.

Inappropriate grazing practices can be tied to the success of medusahead, but eliminating grazing will not eradicate medusahead if it is already present (Wagner et al., 2001). Sheley and Svejcar (2009) reported that even moderate defoliation of bluebunch wheatgrass resulted in increased medusahead density. They suggested that disturbances such as plant defoliation limit soil resource capture, which creates an opportunity for exploitation by medusahead. Avoidance of medusahead by grazing animals allows medusahead populations to expand. This creates seed reserves that can infest adjoining areas and affect the fire regime. Medusahead replaces native vegetation and cheatgrass directly by competition and suppression; it replaces native vegetation indirectly by an increase in fire frequency.

Medusahead litter has a slow decomposition rate because of its high silica content, allowing it to accumulate over time and suppress competing vegetation (Bovey et al., 1961; Davies & Johnson, 2008).

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Major Land Resource Area

MLRA 023X

Malheur High Plateau

Subclasses

- F023XY045NV–JUOS/ARARL3/ACTH7/ACSP12
- F023XY046NV–JUOS/ARTRTW8/ACTH7/ACSP12

Correlated Map Unit Components

21590900, 21590679, 21729557, 21730198, 21730199

Stage

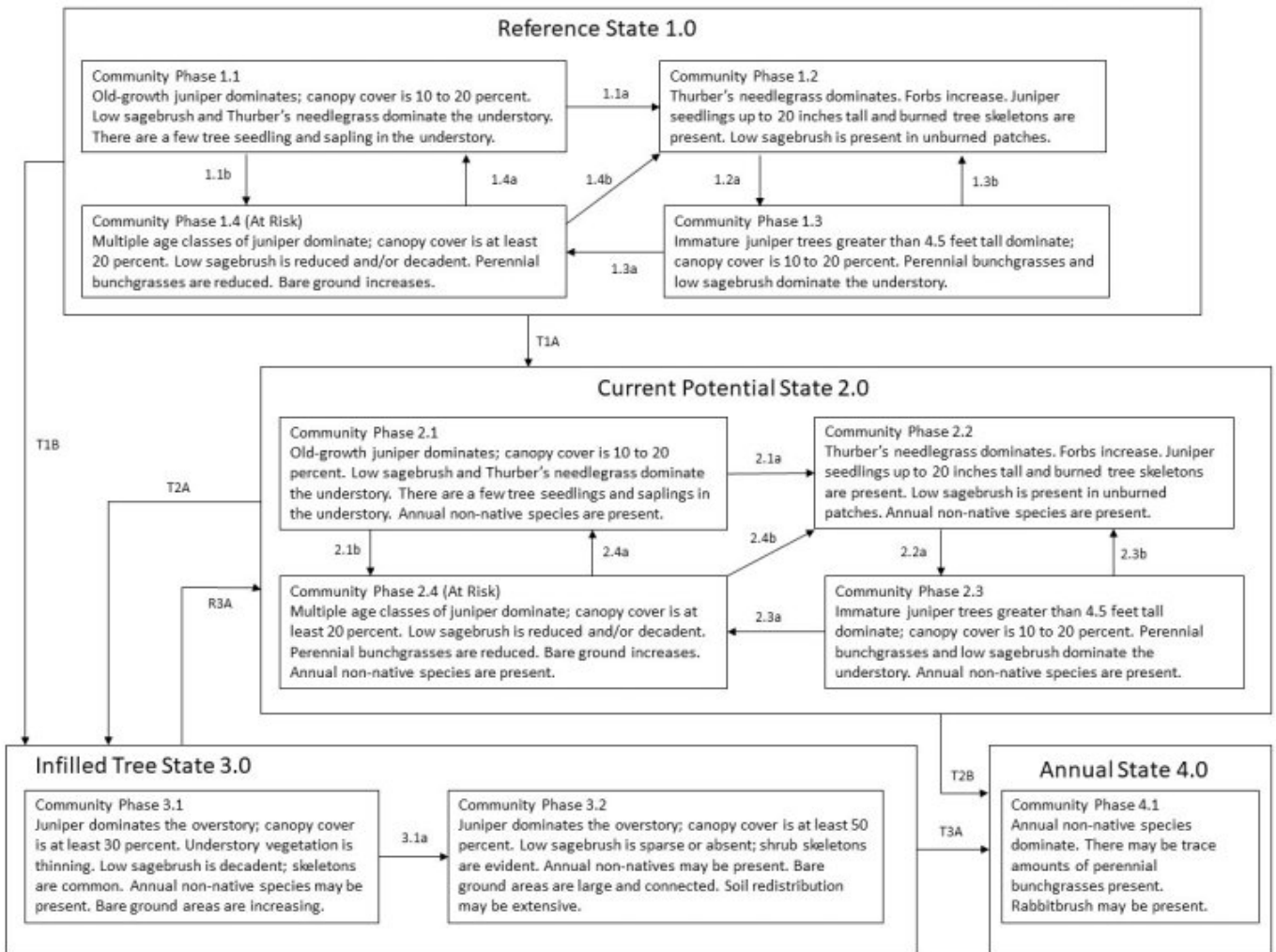
Provisional

Contributors

T Stringham (UNR under contract with BLM)

DMP

State and transition model



Reference State 1.0 Community Phase Pathways

1.1a: High-severity crown fire reduces or eliminates tree cover.

1.1b: Time and lack of disturbance allow younger trees to infill.

1.2a: Time and lack of disturbance allow trees to reestablish. Excessive herbivory may also facilitate this pathway by reducing the perennial grass understory.

1.3a: Time and lack of disturbance allow trees to mature. Excessive herbivory may also facilitate this pathway by reducing the perennial grass understory.

1.3b: Fire reduces or eliminates tree cover.

1.4a: Low-severity fire, insect infestation, or disease removes individual trees and reduces total tree cover.

1.4b: High-severity crown fire reduces or eliminates tree cover.

Transition T1A: This transition occurs following the introduction of non-native annual species.

Transition T1B: Time and a lack of disturbance, which may be coupled with inappropriate grazing management that favors dominance of shrubs and trees, allow trees to dominate site resources.

Current Potential State 2.0 Community Phase Pathways

2.1a: High-severity crown fire reduces or eliminates tree cover.

2.1b: Time and lack of disturbance allow younger trees to infill.

2.2a: Time and lack of disturbance allow trees to reestablish. Excessive herbivory may also facilitate this pathway by reducing the perennial grass understory.

2.3b: Fire reduces or eliminates tree cover.

2.4a: Low-severity fire, insect infestation, or disease removes individual trees and reduces total tree cover.

2.4b: High-severity crown fire reduces or eliminates tree cover.

Transition T2A: Time and a lack of disturbance, which may be coupled with inappropriate grazing management that favors dominance of shrubs and trees, allow for trees to dominate site resources.

Transition T2B: Catastrophic fire causes this transition.

Infilled Tree State 3.0 Community Phase Pathways

3.1a: Time and lack of disturbance such as fire, disease, or drought allow younger trees to infill and mature.

Transition T3A: Catastrophic fire causes this transition.

Restoration R3A: Restoration may be accomplished by thinning trees and seeding. Successful restoration of Community Phase 3.2 is unlikely.

Annual State 4.0 Community Phase Pathways

None.

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