# Ecological site group F023XY920NV
# Mountain Slopes 10-14 PZ Juniper and Mountain Big 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 < 100
- Soils very shallow (depth ≤ 12")

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 backslopes at elevations between 5,500 and 7,500 feet. Slopes are 10 to 75 percent.

## Climate

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

The area receives 10 to 14 inches of annual precipitation as snow in the winter and rain in spring and fall. Summers are generally dry.

The frost-free period is 70 to 100 days. The mean annual air temperature is 40 to 50 °F.

## Soil features

Soils in this group are loamy and very shallow to shallow. They have a mesic temperature regime and classify as Mollisols. Available water holding capacity is moderate to low, but trees and shrub extend their roots into fractures in the bedrock allowing them to utilize deep moisture (Miller & Rose, 1999).

Common soil series in this ecological site group are Berit, Fiddler, Deanran, and Marepas.

## Vegetation dynamics

Understory Dynamics:

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 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).

Juniper:

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 several factors. There factors include:

1. Cessation of the aboriginal burning (Tausch, 1999).
2. Change in climate with rising temperatures (Heyerdahl et al., 2008).
3. The reduced frequency of fire, likely driven by the introduction of domestic livestock.
4. A decrease in wildfire frequency along with improved wildfire suppression efforts.
5. Potentially increased $CO_2$ levels favoring woody plant establishment (Tausch, 1999; Bunting, 1994).

Miller et al. (2008) found pre-settlement tree densities averaged 2 to 11 trees per acre in 6 woodlands studied across the Intermountain West. Current stand densities range from 80 to 358 trees per acre. In Utah, Nevada, and Oregon, trees 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 suggests 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 are the result of both infill in mixed-age tree communities and expansion into shrub-steppe communities. However, the proportion of old growth can vary depending on disturbance regimes, soils, and climate. Some ecological sites can support persistant woodlands, likely due to specific soils and climate resulting 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 stand 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 tree canopy cover, causing a decrease in understory plants like sagebrush (Bates et al., 2000; Miller et al., 2000; Johnsen, 1962; Azuma et al., 2005). 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 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; Tausch, 1999). Additionally, as the understory vegetation declines in vigor, the ability of native perennial plans 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 in conjunction with intensive wildfire the potential for conversion to annual exotics is a serious threat (Tausch, 1999; Miller et al., 2008).

Specific successional pathways after disturbance in juniper stands depend on several variables such as plant species present at the time of disturbance and their individual responses 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 Utah and 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, can kill Utah juniper (Eddleman et al., 1994; Durham, 2014). 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. Witche' 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.
5. Long-horned beetles (Methia juniperi, Styloxus bicolor), round-head borers (Callidium spp.): Girdle branches and can kill branches or entire trees (Tueller & Clark, 1975).

Understory Dynamics:

The ecological sites in this group have understories 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 ranges from 1.0 to over 3.0 meters (Dobrowolski et al., 1990). Root length of mature sagebrush plants measures to a depth of 2 meters in alluvial soils in Utah (Richards & Caldwell, 1987). However, community types with low sagebrush (*Artemisia arbuscula*) as the dominant shrub are found to 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 annual precipitation is received during the winter and early spring. This continental semiarid climate regime favors 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 the duration and severity of drought 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).

Mountain big sagebrush (*Artemisia tridentata* ssp. vaseyana) is generally long-lived; therefore, it is not necessary for new individuals to recruit every year for perpetuation of the stand. Infrequent, large recruitment events and simultaneous low, continuous recruitment is the foundation of population maintenance (Noy-Meir, 1973). Survival of the seedlings depends on adequate moisture conditions.

Native insect outbreaks are also important drivers of ecosystem dynamics in sagebrush communities. Climate is generally believed to influence the timing of insect outbreaks, especially outbreaks of a sagebrush defoliator known as the Aroga moth (Aroga websteri). Aroga moth infestations occurred in the Great Basin in the 1960s, the early 1970s, and have been ongoing in Nevada since 2004 (Bentz et al., 2008). Thousands of acres of big sagebrush have been impacted, with partial to complete die-off observed. Aroga moth can partially or entirely kill individual plants or entire stands of big sagebrush (Furniss & Barr, 1975).

The ecological sites in this group have low to moderate resilience to disturbance and resistance to invasion. Resilience increases with higher 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 with frequent fire (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 publish lands dominated by cheatgrass and suggested that another 76 million acres were susceptible to invasion by winter annuals including cheatgrass and medusahead. By 2003, meduahead occupied approximately 2.3 million acrea in 17 western states (Rice, 2005). In the Intermountain West, the exponential increase in dominance by medusahead has largest been at the expese of cheatgrass (Harris, 1967; Hironaka et al., 1983). Medusahead matures two to three 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 determining 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 and 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 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 (Butler et al. 2011). 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, 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 are rare on these sites. Lightning-ignited fires are common but typically do not affect more than a few individual trees. Stand-replacing fires are uncommon to rare (100 to 600 years) and occur primarily during extreme fire behavior conditions. With low production of the understory vegetation and low density of trees per acre, high-severity fires within this plant community are not likely and rarely became crown fires (Bradley et al., 1992; Miller & Tausch, 2001).

Spreading, low-intensity surface fires have a very limited role in molding stand structure and dynamics. Surface spread was more likely to occur in denser woodlands on more productive sites (Romme et al., 2009). Pre-settlement fire return intervals in Great Basin National Park, Nevada averaged between 50 to 100 years, with north-facing slopes burning every 15 to 20 years and rocky landscapes with a sparse understory burning very infrequently (Gruell, 1999). Limited data exist that describe fire histories across woodlands in the Great Basin. The infilling of younger trees into old-growth stands and the expansion of trees into the surrounding sagebrush steppe communities has increased the risk of losing pre-settlement trees due to increased fire severity and size as a result of the increase in the abundance and landscape level continuity of fuels (Miller et al., 2008).

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 can survive low-severity fires because they have foliage farther from the ground and thicker bark, but mortality occurs when 60 percent or more of the crown is scorched (Bradley et al., 1992).

Western juniper is generally intolerant of fire and historically survived in areas with minimal understory vegetation, due primarily to soil characteristics (Vasek & Thorne, 1977; West, 1984; Miller & Rose, 1995). Therefore, the sites may not have carried fire, and when fire did occur it was low-intensity. With the increased suppression of wildfire and introduction of livestock grazing, which reduces ground fuels and understory competition, regeneration and establishment of western juniper has expanded into sites previously dominated by big sagebrush (Burns & Honkala, 1990). The expansion of western juniper is well documented. In the area surrounding Steens Mountain in southeastern Oregon, the expansion of western juniper coincides with Euro-American settlement. Probable causes of expansion include climate, altered fire frequencies, and grazing of flammable ground fuels (Miller & Rose, 1995). Fire resistance depends on age of the tree: seedlings, saplings, and poles are highly vulnerable to fire. Mature trees have some resistance to fire due to lack of fuels near the trunk, relatively thick bark, and foliage which is high above the ground (Burns & Honkala, 1990).

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 does occur when 60 percent or more of the crown is scorched. With the low production of the understory vegetation, high-severity fires within this plant community were not likely and rarely became crown fires (Bradley et al., 1992; Miller & Tausch, 2001). Tree density on the sites of this group increases with grazing management that favors the removal of fine fuels and management focused on fire suppression. With an increase of cheatgrass in the understory, fire severity is likely to increase. Western and Utah juniper reestablish by seed from nearby seed sources or surviving seeds. Western and Utah juniper begin to produce seed at about 30 years old (Bradley et al., 1992). Seeds establish best with a nurse plant such as sagebrush and rabbitbrush (Everett & Ward, 1984; Tausch & West, 1988; Bradley et al., 1992). Juniper woodlands reach maturity between 85 to 150 years after fire (Barney & Frischknecht, 1974; Tausch & West, 1988).

Mountain big sagebrush is killed by fire (Neuenschwander, 1980; Blaisdell et al., 1982), and does not resprout (Blaisdell, 1953). Post-fire regeneration is from seed and will vary depending on site characteristics, seed sources, and fire characteristics. Mountain big sagebrush seedlings can grow rapidly and may reach reproductive maturity within 3 to 5 years (Bunting et al., 1987). Mountain big sagebrush may return to pre-burn density and cover within 15 to 20 years following fire, but establishment after severe fires may proceed more slowly (Bunting et al., 1987).

Antelope bitterbrush is moderately fire tolerant (McConnell & Smith, 1977). It regenerates from seed and resprouting (Blaisdell & Mueggler, 1956; McArthur et al., 1982). However, sprouting ability is highly variable and is attributed to genetics, plant age, phenology, soil moisture, soil texture, and fire severity (Blaisdell & Mueggler, 1956; Blaisdell et al.,1982; Clark et al., 1982; Cook et al., 1994). Bitterbrush sprouts from a region on the stem approximately 1.5 inches above and below the soil surface. The plant rarely sprouts if the root crown is killed by fire (Blaisdell & Mueggler, 1956). Low-intensity fires may allow bitterbrush to sprout; however, community response

also depends on soil moisture levels at time of fire (Murray, 1983). Lower soil moisture allows more charring of the stem below ground level (Blaisdell & Mueggler, 1956), so sprouting will usually be more successful after a spring fire than after a fire in summer or fall (Murray, 1983; Busse et al., 2000; Kerns et al., 2006). If cheatgrass is present, bitterbrush seedling success is much lower. The factor that most limits establishment of bitterbrush seedlings is competition for water resources with the invasive species cheatgrass (Clements & Young, 2002).

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 the site along with 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 above ground 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).

Fire will remove 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 is considered to experience slight damage from fire but is more susceptible to fire damage in drought years (Young, 1983).

Conversely, Thurber's needlegrass (*Achnatherum thurberianum*) is very susceptible to fire-caused mortality. Burning can decrease the vegetative and reproductive vigor of Thurber's needlegrass (Uresk et al., 1976). Fire can cause high mortality in addition to reducing 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). However, Thurber's needlegrass often survives fire and will continue growth when conditions are favorable (Koniak, 1985). Thus, the initial condition of the bunchgrasses on the site and seasonality and intensity of the fire all factor into the individual species response.

Bluegrasses are a minor component in this group of ecological sites and have been found to increase following fire likely due to its low stature and productivity and may impair reestablishment of more deeply rooted bunchgrasses (Daubenmire, 1975).

The grasses likely to invade the sites in 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, areas dominated by cheatgrass are estimated to have a fire return interval of 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 annual species can take advantage of high nitrogen availability following fire because of their higher growth rates and increased seedling establishment relative to native perennial grasses (Monaco et al., 2003).

Livestock/Wildlife Grazing Interpretations:

Despite low palatability, mountain big sagebrush is eaten in small amounts by sheep, cattle, goats, and horses. Chemical analysis indicates that the leaves of big sagebrush equal alfalfa meal in protein, have a higher carbohydrate content, and yield twelvefold more fat (USDA, 1988).

Antelope bitterbrush, although a small component on these sites, 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.

Bluebunch wheatgrass is moderately tolerant of grazing 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.

Thurber's needlegrass is an important forage source for livestock and wildlife in the arid regions of the West (Ganskopp, 1988). Although the seeds are apparently not injurious, grazing animals avoid them when they begin to mature. Sheep, however, have been observed grazing the leaves closely, leaving stems untouched (Eckert & Spencer, 1987). Heavy grazing during the growing season has been shown to reduce the basal area of Thurber's needlegrass (Eckert & Spencer, 1987). This suggests that both seasonality and utilization are important factors in 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).

Inappropriate grazing practices can be tied to the success of medusahead. However, 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 cause changes to the fire regime. Medusahead replaces native vegetation and cheatgrass directly by competition and suppression; it replaces native vegetation indirectly by increasing 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

- F023XY024NV–JUOC/ARTRV/PSSPS
- F023XY036NV–JUOS/ARTRV/PSSPS

## Correlated Map Unit Components

21501295, 21501254, 21589751, 21589525, 21604425, 21604426, 21604429, 21501437, 21501443
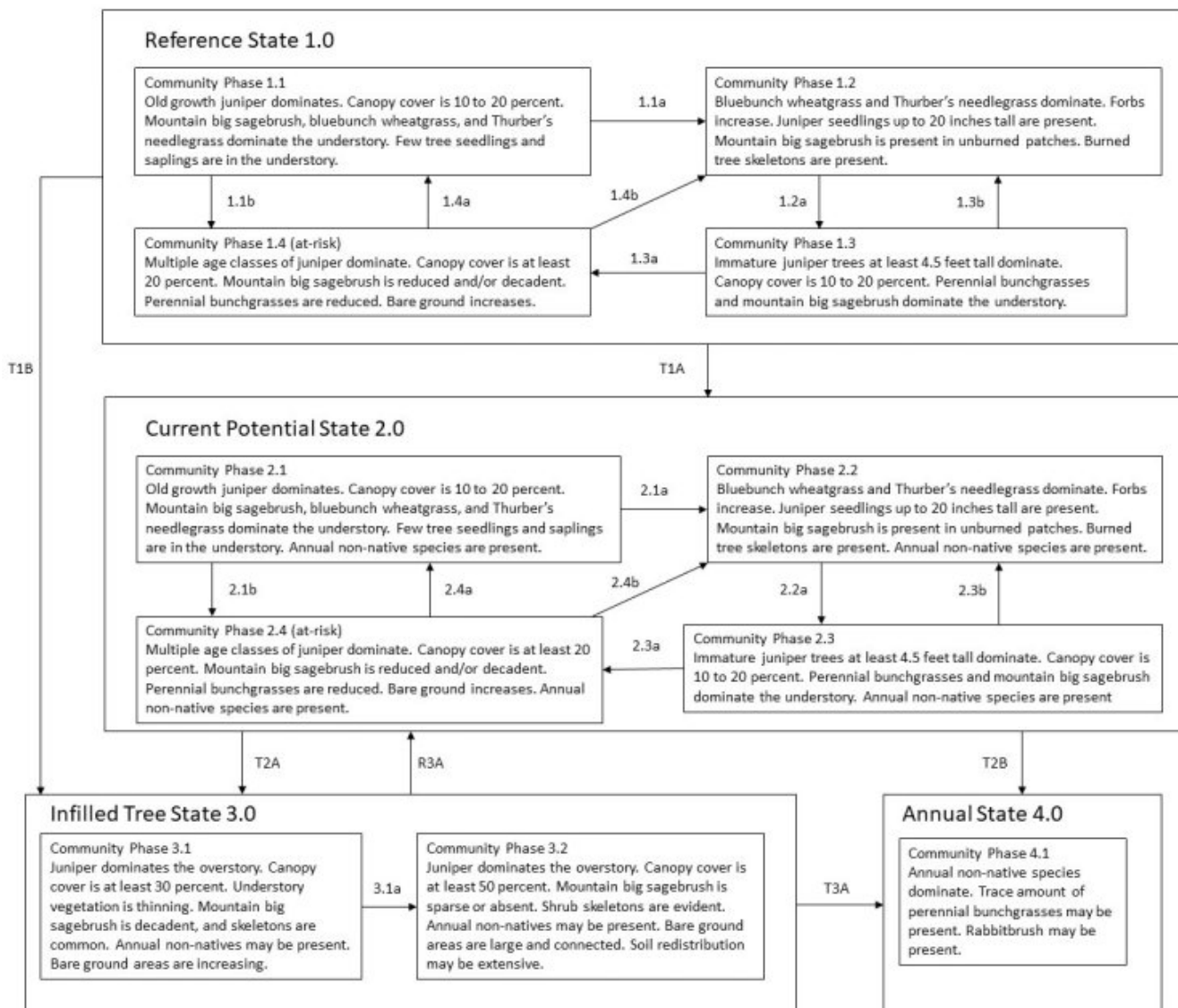
## Stage

Provisional

## Contributors

T Stringham (UNR under contract with BLM)
DMP

## State and transition model

## Reference State 1.0

**Community Phase 1.1**
Old growth juniper dominates. Canopy cover is 10 to 20 percent. Mountain big sagebrush, bluebunch wheatgrass, and Thurber's needlegrass dominate the understory. Few tree seedlings and saplings are in the understory.

1.1a →

**Community Phase 1.2**
Bluebunch wheatgrass and Thurber's needlegrass dominate. Forbs increase. Juniper seedlings up to 20 inches tall are present. Mountain big sagebrush is present in unburned patches. Burned tree skeletons are present.

1.1b · 1.4a · 1.4b · 1.2a · 1.3b

**Community Phase 1.4 (at-risk)**
Multiple age classes of juniper dominate. Canopy cover is at least 20 percent. Mountain big sagebrush is reduced and/or decadent. Perennial bunchgrasses are reduced. Bare ground increases.

1.3a →

**Community Phase 1.3**
Immature juniper trees at least 4.5 feet tall dominate. Canopy cover is 10 to 20 percent. Perennial bunchgrasses and mountain big sagebrush dominate the understory.

T1B · T1A

## Current Potential State 2.0

**Community Phase 2.1**
Old growth juniper dominates. Canopy cover is 10 to 20 percent. Mountain big sagebrush, bluebunch wheatgrass, and Thurber's needlegrass dominate the understory. Few tree seedlings and saplings are in the understory. Annual non-native species are present.

2.1a →

**Community Phase 2.2**
Bluebunch wheatgrass and Thurber's needlegrass dominate. Forbs increase. Juniper seedlings up to 20 inches tall are present. Mountain big sagebrush is present in unburned patches. Burned tree skeletons are present. Annual non-native species are present.

2.1b · 2.4a · 2.4b · 2.2a · 2.3b

**Community Phase 2.4 (at-risk)**
Multiple age classes of juniper dominate. Canopy cover is at least 20 percent. Mountain big sagebrush is reduced and/or decadent. Perennial bunchgrasses are reduced. Bare ground increases. Annual non-native species are present.

2.3a →

**Community Phase 2.3**
Immature juniper trees at least 4.5 feet tall dominate. Canopy cover is 10 to 20 percent. Perennial bunchgrasses and mountain big sagebrush dominate the understory. Annual non-native species are present

T2A · R3A · T2B

## Infilled Tree State 3.0

**Community Phase 3.1**
Juniper dominates the overstory. Canopy cover is at least 30 percent. Understory vegetation is thinning. Mountain big sagebrush is decadent, and skeletons are common. Annual non-natives may be present. Bare ground areas are increasing.

3.1a →

**Community Phase 3.2**
Juniper dominates the overstory. Canopy cover is at least 50 percent. Mountain big sagebrush is sparse or absent. Shrub skeletons are evident. Annual non-natives may be present. Bare ground areas are large and connected. Soil redistribution may be extensive.

T3A →

## Annual State 4.0

**Community Phase 4.1**
Annual non-native species dominate. Trace amount of perennial bunchgrasses may be present. Rabbitbrush may be present.

## 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 reduce the perennial grass understory.

**1.3a**: Time and lack of disturbance allow trees to mature. Excessive herbivory may also reduce the perennial grass understory.

**1.3b**: Fire reduces or eliminates tree cover.

**1.4a**: Low-severity fire, insect infestation, or disease remove individual trees and reduce 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 lack of disturbance allow trees to dominate site resources. This may be coupled with inappropriate grazing management that favors shrub and tree dominance.

## 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 reduce the perennial grass understory.

**2.3a**: Time and lack of disturbance allow trees to mature. Excessive herbivory may also reduce the perennial grass understory.

**2.3b**: Fire reduces or eliminates tree cover.

**2.4a**: Low-severity fire, insect infestation, or disease remove individual trees and reduce total tree cover.

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

**Transition T2A**: Time and lack of disturbance allow trees to dominate site resources This may be coupled with inappropriate grazing management that favors shrub and tree dominance.

**Transition T2B**: This transition is caused by catastrophic fire.

## 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**: This transition is caused by catastrophic fire.

**Restoration Pathway R3A**: Restoration may be accomplished via thinning of trees coupled with seeding. Successful restoration is unlikely for sites in Community Phase 3.2.

## Annual State 4.0 Community Phase Pathways

None.

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