Ashley National Forest Assessment

Terrestrial Ecosystems, System Drivers, and Stressors Report

Prepared by:
Allen Huber, ecologist
Colette Webb, silviculturist
Chris Plunkett, soil, water, air program manager
Brad Gillespie, fuels management specialist (WO Enterprise)
Chris Gamble, fuels specialist
Joe Flores, fire management officer
Dustin Bambrough, ecosystems staff officer

for:
Ashley National Forest

September 2017

We make every effort to create documents that are accessible to individuals of all abilities; however, limitations with our word processing programs may prevent some parts of this document from being readable by computer-assisted reading devices. If you need assistance with this document, please contact the Ashley National Forest at 435-781-5118.

Contents

Introduction 1
Spatial Scale 1
Ecosystem Characteristics 7
Rare and Unique Habitat Types 7
Calcareous or Rich Fens 8
Peatlands or Fens (Glacial Canyons) 10
Peatlands or Fens (Limestone) 11
Composition and Distribution of Vegetation 13
Alpine 13
Coniferous Forest 21
Aspen 31
Sagebrush 43
Pinyon and Juniper Woodlands 65
Desert Shrub 69
Structural Stages of Vegetation 78
Alpine 78
Aspen 82
Coniferous Forest 89
Pinyon Juniper Woodlands 111
Desert Shrub 113
Landscape Disturbances 119
Geomorphic 119
Insects and Disease 121
Fire 139
Literature Cited 154
Appendices 175
Appendix A. Description of Lodgepole Pine Structure 175
Appendix B. Description of Douglas-fir Structure 176
Appendix C. Description of Mixed Conifer Structure 176

Tables

Table 1. National Hierarchical Framework of Ecological Units (domain, division, province, and section) on the Ashley National Forest 2
Table 2. National Hierarchical Framework of Ecological Units, subsection level, on the Ashley National Forest 3
Table 3. Landtype associations on the Ashley National Forest 3
Table 4. Vegetation types on the Ashley National Forest 5
Table 5. Vegetation types evaluated in this assessment 5
Table 6. Vegetation types and acres in the alpine communities on the Ashley National Forest 13
Table 7. Alpine plant communities in the Alpine Moraine and Uinta Bollie landtype associations 13
Table 8. Coniferous forest and associated seral aspen and coniferous. Woodland forest and persistent aspen are not displayed. 21
Table 9. Comparison of the mixed conifer and Engelmann spruce subalpine habitat types on the Ashley National Forest 27
Table 10. Vegetation type and acres in aspen communities on the Ashley National Forest 31
Table 11. Landtype associations that support aspen communities on the Ashley National Forest 31

Table 12. Vegetation type and acres in sagebrush communities on the Ashley National Forest . 44
Table 13. Landtype associations that support mountain big sagebrush (*Artemisia tridentata* var. *vaseyana*) on the Ashley National Forest 44
Table 14. Landtype associations that support Wyoming big sagebrush (*Artemisia tridentata* var. *wyomingensis*) on the Ashley National Forest 45
Table 15. Landtype associations that support black sagebrush (*Artemisia nova*) on the Ashley National Forest 46
Table 16. Vegetation type and acres for desert shrub communities on the Ashley National Forest 69
Table 17. Landtype associations with known natural geomorphic disturbances........ 120
Table 18. Landtype associations with manmade geomorphic disturbances 121
Table 19. Native defoliators and host species on the Ashley National Forest 123
Table 20. Native bark beetles and host species on the Ashley National Forest 124
Table 21. Native bark beetles and host species on the Ashley National Forest 124
Table 22. Stem and branch diseases and host species on the Ashley National Forest 124
Table 23. Root diseases and host species on the Ashley National Forest 124
Table 24. Leaf and needle diseases and host species on the Ashley National Forest........ 124
Table 25. Declines and complexes and host species on the Ashley National Forest 124
Table 26. Fire regime group descriptions from the LANDFIRE classification system 141
Table 27. Fire regime group (FRG) within forested and nonforested vegetation types by percent of area 141
Table 28. Vegetation condition class descriptions from the LANDFIRE classification system . 142
Table 29. Vegetation condition class (VCC) in forested and nonforested vegetation types by percent of area 142
Table 30. Utah fire group descriptions 143
Table 31. Energy release component (ERC) data comparison 1970 to 2014 from the Yellowstone remote automated weather station (kcfast and Fire Family Plus v4.1)........ 149
Table 32. Acres and percent of the non-burnable fire behavior fuel model across the Ashley National Forest 151
Table 33. Acres and percent of the grass fire behavior fuel model across the Ashley National Forest 151
Table 34. Acres and percent of the grass/shrub fire behavior fuel model across the Ashley National Forest 151
Table 35. Acres and percent of the shrub fire behavior fuel model across the Ashley National Forest 151
Table 36. Acres and percent of the timber understory fire behavior fuel model across the Ashley National Forest 151
Table 37. Acres and percent of the timber litter fire behavior fuel model across the Ashley National Forest 151

Figures

Figure 1. Sections classified by the National Hierarchical Framework of Ecological Units on the Ashley National Forest 2
Figure 2. Map of landtype associations on the Ashley National Forest 4
Figure 3. Map of vegetation types on the Ashley National Forest 6
Figure 4. Map coniferous forest vegetation communities on the Ashley National Forest 22
Figure 5. Coniferous forest distribution on the Ashley National Forest........ 21
Figure 6. Landscape view of lodgepole pine belt on the Greendale Plateau landtype association (2011). Past clearcut harvest and fire disturbance are outlined in yellow (regenerated live

trees). Recent tree mortality (red needles) due to mountain pine beetle is visible between areas of past disturbance.. 94
Figure 7. Evidence of mixed severity fire pre-settlement near the Avintaquin Canyon landtype association. Younger stands, presumably a result of past fire, survived the recent Douglas-fir beetle outbreak... 97
Figure 8. Extensive Douglas-fir tree mortality caused by Douglas-fir beetle 98
Figure 9. Trend in Douglas-fir beetle-caused tree mortality, by year, from 1997 to 2015.......... 126
Figure 10. Douglas-fir beetle-caused tree mortality (shown in red) on the Ashley National Forest from 1997 to 2015. Green areas are the Douglas-fir vegetation type................................. 127
Figure 11. Trend in mountain pine beetle-caused lodgepole pine mortality from 1997 to 2015 129
Figure 12. Acres of mountain pine beetle-caused tree mortality in lodgepole pine on the Ashley National Forest from 1997 to 2015 130
Figure 13. Trend in mountain pine beetle-caused ponderosa pine mortality by year since 1997 (no mortality was detected in years 1997 and 1998) 131
Figure 14. Acres of mountain pine beetle-caused tree mortality in ponderosa pine on the Ashley National Forest from 1997 to 2015 132
Figure 15. Trend in piñon Ips beetle-caused piñon pine mortality by year since 1997 (no mortality was detected in years 1997 through 2001).. 133
Figure 16. Acres of piñon Ips beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015.. 134
Figure 17. Trend in aspen dieback, defoliation, and decline by year from 2007 to 2015. No damage was detected in years 1997 through 2006. 135
Figure 18. Acres of aspen dieback, defoliation, and decline on the Ashley National Forest from 1997 to 2015.. 136
Figure 19. Trend in acres of spruce forest affected by spruce beetle from 1997 to 2015 137
Figure 20. Acres of spruce beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015 (South Unit is not the same scale) 138
Figure 21. Changes to natural disturbance regimes resulting from fire exclusion in the Northern and Southern Rockies 140
Figure 22. Average monthly temperature from May to October, 1970-2014 147
Figure 23. Average monthly precipitation from May to October, 1970-2014 147
Figure 24. Average monthly wind direction and speed from May to October, 1970-2014......... 148
Figure 25. Palmer drought severity index from 1901 to 2016 and large fire growth for Utah Climate Division 5, Northern Mountains 150
Figure 26. Ashley National Forest percent of fire by each month from 1970 to 2014................ 152
Figure 27. Ashley National Forest percent of fires by cause class... 153
Figure 28. Ashley National Forest percent of fires by size class from 1970 to 2014.................. 153
Figure 29. Ashley National Forest number of fires and total acres burned from 1970 to 2014 .. 153

Introduction

This assessment evaluates terrestrial ecosystem characteristics and system drivers and stressors on the Ashley National Forest, including current conditions and trends. Terrestrial ecosystems were evaluated by assessing key ecosystem characteristics that sustain the long-term integrity of these ecosystems. Key ecosystem characteristics were evaluated based on the influence of stressors and drivers and the estimated degree of departure from natural range of variation. The current status and trends of key ecosystem characteristics were also considered in relation to the continuation of current management as well as climate associated risks. The terrestrial ecosystems used in this assessment were selected vegetation types and landtype associations that were mapped based on the National Hierarchical Framework of Ecological Units (Cleland and others 1997).

Spatial Scale

This assessment looked at ecosystem characteristics primarily at the vegetation type spatial scale for most key ecosystem characteristics. Landtype associations were also used as a spatial scale in this assessment for other ecosystem characteristics. Further, landtype associations were used to describe the geomorphic influences these landscapes exert on vegetation communities on the Ashley National Forest. The landtype associations were developed from the National Hierarchical Framework of Ecological Units. This type of integrated ecological evaluation provides information regarding the capabilities and limitations of landscapes that is needed for developing plan components at the forest level.

The National Hierarchical Framework of Ecological Units is a systematic land classification and mapping method developed to provide a scientific basis for implementing ecosystem management (Cleland and others 1997). This framework is described as, "a regionalization, classification, and mapping system for stratifying the Earth into progressively smaller areas of increasingly uniform ecological potentials. Ecological types are classified, and ecological units are mapped, based on associations of those biotic and environmental factors that directly affect or indirectly express energy, moisture, and nutrient gradients that regulate the structure and function of ecosystems. These factors include climate, physiography, water, soils, air, hydrology, and potential natural communities."

The Ashley National Forest is made up of diverse ecosystems spanning three physiographic divisions, four sections, and fifteen subsections that are defined by the framework. The four sections are the Uinta Mountains, Green River Basin, Tavaputs Plateau, and a very small portion of the Uinta Basin (see figure 1, table 1, table 2).

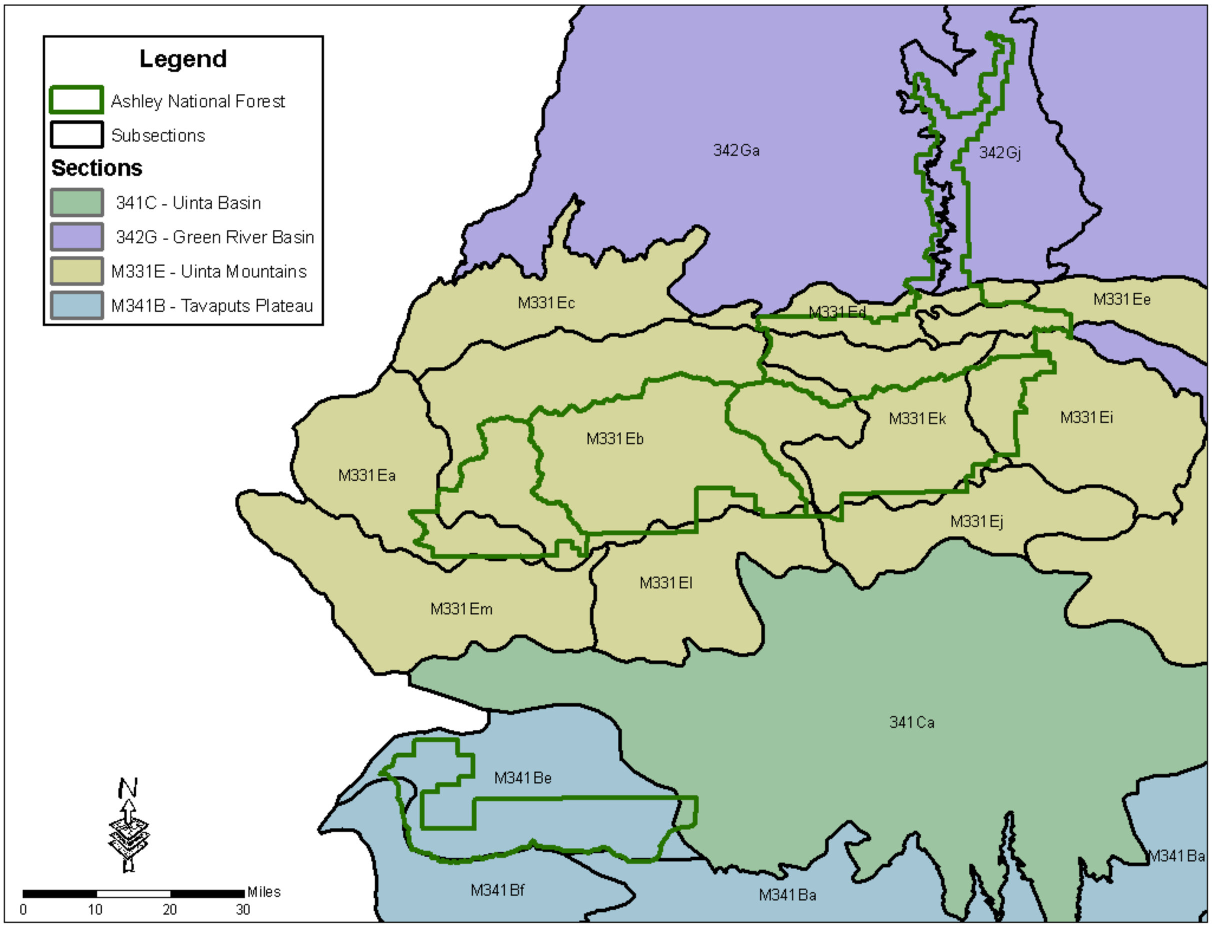


Figure 1. Sections classified by the National Hierarchical Framework of Ecological Units on the Ashley National Forest

Table 1. National Hierarchical Framework of Ecological Units (domain, division, province, and section) on the Ashley National Forest

Ecological Unit[1]	Name
Domain 300	Dry
Division 340	Temperate Desert
Division M330	Temperate Steppe
Division M340	Temperate Desert Regime Mountains
Province 342	Intermountain Semi-Desert
Province M331	Southern Rocky Mountain Steppe
Province M341	Nevada/Utah Mountains
Section 341C	Uinta Basin
Section 342G	Green River Basin
Section M331E	Uinta Mountain
Section M341B	Tavaputs Plateau

[1] Domain, division, province, and section data taken from McNab and others (2007).

Table 2. National Hierarchical Framework of Ecological Units, subsection level, on the Ashley National Forest

Section	Subsection[2,3]	Name
Uinta Basin	Subsection 341Ca	No name
Green River	Subsection 342Ga	No name
Green River	Subsection 342Gj	No name
Uinta Mountain	Subsection M331Ea	West Flank Uintas
Uinta Mountain	Subsection M331Eb	Western High Uintas
Uinta Mountain	Subsection M331Ed	Phil Pico Highlands
Uinta Mountain	Subsection M331Ee	Dutch John Highlands
Uinta Mountain	Subsection M331Ei	Diamond Mountain Highlands
Uinta Mountain	Subsection M331Ej	Upper Ashley Canyons
Uinta Mountain	Subsection M331Ek	Trout Peak Highlands
Uinta Mountain	Subsection M331El	Johnny Star Benches
Uinta Mountain	Subsection M331Em	Wolf Creek Highlands
Tavaputs Plateau	Subsection M341Ba	No name
Tavaputs Plateau	Subsection M341Be	No name
Tavaputs Plateau	Subsection M341Bf	No name

[2] Subsection data taken from Uinta Mountain Ecosystem Management Project, Ashley, Wasatch-Cache, and Uinta National Forests. April 1994

[3] Not all subsections have been named

Within each subsection, the Ashley National Forest mapped landscapes at the landtype association scale. There are 24 distinct landtype associations on the Ashley National Forest (table 3, figure 2). Landtype associations were selected as one spatial scale to describe key ecosystem characteristics on the Ashley National Forest.

Table 3. Landtype associations on the Ashley National Forest

Landtype Association	Total Acres	Percent of Total Acres
Alpine Moraine	25,9135	18.5%
Antelope Flats	7,425	0.5%
Anthro Plateau	10,9261	7.8%
Avintaquin Canyon	82,456	5.9%
Dry Moraine	9,903	0.7%
Glacial Bottom	16,353	1.2%
Glacial Canyons	71,910	5.1%
Green River	70,755	5.0%
Greendale Plateau	53,221	3.8%
Limestone Hills	18,875	1.4%
Limestone Plateau	7,391	0.5%
Moenkopi Hills	2,139	0.2%
North Flank	50,493	3.6%
Parks Plateau	97,442	7.0%
Red Canyon	28,638	2.1%
Round Park	10,465	0.8%

Landtype Association	Total Acres	Percent of Total Acres
South Face	49,144	3.5%
Strawberry Highlands	12,388	0.9%
Stream Canyon	44,046	3.2%
Stream Pediment	8,129	0.6%
Structural Grain	24,359	1.7%
Trout Slope	14,3471	10.2%
Uinta Bollie	174,608	12.5%
Wolf Plateau	6,021	0.4%
(blank)	42,264	3.0%
Grand Total	1,400,292	100%

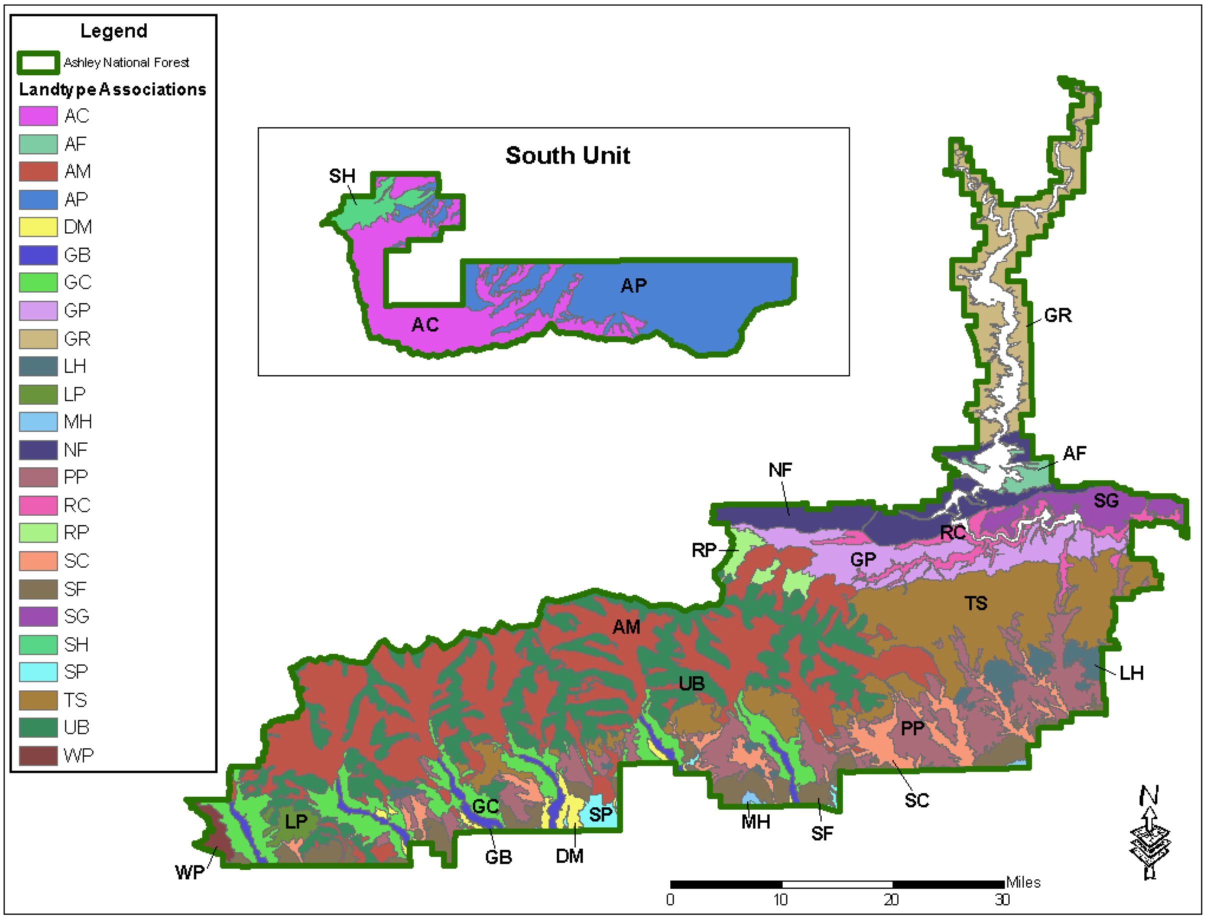


Figure 2. Map of landtype associations on the Ashley National Forest

Vegetation types were selected as another spatial scale to describe ecosystem characteristics on the Ashley National Forest. The Ashley National Forest is composed of the vegetation types described in table 4 and shown in figure 3.

Table 4. Vegetation types on the Ashley National Forest

Vegetation Type	Acres	Percent of Total Acres
Alpine	168,849	12.06%
Coniferous Forest	623,442	44.52%
Deciduous Forest	35,513	2.54%
Seral Deciduous Forest	117,179	8.37%
Mountain brush	43,776	3.13%
Shrubland	125,075	8.93%
Riparian	37,688	2.69%
Grassland	14,660	1.05%
Forb	80	0.01%
Grass/forb	16	0.00%
Woodland	122,444	8.74%
Desert shrub	66,762	4.77%
Water	44,809	3.20%
Total	1,400,293	100.00%

Selected vegetation types were evaluated using landtype associations to distinguish certain characteristics and distinctions of the same vegetation type that span various landscapes on the Ashley National Forest. The vegetation types in the table below were evaluated in this assessment. The vegetation types were selected based on their percentage of representation on the Ashley National Forest (5 percent or more), the ecosystem services they provide, and potential risk to sustainability.

Table 5. Vegetation types evaluated in this assessment

Vegetation Type	Acres	Percent of Total Acres
Alpine	168,849	12.1%
Coniferous Forest	623,442	44.5%
Aspen (Seral)	117,179	8.4%
Aspen (Persistent)	35,513	2.5%
Sagebrush	125,075	8.7%
Pinyon and Juniper Woodlands	122,444	8.7%
Desert Shrub	67,976	4.9%

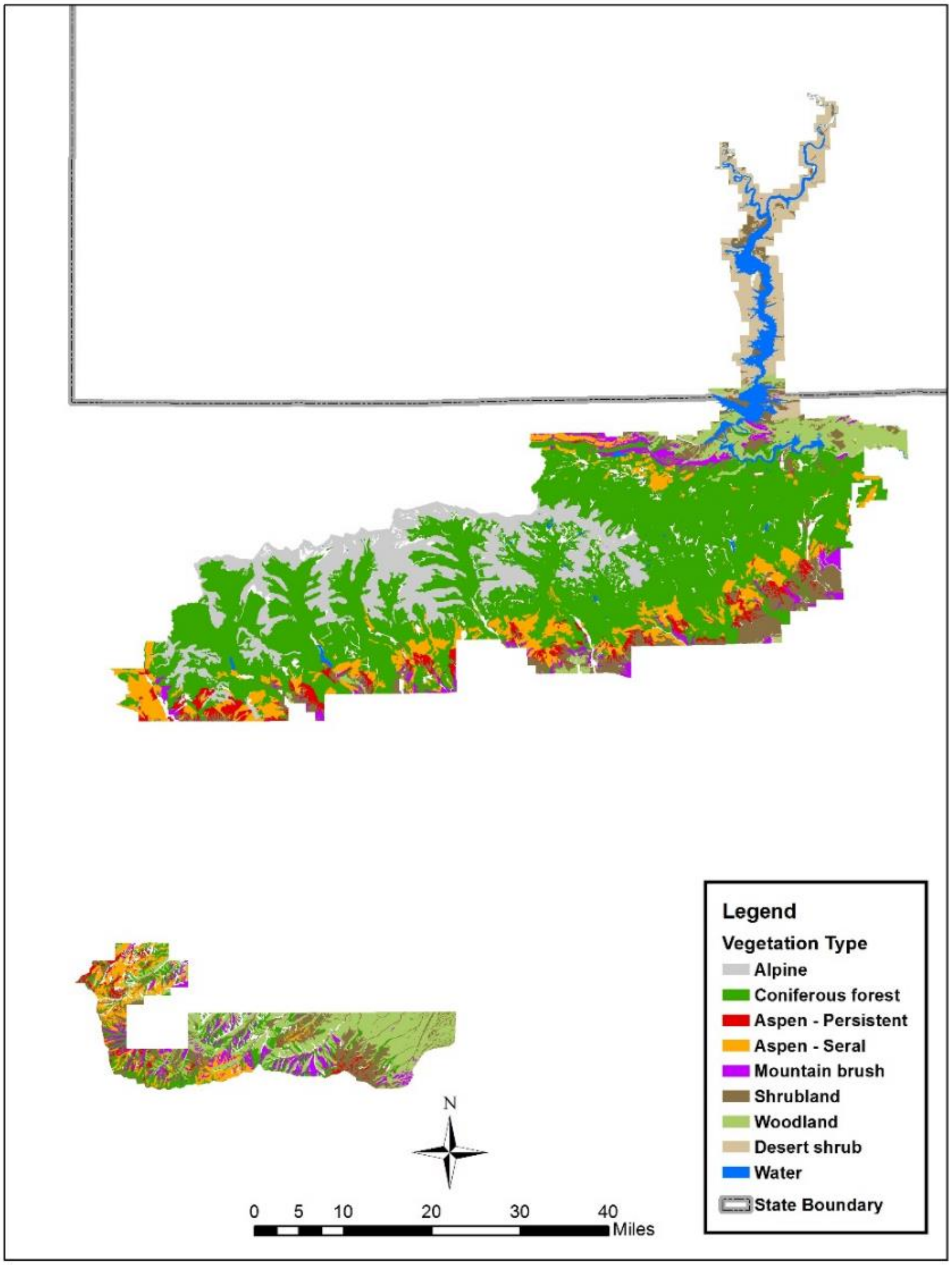


Figure 3. Map of vegetation types on the Ashley National Forest

Ecosystem Characteristics

Ecosystem characteristics were selected based on their influence in sustaining long-term sustainability of ecosystems and their ability to be stressors and drivers of ecosystems. Key ecosystem characteristics that were determined to sustain the long-term integrity of ecosystems on the Ashley National Forest include:

- rare and unique habitat types;
- composition and distribution of vegetation;
- Structure stages of vegetation; and
- landscape disturbances (geomorphic, insects and disease, fire).

These key ecosystem characteristics were evaluated in relation to their natural range of variation and the influence of drivers and stressors on these characteristics. Natural range of variation is defined as, “The variation of ecological characteristics and processes over scales of time and space that are appropriate for a given management application” (Forest Service Handbook 1909.12). The timeframe for natural range of variation is generally considered pre-European influence and should be, “sufficiently long, often several centuries, to include the full range of variation produced by dominant natural disturbance regimes such as fire and flooding and should also include short-term variation and cycles in climate” (Forest Service Handbook 1909.12). The natural range of variation is a tool for assessing the ecological integrity and does not necessarily constitute a management target or desired condition.

Current conditions and trends for key ecosystem characteristics were also described relative to current drivers and stressors. The description of trend for ecosystem characteristics also took into account the continuation of current management activities and uses, as well as climate-associated risks.

Rare and Unique Habitat Types

The Ashley National Forest consists of numerous plant communities, many of which are considered rare because of their limited distribution and infrequent occurrence on the landscape. Some rare plant communities also have qualities that are extraordinary that further distinguish them from others. Rare plant communities are likely more susceptible to stressors, whether natural or anthropogenic, that may threaten their integrity or existence. The susceptibility is caused by the plant community’s limited distribution and occurrence on the landscape. Recognizing and managing for unique habitats conserves landscape diversity, contributes to the integrity of broader ecological systems, and promotes sustainability of existing forest resources.

A filter process was necessary to determine spatial, quantitative, and intrinsic characteristics that distinguish rare and unique habitats from others. Rare and unique habitats are often associated with their size (less than 100 acres) or geographic isolation. Also, occurrence or frequency of habitats on the landscape indicates how rare or common the community is. Habitats with five occurrences or less are considered rare. Habitat that supports flora or fauna that are federally listed, considered for listing, or have narrow endemics add value to the habitat. Intrinsic values of habitats are those characteristics that make the habitat unique or extraordinary. Such characteristics are usually qualitative in description and subjectively assessed.

The following criteria were developed to determine rare and unique habitats on the Ashley National Forest:

1. Is the habitat restricted geographically, both within and outside the plan area?
2. How often does the habitat occur within and outside the plan area?
3. Does the habitat support one or more species of flora or fauna that are listed as threatened, endangered, proposed, candidate, species of conservation concern (SCC), and/or narrow endemic?
4. Are there intrinsic values of the habitat that distinguish it from other habitat?

Habitats that appeared to be rare, infrequent, and/or consist of unique characteristics were identified within and separated by landtype association. Eighteen habitats were initially identified. These were evaluated using the criteria listed above (Huber 2016). In order to qualify as a rare or unique habitat, at least three of the four criteria must apply. Of the 18 habitats evaluated, three qualified as rare and unique habitats and are described below.

Calcareous or Rich Fens

Calcareous or rich fens (Chadde and others 1998) are rare in the Uinta Mountains, in and outside the plan area. A fen is a type of wetland, usually fed by mineral-rich surface water or groundwater. One calcareous or extremely rich fen (pH values exceeding 8.0) is located in the South Fork Rock Creek drainage, within the Alpine Moraine landtype association. This fen is located at approximately 9,300 feet elevation, is about 10 acres in size, located on a gentle to moderate gradient. This fen is watered by an aquifer at the base of a limestone scree slope and mountain. Its features meet the description of a slope fen. The fen is relatively open with stunted Engelmann spruce trees scattered within it, with mature spruce forests established along the perimeter. The fen consists of a patchwork of community types and most of these are rare to the Uinta Mountains. The fen supports about 80 plant taxa, including several species unique to cold and wet conditions. Plant species of note include handsome pussytoes (*Antennaria pulcherrima),* alpine meadowrue *(Thalictrum alpinum)*, subulate sedge *(Carex microglochin)*, wetland kobresia *(Kobresia simpliciuscula),* northern twayblade *(Listera borealis)*, and heartleaf twayblade (*Listera cordata*).

Three clubmosses previously unknown to Utah are also located here. The clubmosses are: bog groove moss (*Aulacomnium palustre),* tall clustered thread moss *(Bryum pseudotriquetrum),* and floating hook moss *(Warnstorfia fluitans*). There are no known threatened, endangered, or sensitive species, but two potential plant species of conservation concern are found in the fen. These plants are handsome pussytoes and wetland kobresia.

Photo of South Fork Rock Creek fen

Influences of Drivers and Stressors

The primary driver of this fen is the extensive karst system of the Blind Stream Plateau. Water runoff drains into sink holes and depression and discharges at the base of the mountain. There are no diversions, dams, or other anthropogenic disturbances that would affect water flow.

Some timber harvesting immediate to the fen occurred historically, but "have not greatly altered the landscape" (Nature Conservancy 1995). No additional timber harvesting has occurred nor been proposed since the early 1990s. Timber harvesting removals within and immediately adjacent to the fen would be a potential stressor. Fire is also a potential stressor, but fire intervals for Engelmann spruce span hundreds of years. A National Forest System road is located down slope and adjacent to the fen, which makes access to the site relatively easy. No notable human impacts have been observed within the fen. Increased recreation use of the area is a foreseeable stressor. Livestock grazing is a stressor and has occurred in the drainage for many decades, with most grazing occurring along the ecotone and in the drier parts of the fen. Some trampling of peat is documented along the margins of the fen, but the interior is minimally impacted by cattle due to water-saturated conditions. Another stressor is the fen's exposure to avalanche disturbance. Parts of the fen are located beneath avalanche chutes from the adjacent slope. The latest event that deposited trees and other debris occurred in winter 2005 (Study 35-15A). Climate-related risks that lead to drying conditions could be a potential stressor. The Intermountain Region's climate vulnerability assessment indicates that this mid-to-high elevation fen has both a moderate-to-high sensitivity and vulnerability ratings and a low-to-moderate adaptive capacity rating regarding climate related risks (USDA Forest Service 2016b). Those species that rely on cold and wet conditions would be at risk with a warmer and drier climate. Plant community composition would be affected by increased water stress, opening niches for more drought-tolerant species.

Status, Trends, and Natural Range of Variation

The fen is a mosaic of plant communities whose vegetation composition are comparative to extremely rich fens found in Idaho and Montana (Chadde and others 1998). Species richness is relatively high (80 plant species). Based on the evaluation protocol used by Chadde and others (1998), the fen would likely receive a moderate to high conservation ranking. Although timber harvest, livestock grazing, and avalanche disturbance are known stressors, long-term monitoring indicates that the fen is in satisfactory condition with stable trends for at least 20 years (Study 35-15). During this time, plant species composition has remained constant, and repeat photography indicates no change in community structure and size of the area. Based on these findings, the South Fork Rock Creek fen is considered to be trending towards its natural range of variation.

Peatlands or Fens (Glacial Canyons)

Peatlands or fens, relatively small in size (i.e., a few acres), are found in glacial canyons (Glacial Bottom landtype association) of the Uinta Mountains at elevations between 7,200 and 8,500 feet. These fens are located within or adjacent to forested areas and are fed by small springs or aquifers found near the base of canyon slopes. The fens are found on gentle to moderate gradients and best fit the description of slope fens. These fens are likely rich fens, but further analysis is necessary to determine their status. Documented sites include Whiterocks, Uinta, and Rock Creek Canyons, but fens can be expected in other glacial canyons in the plan area. This fen community includes some less common plants of the Uinta Mountains and Utah. These include Bebb sedge (*Carex bebbii*), brownish sedge (*Carex brunnescens*), hair-like sedge (*Carex capillaris*), bristlestalked sedge (*Carex leptalea*), green sedge (*Carex oederi*), and chamisso sedge (*Carex pachystachya)*. At a Whiterocks Canyon fen, a high number of sedge species occur within a few acres (less than 10 acres) of habitat, which is a notable feature of the community. There are no known threatened, endangered, or sensitive species from this habitat type, but one potential plant species of conservation concern, bristlestalked sedge, grows here.

Influences of Drivers and Stressors

The primary driver of these fens are springs or seeps located at the toe of canyon slopes. There are no known diversions, dams, or other anthropogenic disturbances that would alter groundwater flow to the fens.

Photo of Whiterocks Canyon fen

For decades, livestock grazing has occurred in most glacially carved drainages of the south slope Uinta Mountains and is a stressor or a potential stressor of these fens. Water-saturated conditions have limited livestock intrusion into or use of the fens and along their margins. Grazing has not occurred for over 25 years in known fens in Uinta Canyon or where grazing could potentially occur in Lake Fork Canyon. Timber harvesting and wildfire are potential stressors, but no evidence of past timber harvesting or recent fire are found in or near known and documented fens. A road runs through the fen located in Whiterocks Canyon and likely has impacted the hydrology and possibly plant species composition of the fen below the road. The other documented fens are remote and are not impacted by roads or trails. No notable human impacts have been observed within the fens; however, increased recreation use of the area is a foreseeable stressor. Climate-related risks that lead to warmer and drier conditions could be a potential stressor. The Intermountain Region's climate vulnerability assessment indicates that this mid-elevation fen has moderate-to-high sensitivity and vulnerability ratings, and a moderate adaptive capacity rating regarding climate related risks (USDA Forest Service 2016b). Mid-elevation fen species may have the ability to move upslope to adaptable habitat as the climate becomes warmer. These fens may be more susceptible to the impacts of stressors due to their small size.

Status, Trends, and Natural Range of Variation

Vegetation composition of its communities reflect those described for intermediate to rich fens in Idaho and Montana (Chadde and others 1998). Based on the site evaluation used by Chadde and others (1998), the fens would likely receive moderate to high conservation rankings. Long-term monitoring indicates that the habitat is in satisfactory condition, with stable trends (Study 42-43G1-G2). No change in plant species composition or structure has been detected over a 20 year period. These fens are considered to be within their NRV, except the fen located within Whiterocks Canyon. Due to a road that crosses the fen, it is considered to be slightly departed from its natural range of variation.

Peatlands or Fens (Limestone)

A few peatlands or fens with limestone influence are found in Sheep Creek and Hickerson Parks of the Greendale Plateau landtype association. These are likely rich fens, but further analysis is necessary to determine their status. The approximate area influenced by fens in Sheep Creek and Hickerson Parks is 85 and 30 acres respectively. The fens are found within depressions, have relatively flat surfaces, and are fed by underground springs that create hydrostatic cones. They fit the description of a basin fen. Hummocks and boggy areas also occur and provide niches for some plant species. No other fens meeting this description are found within or outside the plan area in the Uinta Mountains. The fens include seven less common plants of the Uinta Mountains and Utah, although widespread in other parts of North America. These plants include wetland kobresia (*Kobresia simpliciiuscula*), marsh felwort (*Lomatogonium rotatum*), silvery primrose (*Primula incana*), far northern buttercup (*Ranunculus hyperboreus*), yellow marsh saxifrage (*Saxifraga hirculus*), thick-leaf starwort (*Stellaria crassifolia*), and alpine meadowrue (*Thalictrum alpinum).*

There are no threatened, endangered species or sensitive species associated with the fens, but two potential plant species of conservation concern are found (wetland kobresia and silvery primrose).

Photo of Sheep Creek Park fen

Influences of Drivers and Stressors

The primary driver of these fens are springs that surface periodically throughout the fen. There are no known diversions, dams, or other anthropogenic disturbances that would alter groundwater flow to the fens.

Livestock grazing is a stressor and has occurred in the area for many decades, but ungulate access into the interior of the fens is minimal due to saturated soils. Some trampling and compaction of peat by elk and cattle has occurred along the margins and in the drier areas of the fens. The fens are located near major Ashley National Forest Service roads and their perimeters are easily accessible by off-road recreation vehicles. Long-term studies depict these impacts (Studies 2-2, 3-40). Little to no evidence of vehicle disturbance has been observed within the fens. Increased recreation activities near the fens is a foreseeable stressor. Climate related risks that lead to a warmer and drier climate could be a potential stressor. The Intermountain Region's climate vulnerability assessment indicates that this mid-to-high elevation fen has both a moderate-to-high sensitivity and vulnerability ratings and a moderate-to-high adaptive capacity rating regarding climate related risks (USDA Forest Service 2016b). Mid-elevation fen species may have the ability to move upslope to adaptable habitat as the climate becomes warmer. These fens may be less susceptible to the impacts of stressors due to their relatively large size.

Status, Trends, and Natural Range of Variation

Vegetation composition of its communities reflect those described for rich fens in Idaho and Montana (Chadde and others 1998). Based on the site evaluation used by Chadde and others (1998), the fens would likely receive a high conservation ranking. Long-term monitoring indicates that the habitat is in satisfactory condition, with stable trends (Studies 3-40, 2-2). No change in plant species composition or structure has been detected over several decades. These fens are considered to be within their natural range of variation.

Composition and Distribution of Vegetation

Alpine

Non-forest or alpine plant communities of high elevation are mostly found in the Alpine Moraine and Uinta Bollie landtype associations, and to a small extent in the Trout Slope landtype association.

Table 6. Vegetation types and acres in the alpine communities on the Ashley National Forest

Vegetation type	Acres
Talus, fell, and boulder fields	123,225
Alpine meadow and turf	28,575
Low willow and other shrub	16,489
Snow bed	560
Total	168,849

Table 7. Alpine plant communities in the Alpine Moraine and Uinta Bollie landtype associations

Landtype Association	Comparative Features
Alpine Moraine	Communities occur mostly in areas of glaciation with comparatively young soils. Kobresia, curly sedge (*Carex rupestris*), dryas (*Dryas octopetala*), and cushion plant communities are comparatively uncommon. Plane leaf willow (*Salix planifolia*), glaucous willow (*Salix glauca*), water sedge (*Carex aquatilis*) communities are comparatively common.
Uinta Bollie	Communities occur mostly above areas of glaciation with comparatively old soils. Kobresia, curly sedge, dryas, and cushion plant communities common. Plane leaf willow, glaucous willow and water sedge communities are comparatively uncommon.

Description of the Natural Range of Variation

There is minimal information regarding the natural range of variation for alpine plant communities, but some inferences can be made based on literature, long-term studies, historical photos, and historical accounts. Additionally, vegetation and environmental condition comparisons were made with scientific studies conducted at Niwot Ridge, a UNESCO Biosphere Reserve located in the Rocky Mountains of Colorado. This reserve includes an alpine setting that has had little influence by human impact, including the exclusion of livestock grazing. The reserve is an appropriate reference to help describe the natural range of variation for alpine plant communities.

Alpine vegetation is a complex of communities at high elevation that consist of an array of plants adapted to harsh environmental conditions. Plants are typically low-growing, mat-forming, small or dwarfed in their structure, or some combination of these things. These communities are “uniquely adapted to their environment . . . major adaptations include reduction in growth height, perennial life cycle, herbaceous habit, and below-ground biomass, high photosynthetic efficiency, [and] resistance to drought . . . Alpine vascular plants grow slowly, an adaptation to low temperatures, high desiccation rates, and sudden microclimatic changes” (Romme and others 2009).

In alpine communities, there is typically a diversity of plant communities across the landscape and their presence or absence is conditional on topography, geology, aspect, snow accumulation and persistence, wind exposure, rodent activity, soil moisture, temperature, and other factors that form habitable niches (Baker 1983; Billings 1973; Bryant and Scheinberg 1970; Cox 1933; Douglas and Bliss 1977; Goodrich 2004, Goodrich 2006, Johnson and Billings 1962; Lewis 1970; Marr 1961; Stanton and others 1994;

Walker and others 1993; Willard 1979, Brown 2006, Romme 2009). Such variation creates a mosaic of plant communities that are usually small, differ considerably in species composition and potential ground cover, and where rapid changes in plant composition occurs over short distances (Romme and others 2009). Large, uniform plant communities are rare due to a diversity of microclimates (Osburn 1958). Most communities consist of a dominant species or have co-dominant species that describe the community. Species richness varies by community, from a few plants to 20 or 30 species, with most of these occurring in trace amounts (Brown 2006). Major plant communities of Uinta Mountains, and the environmental conditions that determine their presence, are similar to those described in other Rocky Mountain alpine regions. These regions include Niwot Ridge and Rocky Mountain National Park, Colorado.

Using Cluster Analysis, Brown (2006) described 50 alpine plant communities from 15 major vegetation groups within the Uinta Mountains. A few of the more common or well-distributed alpine communities of the Uinta Mountains will be discussed in relation to natural range of variation below.

Several "wet communities" are present in the Uinta Mountains, but water sedge is the most common. These communities are typically found in wet to saturated areas, often associated with peat-filled lakes or glacial depressions, poor fens, or on floating peatlands adjacent to open water (Lewis 1970, Chadde and others 1998). Since water sedge is a strong community dominant, species richness is relatively low (i.e., about five species). At better drained sites, plane-leaf willow often serves as a co-dominant. Ground cover is at or near 100 percent for these communities (Brown 2006).

Photo of study 12-15E. 2013. Gilbert Creek Basin, Uinta drainage.
View of plane-leaf willow and water sedge community

Snow bed communities have high variation related to several features and these are most influenced by the snow release date. Vegetation composition is a strong indicator of snow duration. Twelve snow bed communities are listed for the Uinta Mountains (Brown 2006). Plants such as tufted hairgrass (*Deschampsia caespitosa*), alpine bluegrass (*Poa alpina*), Parry's rush (*Juncus parryi*), and alpine sagewort (*Artemisia scopulorum*) are found in snow beds with early snow release dates. In late persisting snow beds, Pyrenean sedge (*Carex pyrenaica*), cloud sedge (*Carex haydeniana*), alpine buttercup (*Ranunculus adoneus*), mountain sorrel (*Oxyria digyna*), and Engelmann sedge (*Carex engelmannii*) are quite common. Species richness in snow bed communities range from 10 (e.g., mountain sorrel) to 22 (e.g., Engelmann sedge) plants per community. Ground cover varies from an average of 98 percent in black alpine sedge (*Carex nigricans*), to as low as 62 percent in alpine buttercup communities. Bare soil is more common in snow beds than other alpine communities, and rock represents a large portion of ground cover in many communities (Brown 2006). Pocket gophers are common in a few snow bed communities, which increases bare soil.

Photo of study 35-33F. 2005. Blind Stream Plateau. Alpine snowbed community with alpine buttercup

Low and dwarf shrub communities are represented in dry alpine settings. Of these, gray-leaf willow is most common and grows in well-drained soils. Communities of this willow can be found on steep rocky or talus slopes, to benches or basins of gentle gradient. Willow crown cover can reach as high as 35 percent. Herbaceous cover consists of an array of sedges, grasses and forbs, with timber oatgrass (*Danthonia intermedia*), alpine avens (*Geum rossii*), and single-spike sedge (*Carex scirpoidea*) often present. Species richness averages 23 plants per site and ground cover averages 92 percent (Brown 2006). Vegetation composition of most moist meadows can be variable, but tufted hairgrass is a major component in nearly all of them. Soils are relatively deep, well-watered, but drained (Lewis 1970). These communities are generally of high production and ground cover (i.e., more than 90 percent). Species richness ranges from 15 to 19 plants in moist meadows (Brown 2006). Plane-leaf willow is co-dominant in the wetter communities, while timber oatgrass, alpine sagewort, and single-spike sedge are more common in mesic meadows. Alpine avens is one of the most common plants in the Uinta Mountains. This plant is considered a generalist and can be found in many alpine communities. The plant does form distinct communities in dry alpine turf, stone-nets and boulder fields, and talus outwashes. Alpine avens grows in common with curly sedge, single-spike sedge, and tufted hairgrass. Species richness ranges from 15 to 17 species and ground cover averages about 90 percent (Brown 2006).

Photo of study 14-36C. 2015. Above Reader Lakes, Whiterocks drainage. Alpine avens community

Evidence indicates alpine areas in the Uinta Mountains have fluctuated in size, and the timberline perimeter has changed over time. Comparing photographs taken during the 1870 Hayden geologic survey and replicated in 2001, Munroe (2003) found tree lines shifted upslope from 60 to 180 meters and timberline forests were denser at five different sites since 1870. He reasoned that, following the end of the Little Ice Age around 1850, tree lines began to shift upslope with a warming climate. Dated samples of subfossil wood above modern tree line indicated that tree line on Bald Mountain was approximately 60 meters higher prior to the Little Ice Age (Munroe 2003). He concluded, "a higher treeline in the northern Uintas shortly before A.D. 1550 is consistent with contemporaneous evidence for warmer-than-modern climates in the southwestern United States."

Current Status and Trends

Alpine plant community composition and distribution are currently represented on the landscape consistent with existing geomorphic and climatic features identified above. This is verified by many long-term studies that describe, classify, and determine conditions and trends of alpine communities found in the Uinta Mountains (Brown 2006, Studies 10-10B, 11-14H, 16-1E, 16-20B2, 16-20DROC, 16-20G1, 16-20J, 16-20L, 16-20M, 16-20N, 16-20NET, 23-20C1, 23-21, 28-45D, 28-45E, 37-7F, Goodrich and others 2005). Additionally, Uinta Mountain alpine communities appear similar to those described at other Rocky Mountain alpine sites (Cooper and others 1997, Komarkova 1980, Walker and others 1993, Romme and others 2009). Plant composition is a factor used to compare current conditions to NRV. Study data indicate that native plants totally dominate all alpine communities within the plan area. These communities show satisfactory plant composition and ground cover conditions with mostly stable trends, but some change in trend will be discussed below.

Fens, streams, and other riparian communities show long-term stability in community size and desired plant composition (Goodrich and others 2005, Goodrich and Huber 2009, Goodrich and Huber 2009b, Goodrich and Huber 2009c, Goodrich and Huber 2009d, Goodrich and Huber 2014). Long-term monitoring also indicates that dry meadow, turf, and mesic communities are stable with appropriate ground cover percentages. A couple of trends that are changing community dynamics include an increase in density and canopy cover of low willow in many alpine communities, both wet and dry, that has been documented for at least 50 years (Goodrich and others 2006). This increase of willow has occurred concurrent with livestock grazing. Another notable trend is the increase of and gradual displacement by conifers in low willow communities, mesic meadows, and a few riparian ecotones at or near timberline (Munroe 2003, Studies 11-2B, 11-2C1, 12-22, 23-3A, 23-4B)., Using repeat photography at five different sites, Munroe (2003)found tree lines had moved upslope from 60 to 180 meters since 1870. He also noted that timberline forests were denser in 2003 than depicted in 1870.

Ground cover is also a factor used to help compare current condition to the natural range of variation. Vegetation and litter account for most or nearly all ground cover in riparian, mesic, and most dry alpine communities. Potential ground cover in most communities is high, often greater than 90 percent, especially when pocket gopher activity is absent. Cushion plant and other windswept communities typically have higher rock cover than turf, dry or mesic alpine meadows. In plant communities where pocket gopher activity is present and inherent, bare ground increases considerably. A host of conditions, many interrelated, influence distribution and abundance of pocket gophers. These include plant community type, winter snow cover, ground water, soil drainage, soil texture, and perhaps geologic substrate. Deep persisting snow beds, talus, shale barrens, boulder, and fellfields consist of sparse vegetation, often with high, bare soil; where rock accounts for most ground cover; or both. High, bare soil in these communities is a function of snow cover, shifting surface material, and undeveloped parent material. In summary, alpine plant communities in the plan area are within expected ranges of potential ground cover.

Management Activities and Uses

Alpine plant communities in the Uinta Mountains are remote and relatively inaccessible because topographic barriers limit human uses and activities. Alpine areas are sought for wilderness quality, remoteness, and a variety of recreation opportunities such as hiking, backpacking, camping, sightseeing, fishing, hunting, and riding horses. Overall, recreation use is light except for a few areas near trailheads or other access points. Increased recreation activities are expected during the next plan period. Many alpine communities are capable and suitable for livestock grazing, a use that has occurred for more than 100 years. Domestic sheep grazing is most common in alpine areas, but sheep numbers have fallen considerably. The decrease in sheep grazing on the Ashley National Forest is mostly due to a decline of the sheep industry nationally. Future trends of sheep grazing are expected to either remain stable or possibly trend downward if the sheep industry continues to decline nationally.

Some cattle grazing occurs in alpine areas. Modest increases in areas grazed and cattle numbers have occurred over the last 30 years, mostly resulting from a few allotments converting from sheep to cattle grazing. Most alpine areas are not compatible with cattle grazing and additional increases of cattle use in alpine areas is not predicted. Grazing by cattle in alpine areas is expected to remain stable.

Alpine areas are important watersheds for local communities and the State of Utah. The areas provide for the consumptive needs of people of Utah and others who use water from the Green - Colorado River drainage. Alpine plant communities provide habitat and forage for many wildlife species, including: ptarmigans, pikas, elk, moose, deer, mountain goats, and bighorn sheep.

Influences of Drivers and Stressors

Long-term monitoring has detected increases in density and canopy cover of low willow in many alpine communities and conifer tree lines show upslope movement since 1870. These trends may be interrelated. Munroe (2003) argued that following the conclusion of the Little Ice Age (ca. 1850), climate warming trends in the Uinta Mountains led to increased timberline tree densities and upslope expansion of Engelmann spruce and subalpine fir. Similarly, warming climates may influence the upward trend of low willows in alpine settings. Additionally, new tree establishment at or near timberline consistently occurs in low willow communities where low willows have also shown increase. This seems to indicate that low willows create a favorable micro-environment for conifers (Goodrich and others 2015, Studies 24-15A-B, 11-2A-C1, 11-2E, 11-2E9, 11-2F, 11-2G2).

Lewis (1970) recognized that besides grazing impacts, “natural processes are continually at work breaking down the vegetal cover. Frost, running water, and wind are the more important ones.” These would be considered ecological drivers. Of the three mentioned, running water has the greatest impact on alpine systems. Scouring or headcutting in alpine turf and dry meadow communities is documented at numerous sites across alpine landscapes. Headcutting occurs during run-off events where the volume, concentration, and velocity of water is great enough to cause scouring in the vegetative turf. Headcutting is most common at the base of extensive talus and boulder watersheds. Repeat photography from several head cut sites suggests that runoff events that would cause or increase turf scouring are rare, perhaps 50- to 100-year events. No change in headcuts and surrounding vegetation was detectable from photos over 44 and 59-year periods (Goodrich and Huber 2014, Studies 11-3G1, 12-25, 12-26). Lewis (1970) noted that “head cutting does not appear to progress very readily because of the tough, tenacious sod.” In any event, this erosional process appears to be slow.

Photo of study 11-3G. 1949 and 2008. Head of Yellowstone drainage. Repeat photography depicted slow change in headcut.

Pocket gopher activity in the Uinta Mountains is an inherent disturbance. This activity is indicated to be the major biotic factor controlling plant community dynamics and ground cover in select alpine communities (Willard 1979, Thorn 1982, Stoecker 1976, Goodrich and Cameron 2011, Goodrich and others 2015b). These rodents are present where specific geomorphic settings and desirable plant communities occur. Favorable conditions include winter snow cover, well-drained soils, and desirable plant species. Many persisting snow beds are selected by pocket gophers. In contrast, pocket gophers are essentially absent in turf and meadow communities where ground cover is high and snow cover is marginal. Increased bare soil in the form of deflated eskers and mounds is a function of pocket gopher activity. Additionally, some plant communities, such as boreal sagewort (*Artemisia arctica*), are well developed and maintained where pocket gopher disturbance is abundant. Similar findings were described by Willard (1979) on Trail Ridge in Colorado in the absence of livestock grazing. Finally, there is no apparent correlation between livestock grazing and high levels of pocket gopher activity as some have asserted. Studies indicate high percent ground cover in mesic communities, adjacent to plant communities, with high bare soil occupied by pocket gophers with equal access by domestic sheep to both communities (Goodrich 2006b, Goodrich and Cameron 2011, Goodrich and others 2015b). Also, other studies show total ground cover in communities selected by pocket gophers to be similar in areas grazed by livestock and those not grazed by livestock (Goodrich 2006b). In summary, the distribution and abundance of pocket gophers can be related to plant community type, soil drainage, duration of snow cover, groundwater, soil texture, and perhaps geologic substrate.

Photo of study 35-20B. 2014. Blind Stream Plateau. Alpine snowbed community with pocket gopher activity.

Wild ungulates are capable of impacting some alpine communities, namely riparian, mesic, and dry meadow communities, especially those that have low willows. Elk, moose, and mountain goat populations have increased over the last 30 years. On the South Slope herd unit, elk numbers have increased from 7,470 animals in 2004 to 8,700 animals in 2010 (UDWR 2012). Moose populations show an upward trend in nearly all management units within or adjacent to the plan area (Utah Department of Wildlife Resources management plan). Mountain goats were first introduced to the Uinta Mountains in the 1980s. Goat populations increased fourfold (from 215 to 858 animals) from 2001 to 2011 but are currently at 49 percent of the State's desired management objective. Low willow communities appear to be most susceptible to ungulate browsers. Elk and moose have diminished shrub canopies in a few low willow subalpine communities, but no decreases in alpine communities have been documented (Studies 14-43B, 14-43C1-C4, Goodrich and others 2015). Steady upward trends in wild ungulate populations are considered a potential stressor of alpine communities.

Anthropogenic related impacts in alpine environments have been relatively limited, due to remoteness and harsh conditions associated with high elevations. The most common human disturbances are trails, dispersed camping, and recreation horse use - but these disturbances are minor in magnitude. Human impacts have not influenced vegetation compositions or altered ecological functions. However, increased recreation activities is a foreseeable stressor during the next plan period.

Livestock grazing has occurred in mostly mesic, turf, and dry meadow alpine communities over a 100 year period. However, domestic sheep grazing has fallen considerably in both numbers and the area grazed over the last few decades. Numbers of cattle use have increased in alpine communities, as sheep grazing has declined. Sheep grazing, however, is still the most common type of livestock grazing in alpine plant communities, with cattle grazing more common in the sub-alpine meadows.

Prior to a decline in domestic sheep grazing, Lewis (1970) found “measureable changes” in plant vigor and species composition occurred at one site, “after eleven years of protection from grazing by domestic livestock.” Recent vegetation reports from several high elevation grazing allotments indicate that, “the plant communities grazed by livestock are in satisfactory condition with stable trends or are trending toward desired condition” (Huber 2016). Long-term trend studies for a 50- to 60-year period show that current conditions are satisfactory and the trend is stable concurrent with livestock grazing. Notable disturbances attributed to current livestock grazing are occasional trailing, salt sites, and bedding grounds (Huber 2016). Numerous studies located in plant communities grazed by livestock show dominance of native species that have moderate to high resource value, with plant compositions remaining stable. There are no long-term studies that document changes of plant species composition or unsatisfactory ground cover conditions related to livestock grazing (Goodrich and Huber 2009, Goodrich and Huber 2009b, Goodrich and Huber 2009c, Goodrich and Huber 2005, Goodrich and Huber 2014). An evaluation comparing species richness of similar alpine communities where grazing occurs and does not occur found that “differences in the averages and ranges for species richness of the four plant communities analyzed appear within the range of natural variability for the communities” (Huber 2016b). These trends are concurrent with existing forest uses and management including livestock grazing.

Climate Related Risks and Trends

A warming climate over the last 150 years has influenced the shift of the tree line upslope and forests and the tree lines have become denser (Munroe (2003). Also, low willow communities have increased in canopy cover, density, and size over the last few decades, and changes in low willow communities are likely influenced by a warming climate. If a warming climate continues, gradual upward migration of subalpine communities are predicted to occur, but alpine communities appear topographically limited to upslope migration. If climates become cooler, gradual downward migration of plant communities are predicted to occur. These trends are considered currently within natural range of variation.

Climate change may be considered either an ecological stressor or driver. If the climate becomes warmer, subalpine plants may continue moving upslope to newly habitable environments. Alpine plants may be more susceptible to impacts of a climate warming trend due to topographical limitations in upslope migration. If snow accumulation decreases with a warmer and drier climate, the composition of vegetation in snow bed communities may change as snow melts sooner. Plant communities in traditionally late persisting snow beds may phase out with less snow accumulation and earlier snow release dates. On the other hand, if the climate begins to cool, tree lines and subalpine vegetation may experience downslope migration as discussed by Munroe (2003).

Comparison of Natural Range of Variation and Current Conditions

Current conditions of alpine plant communities in the plan area closely align with the natural range of variation. This is based on environmental conditions and drivers that determine plant species composition, ground cover, species richness, and disturbances that impact vegetation. Anthropogenic impacts are minimal in most alpine areas and have little effect on vegetation resources. Trails, dispersed camping and fire ring impacts are the most common identifiable human impacts, but are small in scope and have shown to be of little consequence in vegetation condition and trends. However, increased use of alpine areas during the next decade is likely, and impacts are expected to increase. Livestock grazing may impact the composition of vegetation, plant vigor, and ground cover of some communities; however, grazing is likely not sufficient to move plant communities outside the natural range of variation. Current grazing management is not expected to move plant communities outside the natural range of variation.

Coniferous Forest

Excluding woodland and persistent aspen forest, approximately 53 percent (748,285 acres) of the Ashley National Forest is coniferous or seral aspen to coniferous forest. Coniferous forest is broadly classified into five major types: ponderosa pine, persistent lodgepole pine, Douglas-fir, mixed conifer, and Engelmann spruce. A summary of acres distribution is listed in table 8. Minor types that occur on the Ashley National Forest are grouped as miscellaneous.

Table 8. Coniferous forest and associated seral aspen and coniferous. Woodland forest and persistent aspen are not displayed.

Community	Acres	Seral Aspen (Acres)	Total
Ponderosa pine	37,870	7,898	45,768
Lodgepole pine (persistent)	76,794	20,228	97,022
Douglas-fir	47,900	40,033	87,933
Mixed conifer	310,828	47,153	357,981
Engelmann spruce	144,500	N/A	144,500
Miscellaneous (subalpine fir, blue spruce, 5-needle pines, riparian forest*)	13,195	1,886	15,081
Total	631,087	117,198	748,285

* May include a mix of conifers or deciduous trees such as aspen, cottonwood, willows, maples, and boxelder.

The following section will address the major coniferous forest vegetation types where aspen may be lacking, approximately 45 percent (617,892 acres) of the Ashley National Forest. Seral aspen and persistent aspen are addressed in another section of this assessment. Although aspen is discussed comprehensively in another section, seral aspen and it's dynamic with the associated coniferous types may be mentioned in this section for context. Coniferous forest distribution is displayed in figure 5.

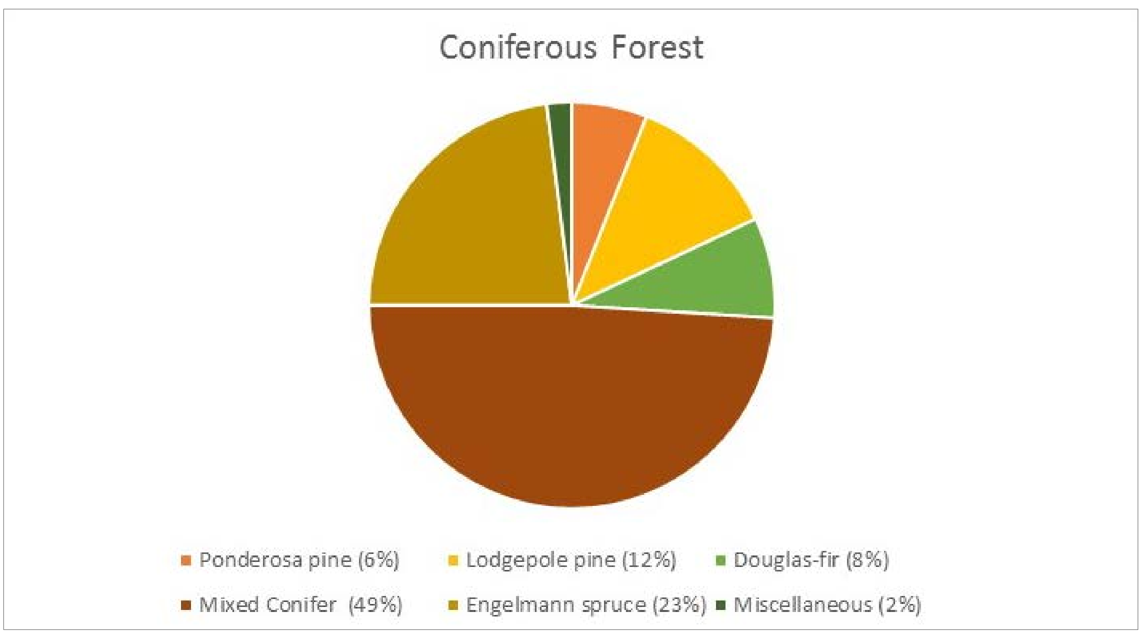


Figure 5. Coniferous forest distribution on the Ashley National Forest

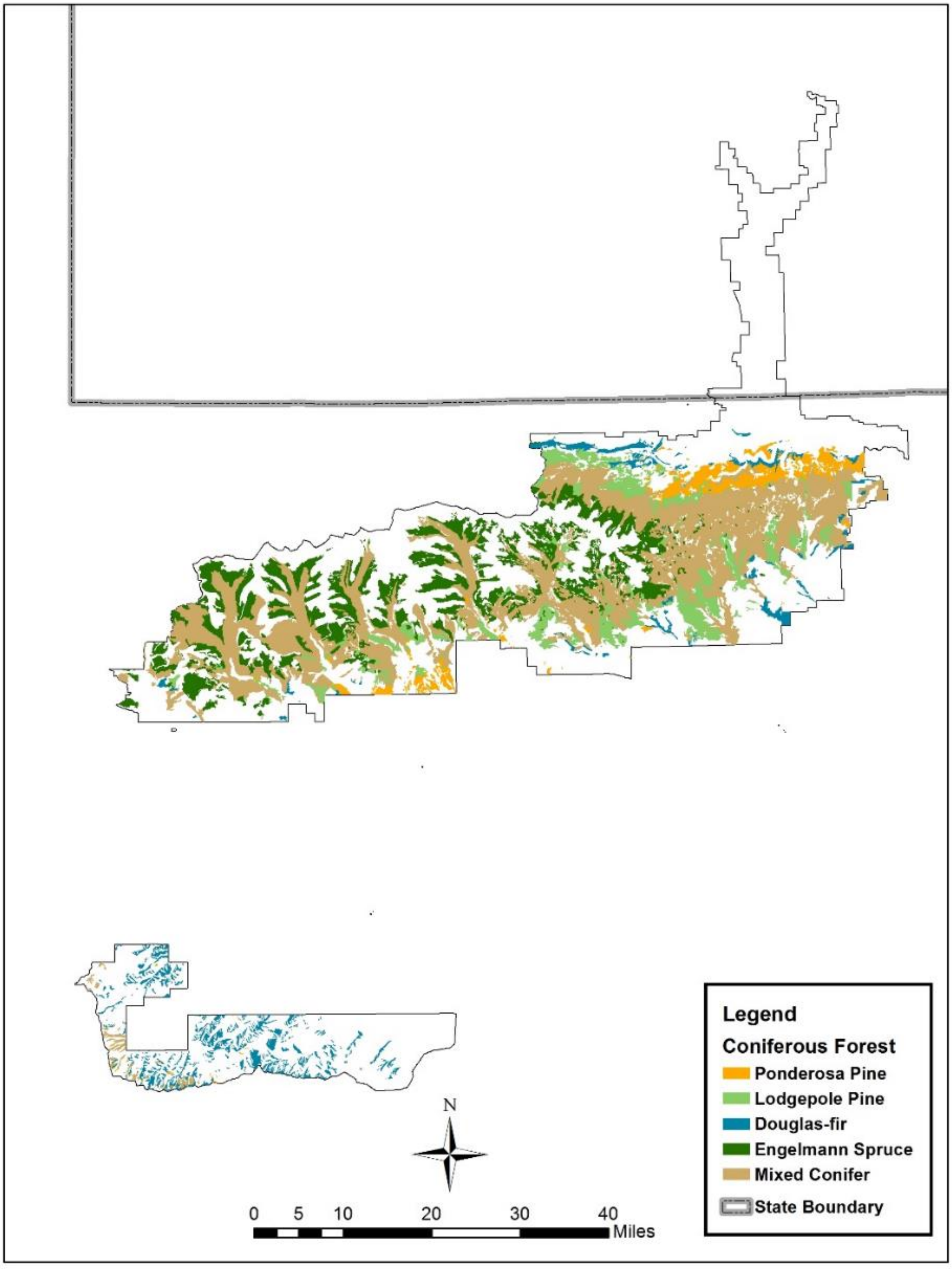


Figure 4. Map coniferous forest vegetation communities on the Ashley National Forest

Ponderosa Pine

Ponderosa pine (*Pinus ponderosa* var. *scopulorum*) occurs in three major landtype associations on the Ashley National Forest: Dry Moraine (2,362 acres), Greendale Plateau (22,658 acres), and Stream Pediment (2,729 acres). Ponderosa pine occurs to a lesser extent on the Red Canyon (2,637 acres) and Structural Grain (1,640 acres) landtype associations. Ponderosa pine on the Red Canyon landtype associations occurs mostly along the fringe where there is a common boundary with the Greendale Plateau.

Description of the Natural Range of Variation

Natural under-burning on a frequent fire return interval eliminates other less fire-tolerant conifers and maintains dominance of the ponderosa pine. The regional properly functioning condition indicator for the ponderosa pine interpreted “dominance” to be more than 75 percent of the tree cover is ponderosa pine (USDA Forest Service 1998, USDA Forest Service 2009). Under the historical nonlethal fire regime, the understories are fairly open with a good diversity of grass, forbs, and shrubs (Fiedler and others 1995). Stands with an aspen component have the potential to maintain both ponderosa pine and aspen. With fire, aspen can become a co-dominant with ponderosa pine. Without fire, aspen would eventually senesce (Bradley 1992).

Current Status and Trends

The Greendale Plateau supports the Ashley’s largest, mostly continuous ponderosa pine forest of the eastern Uinta Mountains. Ponderosa pine on the Greendale Plateau can be found in vegetation communities such as aspen, serviceberry, mountain big sagebrush-bitterbrush, or one-spike oatgrass. Approximately 80 percent of the ponderosa pine forest on this landtype association has received recent disturbances through treatments like underburning (i.e., prescribed fire), pre-commercial thinning, and through salvage harvest accompanied by the burning of activity fuels (i.e., pile burning). Recent harvests of ponderosa pine have been almost exclusively salvage harvest and have occurred in response to tree mortality from mountain pine beetle epidemics. Although salvage harvests reduce buildup of downed woody material, areas that have been underburned have done more to reduce or thin out regenerating conifers, especially conifers that are less fire-tolerant. Approximately 50 to 75 percent of the ponderosa pine on the Greendale Plateau has been treated in the past 20 years by prescribed under burning treatment.

Recent examination of permanent monitoring plots in the Red Canyon and Dowd Mountain areas of the Greendale Plateau show tree composition is well within the natural range of variation. Most of these two areas had been salvage harvested with slash burning and had only been partly treated by prescribed under burning or fuel reduction treatments in the past. The areas show greater than 75 percent ponderosa pine in the overstory conifers. The understory is composed of up to 10 percent of other less fire-tolerant species, such as Douglas-fir and Rocky Mountain juniper. Not all areas are currently set up for monitoring, however, and those areas that lack disturbances, such as areas where fire and timber harvest have not occurred, can be expected to have fire-intolerant species at greater occurrences and species that are more shade-tolerant than ponderosa.

Because fires have been infrequent in the ponderosa pine forest over the last 100 years, saplings of ponderosa pine have increased in density throughout the type, as well as Rocky Mountain juniper at the lower elevations. These species at this stage can act as ladder fuels that increase the potential fuel load of the stand. Such stands are prone to stand-replacement fire that is atypical of the fire regime in this type.

On the Dry Moraine landtype association, manzanita is a common shrub of considerable canopy cover that grows under ponderosa pine. At maturity, this shrub is typically less than 3.5 feet (1 meter) tall. However, it can develop a large canopy, and it is rather volatile under dry conditions. Permanent monitoring plots in the Rock Creek drainage of this landtype association show ponderosa pine is 100 percent of the conifer composition, even prior to prescribed underburning. Similarly, permanent plots in the Mud Springs area show 100 percent ponderosa pine dominance after underburning. Manzanita, however, was very abundant in both these areas, making burn operations difficult. An estimated 90 percent of the ponderosa on this landtype association has been underburned since the 1990s. More than 50 percent of the ponderosa has been salvage harvested, due to mountain pine beetle caused mortality in the 1980s.

On the Stream Pediment landtype association, the basal canopy height, density of trees, and light fuel loading of ponderosa pine stands indicate a relatively high resistance to crown fires and a more park-like appearance of widely spaced trees (Study 40-15E) with shrub understories. Sagebrush and bitterbrush are common species; however, in some areas, mountain brush species dominate the understory in ponderosa pine stands.

Climate Related Risks and Trends

Increase fire frequency and extent increases ponderosa pine’s chances of survival over their less fire tolerant competitors, thus ensuring ponderosa’s dominance in most forests (USDA Forest Service 2016b). An exception might be in areas where fire exclusion has increased stand density and fuel loads that are conducive to crown fires. Also, high severity fires will likely reduce ponderosa pine.

Comparison of Natural Range of Variation and Current Conditions

Tree composition characteristics for Ponderosa pine appear to be in the natural range of variation, at least in areas that have had some form of disturbance like harvest or underburning. However, up to an estimated 20 percent of the ponderosa pine forest has not had recent disturbance. The estimated departure from the natural range of variation based on this characteristic is low to moderate.

Lodgepole Pine

Persistent lodgepole pine (*Pinus contorta* var. *latifolia*) occurs in two major landtype associations: Greendale Plateau (12,372 acres) and Parks Plateau (37,289 acres) and to a lesser extent on the Round Park (3,000 acres). Stands of lodgepole pine also occur on the Trout Slope (8,686 acres), Alpine Moraine (4,700 acres), and Dry Moraine (1,811 acres) landtype associations. On the Trout Slope and Alpine Moraine landtype associations, lodgepole pine occurs more often at the lower to mid-elevations where it can be seral to spruce and subalpine fir. Engelmann spruce has a potential to be more competitive in absence of fire on these landtype associations. The occurrence of lodgepole pine here, and its relationship with Engelmann spruce in subalpine habitat type, may require further examination (see the “Mixed Conifer” and “Engelmann Spruce” sections). The following lodgepole pine discussion will focus on the more extensive and persistent lodgepole pine forests of the Greendale Plateau and the Parks Plateau.

Description of the Natural Range of Variation

Lodgepole pine is typically an early seral tree species with a range extending beyond the Intermountain Region. Persistent lodgepole pine forests often occur at lower elevations (below 9,600 feet), are generally heavily stocked, and comprise large pure stands often exceeding 200 acres (USDA Forest Service 2009). Lodgepole pine is the only conifer tree species present for long periods of time under natural disturbance regimes[1]. If present, other tree species are uncommon and discontinuous. After stand-replacing events such as fires, site conditions are often too dry for establishment of other conifer species. Lodgepole pine’s

[1] See discussion under the ecosystem characteristic, “Structural Stages of Vegetation for Lodgepole Pine:. This characteristic describes the disturbance regime between fire, mountain pine beetle, and lodgepole pine in greater detail.

serotinous cones also allow for dense regeneration and later exclusion of other species. Dense stands of lodgepole pine often prevent other trees from establishing before the next stand-removing event.

Current Status and Trends

Portions of the lodgepole pine type were harvested in the mid-1960s. Since then early harvests were primarily clearcuts. Machine piling of slash often followed these harvests. Harvested areas regenerated well to approximately the same species composition as the original stands.

Persistent lodgepole pine on the Greendale Plateau landtype association extends from Summit Springs to the west end of the landtype association and fits the description of Mauk and Henderson (1984) of a lodgepole pine-kinnikinnik habitat type (Study 2-17B). This is a relatively dry habitat type where subalpine fir and Engelmann spruce are absent or have a minor presence, and lodgepole pine is considered a climax species. Specific to the lodgepole pine stands of the Greendale Plateau landtype association, kinnikinnik appears to be less common in the early-seral stages following fire (Studies 2-17B, 3-39B), but several other herbaceous species are common. Stands of lodgepole pine have a high percentage of closed or serotinous cones, indicating a highly fire-adapted system. As on all landtype associations of this type, lodgepole pine seedlings also establish in great abundance after fire disturbance. Where there is aspen, these lodgepole pine stands would include aspen.

On the Parks Plateau landtype association, lodgepole pine has abundantly regenerated following fire, which is typical of lodgepole pine in other parts of its range. Cones of lodgepole pine also have a relatively high level of serotiny. Lodgepole pine seedlings establish in great abundance the year of, or the year after, disturbance (Studies 18-41, 44-19B). Ross sedge and a few other seedbank species, and species with windborne seeds, generally increase rapidly. Ross sedge is a common seedbank species in early post-burn and post-harvest sites.

On the Round Park landtype association, seral stages for lodgepole pine stands are similar to seral stages described for lodgepole pine forests on the Parks Plateau (appendix A). However, an exception is that, in some of the more open stands of lodgepole pine forests on this landtype association, russet buffaloberry forms a shrub layer. Lodgepole pine occurs mostly at lower elevations on this landtype association and adjacent to the Greendale Plateau landtype association.

Climate Related Risks and Trends

With a warmer climate, the number of fires will likely increase and fire return intervals will decrease. Years with no major fires, which were common historically, are expected to be rare. A more frequent fire regime could change fuel dynamics, making the establishment of Douglas-fir a more likely outcome.

Comparison of the Natural Range of Variation and Current Conditions

The system appears to be operating at low departure from the natural range of variation. This is indicated by the dominance of the lodgepole pine species in the overstory, the lack of other conifer tree species in the understory, and the response of the lodgepole pine type to regenerate lodgepole pine after a disturbance.

Douglas-fir

Douglas-fir (*Pseudotsuga menziesii* var. *glauca* Mirb. Franco) occurs on many landtype associations across the Ashley National Forest in areas having soils with limestone parent material. On the Uinta Mountain Section, it is most prevalent on North Flank (5,751 acres) and on the northerly aspects of the Stream Canyon (4,625 acres) and Red Canyon (4,649 acres) LTAs. Douglas-fir is most common, however, on the Tavaputs Plateau Section of the Ashley National Forest, on the Avintaquin Canyon

(16,959 acres) and Anthro Plateau (9,162 acres) LTAs. The Douglas-fir forest type is well distributed across the Ashley National Forest.

Description of the Natural Range of Variation

Fire histories for moist or cool Douglas-fir types in Utah are lacking (Bradley 1992). Nevertheless, observations are that “Douglas-fir in the Uinta Mountains have multiple fire scars and stands contain much scattered charcoal, indicating fire has occurred at relatively frequent intervals in the past” (Bradley 1992). Evidence suggests that Douglas-fir was historically influenced by mixed-severity fire regimes with mean fire intervals of 35 to 200 years, with a more frequent return interval on dry sites and less frequent on mesic sites (USDA 2009, Landfire 2012). Low-severity fires that thinned the understory were favorable for Douglas-fir. Douglas-fir stands were historically thinned by low-severity fires, where the older, fire-resistant trees would survive and provide seed and partial shade for the establishment of seedlings (Bradley 1992). The bark of these “older” Douglas-fir takes approximately 40 years to become fire resistant (Bradley 1992).

Current Status and Trends

The presence of low- to mixed-severity fire has been largely absent in Douglas-fir forests since settlement and is likely outside the natural range of variation. Fire suppression led to Douglas-fir forests that had become increasingly dense, more mature, and susceptible to bark beetles. Consequently, the Douglas-fir beetle has affected much of the Douglas-fir forest on the Ashley National Forest. Beetle epidemics “serve a role similar to a stand replacing fire, leading to widespread tree mortality, and promote forest succession” (Giunta 2016). Areas of Douglas-fir across the Ashley National Forest have the potential to shift to subalpine fir.

On the Avintaquin Canyon landtype association, open stands of Douglas-fir and abundance of aspen indicate that mean fire return intervals had occurred at a sufficient frequency prior to European settlement to prevent dominance by subalpine fir and sustain aspen communities (Study 64-42D). Areas dominated by Douglas-fir reveal, however, the potential to shift to subalpine fir. Stand exams on the Reservation Ridge of this landtype association, for example, show that although the overstory could be near 100 percent Douglas-fir, the understory is near 90 percent subalpine fir.

Similarly on the western end of the North Flank landtype association, stand exams show a composition of nearly 90 percent subalpine fir in the understory of dominant Douglas-fir stands. As one moves east, however, the composition of the understory becomes predominantly Douglas-fir, indicating that this species can persist and dominate the composition.

Climate Related Risks and Trends

Some studies suggest that Douglas-fir distribution will increase in a warmer climate (USDA Forest Service 2016b). Yet others anticipate more frequent high-severity fire events that will lead to loss of mature trees that serve as seed sources to the next generation of Douglas-fir (Perry and others 2011—from Giunta).

Comparison of the Natural Range of Variation and Current Conditions

Based on the tree composition characteristic of Douglas-fir types on the Ashley National Forest, the successional potential of species other than Douglas-fir varies across the Ashley National Forest. This characteristic appears to be at moderate departure from the natural range of variation.

Mixed Conifer and Engelmann Spruce

Mixed conifer and Engelmann spruce are extensively distributed on the Ashley National Forest, occurring on many landtype associations across the Ashley. These types are most prevalent on the Alpine Moraine, Trout Slope, Glacial Canyon, Stream Canyon, and Uinta Bollies landtype associations. Mixed conifer and Engelmann spruce are considered subalpine habitat types by Bradley (1992) and correspond to fire groups 10, 11, and 12. Mixed conifer generally occurs in the lower elevation moist-wet subalpine habitat, and Engelmann spruce generally occurs in the higher elevation and colder upper subalpine habitat.

Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) are nearly ubiquitous in the subalpine habitat type. These types are mixed with lodgepole pine (*Pinus contorta*), Douglas-fir (*Pseudotsuga menziesii*), blue spruce (*Picea pungens*), and aspen. Lodgepole pine is widespread in drier areas at the mid to lower elevations, often found on shallow soils, and can be seral to spruce and fir. Engelmann spruce becomes dominant in the cold subalpine, often at higher elevations, over 10,000 feet.

Table 9. Comparison of the mixed conifer and Engelmann spruce subalpine habitat types on the Ashley National Forest

Landtype Association	Mixed-conifer (acres)	Engelmann Spruce (acres)	Total
Alpine Moraine	115,674	90,511	206,185
Trout Slope	108,622	10,526	119,148
Glacial Canyon	35,644	4,496	40,140
Stream Canyon	16,306	202	16,508
Uinta Bollies	4,665	27,980	32,645
Total	280,911	133,715	414,626

Depending on the landtype association, the distribution of the vegetation types can vary. For example, the Uinta Bollies is recorded as having only 4,665 acres of mixed conifer but contains 27,980 acres of Engelmann spruce at the higher elevations. The majority of Engelmann spruce is within the High Uintas Wilderness, where there is little to no management. See table 9 for how mixed conifer and Engelmann spruce compare on the Ashley National Forest. Approximately 90 percent of the mixed-conifer on the Ashley National Forest, and 93 percent of Engelmann spruce, are represented in five landtype associations.

Description of the Natural Range of Variation

In the mixed conifer habitat type, the disturbance regime plays a strong role in driving species composition. Specific fire history information for Utah subalpine forests, however, is lacking (Bradley 1992). At lower elevation in the subalpine habitat type, fire intervals of 50 to 130 years have been estimated (Arno 1980). Disturbances lead to a mosaic of different ages and species compositions; multiple successional pathways are possible depending on climate and disturbance. For example, severe fire leads to dominance of seral species such as lodgepole pine and aspen. Aspen can act as a nurse tree for conifers on harsh sites. On the moist-wet subalpine habitats of Utah, severe fires are rare; stand-replacing fire intervals can be as high as 300 to 400 years (Bradley 1992). Severe fires only occur during extremely dry conditions, generally advancing from adjacent upland sites. More frequent smoldering fires remove single trees or groups of trees rather than entire stands, leading to all age condition. Engelmann spruce often persists as a seral species, a dominant climax species, or co-dominant climax with subalpine fir. In addition to spruce, seral species such as Douglas-fir, lodgepole pine, and aspen can be present (Bradley 1992).

In the Engelmann spruce habitat type, severe fires are rare and the interval becomes more infrequent at the higher elevations of this cold, subalpine habitat type. Fire return intervals are estimated at greater than 200 years and can be 300 to 400 year stand-replacing intervals. Engelmann spruce dominates late successional forests and can live 300 years or more when seral. At the highest elevation in cold subalpine habitat type communities, it is too cold for other species such as Douglas-fir, lodgepole pine, blue spruce and aspen to persist. Most of these stands have heavy snowpack well into the summer months; fuels only dry out in August and September about the time late season storms end the fire season. Moist conditions and discontinuous fuels create low fire hazard for this type.

In both the mixed-conifer and Engelmann spruce vegetation types, insects are an important driver of change and composition in the subalpine system. One of the most significant disturbance agents in the spruce-fir ecosystem is spruce beetle (*Dendroctonus rufipennis*). Spruce beetle is a native insect with periodic outbreaks. While spruce beetle populations usually exist at relatively low or endemic levels, they periodically grow to outbreak level[2], causing extensive (less than 90 percent) preferred host tree (*Picea engelmannii*) mortality in affected stands (Jenkins and others 2014). The mortality of spruce beetle-preferred overstory spruce can change species composition and shift stands toward shade-tolerant species such as subalpine fir (Jenkins and others 2014) where it is present in the understory.

Current Status and Trends

Mixed conifer and Engelmann spruce are the dominant vegetation types on the Alpine Moraine landtype association. Engelmann spruce becomes more dominant with increasing elevation, while lodgepole pine becomes less so. Lodgepole pine[3] is common at low elevations, where warmer and drier conditions allow for more frequent fire than at the higher elevations. Subalpine fir is generally the most shade tolerant of the three tree species of this landtype association, and might be expected to dominate large areas with long absences of disturbance. However, this tree rarely forms dominant stands on the quartz-rich sandstones of the Uinta Mountain Group. Engelmann spruce generally outlives subalpine fir by as many as 100 years or more. By longevity, Engelmann spruce commonly dominates on this landtype association, and subalpine fir is sometimes lacking or poorly represented. In places, subalpine fir forms a shrub-like layer that apparently persists for long periods without becoming the dominant tree layer (Goodrich 2005e, USDA 2009). With increasing elevation, aspen decreases and almost disappears. The composition of the forests on this landtype association is consistent with what has been described as the natural range of variation.

Mixed conifer is the dominant vegetation type on the Trout Slope landtype association, while the Engelmann spruce type is less present. Mixed coniferous forest stands of lodgepole pine, Engelmann spruce, and subalpine fir occur throughout much of the area above about 9,600 feet elevation. Lodgepole pine dominated stands[4] are common at lower elevations, and in some places, mid-elevations. Lodgepole pine stands on this landtype association have a lower level of cone serotiny than on the Greendale Plateau and Parks Plateau landtype associations, which may be a function of infrequent fire. As on the Alpine Moraine landtype association, subalpine fir sometimes forms a shrub-like layer that appears to persist for many years without trending toward maturity or dominance (Goodrich 2005e, USDA 2009). In areas underlain by dense clays derived from Red Pine Shale, subalpine fir seems to trend toward maturity in greater abundance and more rapidly than on the sandstone areas.

[2] Literature indicates that in these spruce-fir forests, spruce beetles generally persist in low-level, widespread populations that have little effect on forest structure. However, these forests periodically have very large outbreaks (Jenkins and others 2014)

[3] Lodgepole pine (as typed) is discussed in the persistent lodgepole section and totals 4,700 acres on this landtype association.

[4] Lodgepole pine, as typed and discussed in the persistent lodgepole section, occupies 6 percent of this landtype association, totaling 8,686 acres.

Community dynamics and sustainability on this landtype association are strongly influenced by fire. Lodgepole pine seedlings rapidly establish in many burned areas after fire, forming large areas dominated by even-aged stands of lodgepole pine. Engelmann spruce is also found in abundance following disturbance at higher elevations (Study 17-31A-E). Maturation of these forests will likely take 100 to 150 years for lodgepole pine. Mixed conifer areas of lodgepole pine, displaced by spruce and fir, could take up to 200 to 300 years or longer to reach stand maturation.

In the Stream Canyon landtype association, mixed-conifer forests occur primarily in talus or other rocky areas. Seventy percent of the mixed conifer on this landtype association is typed as conifer in rock. The mix of coniferous forest is more diverse than that of most other forested areas. Douglas-fir, ponderosa pine, lodgepole pine, limber pine, blue spruce, and Rocky Mountain juniper are found here in various combinations with narrowleaf cottonwood, aspen, and other deciduous trees. Blue spruce appears capable of displacing other tree species where there has been a long absence of disturbance. Engelmann spruce appears to play a lesser role on this landtype association.

On the Glacial Canyon landtype association, steep canyon walls with cliffs, rock fall, or talus are common features. Trees are able to colonize on some of the more moderate areas of rock fall and, in other areas with some mantle of soil, stands of aspen, lodgepole pine, Engelmann spruce, and mixed coniferous forests occur. These rock fall forests are likely static over rather long periods where they are protected from fire. Limestone areas are dominated by Engelmann spruce stands, which generally do not support lodgepole pine.

At the highest elevations, the majority of the Uinta Bollies landtype association is above tree line, with inclusions of tree-covered areas on old residual surfaces of the Uinta Mountain Group. Although the tree line has likely fluctuated historically, coniferous forests are typically located at the lower elevations. The Uinta Bollies are heavily weighted to Engelmann spruce forest. Generally only 20 percent has the capability to produce trees, due to a harsh climate and shallow soils. The trees that do grow here are often stunted and of krummholz growth form. Fire is rare, and fire and insects do not appear to be major drivers, although this may change with the advance of high spruce beetle populations. Of all the landtype associations on the Ashley National Forest, vegetative composition has been the least altered on this one.

The majority of this landtype association is within the High Uintas Wilderness; therefore, there is no active timber harvesting. Little to no past management has occurred in the majority of this landtype association.

On all landtype associations with a lodgepole pine or Engelmann spruce component, native bark beetle (*Dendroctonus* sp.) outbreaks are affecting the larger trees, (for example, Engelmann spruce greater than eight inches in diameter at breast height). These outbreaks are shifting tree composition. During the first decade of the 2000s, mountain pine beetle (*Dendroctonus ponderosae*) had been active in the mixed conifer type. Lodgepole pine is a host species for mountain pine beetle. Tree mortality was extensive and apparent in all of the glacial canyons of the south slope of the Uinta Mountains. Currently, a spruce beetle outbreak is affecting the Engelmann spruce component. Host availability and condition will reflect this ecological disturbance with increases in dead standing overstory Engelmann spruce on the landscape. On those landtype associations where subalpine fir is common in the understory of affected stands and Engelmann spruce trees of cone bearing age are attacked by spruce beetle, the composition would shift to subalpine fir or return to an early seral setting.

Climate Related Risks and Trends

Forests will be moderately vulnerable to a warmer climate. Subalpine fir, Engelmann spruce, and blue spruce could migrate to higher elevations. Lodgepole pine would persist in high-elevation landscapes.

Where Douglas-fir is a seral species, it could increase in distribution since it is more fire tolerant than associated species.

Fire may be more frequent and extensive in the subalpine habitat types, given climate projection data for the Intermountain Region (USDA Forest Service 2016b) and host condition, fuel conditions, and fuel availability. In addition to climate projections, increasing temperatures and fuel loads can create conditions conducive for more frequent and extensive fire.

Comparison of the Natural Range of Variation and Current Conditions

The current composition of the mixed conifer and Engelmann spruce vegetation types is consistent with that described in the natural range of variation. Native bark beetles, however, are altering this characteristic. Although the advancing spruce beetle epidemic is not an event that has occurred on the Ashley since Euro-American settlement, we know large-scale spruce beetle epidemics have occurred in this type before[5], but these data are limited. The extent and level of the advancing spruce beetle epidemic is consistent with the extent of available preferred host material (Engelmann spruce greater than eight inches in diameter at breast height) and impacts from a changing climate on insect life history (Bentz and others 2010, Bentz and Jönsson 2015, and Bentz and Munson 2000). Although spruce beetle is a natural part of spruce-fir forest ecology and a key disturbance agent, the impact of the current epidemic is threatening the sustainability and resilience of the subalpine system. Combined with climate projections of increasing temperatures and future fuel loads as dead trees fall, more frequent and extensive fire is expected.

In the mixed conifer vegetation type, some fire suppression has led to greater prevalence of late successional species such as subalpine fir in the mixed conifer vegetation type, especially in the Alpine Moraine and Trout Slope landtype associations. With bark-beetle-caused tree mortality, tree composition is expected to shift to subalpine fir where it is present. With increase in severe fire, lodgepole pine and aspen are expected to persist where they are present. Douglas-fir could increase in distribution where fire is mixed severity. Mixed conifer is therefore estimated at moderate departure, due to some fire suppression and expected climatic trends that would shift tree composition away from historical (e.g., spruce).

In the Engelmann spruce vegetation type, fires have generally not been suppressed; species composition is estimated to be at low departure from the natural range of variation. For reasons listed above, however, the combination of spruce beetle-caused tree mortality and climate projections that include increase in fire, composition in the Engelmann spruce type is estimated to be trending to a moderate departure from the natural range of variation. Tree composition is changing with the advance of spruce beetle-caused tree mortality. Combined with increases in fire, regeneration back to spruce would be delayed where seed tree sources are unavailable.

High-elevation spruce, and spruce growing in uneven-aged condition, may be more resistant or resilient to these potential stand-replacing events (Schmid and Frye 1977, Dymerski and others 2001). Surviving spruce tree seedlings and saplings in an uneven-aged condition provides the regeneration needed to perpetuate the species. Uneven-aged stands, however, are not immune to stand-replacing fires, especially those stands that have had a significant beetle-caused mortality event. The extent of uneven-aged stands is not known throughout the type, although the Trout Slope landtype association is noted for having uneven-aged stands at higher elevations and on wetter areas

[5] Paleo-ecological data of pollen and charcoal (as summarized by Jenkins and others 2014 Mock and Brunelle-Daines 1999, and Morris et al 2013)

Aspen

Aspen communities are found in a number of landtype associations (table 10). Table 11 identifies landtype associations that support aspen communities and comparative features of aspen for each landtype association.

Table 10. Vegetation type and acres in aspen communities on the Ashley National Forest

Vegetation type	Acres
Seral Aspen	117,198
Persistent Aspen	35,513
Total	152,711

Table 11. Landtype associations that support aspen communities on the Ashley National Forest

Landtype Association	Comparative Features
Limestone Hills	Mostly seral aspen. Aspen is seral to Douglas-fir on southerly, warm dip slopes. Common juniper (*Juniperus communis*), elk sedge (*Carex geyeri*), Oregon grape (*Mahonia repens*), and mountain snowberry (*Symphoricarpos oreophilus*) are common understory plants of aspen and Douglas-fir on this landtype association.
Limestone Plateau	Persistent aspen forms parklands with tall forb communities growing in the openings on deep clay-loam soils. This area supports the most developed tall forb communities known on the Ashley National Forest. Seral aspen is rare to nonexistent. Gooseberry (*Ribes* species) is a common understory plant.
Parks Plateau	Both persistent and seral aspen are present on this landtype association. Aspen is seral to lodgepole pine within the conifer belt with a shrub layer of common juniper or snowberry. Below the conifer belt, persistent stands of aspen are common and larger stands of aspen have formed near the ecotone with the coniferous forest. Smaller, isolated stands appear as islands well out into the mountain big sagebrush belt. Common juniper and elk sedge are common understory plants in these long-persistent aspen stand.
South Face	Persistent aspen form stands on leeward snow bed sites and on mesic slopes and drainages of cooler exposures. Stands are nearly pure with little to no coniferous tree presence. Seral aspen is uncommon. Understory species include snowberry and perennial graminoids.
Wolf Plateau	Seral aspen is very common. Aspen is seral to both subalpine fir and Engelmann spruce. Persistent aspen is relatively uncommon. Gooseberry, grasses, and forbs are common understory components. A few tall forb communities form in openings.
Avintaquin Canyon	Mostly seral aspen. Aspen is seral to Douglas-fir on steep canyon slopes and to subalpine fir toward the heads of the drainages. Following fire, ninebark (*Physocarpus malvaceus*) and wild hollyhock (*Iliamna rivularis*) form dense understories. Snowberry and wild rose (*Rosa woodsii*) are also common understory plants.
Strawberry Highlands	Both seral and persistent aspen are present. Seral aspen is mixed with subalpine fir at higher elevations and with Douglas-fir at the lower elevations. Stands of persistent aspen occur in a parklands setting and within drainages.

Description of the Natural Range of Variation

Aspen is a common montane plant community in the Uinta Mountains and is represented across the landscape within a broad range of environments, successional states, and community types. Aspen communities vary in species composition, structure, and their response to disturbances. Complexities within these communities are influenced by genetic variability, environmental conditions, and disturbance mechanisms (Mueggler 1988, Romme and others 2009).

Aspen understory assemblages are quite diverse. Most common understories consist of an array of shrubs, grasses, sedges, and forbs. However, some understories are grass and sedge dominant, while others are dominated by shrubs or forbs. Species richness can be high or consist of a handful of species. Total ground cover in these communities is high, near or at 100 percent in the absence of rodents. Aspen is generally fire dependent and most reproduction occurs after severe fire events (Romme and others 2009, Bartos and Mueggler 1981, Mueggler 1988, Brown and DeByle 1987). This has led many to conclude that fire suppression, increased herbivory, and climate change have led to the decline in aspen over the last century (Bartos, 2001, Kurzel and others 2007).

Aspen develop in clones, where a “single genetic individual” can cover small to large areas with numerous stems interconnected by a common root system (Romme and others 2009). Aspen clones in the plan area range in size from less than a tenth of an acre to dozens of acres. These clones are found across the landscape and range from small, isolated stands near the edge of their range to “broad expanses of pure and mixed stands” in the heart of their range (Mueggler 1988). Stress or disturbances that lead to stem or clone mortality (by fire, drought, or disease) typically trigger new sprouting, a function that perpetuates clone survival and vitality. Following disturbances, new sprouting has been documented to occur not only in, but beyond the original perimeter of, the clone. This new sprouting indicates size expansion of the clone following disturbance (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016). In some settings, where individual clones live in close proximity, clonal coalescing has occurred to form larger aspen forests. Additionally, Long and Mock (2012) found aspen “seedling establishment is common enough to be ecologically important and that genetic diversity is substantially greater than previously thought,” although successful establishment of aspen seedlings has not been verified in the plan area.

Although aspen communities have a high number of different plant associations, two types are generally recognized and discussed based upon successional features: seral aspen and persistent aspen. First, seral aspen functions as a seral species in many conifer communities. In the plan area, seral aspen grows in conjunction with lodgepole pine, Engelmann spruce and subalpine fir, Douglas fir, ponderosa pine, and blue spruce (Mueggler 1988). Over three-quarters of aspen on the Ashley National Forest is classified as seral. Romme and others (2009) found that, on the San Juan National Forest in Colorado, seral aspen was “found primarily at the higher elevations.” This is similar to seral aspen communities on the Ashley National Forest. Seral aspen is dependent upon periodic stand-replacing fire or catastrophic regeneration in order to persist in most conifer communities (Kurzel and others 2007). Following fire, new aspen sprouts mature and dominate communities until the sprouts are gradual displaced by conifers. One exception occurs in ponderosa pine, where aspen is found to be a co-dominant species (Romme and others 2009). In summary, the persistence of seral aspen correlates strongly with the level and duration of conifer encroachment and displacement, as well as the ability of a clone to regenerate following disturbance (Kurzel and others 2007).

Second, persistent aspen accounts for about 24 percent of all aspen in the plan area. Conifers are absent or nearly absent in these communities, which indicates aspen to be the long-term dominant type (Kurzel and others 2007). Other terms to describe these communities include climax, stable, or pure (Powell 1988, Mueggler 1988, Romme and others 2009, UFRWG 2010). There has been considerable discussion about the successional status of persistent aspen. Some view aspen as a climax dominant, citing environmental conditions that preclude conifer presence while others view aspen as stable or persisting due to long-term (centuries) successional processes (Romme and others 2009). Under either scenario, persistent aspen is recognized as a stable plant community. Persistent aspen communities in the plan area are typically found on south-facing or warmer aspects, and at lower to mid-elevations that are usually below the conifer zone, which is compatible with Romme and other's findings. In a survey of 100 aspen stands, Romme and others (2009) found "stable aspen stands were strongly associated with lower elevations" and where median fire intervals were shorter than those in higher elevation conifer communities. Historically and presently, large wildfires have periodically burned in areas where persistent aspen is commonly found. Many of these fires have been documented over a long period of time (Stewart 1911, Hitchcock 1910, Graham 1937, Heyerdahl 2011, Ogle and Dumond 1997, Huber and Goodrich 2016).

In contrast, a study in northwestern Colorado found stand-replacing fires "may not be necessary for aspen regeneration and persistence" in persistent aspen communities (Kurzel and others (2007). Three modes of aspen regeneration occur within the plan area. First, many stands of mainly seral aspen depend on large-scale, stand-replacing disturbances to trigger new sprouting to maintain presence or dominance of a site. This is referred to as stand replacing or catastrophic regeneration. In his study, Kurzel concluded that stand replacing regeneration dominated stand dynamics in seral aspen, but "over 70 percent of the persistent aspen stands regenerated by a regeneration mode that was primarily non-stand replacing."

Second, a few aspen clones perpetually show continuous regeneration, producing new sprouts regardless of disturbance or stress. This regeneration mechanism is relatively uncommon, occurring in only about 11 percent of aspen clones in a northwestern Colorado study (Kurzel and others 2007). Clones with this feature are multi-tiered due to continuous sprouting within relatively open aspen canopies. Continuous regeneration occurs in some clones on the Ashley National Forest, but less than five percent of the clones studied demonstrate this capability (Huber and Goodrich 2016).

Third, the most common non-stand-replacing regeneration mode in persistent aspen is episodic regeneration. In the absence of fire and other stand-replacing events, pulses of new sprouting following senescence has been repeatedly documented across the Ashley National Forest (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016), similar to findings by Kurzel and others (2007). In most persistent aspen stands, apical dominance is the controlling factor in aspen sprout abundance and survival (Schier and others 1985). Apical dominance in aspen is where aspen clones devote energy to upward growth of their terminal shoots (trees) at the expense of new suckering from their root systems below. During canopy development, clones produce a plant hormone auxin, which inhibits or suppresses new suckering.

Closed stands typically produce a few, inconspicuous sprouts each growing season, but these stands seldom survive to maturity. Such conditions persist until stand replacing disturbances or tree dieback during senescence remove or deplete aspen canopies. Senescence is most common in aspen stands that are 120 to 150 years old, where aging stands are at or trending toward decadent conditions. "The period of senescence may produce a loss of apical dominance for canopy trees, leading to increased suckering in conjunction with increased light environment" (Kurzel and others 2007). They concluded that "episodic regeneration is the main mode of stand re-initiation and . . . that at a landscape scale this may be the most widespread mode of aspen regeneration in northwestern Colorado." To expect numerous sprouts in all mature and vigorous aspen stands is contrary to the apical dominance concept described by Schier (1975, 1985). Long-term monitoring on the Ashley National Forest validate these conclusions regarding apical dominance and episodic regeneration (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016).

Current Status and Trends

Aspen communities on the Ashley National Forest have been subject to numerous stressors, disturbances, and management practices during the last 30 years. However, distribution, abundance, and function of aspen have been relatively constant and within expected parameters. There are a few notable conditions and trends that have emerged during the last 30 years.

Aspen continues to be represented across the landscape within its natural elevation and topographical range. This representation consists of diverse and numerous community types, and provides important ecological and anthropogenic services. These conclusions are verified by the numerous long-term studies that monitor condition and trend of aspen communities found on the Ashley National Forest (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016).

Seral aspen is the most common community type, accounting for over 75 percent of aspen communities. However, current monitoring indicates seral aspen is diminishing within the plan area and is being displaced by coniferous trees on the Ashley National Forest. There is even greater risk for aspen loss under current broad-scale fire suppression policies and continued reduction in timber harvest. O'Brien and Tymcio (1997) determined only 31 percent of forest stands once dominated by seral aspen now have enough aspen within these stands to be considered an aspen forest type. This doesn't mean that the other 69 percent cannot become a seral dominant following stand replacing disturbance. Romme and others (2009) "view[ed] these changes in conifer abundance as natural successional processes that have always occurred during long periods without major disturbance." Others have concurred with this argument and concluded that seral aspen was not declining if considered within the context of an entire successional cycle (Smith and Smith 2005). These conclusions are similar to current conditions and trends of seral aspen on the Ashley National Forest. Successful regeneration of seral aspen continues to occur following fire or timber harvest under conifer-dominant conditions (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016). Several examples of successful aspen recruitment following recent stand-replacing fire include the Weyman, Iron Mine, Cow Hollow, Bumper Canyon, Spring, Water Hollow, Church Camp, and Neola North wildfires (Studies 34-5B-C, 34-6, 58-58A-C,F, 59-15A-C, 64-52B,E,G, 58-28A-C,F-G, 66-15F,H-I,M-N, Goodrich and Huber 2008, Zobell and Goodrich 2015).

Photo of study 2-2D. Sheep Creek Park. 2004. View of seral aspen community. Photo taken 26-years post-fire.

Persistent aspen in the plan area continues to be a plant community of high value, providing a number of anthropogenic and ecological services. During the last 30 years, nearly all these communities have demonstrated resilience concurrent with contemporary stressors and drivers. Successful regeneration of persistent aspen has been documented through stand replacing fire, continuous, and episodic regeneration pathways. Numerous stands of persistent aspen, both small and large, have been burned by wild and prescribed fire over the last 30 to 40 years (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016). In every instance but two, long-term studies have documented new sprouting post-fire and sufficient sprout survival to perpetuate the long-term viability of aspen. Exceptions include an isolated aspen clone of less than an acre that did not regenerate following a 1998 prescribed fire treatment and a small aspen stand on Anthro Mountain where sprout density was severely reduced by elk herbivory (Study 32-85H, 68-7B). Within five years post-fire, sprouts have routinely exceeded six feet in height, with 50 percent canopy cover or greater, indicating successful aspen recruitment of burned stands. Additionally, new sprouting in many burned aspen stands exceeded their pre-burn perimeters by five to 50 feet, indicating stand expansion. Plants of moderate to high resource value have dominated understory plant composition within a year following fire. Satisfactory conditions regarding aspen recruitment and understory species composition strongly suggest an upward trend in nearly all burned aspen stands on the Ashley National Forest.

Photo of study 68-53B. Anthro Mountain. 2011. View of persistent aspen.

Some clearcut treatments or harvests in persistent aspen have occurred since 1980. Most treatments were small (less than 5 acres) and occurred within or along the perimeter of larger persistent aspen stands. Following treatment, most clearcuts exhibited sprouting similar to burned aspen stands, but a couple of treatments failed to initiate sprouting (Studies 41-4F, 43-25A). Failure in these stands is likely due to small treatment sizes, failure to treat the entire clone, coupled with possible wild ungulate herbivory at one of sites. Few aspen timber harvests have occurred during the last 30 years. However, sprouting and understory vegetation response post-harvest is similar to clearcut treatments. One exception is where heavily used vehicle pathways and excessive vehicle use occurred within the harvest area (Studies 64-20A, 64-33). The risk of unsuccessful aspen regeneration is higher in treatments that are small and where excessive vehicle use within the clone takes place following treatment.

Few persistent aspen clones on the Ashley National Forest demonstrate the ability to perpetually regenerate. Clones with this capacity have multiple age or height classes of aspen. This occurs where height classes of trees range from six to 20 feet in height under a canopy of mature trees and where many saplings and young trees appear to be trending towards maturity (Huber and Goodrich 2016, Studies 30-24E-F, 40-1). In these clones, stand replacing disturbance or clonal senescence is not necessary to trigger a sprouting event. Some researchers hypothesize that continuous regeneration may be due to a “relatively high-light environment” found under relatively open aspen canopies (Kurzel and others 2007). Based on the numerous studies established within persistent aspen stands, less than 5 percent of aspen clones in the Uinta Mountains exhibit continual regeneration characteristics, far less than reported by Kurzel and others (2007). Understory species composition in these communities are dominated by native plants of moderate to high value for watershed protection, and fall in the natural range of variation. In summary, satisfactory conditions regarding aspen recruitment and understory species composition indicates stable trend in continuous regenerating aspen stands on the Ashely National Forest (Huber and Goodrich 2016, Studies 30-24E-F, 40-1).

Photo of study 30-24F. Dry Fork Mountain. 2010. View of aspen with continuous regeneration.

Episodic regeneration is the most common mode of regeneration of persistent aspen on the Ashley National Forest (Huber and Goodrich 2016). In most stands of persistent aspen, sprout development is largely suppressed by apical dominance (Schier and others 1985). These stands usually consist of one height class or cohort of trees, and clones produce a few inconspicuous sprouts annually; but these usually die before reaching maturity unless they occur in a canopy opening. In the absence of stand-replacing disturbances, pulses of abundant and successful sprouting occur following die-back within the mature canopy. This usually occurs during senescence of aspen cohorts. Numerous events of episodic regeneration are documented within persistent aspen, with most occurring after the year 2000. Die-off of persistent aspen increased during and after severe droughts in the years 2002, 2012, and 2013 (Huber and Goodrich 2016, Huber and Goodrich 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b). Some aspen stands experienced nearly 100 percent mortality of mature trees followed by successful sprouting events (Studies 32-17D, 32-31D), while most stands experienced only partial die-back of the canopy with similar results (Studies 31-40B, 32-67C-D, 3-23J, 64-11E, 57-3B, 68-66F). In most instances of episodic regeneration, new sprouting exceeded the perimeter of the original clone by up to 50 feet, resulting in stand expansion. Similar to burned aspen stands, sprouts have routinely exceeded six feet in height with canopy covers at or greater than 50 percent three to five years after the sprouting event. Plant assemblages in the understory show little to no change during episodic regeneration. Plants of moderate to high resource value dominate understory plant composition. Satisfactory conditions associated with aspen recruitment and understory species composition implies there is an upward to stable trends in persistent aspen experiencing episodic regeneration. In conclusion, "episodic regeneration via replacement dieback may allow many of these stands that appear to be "decadent" or deteriorating to persist in the landscape in coming decades"(Kurzel and others 2007). The concept of apical dominance with consideration to modes of regeneration should be the focal point when determining standards and guidelines that pertain to persistent aspen management.

Photo of study 32-67D. Diamond Mountain. 2012. View aspen clone with episodic regeneration occurring.

Monitoring of aspen regeneration in the Uinta Mountains is summarized in PowerPoint presentations for the north slope (Zobell and Goodrich 2015), south slope east (Goodrich and Huber 2015b), south slope west (Huber and Goodrich 2015b) and West Tavaputs Plateau (Huber and Goodrich 2015) for more than 200 study sites.

Management Activities and Uses

Aspen contain diverse and numerous community types, which provide many valuable ecological and anthropogenic services. These communities occupy several million acres of rangeland and occur on mountain landscapes where annual precipitation rates average about 16 inches, increasing their watershed value, especially in the arid Intermountain West (DeByle and Winokur 1985). Aspen tends to occur where intermediate to high winter snowpack accumulates and summer precipitation is higher. It has been reported that 5 to 70 percent more water is found in snowpack under persistent aspen than in snowpack under conifer-dominated communities (DeByle 1985b). Aspen watersheds yield valuable run-off for consumptive uses in arid lowlands (irrigation, culinary use, storage). Soils in aspen communities have a high capacity to absorb and hold water. Furthermore, aspen protects watersheds and provides excellent water quality because little to no erosion occur within these communities. During the next plan period, aspen will continue to be an important vegetation community for watersheds, and monitoring the amount of maintenance and quality of aspen appears imperative.

Aspen communities are very productive, diverse and structurally complex, which is attractive to wildlife. These qualities make aspen prime habitat for numerous wildlife species, especially birds, where species richness is relatively high. Behle and Perry (1975) found that about 60 bird species use persistent aspen in Utah. A variety of birds depend on aspen for nesting, foraging, and cover. These include cavity nesters, songbirds, raptors, and game birds. Numerous mammals also use aspen communities, with at least 55 species identified (DeByle 1985). These include big game, predators, and small mammals such as mice and voles. Beaver depend on aspen for food and building material.

Aspen communities provide a range of anthropogenic services, with livestock grazing as the most extensive and enduring use of these communities. Almost all persistent aspen communities are classified “capable” for livestock grazing. These communities are very productive, producing from 500 to more than 2,000 pounds of forage per acre. Persistent aspen on the Ashley National Forest is readily accessible to wild and domestic ungulates, which are usually found on gentle to intermediate slopes. These aspen understories are usually composed of a variety shrubs, grasses, sedges, and forbs, which makes these communities attractive rangeland for both cattle and sheep. Historically, cattle and sheep have grazed all or nearly all persistent aspen within the plan area. Light to intensive livestock grazing in these communities has occurred for more than a 100 years. During the last 30 years, domestic sheep grazing declined considerably, and all but one domestic sheep allotment with persistent aspen have been either closed or converted into cattle allotments. Over the last 30 years, decreases in the numbers of domestic sheep have been voluntary, which is correlated with the economic decline in the sheep industry. Future trend of domestic sheep grazing is expected to either remain stable or continue downward. On the other hand, cattle grazing has increased in aspen communities, but intensity has either remained stable or has decreased on some allotments over the last 30 years. During the next plan period, the high value of persistent aspen as a producer of livestock forage is expected to remain constant.

Aspen also provides fuelwood and wood fiber for manufacturing (Wengert and others 1985). However, aspen is in low demand for fuelwood and the demand for wood fiber is almost non-existent within the plan area. Most people who harvest fuelwood on the Ashley National Forest opt for conifers such as lodgepole pine, Engelmann spruce, pinyon pine, ponderosa pine, and Douglas fir. Very few aspen fuelwood harvests have occurred during the last 30 years, with none occurring within the last couple of decades. Future demand for aspen as fuelwood is expected to remain constant or perhaps slightly decline. There currently is little to no demand for aspen wood fiber within the plan area. This industry locally is nonexistent, and the demand for wood fiber during the next decade is expected to remain constant or possibly slightly increase.

Aspen communities also provide a recreation value. These areas are sought out by many for their scenic beauty, especially during autumn months. Aspen communities provide a variety of recreation opportunities such as camping, hiking, sightseeing, all-terrain vehicle use, horseback riding, hunting, snowmobiling, and cross-country skiing. Recreational use of these areas is highest during the autumn big game hunts, and summer when most camping occurs. Persistent aspen stands nearest to roads receive the highest use, especially for recreational vehicle (RV) camping. Many long-term, dispersed campsites are located within aspen communities. Increased recreation use in aspen is expected during the next plan period.

Influences of Drivers and Stressors

The most recognized and understood driver of aspen communities is fire. Numerous stands of persistent and seral aspen, both small and large, have burned with wild and prescribed fire over the last 30 to 40 years (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016). Successful sprouting following fire has routinely occurred concurrent with other drivers and stressors of aspen. Aspen stands often increase in size following fire, when new sprouting occurs beyond the perimeter of the original stand. Most fires have been less than 200 acres in size overall, affecting less than 100 acres of aspen in one event, but there have been several larger fires that have burned hundreds of acres of the aspen component in one event. These fires include the Weyman, Neola North, and Church Camp wildfires and the Petty Mountain prescribed fire (Zobell and Goodrich 2015, Goodrich and Huber 2008, Studies 3-9A, 3-17A,D, 38-58A-B,E-F, 66-15F,H-I,M-N). Historical accounts reveal that stand replacing wildland fire occurred periodically in the Uinta Mountains, both at large and small scales. For example, S.S. Stewart surveyed the east end of the Ashley National Forest in 1909. His subsequent report listed burned areas for Cart Creek, Little Brush Creek, Big Brush Creek, an area between Sims Peak and Taylor Mountain, Mosby Mountain, and lower portions of Dry Fork (Stewart 1911). Stewart reported one fire in 1879 that covered the greater part of four townships in the southeast portion of the forest, and that smaller fires subsequently burned over much of the same area. The 1879 burn area was estimated to be about 24 miles long, an area that spans from about Mosby Mountain eastward, to the forest boundary on the Diamond Mountain Plateau. Other reports of historic fires include the Uinta and Pole Creek drainages and large tracts of forest in Lake Fork and Dry Gulch drainages (Hitchcock 1910). Furthermore, Graham (1937) included the following comment about the Wheeler Expedition of 1871. When this expedition reached the Uinta Mountains, “the full complement of topographical work was prevented by forest fires of great extent”. Heyerdahl and others (2011) also documented a large fire in the Brownie Canyon - Dry Fork Canyon area that same year. Prescribed fire events within persistent and seral aspen are expected to either occur at current rates or possibly increase during the next plan period. Wildfire occurrence is strongly related to environmental and climatic conditions. If the climate continues to warm, fire frequency is predicted to increase. On the other hand, if the climate cools, fire frequency is predicted to decrease. The current rate of fire is not likely to induce an upward trend in seral aspen acres within the plan area. Increased fire frequency in a warmer climate is predicted to benefit seral aspen communities. In contrast, persistent aspen is expected to be sustained or show increases in area under current fire return intervals. Increased fire frequency, which implies decreased precipitation, would likely result in a decline persistent aspen.

Anthropogenic disturbance, such as timber harvest, can parallel the regeneration effects of fire. During the last 30 years, hundreds of acres of seral aspen have started new sprouting, following timber harvests of conifer trees (Goodrich and others 2015b). Aspen presence in these settings have been sustained through timber harvesting in conifer-aspen forest types. Timber harvesting trended downward between 1987 and 2008, where it has remained level since. Timber harvesting is expected to remain level or possibly trend downward during the next plan period. Future effects of timber harvesting on seral aspen is dependent upon this trend.

An important ecological feature of persistent aspen is that it benefits from disturbances and other die-back mechanisms that cause tree mortality. Die-off or die-back of aspen usually triggers new sprouting at levels that ensure its persistence or dominance of a site. The condition, called “cohort senescence”, makes persistent aspen more susceptible to insects, disease, or drought; factors commonly attributed to aspen die-back (Kurzel and others 2007). Die-back processes are better understood when stand age of aspen is considered. In the absence of stand replacing disturbance, aspen trees or cohorts begin to senesce or become decadent after about 100 years. Senescing cohorts lose vigor and are less resistance to disease, insects, and drought, ultimately succumbing to these stressors or old age. Die-back of senescing aspen is well documented within the plan area (Huber and Goodrich 2016). Die-back events increased during severe droughts in 2002 and 2012. Loss in apical dominance of canopy trees due to senescence and subsequent die-back resulted in increased and abundant sprouting in affected aspen stands. Additionally, new sprouting occurred beyond the perimeter of dying cohorts in most stands, which demonstrates aspen expansion. Episodic regeneration is common in persistent aspen within the plan area. The regeneration is expected to continue during the next plan period, as aspen clones age and begin to senesce. Persistent aspen is expected to be sustained or show modest increases in areas under processes of senescence and die-back, coupled with occasional stand-replacing fire.


Photos of study 31-32D1. Taylor Mountain. 2006 and 2016. View of episodic regeneration triggered with die-back of aspen clone.

Aspen communities are important for forage and cover by both domestic and wild ungulates. Most livestock grazing occurs in persistent aspen as opposed to seral aspen. This is because persistent aspen are very productive, readily accessible, and their understories usually consist of palatable grasses and sedges. Some livestock grazing does occur in seral aspen. Grazing increases after fire when forage and aspen is more abundant, but diminishes over time as conifers displace aspen and understory species. Various intensities of livestock grazing have occurred in aspen for more than a 100 years. Little to no information is available pertaining to livestock effects on aspen prior to the 1950s, but repeat photography indicates that aspen persisted during decades of heavy grazing by cattle and sheep. Domestic sheep grazing has decreased substantially in aspen communities since 1970, but cattle grazing has increased during the same time period. During and prior to the last plan period, successful recruitment of aspen following stand replacing fire and periodic die-back has occurred throughout the plan area, concurrent with the presence of livestock (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016). Additionally, successful aspen recruitment post-fire occurred regardless of livestock stocking rates, seasons of use, or livestock management strategies. Livestock browsing of aspen sprouts has been minimal and not sufficient to affect successful recruitment or diminish stand persistence. Livestock grazing in the terms of numbers, class of livestock, and management is expected to remain relatively constant during the next plan period. Persistent aspen is expected to be sustained and successful aspen recruitment is expected to occur concurrent with contemporary livestock stocking rates and management strategies. Livestock grazing is expected to minimally effect seral aspen communities.

Aspen is considered prime habitat for both elk and mule deer. These species have impacted some aspen historically and recently. Prior to 1960, elk populations were very low, while mule deer populations were high. Unlike other areas in the West, there is no information indicating that mule deer suppressed aspen recruitment or diminished persistence during that period, although that may have occurred (Romme and others 2009). The only notable impacts during this time period are high-lining (i.e., browsing as high as can be reached) of some aspen by mule deer (Study 37-11A). During the last plan period, mule deer populations have decreased and elk populations have increased considerably. On the South Slope herd unit, elk numbers increased from 7,470 animals in 2004 to 8,700 animals in 2010 (UDWR 2012). On the Anthro Mountain subunit, elk numbers increased from a couple of dozen animals in the 1970s to 1,450 by 2009, which is about 700 animals above the current herd management objective (UDWR 2012b). Numerous studies on the Ashley National Forest indicate that current elk populations have minimal impact on aspen recruitment following fire, timber harvest, or die-back in most stands in the plan area (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich

2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016). For the most part, elk browsing of aspen sprouts is not sufficient to suppress successful recruitment or diminish stand persistence. One exception is Anthro Mountain. Due to the limited number of acres of aspen on the Anthro Plateau landtype association, aspen is more susceptible to elk browsing than on other aspen-bearing landtype associations. Following a 1996 prescribed fire treatment, elk browsing of new sprouts severely curtailed recruitment and threatens aspen persistence (Huber and Goodrich 2015, 68-7A). Elk populations are predicted to increase during the next plan period. If elk populations continue an upward trend, more aspen stands within the plan area would be susceptible to elk browsing following disturbances, which may threaten successful aspen recruitment and diminish aspen persistence. Since elk populations are greatly influenced by management objectives, this upward trend in elk populations is considered a potential stressor of aspen communities.

Recreation is a minor stressor of aspen communities. Recreation use such as camping, hiking, and all-terrain vehicle use have had minimal effects on aspen overall and impacts are site specific. For example, some mortality of mature trees or new sprouts have occurred at some dispersed camping sites in aspen. However, the mortality is not extensive enough to impact regeneration processes or diminish persistence of aspen. Increased recreation use of the Ashley National Forest is predicted during the next plan period, but this trend is not expected to adversely affect aspen persistence within the plan area.

Climate Related Risks and Trends

Climate change is potential driver of aspen. Seral aspen may benefit from a warmer and drier climate. The frequency and extent of fires would increase in montane forests, including conifer-aspen communities, where conditions become consistently warmer and drier (Rice and others 2016). More frequent and larger fires would move conifer-aspen communities toward early seral and mid-seral stages, which would favor greater aspen dominance of these communities. Persistent aspen may also benefit from a warming climate. The extent of these communities may increase as conifer communities migrate upslope under warming climates, if annual precipitation remains within its range of variability. Higher fire frequency would likely reduce the number of older-age, senescing aspen stands and perhaps improve clone vigor and health with more frequent cohort turnover. Since stand expansion often occurs with new sprouting, frequent fire would increase total area of persistent aspen over time. On the other hand, if the changing climate becomes drier and precipitation falls below required water needs, affected persistent aspen may be displaced by other communities such as mountain big sagebrush (*Artemisia tridentata var. vaseyana*). In this case, upslope migration of persistent aspen must occur to maintain these communities, and/or seral aspen becomes persistent as conifers die-off and these communities migrate upslope. If a cooler and wetter climatic trend occurs, seral aspen would be adversely affected. Less frequent and smaller fires resulting from cool and wet conditions would favor late-seral conifer dominance, with loss of seral aspen over time. Lower fire frequency would likely increase the number of older-age persistent aspen stands and episodic regeneration from senescing aspen would become increasingly common. Some downslope migration of conifers may occur and establish within higher elevation persistent aspen, but these communities are expected to persist within a favorable climatic zone. In conclusion, long-term monitoring has not detected change in aspen recruitment, persistence, or elevational movement directly related to current trends in climate.

Comparison of the Natural Range of Variation and Current Conditions

Seral Aspen

Similar to south-central Colorado, current monitoring on the Ashley National Forest indicates that seral aspen is diminishing within the plan area in terms of a “decrease in aspen density, basal area (size of the stem), or cover” because the frequency and extent of fires have been reduced, which indicates moderate departure from the natural range of variation (Romme and others 2009). Natural range of variation shows a higher fire frequency with larger fires similar to those documented during the 1870s. Large and small scale fires and timber harvest have temporarily removed conifers at a number of seral aspen sites on the Ashley National Forest, but their occurrence is not sufficient enough to maintain seral aspen long term. If the climate continues to warm and become drier, fire frequency and size is predicted to increase, which would be beneficial to seral aspen. More conifer-aspen communities would transition to early and mid-seral stages, from an increase in stand-replacing fires.

Persistent Aspen

Persistent aspen has persisted on the landscape at levels equal to or greater than at the beginning of the last plan period, which indicates sustainability over a 30-year period. Successful regeneration of these communities has occurred under stand replacing fire, continual, and episodic regeneration processes. Successful recruitment following stand replacing fire are documented at numerous sites and very few failures have occurred. Many persistent aspen stands experienced die-back following severe drought in the year 2002. Abundant sprouting in affected stands began a year later and continues to occur today. Successful episodic regeneration has occurred in the absence of stand replacing fire, but concurrent with other known stressors such as insects, disease, wild ungulate browsing, and livestock grazing. Many persistent aspen stands show expansion due to new sprouting outside the perimeter of the dying cohort (Studies 31-14D, 32-86C, 32-86I1, 34-6A, 34-17H, 37-28C, 39-32B, 41-4B3, 41-4B4, 50-6, 57-3B, and 68-66F). No complete die-off of persistent aspen related to drought stress has been documented, as it has been in southwestern Colorado (Worrall and others 2010). In some cases of minor scale, moderate to heavy ungulate impact to aspen regeneration has been documented (Studies 31-14C, 32-5, 34-11C, 34-11D, 39-17Q, 43-25A, 68-7B). However, long-term monitoring demonstrates aspen sustainability within the plan area concurrent with permitted livestock grazing and wild ungulate use. Elk numbers within in the State of Utah are expected to increase over the next decade based on the current trend to manage for increased state population objectives. Although current elk populations present little risk to sustainability of aspen at a landscape level, increases in elk populations may curtail future regeneration of many persistent aspen stands. If the climate gets warmer and drier, persistent aspen may experience higher fire frequencies, community shifts upslope, and displacement of lower elevation stands by shrub and grass communities. However, long-term monitoring has not detected change in aspen recruitment, persistence, or elevational movement directly related to current trends in climate. Shrub and herbaceous understories in persistent aspen consist of native plants with moderate to high resource value, and total ground cover at or near 100 percent. Existing conditions and current trends indicate that persistent aspen is near or within NRV and is expected to remain so during the next plan period.

Sagebrush

Sagebrush communities are found in a number of landtype associations on the Ashley National Forest (table 12). Table 13, table 14, and table 15 identify landtype associations that support mountain big, Wyoming, and black sagebrush communities and the comparative features of these sagebrush types for each landtype association.

Table 12. Vegetation type and acres in sagebrush communities on the Ashley National Forest

Vegetation type	Acres
Mountain big sagebrush	89,839
Wyoming big sagebrush	15,500
Basin big sagebrush	2,988
Spiked big sagebrush	257
Black sagebrush	10,695
Fringe sagebrush	960
Silver sagebrush	1,406
Total	121,645

Table 13. Landtype associations that support mountain big sagebrush (*Artemisia tridentata* var. *vaseyana*) on the Ashley National Forest

Landtype Association	Comparative Features
Parks Plateau	Vulnerability to cheatgrass (*Bromus techtorum*) is low. Potential for displacement by conifer is low. High number of herbaceous understory species; recent burned areas with 30 or more herbaceous species. Comparatively light use of sagebrush and other shrubs by wild ungulates. Greater sage-grouse nesting and brooding occurs at lower and mid elevations.
South Face	Vulnerability to cheatgrass is moderate to high. Potential for conifer displacement is low. Comparatively low plant species richness found in the herbaceous understory; recent burns usually with less than 15 herbaceous species. Sagebrush use by wild ungulates is moderate. Use of bitterbrush (*Purshia tridentata*) by ungulates is high. Greater sage-grouse nesting and brooding occurs on this landtype association.
Glacial Canyon	Vulnerability to cheatgrass is moderate to high. Potential for conifer displacement is moderate to high. Comparatively low plant species richness found in the herbaceous understory. Sagebrush use by wild ungulates is typically light. Greater sage-grouse use likely rare to non-existent.
Stream Pediment	Vulnerability to cheatgrass is moderate to high. Displacement by conifer is light to moderate. More serviceberry (*Amelanchier alnifolia*) occurs here than in other landtype associations. Use of sagebrush and other shrubs by wintering ungulates is moderate to high. Greater sage-grouse nesting and brooding occurs on this landtype association.
Anthro Plateau	Vulnerability to cheatgrass is currently low. Potential for conifer displacement is high at lower elevations and low at higher elevations. Moderate plant species richness found in the herbaceous understory. Herbaceous species composition somewhat different from landtype associations of the Uinta Mountains (e.g., Salina wildrye [*Elymus salina]*) is not found in landtype associations of Uinta Mountains). Potential for ground cover is likely lower than other landtype associations due to geologic formations. Greater sage-grouse breeding, nesting, and brooding occurs on this landtype association.
Avintaquin Canyon	Vulnerability to cheatgrass is low. Potential for conifer displacement is high. Moderate plant species richness found in the herbaceous understory. Herbaceous species composition somewhat different from landtype associations of the Uinta Mountains (e.g., Salina wildrye and spike fescue [*Leucopa kingii*] are not found in landtype associations of Uinta Mountains). Lower elevations preferred by wintering ungulates. Greater sage-grouse nesting limited to lower edge of this landtype association.

Landtype Association	Comparative Features
Strawberry Highlands	Vulnerability to cheatgrass is low. Potential for conifer displacement is moderate to high. Moderate to high plant species richness found in the herbaceous understory. Use by wintering wild ungulates is very low except in unusually mild winters. No greater sage-grouse wintering due to high elevations and little, if any, nesting or brooding. There are no records of greater sage-grouse use of this landtype association, but use is expected in the late summer and early autumn.
Structural Grain	Vulnerability to cheatgrass is currently low to moderate. Comparatively high plant species richness found in the herbaceous understory. Potential for conifer displacement is moderate to high. Use by wintering ungulates is moderate to heavy in mild winters. Greater sage-grouse breeding, nesting, and brooding occurs on this landtype association.
Dry Moraine	Vulnerability to cheatgrass is moderate to high. Potential for conifer displacement is low. Low to moderate plant species richness found in the herbaceous understory. Sagebrush use by wild ungulates is low to moderate. Greater sage-grouse nesting and brooding occurs on this landtype association.

Table 14. Landtype associations that support Wyoming big sagebrush (*Artemisia tridentata* var. *wyomingensis*) on the Ashley National Forest

Landtype Association	Comparative Features
Green River	Vulnerability to cheatgrass is moderate to high. Annual precipitation about eight inches. Fire frequency is very low. Vegetation production is low. Low plant species richness; usually with less than seven herbaceous species. Total ground cover is low, rarely exceeding 65 percent. Comparatively light to moderate use of sagebrush by wild ungulates, mainly pronghorn antelope. Greater Sage-grouse nesting and brooding occurs on this landtype associations.
Antelope Flat	Vulnerability to cheatgrass is high. Potential for conifer displacement is low. Annual precipitation about nine inches. Fire frequency is low. Vegetation production is low. Low plant species richness found in the herbaceous understory; usually with 10 herbaceous species or less. Total ground cover is low, rarely exceeding 65 percent. Sagebrush use by pronghorn antelope is moderate to heavy. Greater sage-grouse breeding, nesting, and brooding occurs on this landtype association.
North Flank	Vulnerability to cheatgrass is low to moderate. Annual precipitation about 12.5 inches. Fire frequency is low, but higher than Green River and Antelope Flat landtype associations. Potential for conifer displacement is moderate to high. Vegetation production is moderate. Comparatively moderate to high plant species richness found in the herbaceous understory; usually 10 to 20 herbaceous species. Sagebrush use by wild ungulates is light to moderate. Greater sage-grouse use likely rare to nonexistent.
Structural Grain	Vulnerability to cheatgrass is moderate to high. Annual precipitation about 12.5 inches. Fire frequency is low, but higher than Green River and Antelope Flat landtype associations. Potential for conifer displacement is moderate. Vegetation production is moderate. Comparatively moderate to high plant species richness found in the herbaceous understory; usually 10 to 20 herbaceous species. Total ground cover is low to moderate, rarely exceeding 70 percent. Sagebrush use by wild ungulates is light to moderate. Greater sage-grouse use is rare.

Landtype Association	Comparative Features
Anthro Plateau	Vulnerability to cheatgrass is moderate to high. Annual precipitation between 12 to 13 inches. Fire frequency is relatively low, but higher than other landtype associations. Potential for conifer displacement is moderate to high. Vegetation production is moderate. Comparatively moderate to high plant species richness found in the herbaceous understory; usually 10 to 20 herbaceous species. Total ground cover is low to moderate, rarely exceeding 70 percent. Sagebrush use by wild ungulates is light to moderate. Greater sage-grouse use is rare, but some winter use may occur.

Table 15. Landtype associations that support black sagebrush (*Artemisia nova*) on the Ashley National Forest

Landtype Association	Comparative Features
South Face	Vulnerability to cheatgrass is high. Potential for conifer displacement is moderate to high. Comparatively high plant species richness found in the herbaceous understory; usually 20 to 35 herbaceous species. Sagebrush use by wild ungulates is moderate to high. Greater sage-grouse breeding, nesting, and brooding occurs
North Flank	Vulnerability to cheatgrass is low. Potential for conifer displacement is moderate to high. Comparatively moderate plant species richness found in the herbaceous understory; usually 15 to 25 herbaceous species. Sagebrush use by wild ungulates is moderate to high. Greater sage-grouse use is rare.
Anthro Plateau	Vulnerability to cheatgrass is low in communities located above the pinyon-juniper belt, but moderate in communities in and below the pinyon-juniper belt. Potential for conifer displacement is moderate to high. Comparatively moderate to high plant species richness found in the herbaceous understory; usually 20 to 30 herbaceous species. Sagebrush use by wild ungulates is light. Greater sage-grouse breeding, nesting, and brooding occurs.
Green River	Vulnerability to cheatgrass is low to moderate. Potential for conifer displacement is low. Comparatively low plant species richness found in the herbaceous understory; usually 7 to 15 herbaceous species. Sagebrush use by wild ungulates is light to moderate. Greater sage-grouse use is rare.

Description of the Natural Range of Variation

Communities of sagebrush are common to the Uinta Mountains, Tavaputs Plateau, and high deserts of Wyoming. These communities are represented across the landscapes of the Ashley National Forest within a broad range of environments, successional states, and community types. Sagebrush prefer drier and cooler environments generally, but diversification of the taxa exists because of variability in geography, climate, and topography (Kitchen and McArthur 2007, USDA Forest Service 2016a). In a more refined scale, precipitation, temperature, and soil strongly influence distribution and species composition of the major sagebrush taxa in the plan area (Goodrich and others 1999, Kitchen and McArthur 2007, USDA Forest Service 2016). Sagebrush communities - where annual precipitation is above 13 inches - have higher species richness, herbaceous vegetation cover, and total ground cover than sagebrush communities where annual precipitation is less than 13 inches (Goodrich and Huber 2001, Goodrich and Other 1999). Natural disturbances play an important role in many sagebrush communities. Disturbances such as insects, disease, winter exposure, or snow lay are relatively minor in their effects to sagebrush spatially, but fire and extreme climate events have impacted sagebrush distribution, structure, and composition at larger scales (Kitchen and McArthur 2007, USDA Forest Service 2016). Kitchen and McArthur (2007) stated that “fire is believed to be the dominant disturbance force in natural populations.” Fire occurs in black sagebrush and all big sagebrush taxa, but fire frequency of these communities varies and correlates closely to annual precipitation, vegetation production, community structure, among other temporal and spatial parameters. Rate of return of sagebrush following fire is also variable, both among and within taxa

(Goodrich and others 2008, Goodrich 2012). Herbaceous plants and sprouting shrubs dominate vegetation cover following fire, but decrease over time as sagebrush density and canopy cover increase. Fire appears most important in sagebrush communities of mid elevations where "recurrent invasions" of pinyon, juniper, and Douglas fir have the propensity to displace shrubs (Kitchen and McArthur 2007). Historic fire regimes were sufficient to maintain sagebrush within these settings. Historically and presently, fires have burned in areas where sagebrush is commonly found (Stewart 1911, Hitchcock 1910, Graham 1937, Ogle and Dumond 1997, Huber and Goodrich 2011, Goodrich and Huber 2011). Sagebrush, with its respective plant assemblages, returned in expected return intervals following fire. Mortality of sagebrush has occurred during severe and prolonged drought (Goodrich and others 2005, Goodrich and Zobell 2012, Goodrich 2005). Initially, herbaceous understories have increased in density and productivity following shrub die-back, but surviving shrubs responded with greater foliage and seed production. Although not well documented, sagebrush die-back events during extreme climate events have likely occurred historically, followed by return of sagebrush in expected return intervals.

Mountain and Wyoming big sagebrush and black sagebrush account for about 95 percent of sagebrush found in the plan area. Hence, only these communities will be discussed in further detail in regards to the natural range of variation. Mountain big sagebrush is a montane landscape dominant that extends from the foothills of mountains and plateaus, up to aspen and coniferous forests of higher elevations. These communities grow in moderately deep loamy soils and are often associated with mollisols (Kitchen and McArthur 2007). The shrub is usually found in and above the pinyon-juniper thermal belt and occasionally interfaces with Wyoming big sagebrush in this transition zone (Goodrich and others 1999b). Mountain big sagebrush typically grows where depth and duration of snowpack exceeds 15 inches, which is considerably greater than for Wyoming big sagebrush (Sturges and Nelson 1986). Annual precipitation ranges from 12 to about 28 inches, which is correlated to comparatively high vegetation production, species richness, total ground cover, and shrub canopy cover of these communities. Vegetation production in Uinta Mountain communities is between 700 and 1,200 pounds/acre, similar to findings from northeastern Nevada (Goodrich 2016, Jensen 1989). Species richness is considerably higher in mountain big sagebrush than in Wyoming big sagebrush. An average of 47 species (four shrubs, 12 graminoids, 31 forbs) per site was determined from data analyzed from several mountain big sagebrush communities of the Uinta Mountains (Goodrich and Huber 2001). This is consistent with findings from the west flank of the Wind River Mountains, Wyoming (Tart 1996). Mountain big sagebrush often exists with other shrubs, many of these with sprouting capabilities such as snowberry, bitterbrush, and low yellowbrush (*Chrysothamnus viscidiflorus* var. *lanceolaus*). Graminoids are common and account for most of the herbaceous cover. Forbs, on the other hand, add significantly to species richness but they, "react to community dynamics more than to drive these dynamics. [Forbs] decrease or increase in reaction to shrub and graminoid changes" (Goodrich and Huber 2001). Total ground cover in mountain big sagebrush commonly equals or exceeds 85 percent under the natural range of variation. Tart (1996) found that potential ground cover for mountain big sagebrush to be greater than 80 percent in the Wind River Mountains. Not all mountain big sagebrush sites have potential for ground cover of 85 percent (Goodrich 2012, Study 64-1). In rare instances, total ground cover of less than 65 percent is inherent at marginally productive sites. For example, at two adjacent sites on Bare Top Mountain, in an area closed to livestock grazing for over last 30 years, total ground cover was measured at 62 and 85 percent respectively (Study 5-13). Mountain big sagebrush canopy cover equal to or greater than 25 percent is achievable in the absence of fire or other crown reducing disturbances of 25 years or greater, but seldom reaches 35 to 40 percent under NRV (Goodrich and Huber 2015, Studies 31-3A1,D, 32-94A, 32-100A1). Tart (1996) found canopy cover of all shrubs in mid to late seral communities between 23 and 40 percent. In some mountain big sagebrush communities under the natural range of variation, native perennial grasses dominate vegetation cover and late seral sagebrush canopy cover remains below 20 percent (Kitchen and McArthur 2007, Studies 5-13, 31-3A1,D, 32-94A, 32-100A1). Higher vegetation productivity, including comparatively high shrub canopy cover, supports higher fire frequency in mountain big sagebrush than in

Wyoming big sagebrush. Fire frequency for mountain big sagebrush ranges between 15 and 40 years (Winward 1991). This interval is supported by work from southwest Montana, eastern Idaho, Oregon, and other sites in the Intermountain West (Lesica and others 2007, Woods and others 2013, Sankey and others 2008, Miller and Rose 1999, Miller and others 1999). A 15 to 40 year interval is also supported by data from the Uinta Mountains and Tavaputs Plateau, where canopy cover of mountain big sagebrush has returned to pre-burn levels within 20 to 30 years (Goodrich and others 2008). Under the natural range of variation, rapid recovery in vegetation and total ground cover typically occur following fire, especially in sagebrush communities with pre-burn canopy cover at 30 percent or less (Goodrich and others 2008, Goodrich and Huber 2015). Herbaceous plants and sprouting shrubs, well adapted to fire, dominate cover within two to five years post-burn (Goodrich and Huber 2001). In summary, the short recovery time for mountain big sagebrush following fire is also indicative of an ecological history of greater fire frequency than other sagebrush communities.

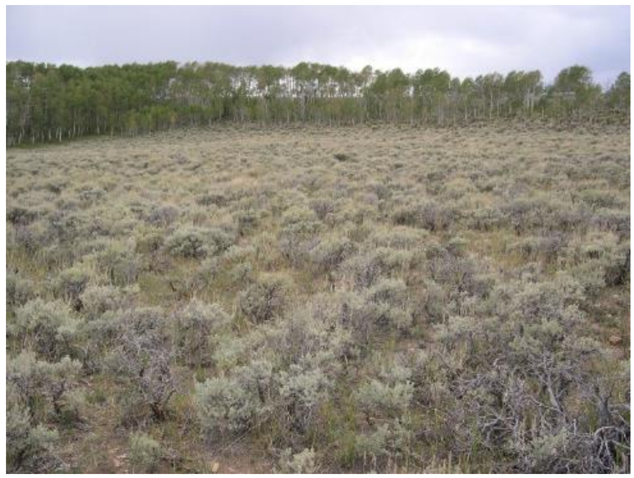
Wyoming big sagebrush is a landscape dominant in semi-arid environments throughout the Intermountain West. Within the plan area, most Wyoming big sagebrush communities are found in the Lower Green River Basin Subsection in southwest Wyoming. To a lesser extent, these communities grow in and below the pinyon-juniper thermal belt at lower elevations of the Tavaputs Plateau and Uinta Mountains (Goodrich and others 1999b). Most Wyoming big sagebrush communities in the Uinta Basin are located outside the plan area. Wyoming big sagebrush grows in deep to moderately deep soils or alluvial depositions and are usually associated with aridisols (Kitchen and McArthur 2007, Barker and McKell 1983, Winward 1983). Sagebrush communities are found where depth and duration of snow pack is less than 16 inches (Sturges and Nelson 1986) and annual precipitation ranges from 8 to 12.5 inches (Goodrich and others 1999). Where annual precipitation is at or slightly less than eight inches, Wyoming big sagebrush intermixes with salt-desert shrubs and is confined to small stands in drainage bottoms, cool exposures, or other areas where soil moisture may persist (Goodrich 2016, Kitchen and McArthur 2007). Where annual precipitation is greater than eight inches, Wyoming big sagebrush forms extensive communities across the landscape. At the upper end of its precipitation zone the shrub exists with or, in absence of fire, is displaced by pinyon and juniper. Introgression with mountain big sagebrush also occurs within this ecotone. Inherent ecological features associated with Wyoming big sagebrush indicate communities of low productivity, total ground cover, and species richness, especially in comparison to other big sagebrush taxa. In Daggett County, communities produce between 224 to 560 pounds/acre, which falls within production ranges at other Intermountain sites (Goodrich and others 1999, Peters and Bunting 1994, West and Yorks 2006, McDaniel and others 2005). Goodrich and others (1999) determined potential ground cover at Wyoming big sagebrush sites - protected from livestock grazing in northeastern Utah and southwestern Wyoming - ranged from 49 to 65 percent, with an average of 55 percent, which represents most sagebrush sites. In the upper limits of the Wyoming big sagebrush belt, where annual precipitation is about 12 inches, potential ground cover may be as high as 75 percent (Study 4-3). Goodrich and Huber (2001) found average ground cover to be 58 percent for nine sagebrush sites with moderate to no livestock grazing. Winward (1983) noted as much as 25 percent bare ground for Wyoming big sagebrush communities, even under undisturbed conditions. In southeastern Oregon, Kindschy (1994) reported 38 percent bare soil for a pristine site. Davies and Bates (2010) recorded average bare ground of 40 percent for about 50 Wyoming big sagebrush sites with limited livestock use. Species richness is significantly lower in Wyoming big sagebrush than in mountain big sagebrush, mostly because of low forb presence. An average of 18 species (i.e., four shrubs, six graminoids, seven forbs) per site was determined from data analyzed from several Wyoming big sagebrush communities of the Green River Basin (Goodrich and Huber 2001). Winward (1983) noted relatively few perennial forbs for undisturbed Wyoming big sagebrush communities. Others have found low perennial forb presence, cover, production, and frequency at any successional stage, even late seral communities, and on relic sites (Goodrich and others 1999, Bunting 1985, Davies and others 2006, West and Yorks 2006, Marquiss and Lang 1959, Rowlands and Brian 2001).

Several shrubs grow with Wyoming big sagebrush. These include snakeweed (*Gutierrezia sarothrae*), pricklypear (*Opuntia*), shadescale (*Atriplex confertifolia*), winterfat (*Krascheninnikovia lanata*), and sticky yellowbrush (*Chrysothamnus viscidiflorus* var. *viscidiflorus*). Graminoid species are few but account for most of the herbaceous cover, with needle-and-threadgrass (*Stipa comata*), Sandberg bluegrass (*Poa secunda*), and thickspike wheatgrass (*Elymus lanceolatus*) being most common. In almost all communities, few to very few forbs are present. Usually there are less than 10 species but rarely surpassing 12 (Goodrich and others 1999, Goodrich and Zobell 2016). Potential canopy cover of Wyoming big sagebrush is inherently lower than other big sagebrush taxa. Goodrich and others (1999) found sagebrush canopy cover ranging from 0.3 to 22 percent, with an 11 percent average at nine sites protected from domestic livestock grazing. In this study, the highest canopy cover score occurred within an exclosure where all ungulates were excluded. Outside the exclosure, where pronghorn antelope and mule deer browsing occurred, sagebrush canopy cover was 11 percent. Wild ungulates can strongly influence Wyoming big sagebrush canopy cover through heavy browsing. In Oregon, Utah, and Wyoming, others found similar canopy cover ranges and averages comparable to those reported by Goodrich and others (Davies and others 2006, Bates and others 2009, Anderson and Holte 1981, Perryman and others 2002). Low vegetation productivity lends to a low fire frequency in Wyoming big sagebrush. A fire interval of 50 to 110 years is common for many pre-settlement communities (Winward 1991; Whisenant 1990; Mensing and others 2006). Another source suggests that the interval may be as high as 200 to 350 years (USDA Forest Service 2016a). Recovery of sagebrush following fire is considerably slower for Wyoming big sagebrush than for mountain big sagebrush, even at sites with higher annual precipitation (Britton and Clark 1984, Bunting 1984, Boyd and Svejcar 2011, Lesica and others 2007, West 2000, Winward 1991). Bare ground is higher and herbaceous vegetation cover is much lower three to five years post-fire.

Black sagebrush has a relatively wide elevational amplitude within the plan area, ranging from 6,100 to 9,100 feet (Studies 79-14A, 67-70B). Its elevational range is "nearly equal to the combined range of the three sub-species of big sagebrush (Kitchen and McArthur 2007). Within the plan area, black sagebrush is found on stream pediments on the Green River landtype association and often on soils derived from limestone or other basic substrates elsewhere in the plan area. In general, soils that support black sagebrush are typically shallow, rocky, or gravelly, often poorly developed with overlying bedrock (Fryer 2009). In northeastern Nevada, Jensen (1990) found these soils had the lowest water holding capacity, organic matter, nitrogen content, and thinnest mollic epipedons. At low elevations of the Lower Green River Basin, black sagebrush is an aspect dominant where it forms both ecotones with the salt-desert shrub and Wyoming big sagebrush communities and as well as nearly pure stands where more distinct topographical boundaries exist (Fryer 2009, Study 79-14A). Because of low annual precipitation (8 inches or less), these communities consist of scattered shrubs, high bare ground, and a "sparse herbaceous understory" (Kitchen and McArthur 2007, Fryer 2009, Goodrich 2001). Nevertheless, in this environment, black sagebrush appears to be very competitive and long persisting with little to no displacement by other shrubs (Goodrich 2001). Canopy cover of black sagebrush rarely exceeds 15 percent, even without browsing pressure; but canopy cover can be greatly reduced where wild ungulate use is prevalent (Goodrich 2001). Potential ground cover for low elevation black sagebrush ranges between 55 to 60 percent; but surface rock makes up a high percent of ground cover and herbaceous cover is low. (Studies 79-14B, 79-17). Fire frequency of these communities is low (i.e., 100 to 200 years) (Fryer 2009). An average of 18 species (six shrubs, four grasses, nine forbs) per site was determined from five black sagebrush communities of the Green River Basin (Studies 75-33, 78-27, 78-28, 79-14B, 79-17). Common plant associates include desert shrubs such as shadscale, snakeweed, and sticky low yellowbrush, and grasses such as needle-and-threadgrass, galleta (*Hilaria jamesii*), squirreltail (*Elymus elymoides*), sandberg bluegrass, and bluebunch wheatgrass (*Elymus spicatus*). Kitchen and McArthur (2007) noted that "the diversity and abundance of associated shrubs and understory species increases with elevation," which reflects findings within the plan area. At middle elevations, black sagebrush communities often integrate with pinyon and juniper. At many sites, the shrub is seral to pinyon and juniper and is displaced by these trees in the absence of fire. However, some communities have persisted in pinyon and juniper without stand-replacing fire (Goodrich 2001, Fryer 2009). Thompson (1999) identified pinyon-juniper and black sagebrush community types for central Utah, which mirrors similar communities within the plan area. Overall, most black sagebrush communities in the pinyon-juniper belt require periodic fire to maintain shrubs (Goodrich 2001). At higher elevations, black sagebrush communities of considerable size are found on moderate to steep slopes of both southerly and northerly aspects and atop wind swept ridges and plateaus. Because of higher annual precipitation, potential ground cover for montane black sagebrush communities ranges from 60 to 80 percent (Studies 67-25A, 68-1). Rock cover is similar for low elevation black sagebrush communities, but vegetation and litter cover is higher for montane black sagebrush communities. Canopy cover of black sagebrush at sites with high potential can exceed 25 percent (Goodrich 2001, Studies 68-1, 68-69). Black sagebrush can withstand moderate wildlife browsing pressure, but can decline with long-term overuse (Fryer 2009). Species richness for high elevation communities is also higher than low elevation communities. Based on 15 sites, an average of 29 plants (i.e., five shrubs, six grasses, and 18 forbs) were documented. Common plant associates include bluebunch wheatgrass, junegrass (Koeleria macrantha), needle-and-threadgrass, muttongrass (*Poa fendleriana*), sandberg bluegrass, and thickspike wheatgrass. Fryer (2009) found that "montane black sagebrush communities…. generally those above the pinyon-juniper zone….. are usually stable" because of topography and edaphic features. This finding correlates with most communities within the plan area, but some black sagebrush communities are susceptible to displacement by Douglas fir (Goodrich 2001). Due to higher production, canopy cover, and shrub density, montane black sagebrush communities have a higher fire frequency (30 years or more) than low-elevation communities (Fryer 2009). Herbaceous cover increases following fire, but decreases over time as shrubs increase. Goodrich (2001), found that black sagebrush returned to pre-treatment status within 20 years, while others determined a return interval range from 15 to 60 years to regain pre-fire cover (Fryer 2009).

Current Status and Trends

Mountain big sagebrush of the Parks Plateau, Anthro Plateau, Avintaquin Canyon, and Strawberry Highlands landtype associations are located mostly above 8,000 feet where higher annual precipitation and lower summer temperatures occur. Mountain big sagebrush communities of considerable size occur on these landtype associations. Since the early 1900s, almost all mountain big sagebrush communities within the plan area have been impacted by anthropogenic uses and management. Livestock grazing has occurred in various forms and intensities for more than 100 years. Since the 1940s, thousands of acres of mountain big sagebrush have been plowed and seeded into introduced grasses, sprayed with herbicide, and treated with prescribed fire (Goodrich and others 2005, Goodrich and Huber 2015). Seeding and herbicide treatments were implemented, mostly during the 1950s and 1960s, to reduce dense sagebrush canopy cover and enhance livestock forage. In plow and seed treatments, seeded grasses successfully established, but canopy cover of sagebrush returned to pre-treatment levels within expected return intervals regardless of the production of the herbaceous understory (Goodrich and others 2005, Goodrich and Huber 2015). Similar results were documented in herbicide spray treatments, but native herbaceous species increased where no seeding treatments occurred. Few seeding and herbicide treatments, especially on a large scale, have occurred over the last 30 years, however, thousands of acres of mountain big sagebrush were treated with prescribed fire or conifer lop-and-scatter treatments, especially since the year 1995 (Huber and others 2010). These communities are currently in satisfactory condition and have demonstrated resilience to disturbances both past and present. Herbaceous understories have responded rapidly and vigorously following disturbance (Goodrich and Huber 2001). Native and seeded perennial forbs and grasses that have moderate to high resource value, and dominate herbaceous vegetation cover and species richness values, have been maintained following disturbance events (Goodrich and Huber 2001, Goodrich and others 2008). Even some sagebrush communities that are currently dominated by seeded grasses demonstrate relatively high herbaceous species richness (Studies 68-2E,G, 68-18B, 68-73D3,E). Mountain big sagebrush and its shrub associates have and are returning to pre-disturbance canopy cover within 15 to 30 years following disturbances, which falls within expected return intervals (Goodrich and Huber 2001, Goodrich and others 2008, Goodrich and others 2005). As shrub cover increases, herbaceous understories decrease in vegetation cover and production (Goodrich and Huber 2001). Following disturbance, ground cover has returned to 80 percent or greater in five years at most sites (Goodrich and others 2008). High percent ground cover in mountain big sagebrush has been maintained long term concurrent with present drivers and stressors. These communities currently show high resilience to annual invasive plants, following episodes of disturbance such as fire or drought. Little to no cheatgrass or other invasive annuals are documented within early, mid, or late seral mountain big sagebrush communities of these landtype associations.



Photos of study 68-9F. Anthro Mountain, 1959 and 2004. Plow and seeding treatment of mountain big sagebrush community.

Many communities of mountain big sagebrush of the South Face, Dry Moraine, Glacial Canyon, Stream Pediment, and Structural Grain landtype associations are currently in satisfactory condition in regards to plant species composition, species richness, shrub cover, and total ground cover. However, these communities are potentially at risk due to their moderate to high susceptibility to annual invasive plants. Annual invasive plants degrade sagebrush communities by changing plant composition and structure, lowering species richness, and narrowing fire frequency. Long-term monitoring shows that cheatgrass is present and increasing in mountain big sagebrush communities with native herbaceous understories, especially following fire and severe drought (Studies 6-46A-D, 38-7A,E-G, 38-12B, 39-8A-D, 39-9, 39-16, 41-1, 41-2A, 41-3E3-F, 41-7A,C-G,N, 42-17A4A, 42-17F, 43-16A-B, 51-17). Although cheatgrass was present prior to the year 2002, most of its spread has occurred since that time, which indicates low resilience (USDA Forest Service 2015a, 2015b, Goodrich and Huber 2015). Cheatgrass has increased considerably over the last few years in native shrub communities on Mosby Mountain, Dry Fork Mountain, Lake Fork Canyon, and Yellowstone Canyon. The most affected mountain big sagebrush communities are located on steep, southerly aspects of the South Face landtype association where cheatgrass often dominates herbaceous cover and is most abundant where fire, drought, or both has recently occurred (Goodrich and Huber 2015, Studies 37-21A-B,D, 37-24A-E, 44-16C, 44-23A-C, 44-24G, 44-26A-B). In contrast, communities where seeded non-native grasses dominate herbaceous cover, cheatgrass is absent or has minor presence with no indication of spread or increase (Studies 38-16, 41-8A-F,I, 41-10A--E, 42-1A,C-G,I-K, , 42-13A-C, 43-1A-B, 43-7B-C). These communities typically have satisfactory plant composition, species richness, and total ground cover. Historical seeding treatments of these shrublands, with non-native grasses, have demonstrated high resilience to invasive annuals.

Photo of study 42-8C. Mosby Mountain. 2015. Mountain big sagebrush with cheatgrass filling shrub interspaces.

Montane black sagebrush, mostly found within the Anthro Plateau and North Flank landtype associations, benefit from a cooler and wetter environment found at higher elevations. These communities have higher herbaceous cover, species richness, and total ground cover than black sagebrush communities of lower elevation. Black sagebrush has been less impacted by anthropogenic activities than mountain big sagebrush due to lower productivity; soils that are more shallow, rocky, and poorly developed; and/or steep gradients. Because of soil conditions and inherent low vegetation productivity, few acres of black sagebrush were plowed-and-seeded, sprayed with herbicide, or treated with prescribed fire to enhance livestock forage production prior to the year 1980. For the same reason, black sagebrush has not been targeted for treatments during the last 30 years. Small treatments that did occur prior to 1980 showed that black sagebrush returned to pre-treatment levels within 25 years. (Goodrich 2001, Studies 67-25, 68-1). During the last 30 years, a few acres of black sagebrush were burned inadvertently during prescribed fire treatments (Studies 68-72A, 68-11F, 68-96E). At two burned sites, black sagebrush canopy cover was two percent and one percent, 17 and 8 years post-fire respectively, which is a relatively long return interval (Study 68-11F, 68-72A, 68-96E). Overall, most montane black sagebrush communities are in late seral stages (Studies 67-25A-B, 67-52, 67-70B, 67-86D, 68-1A, 68-21A, 68-62E, 68-69A,C-D, 68-72D, 70G-2). These communities are in satisfactory condition in regards to plant species composition, ground cover, shrub canopy cover, and absence of annual invasive plants. Native plants of moderate to high resource value dominate herbaceous vegetation cover in montane black sagebrush communities and total ground cover averages 75 percent, which is within NRV (Studies 67-52, 67-53, 67-25, 68-1, 68-1B, 68-72D, 4-4, 4-29B). At these sites, black sagebrush canopy cover ranges from 3 to 30 percent, with an average of 20 percent. Shrub die-back, following drought, has also been documented (Study 68-21, 69-54A). In addition, sagebrush canopy cover averaged higher (20 percent) in communities located above critical winter range for wild ungulates than in communities located within critical winter range (13.5 percent). This higher average likely indicates shrub decrease from wild ungulate browsing (Studies 5-63D, 4-2J, 4-26B, 69-48B, 69-54A, 68-77A). Montane black sagebrush communities of the Anthro Plateau and North Flank landtype associations currently show resilience to annual invasive plants in late seral communities and in burned or drought-impacted communities (Studies 68-21A, 68-72A, 68-96E). Cheatgrass and other annual invasive plants are either absent or have minor presence, with no indication of spread or increase.

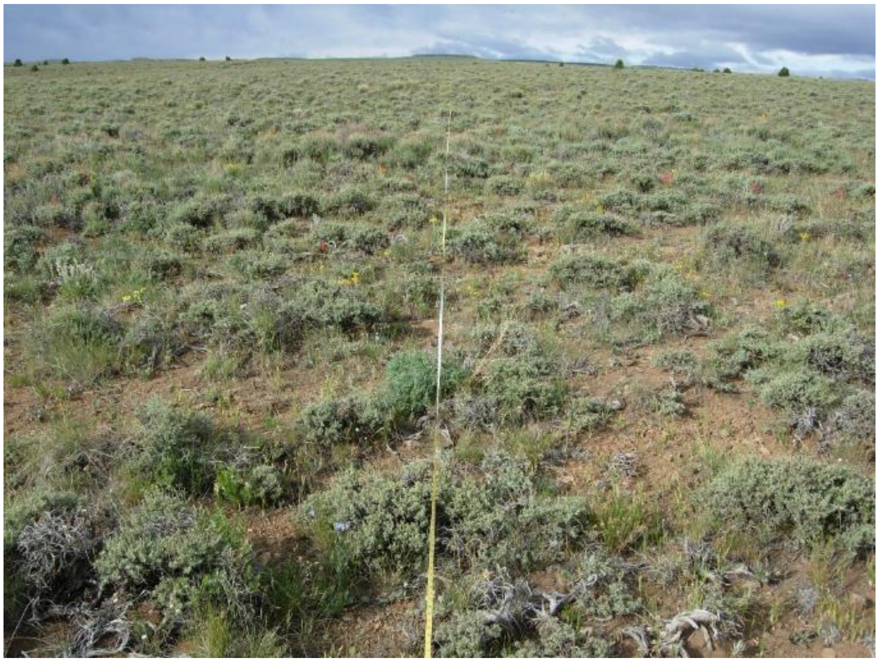


Photo of study 68-1B. Anthro Mountain. 2007. Montane black sagebrush community within the natural range of variation.

Black sagebrush communities of the South Face landtype association are located within or just above the pinyon-juniper belt. These communities currently have satisfactory plant compositions, species richness, ground cover, and shrub cover (Studies 31-89A, 32-64, 45-6T, 45-8D-E,L,N-Q,S, 45-19C-D,H). Based on eight sites, black sagebrush canopy cover averaged 20.5 percent, with total ground cover averaging 77 percent. An average of 29 plant species (i.e., four shrubs, six graminoids, and 19 forbs) were present at each site, which is in the natural range of variation. But two of the eight sites have cheatgrass present, with monitoring indicating recent spread (Studies 45-6U, 45-8). Black sagebrush communities of the South Face landtype association have a moderate to high susceptibility to invasive annuals. Long-term monitoring shows that, within the last 10 years, cheatgrass has recently established here and spread into black sagebrush communities of the South Face landtype association. This indicates low to moderate resilience of this shrub type (Studies 32-64, 45-6U, 45-8, 45-8R,U-V, 45-17D, 45-19I). Cheatgrass spread appears to accelerate during and shortly following severe drought. Similar to mountain big sagebrush, cheatgrass can change plant composition, reduce species richness, and increase fire frequency. This places these communities at risk of loss of resilience, capability, and function. No black sagebrush communities have introduced seeded grasses within them.

Photo of study 45-17D. Taylor Mountain. 2015. View of black sagebrush on South Face landtype association with cheatgrass established between shrub interspaces.

At low elevations, black sagebrush of the Green River landtype association has long persisted on favorable aspects and soil types (Goodrich 2005). These communities are currently in satisfactory condition in regards to plant composition, species richness, total ground cover, and shrub cover. Native grasses that have moderate to high resource value dominate herbaceous vegetation cover, with sagebrush canopy cover averaging 11 percent. Browsing by pronghorn antelope may influence sagebrush cover at some sites. Total ground cover averaged 62.5 percent, with rock cover making up 35 percent of the total (Studies 75-33, 79-14B, 79-17, 78-28). Cheatgrass is present within desert shrub communities of the Green River Basin, including black sagebrush (Study 79-17). Black sagebrush communities have moderate to high susceptibility to cheatgrass invasion. This places these communities at risk of loss of resilience, capability, and function. Cheatgrass spread appears to increase during and shortly following severe drought. Black sagebrush communities have low to moderate resilience to annual invasive plants.

Photo of study 79-17A. Green River Basin. 1999. Black sagebrush community on favorable aspect in cold desert environment. The community is currently in satisfactory condition.

Wyoming big sagebrush communities associated with the Antelope Flat, Green River, North Flank, and Structural Grain landtype associations have shown resilience to disturbance during the last 30 years. For the most part, these communities are currently in satisfactory condition concerning plant composition, species richness, total ground cover, and shrub cover. Native grasses that have moderate to high resource value dominate herbaceous vegetation cover. These grasses include needle-and-threadgrass, Indian ricegrass (*Stipa hymenoides*), western wheatgrass (*Elymus smithii*), Sandberg bluegrass, squirreltail, and thickspike wheatgrass. Site capability influences plant composition, total ground cover, and perhaps shrub cover. However, in most instances, the abundance of native grasses corresponds most closely with sagebrush canopy cover. A strong inverse relationship between Wyoming big sagebrush canopy cover, perennial grass cover, and total ground cover has been documented. Herbaceous and total ground cover is most abundant when sagebrush cover is less than 12 percent, but trends downward as sagebrush canopy cover increases (Goodrich and Zobell 2012, Goodrich 2005). Minimal percentages of herbaceous and total ground cover occur when sagebrush canopy cover equals or exceeds 20 percent. At nine sites where sagebrush canopy cover was 12 percent or greater, graminoid and total ground cover averaged 13 and 55 percent respectively.

Photo of study 81-3. Green River Basin. 2010. Wyoming big sagebrush community with 17 percent canopy cover and a sparse herbaceous understory.

In contrast, at 10 sites where sagebrush canopy cover was 10 percent or less, graminoid and total ground cover averaged considerably higher, at 48 and 80 percent respectively (Goodrich and Zobell 2012). Although most Wyoming big sagebrush communities are currently in late seral stages with relatively high canopy cover, some have been affected by disturbance. For instance, pronghorn antelope browsing strongly influences canopy cover of many Wyoming big sagebrush communities of the Antelope Flat landtype association. In Lucerne Valley, browsing pressure has reduced Wyoming big sagebrush canopy cover to well below 10 percent in many communities (Goodrich and Zobell 2012). Native grass and total ground cover is substantially higher at these sites than in late seral communities minimally influenced by ungulate browsing. Drought has also affected Wyoming big sagebrush. Following the drought of 2002, considerable sagebrush/perennial grass die-back occurred in many communities of the Antelope Flat landtype association. At the same time, minor to no die-back occurred in communities of the North Flank, Green River, or Structural Grain landtype associations (Drought White paper XX, Goodrich and Zobell 2012, Goodrich 2005, Studies 4-42B, 5-1A, 5-2A, 5-3B2,D, 5-27, 5-52A, 6-46A-C, 72-33). Greater die-off occurred in communities with low annual precipitation and communities located near or intergrading with desert shrubs (Goodrich 2005). At some sites, Wyoming big sagebrush and needle-and-threadgrass mortality was as high as 90 percent. By 2005, rapid recovery of drought-impacted communities was documented. At most sites, both sagebrush and pre-drought dominant perennial grasses increased in cover and in vegetation and seed production. A few sites showed increases in winterfat, western wheatgrass, and other pre-drought nondominant plants, where marginal recovery in sagebrush/dominant perennial grasses occurred (Goodrich and Zobell 2012, Goodrich 2005). As of 2005, rapid recovery of perennial grasses and Wyoming big sagebrush following drought indicated resilience, capability, and proper function of these communities. A subsequent die-back of perennial grasses and sagebrush occurred following the 2012 drought. Since 2012, perennial grasses showed recovery at some sites, similar to the 2002 drought, but additional monitoring is needed to better determine future trend (Goodrich 2005). Unlike the 2002 drought, cheatgrass and other annual invasive plants are more common in Wyoming big sagebrush communities with indications of potential spread into all communities. This occurs regardless of seral stage, annual precipitation rates, or other ecological features (Studies 5-27E, 5-27J, 72-8, 72-8B, 72-32, 78-13C, 78-26). Cheatgrass spread appears to increase during and shortly following severe drought or fire. Wyoming big sagebrush communities currently have moderate to high susceptibility to cheatgrass invasion. This susceptibility increases these communities' risk to loss of resilience, capability, and function.

Wyoming big sagebrush communities of the Anthro Plateau landtype association are located within the pinyon-juniper belt, where annual precipitation exceeds 12 inches. These sagebrush communities typically grow in shallow drainages or swales, and on alluvial fans of canyon bottoms where periodic fire maintains shrubs. Sagebrush communities are also located within openings that were chained to remove sagebrush in the mid-1970s. Over the last 30 years, Wyoming big sagebrush communities demonstrated resilience following fire and drought. However, since 2002, annual invasive plants have established and have spread into many of these communities. Most sagebrush communities within the shallow drainages or swales have been burned with prescribed fire from the years 1970 to 1999. In Wyoming, big sagebrush communities that have not burned, average canopy cover was 18 percent, with graminoid and total ground cover averaging 18 and 65 percent respectively (Studies 69-31, 69-31B, 69-95C, 70-10A). Herbaceous species richness averages about 10 plants (five graminoids and five forbs) per site. Thickspike wheatgrass, needle-and-threadgrass, Indian ricegrass, muttongrass, Sandberg bluegrass, and blue gramma (*Bouteloua gracilis*) are the most common grasses, and winterfat is often present in a minor capacity. In burned Wyoming big sagebrush communities, graminoid and total ground cover is higher than in unburned communities, averaging 30 and 78 percent for five sites respectively (Studies 69-21B, 69-58A, 70-3D, 70-8A, 70-9B). Winterfat often increases in cover as well, following fire. Recovery of Wyoming big sagebrush to pre-burned conditions at burned sites is typically greater than 50 years. For example, two sites that were burned in about 1970 have sagebrush canopy cover of two and seven percent, and two sites burned in the 1980s have sagebrush canopy cover of zero and two percent, which indicates a slow return (Studies 69-21B, 69-72A, 70-9B, 69-58A). Drought has impacted plant composition and of Wyoming big sagebrush. Severe drought occurred in 2002, and from 2012 to 2014. Minimal die-back of Wyoming big sagebrush occurred during these droughts; however, considerable die-back of perennial native grasses occurred during the drought of 2002, with additional die-back occurring from 2012 to 2014. After 2002, winterfat increased in density and cover, replaced dead perennial grasses, and became an dominant understory species within two years following drought (Studies 69-21A, 69-21B, 69-21D, 69-21F, 69-31, 69-58A, 69-58B, 69-58D). Although cheatgrass was present in 2002, it did not alter the winterfat community. However, during and following the drought of 2012-2014, cheatgrass became more common and widespread in Wyoming big sagebrush and in some cases, cheatgrass has displaced both perennial grasses and winterfat, especially in burned communities (Studies 69-1, 69-1E, 69-1F, 69-2B, 69-3, 69-3B, 69-21A, 69-21F, 69-31, 69-56, 69-58A, 69-58B, 69-72B, 69-94, 69-94D, 69-94E, 69-100B, 70-3D, 70-8A, 70-8B, 70-9B, 70-10A). Similar to other sagebrush communities, the spread of cheatgrass into Wyoming big sagebrush has increased during the last drought. Wyoming big sagebrush communities of the Anthro Plateau landtype association currently have moderate to high susceptibility to cheatgrass invasion, which places them at risk of loss of resilience, capability, and function.


Photo of study 69-94> Gilsonite Ridge. 2010 and 2015. Burned Wyoming big sagebrush community. 2015 photo taken 22-years post-fire and after three consecutive years of drought. Cheatgrass has displaced many of the perennial grasses.

Management Activities and Uses

Sagebrush communities provide a range of anthropogenic and ecological services. A primary management activity is livestock grazing, which is considered the most extensive and enduring anthropogenic use of these communities. Almost all sagebrush communities are classified as “capable” for livestock grazing, providing both summer and winter forage for cattle and sheep. These sagebrush communities are productive, producing from 200 to more than 1,200 pounds of forage per acre. To increase forage production for livestock use, many acres of mountain big sagebrush were plowed and seeded with introduced perennial grasses between the 1940s and the 1960s. Most sagebrush communities are readily accessible to ungulates and are usually found on gentle to intermediate slopes. Their understories consist of shrubs, palatable grasses, and forbs, which makes these communities attractive rangeland for both the cattle and sheep. Historically, livestock have grazed all or nearly all sagebrush communities within the plan area, with continual use spanning more than 100 years. Most sagebrush communities are allocated as summer range for cattle, but a few allotments provide winter forage and browse for both cattle and sheep. Summer sheep grazing has decreased over the last 30 years, but winter use by domestic sheep in Wyoming big and black sagebrush communities of the Green River landtype association has remained constant. Some browsing of sagebrush and other shrubs by domestic sheep occurs on winter ranges. Future trend of sheep grazing is expected to either remain stable or continue downward. Cattle grazing has increased in mountain big sagebrush by area, but intensity has either remained stable or has decreased on some allotments over the last 30 years. During the next plan period, the high value of sagebrush, as a producer of forage, is expected to remain constant.

In relative terms, sagebrush has limited recreation value. In the Uinta Mountains, forest visitors typically drive through these communities to access forests, lakes, and streams of higher elevations. The highest recreation use is big game hunting, which occurs in the autumn months. Impacts from this use has increased over the last plan period. Many unauthorized roads and trails in sagebrush are either created, maintained, or expanded by all-terrain vehicles and pickups used by hunters. Camping is most popular during the big game hunts. Other minor recreation opportunities include hiking, wildlife viewing, horseback riding, snowmobiling, and cross-country skiing. Within the Green River landtype association, sagebrush communities receive more year round use by those who visit the Flaming Gorge Recreation Area. These consist mostly of fishermen, campers, and boaters. Many unauthorized roads are created and maintained near the shores of the reservoir. Overall, sagebrush communities nearest to main roads receive the highest use. Most new noxious weed infestations in sagebrush communities are located along authorized and unauthorized roads; a direct result of vehicle and recreation use. Similarly, cheatgrass and other invasive annual plants are typically found along roadways and eventually spread into nearby sagebrush communities. Increased recreation use in sagebrush is expected during the next plan period. With this increase, additional unauthorized roads and trails and noxious weed infestations are also predicted.

Oil and gas exploration and development occurs in sagebrush communities on the Anthro Plateau landtype association. Many acres of mountain big, Wyoming big, and black sagebrush have been impacted by this management activity. Numerous well pads, service roads, and pipeline corridors have been constructed within sagebrush communities over the last 15 years, with additional infrastructure planned. These sagebrush communities are common and some loss in acreage is acceptable. However, the greatest impact of oil and gas exploration is the introduction, establishment, and spread of noxious weeds and invasive annuals into the area. Almost all new noxious weed infestations occur along roads. Increased vehicle use, mostly from the oil and gas industry, has accelerated noxious weed infestations. Additionally, cheatgrass and halogeton (*Halogeton glomerata*) are relatively new to the Anthro Mountain area. Cheatgrass currently grows along road sides, with minimal to no spread into sagebrush communities, and halogeton has been found on well pads (Study 68-27K, 69-9F). Most sagebrush communities with native understories are moderately to highly susceptible to cheatgrass invasion. Oil and gas exploration and development is expected to increase during the next plan period.

Sagebrush communities provide quality habitat and forage for a number of wildlife species, some of which are sagebrush obligates. Big game species such as mule deer, pronghorn antelope, and elk are known to browse sagebrush, associated shrubs, forage, or all three on the herbaceous understory. Mountain big sagebrush communities are preferred by wild ungulates during the snow-free months of the year. On the other hand, Wyoming big and black sagebrush communities not covered by snow during winter months become important winter habitat for wild ungulates. Browsing intensity of these shrubs is subject to shrub preference, winter conditions, availability of other forage, and ungulate population densities. Pronghorn antelope use Wyoming big sagebrush year round in the Green River landtype association. Heavy browsing in some communities has reduced Wyoming big sagebrush canopy cover to less than five percent (Goodrich and Zobell 2012). Many black sagebrush and Wyoming big sagebrush communities in the Uinta Mountains are important winter habitat for elk and mule deer (Studies 3-16, 4-4, 6-18, 32-74). At one study, mule deer and elk have kept canopy cover of black sagebrush well below potential, which has favored grass production and herbaceous species richness (Study 32-74). Elk numbers have increased significantly during the last 30 years. An upward trend in the elk population is predicted for the next plan period. This trend may impact sagebrush communities, especially those that provide winter habitat. Although populations fluctuate, mule deer and pronghorn antelope numbers have remained relatively constant over the last couple of decades and their populations are expected to follow a similar trend during the next plan period.

Greater sage-grouse is a sagebrush obligate species and requires sagebrush to breed, nest, raise broods, and winter. Greater sage-grouse use the major sagebrush communities within the plan area at high, as well as low elevations. Quality greater sage-grouse habitat is defined in terms of plant composition, species richness, shrub and herbaceous cover, and sagebrush seed production. Greater sage-grouse populations have steadily declined during the last 30 years, especially in Wyoming big sagebrush communities. During the next plan period, populations are expected to level out, but this is dependent upon the conservation of quality sagebrush habitat that is resilient to disturbance. A number of other mammals and birds use sagebrush to meet life cycle needs.

Influences of Drivers and Stressors

Fire is an important ecological driver in sagebrush communities. As stated earlier, “fire is believed to be the dominant disturbance force in natural populations” (Kitchen and McArthur 2007). Many communities, especially those within and above the pinyon-juniper belt are susceptible to conifer encroachment and displacement. Under natural conditions, fire frequency correlates closely to annual precipitation, vegetation production, community structure, among other temporal and spatial parameters. With these conditions, fire occurred often enough to maintain sagebrush within and above the pinyon-juniper belt. Additionally, sagebrush communities with higher fire frequency, such as mountain big sagebrush, demonstrated faster and more vigorous vegetation responses following fire and sagebrush return intervals are considerably shorter than communities with low fire frequency. Ironically, fire frequency has decreased in mountain big sagebrush communities but has increased or is conditioned to increase in communities with historically low fire frequencies such as Wyoming big sagebrush. Fire suppression policies, intensive grazing management, and other anthropogenic activities have decreased fire frequency in mountain big sagebrush over the last 100 years. Notable trends include most mountain big sagebrush communities in late seral stages with canopy cover exceeding 20 percent and widespread encroachment and displacement of conifers in many of these communities. Some sagebrush communities have transitioned into conifer forest or woodland types with prolonged absence of fire. Wildfire from natural ignitions continues to occur on montane landscapes, but not at the rate documented historically (Stewart 1911, Graham 1937, Hitchcock 1910, Heyerdahl and others 2011). To compensate for lowered fire frequency, management strategies were developed to maintain montane sagebrush communities. During the last 30 years, especially since 1995, prescribed fire and lop-and-scatter treatments were implemented to imitate natural fire frequencies or to curtail conifer encroachment within sagebrush communities (Huber and others 2010, Bistryski 1996, Bistryski 2000, Kirkaldie 2009, Kirkaldie 2009b, Kirkaldie 2009c). These prescriptions, including a few wildfire events, has curtailed conifer encroachment within a few thousand acres of sagebrush. Additional treatments have been planned, analyzed, and are awaiting implementation. During the next plan period, natural fire frequency of montane sagebrush communities (i.e., Parks Plateau, Anthro Plateau, Avintaquin Canyon, Strawberry Highlands landtype associations) is expected to remain outside the natural range of variation, but management prescriptions would be implemented to maintain these communities.

On the South Face, Glacial Canyon, and Stream pediment landtype associations, plant compositions within burned mountain big sagebrush communities have and are being displaced by invasive annuals, particularly cheatgrass. As stated above, under the natural range of variation, native herbaceous understories of mountain big sagebrush respond quickly and vigorously following fire. These processes were documented following wild and prescribed fire prior to about the year 2002 (Goodrich and others 2008, Goodrich and Huber 2007, Goodrich and Huber 2016). Since that time, cheatgrass has not only established and spread within recent burns, but has displaced standing native vegetation in burns that occurred prior to 2002. On the other hand, cheatgrass has not established in burns where introduced seeded grasses dominate the understory (Studies 38-16, 41-8A-F,I, 41-10A--E, 42-1A,C-G,I-K, , 42-13A-C, 43-1A-B, 43-7B-C). The presence and dominance of cheatgrass in mountain big sagebrush communities will likely increase fire frequency of these communities. During the next plan period, cheatgrass is expected to increase in sagebrush communities with native herbaceous understories of the South Face, Glacial Canyon, and Stream pediment landtype associations, especially following fire.

Under the natural range of variation, Wyoming big and black sagebrush communities of low elevation have low fire frequency and sagebrush return intervals. Since fire is rare in these communities, native herbaceous plants respond to fire less efficiently than those in montane sagebrush communities, which makes sagebrush communities that have infrequent fire frequencies more susceptible to change with the introduction of annual invasive plants. Since 2002, cheatgrass has established and is spreading within many Wyoming big and black sagebrush communities. Cheatgrass is very capable of altering fire regimes of low elevation sagebrush. This annual plant does this by growing within and filling shrub interspaces, which creates continuous fuel between shrubs that would otherwise be separated by bare or rocky soil, pediments, or sporadic herbaceous cover. Cheatgrass is a fine fuel and is highly flammable. As a result, sagebrush communities become more prone to fire. When infested sites eventually burn, cheatgrass out competes and displaces native vegetation, dominates cover, and is capable of creating annual monocultures. Under these conditions, fire frequency increases, often at intervals that preclude shrubs from reestablishing. Although cheatgrass is present in some Wyoming big and black sagebrush communities of lower elevation, these communities are currently intact due to the absence of fire. Further spread of cheatgrass is predicted during the next plan period, which will make more communities prone to fire. In addition, as cheatgrass becomes more widespread, the incidence of fire and the size of burns are predicted to increase.

Drought is a natural disturbance that occurs regularly in arid regions of North America. Under the natural range of variation, response mechanisms to drought have successfully maintained sagebrush on western landscapes. Most drought events minimally impact shrub cover or change plant compositions; however, high mortality of sagebrush and herbaceous plants have occurred during the most severe and/or prolonged droughts. In most cases, herbaceous understories have increased in density, cover, and productivity following shrub die-back and surviving shrubs responded with greater foliage and seed production. Where die-back of herbaceous plants has occurred, two responses have been documented. In most cases, herbaceous plants reestablish or recover from die-back within two to five years post-drought. At some sites, surviving understory species have replaced, at least temporarily, those that experienced die-back. During the next plan period, sagebrush and herbaceous plants are expected to respond to drought events similar to the natural range of variation if annual invasive plants are not present. Where present, annual invasive plants have and are beginning to alter sagebrush community’s ability to respond to drought. Cheatgrass spread has accelerated and native perennial understories have been displaced by the annual grass during severe drought events. During the next plan period, cheatgrass is expected to increase during drought events in sagebrush communities of the Green River, South Face, Glacial Canyon, Stream Pediment, Dry Moraine, Antelope Flat, and Structural Grain landtype associations.

Livestock grazing is a stressor of sagebrush communities. This long-term and enduring practice has impacted plant compositions in most communities. Historically, heavy livestock grazing depleted and/or reduced vigor of native herbaceous understories, decreased ground cover, and increased sagebrush canopy cover. To maintain or maximize forage production, mechanical, prescribed fire, and herbicide treatments were implemented in the 1940s through 1980s to remove sagebrush. This included plow-and-seeding treatments on thousands of acres of mostly mountain big sagebrush, the most productive sagebrush community. Smooth brome, crested wheatgrass, intermediate wheatgrass, and other seeded non-native grasses have been successful in dominating herbaceous cover, increasing forage production, depleting native herbaceous understories, and lowering species richness. Since 1980, few sagebrush treatments, seedings to enhance livestock forage, or both were implemented. Sagebrush treatments that did occur were mostly prescribed fire. Instead, adjustments in grazing management to lower grazing intensity and improve range conditions of sagebrush communities has become more common. On livestock allotments where these adjustments were made, plant compositions improved and total ground cover increased in sagebrush communities with native herbaceous understories. In some instances, higher species richness was documented in sagebrush communities with introduced grass understories and where light to moderate grazing intensities are managed for (Studies 68-2E,G, 68-18B, 68-73D3,E). In addition, some research has shown that targeted livestock grazing can enhance habitat for wildlife, especially elk habitat (Anderson and Scherzinger 1975, Frisina 1992, Vavra 2005, Crane and others 2016). In sagebrush communities currently with light to moderate grazing intensities, plant composition and total ground cover are expected to be maintained relative to the natural range of variation and livestock grazing is predicted to remain constant. In sagebrush communities with heavy grazing intensities, grazing management adjustments targeted to improve range conditions is predicted to occur. In summary, native plant compositions of sagebrush communities can remain in the natural range of variation with appropriate livestock management.

Sagebrush communities provide quality habitat and forage for a number of wildlife species. Most wildlife species minimally impact sagebrush condition or demonstrate the capacity to alter plant composition or shrub cover. Similar to livestock, wild ungulates are capable of depleting native plant communities and/or decreasing shrub cover. During eras of high population, pronghorn antelope and mule deer have decreased Wyoming big and black sagebrush cover (Goodrich and Zobell 2012). Populations of these ungulates have remained relatively constant over the last couple of decades. Although areas of heavy use are documented, current populations of pronghorn antelope and mule deer are not sufficient to deplete sagebrush cover across large landscapes. During the next plan period, pronghorn antelope and mule deer populations are predicted to remain relatively constant. On the other hand, elk populations have increased substantially over the last 30 years. Elk use occurs in all sagebrush communities and elk operate as both a grazer and a browser. Like pronghorn antelope and mule deer, areas of heavy use by elk occur within the plan area, but current populations are not sufficient to alter plant compositions or deplete shrub cover across the landscape. Based on current trend, elk populations are predicted to continue to increase during the next plan period. As populations rise, elk use in sagebrush communities is predicted to increase, which indicates elk to be a potential stressor of these communities.

Recreation travel is a stressor of sagebrush communities. Considerable increase in all-terrain vehicle use and other off-road travel has occurred in these communities over the last 30 years, resulting in miles of new unauthorized trails and roads. Off-road travel reduces sagebrush cover, impacts plant composition, and decreases ground cover. In addition, recreation travel is the principal agent of new noxious weed and annual invasive infestations on the forest. Off-road travel use will increase distribution and accelerate spread of noxious weeds and invasive annuals in sagebrush communities. Recreation travel with its impacts are expected to increase during the next plan period. Other recreation use such as camping, hiking, and horseback riding is expected to minimally affect sagebrush overall and impacts would be site specific.

Oil and gas exploration and development is a present and foreseeable stressor of sagebrush of the Anthro Plateau landtype association. Numerous well pads, service roads, and pipeline corridors have been constructed within sagebrush communities over the last 15 years with additional infrastructure planned. These disturbances reduces sagebrush cover, diminish plant composition, and decreases ground cover, which opens niches for the spread of annual invasive plants. The oil and gas industry has accelerated the introduction, establishment, and spread of noxious weeds and invasive annuals into the area. As the demand for energy increases, expansion of oil and gas exploration and production is predicted to increase during the next plan period.

Climate-related risks could be a foreseeable stressor. Sensitivity to climate changes could vary between and within sagebrush taxa, with respect to their elevational ranges. For example, the effects of the amount, distribution, and timing of precipitation could have variable impacts on sagebrush communities depending on a site’s aspect, soil condition, and other and ecological conditions. Mountain big sagebrush’s sensitivity and vulnerability to climate related risks is moderate to high with an adaptive capacity of low to moderate (Padgett and others 2016). Optimal conditions for plant growth of this taxa is winter snow depth that covers shrubs, cool to moderate temperatures during the growing season, and annual precipitation greater than 13 Inches. In a warming climate, mountain big sagebrush below 8,000 feet elevation (i.e., drier environments) would be most susceptible to climate related risks. Under this conditions, shifts in herbaceous compositions from perennial to annual invasive plants is predicted. Fire frequency would likely increase, and woodland and Wyoming big sagebrush communities may shift upslope. Resilience of these communities would be low and with widespread loss predicted. Mountain big sagebrush communities above 8,000 feet are most likely to persist under a warming climate, but may be constantly stressed with increased drought. Resilience of these communities may diminish under a warming and drying climate.

Wyoming big sagebrush communities are the most vulnerable to a warming and drying climate. Sensitivity to climate related risks is high, adaptive capacity is low, and vulnerability is very high for these communities (Padgett and others 2016). A warming and drying climate would increase the frequency and duration of drought in Wyoming big sagebrush. Under these conditions, shrub and perennial grass die-back would occur more frequently, seed production would likely diminish, seedling establishment and survival would become increasingly difficult, annual invasive plants would displace native plants and dominate the understory, and fire frequency would likely increase. The expansion of desert shrubs into Wyoming big sagebrush is also a likely event under a warming climate. Resilience of these communities would be low and widespread loss of these communities is likely.

Black sagebrush communities are considered moderate to high in both sensitivity to climate related risks and adaptive capacity is considered moderate in vulnerability (Padgett and others 2016). Since black sagebrush has a wide elevational amplitude and inherent low productivity, these communities have a greater ability to adapt or resist changes of a warming and drying climate. Under warming conditions, black sagebrush is susceptible to more frequent and persisting droughts, but these communities are inherently fire resistant and less likely to accommodate annual invasive plants. Community resilience may be lower with a warming and drying climate, but black sagebrush is predicted to persist at most sites.

Climate Related Risks and Trends

During the next plan period, many sagebrush communities could be susceptible to changes in conifer displacement, fire frequency, drought, invasive annual plants, and due to climate change. For decades, conifer encroachment in, and displacement of sagebrush communities has progressively increased with decreased fire frequency, which indicates moderate departure from the natural range of variation. This trend is most common in mountain big, Wyoming big, and black sagebrush communities in and above the pinyon-juniper belt and in upper elevations of mountain big and black sagebrush near Douglas-fir and ponderosa pine forests. Wildfire events during the 30 years were not frequent enough to reverse this trend, but an increase in prescribed fire and lop-and-scatter treatments were implemented across the forest to help reduce conifer encroachment into many sagebrush communities. If fire frequency remains constant during the next plan period additional treatments would be necessary to curtail conifer displacement of sagebrush and help neutralize current trend. But fire frequency may increase in many sagebrush communities if the climate becomes warmer and drier, if severe and prolonged drought events become more common, and if the establishment and spread annual invasive plants increases. Currently, cheatgrass is present and spreading in Wyoming big sagebrush of the Green River, Antelope Flat, Structural Grain, and Anthro Plateau landtype associations and in mountain big and black sagebrush (below 8,000 feet) of the South Face, Glacial Canyon, Dry Moraine, Structural Grain, and Stream Pediment landtype associations. Invasive annuals are the primary cause of sagebrush communities not being in, or trending away from, the natural range of variation. At this time, many of these communities are currently within or of low departure from the natural range of variation, some are currently of low to high departure, but many are predicted to depart away from the natural range of variation during the next plan period. Spread of annual invasive plants in sagebrush of the landtype associations listed above is predicted to continue whether or not the climate becomes warmer and drier or droughts become more frequent; however, if the climate does become warmer and drier and droughts do become more frequent, the rate of spread is predicted to accelerate. Spread of invasive annuals appeared to accelerate during the droughts of 2002 and 2012 to 2014. Presence and abundance of cheatgrass and other annuals in sagebrush makes these communities more susceptible to fire. If cheatgrass spread increases in sagebrush during the next plan period, fire frequency is also predicted to increase. If fire frequency increases, increased fire suppression and management actions implemented to mitigate fire ignition and spread would be necessary to conserve sagebrush.

Comparison of the Natural Range of Variation and Current Conditions

Most mountain big and black sagebrush communities (above 8,000 feet) of the Parks Plateau, North Flank, Anthro Plateau, Avintaquin Canyon, and Strawberry Highlands landtype associations are currently in the natural range of variation or of low departure from it because of past plow-and-seed treatments. Sagebrush of higher elevation function properly following disturbances, with sagebrush and herbaceous vegetation returning within return intervals described for the natural range of variation. Conifer encroachment and displacement occurs within some sagebrush communities, but not of the magnitude within and directly above the pinyon-juniper belt. These communities show low to moderate departure from the natural range of variation. Prescribed fire and lop-and-scatter treatments have occurred during the 30 years to curtail conifer displacement. Additional treatments are recommended during the next plan period to address conifer displacement trend. Drought has not changed native plant composition, total ground cover, or sagebrush canopy cover of these communities. Herbaceous understories respond quickly and vigorously following fire. Return of sagebrush are within expected return intervals for mountain big and black sagebrush. Invasive annual plants are rare to nonexistent in these communities and are not predicted to increase during the next plan period. Introduced perennial grasses continue to dominate herbaceous cover in sagebrush plowed and seeded decades ago, but these have adequate species richness, high total ground cover, and normal sagebrush canopy cover. Resilience is high and is expected remain high during the next plan period. If the climate becomes warmer and drier, these communities may become more stressed and less resilient. Invasive annuals may become established and begin to spread within these communities.

Other stressors of sagebrush such as livestock grazing, recreation travel, wild ungulate use, and oil and gas exploration may impact sagebrush communities and may even contribute to the spread of invasive annual plants; but these can be or have been appropriately managed or mitigated to maintain sagebrush communities in the natural range of variation. For instance, management adjustments of a few livestock allotments could improve conditions of some sagebrush communities and seeding along service roads, pipeline right-of-ways, and well pads could reduce or eliminate the establishment and spread of noxious weeds or invasive annuals.

Existing conditions and current trends indicate that sagebrush communities of higher elevations are near or in the natural range of variation and are expected to remain so during the next plan period until long-term monitoring indicates otherwise. Sagebrush communities below 8,000 feet are more susceptible to drought, fire, and invasive annuals than those of higher elevation. Resilience of these communities and departure from the natural range of variation is low to moderate. Although many sagebrush communities within the plan area are currently near or in the natural range of variation, many of these are likely to trend away from it during the next plan period.

Pinyon and Juniper Woodlands

Persistent pinyon-juniper[6] woodlands occur on many landtype associations across the Ashley National Forest, totaling 122,383 acres. On the Uinta Mountain Section, it is most prevalent on Structural Grain (17,921 acres), Red Canyon (10,402 acres), and North Flank (18,868 acres) landtype associations, with some moderate presence on South Face (2,322 acres). On the Tavaputs Plateau Section, this type occurs primarily on the Anthro Plateau (61,761 acres) and the Avintaquin Canyon (6,368 acres) landtype associations. Pinyon-juniper consists primarily of Utah juniper (*Juniperus osteosperma*) and two-needle pinyon pine (*Pinus edulis*) but may also include Rocky Mountain juniper (*Juniperus scopulorum*).

Description of the Natural Range of Variation

Both pinyon and juniper have large ecological amplitudes (Tausch 1999) and are useful indicators of a rather broad climatic regime or thermal belt. Pinyon and juniper are also extreme generalists across a wide range of soils and other ecological features. Harper and Davis (1999) recognized Utah juniper and associated pinyons as insensitive to differences in geologic parent materials and soils derived there from. In the Great Basin, these pinyon and juniper range from the upper fringes of the Mohave Desert, where they occur with Joshua tree and blackbrush, to the lower fringes of high mountain forests (West 1988). On the Colorado Plateau, pinyon-juniper communities “constitute one of the most widespread vegetation types within the Four Corners states of Arizona, Colorado, New Mexico, and Utah in the American Southwest” (Jacobs and others 2008).

The primary disturbance of landscape change within the pinyon-juniper woodland and sagebrush ecosystem complex was fire, and fire patterns were generally tied to topography, soils, and existing vegetation composition (Tausch and Hood 2007). Fires ranged from low-intensity surface fires to stand-replacement fires depending on the existing understory vegetation and density of pinyon and juniper (Tausch and Hood 2007). Spreading, low-intensity fires, however, had a very limited historical role in shaping persistent pinyon-juniper woodlands (Romme and others 2007): “In very sparse woodlands, especially on rocky terrain, fires typically burned individual trees but did not spread extensively because of lack of surface fuels”. It is known with high confidence that fire did not generally “thin from below”, but typically spread crown to crown, especially when wind-driven, killing most or all trees (Romme and others 2007): Historical fire rotations were generally very long, approximately two to six centuries.

Stand dynamics in some persistent woodlands were “driven more by climatic fluctuation, insects, and disease than by fire” (Romme and others 2007). This is also known with high confidence, but its geographic applicability is not adequately known. Over the past 18,000 years or so, climate change drove pinyon juniper expansion and contraction, as well as changes in crown closure, tree density, and composition of juniper versus pinyon pine (Tausch 1999).

Disturbance in pinyon and juniper types may be necessary to maintain plant diversity. Pinyon and juniper have the capability to deplete shrub and herbaceous species as crown cover of these trees increase. With canopy closure of pinyon, juniper, or both, numbers, biomass, and seedbanks of understory species are reduced (Clary 1971; Koniak and Everett 1982; Everett 1987; Naillon and others 1999; Bunting and others 1999; Poulsen and others 1999). In Arizona, for example, Tress and Klopatek (1987) found the greatest plant community diversity in 35-year-old stands of pinyon and juniper, and decreasing diversity in 35 to 300 year old stands. In New Mexico, Short and others (1977) found a rapid decrease in

[6] Persistent PJ sites are distinguished from “expansion” PJ sites. Persistent PJ sites are primarily those sites where pre-settlement aged trees are present or trees had once occupied the site (e.g. skeletal remains may be present showing evidence of past fire). Potential expansion sites are areas where site conditions (e.g. climate) are intermittently suitable for pinyon and/or juniper. Areas of PJ “expansion” or “encroachment” may be mentioned in this segment, but are discussed in the segments on sagebrush communities in this assessment. Romme and others (2007) provides an identification key to distinguish between these two historical types of pinyon-juniper vegetation. Also see Intermountain SAF 2013

herbaceous production as tree density increased to around 200 trees per acre. In areas where density was greater than 200 trees per acre, they recorded a slow but steady decrease in herbaceous production.

In a study on the Ashley National Forest by Huber and others (1999), a high diversity of understory plants was found with less than 20 percent canopy cover of pinyon-juniper. Plant diversity was determined to be at risk when pinyon-juniper reach crown cover levels of 20 percent. Beyond 40 percent crown cover of pinyon-juniper, the understory was greatly depleted. Regression analysis between tree and understory cover by Tausch and Nowak (1999) also indicates that greater than 30 percent crown cover of pinyon-juniper typically depresses understory species. In addition, Loftin (1999) also found greater plant species richness with less pinyon-juniper cover. Other studies in the area indicate that depletion of understory species is associated with loss of resilience or ability of the native community to recover after fire. Without an abundance of resilient perennial herbaceous species and sprouting shrubs, burned pinyon-juniper sites are vulnerable to invasive annuals.

Current Status and Trends

During the 20th century, most persistent pinyon-juniper woodlands throughout the west have increased in tree density and crown cover (Romme 2007). Tausch and Nowak (1999) concluded that, "pinyon-juniper woodlands in the Great Basin generally had more open areas" during the Little Ice Age. Crown cover increased following the Little Ice Age with the combination of climate change and settlement impacts. Tausch (1999) suggested that there is potential for additional large, stand-replacing crown fires in pinyon-juniper woodlands over the next 150 years, due to current trends toward larger trees with increasing crown cover. Under the right conditions, many thousands of acres of mature pinyon and juniper woodlands can now burn in a day (Tausch 1999). This prediction has been validated to some degree by recent large stand-replacing fires that have occurred in pinyon-juniper forests on the Ashley National Forest (such as the Mustang fire of 2002 and the Neola North Fire of 2007). This recent increase in large stand-replacing fires is also supported by fire reports of the Forest Service and Bureau of Land Management (Gruell 1999).

On the Structural Grain and Red Canyon landtype associations, the Mustang Fire of 2002 is a good example of a recent large, stand-replacing fire. The Mustang Fire burned over about 20,000 acres of an extensive and nearly continuous mature woodland of pinyon-juniper that provided crown fuels for this intense, large fire. This large fire is consistent with the patterns and processes discussed by Tausch (1999) who indicates that these types of fires are to be expected where large, continuous stands of mature pinyon and juniper have developed. The Mustang Fire is indicated to be within an expected fire regime for this vegetation type. In addition, based on the above references, the Mustang Fire was likely not outside the expected natural range of variation.

The vulnerability of pinyon-juniper sites that are lacking resilient perennial herbaceous species was vividly demonstrated by the great abundance of invasive annual species found following the Mustang Fire of 2002. Tumble mustard and, to a lesser extent, Russian thistle became the dominant species over large areas two years after the burn.

Of all landtype associations of the Ashley National Forest, the greatest departure from historical vegetative composition is found on the Structural Grain landtype association. This departure is a function of the low resilience and high vulnerability of pinyon and juniper communities to invasive plant species. Much of this departure was observed after the 2002 Mustang Fire. Following this fire, tumble mustard, prickly Russian thistle, and cheatgrass increased rapidly. Within two years of the burn, these invasive species dominated large areas (Studies 6-2L, 6-21H, 6-24E, 6-24E3, 7-2D). This rapid colonization by invasive annuals is currently a pattern common of pinyon-juniper communities across the West (Chambers and others 2005, Goodrich and Rooks 1999, Svejcar 1999).

The flush of invasive annual forbs in the area burned by the Mustang Fire on the Structural Grain landtype association appears to have reached maximum distribution, and these annuals can be expected to decrease in some areas as perennial grasses, including seeded species, increase. However, this might not be the case in areas dominated by cheatgrass. This invasive species has increased more slowly than tumble mustard and prickly Russian thistle, however, it has continued to increase steadily after the burn. The response of cheatgrass in other areas of the West after disturbance indicates it can be expected to increase in the burned area.

On the Structural Grain landtype association, pinyon and juniper do have the capacity to dominate all lands. Johnson (2000) notes that juniper appeared in the Dutch John area of this landtype association at least by 6310 BP and pinyon pine by 4110 BP, and that juniper and pinyon likely began to dominate on the wooded ridges around 4100 BP. By 1980 the only sagebrush areas on this landtype association that were free of trees were the flat at the town of Dutch John, Dutch John Airport, the flat at Mustang Junction, and the tops of Goslin and Bare Top Mountains. Even the margins of these tree-free areas showed evidence of advancing pinyon and juniper.

On the Red Canyon landtype association, warmer southerly aspects dominated by pinyon and juniper, bluebunch wheatgrass and other perennial herbaceous species are common at the early-seral stage and persist into later stages in some degree depending on crown closure of pinyon-juniper. Sprouting shrubs are comparatively uncommon. Pinyon and juniper move slowly into burned areas. Recruitment of pinyon and juniper from seeds that survived fire is comparatively rare on this landtype association.

Native species presently dominate northerly aspects and most of the tributary drainages of the Red Canyon landtype association (Table 6-23, studies 5-64A, 5-64B). This indicates composition of these areas is similar to that of periods of the past, with the exception of cheatgrass that is abundant to locally dominant on southerly aspects of Red Canyon in the pinyon-juniper belt (Study 5-26).

With the existing high levels of recreation use on the Flaming Gorge Reservoir and along the Green River below the Flaming Gorge Dam, high rates of new invasive plant infestations can be expected for these areas. Spread of these infestations can be expected to be comparatively rapid along the reservoir and high use points along the river, but likely slow in the drier parts of the canyons. However, there are other species, including cheatgrass and musk thistle, with a high capacity to spread into the drier areas.

The abundance of cheatgrass and other invasive annuals, especially in the area burned by the Mustang Fire (2002), indicates more frequent fire return intervals in the future. Fire return intervals typically driven by cheatgrass are too frequent for the development of pinyon-juniper communities and, in many cases, too frequent for the recovery of big sagebrush. These studies suggest that there could be major changes in these plant communities in the future, and presents a risk to adjacent woodland and sagebrush communities.

Pinyon-juniper woodlands that occur on the North Flank landtype association include small isolated stands and some rather large stands that are mixed on the landscape with sagebrush and mountain brush types. Stands of these woodland species are common on erosional surfaces of dip and scarp slopes of the Navajo Sandstone, Carmel, Curtis, and other geologic formations. Density of trees or crown cover varies with geologic strata, geomorphic setting, aspect, and other features. Low-density stands are typical of scarp slopes of Navajo Sandstone, Carmel, and other formations. Stands of higher density and greater crown cover are more typical of dip slopes. Stands of pinyon and juniper are not common on valley-fill and lower colluvial slopes where big sagebrush is more common. Utah juniper with little or no pinyon forms stands on the Park City Formation and in a few other places. This condition occurs on approximately five percent of the North Flank landtype association.

Native plants generally dominate pinyon and juniper communities on the North Flank landtype association. Only small and relatively few infestations of invasive plants have been found here so far. Cheatgrass and other invasive annuals are of minor extent. Compared to pinyon-juniper woodlands of the Structural Grain landtype association, those of the North Flank landtype association have a lower presence of cheatgrass and other invasive species. A low departure from the natural range of variation is based on relatively low presence of cheatgrass and other invasive species. However, a moderate confidence estimate is based on the unknown potential of cheatgrass to dominate this type quickly.

On the Anthro Plateau landtype association, the steep canyon walls of lower and moderate elevations are nearly all dominated by scattered stands of pinyon and juniper. Fire ignition occurs frequently on this landtype association; however, a majority of ignitions result in only a single tree or otherwise small fires. The frequency of large fires in pinyon-juniper woodlands of this landtype association is likely in the order of 200 to 300 years. Large stand-replacing fires most likely occur within areas of mature pinyon-juniper woodlands that have developed an average crown cover greater than 50 percent. This level of canopy cover is beyond the capability of the woodlands of the steep canyon walls set in the Green River and Uinta Formations. Consequently, these areas tend to serve as fuel breaks. However, large stands with a high percent crown cover do develop on ridges and lower gradient slopes. The possibility of larger fires increases where these fire-prone pinyon-juniper stands contact the more flammable shrub and grass communities of the swales and canyon bottoms.

Fire has historically kept trees off most swales and canyon bottoms on this landtype association. The rockier and sparsely vegetated canyon walls generally limit most fires to the bottom of the canyons. The fire return interval for pinyon-juniper communities on this landtype association is estimated to be a low departure from the natural range of variation based on recent assessments of fire in pinyon-juniper communities on the Ashley National Forest and in other areas in the West.

Specific to this landtype association, recent large fires in pinyon-juniper woodlands have been absent. Perhaps more fuel breaks are a factor that has reduced the amount of large-scale fires on this landtype association. Prescribed fire has been applied several times over most of the canyon bottoms without fire advancing up the sides of canyons that are covered with pinyon-juniper (Studies 69-76B, 69-91A1, 67-74B). Successful control of prescribed burns has been demonstrated throughout the canyon bottom type. However, a recent example of fire burning the side of a canyon occurred at the junction of the Left Fork Antelope Canyon and Chokecherry Canyon. There is also some historical evidence that fire occasionally burned narrow strips of pinyon-juniper on the canyon walls.

The major difference between pinyon-juniper woodlands of this landtype association and those of the Structural Grain landtype association is the presence of invasive species. Compared to the Structural Grain, the Anthro Plateau has a low presence of cheatgrass or other invasive species that have the potential to alter plant community dynamics and function. Wildfires and prescribed fires have burned relatively small areas and native plants generally dominate after fire. The lower presence of invasive species is an indication of a moderate departure from the natural range of variation.

Climate Related Risks and Trends

Although large-scale severe fires are typical fires for this vegetation type, the frequency of these types of fires appears to have increased throughout the West in the last 20 years (Romme 2007). This increase "may be a consequence of warmer temperatures and longer fire seasons, greater fuel continuity developing from increasing tree cover" but may occur in combination with other factors, such as invasion of highly flammable annual grasses like cheatgrass (Romme 2007). Pinyon-juniper woodlands are highly sensitive to climate related risks. Lower elevations of the Pinyon-juniper range will likely be most affected by rapidly increasing temperatures. Climatically suitable habitat for two-needle pinyon and Utah juniper will likely contract significantly by the end of the century (USDA Forest Service 2016b, Rehfeldt and others 2012). There is likely to be a great reduction in area dominated by pinyon-juniper as a result of increased drought and temperatures. There is indication that historical woodlands are already in decline on the Colorado Plateau[7] (Arendt and Baker 2013) due to an excess of fire since Euro-American settlement.

Comparison of the Natural Range of Variation and Current Conditions

The fire return interval for pinyon-juniper communities is estimated to be a low departure from the natural range of variation, based on recent assessments of fire in pinyon-juniper communities on the Ashley National Forest and in other areas in the West.

The departure from the natural range of variation for pinyon juniper communities on the Ashley National Forest varies according to landtype association. Most pinyon and juniper communities are considered to be at a low departure from the natural range of variation. The exception is pinyon and juniper communities found on the Structural Grain landtype association. These communities are considered to be at high departure from the natural range of variation. This high departure is due to the presence and high frequency of invasive species, including cheatgrass, and their strong biological capability to alter plant community dynamics.

Desert Shrub

Desert shrub communities are found mainly within the Green River landtype association, with representation from the Antelope Flat, North Flank, and Moenkopi Hills landtype associations.

Table 16. Vegetation type and acres for desert shrub communities on the Ashley National Forest

Vegetation type	Acres
Wyoming big sagebrush	38,473
Semi-barrens	14,408
Saltbush	6,504
Shadscale	2,671
Spiny hopsage	2,048
Desert shrub	1,891
Winterfat	587
Greasewood	1,252
Gray molly	111
Bud sagebrush	31
Total	67,976[1]

[1] Includes greasewood, which is not part of the total for desert shrub

Description of the Natural Range of Variation

Most desert shrub communities on the Ashley National Forest are located in cold desert environments along Flaming Gorge Reservoir in southwestern Wyoming. Summers are typically warm and winters consist of below freezing temperatures (Blaisdell and Holmgren 1984). Annual precipitation ranges between 6 to 12 inches annually, and generally comes during the winter as snow. Geologic material and geomorphic processes are factors that influenced landforms of the Green River landtype association. Such

[7] Persistent pinyon-juniper woodlands on the Ashley National Forest occurs on the northern fringe of the Colorado Plateau as mapped by Jacobs and others (2008)

processes created land features that include moderately to highly dissected slopes, benches, and plateaus; scarps; and saline or alkaline flats. Ephemeral channels or gullies dissect many of the landforms. Most channels are low-to-moderate gradient, narrow, and V-shaped. In contrast, few channels are found in areas of gently sloping topography (Buckboard Marina, Anvil Draw). Residual surface materials, such as eolian sands, slopewash colluvium, pediments, and partially decomposed shales, influence composition and distribution of plant communities within this region. Vegetation is dominated by cold desert shrubs of the Chenopod family, sagebrush, early-season grasses, and a few forbs adapted to semiarid environments. Desert shrub communities of the Green River landtype association closely align to those described by Blaisdell and Holmgren (1984). Desert shrub plants only grow when temperatures are favorable and soil moisture is present, which indicates weather as a primary driver in community dynamics (Blaisdell and Holmgren 1984). Vegetation production and cover is dependent upon the amount and timing of precipitation, and dramatic shifts can occur annually. Spring and early summer precipitation is favorable for most desert shrub plants, but especially for herbaceous and sub-shrub species. Certain plants germinate, grow, and produce seed only if adequate soil moisture is available during a certain time of the growing season. Otherwise, these plants may remain dormant until favorable conditions exist. Blaisdell and Holmgren (1984) stated that, "because the moisture regime is so variable from year to year, and different species flourish under different seasons of soil moisture, there is irregularity in thriftiness of species and combinations from year to year. Only rarely do all components of the vegetation thrive at their best in the same year." Years of drought usually lead to meager vegetation and seed production, but can cause dieback of shrubs and herbaceous plants if the drought is severe and/or persisting. Resurgence of vegetation production, cover, and seed production occurs with subsequent years of abundant moisture (Studies 72-2A, 72-13B, 72-14A, 72-18A, 72-21D, 72-33A, 75-17A- C, 75-18, 75-23C, USDA Forest Service 2006).

Most desert shrub communities are sensitive to soil features. For example, spiny hopsage (*Grayia spinosa*) communities are tightly aligned with eolian sand and gravelly slope-wash colluvium. On erosional surfaces of the Green River Formation, where eolian and slope-wash materials are absent, Wyoming big sagebrush and shadscale communities dominate. On the other hand, where gravelly pediment surfaces occur, perennial grasses such as needle-and-threadgrass, Indian ricegrass, and bluebunch wheatgrass are abundant and these often dominate vegetation cover. Winterfat communities are found in silty loam soils of the valley bottoms and Gardner saltbush (*Atriplex gardneri*) grows in fine-textured clays with relatively high pH.

In general, desert shrub communities consist of a shrub or sub-shrub component, with few herbaceous plants (less than 15). Herbaceous vegetation production is relatively low and perennial grasses make up most of that production. Total ground cover is usually low (less than 60 percent) unless surface rock or pediments are present. Bare spaces between shrubs, often consisting mostly of bare soil, is typical. Blaisdell and Holmgren (1984) found "plant communities [to be] normally distinct, but sometimes merge imperceptibly into one another." Mixed shrub communities, such as shadscale and Wyoming big sagebrush, are common. Fire rarely occurs in desert shrub communities because vegetation production is low and bare soil to intermittent herbaceous cover between shrub interspaces is typical. In shadscale and similar communities, fire is infrequent at best but possibly nonexistent (Knapp 1996). For Wyoming big sagebrush, a fire interval of 50 to 110 years is indicated by many but may occur no often than 200 to 350 years (USDA Forest Service 2016, Winward 1991, Mensing and others 2006).

Although several desert shrub communities are found within the Green River landtype association, only those of 2,000 acres or greater will be discussed in further detail in regards to the natural range of variation. An exception will be semi-barrens, which represents a land feature rather than a plant community. Spiny hopsage can form pure stands. Spiny hopsage, however, is more often the dominant shrub in eolian sands and slope-wash colluvium, with the presence of sticky yellowbrush, Wyoming big sagebrush, shadscale, or greasewood (*Sarcobatus vermiculatus*). Average crown cover of this shrub is about 12 percent, but can be as high as 20 percent. Herbaceous species richness averages six plants per site and most are perennial grasses. These include squirreltail, western wheatgrass, Indian ricegrass, needle-and-threadgrass, and galleta. Total ground cover is very low, averaging about 36 percent. Nearly all cover comes from live vegetation (Studies 72-21B, 74-2, 75-14B, 75-17A-B, 75-39A, 75-46, 77-2, 78-9, 78-16, 78-16C, 78-18). Shadscale seldom forms pure stands, but is often dominant and occasionally co-dominant in mixed desert shrub communities. It mixes with winterfat, greasewood, gray molly (*Bassia americana*), Wyoming big and black sagebrush, bud sage (*Artemisia spinescens*), spiny hopsage, low rabbitbrush, and other saltbush taxa (Blaisdell and Holmgren 1984). Shadscale communities typically form extensive stands on flats or gently sloping topography. It is a short-lived shrub and is more vulnerable to drought and excessive soil moisture than other salt-desert shrubs (Blaisdell and Holmgren 1984, Goodrich 2012). Average crown cover of this shrub is about seven percent. Herbaceous species richness averages six plants per site. That number is considerably lower than Wyoming big sagebrush, comparable to spiny hopsage, but higher than Gardner sagebrush. Grass cover averages 10 percent, with a wide range from zero to 30 percent. Perennial grasses include squirreltail, Sandberg bluegrass, Indian ricegrass, and needle-and-threadgrass. Total ground cover averages 40 percent (Studies 72-1A,C-D, 72-2, 72-21C, 72-24, 72-28B, 75-18B, 75-18E, 75-21B2-C, 75-23C-D, 75-28B, 75-32, 75-50B). Gardner saltbush is more limited in its habitat than other saltbrush species. This low-growing subshrub forms nearly pure stands on “badland” clay soils with relatively high soluble salt content (Blaisdell and Holmgren 1984) and “where few other plants have capacity to grow” (Goodrich and Zobell 2011). At marginal sites, saltbrush occasionally mixes with shadscale, winterfat, and bud sage. Gardner saltbush crown cover can range from 10 to more than 35 percent, but averages 19 percent. Little to no herbaceous cover occurs in most communities and herbaceous species richness averages less than five plants per site. Perennial grasses that occasionally grow with Gardner saltbush include Sandberg bluegrass and squirreltail. Total ground cover averages 46 percent (Studies 6-43A-B, 6-16A, 71-11, 72-13A, 72-18A, 72-21D, 72-26, 72-31A-B, 75-4A, 75-21B1, 77-1). Wyoming big sagebrush is the most common community of the Green River LTA. Where annual precipitation is at or slightly less than eight inches, Wyoming big sagebrush intermixes with salt-desert shrubs such as shadscale, spiny hopsage, and sticky yellowbrush. In this setting, Wyoming big sagebrush dominated areas are similar to desert shrub communities in terms of precipitation, comparatively low herbaceous species richness, total ground cover, and production. Refer to sagebrush composition for a complete description of Wyoming big sagebrush communities.

Current Status and Trends

For 30 years, most desert shrub communities showed resilience to drought, ungulate grazing, and other disturbances up to about the year 2002. These communities responded successfully to boom and bust cycles that were driven by precipitation patterns and other weather-related events, concurrent with livestock grazing and other stressors. Some drought events caused occasional die-back in communities of shadscale and Wyoming big sagebrush, but recovery occurred with the return of abundant and timely moisture. Invasive annuals such as halogeton and cheatgrass were present as early as the 1970s, but did not impact ecological processes of desert shrubs (Goodrich 2005, Goodrich and Zobell 2011).

Desert shrub communities have been influenced by a long history of domestic sheep and cattle grazing. Livestock use of shrubs and herbaceous forage occurs during winter, spring, and early summer. Reductions in shrub, herbaceous cover and total ground cover have occurred (Study 72-1), but grazing practices did not interrupt ecological processes of desert shrubs. Vegetation evaluations of the Cedar Mountain, Rock Springs, and Sugar Loaf Allotments indicated satisfactory composition and ground cover conditions for desert shrub communities as recent as 2005, with an exception of some Gardner saltbush sites (Goodrich 2005, Goodrich 2005b, Goodrich 2005c, Goodrich 2013, Goodrich and Zobell 2013, Goodrich and Zobell 2013b, Goodrich and Zobell 2013c).


Photos of study 75-21C. Green River Basin. 2003 and 2005. Photos depict a boom-and-bust cycle in shadscale community. Shadscale die-off depicted in 2003 occurred following severe drought of 2002.

Since 2002, some desert shrub communities have become particularly susceptible to invasive annual plants, while others show signs of susceptibility. Long-term monitoring has documented the spread of invasive annuals during and shortly following periods of severe drought. Notable recent droughts include years of 2002, and 2012 through 2014. Gardner saltbush communities are most vulnerable to invasive annuals and were the first communities to be negatively affected. Over the last 15 years, many Gardner saltbush communities have been entirely displaced by halogeton (Goodrich and Zobell 2009, Goodrich and Zobell 2011, Studies 72-13, 72-13G, 72-18A, 75-21B, 72-21D, 72-31A-B, 75-4A-B, 75-21B1-B4). Halogeton has the capacity to persist on clay soils with high soluble salt content, soils that Gardner saltbush thrives on (Goodrich and Zobell 2011). This invasive annually changes soil chemistry and ecology enough to make the soil incompatible for Gardner saltbush (Goodrich and Zobell 2011). Additionally, die-off of Gardner saltbush and displacement by halogeton has occurred concurrent with, and in the absence of, livestock grazing, indicating little to no influence from grazing (Study 75-7). Gardner saltbush communities are outside of or are trending away from the natural range of variation.

Gardner saltbush is preferred by domestic sheep. With the loss of many Gardner saltbush communities, surviving communities and other desert shrubs may receive additional browse pressure by domestic sheep. Halogeton has been documented within shadscale, winterfat, spiny hopsage, sticky yellowbrush, and Wyoming big sagebrush communities. Halogeton has increased in winterfat and shadscale, especially communities that have Gardner saltbush mixed with them (Studies 72-1, 72-2, 72-13E-G, 72-14A1-A2, 72-18B-D, 72-28B, 72-31G, 72-43, 75-14A, 75-18A, 75-18D-E, 75-21A-C, 78-18A). Winterfat also appears vulnerable to halogeton displacement, likely because of similar soil characteristics that supports Gardner saltbush. Winterfat appears to have a negative relationship with halogeton and cheatgrass; as these invasive annuals increase, winterfat decreases (Goodrich 2012). In addition, Goodrich (2012) determined that excessive browsing of mature winterfat may facilitate displacement by halogeton (Blaisdell and Holmgren 1984). A few communities of shadscale have halogeton in them and shrub displacement by halogeton has likely occurred (Study 75-18E). It is unknown whether halogeton is capable of displacing shadscale in this setting.

Photos of study 72-013A. Green River Basin. 1993 and 2011. Gardner saltbush community that has been displaced by halogeton. Tipping point occurred following severe drought in 2002.

Additional monitoring will be required to make that determination. Halogeton appears to have little to no influence of ecological processes of spiny hopsage, Wyoming big sagebrush, and most shadscale communities. Currently, most shadscale, spiny hopsage, Wyoming big sagebrush, and other desert shrub communities are in satisfactory condition in regards to plant composition, species richness, total ground cover, and shrub cover. These communities fall within or are trending toward the natural range of variation as described above; but cheatgrass is becoming more prevalent and is spreading into these communities (Studies 6-16A, 6-43A-B, 72-8, 72-8B, 72-14B, 72-32, 75-23A, 75-23C-E, 75-32, 75-39A, 75-45A-B, 78-9, 78-13C, 78-16, 78-26). Although Wyoming big sagebrush and winterfat communities appear most susceptible to cheatgrass, essentially all other desert shrub communities will also be susceptible if cheatgrass continues to increase. Cheatgrass is very capable of altering fire frequency and disturbance response sequences of desert shrubs, especially since most desert shrubs do not sprout following fire (Blaisdell and Holmgren 1984).

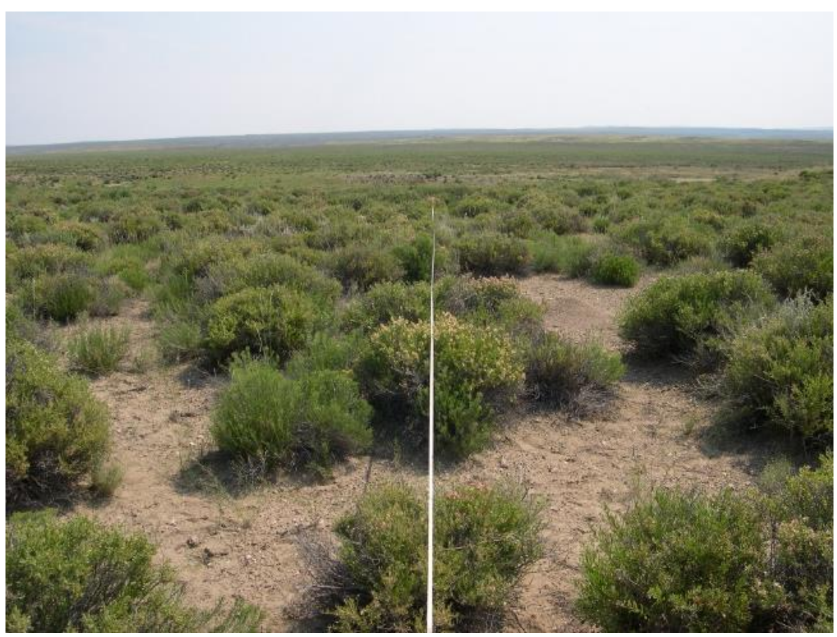


Photo of study 75-14A. Green River Basin. Spiny hopsage community in cold desert environment.

Management Activities and Uses

Because of its harsh environment, low annual precipitation, and limited vegetation production, desert shrub communities have fewer anthropogenic and ecological services than their montane or high elevation counterparts. Similar to many other vegetation types, livestock grazing is the most extensive and enduring human use of desert shrub communities. Historically, livestock have grazed all or nearly all desert shrub communities with continual use spanning more than a 100 years. Almost all desert shrub communities are classified as “capable” for livestock grazing, providing both summer, spring, and winter forage for cattle and sheep. Many of these communities are relatively low in herbaceous production, but most desert shrubs provide palatable browse for ungulates (Blaisdell and Holmgren 1984, Goodrich 2012). No known vegetation treatments to increase livestock forage production were implemented during or before the last plan period. Cattle and sheep grazing of the Green River landtype association has remained relatively constant over the last 30 years, and grazing is expected to either remain constant in terms of numbers, time, and intensity. Forage production, however, will likely trend downward if invasive annuals continue to increase.

Desert shrub communities usually have limited recreation value; however, Flaming Gorge Reservoir continues to attract hundreds of thousands of visitors annually. Much of the recreation area is located within these desert shrub communities and occurs during the summer and autumn months. These consist mostly of fishermen, campers, and boaters. Other minor recreation opportunities include hiking, wildlife viewing, and hunting. Many unauthorized roads are created and maintained near the shores of the reservoir. Overall, desert shrub communities nearest to main roads receive the highest use and greatest impacts. Most new noxious weed infestations are located along the drawdown basin of the reservoir and along authorized and unauthorized roads. Similarly, cheatgrass, halogeton, and other invasive annual plants establish along the reservoir shoreline and roadways. These plants eventually spread into nearby desert shrub communities. Increased recreation use of the Flaming Gorge Recreation Area is expected during the next plan period. With increased visitation, additional noxious weed and invasive annual plant infestations are predicted to occur. Regardless of the trend in recreation use, new weed infestations along the drawdown basin of the reservoir are predicted to remain constant or increase.

Desert shrub communities provide habitat and forage for some wildlife species. Shrub communities not covered by snow during winter months become important winter habitat for mainly pronghorn antelope and mule deer. Browsing intensity of desert shrubs is subject to shrub preference, winter conditions, availability of other forage, and ungulate population densities. Pronghorn antelope and Greater Sage-grouse use Wyoming big sagebrush year round in the Green River landtype association. Heavy browsing in some communities has reduced Wyoming big sagebrush crown cover to less than five percent (Goodrich and Zobell 2012). Populations of greater sage-grouse have declined over the last 30 years. Greater Sage-grouse populations are expected to either remain stable or trend downward if Wyoming big sagebrush communities burn as a result of invasive annual infestations. A number of other mammals and birds use desert shrubs to meet life cycle needs.

Influences of Drivers and Stressors

Invasive annual plants are the greatest threat to desert shrub communities. These invasive species are capable of not only changing the composition of desert shrub communities, but to eradicate community drivers such as shrubs and perennial grasses. Since the introduction and rate of spread of invasive annuals is usually influenced by other stressors and drivers of desert shrub communities, invasive annuals will be discussed in concert with stressors and drivers that are beneficial to their increase.

Drought is a natural disturbance that occurs regularly in cold deserts of the Intermountain West, indicated by the boom and bust cycles of vegetation that occur in these environs. Under the natural range of variation, response mechanisms to drought have successfully maintained desert shrubs on western landscapes, concurrent with most other stressors and drivers. Response to drought is not equal among desert shrubs. For instance, shadscale appears most susceptible to drought, where shrub die-back often occurs following these events (Blaisdell and Holmgren 1984). On the other hand, winterfat and Gardner saltbush are relatively drought tolerant and show limited to no die-back in drier-than-normal years. Generally, most drought events minimally impact shrub cover or change plant compositions long-term. However, high mortality of Wyoming big sagebrush and shadscale, and/or their herbaceous understories have occurred during the most severe or prolonged droughts (Studies 72-1A, 75-18A, Goodrich and Zobell 2013). In most cases, herbaceous understories have increased in density, cover, and productivity following shrub die-back. Surviving shrubs responded with greater foliage and seed production with the return of abundant and timely moisture. Where die-back of herbaceous plants has occurred, two responses have generally occurred. First, herbaceous plants reestablish or recover from die-back in two to five years post drought. Second, surviving understory species replace, at least temporarily, those that experienced die-back. The introduction and spread of invasive annuals have altered drought response mechanisms of some communities. Since about 2002, some desert shrubs have not successfully recovered from drought events with the presence of and interactions with invasive annuals such as halogeton and cheatgrass. With the presence of halogeton, Gardner saltbush has experienced widespread die-off because of halogeton's ability to alter soil chemistry. Shrub die-off has been followed by an increase of halogeton, which then dominates the site (Goodrich and Zobell 2009, Goodrich and Zobell 2011, Studies 72-13, 72-13G, 72-18A, 75-21B, 72-21D, 72-31A-B, 75-4A-B, 75-21B1-B4). Halogeton is present in other communities and has increased and sometimes displaced winterfat communities and has increased in shadscale communities. The long-term effects of drought and halogeton in these communities are not fully understood. Cheatgrass spread accelerates during drought events, intermixing or often displacing herbaceous understories. Wyoming big sagebrush and winterfat communities appear most susceptible to cheatgrass spread during drought events. During the next plan period, desert shrubs and herbaceous plants are expected to respond to drought events similar to the natural range of variation if annual invasive plants are not present. Where present, annual invasive plants have, are beginning, and will continue to alter some desert shrub communities' ability to respond to drought and other disturbances. Gardner saltbush communities are predicted to decrease as halogeton spreads and new drought events occur. Some winterfat and shadscale communities may also reflect this trend. Cheatgrass is also predicted to increase in desert shrub communities as new drought events occur.

Recreation travel is a stressor of desert shrub communities. The Flaming Gorge National Recreation Area is serviced by a network of paved and unpaved roads, including a major highway. These road networks are likely the most efficient vectors for spread of noxious weeds, halogeton, cheatgrass, and other invasive annual plants into the area. Visitation to the area has increased over the 30 years as has an increase in all-terrain vehicle use and other off-road travel. Off-road travel disturbances decreases vegetation and ground cover, and increases the spread of noxious weeds and invasive annuals. Recreation travel with its impacts are expected to increase during the next plan period.

Flaming Gorge Reservoir is also a vector of spread for noxious weeds and invasive annual plants. Seeds and plant materials from the Green River watershed are deposited along the shoreline of the reservoir. The fluctuating water level of the reservoir opens niches for plants to establish. Nearly pure stands of halogeton and other undesirable plant species are found in some low-gradient areas of the draw-down basin of the reservoir (Study 83-2B). Additional noxious weed and invasive annual infestations are predicted to occur during the next plan period.

Fire does occur in desert shrub communities; however, these events are quite rare. Under the natural range of variation, desert shrub communities have low to almost nonexistent fire frequencies. Most desert shrubs do not sprout following fire and shrub return intervals for most are relatively long. Community response following fire is similar to response following drought. This response is an increase in density, cover, and productivity, and with temporary dominance of the herbaceous understory (Study 83-2B). Since fire is rare, vegetation production is minimal, and precipitation is low and intermittent, desert shrub communities do not respond as quickly nor as vigorously following fire as do most montane shrublands. This makes these communities greatly susceptible to change with the presence of annual invasive plants. Since 2002, cheatgrass has established and is spreading within many desert shrub communities, especially Wyoming big sagebrush and winterfat communities. The annual has the ability to alter fire regimes of desert shrubs. Cheatgrass does this by growing within and filling shrub interspaces, which creates continuous fuel between shrubs that would otherwise be separated by bare or rocky soil, pediments, or sporadic herbaceous cover. Desert shrubs communities become more prone to fire as cheatgrass spreads. When infested sites burn, cheatgrass out competes and displaces native vegetation, dominates cover, and is capable of creating annual monocultures. Fire frequency of impacted communities increase, often at intervals that preclude shrubs from reestablishing. Although cheatgrass is present in some desert shrub communities, these communities are currently relatively intact due to the absence of fire. However, further spread of cheatgrass is predicted during the next plan period, which makes desert shrubs more susceptible to fire. In conclusion, as cheatgrass becomes more widespread, the incidence of fire and the size of burns are predicted to increase.

Livestock grazing is a stressor of desert shrub communities. This long-term and enduring practice impacted plant composition and ground cover in many desert shrub communities, but grazing intensities were not great enough to inhibit these communities from functioning under the natural range of variation. Prior to 2002, monitoring indicated that desert shrub communities remained resilient concurrent with grazing, drought, and other disturbances. Communities consisted of native perennial plants and ground cover was sufficient to keep hydrologic function from changing (Goodrich 2005, Goodrich 2005b, Goodrich 2005c, Goodrich 2013, Goodrich and Zobell 2013, Goodrich and Zobell 2013b, Goodrich and Zobell 2013c). Along with other forest uses, livestock have contributed to the spread of weeds and invasive annuals. The spread of invasive annuals over the last 15 years has altered composition and function. The spread has also diminished resilience of some desert shrub communities grazed by livestock, particularly Gardner saltbush. As invasive annuals alter composition of additional Gardner saltbush and other desert shrub communities, available forage for livestock will decrease. Invasive annuals are predicted to increase and further spread during the next plan period. Under this scenario, livestock grazing in terms of numbers, time of use, class, intensity, or some combination of these things are likely to decrease over the next plan period.

During eras of high population, pronghorn antelope have decreased Wyoming big sagebrush cover (Studies 72-18D, 75-23). Although areas of heavy use are documented, browsing by pronghorn antelope was not sufficient to deplete sagebrush cover across large landscapes. On the other hand, cheatgrass is capable of altering shrub cover and herbaceous compositions of Wyoming big sagebrush communities across large landscapes. Considerable loss of sagebrush due to cheatgrass spread and increased fire frequency would negatively impact pronghorn antelope populations. During the next plan period, pronghorn antelope are predicted to remain relatively constant if Wyoming big sagebrush shrub cover remains relatively constant. Climate related risks could be a foreseeable stressors in desert shrub communities. Although desert shrubs have wide ecological distributions in semi-arid regions and are adapted to harsh, dry climates, the shrubs' resilience with a warmer climate is in question due to the presence and spread of invasive annual plants (Padgett and others 2016). Desert shrubs' sensitivity to climate-related risks is considered moderate because of their capacity to adapt to dry, highly variable environments. However, their vulnerability to climate-related risks is considered moderate to high because of their susceptibility to invasive annuals such as halogeton and cheatgrass.

Invasive annuals successfully function with drought and are predicted to function with warming climates. A warmer and drier climate would likely increase the frequency and duration of drought. Under these conditions, Gardner saltbush displacement by halogeton would accelerate. Die-back of other desert shrubs and their perennial grass understories would occur more frequently, seed production would likely diminish, and seedling establishment and survival would become increasingly difficult. Consequently, shifts from perennial herbaceous compositions to annual invasive plants is predicted to occur if invasive annuals are present. Fire frequency would increase in desert shrub communities infested with cheatgrass. The accelerated spread of cheatgrass is predicted to occur following every drought and fire event. Fire intolerant desert shrubs could not reestablish if fire occurs too frequently. In summary, the presence of invasive annuals would diminish desert shrub resilience and widespread loss of these communities is predicted to occur. Because of this, a low to moderate rating for adaptive capacity in desert shrub communities is likely (Padgett and others 2016).

The following series of photos show loss of sagebrush canopy cover from 1972 to 2012, mostly from pronghorn antelope browsing in absence of livestock grazing. Needle-and-threadgrass dominated herbaceous cover in 2012. Perennial grass die-off depicted in 2013 occurred following severe drought of 2012. By 2016, cheatgrass and other annuals are present at the site with minimal needle-and-threadgrass.

Photos of study 5-3D. Lucerne. 1972, 2012, 2013, and 2016. Photos depict loss of sagebrush canopy cover from 1972 to 2012 mostly from pronghorn antelope browsing in absence of livestock grazing.

Climate Related Risks and Trends

A warmer and drier climate associated with frequent droughts will increase the spread of invasive non-native plants in desert shrub communities. Evidence for this is the spread of invasive annuals that accelerated during the droughts of 2002, and 2012-2014. Since 2002, many Gardner saltbush communities have been displaced by halogeton. This trend is predicted to continue during the next plan period. Additionally, some winterfat and shadscale appear vulnerable to halogeton displacement.

Cheatgrass is present and spreading in Wyoming big sagebrush, winterfat, shadscale, and other desert shrub communities. Cheatgrass spread is predicted to occur during the next plan period and spread will accelerate if additional droughts and a warming climate occur. Presence and abundance of cheatgrass make these communities susceptible to fire. If cheatgrass spread increases during the next plan period, fire frequency is also predicted to increase, especially under a warmer and drier climate. In addition, the expansion of desert shrubs into Wyoming big sagebrush is also predicted with a warming climate. However, expansion of these communities would be precluded with the presence of invasive annuals.

Comparison of the Natural Range of Variation and Current Conditions

Invasive annual plants are the primary cause of desert shrub communities' departure or trend away from the natural range of variation. Where invasive plant communities are absent, there is a low departure from the natural range of variation. Where invasive annuals are present, desert shrubs communities are of moderate to high departure and are trending away from the natural range of variation. As invasive annuals increase, resilience of these desert shrub communities rapidly diminishes.

Desert shrub communities that appear to be within the natural range of variation include spiny hopsage, gray molly, and black sagebrush communities. Desert shrub communities that appear to have a low departure from the natural range of variation include shadscale and Wyoming big sagebrush communities. Winterfat communities appear to be moderately departed from the natural range of variation, and Gardner saltbush communities appear to be not functioning and have a high departure from the natural range of variation. All these desert shrub communities will likely trend outside the natural range of variation during the next plan period. Other stressors such as drought and fire will accelerate this trend. Within the plan period, substantial change in vegetation composition of many desert shrub communities is predicted to occur because of the presence and spread of invasive annual plants.

Structural Stages of Vegetation

Alpine

Description of the Natural Range of Variation

Within the plan area, alpine communities are only found in the Uinta Mountains and represent about 169,000 acres. Alpine structure is described at a landscape scale spatially and temporally and at a community scale by height, density, canopy cover, or a combination of those things if the community consists of a shrub component with average height greater than six inches. Alpine communities are found above timberline with an elevational range of 11,200 to over 13,500 feet for the Uinta Mountains. Plants are "uniquely adapted to their environment" (Romme and others 2009). Most are herbaceous, perennial, drought resistant, and of low growth form. Some plants are mat-forming, and most shrubs are dwarfed in their structure (less than 6 inches). Alpine communities are diverse, and their presence or absence is conditional on topography, geology, aspect, snow accumulation and persistence, wind exposure, rodent activity, soil moisture, temperature, and other factors that form habitable niches (Baker 1983, Billings 1973, Bryant and Scheinberg 1970, Cox 1933, Douglas and Bliss 1977, Goodrich 2004, Goodrich 2006, Johnson and Billings 1962, Lewis 1970, Marr 1961, Stanton and others 1994, Walker and others 1993, Willard 1979, Brown 2006, Romme and others 2009). Alpine plant communities are often small and "community types change abruptly over short distances" (Romme and others 2009). Although alpine communities are diverse compositionally, most of these are, or appear, uniform structurally. Few communities consist of a shrub component with height exceeding six inches. Alpine community structure of Uinta Mountains and the environmental conditions that determine their presence are similar to those described in other alpine areas in the region (Romme and others 2009, Cooper and others 1997).

Photo of study 12-44. Milk Lake Ridge. Uinta drainage. 2003. View of *Bellardi kobresia* community that is within the natural range of variation from structure and composition.

Most alpine communities are similar in regards to structure, where plant heights seldom exceed 12 inches and vegetation seldom consists of more than one structural layer. Uniform structure is found in most wet, dwarf shrub, meadow, mesic, and dry alpine communities (Brown 2006). Some communities consist of an additional shrub or woody layer. Plane-leaf willow is the most common shrub in alpine communities. It grows in wet, meadow, and mesic communities as described by Brown (2006). These willows form tall thickets along streams and in areas where water and soil conditions are ideal, but are also found in many other wet communities, in various densities and heights, where conditions are tolerable. Shrubs range in height from six inches to six feet (Brown 2006, Study 16-20B2). Plane-leaf willow canopy cover varies from less than 5 percent to as high as 95 percent. Shrub height and canopy cover are dependent upon the depth of the water table and the duration that soils are water saturated. Gray-leaf willow grows in well-drained soils. This willow can be found on steep rocky or talus slopes to benches or basins of gentle gradient. Shrubs range in height from six inches to three feet (Brown 2006, Studies 10-13B). Willow canopy cover can reach as high as 35 percent. Shrubby cinquefoil (*Potentilla fruticosa*) and barren-ground willow (*Salix brachycarpa*) are shrubs with minor representation in alpine communities and are not discussed in this analysis. Plant density of alpine can have a lot of variation. Plant density is high in wet, low and dwarf shrub, meadow, mesic, and dry alpine communities, which coincides with relatively high vegetation cover. Densities are low in snow bed and talus, fellfield, and barren communities.

Photo of study 10-13B. Upper Oweep Basin. Lake Fork drainage. 2006. View of structure in gray-leaf willow community.

Alpine areas within the Uinta Mountains have likely fluctuated in size, and their perimeter has advanced and receded above or below timberline over time (Munroe 2003). Munroe (2003) found tree lines shifted upslope from 60 to 180 meters and timberline forests were denser at five different sites since 1870. He reasoned that, following the end of the Little Ice Age around 1850, tree lines began to shift upslope with a warming climate. Additional evidence, such as dated samples of subfossil wood above modern tree line, indicated the tree line on Bald Mountain was approximately 60 meters higher prior to the Little Ice Age, approximately 1550 (Munroe 2003). Munroe concluded "a higher treeline in the northern Uintas, shortly before A.D. 1550, is consistent with contemporaneous evidence for warmer-than-modern climates in the southwestern United States."

Current Status and Trends

Alpine community structure is currently represented on the landscape consistent with existing geomorphic and climatic features that were identified under the natural range of variation. This is verified by the many long-term studies that describe, classify, and determine condition and trend of alpine communities found in the Uinta Mountains (Brown 2006, Studies 10-10B, 11-14H, 16-1E, 16-20B2, 16-20DROC, 16-20G1, 16-20J, 16-20L, 16-20M, 16-20N, 16-20NET, 23-20C1, 23-21, 28-45D, 28-45E, 37-7F Goodrich and others 2005). Alpine communities show long-term stability within structural parameters described for the natural range of variation, but a couple of trends are noted. First, increase in density and canopy cover of low willow in many alpine communities, both wet and dry, has been documented for at least 50 years (Goodrich and others 2006). These increases have occurred concurrent with livestock grazing and wild ungulate browsing. Also, an increase of and gradual displacement by conifers in low willow communities, mesic meadows, and a few riparian ecotones at timberline has been documented (Munroe 2003, Studies 11-2B, 11-2C1, 12-22, 23-3A, 23-4B). During the next plan period, continued increase in low willow and conifer is predicted to occur.

Influences of Drivers and Stressors

Long-term monitoring has detected increases in density and canopy cover of low willow in many alpine communities. Also, conifer tree lines show upslope movement since 1870. These trends may be interrelated. Munroe (2003) suggested that, following the conclusion of the Little Ice Age (ca. 1850), climate warming trends in the Uinta Mountains led to increased timberline tree densities and upslope expansion of Engelmann spruce and subalpine fir. Similarly, warming climates may influence the upward trend of low willows in alpine settings. Additionally, new tree establishment at or near timberline consistently occurs in low willow communities where low willows have increased. This suggests that low willows create a favorable microenvironment for conifers (Goodrich and others 2015, Studies 24-15A-B, 11-2A-C1, 11-2E, 11-2E9, 11-2F, 11-2G2).

Wild ungulates are capable of impacting some alpine communities including riparian, mesic, and dry meadow communities, especially those that have low willows. Elk, moose, and mountain goat populations have increased over the 30 years. On the South Slope herd unit, elk numbers have increased from 7,470 animals in 2004 to 8,700 animals in 2010 (UDWR 2012). Moose populations show an upward trend in nearly all management units in, or adjacent to, the plan area (UDWR 2012). Mountain goats were first introduced to the Uinta Mountains in the 1980s. Goat populations have increased fourfold (215 to 858 animals) from 2001 to 2011, but are currently at 49 percent of the State's management objective. Low willow communities appear to be most susceptible to ungulate browsers. Elk and moose have diminished shrub canopies in a few low willow subalpine communities, but no decreases in alpine communities have been documented (Studies 14-43B, 14-43C1-C4, Goodrich and others 2015). Steady upward trends in wild ungulate populations is considered a potential stressor of alpine communities, particularly low willows.

Human-related impacts in alpine environments have been relatively limited, due to remoteness and harsh conditions associated with high elevations. Human impacts have not influenced or changed vegetation structure during the last 30 years. Increased recreation use is predicted during the next plan period.

Livestock grazing has occurred in mostly mesic, turf, and dry meadow alpine communities over 100 year period, but has minimally affected community structure, including low willows. Low willows have increased over the last 30 years concurrent with livestock grazing. Domestic sheep grazing has declined over the last 30 years, but cattle grazing has had modest increases in both area and numbers in alpine communities. Livestock grazing is not expected to increase in alpine areas during the next plan period. Current livestock grazing may impact vegetation structure during the next plan period but not sufficient to move plant communities outside the natural range of variation.

Climate Related Risks and Trends

Climate change may be considered either an ecological stressor or driver in alpine plant communities. If the climate becomes warmer, low willow may continue to increase and coniferous forests may continue moving upslope. On the other hand, if the climate begins to cool, tree lines and subalpine vegetation may experience downslope migration as discussed by Munroe (2003). Environmental conditions are drivers that determine plant species composition, ground cover, and disturbances that impact vegetation.

Warming climates over the last 150 years have shifted tree lines upslope and forests near timberline have become denser. Also, low willow communities have increased in canopy cover, density, and size over the last few decades. Changes in low willow communities are likely influenced by a warming climate. If a warming climate continues, gradual upward migration of subalpine communities is predicted to occur, however sub-alpine communities may be topographically limited to upslope migration. If climates become cooler, gradual downward migration of plant communities are predicted. These trends are considered currently within the natural range of variation.

Comparison of the Natural Range of Variation and Current Conditions

Current conditions of the structure of alpine plant communities in the plan area closely align with the natural range of variation.

Aspen

Description of the Natural Range of Variation

Aspen occurs on montane landscapes in the form of small, isolated stands near the fringe of their range to, "broad expanses of pure and mixed stands" of upper elevations (Mueggler 1988). Stands range in size from less than a tenth of an acre to dozens of acres. Extensive aspen forests occur where stands of close proximity have coalesced over time. Tree structure, height, and density vary from stand to stand. Some aspen stands consists of a single cohort or height or age class of trees, while others consist of two to multiple cohorts. Tree heights range from 20 to 70 feet, but seldom exceed 50 feet on the Ashley National Forest (Mueggler and Campbell 1986). Stand structure is greatly influenced by community function and condition (Kurzel and others 2007). In regards to structure and function, two general types of aspen are recognized: seral and persistent. Seral aspen is the most common aspen type on the Ashley National Forest. These communities consist of a coniferous component that functions as the late-successional dominant. Seral aspen typically occurs at high elevation zones where aspen and conifer are cohabitants (Romme and others 2009, Mueggler and Campbell 1986). Aspen is known to grow with lodgepole pine, Engelmann spruce, sub-alpine fir, Douglas fir, ponderosa pine, and blue spruce (Mueggler 1988). In seral communities, aspen dominates crown cover, following "coarse-scale disturbances" that remove conifer dominance (Kurzel and others 2007, Romme and others 2009). Aspen dominance may last several decades as conifer species such as sub-alpine fir, Douglas fir, and Engelmann spruce slowly return. As conifer species increase, a gradual shift eventually occurs as reestablished conifers increase in density, height, and cover. In contrast, lodgepole pine and aspen regenerate almost simultaneously following disturbance. Aspen dominance in this setting is relatively short (Study 30-7). As conifers increase in dominance, aspen usually decreases in "density, basal area, or cover over time." But this "natural successional process" is considered cyclical where disturbance events may span from several decades to centuries (Romme and others 2009). Romme and others (2009) determined increasing dominance of conifer during long absences of fire, "does not pose a serious threat to long-term persistence or ecological function of aspen." Kurzel and others (2007) found "signs of aspen self-replacement despite conifer competition." They determined that some self-replacing seral aspen may exist with conifers for decades, "with no signs of emerging dominance by either species," and that the role of fire "may be to eliminate competition rather than serve as a necessary stimulus for aspen regeneration."

Aspen functions as the late-successional dominant in persistent aspen communities. Conifers are absent or nearly absent in these settings because persistent aspen mostly occurs below the conifer belt (Romme and others 2009). Community dynamics are influenced by genetic variability, environmental conditions, and disturbance mechanisms (Mueggler 1988, Romme and others 2009). Kurzel and others (2007) described several modes of regeneration that facilitates aspen's ability to persist on the landscape. These are stand replacing, continuous, and episodic. The expression of each mode is in response to community function or condition and each mode influences community structure.

Stand-replacing fire is the most common and important form of stand-replacing regeneration (Romme and others 2009). Tree die-off from fire initiates sprouting within one to two years following disturbance, which initiates a single-age cohort. Jones and DeByle (1985) suggested "almost all even-aged aspen stands in the West appear to be the result of severe fire." Many persistent aspen stands on the Ashley National Forest consist of one cohort, indicating one sprouting episode since the last disturbance event. Historically and presently, wildfires have burned areas where persistent aspen is established (Stewart

1911, Hitchcock 1910, Graham 1937, Heyerdahl 2011, Ogle and Dumond 1997, Huber and Goodrich 2016).

Continuous regeneration is considered a relatively minor form of aspen regeneration, since few clones exhibit this feature (Studies 30-24E-F, 40-1, Huber and Goodrich 2016). Aspen clones with this capability perpetually produce new sprouts regardless of disturbance or stress. These communities are multi-tiered, and continuous sprouting is perhaps “due to the relatively high-light environment” of open aspen canopies (Kurzel and others 2007).

Episodic regeneration is likely the least understood mode of regeneration. Aspen is capable of self-replacement when coarse-scale disturbance does not occur over the life span of an aspen cohort. Most persistent aspen consist of a single cohort because apical dominance suppresses new sprout abundance and survival (Schier and others 1985). Apical dominance may persist for decades until cohort die-back takes place. Successful pulses of new sprouting occur when apical dominance is weakened or removed during “periods of widespread die-back” (Kurzel and others 2007). Aspen die-back mostly occurs in stands that are in stages of senescence. New sprouting initiates a new cohort that ultimately replaces the senescing cohort. Kurzel and others (2007) also found that “recent cohorts are occurring roughly 100 years after the initial cohorts, a duration that is the same length as the average aspen life span.” Kurzel and others (2007) state that, “episodic regeneration is the main mode of stand re-initiation and . . . that at a landscape scale this may be the most widespread mode of aspen regeneration.”

Photo of study 32-86K. Little Brush Creek. 2015. View of single cohort aspen clone where apical dominance suppresses new sprout abundance and survival. Livestock grazing is absent in this setting.

Current Status and Trends

Aspen communities on the Ashley National Forest have been subject to numerous stressors, disturbances, and management practices during the 30 years. However, distribution, abundance, structure, and function of aspen have been relatively constant and within expected the natural range of variation parameters. Aspen continues to be represented across the landscape within its natural elevation and topographical range. These conclusions are verified by the numerous long-term studies that monitor condition and trend of aspen communities found on the Ashley National Forest (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016).

Current monitoring indicates that seral aspen is diminishing in terms of density, basal area, and cover within the plan area and is being displaced by coniferous trees on the Ashley National Forest. There may be potential for aspen loss under current broad-scale fire suppression policy and continued reduction in timber harvest. In contrast, Romme and others (2009) viewed increases, “in conifer abundance as natural successional processes that have always occurred during long periods without major disturbance.” Considered in terms of succession, seral aspen may not be declining at a landscape scale, due to its ability to persist under the dominance of conifer for long periods of time. Successful regeneration of seral aspen continues to occur following fire or timber harvest under conifer-dominant conditions, even after a 100 years of fire suppression policy (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016). Several examples of successful aspen recruitment following recent stand replacing fire includes the Weyman, Iron Mine, Cow Hollow, Bumper Canyon, Spring, Water Hollow, Church Camp, and Neola North Wildfires (Studies 34-5B-C, 34-6, 58-58A-C,F, 59-15A-C, 64-52B,E,G, 58-28A-C,F-G, 66-15F,H-I,M-N, Goodrich and Huber 2008, Zobell and Goodrich 2015). Seral aspen appears to be functioning in or near the natural range of variation, but continued monitoring during the next plan period should provide additional information regarding the effects of fire suppression and reduced timber harvest in seral aspen communities.

Photos of study 64-52G. Reservation Ridge. 2011 and 2015. View of successful aspen regeneration in Douglas fir and aspen stand following wildfire.

During the last 30 years, nearly all persistent aspen communities demonstrated resilience concurrent with contemporary stressors and drivers. Successful regeneration of persistent aspen has been documented through stand-replacing, continuous, and episodic regeneration pathways. Numerous stands of persistent aspen have burned with wild or prescribed fire, harvested for wood products, or clearcut for other purposes over the last 30 to 40 years (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016). In nearly every instance, new sprouting occurred and sprout survival was sufficient to perpetuate the long-term viability of aspen. Within five years post-fire, sprouts have routinely exceeded six feet in height, with 50 percent canopy cover or greater. This indicates successful aspen recruitment of burned stands. Additionally, new sprouting in many burned aspen stands exceeded their pre-burn perimeters by five to 50 feet, indicating stand expansion. Most clear cuts or harvests exhibited sprouting similar to burned aspen stands, but a couple of treatments failed to start sprouting due to their small size (less than 5 acres), other impacts, or both (Studies 41-4F, 43-25A, 64-20A, 64-33). Most treatments occurred within or along the perimeter of larger persistent aspen stands. In conclusion, aspen response to coarse-scale disturbances is within parameters described for the natural range of variation.

Study 58-14. Slab Canyon. Reservation Ridge. 1980, 1998, and 2011. Depiction of successful episodic regeneration of small aspen clone. New sprouting occurred beyond the perimeter of 1980 stand leading to the expansion of the clone.

Very few persistent aspen clones on the Ashley National Forest demonstrate the ability to perpetually regenerate. Clones with this capacity consist of multiple age or height classes of aspen, where height classes of trees range from six to 20 feet in height, under a canopy of mature trees and many saplings, and young trees appear to be trending towards maturity (Huber and Goodrich 2016, Studies 30-24E-F, 40-1). Aspen self-replacing under the continuous regeneration mode appears to be functioning within the natural range of variation in regards to successful recruitment and multi-tiered structure.

In the absence of stand-replacing disturbances, pulses of abundant and successful sprouting occurred in persistent aspen following die-back of the initial cohort. Numerous events of episodic regeneration are documented and most occurred after 2000. Die-back or die-off of persistent aspen accelerated during and after severe droughts in 2002, 2012 and 2013 (Huber and Goodrich 2016, Goodrich and Huber 2015b). Some aspen stands experienced nearly 100 percent mortality of the initial cohort, followed with successful sprouting events (Studies 32-17D, 32-31D). Most stands experienced only partial die-back of the canopy with similar results (Huber and Goodrich 2016, Studies 31-40B, 32-67C-D, 3-23J, 64-11E, 57-3B, 68-66F). In many instances, new sprouting exceeded the perimeter of the original clone by up to 50 feet, resulting in stand expansion. Similar to burned aspen stands, sprouts have routinely exceeded six feet in height, within three to five years after the sprouting event. Episodic regeneration in dying persistent aspen stands has been successful in terms of recruitment, which indicates these communities are functioning within the natural range of variation.

Influences of Drivers and Stressors

An important ecological feature of aspen structure is that it benefits from disturbances and other die-back mechanisms that cause tree mortality. Die-off or die-back of aspen usually triggers new sprouting at levels that ensure its persistence or dominance of a site. The most recognized and understood driver of aspen communities is fire. Numerous stands of persistent and seral aspen, both small and large in size, have burned with wild and prescribed fire over the last 30 to 40 years (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016). Successful sprouting following fire has routinely occurred concurrent with other drivers and stressors of aspen. Size of aspen stands often increase following fire because new sprouting often occurs beyond the perimeter of the original stand. Most fires have been less than 200 acres in size, affecting less than a 100 acres of aspen in one event, but there have been several larger fires that have burned hundreds of acres of aspen in one event. These fires include the Weyman, Neola North, and Church Camp wildfires and the Petty Mountain prescribed fire (Zobell and Goodrich 2015, Goodrich and Huber 2008, Studies 3-9A, 3-17A,D, 38-58A-B,E-F, 66-15F,H-I,M-N). Historical accounts revealed that stand replacing fire occurred periodically in the Uinta Mountains, both at large and small scales. For example, S.S. Stewart surveyed the east end of the Ashley National Forest in 1909. His subsequent report listed burned areas for Cart Creek, Little Brush Creek, Big Brush Creek, an area between Sims Peak and Taylor Mountain, Mosby Mountain, and lower portions of Dry Fork (Stewart 1911). Stewart reported one fire in 1879 that covered the greater part of four townships in the southeast portion of the forest, and that smaller fires subsequently burned over much of the same area. The 1879 burn area was estimated to be about 24 miles long, an area that spans from about Mosby Mountain eastward to the forest boundary on the Diamond Mountain Plateau. Other reports of historic fires include the Uinta and Pole Creek drainages and large tracts of forest in Lake Fork and Dry Gulch drainages (Hitchcock 1910). Furthermore, Graham (1937) included the following comment about the Wheeler Expedition of 1871. When this expedition reached the Uinta Mountains "the full complement of topographical work was prevented by forest fires of great extent" (Graham citing Wheeler 1889 page 657). This is the same year Heyerdahl and others (2011) documented a large fire in the Brownie Canyon - Dry Fork Canyon area.

Wildfire occurrence is strongly related to environmental and climatic conditions. If the climate continues to warm, fire frequency is predicted to increase. On the other hand, if the climate cools, fire frequency is predicted to decrease. The current rate of fire is not likely to induce an upward trend in seral aspen acres or area in the plan area. Increased fire frequency in a warmer climate is predicted to benefit seral aspen communities. In contrast, persistent aspen is expected to be sustained or show increases under current fire return intervals, also with increased or decreased fire frequency; coupled with other disturbance or die-back events. If fire frequency is 20 years or less, aspen may be negatively affected. Stand mortality may occur if fire burns aspen sprouts or very young trees.

A human-caused disturbance that parallels the effects of fire is timber harvest. During the last 30 years, hundreds of acres of seral aspen have initiated new sprouting following timber harvests of conifer trees (Goodrich 2015b). Persistence of aspen in these settings have been sustained through timber harvesting in conifer-aspen forest types. Timber harvesting has trended downward during the last 30 years and is expected to continue downward or level off during the next plan period. Future effects of timber harvesting on seral aspen is dependent upon this trend.

“Cohort senescence” is a condition that makes persistent aspen more susceptible to insects, disease, or drought, which are factors commonly attributed to aspen die-back (Kurzel and others 2007). Die-back processes are better understood when stand age of aspen is considered. With the absence of stand replacing disturbance, aspen trees or cohorts begin to senesce or become decadent after about 100 years. Senescing cohorts lose vigor and are less resistance to disease, insects, and drought. The cohorts ultimately succumb to these stressors or old age. Die-back of senescing aspen is well documented within the plan area, with most recent die-backs occurring since 2000 (Goodrich and Huber 2015b, Huber and Goodrich 2016). Die-back events increased during severe droughts in 2002 and 2012. Loss in apical dominance of canopy trees, due to senescence and subsequent die-back, resulted in increased and abundant sprouting in affected aspen stands. Additionally, new sprouting occurred beyond the perimeter of dying cohorts in most stands, which indicates aspen expansion. Episodic regeneration is common in persistent aspen within the plan area and is expected to continue to occur during the next plan period, as aspen clones phase into old age and begin to senesce. Persistent aspen is expected to be sustained or show modest increases in area under processes of senescence and die-back, coupled with occasional stand replacing fire.

Aspen communities are used for forage and cover by both domestic and wild ungulates. Livestock grazing has been a long-term and enduring use in aspen for more than a 100 years. Little to no information is available pertaining to livestock effects on aspen recruitment or structure prior to the 1950s, but repeat photography indicates that aspen persisted during decades of heavy grazing by cattle and domestic sheep. During and prior to the last plan period, successful recruitment of aspen, following stand replacing fire and periodic die-back, has occurred throughout the plan area, concurrent with the presence of livestock (Huber and Goodrich 2015, Zobell and Goodrich 2015, Goodrich and Huber 2015, Huber and Goodrich 2015b, Goodrich and Huber 2015b, Huber and Goodrich 2016). Additionally, successful aspen recruitment occurred within allotments where few to no changes in stocking rates, seasons of use, and management strategies (temporary or permanent), were implemented following fire events. Livestock browsing of aspen sprouts has been minimal and not sufficient to affect successful recruitment, stand structure, or diminish aspen persistence. Livestock grazing in the terms of numbers, class of livestock, and management is expected to remain relatively constant during the next plan period. Persistent aspen is expected to be sustained and successful aspen recruitment is expected to occur concurrent with contemporary livestock stocking rates and management strategies.

Elk and mule deer have impacted some aspen historically and recently. Elk and mule deer populations have shown reverse trends over the last 60 years. Prior to 1960, elk populations were very low, while mule deer populations were high. Unlike other areas in the West, there is no information indicating that mule deer suppressed aspen recruitment or diminished persistence during that time period - although that may have occurred (Romme and others 2009). Ultimately, aspen has persisted concurrent with high mule deer populations. The only notable impacts during this time period are high-lining of some aspen by mule deer. During the last 30 years, mule deer populations have decreased and elk populations have increased considerably. Numerous studies indicate that current elk populations have minimal impact on aspen recruitment following fire, timber harvest, or die-back in most stands within the plan area. For the most part, elk browsing of aspen sprouts is not sufficient to suppress successful recruitment or diminish stand persistence. One exception is Anthro Mountain. Due to the limited number of acres of aspen on the Anthro Plateau landtype association, aspen is more susceptible to elk browsing than on other aspen-bearing landtype associations. Following a 1996 prescribed fire treatment, elk browsing of new sprouts severely curtailed recruitment, thinned stand structure, and threatened aspen persistence (Huber and Goodrich 2015, Study 68-7A). Elk populations are predicted to increase during the next plan period. If populations continue an upward trend, more aspen stands within the plan area would be susceptible to elk browsing following disturbances. This trend may threaten successful aspen recruitment, alter stand structure, and diminish aspen persistence. Upward trend in elk populations is considered a potential stressor of aspen communities.

Recreation is a minor stressor of the structure of aspen communities. Recreation use such as camping, hiking, and all-terrain vehicle use have had minimal effects on aspen overall and impacts are site specific. Some mortality of mature trees or new sprouts have occurred at some dispersed camping sites in aspen, but the use is not sufficient nor extensive enough to impact regeneration processes or diminish persistence of aspen. Increased recreation use of the Ashley National Forest is predicted during the next plan period, but this trend is not expected to adversely affect aspen recruitment, structure, or persistence within the plan area.

Climate Related Risks and Trends

Climate change is a potential driver of aspen structure. Seral aspen may benefit from a warmer and drier climate. The frequency and extent of fires would increase in montane forests, including conifer-aspen communities, where conditions become consistently warmer and drier over time (Rice and others 2016). More frequent and larger fires would move conifer-aspen communities toward early seral and mid-seral stages, which would favor greater aspen dominance of these communities. Persistent aspen may also benefit from a warming climate. The extent of these communities may increase as conifer communities migrate upslope under warming climates, if annual precipitation remains within its range of variability. Higher fire frequency would likely reduce the number of older-age, senescing aspen stands. Higher frequency could also perhaps improve clone vigor and health, with more frequent cohort turnover. Since stand expansion often occurs with new sprouting, frequent fires would increase total area of persistent aspen over time. On the other hand, if the changing climate becomes drier and precipitation falls below required water needs, affected persistent aspen may be displaced by other communities, such as mountain big sagebrush. In this case, upslope migration of persistent aspen must occur to maintain these communities and/or seral aspen becomes persistent as conifers die-off and these communities migrate upslope. In addition, aspen mortality may occur if a warmer climate induces fire intervals to 20 years or less (Romme and others 2009). If a cooler and wetter climatic trends occur, seral aspen would be adversely affected. Less frequent and smaller fires resulting from cool and wet conditions would favor late-seral conifer dominance, with loss of seral aspen over time. Lower fire frequency would likely increase the number of older-age persistent aspen stands and episodic regeneration from senescing aspen would become increasingly common. Some downslope migration of conifers may occur and establish within higher elevation persistent aspen, but these communities are expected to persist within a favorable climatic zone. In conclusion, long-term monitoring has not detected change in aspen recruitment, structure, persistence, or elevational movement directly related to current trends in climate.

Comparison of the Natural Range of Variation and Current Conditions and Trends

Current monitoring indicates seral aspen is diminishing in the plan area in terms of “decrease in aspen density, basal area, or cover” because the frequency and extent of fires have been reduced over the last 100 years (Romme and others 2009). Consequently, seral aspen is of moderate departure from the natural range of variation. Natural range of variation indicates a higher fire frequency, with larger fires similar to those documented during the 1870s. Large and small scale fires and timber harvest has temporarily removed conifers at a number of seral aspen sites, but their occurrence is not sufficient enough to maintain seral aspen long-term. If the climate continues to warm and become drier, fire frequency and size is predicted to increase, which would be beneficial to seral aspen. More conifer-aspen communities would transition to early and mid-seral stages from increased stand replacing fire.

Persistent aspen has persisted on the landscape at levels equal to or greater than at the beginning of the last 30 years. This indicates that persistent aspen is within or of low departure from the natural range of variation. Successful regeneration of these communities has occurred under stand-replacing fire, continual, and episodic regeneration processes. Successful recruitment following stand replacing fire are documented at numerous sites, with very few failures. Many persistent aspen stands experienced die-back following a severe drought in 2002. Abundant sprouting in affected stands began a year later and continues today. Successful episodic regeneration has occurred in the absence of stand-replacing fire but concurrent with other known stressors such as insects, disease, wild ungulate browsing, and livestock grazing. Many persistent aspen stands show expansion due to new sprouting outside the perimeter of the dying cohort (Studies 31-14D, 32-86C, 32-86I1, 34-6A, 34-17H, 37-28C, 39-32B, 41-4B3, 41-4B4, 50-6, 57-3B, and 68-66F). In relatively few cases, moderate to heavy ungulate grazing has impacted aspen regeneration (Studies 31-14C, 32-5, 34-11C, 34-11D, 39-17Q, 43-25A, 68-7B). However, long-term monitoring demonstrates aspen sustainability in the plan area concurrent with existing levels of permitted livestock grazing and wild ungulate use. Permitted livestock use is expected to remain relatively unchanged, but elk numbers are expected to increase during the next plan period. Although current elk populations present little risk to sustainability of aspen at a landscape level, upward trend in populations may curtail future regeneration of many persistent aspen stands. If climates get warmer and drier, persistent aspen may experience higher fire frequencies, community shifts upslope, and displacement of lower elevation stands by shrub and grass communities. Long-term monitoring has not detected change in aspen recruitment, persistence, or elevational movement directly related to current trends in climate. In conclusion, existing conditions and current trends indicate that persistent aspen is within or of low departure from the natural range of variation. These conditions and trends are expected to remain so during the next plan period until long-term monitoring indicates otherwise.

Coniferous Forest

Ponderosa Pine

Description of the Natural Range of Variation

Under ponderosa pine’s historical non-lethal fire regime, structure characteristics are largely expressed by the fire return interval for this species, which is described as about 5 to 45 years (USDA 2009). Periodic fires can create all-aged structure or uneven-aged stands comprised of a mosaic of various even-aged groups, periodically thinning the understory and removing patches of seedlings (Bradley 1992, Fiedler and others 1995, Hood 2010). Specifically, the uneven-aged structure of ponderosa pine can be described using a *q* factor. The value *q* is the ratio between trees in successive diameter classes and is used to describe the diameter distribution of an uneven age stand. For example, a *q* factor of 1.1 to 1.2 for two-inch-diameter classes allocates a larger proportion of the basal area to large trees (Fiedler and others 1988, USDA 1998, USDA 2009) typical of ponderosa pine. Frequent surface fires “kept most stands in an open park-like condition dominated by large old trees,” as summarized by Fiedler and others (1995), and stand densities generally do not exceed about 140 ft^2 basal area (USDA 2009, USDA 1990). Basal areas that are at lower densities (40 to 60 square feet per acre) ensure regeneration of shade-intolerant ponderosa pine (Fiedler and others 1988).

Frequent fire may also influence the incidence of insects and disease damage as natural drivers of structure in ponderosa pine. Ponderosa pine has a history of mountain pine beetle epidemics. Trees are characteristically killed in groups—primarily in dense, over-stocked stands or clumps of pure, even-aged pines. For ponderosa pine and its susceptibility to mountain pine beetle attack, tree diameter is not as significant as tree densities (USDA 2009, USDA 1990, Olsen and others 1996). Effects of fire may include an increased resistance to bark beetle attack. “Increased resin production is thought to be a measure of a tree’s resistance to bark beetle attacks. Many studies have found [significantly] increased resin production after burning” (Hood 2010). In addition, frequent surface fires may also help “control dwarf mistletoe (*Arceuthobium campylopodum* and *A. vaginatum*) infestations by pruning back infected branches in the lower crowns” (Bradley 1992). This parasitic plant can cause growth loss in all age classes.

Current Status and Trends

The Greendale Plateau supports the Ashley National Forest’s largest, mostly continuous ponderosa pine forest of the eastern Uinta Mountains. Ponderosa pine forms a more or less continuous belt on the eastern side of the Greendale Plateau landtype association, from near Ute Lookout Tower to the eastern end of the Ashley National Forest. Ponderosa pine forest occupies 43 percent of this landtype association, while an additional six percent is seral aspen to ponderosa pine. The historic fire return interval for the Greendale Plateau is supported by a site-specific study by Palmer (1993). Palmer determined a mean fire return interval of about 21 years for the ponderosa pine belt of the Greendale area, based on a number of fire-scarred stumps. This fire return interval is consistent with the ranges reported in several other studies (Bradley and others 1992).

Much of this landtype association has received some sort of past treatment. Historical documents describe much of the Greendale Plateau area as being salvage harvested in the 1920s and again in the 1950s in order to remove trees killed by mountain pine beetle epidemics (USDA 2005). In the more recent recorded history of harvests at the stand scale, about 60 percent of the ponderosa pine on this landtype association has been timber harvested since about the 1980s. These harvests were mostly salvage logged due to a mountain pine beetle epidemic at that time. In addition, salvage harvest has taken place along Highway 44, due to mortality associated with salt damage, which is an ongoing problem. Only one percent of timber harvest recorded on this landtype association was clearcutting harvest, which results in even-aged condition. In addition, about 50 to 75 percent of the ponderosa pine in the Greendale Plateau landtype association has been treated in the past 20 years by prescribed burn or other fuel reduction treatment.

In 2010, yet another mountain pine beetle epidemic in the adjacent lodgepole pine belt, on this same landtype association, made its way into the ponderosa pine belt. Primarily dense pockets of ponderosa pine were attacked and killed. Compared to the epidemic in the 1980s, the damage to the ponderosa pine was not as severe, indicating some resistance to spread of disturbance.

Some of the past prescribed burn treatments and timber harvesting overlapped. This resulted in at least 20 percent of the remaining area having no record of having been harvested, mechanically thinned, prescribed burned, or otherwise treated in recent history. Stands in this area are likely departed from the natural range of variation. Forest managers continue to identify areas that need prescribed under burning treatment or timber harvest entries (USDA 2013).

Low intensity prescribed burns have reduced available fuels in this type to levels that are closer to the natural range of variation. However, fire has likely been suppressed in the ponderosa pine type for the past 100 years. There is still a moderate degree of departure from the natural range of variation in ponderosa stands, due to a structure that includes high densities in the younger tree layers (GreendalePlateau.ppt, USDA 2015, USDA 2017). Conversely, although larger trees are present, stands have few large-diameter trees. This type of uneven aged distribution, where few large trees are present, is inconsistent with expected densities in ponderosa pine stands. For example, tree data from permanent plot transects in the Red Canyon area of the Greendale Plateau landtype association show *q* factors as high as 1.6. This indicates a disproportionate weight to smaller trees that is uncharacteristic of healthy ponderosa pine stands. This area had missed one or more fire return intervals. On the other hand, tree data from permanent plot transects in the Dowd Mountain area of the same landtype association show *q* factors of 1.1 to 1.3, indicating structures that are characteristic of healthy ponderosa pine. This entire area had been salvage harvested in the 1980s and had been partially underburned in the 1990s. In areas that missed one or more fire return intervals or other form of disturbance, the return of disturbance is not expected to produce the desired density of trees or distribution of tree sizes in a short time period. Thick growth of ponderosa pine after decades of fuel accumulation, makes prescribed burning difficult without losing an unacceptable number of older ponderosa pine trees (Fiedler and others 1995). This is mitigated by implementing relatively low intensity under burning.

Site-specific fire studies have not been conducted on the other landtype associations where ponderosa pine is present. However, the fire return interval under the natural range of variation can be inferred by reference to that reported in Palmer (1993) and other studies (Bradley and others 1992) for this type.

The Dry Moraine landtype association also supports a large percentage of the ponderosa pine type. Ponderosa pine forest occupies 24 percent of the landtype association, while an additional 14 percent is seral aspen to ponderosa pine. The bark beetle epidemic of the 1980s affected the entire ponderosa pine belt of the Dry Moraine landtype association. The result of this beetle epidemic was a high percent mortality of larger trees. Larger trees that survived are found scattered throughout some stands, but the trees are hardly of sufficient density or basal area to be a dominate feature of these stands. Relatively young trees can now dominate these stands. As an example, stand exam in the Rock Creek drainage of this landtype association revealed a ponderosa pine structure weighted heavily to saplings with a q-factor of 1.5 pre-treatment, having missed one or more fire return intervals. Some areas have been managed to retain larger trees through prescribed under-burning. Within the past 20 years, nearly 90 percent of the ponderosa pine type of this landtype association, including the Rock Creek area, has been under-burned. However, in other places, potential ladder fuels such as understory shrubs including manzanita, Rocky Mountain juniper, and regeneration of ponderosa pine saplings are at levels that could lead to stand-replacing fires.

Salvage harvest followed the bark beetle epidemic of the 1980s. More than 50 percent of the ponderosa pine type of this landtype association has had some harvest. Fuel loadings of understory vegetation in the Mud Springs and adjacent Dry Gulch area, before fire ranged from two to 14 tons per acre when measured in 2001, are well within the range reported in other studies (Bradley and others 1992) for this type.

The Stream Pediment landtype association is another large supporter of ponderosa pine forest. Ponderosa pine forest occupies 34 percent of the landtype association, while an additional 13 percent is seral aspen to ponderosa pine. Like the other landtype associations, insect epidemics are an inherent disturbance process on this landtype association, and a major factor affecting vegetation structure. The massive mountain pine beetle epidemics of the 1920s and 1980s are evidence of this. The magnitude and subsequent recovery from these events indicates that the vegetation on this landtype association has been resilient enough to recover from mountain pine beetle epidemics in the past.

Perhaps as much as 80 percent of the ponderosa pine on this landtype association has been salvaged logged, due to the pine beetle epidemic of the 1980s. Most of the heaviest 1,000-hour time lag fuels loadings were slashed and burned at this time. Since this time, extensive under burns have occurred, removing much of the smaller diameter woody debris. Remaining fuels are needle beds and naturally thinned smaller diameter branch wood.

As in other landtype associations, regeneration can be at high densities, contributing to ladder fuels to the main canopies of the larger, older ponderosa. Instead of being pruned by fire, younger trees generally develop limbs extending to near ground level. As a result, these younger trees with low limbs contribute to ladder fuels that enable fire to affect the upper canopy of these stands in the event of a fire (Study 40-

15K). Some efforts were made starting in the 1990s to mechanically thin ponderosa pine saplings, to reduce this ladder fuel on this landtype association.

Climate Related Risks and Trends

A warmer climate could increase stress and mortality, due to bark beetles. The Intermountain Adaptation Partners (2016) stated, "Increased frequency and magnitude of drought, and consistently drier soils will cause ponderosa pine to grow slower, but mortality will be rare unless drought lasts for several consecutive years and bark beetles cause additional stress… If bark beetles become more prevalent in a warmer climate, they could increase stress and mortality in pine species, especially during drought periods."

Comparison of the Natural Range of Variation and Current Conditions

Although fire has been recently reintroduced, fires have otherwise been infrequent in the ponderosa pine forest over the last 100 years. Although some areas have an uneven-aged structure that is closer to historical, the larger tree component has become deficient in general. Other areas have an uneven-aged structure uncharacteristic of ponderosa pine, where saplings are greater in number than they should be. Fewer large trees are largely due to mountain pine beetle outbreaks that have targeted dense stands. These stands have likely missed a fire disturbance that would have otherwise kept tree density lower historically and less susceptible to bark beetle attack. Although the return of fire is trending the structure in this type back to historic conditions, there is still a moderate degree of departure from the natural range of variation.

Lodgepole Pine

Description of the Natural Range of Variation

Lodgepole pine (*Pinus contorta*) is typically an early seral tree species that ranges over extensive areas in the Intermountain Region. The fire regime consists of both low and high severity fires. Fire frequency is 35 to 200 years and fires are generally high severity and stand replacing (Landfire 2012). Low severity occurs at more frequent intervals where fires smolder and creep in the litter and duff (Bradley 1992). High severity occurs at less frequent intervals as crown fires. The latter is more common in the lodgepole pine type on the Ashley National Forest.

Prior to 2009, most lodgepole pine forests in the Intermountain Region were in the mature and old structure classes, except for harvested and wildfire-burned areas (USDA 2009). Since then, recent widescale mountain pine beetle outbreaks in the Intermountain Region, as on the Ashley National Forest, have altered this condition. With the impact of insects and fire, large fluctuations in the distribution of structural classes are more common than a balanced distribution in lodgepole pine (USDA 2009).

Depending on maturation of lodgepole pine at a landscape scale, large epidemics of mountain pine beetles are to be expected. When lodgepole pine stems reach about eight inches in diameter at breast height (DBH), mature stands become highly vulnerable to mountain pine beetles. Lodgepole pine has a history of extensive mountain pine beetle epidemics at elevations generally below 9,600 feet (USDA 2009). "Trees greater than eight inches DBH with stand densities exceeding 120 square feet of basal area per acre are the most susceptible to mountain pine beetle attack" (USDA 2009).

The classic natural range of variation pathway for the lodgepole pine forest is that it grows, beetles attack it, it burns, and it grows again. Therefore, a sustainable and resilient condition for lodgepole pine types includes a mosaic of large patch sizes of even-aged stands of lodgepole pine. Where there is potential for aspen, these lodgepole pine stands would include aspen. This concept is consistent with the dynamics inherent to this lodgepole pine system.

Dwarf mistletoe (*Arceuthobium americanum*) is the most common and damaging disease in lodgepole pine. This parasitic plant affects growth in all age classes, causing significant deformation and loss of branches (van der Kamp and Hawksworth 1985; Hawksworth and Johnson 1989). Diseased trees have increased mortality rates. If trees are infected as saplings, there is significant growth loss (USDA 2009). Large stand-replacing fires often remove stands of mistletoe-infected lodgepole pine (Bradley and others 1992).

Current Status and Trends

Portions of the lodgepole pine type have been harvested since the 1960s. Early harvests were primarily clearcuts. Machine piling of slash often followed these harvests. Harvested areas regenerated well to high densities. Thinning of seedlings and saplings in some of these young stands has maintained, or in some cases, increased individual tree growth. The unthinned stands are becoming overstocked; individual tree growth is starting to slow due to the high stand densities. Overall, growth rates of lodgepole pine on the Ashley National Forest is low to moderate. Seral stages of lodgepole pine tree structure are similar to those of most lodgepole pine dominated systems and are described in appendix A.

On the Greendale Plateau landtype association, structure of vegetation—including size, shape, and patterns—is highly influenced by insects and fire (Studies 2-21, 16-31A). Mountain pine beetle caused tree mortality has been very heavy on this landtype association in the 1980s, and recently from 2008 through 2012. Evidence of historical and recent large, stand-replacement fires is also ubiquitous in the lodgepole pine belt of this landtype association. Stands of lodgepole pine, with a high percentage of closed or serotinous cones, indicate a highly fire-adapted system for this landscape. This relatively high level of serotiny also indicates relatively short fire return intervals, perhaps as short as 50 years (Bradley and others 1992).

Stand structure distributions on this landtype association are weighted more heavily to the smaller structure stages than to the larger. This result is due to relatively recent fire, insect, and harvest disturbance. Lodgepole pine stands in the seedling and sapling structure stages occur at approximately 55 percent of the distribution. Mature classes are at 25 percent with the largest, usually oldest, classes almost nonexistent.

Relatively recent fire and harvest disturbance, however, contribute to resilience in lodgepole pine structure on the landscape of the Greendale Plateau. These smaller structure stages, which are generally younger trees, have provided a head start on structure mix, having largely survived a recent mountain pine beetle epidemic in the area that peaked near 2010 (figure 6).

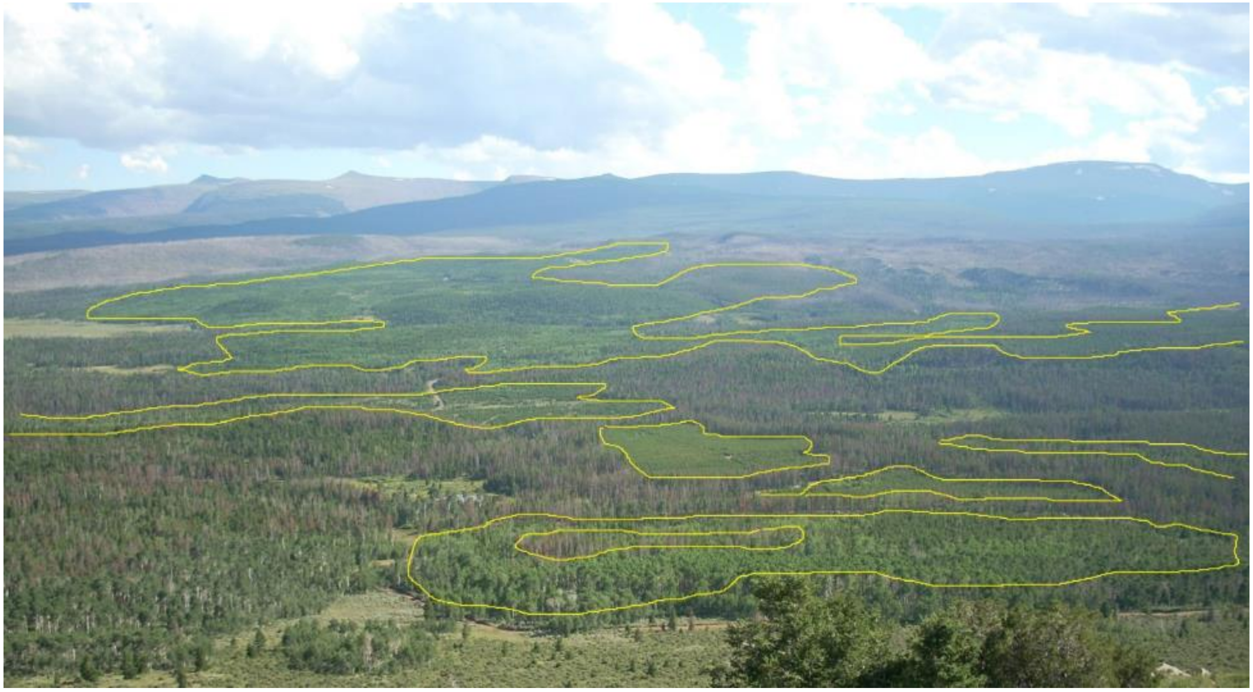


Figure 6. Landscape view of lodgepole pine belt on the Greendale Plateau landtype association (2011). Past clearcut harvest and fire disturbance are outlined in yellow (regenerated live trees). Recent tree mortality (red needles) due to mountain pine beetle is visible between areas of past disturbance.

Areas impacted by the recent epidemic now have canopies composed mostly of dead trees that are not fully opened to sunlight, inhibiting immediate regeneration response. Only the live conifers, primarily smaller diameter lodgepole pine (less than six inch in diameter at breast height) that survived the outbreak are left but at low health, low densities, or sparse distribution. Aspen is the only other tree species left at significant amounts. Lodgepole pine is extremely shade intolerant and reproduces best when the stand is opened (Lotan 1983). Lodgepole pine needs large enough openings to admit adequate light to the forest floor. Opening sizes of a least one and a quarter acre are required or lodgepole pine is significantly inhibited (Koch 1996, p 160). In 10 to 30 years, as dead trees start to topple, openings created by dead trees are commonly filled in with lodgepole pine seedlings that grow up through the deadfall. The result of the mortality caused by mountain pine beetle epidemics is an increase in coarse woody debris and ladder fuels in various forms. These coarse woody debris and ladder fuels provide fuel to the large stand-replacing fires common to the lodgepole pine belt of the Greendale Plateau landtype association.

Lodgepole pine dwarf mistletoe is a parasitic plant that commonly infects lodgepole pine on the Greendale Plateau landtype association. Dwarf mistletoe is contributing to the small diameter of trees on this association. Some stands with a very low growth rate have been heavily infected with lodgepole pine dwarf mistletoe for many years. Some stands east of Sheep Creek Park provide examples of this condition (GreendalePlateau.ppt). Large stand-replacing fires on the landtype association have reduced the incidence of mistletoe-infected lodgepole pine.

Patch size on the landtype association ranges from small (tens of acres) to large (hundreds to thousands of acres). Since 1950, about 50 percent of the lodgepole pine type on this landtype association have been harvested under a 40-acre-or-less, patch-size design. Conversely, fires that have occurred on this landtype association have been mostly large (hundreds to thousands of acres) and stand replacing, even though some would likely have been larger had they not been suppressed. The potential area that could have been affected by these fires indicates that patch size of several thousand acres or more in lodgepole pine was common for this landtype association historically.

Patch size has an effect on dwarf mistletoe infection spread and impacts. In dwarf mistletoe-infected forest, larger patch size reduces the amount of edge or interface between infected overstory trees and uninfected regeneration on the patch. Dwarf mistletoe is "spatially limited. Most dwarf mistletoe seeds fall within 60 feet from a tall single tree…In even aged stands, spread is only 1 to 2 feet per year due to screening by existing trees" (Guyon 2013). Clearcut harvests at smaller patch size may be expected to contribute more to dwarf mistletoe spread where it is present than would have otherwise occurred, had patch size been closer to historical.

As on the Greendale Plateau landtype association, the lodgepole pine forest on the Parks Plateau landtype association includes large stands of lodgepole pine with an obvious disturbance history of stand replacement fire and mountain pine beetle epidemics. Ogle and DuMond (1997) summarized reports by S.S. Stewart of 1909 and L.E. Hitchcock of 1910 that documented extensive fire for the area of the Parks Plateau landtype association. The Stewart report included a note of a large burn across four sections, or 24 miles. Forest fires of great extent were also reported in 1871 for the Uinta Mountains by George M. Wheeler while on a U.S. Geographical Survey (Graham 1937). As on the Greendale Plateau, cones of lodgepole pine on this landtype association likewise have a relatively high level of serotiny, indicating a highly fire-adapted system. Serotinous cones also contribute to stagnant stands of dense even-aged lodgepole (2 to 5 inches in diameter at breast height) on this landtype association. Mortality in these stands of "doghair" lodgepole pine comes in the form of competition and self-thinning. Unlike the Greendale Plateau, the lodgepole pine on the Parks Plateau has not had a relatively recent fire or insect disturbance history, although mountain pine beetle caused wide-scale tree mortality in the landtype association during the 1980s.

Timber harvest appears to have been a major contributor to smaller seedling and sapling structure stages, increasing these stages to about 40 percent of the total structure distribution. Consequently, the structure distribution is more balanced. Distributions are weighted more heavily to the mature structure stages than on the Greendale Plateau, with 47 percent in the mature sizes and 9 percent in the large structure sizes. Like the Greendale Plateau, patch size in many of the harvested areas has been limited to less than 40 acres on the Parks Plateau.

Climate Related Risks and Trends

The recent mountain pine beetle epidemic in the Greendale Plateau and surrounding landscape is a disturbance event that shows indications of being within the natural range of variation, according to the dynamics inherent to a lodgepole pine system. However, tree damage has occurred in stands of this landtype association with lower basal areas, and in trees with smaller diameters (USDA 2013a) than as indicated in the past.

Under a warming climate, the number of wildland fires is expected to increase and the fire return interval is expected to become more frequent (USDA 2016b). Years with no major fires, which were common historically, are expected to be rare. A more frequent fire regime could change fuel dynamics. Reduced fire intervals are likely to preclude tree regeneration, allowing for some conversion to non-forest (USDA 2016b). One would expect less mature and older forests under such a change in regime.

Comparison of the Natural Range of Variation and Current Conditions

Based on the structure characteristic in the persistent lodgepole pine type on the Ashley National Forest, the system appears to be operating at moderate departure from the natural range of variation, due to some fire suppression. Stand replacing fires that have occurred in the lodgepole pine types would likely have been larger, had they not been suppressed. Contributing to departure is patch size due to timber harvest, which occurred at much smaller size than would have occurred historically with wildland fire.

Douglas-fir

Description of the Natural Range of Variation

Historic fire regimes of nonlethal and mixed-severity fires with mean fire return intervals of 35 to 200 years once dominated Douglas-fir forests. More frequent intervals dominated drier sites, while the longer intervals dominated mesic sites. Variable fire intensities in such a regime drive the uneven-aged structure characteristic of Douglas-fir (Agee 1998; Arno and others 2000). The uneven-aged structure can often resemble even-aged groups of mature and younger trees rather than a mixture of age classes by single trees. The diverse structure characteristic inherent of a mixed severity fire regime "limit the availability for [Douglas-fir beetle] to cause landscape scale Douglas-fir mortality" (Giunta 2016).

Current Status and Trends[8]

The presence of low- to mixed-severity fire has been largely absent in the Douglas-fir forest on the Ashley National Forest since settlement and is outside historical ranges. Fire suppression leads to Douglas-fir forests that become increasingly dense and more mature. These forests are subject to forest health issues that include widespread Douglas-fir beetle epidemics and increased incidents of dwarf mistletoe infection (Giunta 2016). Consequently, the Douglas-fir beetle has affected most stands of the Douglas-fir forest on the Ashley National Forest in the past decade. Beetle epidemics, "serve a role similar to a stand replacing fire, leading to widespread tree mortality, and promote forest succession" (Giunta 2016).

Although most of the trees killed were merchantable, very little has been salvaged. Most of the areas impacted by the Douglas-fir beetle are located in Inventoried Roadless Areas, particularly in the South Unit.

On the North Flank landtype association, the presence of seral aspen in Douglas-fir stands is strong evidence that fire is a dominant process for this type. Individual trees and groups of trees of mature Douglas-fir with fire scars, surrounded by thickets of smaller trees, indicate that mixed-severity fires also occurred in some Douglas-fir stands on this landtype association.

However, except for a few clearcuts and the Sols Canyon Fire of 1980 (i.e., 115 acres of opened forest), Douglas-fir formed nearly continuous mature stands on this landtype association, indicating a departure from the natural range of variation. Fire appears to have played a minor role on the North Flank landtype association in the last 100 years. Harvest disturbance is recorded on five percent of the Douglas-fir forest on this landtype association.

As in other Douglas-fir forests across the Ashley National Forest, a bark beetle epidemic that peaked near year 2008 has affected most of the Douglas-fir type on this landtype association. A high percentage of larger trees are dead following this epidemic. The amount of available fuel that will be generated in the next several decades as these beetle-killed trees topple to the ground, combined with other woody debris that has not burned due to fire suppression of the past 100 years, would likely be outside the natural range of variation for Douglas-fir stands.

With the increase in available fuel that can be expected to follow the recent bark beetle epidemic in several decades, fire might be expected to be a much greater factor in the next 50 to 100 years: "Changes in forest conditions can have a great impact on interior Douglas-fir stands, where the prevalence of high-severity fires may become more commonplace in these forests which previously experienced historic low-severity fire regimes" (Covington and Moore 1994; Daniels 2004 as summarized by Giunta 2016).

[8] Seral stages of mixed conifer tree structure are further described in appendix B.

High-severity fires are expected to lead to stand replacement. This could possibly result in a nonforested condition where aspen is not present, or delayed regeneration where nearby surviving seed trees are not available. Recognizing that fire on the landtype association is needed, managers have attempted to reintroduce fire on this landtype association in a way that resembles a mixed severity fire (USDA 2011).

On the Red Canyon landtype association, the observed level of Douglas-fir beetle activity has been relatively light when compared to the other landtype associations, which have been widespread. This could be due to the isolated stands of Douglas-fir on this landtype association, compared to the large continuous stands on the others. Some older stands of Douglas-fir, however, have recently been attacked by bark beetles. This has resulted in a high percentage of mortality of the larger trees and an increase in coarse woody debris and potential available fuel. There has also been little fire suppression on this landtype association, but prior to the Flaming Gorge Reservoir, the dense vegetation of the canyon bottom was likely a heavy fuel source that generated fires of sufficient intensity to carry up the canyon sides. With the canyon bottom now covered with water, fire frequency and size might be reduced, resulting in less diversity of structural stages.

The Stream Canyon landtype association is influenced by high rock component. Consequently, there has been little timber harvest and minimal fire suppression. Fire likely played a minor role in the size and shape of Douglas-fir patterns, due to large expanses of talus or other rocky areas. These areas are largely devoid of available fuel, likely preventing most fire. Douglas-fir beetle epidemics have occurred in the past 25 years on this landtype association.

The Avintaquin Canyon and Anthro Plateau landtype associations occur on the Tavaputs Plateau and support most of the Douglas-fir on the Ashley National Forest. There has been very little timber harvest on these two landtype associations. Fire had once played a role in the Avintaquin Canyon landtype association (Study 64-42D, figure 7). Recent fires (2008 to 2012) that were of mixed severity have occurred in about seven percent of the Douglas-fir forest on this landtype association, contributing to structural diversity.

Figure 7. Evidence of mixed severity fire pre-settlement near the Avintaquin Canyon landtype association. Younger stands, presumably a result of past fire, survived the recent Douglas-fir beetle outbreak.

However, a Douglas-fir beetle epidemic that had swept across the areas, peaking in 2005, indicates that fire suppression since settlement has led to a departure from the natural range of variation. The lack of structural diversity, otherwise inherent in a mixed-severity fire regime, had allowed for extensive late-seral forests highly susceptible to bark beetle epidemics (figure 8). As in the other landtype associations where beetles have impacted Douglas-fir, very high fuel loads will likely begin appearing in about 30 years when the dead trees begin to fall.

Figure 8. Extensive Douglas-fir tree mortality caused by Douglas-fir beetle

Climate Related Risks and Trends

Under a warming climate, more frequent high-severity fires are expected. These fires could lead to loss of mature trees that serve as seed sources to the next generation of Douglas-fir (Perry and others 2011).

Comparison of the Natural Range of Variation and Current Conditions

The diverse structure characteristic, inherent of a mixed severity fire regime in the Douglas-fir type, appears to be lacking on most landtype associations of the Ashley National Forest. This characteristic appears to be at moderate to high departure from the natural range of variation.

Mixed Conifer and Engelmann Spruce

Description of the Natural Range of Variation

Mixed conifer: The disturbance regime plays a strong role in driving structure in the mixed conifer vegetation type. As described in the composition section, fire intervals of 50 to 130 years have been estimated (Arno 1980) in the lower elevation subalpine habitat type. These disturbances lead to a mosaic of different ages and species compositions. Multiple successional pathways are possible depending on climate and disturbance. On the moist-wet subalpine habitats, severe fires are rare and stand-replacing fire intervals can be longer, as high as 300 to 400 years (Bradley 1992). More frequent smoldering fires remove single trees or groups of trees rather than entire stands, leading to condition where trees from almost all age classes are represented.

Engelmann spruce: At the higher elevations of cold, subalpine habitat type, severe fires are rare. The fire return intervals becomes more infrequent, estimated at greater than 200 years - and can be 300 to 400 year stand-replacing intervals. Most of these stands have heavy snowpack well into the summer months. Fuels only dry out in August and September, about the time late-season storms end the fire season. Moist conditions and discontinuous fuels create low fire hazard for this type.

In both the mixed conifer and Engelmann spruce vegetation types, insects are an important driver of change and structure in the subalpine system. As described in the “Composition” section, one of the most significant disturbance agents in the spruce-fir ecosystem is spruce beetle (*Dendroctonus rufipennis*). The spruce beetle is a native insect with periodic outbreaks. The mortality of spruce-beetle-preferred host material (overstory spruce) can change forest structure and shift stands from older and mature, to younger structure classes.

Current Status and Trends[9]

On the Alpine Moraine landtype association, fire is a driver of structure, more so at the lower elevations where the mixed conifer vegetation type is prevalent. The abundance of lodgepole pine at lower and mid elevations, where spruce and fir are also present, indicates mean fire return intervals are more frequent, between 35 and 200 years (Landfire 2012). In spite of this, a majority of the forest was in the late-seral stage for much of this landtype association, exhibiting large structure sizes typical of mature and old forests. Mountain pine beetle, and now spruce beetle, is changing that. Photographic records also show a moderate displacement of subalpine meadows by conifer. Similar to the climate related trends that caused the fluctuation of the tree line in alpine plant communities, the encroachment of conifer into these subalpine meadows may be within the natural range of variation.

Fire, as a driver of structure, has less influence in the Engelmann spruce vegetation type of this landtype association at higher elevations. The lack of or absence of lodgepole pine in the Engelmann spruce belt indicates fire return intervals are typically greater than 200 years (Volland 1985). The upper-elevation forests are dominated by Engelmann spruce, and lodgepole pine has been found with 24 inch in diameter at breast height with an average age between 250 to 300 years. Patch sizes of mature trees are large over much of the landtype association. Also, old forests covered thousands of acres prior to the advancing spruce beetle epidemic. In addition, some forest stands at the upper elevations of the landtype association are separated by large alpine areas that include talus and boulder fields of the Uinta Bollie landtype association that which are often highly effective fire barriers. These characteristics indicate that fire size has historically been comparatively small on this landscape, especially at the upper elevations. Other factors that tend to decrease the occurrence of fire on this landtype association are persistent snowpack and precipitation. Snowpack on the Alpine Moraine landtype association commonly persists to the end of June or early July.

Timber harvests have also driven structure change on the Alpine Moraine landtype association in the mixed-conifer type. Patch size has mostly been limited to less than 40 acres, due to timber harvest on this landtype association. Timber harvest has been comparatively minor, but where clearcutting has occurred at higher elevations, regeneration has been very slow (e.g., Whiterocks drainage). Most harvest has been in the Chepeta Lake area where there is road access. Less than five percent of the landtype association has been harvested.

Conversely, fire is a major disturbance process that drives structure on most of the Trout Slope landtype association. Evidence of this is demonstrated by large areas in the mixed conifer types, dominated by even-aged stands of lodgepole pine where Engelmann spruce has the potential to be more competitive in absence of fire. A strong history of fire is also indicated by the numerous trees with fire scars. The abundance of these scarred trees seems to indicate more ground fire and less stand-replacement fires with

[9] Seral stages of mixed conifer tree structure are further described in appendix C.

increasing elevations (Study 17-22B) where the Engelmann spruce vegetation type becomes more prevalent.

Even-aged stands of lodgepole pine are generally more common at lower elevations and stands of Engelmann spruce are more frequent at higher elevations. These observations suggest that larger and more intense fires occur at lower elevations and smaller, less severe fires occur at higher elevations.

The multiple size classes of trees, and abundance of Engelmann spruce, is an indication of the relative long-absence of stand-replacing fires at higher elevations in the Engelmann spruce vegetation types. In addition, wet areas in parts of the Trout Slope landtype association (e.g., TS4 landtype) apparently reduce the intensity of fires that start here. Consequently, there is a greater potential for sustainable multiple-aged stands, with greater structural diversity in these areas, than in upland sites that appear to be more prone to stand-replacing fires. However, these stands are not immune to stand-replacing fires, and these stand-replacing fires are likely the principal means by which lodgepole pine is maintained here.

Timber harvests have also driven structure change on the Trout Slope landtype association in the mixed conifer type. Much of the Trout Slope landtype association has been a major source of timber products that have been removed from the Ashley National Forest (Studies 17-31A, 17-31P, TroutSlope.ppt). Timber harvest has been concentrated on certain landtypes of this association (e.g. TS1 and TS2), while other landtypes have had very little harvest. Harvests (e.g., clearcuts) that have occurred on this landtype association over the last 50 years have been mostly limited to 40 acres.

If there is a shortage of a particular structural stage on the Trout Slope landtype association, it is likely the late-seral stage as dominated by live trees. The mountain pine beetle epidemic of the 1980s devastated mature lodgepole pine in the mixed conifer type of the east Uinta Mountains, and again in the first decade of the 2000s in the west Uinta Mountains. Spruce beetle is currently adding to the mortality of late-seral stages by killing the majority of Engelmann spruce greater than five inches in diameter. The disparity of the late-seral stage dominated by mature live trees is likely a landscape-scale condition.

In the Stream Canyon and Glacial Canyon landtype associations, mixed-conifer forests occur mostly in talus or other rocky areas. There has been little harvest of mixed conifer in these landtype associations. Fire suppression has also been minimal. Due to the rocky areas where mixed conifer stands are generally found in these landtype associations, fire likely played a minor role in the size, shape and patterns of the mixed conifer type. These rock fall forests are likely static over rather long periods where they are protected from fire. However, the Uinta Canyon Fire of 1989 (Study 26-16F) in the Glacial Canyon landtype association is an indication of the potential for large fires in at least some areas.

At the highest elevations, the Uinta Bollies landtype association is heavily weighted to Engelmann spruce forest. Engelmann spruce dominates these tree-covered areas, often stunted and of krummholz growth form, where fire and insects do not appear to be major drivers of structure on this landtype association. However, this may change with the advance of high spruce beetle populations. Fire is rare. Fire frequency is likely much longer in the Engelmann forests on this landtype association due to cool temperatures, the persistence and early return of snow, and an annual pattern of summer rains. Lightning strikes might be abundant due to the high elevation, but actual starts are generally few. Likewise, the size of fires is relatively small due to the presence of rocky interspaces, a relatively low amount of coarse woody debris, and a low amount of fine fuels. Of all the landtype associations on the Ashley National Forest, vegetative structure has been the least altered on this landtype association. There has been little to no harvest and generally, no fire suppression.

On all landtype associations with a lodgepole pine or Engelmann spruce component, native bark beetles are a major driver of structure. The beetles affect the larger diameter trees in the mixed conifer and

Engelmann spruce vegetation types. Mountain pine beetle, and now spruce beetle, are driving structure change in the subalpine habitat type on the Ashley National Forest. Stands that have experienced high levels (more than 65 percent stand affected) of tree mortality can be expected to have a build-up of coarse woody debris over the next several decades, leading to increased fuels for wildland fire. Structure is likely to shift from continuous mature and old structure, to younger or even to delayed regeneration. While large diameter lodgepole pine is still represented across the Ashley National Forest, current size class distributions indicate a trend toward diameter classes that are not the preferred host size for mountain pine beetle.

Climate Related Risks and Trends

Forests will be moderately vulnerable to a warmer climate. Crown fires may be more prevalent, eliminating mature trees across the landscape. Lodgepole pine could be more susceptible to mountain pine beetle, but serotinous cones may allow for rapid, dense regeneration.

The advance of conifer into the subalpine meadow communities on the Alpine Moraine landtype association may be climate related. Generally, soils that develop in areas displaced by conifer are typically less resilient and not as productive as soils of existing subalpine meadow communities because less organic matter is deposited on the soil surface. Under these conditions, the soil surface horizon generally becomes shallower and less productive.

Comparison of the Natural Range of Variation and Current Conditions

In both the mixed conifer and Engelmann spruce types, the extent and level of the advancing spruce beetle epidemic is consistent with the extent of available preferred host material (Engelmann spruce larger than eight inches in diameter at breast height) and impacts from a changing climate on insect life history (Bentz and others Bentz and Jönsson 2015, and Bentz and Munson 2000). Although spruce beetles are a part of spruce-fir forest ecology and a key disturbance agent, the impact of the current epidemic is threatening the sustainability and resiliency of the subalpine system. Combined with climate projections of increasing temperatures and future fuel loads as the dead trees topple, more frequent and extensive fire is expected.

Mixed conifer: Some fire suppression had led to greater prevalence of older structure classes, especially in the Alpine Moraine and Trout Slope landtype associations. With bark beetle caused tree mortality and subsequent wildland fire, tree structure is expected to shift from continuous mature and old structure to younger, especially where lodgepole pine and aspen are present. In addition, the timber harvests that have occurred on this landtype association over the last 50 years have been mostly limited to less than 40 acres. This disturbance pattern and subsequent regeneration is considerably different from that created by historical stand-replacing fires that typically burned thousands of acres in one event. The difference in extent of area in early and mid-seral condition as a function of harvest, compared to the area of equal seral status under fire regimes of the past, is unknown. However, it is likely that the size, shape, and patterns of stands in a particular structural stage created by harvest are much different from stands driven by fire of the past. Due to these reasons, the mixed conifer is estimated at moderate departure from the natural range of variation.

Engelmann spruce: Fires have generally not been suppressed in the Engelmann spruce type; tree structure is estimated to be at low departure from the natural range of variation. For reasons listed above, however, structure in the Engelmann spruce type is estimated to be trending at moderate departure from historical. Tree structure is changing with the advance of spruce-beetle-caused tree mortality. Combined with projected increases in fire, regeneration back to spruce would be delayed where seed tree sources are unavailable, possibly creating nonforest-like conditions.

High-elevation spruce, and spruce that is growing in an uneven-aged condition, may be more resistant or resilient to stand-replacing events (Schmid and Frye 1977, Dymerski and others 2001). The reason for this is that seedlings or saplings in uneven-aged stands would survive a bark beetle outbreak, which would give the stand a head start on regeneration. Uneven-aged stands, however, are not immune to stand replacing fires, especially those stands that have had a significant beetle-caused mortality event. The extent of uneven-aged stands is not known throughout the Engelmann spruce type, although the Trout Slope landtype association is noted for having uneven-aged stands at higher elevations and on wetter areas.

Sagebrush

Description of the Natural Range of Variation

Sagebrush communities are common to the Uinta Mountains, Tavaputs Plateau, and high deserts of Wyoming. These sagebrush communities are represented across the landscape within a broad range of environments, successional states, and community types. Mountain and Wyoming big sagebrush and black sagebrush account for about 95 percent of sagebrush found within the plan area. Because of that, only these communities will be discussed in further detail in regards to the natural range of variation and sagebrush structure.

Sagebrush structure is described at a landscape scale - spatially and temporally - and at a community scale by height, density, canopy cover, or some combination of these things. Mountain big sagebrush is a montane landscape dominant that extends from the foothills of mountains and plateaus up to aspen and coniferous forests of higher elevation. Its elevational range is 6,200 to 9,850 feet (Kitchen and Durrant 2007). Mountain big sagebrush communities are found within its elevational zone across the Uinta Mountains and the Tavaputs Plateau. The shrub usually grows in and above the pinyon-juniper thermal belt and occasionally interfaces Wyoming big sagebrush within this transition zone (Goodrich and others 1999b). Wyoming big sagebrush a landscape dominant species in drier environments, with an elevational range of 4,900 to 6,500 feet (Kitchen and Durrant 2007). Within the plan area, most Wyoming big sagebrush communities are found in Green River landtype association in southwest Wyoming, but the shrub also grows in and below the pinyon-juniper thermal belt of low elevations of the Tavaputs Plateau and Uinta Mountains. Black sagebrush has a relatively wide elevational amplitude, ranging from 6,100 to 9,100 feet. Its elevational range is “nearly equal to the combined range of the three sub-species of big sagebrush” (Kitchen and McArthur 2007). Unlike mountain and Wyoming big sagebrush, “black sagebrush is sharply limited as to where it can grow” based more on soil features and less on elevational range (Fryer 2009). Soils that support black sagebrush are typically shallow, rocky, or gravelly, often poorly developed, and these have the lowest water holding capacity, organic matter and nitrogen content (Fryer 2009). Black sagebrush often integrates with pinyon and juniper at mid-elevations and has been encroached upon by Douglas fir at higher elevations.

Natural disturbances play an important role in many sagebrush communities. Disturbances such as insects, disease, winter exposure or snow lay, are relatively minor in their effects to sagebrush spatially. However, fire and extreme climate events have impacted or influenced sagebrush distribution, structure, and composition at larger scales (Kitchen and McArthur 2007, USDA Forest Service 2016). Fire frequency and rate of return of sagebrush following fire is variable and correlates closely to several temporal and spatial parameters, but these features are lower for mountain big sagebrush than for Wyoming big or black sagebrush (Kitchen and McArthur 2007). Sagebrush communities, adjacent to or growing within conifer zones, are susceptible to encroachment and displacement by trees. On average, fire occurred often enough to maintain sagebrush in these settings. Although not well documented, sagebrush die-back during extreme climate events, such as severe drought, have likely occurred historically. The rate of return of sagebrush following drought related die-back would be similar to return intervals related to fire under the natural range of variation.

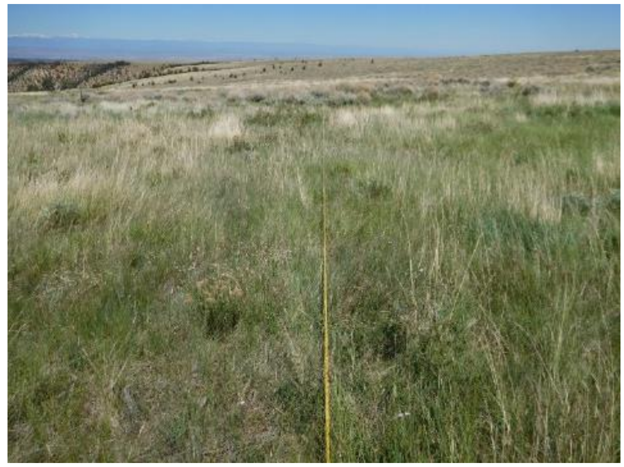
Mountain big sagebrush is of medium height, where shrubs vary from two to five feet. Late succession communities usually consist of two or more layers of cover. Unlike other big sagebrush, mountain big sagebrush appears flat-topped, since seed stalks and foliage are of comparative height (Kitchen and McArthur 2007, Johnson 2000). Extensive stands of mountain big sagebrush grow within the plan area. Communities adjacent to or integrating with conifers are susceptible to conifer displacement. Canopy cover of the shrub is relatively variable from site to site. Canopy cover equal to or greater than 25 percent in the absence of fire is common, but seldom reaches or exceeds 40 percent under NRV (Goodrich 2012, Goodrich and Huber 2001). Tart (1996) found canopy cover of all shrubs in mid to late seral communities to be between 23 to 40 percent. In some communities, native perennial grasses dominate vegetation cover and late seral sagebrush canopy cover may not exceed 20 percent (Kitchen and McArthur 2007). Mountain big sagebrush is a fire driven system and most fires within these communities are stand-replacing. The fire return interval for these communities varies from 15 to 40 years but may extend to 40 to 80 years (Winward 1991, Johnson 2000, Kitchen and McArthur 2007). Fire influences community structure, especially in regards to canopy cover. Historic and recent wildfires have burned mountain big sagebrush communities within the plan area (Stewart 1911, Hitchcock 1910, Graham 1937, Heyerdahl 2011, Ogle and Dumond 1997, Huber and Goodrich 2011, Goodrich and Huber 2011). Mountain big sagebrush is self-replacing following fire and is capable of returning to pre-burn conditions within 15 to 25 years (Goodrich and Huber 2001). Under the natural range of variation, landscape patterns of fire with various stages of sagebrush canopy cover in recovery is not well understood (USDA Forest Service 2016). In summary, the short recovery time for mountain big sagebrush following fire is also indicative of an ecological history of higher fire frequency.

Wyoming big sagebrush is comparable in height to mountain big sagebrush, ranging from two to three feet. Shrubs appear rounded with uneven crowns. Late succession communities usually consist of two layers of cover, but the herbaceous component is often sparse. Community structure of this shrub is significantly influenced by annual precipitation. Where annual precipitation is at, or slightly less than, eight inches, Wyoming big sagebrush intermixes with salt-desert shrubs (Goodrich 2012, Kitchen and McArthur 2007). Where annual precipitation is greater eight inches, Wyoming big sagebrush forms extensive communities. At the upper end of its precipitation zone, the shrub integrates with, and is susceptible, to displacement by pinyon and juniper trees. Potential canopy cover of Wyoming big sagebrush is inherently lower than other big sagebrush taxa, which seldom exceeds 25 percent. Goodrich and others (1999) found sagebrush canopy cover ranging from 0.3 to 22 percent at sites protected from domestic livestock grazing. In Oregon, Utah and Wyoming, others found similar canopy cover ranges and averages comparable to those reported by Goodrich and others (Davies and others 2006, Bates and others 2009, Anderson and Holte 1981, Perryman and others 2002). Fire frequency of Wyoming big sagebrush is lower than for mountain big sagebrush, due low precipitation and vegetation productivity. A fire interval of 50 to 110 years is likely the return interval for many presettlement Wyoming big sagebrush communities, but may be as high as 200 to 350 years in some settings (Winward 1991; Whisenant 1990; Mensing and others 2006, NRV paper 2016, Howard 1999). Likewise, rate of shrub return is considerably slower for Wyoming big sagebrush than for mountain big sagebrush (Britton and Clark 1984, Bunting 1984, Boyd and Svejcar 2011, Lesica and others 2007, West 2000, West and Yorks 2002, Winward 1991, Studies XX). Landscape patterns consisted of “patchy fires that produced a mosaic of burned and unburned lands” (Howard 1999).

Black sagebrush is low “growing and decumbent,” where shrubs are generally two feet or shorter (Fryer 2009, Kitchen and McArthur 2007). “Black sagebrush crowns are typically irregular, spreading, and U-shaped,” but may appear rounded with ungulate browsing (Fryer 2009). Late succession communities usually consist of two layers of cover. Kitchen and McArthur (2007) found that “black sagebrush segregation from big sagebrush is due to its ability to grow in shallow, rocky soils and mixing of the two species is generally limited to narrow ecotones.” In desert shrub environs, black sagebrush is a dominant that forms nearly pure stands (Fryer 2009). These communities consist of scattered shrubs with “sparse herbaceous understory” (Kitchen and McArthur 2007, Fryer 2009, Goodrich 2001). In this environment, black sagebrush is long persisting with little to no displacement by other shrubs (Goodrich 2001). Canopy cover of black sagebrush rarely exceeds 15 percent, even without browsing pressure, but canopy cover can be greatly reduced where wild ungulate use is prevalent (Goodrich 2001). Fire frequency of these communities is estimated between 100 to 200 years (Fryer 2009). Black sagebrush often integrates with pinyon and juniper at mid-elevations and functions as a seral species (Goodrich 2001, Fryer 2009). In this setting, fire occurred at intervals that maintained black sagebrush communities in the pinyon-juniper belt (Goodrich 2001). At high elevations, stable black sagebrush communities of considerable size occur. Some communities are susceptible to displacement by Douglas fir (Goodrich 2001, Fryer 2009). Canopy cover of black sagebrush and herbaceous vegetation cover is considerably higher in these communities than those of lower elevation. Black sagebrush canopy cover can exceed 25 percent (Goodrich 2001, Studies 68-1, 68-69). Black sagebrush can withstand moderate wildlife browsing pressure, but canopy cover can decline with long-term overuse (Fryer 2009). Montane black sagebrush communities have a higher fire frequency (30 years or more) than low elevation communities (Fryer 2009). Rate of return of black sagebrush is not well understood, but 15 to 60 years to regain pre-fire cover is estimated (Freyer 2009, Goodrich 2001). Information regarding landscape fire patterns is limited, but “they were probably mostly patchy and of mixed severity, although stand-replacement fires may have occurred rarely” (Fryer 2009).

Current Status and Trends

Almost all mountain big sagebrush communities have been impacted by human uses since at least the early 1900s. Livestock grazing is a long-term and enduring practice that has occurred for more than 100 years. In regards to community structure, consequences of long-term livestock grazing in mountain big sagebrush have been: the increase in sagebrush density and canopy cover, the depletion of the herbaceous understory, reduced fire frequency, and the acceleration of conifer invasion (Kitchen and McArthur 2007). In response to these changes, thousands of acres of mountain big sagebrush were plowed and seeded into introduced grasses, sprayed with herbicide, treated with prescribed fire and conifer lop-and-scatter (Goodrich and others 2005, Goodrich and Huber 2015). These treatments were successful temporarily in reducing sagebrush canopy cover and increasing forage species for ungulates. In nearly all cases, sagebrush returned to pre-treatment levels within expected return intervals, regardless of the herbaceous understory or treatment type (Goodrich and others 2005, Goodrich and Huber 2015). However, there are examples on the Ashley where sagebrush return was longer than the expected return interval (Studies 31-36, 31-37, 32-15B, 32-72F-H). Since 1995, thousands of acres of mountain big sagebrush were treated with prescribed fire or conifer lop-and-scatter to curtail conifer encroachment and displacement of sagebrush (Huber and others 2010, Bistryski 1996, Bistryski 2000, Kirkaldie 2009, Kirkaldie 2009b, Kirkaldie 2009c). Landscape fire patterns of wild and prescribed fire over the last 30 years mostly appear mosaic in nature, with relatively uneven edges; where fire spread followed natural topography and was limited by natural barriers. Burned areas were variable in size, from five acres to about 4,000 acres (Goodrich and others 2005). Mountain big sagebrush and its shrub associates of the Parks Plateau, Anthro Plateau, Avintaquin Canyon, and Strawberry Highlands landtype associations have and are returning to pre-disturbance canopy cover within 15 to 30 years following fire, which falls within expected return intervals (Goodrich and Huber 2001, Goodrich and others 2008).

Photos of study 68-26D. Anthro Mountain. 2007 and 2016. View of prescribed fire treatment of mountain big sagebrush. Fire treatment created a mosaic of burned and unburned areas.

Most mountain big sagebrush communities of the South Face, Dry Moraine, Glacial Canyon, Stream Pediment, and Structural Grain landtype associations are currently in satisfactory condition in regards to sagebrush canopy cover and shrub return intervals following fire. However, structure of sagebrush is at risk because of its moderate to high susceptibility to annual invasive plants such as cheatgrass. Cheatgrass is capable of shortening fire return intervals sufficient to preclude reestablishment of sagebrush following fire. Long-term monitoring shows cheatgrass is present and increasing in mountain big sagebrush communities with native herbaceous understories, especially following fire and severe drought (Studies 6-46A-D, 38-7A,E-G, 38-12B, 39-8A-D, 39-9, 39-16, 41-1, 41-2A, 41-3E3-F, 41-7A,C-G,N, 42-17A4A, 42-17F, 43-16A-B, 51-17). Although cheatgrass was present prior to 2002, most of its spread has occurred since that time (USDA Forest Service 2015a, 2015b, Goodrich and Huber 2015). Cheatgrass has increased considerably over the last few years in native shrub communities on Mosby Mountain, Dry Fork Mountain, Lake Fork Canyon, and Yellowstone Canyon. The most affected mountain big sagebrush communities are located on steep, southerly aspects of the South Face landtype association. Here, cheatgrass often dominates herbaceous cover and is most abundant where fire, drought, or both has recently occurred (Goodrich and Huber 2015, Studies 37-21A-B,D, 37-24A-E, 44-16C, 44-23A-C, 44-24G, 44-26A-B). The resilience of native mountain big sagebrush communities of the South Face, Dry Moraine, Glacial Canyon, Stream Pediment, and Structural Grain landtype associations is low to moderate. In contrast, in communities where seeded nonnative grasses dominate herbaceous cover, cheatgrass is absent or has minor presence with no indication of spread or increase (Studies 38-16, 41-8A-F,I, 41-10A--E, 42-1A,C-G,I-K, , 42-13A-C, 43-1A-B, 43-7B-C). These communities typically have satisfactory plant composition, species richness, and total ground cover. Historic seeding treatments of these shrublands with nonnative grasses have demonstrated high resilience to invasive annuals.

Most Wyoming big sagebrush associated with the Antelope Flat, Green River, North Flank, and Structural Grain landtype associations are currently in late seral stages with relatively high canopy cover. However, some sagebrush communities have been affected by disturbances over the last 15 years. For instance, pronghorn antelope browsing reduced canopy cover of some Wyoming big sagebrush communities of the Antelope Flat landtype association. (Goodrich and Zobell 2012). Droughts of 2002 and 2012 to 2014 have affected some Wyoming big sagebrush communities. Following the drought of 2002, considerable sagebrush die-back occurred in some communities (Drought White paper, Goodrich and Zobell 2012, Goodrich 2005, Studies 4-42B, 5-1A, 5-2A, 5-3B2,D, 5-27, 5-52A, 6-46A-C, 72-33). Most die-off occurred in communities with low annual precipitation (Goodrich 2005). With the return of abundant and timely precipitation, sagebrush in these communities increased in vegetative growth, cover, and seed production, indications of resilience, capability, and proper function. Although present, cheatgrass showed little to no spread prior to 2012. Since the start of drought in 2012, cheatgrass has spread considerably into many Wyoming big sagebrush communities (Studies 5-27E, 5-27J, 72-8, 72-8B, 72-32, 78-13C, 78-26). Cheatgrass can change community function and lower resilience by creating continuous, flammable fuels, which potentially increases fire frequency. Shorter fire return intervals would impede, or potentially stop, reestablishment of sagebrush. In summary, Wyoming big sagebrush communities currently have a moderate to high susceptibility to cheatgrass invasion. This susceptibility places the communities at risk of loss of resilience, capability, and function.

Wyoming big sagebrush communities of the Anthro Plateau landtype association grow in shallow drainages or swales, on alluvial fans of canyon bottoms where occasional fire maintained shrubs. Most sagebrush communities within the shallow drainages or swales were burned with prescribed fire from 1970 to 1999. The return of Wyoming big sagebrush is slow, with little to no recruitment of shrubs following treatments. Two sites that were burned in about 1970 have sagebrush canopy cover of 2 and 7 percent, and two sites burned in the 1980s have sagebrush canopy cover of 0 and 2 percent. This indicates a sagebrush return interval greater than 50 years (Studies 69-21B, 69-72A, 70-9B, 69-58A). In communities that have not burned, average sagebrush canopy cover averaged 18 percent, (Studies 69-31, 69-31B, 69-95C, 70-10A). Severe droughts occurred in 2002 and from 2012 to 2014, but minimal die-back of Wyoming big sagebrush occurred. In a few sagebrush communities burned with prescribed fire, winterfat increased in density and cover and replaced dead perennial grasses as the early-to-mid seral dominant after the 2002 drought (Studies 69-21A, 69-21B, 69-21D, 69-21F, 69-31, 69-58A, 69-58B, 69-58D, 69-94). Cheatgrass was present during this time, but showed little to no spread following drought. During the drought of 2012-2014, cheatgrass has spread into many burned and unburned Wyoming big sagebrush communities, and has displaced both perennial grasses and winterfat (Studies 69-1, 69-1E, 69-1F, 69-2B, 69-3, 69-3B, 69-21A, 69-21F, 69-31, 69-56, 69-58A, 69-58B, 69-72B, 69-94, 69-94D, 69-94E, 69-100B, 70-3D, 70-8A, 70-8B, 70-9B, 70-10A). Similar to other sagebrush communities, the spread of cheatgrass into Wyoming big sagebrush accelerated during the last drought. Fire frequency is expected to be higher with the presence and spread of cheatgrass. Wyoming big sagebrush communities of the Anthro Plateau landtype association currently have moderate to high susceptibility to cheatgrass invasion. This places the communities at risk of loss of resilience, capability, and function.

Montane black sagebrush, mostly found within the Anthro Plateau and North Flank landtype associations, have been less impacted by anthropogenic activities than big sagebrush communities. This is due to marginal soil conditions and inherent low vegetation productivity. Black sagebrush has seldom been targeted with treatments to reduce sagebrush and enhance herbaceous production. Small treatments prior to 1980 were for study. Black sagebrush returned to pre-treatment levels within 25 years. (Goodrich 2001, Studies 67-25, 68-1). A few acres of black sagebrush were burned inadvertently during prescribed fire treatments since 1995 (Studies 68-72A, 68-11F, 68-96E). At two burned sites, black sagebrush canopy cover was two percent and one percent, 17 and eight years post-fire respectively. This indicates a relatively long return interval (Study 68-72A, 68-11F, 68-96E). On the Anthro Plateau landtype association, lop-and-scatter treatments removed pinyon, juniper, and Douglas fir trees from black sagebrush communities threatened by conifer displacement (Huber and others 2010). Overall, most montane black sagebrush communities are in late seral stages, with sagebrush canopy cover ranging from three to 30 percent - with an average of 20 percent (Studies 4-4, 4-29B, 67-25A-B, 67-52, 67-70B, 67-86D, 68-1A, 68-21A, 68-62E, 68-69A,C-D, 68-72D, 70G-2). Shrub die-back following drought has also been documented, but no invasive annuals are present and spreading (Study 68-21, 69-54). In addition, sagebrush canopy cover averaged higher (20 percent) in communities located above critical winter range for wild ungulates than in communities located within critical winter range (13.5 percent). This likely indicates shrub decrease from wild ungulate browsing (Studies 5-63D, 4-2J, 4-26B, 69-48B, 69-54A, 68-77A). Montane black sagebrush communities of the Anthro Plateau and North Flank landtype associations currently show resilience in late seral communities, burned, and drought-impacted communities (Studies 68-21A, 68-72A, 68-96E). Cheatgrass and other annual invasive plants are either absent or have minor presence, with no indication of spread or increase.

Photo of study 68-69B. Wild Horse Ridge. 2005 and 2016. Before and after photos of lop-and-scatter treatment of black sagebrush community (flat below).

Black sagebrush communities of the South Face landtype association are located within or just above the pinyon-juniper belt. At eight sites, canopy cover averages 20.5 percent, similar to those found in montane communities (Studies 31-89A, 32-64, 45-6T, 45-8D-E,L,N-Q,S, 45-19C-D,H). Two of eight sites have cheatgrass present, with additional spread predicted (Studies 45-6U, 45-8). Cheatgrass has been present on steep south-facing slopes for a couple of decades, however spread into black sagebrush communities of different aspects and gentle gradients has occurred over the last 10 years (Studies 32-64, 45-6U, 45-8, 45-8R,U-V, 45-17D, 45-19I). Spread appears to increase during severe drought. Similar to mountain big sagebrush, black sagebrush communities of the South Face landtype association have moderate to high susceptibility to cheatgrass. Cheatgrass has the capacity to change sagebrush canopy cover by increasing fire occurrence to levels that would preclude the reestablishment of sagebrush after fire. In conclusion, black sagebrush of the South Face landtype association are at risk of loss of resilience, capability, and function due to the presence and spread of cheatgrass. No black sagebrush communities have introduced seeded grasses within them.

At low elevations, black sagebrush of the Green River landtype association has persisted on favorable aspects and soil types (Goodrich 2005). These communities are currently in satisfactory condition in regards to shrub canopy cover. Black sagebrush canopy cover averages 11 percent, which within the expected range for communities with low precipitation (Studies 75-33, 79-14B, 79-17, 78-28). Pronghorn antelope browsing may reduce sagebrush cover at some sites. Cheatgrass is present in black sagebrush of the Green River landtype association (Study 79-17). Similar to communities of the South Face landtype association, black sagebrush has moderate to high susceptibility to cheatgrass invasion. This places sagebrush at risk of loss of resilience, capability, and function.

Influences of Drivers and Stressors

Fire is an important ecological driver in sagebrush communities. Where conifers are present, sagebrush communities decrease in shrub canopy cover and density. This occurs as conifers increase in density and canopy cover in the absence of fire. Under natural conditions, fire occurred often enough to maintain sagebrush where it is presently found. Fire regimes in sagebrush communities are moving away from the natural range of variation and fire frequency has decreased in mountain big sagebrush communities. Fire frequency has increased or is conditioned to increase in communities with historically low fire frequencies, such as Wyoming big and black sagebrush. Fire suppression policies, grazing management, and other anthropogenic activities have lowered fire frequency in mountain big sagebrush. Notable trends include many mountain big sagebrush communities with canopy cover exceeding 20 percent, and widespread encroachment and displacement of conifers in communities adjacent to or integrating with conifers. Some sagebrush communities have transitioned into conifer forest or woodland types with prolonged absence of fire. Wildfire from natural ignitions continue to occur on montane landscapes, but not likely at the rate documented historically (Stewart 1911, Graham 1937, Hitchcock 1910, Heyerdahl and others 2011). To compensate for lowered fire frequency, treatments were implemented to maintain montane sagebrush communities. Since 1995, prescribed fire and lop-and-scatter treatments were implemented to imitate natural fire frequencies or to curtail conifer encroachment within sagebrush communities (Huber and others 2010, Bistryski 1996, Bistryski 2000, Kirkaldie 2009, Kirkaldie 2009b, Kirkaldie 2009c). These prescriptions, including a few wildfire events, have curtailed conifer encroachment within a few thousand acres of sagebrush. Additional treatments have been planned, analyzed, and are awaiting implementation. During the next plan period, natural fire frequency of montane sagebrush communities (Parks Plateau, Anthro Plateau, Avintaquin Canyon, Strawberry Highlands landtype associations) is expected to remain outside the natural range of variation, but management prescriptions would be implemented to maintain these communities.

On the South Face, Glacial Canyon, and Stream Pediment landtype associations, cheatgrass is spreading into mountain big and black sagebrush communities. Prior to 2002, mountain big and black sagebrush returned to pre-disturbance conditions, in absence of cheatgrass and within expected return intervals described for the natural range of variation. Since 2002, cheatgrass has established and spread into unburned and older and recent burns. The presence and spread of cheatgrass in these communities will likely increase fire frequency to levels that would diminish sagebrush’s ability to reestablish and persist within the community. On the other hand, cheatgrass has not established in burns where introduced seeded grasses dominate the understory (Studies 38-16, 41-8A-F,I, 41-10A--E, 42-1A,C-G,I-K, , 42-13A-C, 43-1A-B, 43-7B-C). During the next plan period, cheatgrass is expected to increase in sagebrush communities with native herbaceous understories of the South Face, Glacial Canyon, and Stream Pediment landtype associations, especially following fire and drought.

Under the natural range of variation, Wyoming big and black sagebrush communities of low elevation have low fire frequency and sagebrush return intervals. Since 2012, cheatgrass has established and is spreading into many of these communities. Cheatgrass has the ability to alter fire regimes of low-elevation sagebrush by creating continuous fuel between shrubs that would otherwise be separated by bare or rocky soil, pediments, or sporadic herbaceous cover. As a result, sagebrush communities become more prone to fire. When infested sites eventually burn, cheatgrass out competes and displaces native vegetation and dominates cover. Fire frequency increases, often at intervals that preclude shrubs from reestablishing. Further spread of cheatgrass is predicted during the next plan period, which will make more communities prone to fire.

Drought is a natural disturbance that occurs regularly in sagebrush communities. Under the natural range of variation, response mechanisms to drought have successfully maintained sagebrush on western landscapes. Most drought events minimally impact shrub cover or density, but high mortality of sagebrush occasionally occurred in Wyoming big sagebrush during the most severe and prolonged drought events (Goodrich 2005). In most cases, surviving shrubs responded with greater foliage and seed production with the return of abundant and timely precipitation. Big and black sagebrush is naturally resilient to drought events if annual invasive plants are not present. Where annual invasive plants are present, these plants are beginning to reduce the sagebrush community’s resiliency to drought. Cheatgrass spread has increased during drought events in sagebrush communities of low-to-mid elevation. During the next plan period, cheatgrass is expected to continue to increase during drought events in sagebrush communities of the

Green River, South Face, Glacial Canyon, Stream Pediment, Dry Moraine, Antelope Flat, and Structural Grain landtype associations. Mountain big and black sagebrush of the Parks Plateau, Anthro Plateau, Avintaquin Canyon, North Flank, and Strawberry Highlands landtype associations are likely to be more resilient and recover from drought. This is because cheatgrass and other invasive annual plants are not expected to spread into these communities.

Livestock grazing has impacted sagebrush canopy cover in many communities. Historically, heavy livestock grazing depleted and/or reduced vigor of native herbaceous understories, and increased or accelerated the increase of sagebrush canopy cover. To maintain or maximize forage production in the most productive sagebrush communities (for example, mountain big sagebrush) mechanical, prescribed fire, and herbicide treatments were implemented to remove sagebrush. This included plow and nonnative grass seeding treatments of thousands of acres of mostly mountain big sagebrush. Since 1980, most sagebrush treatments were prescribed fire. Sagebrush returned to, or is currently returning to, pre-treatment cover percentages within sagebrush return intervals of the natural range of variation. Sagebrush canopy cover is expected to remain within the natural range of variation under light to moderate grazing intensities. In sagebrush communities with heavy grazing intensities, sagebrush canopy cover may surpass normal ranges found under the natural range of variation. Browsing of low elevation Wyoming big and black sagebrush by domestic sheep occurs in the Green River landtype association, but browsing intensity has not been heavy enough to induce a downward trend in canopy cover of these sagebrush taxa. Browsing of sagebrush by domestic sheep is not expected to increase during the next plan period.

Most wildlife species minimally impact sagebrush condition or demonstrate the capacity to reduce shrub canopy cover. However, during eras of high population, pronghorn antelope and mule deer have decreased Wyoming big and black sagebrush cover (Goodrich and Zobell 2012). Populations of these ungulates have remained relatively constant over the last couple of decades. Although areas of heavy use are documented, current populations of pronghorn antelope and mule deer are not sufficient to deplete sagebrush cover across large landscapes. During the next plan period, pronghorn antelope and mule deer populations are predicted to remain relatively constant. Although elk populations have increased substantially over the last 30 years, current populations are not sufficient to deplete sagebrush canopy cover across the landscape. Based on current trend, elk populations are predicted to increase during the next plan period. As populations rise, elk use in sagebrush communities is predicted to increase, which indicates elk to be a potential stressor of these communities.

Recreation travel is a stressor of sagebrush canopy cover and structure. Considerable increase in all-terrain vehicle use and other off-road travel has occurred in these communities over the last 30 years, resulting in miles of new unauthorized trails and roads in sagebrush. Off-road travel reduces sagebrush canopy cover by breaking or crushing shrubs with vehicle wheels and frames. Recreation travel expands distribution and increases the spread of noxious weeds and invasive annuals in sagebrush. Recreation travel, with its impacts, is expected to increase during the next plan period. Other recreation uses are expected to minimally affect sagebrush overall and impacts would be site specific. These uses include camping, hiking, and horseback riding

Oil and gas exploration and development is a present and foreseeable stressor of sagebrush of Anthro Plateau landtype association. Numerous well pads, service roads, and pipeline corridors have been constructed within sagebrush communities over the last 15 years, with additional infrastructure planned. These disturbances reduce sagebrush canopy cover and open niches for the spread of annual invasive plants. Oil and gas activities have increased the introduction, establishment, and spread of noxious weeds and invasive annuals into the area. As the demand for energy increases, expansion of oil and gas exploration and production is predicted to increase during the next plan period.

Climate Related Risks and Trends

Effects to sagebrush structure due to climate change vary between and within sagebrush taxa. Mountain big sagebrush's sensitivity and vulnerability to climate change is likely moderate to high, with an adaptive capacity of low to moderate (Padgett and others 2016). In a warming climate, mountain big sagebrush below 8,000 feet elevation (i.e., drier environments) would be most susceptible to climate change. Under these conditions, shifts in herbaceous compositions from perennial to annual invasive plants are predicted. Fire frequency would likely increase, and woodland and Wyoming big sagebrush communities may shift upslope. Resilience of these communities would be low and with widespread loss in canopy cover and density.

Mountain big sagebrush communities above 8,000 feet are most likely to persist under a warming climate but may be constantly stressed with increased drought. These communities may become more susceptible to invasive annual plants like their lower-elevation counterparts, if cheatgrass migrates upslope. Resilience of these communities may diminish under a warming and drying climate. Wyoming big sagebrush communities are most vulnerable to a warming and drying climate. Sensitivity to climate change is high, adaptive capacity is low, and vulnerability is very high for these communities (Padgett and others 2016). A warming and drying climate would increase the frequency and duration of drought in Wyoming big sagebrush. Under these conditions, shrub die-back would occur more frequently, seed production would likely diminish, seedling establishment and survival would become increasingly difficult, annual invasive plants would likely dominate the understory, and fire frequency would likely increase. The expansion of desert shrubs into Wyoming big sagebrush is also predictable, but may not occur with increased fire frequency. Resilience of these communities would be low, and widespread loss of sagebrush canopy cover and density are predicted to occur. Black sagebrush communities are considered moderate to high in both sensitivity to climate-related risks. Black sagebrush adaptive capacity is considered moderate in vulnerability (Padgett and others 2016).

Black sagebrush has a wide elevational amplitude and inherent low productivity. Therefore, these communities have the greater ability to adapt or resist changes of a warming and drying climate. Under warming conditions, black sagebrush is susceptible to more frequent and persisting droughts, but these communities appear more fire resistant than Wyoming big sagebrush. These communities are susceptible to cheatgrass, but less than many big sagebrush communities. Community resilience may be lower with a warming and drying climate, but black sagebrush is predicted to persist at many sites.

Comparison of the Natural Range of Variation and Current Conditions

During the next plan period, the structure of sagebrush communities could be susceptible to changes in conifer displacement, fire frequency, drought, invasive annual plants, and climate related risks. For decades, conifer encroachment in and displacement of sagebrush communities has progressively increased with decreased fire frequency. This increase in conifer encroachment suggests a low to moderate departure from the natural range of variation. This trend is most common in mountain big, Wyoming big, and black sagebrush communities within and above the pinyon-juniper belt. The trend is also common in upper elevations of mountain big and black sagebrush, near Douglas fir and ponderosa pine forests. Wildfire events during the last 30 years were not frequent enough to reverse this trend, but there has been an increase in prescribed fire and lop-and-scatter treatments implemented across the Ashley that has helped curtail conifer encroachment into some sagebrush communities. If fire frequency remains constant during the next plan period, additional treatments would be necessary to curtail conifer displacement of sagebrush and help neutralize current trend. However, fire frequency may increase in many sagebrush communities if the climate becomes warmer and drier, if severe and prolonged drought events become more common, and if the establishment and spread of annual invasive plants increases.

Currently, cheatgrass is present and spreading in Wyoming big sagebrush of the Green River, Antelope Flat, Structural Grain, and Anthro Plateau landtype associations. Cheatgrass is doing the same in mountain big and black sagebrush (below 8,000 feet) of the South Face, Glacial Canyon, Dry Moraine, Structural Grain, and Stream Pediment landtype associations. Cheatgrass, exacerbated by drought and fires, is the primary stressor moving or likely to move sagebrush communities away from the natural range of variation. During the next plan period, additional sagebrush communities of lower-to-mid elevations are predicted to move outside the natural range of variation. If the climate become warmer and drier and droughts do become more frequent, the rate of spread is predicted to increase. If cheatgrass spread increases in sagebrush during the next plan period, fire frequency is also predicted to increase. This increase may prohibit sagebrush from reestablishing and persisting within its natural range.

Most mountain big and black sagebrush communities (above 8,000 feet) of the Parks Plateau, North Flank, Anthro Plateau, Avintaquin Canyon, and Strawberry Highlands landtype associations are currently within, or of low departure from, the natural range of variation. Sagebrush of higher elevation show resilience following disturbances. Conifer encroachment and displacement occurs within some communities, but not of the magnitude within and directly above the pinyon-juniper belt. Natural fire intervals are likely outside the natural range of variation, but prescribed fire and lop-and-scatter treatments have been implemented during the last 30 years to mimic fire effects and conserve intact communities. Drought has minimally affected sagebrush canopy cover. Sagebrush cover is returning within the expected return intervals for mountain big and black sagebrush. Invasive annual plants are rare to non-existent and are not predicted to increase during the next plan period. Sagebrush canopy cover is similar to the natural range of variation in areas where was sagebrush plowed and seeded, sprayed with herbicide, and treated with prescribed fire. These areas are at a low departure from the natural range of variation, and resilience to invasive annuals is high and is expected to remain high during the next plan period. If the climate becomes warmer and drier, these communities may become more stressed, more susceptible to invasive annuals, and less resilient.

Other stressors of sagebrush may impact the structure of sagebrush communities and may contribute to the spread of invasive annual plants. These stressors include livestock grazing, recreation travel, wild ungulate use, and oil and gas exploration. However, these stressors can be or have been appropriately managed or mitigated to maintain sagebrush communities within or near a low departure of the natural range of variation.

Existing conditions and current trends indicate that the structure of sagebrush communities of higher elevations are within or of low departure from the natural range of variation. These conditions and trends are expected to remain so during the next plan period until long-term monitoring indicates otherwise. Structure of sagebrush communities below 8,000 feet are more susceptible to drought, fire, and invasive annuals than those of higher elevation. Resilience of these communities are low to moderate. Although many sagebrush communities are currently within or of low departure from the natural range of variation, many of these are likely to trend away from the natural range of variation during the next plan period.

Pinyon Juniper Woodlands

Description of the Natural Range of Variation

The primary disturbance of landscape change within the pinyon-juniper woodland and sagebrush ecosystem complex was fire. The type of fire depended on the existing understory vegetation and density of pinyon and juniper (Tausch and Hood 2007). As mentioned in the “Composition” segment, spreading, low-intensity fires had a very limited historical role in shaping persistent pinyon-juniper woodlands (Romme and others 2007). Instead, fires were generally stand replacing—spreading crown to crown, especially when wind-driven, killing most or all trees (Romme and others 2007): Historical fire rotations were generally very long, approximately two to six centuries.

Stand dynamics in persistent woodlands were also driven "by climatic fluctuation, insects, and disease…" (Romme and others 2007). Over the past 18,000 years or so, climate change drove pinyon-juniper woodland expansion and contraction[10] and structure changes such as crown closure and tree density (Tausch 1999).

As described in the "Composition" section for pinyon juniper woodlands, plant diversity is generally at risk when pinyon-juniper reaches crown cover levels of 20 percent. Beyond 40 percent crown cover of pinyon-juniper, the understory was greatly depleted. Studies in the pinyon-juniper on the Ashley National Forest indicate that depletion of understory species is associated with loss of resilience or ability of the native community to recover after fire. Without an abundance of resilient perennial herbaceous species and sprouting shrubs, burned pinyon-juniper sites are vulnerable to invasive annuals.

Current Status and Trends

During the 20th century, most persistent pinyon-juniper woodlands throughout the West have increased in tree density and crown cover (Romme 2007). Tausch and Nowak (1999) concluded that, "pinyon-juniper woodlands in the Great Basin generally had more open areas" during the Little Ice Age. Crown cover increased following the Little Ice Age with the combination of climate change and settlement impacts (Tausch and Nowak 1999).

Historical patterns and future trends described by Tausch (1999) for the Great Basin seem highly applicable to the pinyon-juniper belt of the Structural Grain LTA of the Green River Corridor, and includes an increase in crown closure and density of pinyon-juniper woodlands since European settlement (Tausch 1999). Tausch (1999) suggested that there is potential for additional large, stand-replacing crown fires in pinyon-juniper woodlands over the next 150 years, due to current trends toward larger trees with increasing crown cover.

Because of large stand-replacing fire, structure for pinyon-juniper has changed greatly over much of the Structural Grain landtype association, due to the change in dominance from woodland species to herbaceous species. The Mustang Fire of 2002 set back approximately 65 percent of pinyon-juniper forest on this landtype association (almost 12,000 acres) to early successional structure stages; 60 percent is currently in the perennial forb and grass successional stage.

In the future, fire regimes will be different than they have been since pinyon-juniper came to dominate this landtype association, especially in areas dominated by invasive annuals. Succession to mature and old stands of pinyon-juniper commonly takes 200 years or longer. However, fire intervals that will prevent mature pinyon-juniper from dominating large areas are a likely future trend in this area. This prediction is partially because cheatgrass is a common and locally abundant species and with high levels of recreation activity providing sources of ignition.

On the Red Canyon landtype association, closed canopy stands generally do not develop until the late-seral stages. Although fires of recent decades have been small in this landtype association, the massive area burned by the Mustang Fire of 2002 carried across areas of the Red Canyon landtype association. This fire gained sufficient intensity on the Structural Grain landtype association to carry onto the Red Canyon landtype association. This fire burned with stand-replacing intensity, changing structure to perennial forb and grass successional stage. This change occurred on approximately 16 percent of the

[10] Miller and others 2008, Romme and others 2007, Miller and others 2005, Eisenhart 2004, Miller and Rose 1999a, Tausch 1999, Allen and Breshears, 1998, Miller and Wigand 1994

pinyon-juniper forest. Structure distribution for pinyon-juniper is still weighted heavily to the later successional stages at 78 percent, where older pinyon and juniper trees dominate.

In Red Canyon, mature pinyon and juniper currently dominate the southerly aspects of this canyon above the Flaming Gorge Reservoir. There are a few burned areas where tree density has been reduced. The largest of these is the area created by Hideout Fire (Study 5-44), which was prescribed to enhance bighorn sheep habitat. A small burn of about 80 to 100 acres occurred near Skull Creek, and a few small openings were created manually (Study 5-22). Vegetative trend in this landtype association is mostly stable with stand changing events rather infrequent.

Likewise, on the North Flank landtype association, pinyon and juniper can be expected to take 200 years or longer to return to mature stands of high percentage of crown cover after stand-replacement fires. Stand-replacement fires are common for pinyon-juniper areas and under-burns are unlikely. Less than five percent of the pinyon-juniper forest is currently in the perennial grass stage.

On the Anthro Plateau landtype association, fire ignition occurs frequently. However, a majority of ignitions result in only a single tree or otherwise small fires. The frequency of large fires in pinyon-juniper woodlands of this landtype association is likely in the order of 200 to 300 years. Consequently, later successional stage pinyon-juniper forest occurs at almost 92 percent on this landtype association.

Climate Related Risks and Trends

Although large-scale severe fires are typical fires for this vegetation type, the frequency of these types of fires appears to have increased throughout the West in the last 20 years (Romme 2007), contributing to structural changes in pinyon-juniper. This increase “may be a consequence of warmer temperatures and longer fire seasons, greater fuel continuity developing from increasing tree cover” but may occur in combination with other factors, such as invasion of highly flammable annual grasses like cheatgrass (Romme 2007). Pinyon-juniper woodlands are sensitive to climate change. There is likely to be a reduction in area dominated by pinyon-juniper as a result of increased drought and temperature (USDA 2016b, Rehfeldt and others 2012). There is indication that historical woodlands are already in decline on the Colorado Plateau (Arendt and Baker 2013) due to an excess of fire since Euro-American settlement.

Comparison of the Natural Range of Variation and Current Conditions

The pinyon-juniper woodland vegetation type is likely low to moderately departed from the natural range of variation. This is due to the presence of nonnative invasive annuals like cheatgrass and the potential for cheatgrass to alter the pinyon-juniper woodland fire regime and system, impacting pinyon-juniper woodland structure.

Desert Shrub

Description of the Natural Range of Variation

Community structure of desert shrubs are described at a landscape scale - spatially and temporally - and at a community scale by height, density, canopy cover, or some combination of these things. Land features, surface materials, soil, and in some cases, annual precipitation influence the structure of desert shrub plant communities. For example, spiny hopsage communities are tightly aligned with eolian sand and gravelly slope-wash colluvium. On erosional surfaces of the Green River Formation, Wyoming big sagebrush and shadscale communities dominate. However, where gravelly pediment surfaces occur, perennial grasses such as needle-and-threadgrass, Indian ricegrass, and bluebunch wheatgrass are abundant and occasionally dominate vegetation cover. Black sagebrush is found mostly on pediment surfaces where soils are largely undeveloped, shallow, and rocky. Winterfat communities are found in silty loam soils of the valley bottoms. Gardner saltbush grows in fine-textured clays with relatively high pH. Wyoming big sagebrush forms extensive communities across the landscape where annual precipitation is greater than eight inches. However, Wyoming big sagebrush intermixes with or integrates into other with salt-desert shrubs, like shadscale, where annual precipitation is at, or slightly less than, eight inches. Blaisdell and Holmgren (1984) found “plant communities [to be] normally distinct, but sometimes merge imperceptibly into one another.”

Drought, fire, and other disturbances can affect desert shrub structure in regards to canopy cover and shrub density. A relatively common climate event in desert shrub environments is drought. Depending upon its severity and duration, drought has caused die-back or die-off in many desert shrub communities including shadscale, Wyoming big and black sagebrush, budsage, spiny hopsage, winterfat, greasewood, and low yellowbrush (Nelson and other 1989). Shadscale is most susceptible to drought and die-back of shrubs occurs regularly (about every 10 years), especially when drought persists for two years or more (Nelson and other 1989, Blaisdell and Holmgren 1989). Shadscale also has a short return interval following die-back, which reflects a boom-and-bust cycle for the community. Hence, canopy cover of shadscale can vary greatly over a 10 to 20 year period (Study 72-1).

Gardner saltbush appears least susceptible to drought. Fire rarely occurs in desert shrub communities because vegetation production is low and bare soil to intermittent herbaceous cover between shrub interspaces is typical. In shadscale and similar communities, fire is infrequent at best, but possibly nonexistent in some communities (Knapp 1996). For Wyoming big sagebrush, a fire interval of 50 to 110 years is common but may occur no often than 200 to 350 years (USDA Forest Service 2016, Winward 1991, Mensing and others 2006).

Although several desert shrub communities are found within the Green River landtype association, only those of 2,000 acres or greater will be discussed in further detail in regards to structure under the natural range of variation. Spiny hopsage is a shrub that can form nearly pure stands but more often is a community dominant that intermixes with a number of other desert shrubs. Shrubs have an erect form; they are deciduous, spinescent, and diffusely branched (Blaisdell and Holmgren 1984). Height ranges from one to four feet, and canopy cover averages about 12 percent but can be as high as 20 percent (Studies 72-21B, 72-28, 74-2, 75-14B, 75-17A-B, 75-39, 75-46, 77-2, 78-9, 78-16, 78-16C, 78-18).

Shadscale seldom forms pure stands, but is often dominant and occasionally co-dominant in with many other desert shrubs. It’s a short-lived shrub and is more vulnerable to drought and excessive soil moisture than other salt-desert shrubs (Blaisdell and Holmgren 1984, Nelson and others 1989, Goodrich 2012). Shrubs are “compact, spinescent . . . [and] growing typically in dense clumps” (Blaisdell and Holmgren 1984). Height ranges from 8 to 32 inches (Blaisdell and Holmgren 1984). Average canopy cover of this shrub is about 7 percent but may reach as high as 15 percent. (Studies 72-1, 72-2, 72-21C, 72-24, 75-18B, 75-18E, 75-21B2-C, 75-23C-D, 75-28B, 75-32, 75-50B). Shadscale canopy cover is variable due to its sensitivity to drought and other disturbances (Nelson and others 1989).

Gardner saltbush is a low-growing subshrub with heights ranging from about four to 15 inches. Gardner saltbush forms nearly pure stands on clay soils “where few other plants have capacity to grow” but occasionally mixes with shadscale, winterfat, and bud sage (Goodrich and Zobell 2011). Gardner saltbush canopy cover can range from 10 to over 35 percent but averages 19 percent (Studies 6-43A-B, 6-16A, 71-11, 72-13, 72-18, 72-21D, 72-26, 72-31A-B, 75-4A, 75-21B1, 77-1).

Wyoming big sagebrush is the most common community of the Green River landtype associations. Shrubs range in height from two to three feet and their growth habit is rounded with uneven crowns. Where annual precipitation is at or slightly less than eight inches, Wyoming big sagebrush intermixes with salt-desert shrubs. Where annual precipitation is greater eight inches, Wyoming big sagebrush forms extensive communities. Wyoming big sagebrush canopy cover ranges from one to 22 percent, with an average of about 11 percent (Goodrich and others 1999).

Current Status and Trends

During the last 30 years, all desert shrub communities demonstrated resilience to drought, ungulate grazing, and other disturbances up to about the year 2002. Community structure was maintained or responded within natural range of variation parameters, concurrent with existing drivers and stressors. Drought events caused occasional die-back in communities of shadscale and Wyoming big sagebrush, but shrub canopy cover and density rebounded with the return of abundant and timely moisture (Goodrich 2005d, Goodrich and Zobell 2013). Since 2002, some desert shrub communities have become susceptible to invasive annual plants, while others show signs of susceptibility. Additionally, long-term monitoring documented the spread of invasive annuals during and shortly following periods of severe drought in 2002 and 2012 to 2014.

Gardner saltbush die-off or severe die-back has occurred over the last 15 years, due to the presence and increase of halogeton (Goodrich and Zobell 2011, Goodrich and Zobell 2009, Studies 72-13, 72-13G, 72-18A, 75-21B, 72-21D, 72-31A-B, 75-4A-B, 75-21B1-B4). Halogeton changes soil chemistry and ecology enough to make the soil incompatible for Gardner saltbush (Goodrich and Zobell 2011). Halogeton occupies sites where Gardner saltbush die-off has occurred. Drought is not considered a primary driver of saltbush die-off but may have accelerated the conversion. Livestock grazing is considered to have little to no influence on shrub die-off. This is because die-off occurred at grazed and ungrazed sites and Gardner saltbush is known to tolerate severe use by livestock and wild ungulates (Blaisdell and Holmgren 1984). Gardner saltbush communities are at high departure from NRV. Halogeton has also been documented in shadscale, winterfat, spiny hopsage, sticky yellowbrush, and Wyoming big sagebrush communities but has drastically increased specifically in winterfat and in some shadscale communities (Studies 72-1, 72-2, 72-13E-G, 72-14A1-A2, 72-18B-D, 72-28B, 72-31G, 72-43, 75-14A, 75-18A, 75-18D-E, 75-21A-C, 78-18A). Halogeton appears to have little to no influence of ecological processes of spiny hopsage, Wyoming big sagebrush, and most shadscale communities. However, additional monitoring will be required to make these determinations. Currently, shadscale, spiny hopsage, Wyoming big sagebrush, and other desert shrub communities are in satisfactory condition, in regards to community structure. These fall within or show low departure from the natural range of variation. Cheatgrass, however, is becoming more prevalent and is spreading into shadscale, spiny hopsage, winterfat, and Wyoming big sagebrush communities (Studies 6-16A, 6-43A-B, 72-8, 72-8B, 72-14B, 72-32, 75-23A, 75-23C-E, 75-32, 75-39A, 75-45A-B, 78-9, 78-13C, 78-16, 78-26).

Photo of study 72-14E. Green River Basin. Depiction of winterfat community with satisfactory structure and composition.

Wyoming big sagebrush and winterfat communities appear most susceptible to cheatgrass, but cheatgrass appears capable of establishing in all other desert shrub communities. The invasive annual can alter fire frequency and disturbance response sequences of desert shrubs, especially since most desert shrubs do not sprout following fire (Blaisdell and Holmgren 1984).

Influences of Drivers and Stressors

Invasive annual plants are the greatest threat to the structure of desert shrub communities because these invasive plants are capable of diminishing or eradicating desert shrub community drivers, such as shrubs and perennial grasses. Other stressors and drivers may also increase the spread of invasive annuals.

Drought is a natural disturbance that occurs regularly in cold deserts. Under the natural range of variation, response mechanisms to drought have successfully maintained desert shrubs, concurrent with most other stressors and drivers. Shadscale is most susceptible to drought, where shrub die-back often occurs following these events (Blaisdell and Holmgren 1984). Rapid increase in shadscale canopy cover occurs following the return of abundant and timely moisture. Gardner saltbush, and to some degree, winterfat appear relatively drought tolerant, with limited to no shrub die-back in drier-than-normal years. Nelson and others (1989) found that most desert shrubs are susceptible to drought, depending upon severity and duration of these events. High mortality of Wyoming big sagebrush and shadscale have recently occurred in 2002, and 2012 to 2014 (Goodrich 2005, Goodrich and Zobell 2013). In these cases, surviving shrubs responded with greater foliage and seed production with the return of abundant and timely moisture. The introduction and spread of invasive annuals have altered drought response mechanisms of some communities and drought may have facilitated the spread of invasive annuals. Since about 2002, Gardner saltbush has experienced widespread die-off because of halogeton’s ability to alter soil chemistry. Shrub die-off has been followed by an increase of halogeton, which then dominates the site (Goodrich and Zobell 2011, Goodrich and Zobell 2009). Drought may or may not accelerate halogeton spread into Gardner saltbush.

Halogeton is present in other communities and has shown increase and displacement of some winterfat, and increase in some shadscale. The long-term effects of drought and halogeton in these communities are not fully understood. Cheatgrass spread has increased during recent drought events, especially into Wyoming big sagebrush and winterfat communities. During the next plan period, desert shrubs are expected to respond to drought events similar to the natural range of variation if annual invasive plants are not present. Where present, annual invasive plants have and will continue to alter some desert shrub's ability to respond to drought and other disturbances. Gardner saltbush communities are predicted to decrease as halogeton spreads and new drought events occur. Some winterfat and shadscale communities may also reflect this trend. Cheatgrass in particular is predicted to increase in desert shrub communities, as new drought events occur.

Recreation travel is a stressor of desert shrub communities. The numerous network of roads in the Flaming Gorge National Recreation Area are likely to increase the spread of noxious weeds, halogeton, cheatgrass, and other invasive annual plants into the area. All-terrain vehicle use and other off-road travel has increased over the last 30 years. Off-road travel can decrease shrub canopy cover and density, and contribute to the spread of noxious weeds and invasive annuals. Recreation travel, with its impacts, are expected to increase during the next plan period.

Flaming Gorge Reservoir and the Green River watershed are likely the greatest source of noxious weeds and invasive annuals to the Green River landtype association. Seeds and plant materials are deposited along the shore line of the reservoir. Halogeton and other undesirable plant species are found in some low-gradient areas of the draw-down basin of the reservoir (Study 83-2B). Additional noxious weed and invasive annual infestations are predicted to occur during the next plan period.

Although fire is a feature of desert shrub communities, wildfire events are quite rare. Desert shrub communities have low to almost non-existent fire frequencies. Most desert shrubs do not sprout following fire and shrub return intervals for most are relatively long. Since 2002, cheatgrass has established and is spreading within many desert shrub communities, especially Wyoming big sagebrush and winterfat. This annual grass can change community structure by altering fire regimes of desert shrubs. Cheatgrass does this by growing within and filling shrub interspaces. This type of establishment creates continuous fuel between shrubs that would otherwise be separated by bare or rocky soil, pediments, or sporadic herbaceous cover. When infested sites burn, cheatgrass out competes and displaces herbaceous vegetation and creates extremely flammable monocultures. Fire frequency increases, often at intervals that preclude shrubs from reestablishing. Currently, most desert shrub communities are structurally intact due to the absence of fire, and shrub canopy cover and densities are within ranges described for the natural range of variation. Further spread of cheatgrass is predicted during the next plan period, which makes desert shrubs more susceptible to fire. As cheatgrass becomes more widespread, the incidence of fire increases. Frequent fire will significantly impact the structure of desert shrub communities, to the extent of possibly eliminating these communities from the ecosystem.

Livestock grazing is a stressor of desert shrub communities. Many desert shrubs are browsed by livestock. These shrubs include Gardner saltbush, winterfat, spiny hopsage, shadscale, gray molly, greasewood, black sagebrush, and to a small degree Wyoming big sagebrush (Goodrich 2012). This long-term and enduring practice has impacted shrub canopy cover, but browsing is not great enough to cause departure from NRV (Goodrich 2005, Goodrich 2005b, Goodrich 2005c, Goodrich 2013, Goodrich and Zobell 2013, Goodrich and Zobell 2013b, Goodrich and Zobell 2013c). Along with other forest uses, livestock have contributed to the spread of weeds and invasive annuals. Considerable loss of sagebrush, due to cheatgrass spread and increased fire frequency, may reduce livestock grazing when available forage is displaced by invasive annuals.

During eras of high population, pronghorn antelope have decreased Wyoming big sagebrush cover (Studies 72-18D, 75-23). Although areas of heavy use are documented, pronghorn antelope browsing was not sufficient to deplete sagebrush cover across large landscapes. On the other hand, cheatgrass can alter shrub cover and herbaceous compositions of Wyoming big sagebrush communities across large landscapes. Considerable loss of sagebrush canopy cover, due to cheatgrass spread and increased fire frequency, would negatively impact pronghorn antelope populations. During the next plan period, pronghorn antelope are predicted to remain relatively constant if Wyoming big sagebrush shrub cover remains relatively constant.

Climate Related Risks and Trends

Climate related risks could be a foreseeable stressor for the structure of desert shrub communities. Desert shrubs have “wide ecological distributions” in semi-arid regions and are adapted to harsh, dry climates. However, the resilience of desert shrub communities under a warming climate will likely decrease, due to the presence and spread of invasive annual plants (Padgett and others 2016). Desert shrubs’ sensitivity to climate related risks is considered a moderate risk. This risk is due to the desert shrubs communities’ capacity to adapt to dry, variable environments. The desert shrub communities’ vulnerability to climate related risks is considered moderate to high because of their susceptibility to invasive annuals such as halogeton and cheatgrass. Invasive annuals successfully function with drought and are predicted to function with warming climates. A warmer and drier climate would likely increase the frequency and duration of drought. Under these conditions, Gardner saltbush displacement by halogeton may increase. Die-back of desert shrubs would occur more frequently, seed production would likely diminish, and seedling establishment and survival would become increasingly difficult. Fire frequency would increase in desert shrub communities infested with cheatgrass. The accelerated spread of cheatgrass is a likely outcome following every drought and fire event. Fire intolerant desert shrubs could not reestablish if fire occurs too often. The presence and spread of invasive annuals would diminish desert shrub resilience and widespread loss of these communities is predicted. Because of this, the adaptive capacity of desert shrub communities is moderate (Padgett and others 2016). In addition, the expansion of desert shrubs into Wyoming big sagebrush is also predicted with a warming climate. However, expansion of these communities would be precluded with the presence of invasive annuals and more frequent fire.

Comparison of the Natural Range of Variation and Current Conditions

Invasive annual plants are the primary cause of desert shrub communities departing or trending away from the natural range of variation. At this time, community structure of most desert shrub communities are within or show low departure from the natural range of variation. These include spiny hopsage, Wyoming big sagebrush, gray molly, black sagebrush, and most shadscale and winterfat communities. A few shadscale and winterfat communities show low to moderate departure from the natural range of variation because of halogeton presence and spread. Some desert shrub communities are predicted to depart or further depart from the natural range of variation during the next plan period. Most Gardner saltbush communities show high departure from the natural range of variation because of their displacement by halogeton. This trend is predicted to continue during the next plan period. Within the plan period, decreases in shrub structure of many desert shrub communities is likely because of predicted increases in fire frequency due to the presence and continued spread of cheatgrass. Invasive annuals’ spread in desert shrubs is predicted to continue whether or not other stressors or drivers are present. However, if stressors such as a warmer and drier climate and/or frequent droughts occur, the rate of spread is predicted to increase. Standing alone, other stressors of desert shrubs would impact the structure of desert shrub communities but not cause a departure from the natural range of variation. These stressors include livestock grazing, recreation travel, and wild ungulate use. However, these uses could be a vector and contribute to the spread of invasive annual plants.

Landscape Disturbances

Geomorphic

Description of the Natural Range of Variation and Influences of Drivers and Stressors

Natural geomorphic disturbances in the Ashley National Forest include flood events, mass wasting (slope movement), and inherent and accelerated erosion on the landscape. The processes that formed and shaped the current landforms include periods of deposition, uplift, deformation, erosion, and glaciation. Present landforms were also formed by prehistoric mass wasting and flood events that have been identified and sometimes dated (Kowallis and Bradfield 2005; Munroe and others 2005).

Prehistoric landslides, often in slide complexes, have been documented by Kowallis in the Uinta Mountains on Mosby Mountain, Lake Mountain, and Little Mountain and in Dry Fork Canyon (Kowallis and Bradfield 2005.) Munroe describes landforms from large mass wasting events in Yellowstone and Lake Fork Canyons. These landforms include Raspberry Draw, the Cow Canyon landslide, and the slump and alluvial fan that originally formed Moon Lake (Munroe and others 2005). Geologists determined the recent Dry Fork debris flow contained Holocene and Pleistocene landslide deposits that were reactivated when they became saturated (Christenson, 1998). Prehistoric landslides have removed or dissected glacial moraine deposits in Rock Creek Canyon, Blind Stream basin, and the North Fork of the Duchesne River (Laabs and Carson 2005). Numerous former slumps that could fail again have been identified in Uinta Canyon (Herron 2007).

Both prehistoric and recent mass wasting are linked to specific geologic formations that are inherently erosive, or combinations of permeable and impermeable deposits. Many landslides in the Uinta Mountains are in areas where the permeable Bishop Conglomerate formation overlies a less permeable formation high in mudstone or siltstone, and the less permeable layer fails once saturated (Kowallis and Bradfield 2005.) Other major landslides commonly involve deformation of weak shales of the Doughnut Formation, overlying impermeable limestone and dolomite of the Humbug Formation (Munroe and others 2005). Some rock types are inherently erosive and unstable, including Hilliard, Manning, and Red Pine shales, and unconsolidated glacial deposits in the Uinta Mountains (Kowallis and Bradfield 2005). Shale and other inherently erosive parent materials also correspond to semi-barren or badland areas of accelerated erosion within several landtype associations. The Green River and Uinta Formations compose the Tavaput Plateau, located on the South Unit area of the Ashley National Forest. These lacustrine and alluvial formations include unconsolidated deposits, prone to accelerated erosion and mass wasting during storm events (Herron 2016).

Flood and mass wasting events have often resulted from saturation of slopes due to precipitation and snowmelt conditions. Historic records indicate summer or autumn thunderstorm events, or a combination of heavy precipitation on top of spring snowmelt, has triggered flooding and mass wasting (Christenson 1998; Giraud 2006). Flood events commonly include mass wasting in the form of slumps, slides, debris, or mudflows. This leaves deposits ranging from boulders to alluvial fans of sediment. Flooding on the Ashley National Forest has occurred within varied landscapes, mainly within six landtype associations. Mass wasting has been documented within many landtype associations, both prehistoric and current, including processes classified as falls, slides, slumps and flows and the slow downward movement of creep and solifluction. Solifluction is the slow downslope movement of water-saturated sediment due to recurrent freezing and thawing of the ground.

In recent history, a fatal debris flow and flood occurred in Sheep Creek. Heavy precipitation and snowmelt saturation in the summer of 1965 reactivated an old landslide and the debris flow was carried by the creek for approximately five miles. Under similar conditions, the spring of 1983 may have had the greatest number of drainages flooded. Impacts included altered stream and floodplain morphology, and damage to campgrounds, dams, spillways. and roads (Price and others 1983). The 1997 Dry Fork flood resulted from a breach in the Mosby Canal that added to spring saturation conditions. The failure began as a landslide and debris flow, and the failure undercut and reactivated old landslide deposits in the canyon. When the deposits breached the retained sediments and water flooded Dry Fork Canyon. The domestic and irrigation water supplies of the Vernal area were disrupted and the event left a large and unstable gully in unconsolidated deposits (Christenson 1998). Most recently, rapid snowmelt and a summer storm event triggered 2005 and 2007 debris flows in Uinta Canyon that damaged the U Bar Ranch and Wandin campground (Herron 2007).

In the future, areas that once failed through mass wasting or other instability issues could reactivate. Geologists examine many factors to evaluate potential for mass wasting, including fracture zones, earthquake potential, groundwater levels and the presence of springs and seeps, the stability of geology and soils, and slope gradient and length (Hylland 1996).

Table 17. Landtype associations with known natural geomorphic disturbances

Natural (Drivers) Geomorphic Processes and Conditions	Landtype Association
Flooding	Avintaquin Canyon, Anthro Plateau, Glacial Bottom, Red Canyon, Stream Canyon, North Flank
Mass wasting	Avintaquin Canyon, Anthro Plateau, Dry Moraine, Glacial Bottom, Glacial Canyon, Limestone Plateau, Moenkopi Hills, North Flank, Stream Canyon, Strawberry Highlands, Wolf Plateau
Creep or solifluction mass wasting	Glacial Canyon, Limestone Hills, Stream Canyon, Uinta Bollie, Wolf Plateau
Geologic hazards: mass wasting prone geology	Avintaquin Canyon, Antelope Flat, Anthro Plateau, Dry Moraine, Glacial Canyon, Moenkopi Hills, South Face, Strawberry Highlands, Wolf Plateau
Semi-barrens, barrens, badlands inherent to geology	Antelope Flat, Glacial Canyon, Moenkopi Hills,

Current Status and Trends

Human activities also result in geomorphic disturbances. Numerous dams, canals and ditch lines have been constructed on the Ashley National Forest. The only area that has not had infrastructure to impound and control water is the Tavaputs Plateau area of the Duchesne Ranger District. Present infrastructure provides electric power and water for irrigation and domestic use. However, this infrastructure also impacts watersheds with changes in channel morphology and migration, and sediment and water transport (Munroe and others 2005).

Several dams on the Ashley National Forest fall are in the High Uintas Wilderness and were constructed prior to the wilderness designation. Some of these dams are still present and have plans to be decommissioned (Milk Lake) or repaired (Atwood). Thirteen wilderness dams that had been constructed on mountain lakes between 1910 and 1930 have been breached, stabilized, and returned to natural water storage and flow. This project was done as a high lakes stabilization project, completed in 2010 (Utah Reclamation Mitigation Conservation Commission 2016). Other dams are operational and scattered across the Ashley.

The Stillwater and Hades Rhodes tunnels convey water from the Ashley National Forest Upper Stillwater Reservoir off forest to Strawberry Reservoir. Water from Strawberry Reservoir maintains flow for

Strawberry River and provides water for cities along the Wasatch Front (Utah Reclamation Mitigation Conservation Commission 2016).

The most concentrated human impact on the geomorphology of the Ashley National Forest may be in the approximate 25,900 acres of oil and gas leases on the South Unit of the Duchesne Ranger District. Berry Petroleum (Linn Energy) and Vantage Energy are permitted to develop approximately 165 oil and gas pad sites, approximately 57 miles of new roads and 21 miles of upgraded road area.

This development allows for 162 oil and natural gas pad sites, 57 miles of new roads and 20 miles of improved road area. An estimated 836 acres could be disturbed by this new construction (USDA Forest Service 2012). Impacts from development to date include altered topography and additional mass wasting and erosion. Topography has been changed by the construction of pads and roads, with deep benches cut in some ridges and alluvial fans. Soil erosion rates increase by continual road grading and storm events moving unconsolidated materials. This area is within the Anthro Plateau landtype association.

Table 18. Landtype associations with manmade geomorphic disturbances

Human Impacts (Stressors)	Landtype Association
Oil and gas development	Anthro Plateau
Infrastructure: active dams	Alpine Moraine, Greendale Plateau, North Flank, Parks Plateau, Trout Slope, Glacial Bottom
Infrastructure: canals and ditch lines	Alpine Moraine, Glacial Bottom, Glacial Canyon, Greendale Plateau, Limestone Hills, North Flank, Parks Plateau, Uinta Bollie
Climate related risks	All

Climate Related Risks and Trends

Climate modeling that applies to the Intermountain Region indicates a potential 4 to 5 degree Fahrenheit increase in average annual temperature over the next 40 years. Precipitation modeling indicates no clear trend and the trend is expected to be heavily influenced by local topography. The continued warming trend on the Ashley National Forest is expected to stress and change vegetation communities, allow invasive species to expand where they dominate, and result in more frequent and severe wildfires. All these changes could increase bare soil, surface runoff, and soil erosion rates. Changes in vegetation canopy cover and root support can also contribute to reduced slope stability and mass wasting events such as slides, flows and slumps. Potential shifts in precipitation patterns may include additional rain, more concentrated precipitation in seasons, and more rapid snowmelt. Changes in these patterns could result in and more frequent and severe flooding events, and more mass wasting, particularly with lithologies known to fail when saturated (USDA Forest Service 2016b).

Insects and Disease

Description of the Natural Range of Variation and Influences of Drivers and Stressors

Native forest insects and diseases are an integral part of forest ecosystems. Current ecological theories postulate endemic amounts of insects and disease are found in properly functioning forest ecosystems (Teale and Castello 2011). One key role is that insects and diseases often act as part of a natural disturbance regime.

Ecological disturbance creates and maintains biological diversity in the form of shifting mosaics of diverse communities and habitats across a landscape. Natural forest disturbance regimes have changed due to human influences such as fire suppression and fire exclusion without corresponding increases in timber management consistent with the natural disturbance regime. Forest disturbance regimes have also changed due to other human activities such as timber harvesting, mining, and grazing. Influences such as climate, fire suppression and exclusion, recreation, mining, timber management (e.g., policy, markets, availability of infrastructure), and grazing impact the availability of susceptible host type on the landscapes managed by the Forest Service. Multiple use management can directly and indirectly impact forest structure and composition (Fettig et al. 2014). Regardless of anthropogenic influences and management direction, natural disturbance regimes are part of the landscapes we manage and will continue to impact these ecosystems (DeRose 2014). In addition, the effects of individual insects and disease species may change due to effects of climate on stressor and driver organisms, their hosts, and environmental predisposition (Guyon IAP, in review).

As with wildland fire, insect outbreaks can interact with climate to generate long-term changes in vegetation structure (Allen and Breshears 1998). Insect- or pathogen-generated changes in vegetation structure can have multiple ecosystem-level consequences on nutrient cycling, hydrologic function, and geomorphic processes.

Endemic levels of insect and disease species are natural drivers of vegetative patterns and dynamics. All diseases currently active on the Ashley National Forest are native pathogens and could be considered endemic. Both endemic and epidemic populations of native forest insects can affect important ecosystem processes. These processes include the allocation of water and nutrients in a stand or a watershed, as well as forest structure and composition (Collins and others 2011, Mikkelson and others 2013). For example, endemic populations of bark beetles kill old, suppressed or otherwise unhealthy and stressed host trees. Tree mortality permits succession of understory plants and promotes new regeneration by opening gaps in the canopy. When dead trees fall, younger or previously suppressed understory trees gain access to an increased availability of resources including light, water, and nutrients (Hansen 2014).

Insect and disease epidemics can result in extensive tree stress and mortality. Mechanisms that trigger outbreaks of some insect species and pathogens are poorly understood, while others have been well studied. Nonetheless, suitable stand structures and sufficient amounts of preferred host vegetation must be available in a forested ecosystem to accommodate epidemics. Consequently, as forests change (e.g., structure, type, and species diversity), so do their associated insect and disease communities and the subsequent likelihood of serious impacts.

The impact of insect and disease epidemics can stress both host and nonhost vegetation, depending on their magnitude, frequency, and intensity. The impact can also reduce the capability of forests to provide ecologic and resource benefits. Although large scale host mortality caused by insect and disease epidemics are natural events, the cumulative effects of those events, and their interactions with other disturbance factors, can lead to undesirable impacts. These impacts are expressed in terms of short and long term.

Increased drought severity and frequency increases a tree’s vulnerability to both direct (reduced growth and mortality) and indirect (insect outbreaks, pathogens, and wildfire) impacts (Schlesinger and others 2016, Dale and others 2001, Weed and others 2013). For example, bark-beetle-caused tree mortality, combined with decades of fire suppression, can result in fires that are both geographically larger in size and more intense than normal.

Endemic populations of insects and diseases are important components of properly functioning ecosystems and help to sustain the resilience of stands to negative impacts. Effects of large-scale bark beetle and disease-caused tree mortality change composition, age, and stand structure of affected forests. Mortality altered fuels complexes and stands may become less resilient to invasions of undesirable species. The severity and intensity of these impacts may increase in the face of a changing climate

(Peterson et.al. 1998). Large mortality events, however, are often considered undesirable when they conflict with human resource objectives and ecosystem services (Malesky et al., in review).

Current Status and Trends

The insects and diseases in table 19 through table 25 are found on the Ashley National Forest. Most of these agents occur at endemic levels and pose a low risk to important ecosystem services. These agents do not require intensive management. Insect and disease species that meet the criteria listed below are denoted by an asterisk in the table. If insects and diseases meet thresholds in these criteria, management may be warranted to mitigate their impacts.

1. They pose an immediate threat to critical ecosystems and services
2. The present status of forest health, projected vegetative trends, or factors contributing to increased risk elevate concerns associated with undesirable levels of tree mortality
3. Effective management is feasible

Table 19. Native defoliators and host species on the Ashley National Forest

Category	Scientific Name	Host
Douglas-fir tussock moth	*Orgyia pseudotsugata* (McDunnough)	Douglas–fir, spruce, and all true firs
Western spruce budworm*	*Choristoneura freemani* (Freeman)	Douglas-fir, subalpine fir, white fir, blue spruce, and Engelmann spruce
Tent caterpillars	*Malacosoma californicum* (Packard) *Malacosoma disstria* (Hubner)	Aspen, willow, cherry, cottonwood, mountain mahogany, oak, alder, and birch

*Meets criteria listed above

Table 20. Native bark beetles and host species on the Ashley National Forest

Category	Scientific Name	Host
Western pine beetle	*Dendroctonus brevicomis* (LeConte)	Ponderosa pine
Roundheaded pine beetle	*Dendroctonus adjunctus* (Blandford)	Ponderosa pine
Spruce beetle*	*Dendroctonus rufipennis* (Kirby)	Engelmann and occassionally blue spruce
Douglas-fir beetle*	*Dendroctonus pseudotsugae* (Hopkins)	Douglas-fir
Mountain pine beetle*	*Dendroctonus ponderosae* (Hopkins)	Limber and ponderosa pine
Fir engraver beetle	*Scolytus ventralis* (LeConte)	True firs
Piñon engraver beetle*	*Ips confusus* (LeConte)	Piñon pine

*Meets criteria listed above

Table 21. Native bark beetles and host species on the Ashley National Forest

Category	Scientific Name	Host
Pitch mass borer	*Dioryctria spp.*	Piñon pine,

Table 22. Stem and branch diseases and host species on the Ashley National Forest

Category	Scientific Name	Host
Dwarf mistletoes*	*Arceuthobium spp.*	Douglas-fir, pines
Stem rusts	*Cronartium comandrae, Endocronartium harknessii*	Ponderosa and lodgepole pines

*Meets criteria listed above

Table 23. Root diseases and host species on the Ashley National Forest

Category	Scientific Name	Host
Annosum root disease	*Heterobasidion occidentale*	Douglas-fir and subalpine fir
Armillaria root disease	*Armillaria solidipes*	Douglas-fir, Engelmann spruce, subalpine fir, white fir, and pines
Black stain root disease	*Leptographium wageneri*	piñon pine

Table 24. Leaf and needle diseases and host species on the Ashley National Forest

Category	Scientific Name	Host
Aspen leaf spot	*Marssonina populi*	Aspen
Needle casts	*Lophodermella spp*	Lodgepole pine

Table 25. Declines and complexes and host species on the Ashley National Forest

Category	Scientific Name	Host
Aspen decline*	Several agents	Aspen
Subalpine fir mortality complex	Bark beetle-*Dryocoetes confusus* (Swaine); root disease *H. occidentale and several other agents*	True fir

*Meets criteria listed above

Given suitable stand conditions and susceptible landscapes, endemic populations of eruptive bark beetles can achieve exponential growth. This growth can affect hosts at the landscape level in relatively short periods of time (Safranyik et al. 2006; Lundquist and Reich 2014). One of the factors fostering epidemic insect population growth is an abundance of suitable hosts.

Spatial scales are measures of the aerial extent of disturbance events including the size of disturbance patches, the area per disturbance event per time period, and the percentage of area within a cover type affected. The scale of disturbances is reported in this assessment in acres and percent of acres affected by vegetation types. The temporal scale refers to the mean number of disturbance events per time period within a specified area. For this assessment, the temporal scale spans from 1997 to 2016 (19 years).

Aerial insect and disease detection surveys[11] are conducted annually to detect and monitor visible vegetation damage primarily caused by insects. Aerial detection surveys are intended to: detect new activity, monitor the trend of ongoing activity, provide general location information and subjectively rate levels of defoliation and mortality. The numbers do not reflect the current year's beetle population or number of currently attacked trees. Observers have just a few seconds to recognize, identify and document observed activity. Air turbulence, cloud shadow, haze, smoke, and observer experience can all affect the quality of the survey.

Information on forest susceptibility (hazard) and cumulative damage impact renditions of the major insect damage agents were not available in time to place into this assessment. These damage agents include mountain pine beetle, spruce beetle, and Douglas-fir beetle. The Ashley National Forest is currently working with Forest Health Protection staff to obtain and interpret these renditions and to provide susceptibility summaries at sub-watershed and forest levels to help inform Ashley National Forest personnel.

Dwarf Mistletoes: Older ground surveys found 58 percent of lodgepole pine stands on the Ashley National Forest, and eight percent of the ponderosa pine are infected with dwarf mistletoes (Hoffman 1978). More recent but spatially coarse data found 36 percent of the lodgepole pine and 16 percent of the ponderosa pine plots in forests across the state of Utah to be dwarf mistletoe infected. This data came from sampling completed by the Forest Inventory and Analysis group. The largest impact of dwarf mistletoes on their host tree is growth reduction. But dwarf mistletoes also cause some direct mortality, and can cause stress predisposing their host to damage by other agents. The impacts of dwarf mistletoes can be expected to increase under warming scenarios, drying scenarios, or both (Sturrock et al 2011).

Douglas-fir Beetle: Douglas-fir beetle is the most destructive bark beetle affecting Douglas-fir in western North American forests. At endemic levels this bark beetle infests blowdown and scattered trees of low vigor and poor health. Where there is an abundance of trees due to windthrow, populations can build rapidly and spread to adjacent healthy, green standing trees (Guinta 2015, Munson 2001). In drought conditions, Douglas-fir beetle outbreaks may be prolonged for several years. Catastrophic events (fire, windthrow, and avalanche) have led to exponentially expanded populations (Kegley, 2011).

The extent of forested area typed as Douglas-fir totals 47,900 acres and that typed as seral aspen to Douglas-fir totals 40,033 acres. It is estimated that Douglas-fir beetle-caused tree mortality has affected approximately 28,000 acres of this area, approximately 32 percent of the total area since 1997. Tree mortality has modified stand and age-class structure and altered the amount, composition, and

[11] Data provided by aerial detection survey (ADS) should be used only as an indicator of insect and disease activity. Surveys are often coarsely mapped and mapped polygons of insect and disease disturbance should be ground–truthed for actual location and causal agent. Polygons indicate locations of tree mortality, defoliation and/or other damage. Intensity of damage is variable and not all trees and areas mapped are dead or damaged. Using this data for purposes other than those for which it was intended may yield inaccurate or misleading results.

arrangement of living and dead biomass. In 2015, aerial surveys detected decreases in Douglas-fir beetle-caused tree mortality on the Ashley National Forest—specifically on the Vernal, Duchesne, and Roosevelt Ranger Districts. Mortality was estimated at two trees per acre, on 194 acres forestwide (USDA 2016). New pockets of Douglas-fir mortality were detected near Youngs Spring, Cub Creek, just north of Ashley National Forest boundary and Greendale Junction on the Flaming Gorge Ranger District.

Figure 9 shows the trend in Douglas-fir beetle-caused tree mortality by year since 1997. Tree mortality peaked in 2005. Douglas-fir beetle-caused mortality has affected approximately 28,000 acres of the forested area typed as Douglas-fir and seral aspen to Douglas-fir (approximately 32 percent of the total area) since 1997. Mortality has declined since 2008. Figure 10 shows the area of Douglas-fir beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015 and the locations of unaffected Douglas-fir stands.

Mapped acreages displayed in figure 9 and figure 10 were coarsely (and broadly) drawn resulting in area totals that encompassed other vegetation types, such as nonforest. Although this can be true for all aerial detection survey data, it is especially the case for mapped Douglas-fir beetle damage in year 2005. The actual footprint of the Douglas-fir type is clipped to the insect damage and totals 28,000 acres.

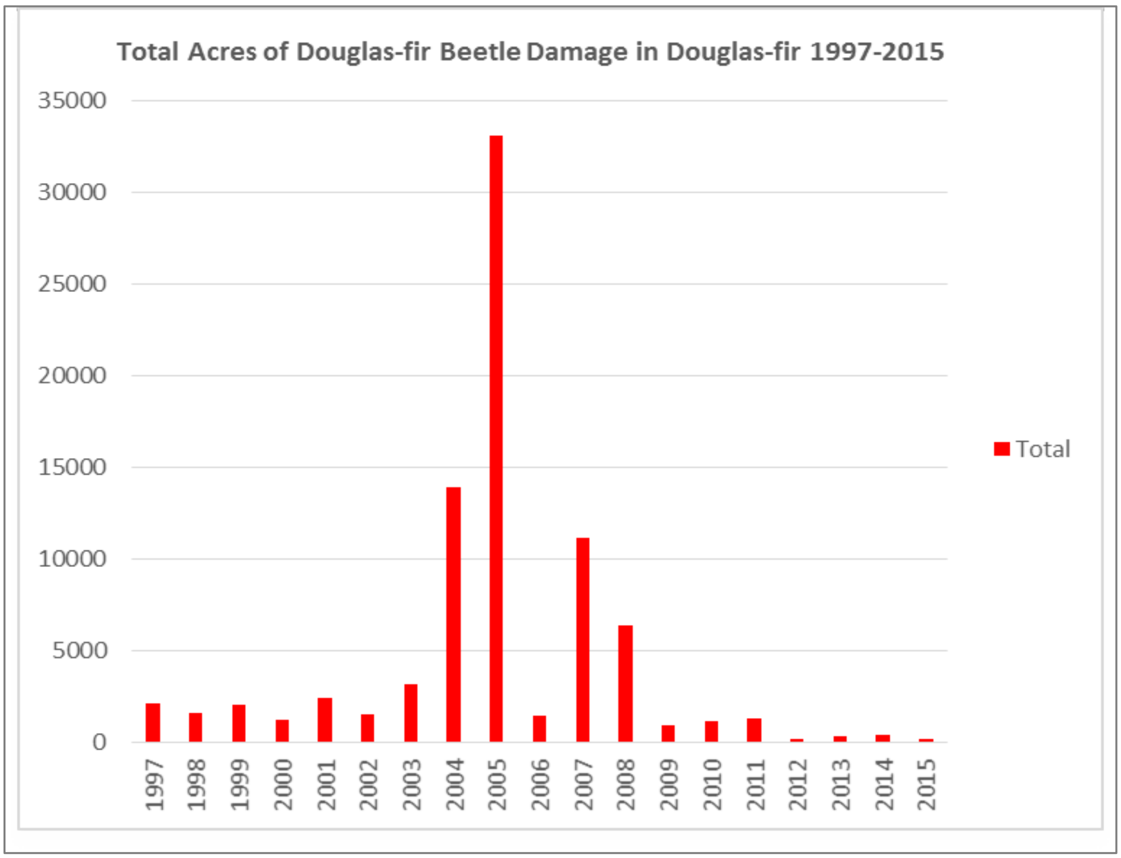


Figure 9. Trend in Douglas-fir beetle-caused tree mortality, by year, from 1997 to 2015

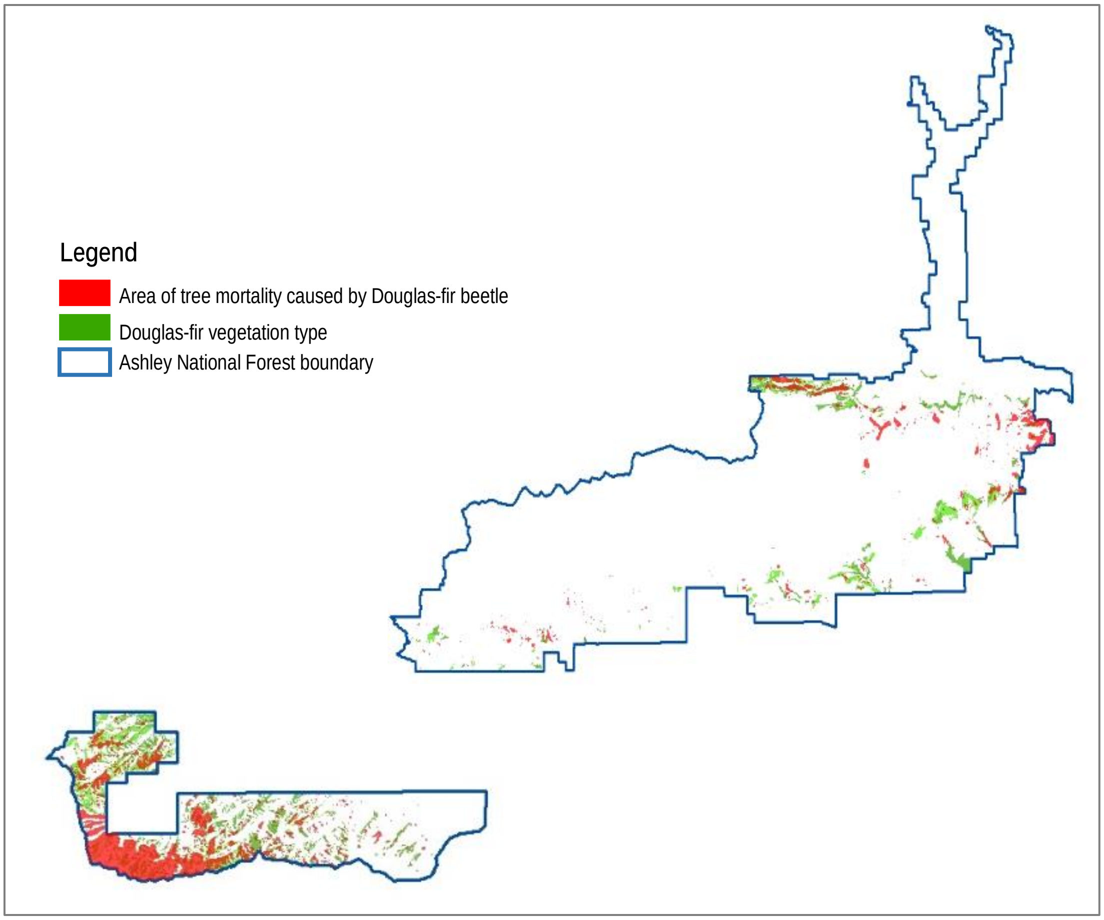


Figure 10. Douglas-fir beetle-caused tree mortality (shown in red) on the Ashley National Forest from 1997 to 2015. Green areas are the Douglas-fir vegetation type.

Mountain Pine Beetle: Mountain pine beetle is the most destructive bark beetle affecting pines in western North America. On the Ashley National Forest, most of the damage due to mountain pine beetle has been documented in lodgepole pine and ponderosa pine. Mountain pine beetle has recently expanded its range due to changes in climate (Bentz and others 2010). In areas of greater impact, tree mortality has modified stand and age-class structure and altered the amount, composition, and arrangement of living and dead biomass in both the pine and mixed conifer communities that were affected.

Mountain pine beetle affects lodgepole pine, ponderosa pine, limber pine, and bristlecone pine on the Ashley National Forest. The amount[12] of forested area considered persistent lodgepole pine is approximately 97,000 acres. Lodgepole pine also occurs among mixed conifer (spruce, lodgepole pine, Douglas-fir, subalpine fir) on about 358,000 acres. For ponderosa pine, the extent of forested area totals 45,770 acres.

On the Ashley National Forest, limber pine and bristlecone pine occur as a minor species in other forest types. The Ashley has 40 acres of mapped stands of limber pine forest type and no mapped bristlecone pine stands. Limber pine is estimated to occur on 287,500 acres of forested land, as part of the limber pine and subalpine fir habitat series (O'Brien and Tymcio 1997). Bristlecone pine is confined to a few isolated

[12] Acreage totals in this paragraph include the seral aspen types to the lodgepole, mixed-conifer, and ponderosa types

locations on the western Tavaputs Plateau of the South Unit of the Ashley National Forest (Goodrich and Neese 1986). The acreage it occupies is not determined.

Using spatial data since 1997, aerial surveys have mapped tree mortality caused by mountain pine beetle in the above forested types. It is estimated that mountain pine beetle has killed lodgepole pine trees on approximately 274,600 acres (approximately 60 percent of the area affected) of lodgepole pine forests and mixed-conifer forests in this period. In ponderosa pine, mountain pine beetle has killed trees on about 16,175 acres (approximately 35 percent) of ponderosa pine forests. Mountain pine beetle has also attacked and killed susceptible limber pine scattered throughout other forest types on approximately 2,900 acres (one percent of area affected) since 1997; no detection of mountain pine beetle-related tree mortality has been mapped on the 40 acres of limber pine identified on the Flaming Gorge District. Aerial surveys have also not detected incidents of beetle-caused tree mortality in bristlecone pine.

While decreases in mountain pine beetle-killed lodgepole pine were detected across the Ashley in 2015, increases in mountain pine beetle-killed limber pine were noted on the Duchesne and Roosevelt Districts (USDA 2016). Lodgepole pine mortality occurred on 577 acres forest wide in 2015. Limber pine mortality occurred on 862 acres, compared to 67 acres in 2014 (USDA 2016). Recent surveys in the summer of 2016, however, have since indicated a decrease to 200 affected acres where mountain pine beetle caused an estimated two trees per acre of mortality in limber pine[13]. Similar decreases in lodgepole pine and ponderosa pine mortality were observed at 22 acres in lodgepole pine and zero acres in ponderosa pine in 2016.

Figure 11 shows the trend in mountain pine beetle mortality in lodgepole pine on the Ashley from 1997 to 2015. Figure 12 shows acres of mountain pine beetle-caused tree mortality in lodgepole pine on the Ashley National Forest from 1997 to 2015 and the locations of unaffected lodgepole stands on the east and west Uintas of the Ashley. Not displayed are approximately 4,700 acres typed as mixed conifer on the South Unit of the Ashley, of which approximately 8 acres were reported as having mountain pine beetle-caused tree mortality.

[13] Findings have not yet been confirmed or summarized by Forest Health Protection staff.

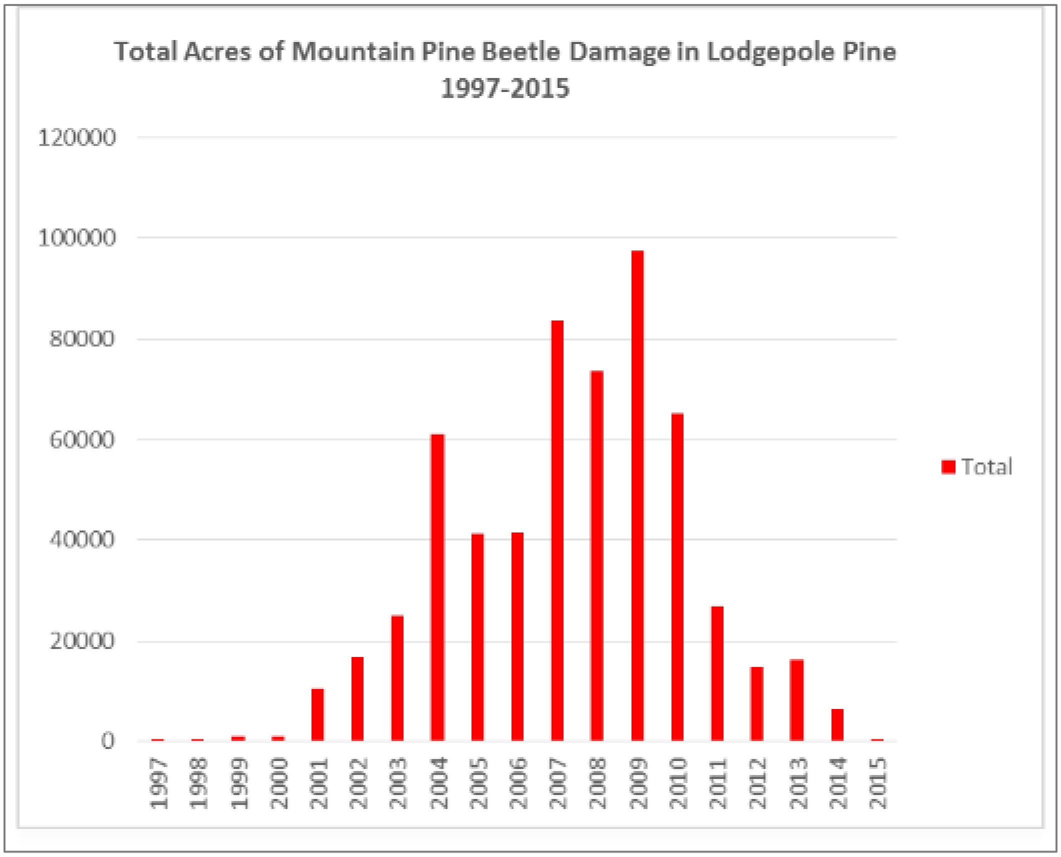


Figure 11. Trend in mountain pine beetle-caused lodgepole pine mortality from 1997 to 2015

Historically, mountain pine beetle has infested extensive areas of lodgepole pine and has likely been active in the ecosystem as long as lodgepole pine has existed (Roe and Amman 1972). Susceptible stands facilitate landscape-scale outbreaks because stand characteristics favor successful dispersal and aggregation by beetles (Fettig et al. 2014).

Mountain pine beetle has caused significant lodgepole pine mortality in the Uinta Mountains since 2000. It is estimated that mountain pine beetle has killed lodgepole pine trees on approximately 274,600 acres (approximately 60 percent) of lodgepole pine forests and mixed conifer forests with a lodgepole pine component since 1997 on the Ashley National Forest. Damage peaked on the Roosevelt-Duchesne Ranger Districts in 2004 and on the Flaming Gorge Ranger District between 2007 and 2009. Mortality has declined since 2011. Susceptible hosts still occur in the mixed conifer and lodgepole pine types and mortality is anticipated in the next planning period.

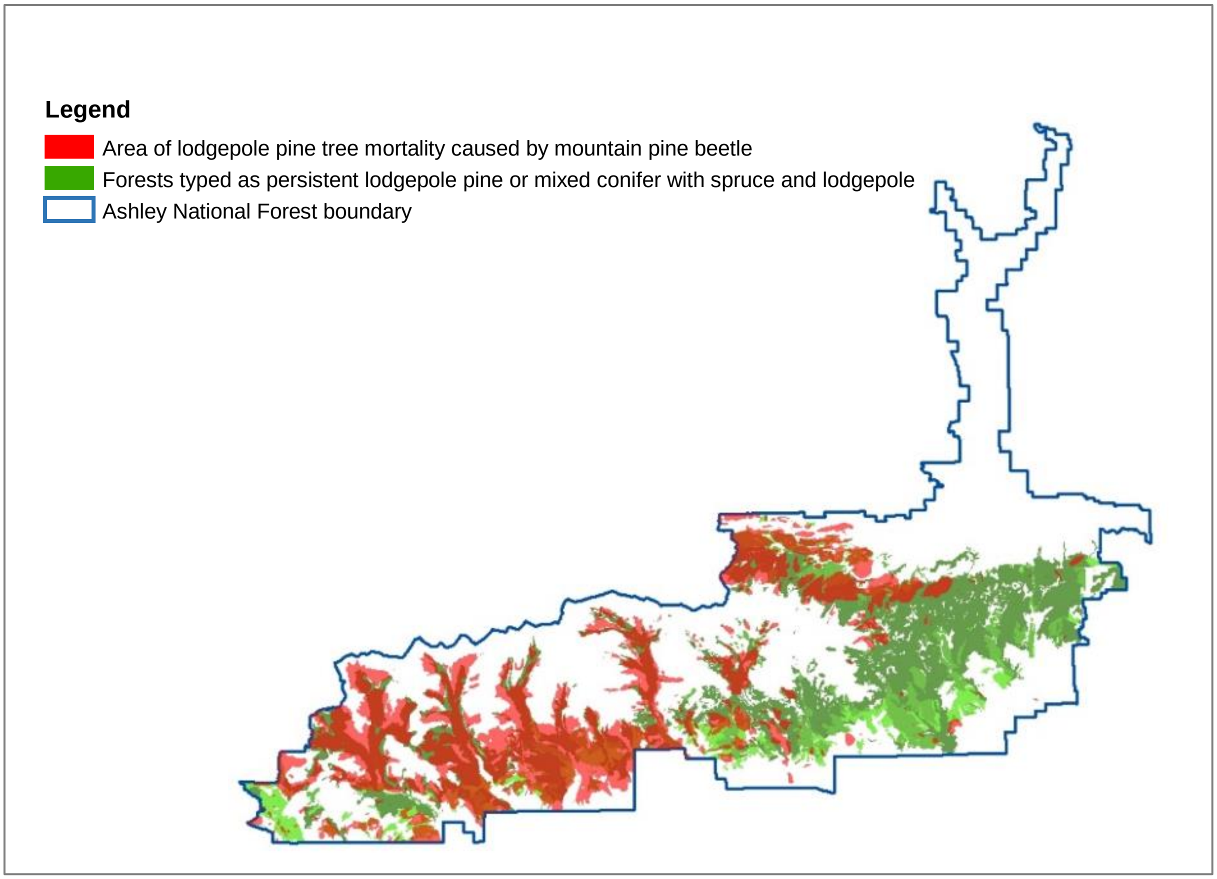


Figure 12. Acres of mountain pine beetle-caused tree mortality in lodgepole pine on the Ashley National Forest from 1997 to 2015

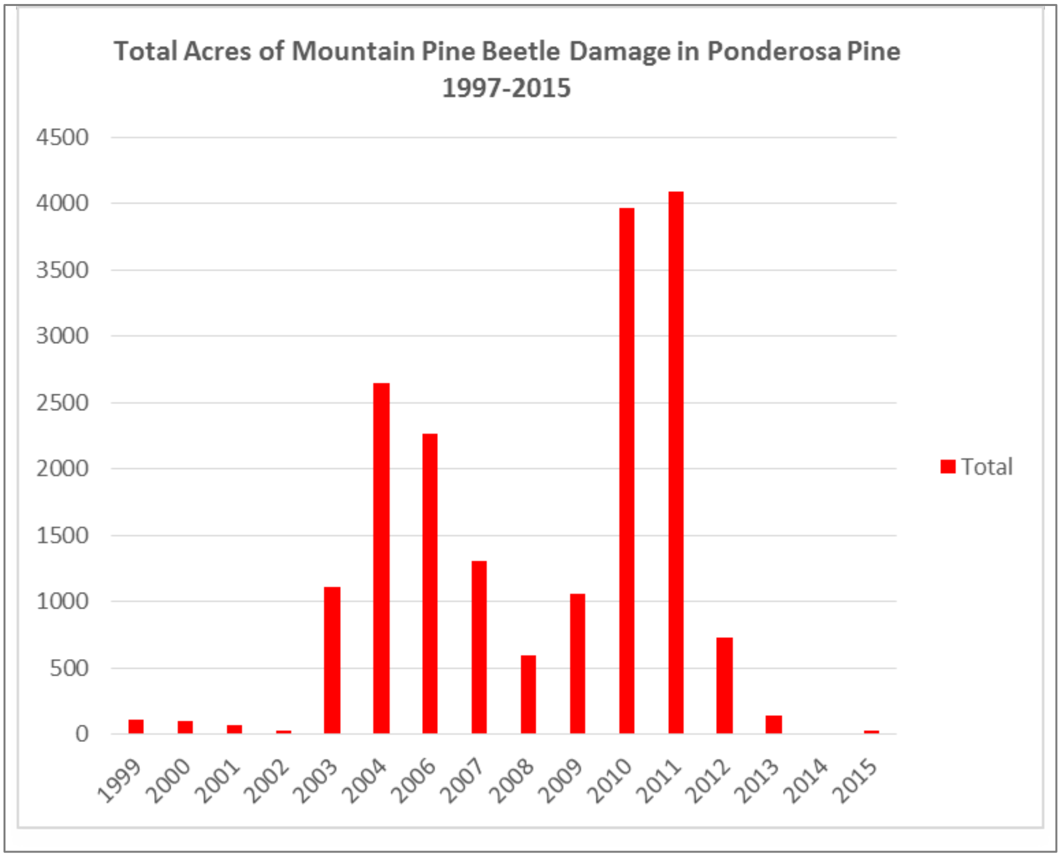


Figure 13. Trend in mountain pine beetle-caused ponderosa pine mortality by year since 1997 (no mortality was detected in years 1997 and 1998)

Figure 13 shows the trend in mountain pine beetle-caused ponderosa pine mortality by year since 1997. Damage peaked on the Roosevelt-Duchesne Ranger Districts in 2004 and on the Flaming Gorge Ranger District in 2011. Mountain pine beetle-caused tree mortality has affected approximately 16,170 acres of the ponderosa pine forest (approximately 35 percent) since 1997. Field observations and aerial detection survey data indicate that mountain pine beetle-caused mortality has declined. Susceptible hosts still occur in the ponderosa pine types and mortality is anticipated in the next planning period.

Figure 14 shows the area of mountain pine beetle-caused tree mortality in ponderosa pine on the Ashley National Forest from 1997 to 2015 and the remaining locations of unaffected ponderosa pine stands on the east and west Uintas of the Ashley. Not displayed in the map are approximately 600 acres typed as ponderosa pine on the South Unit of the Ashley, of which approximately 15 acres were reported as affected by mountain pine beetle within this period.

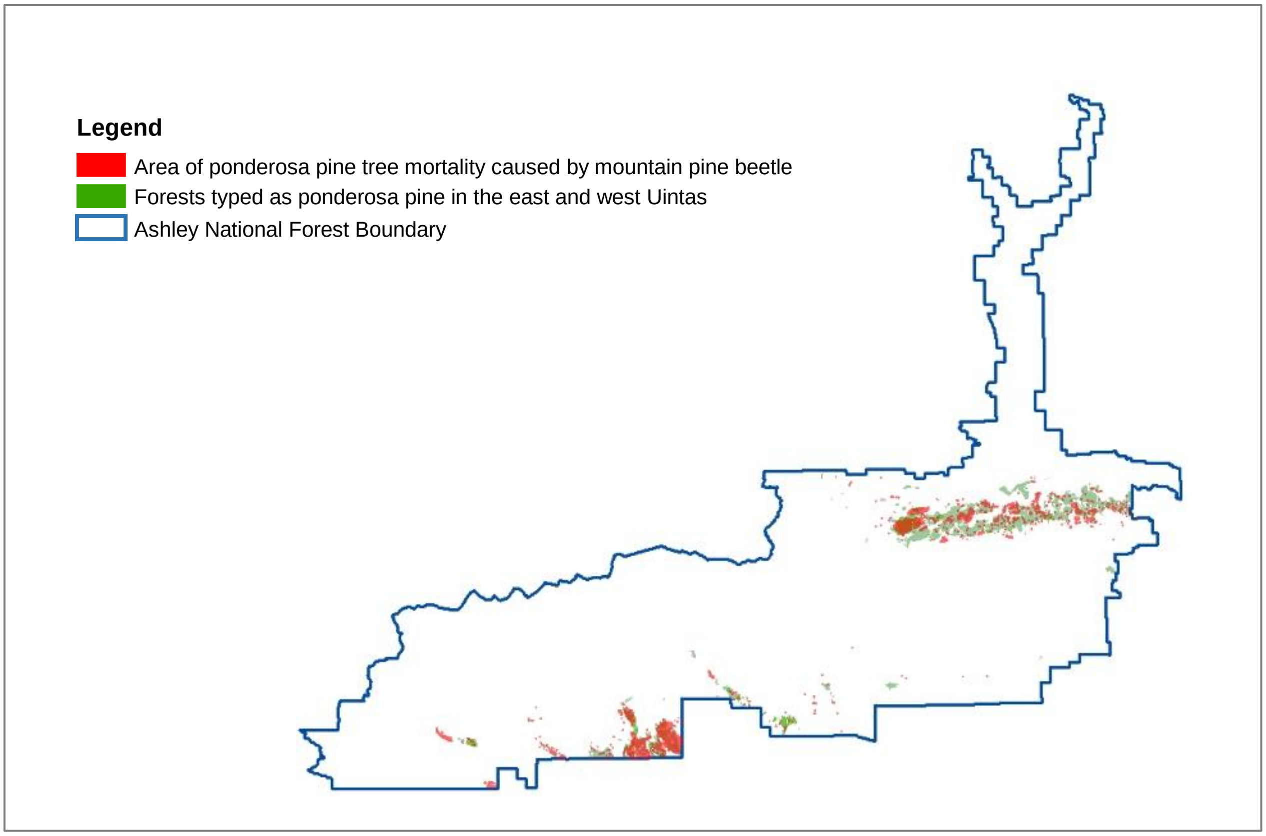


Figure 14. Acres of mountain pine beetle-caused tree mortality in ponderosa pine on the Ashley National Forest from 1997 to 2015

Piñon Ips: Piñon pine is part of the piñon-juniper woodland vegetation type and totals 122,383 acres on the Ashley National Forest. Piñon *Ips* populations persist in piñon-juniper woodlands by attacking damaged or stressed host trees. Endemic or low populations of this insect are associated with top-kill, branch mortality, or kills of widely scattered single trees or small groups of trees numbering less than 10. Often these trees have been previously damaged by wind, snow, fire, or lightning. In a drought-triggered outbreak, landscape scale mortality can occur in stands affected by drought and other stress factors (Shaw et al. 2005).

Tree mortality from the piñon *Ips* beetle increased slightly in 2015, where it was isolated to the Duchesne District. There was an increase of one tree and one additional acre affected (11 trees on six acres). Several pockets of mortality were mapped near Sowers Canyon and Bad Lands Cliffs.

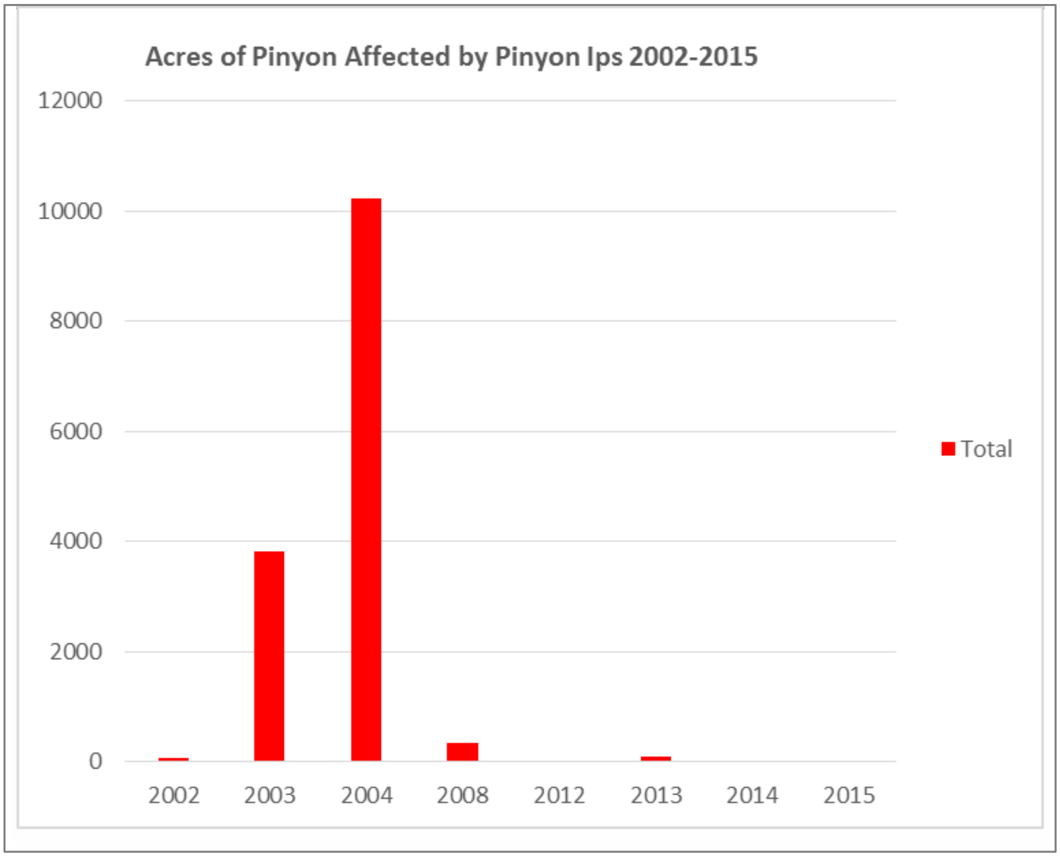


Figure 15. Trend in piñon Ips beetle-caused piñon pine mortality by year since 1997 (no mortality was detected in years 1997 through 2001)

Figure 15 shows the trend in piñon Ips beetle-caused piñon pine mortality by year since 1997 (no mortality was detected in years 1997 through 2001). Drought combined with outbreak piñon Ips populations contributed to heavy piñon pine mortality in 2003 and 2004 (USDA 2012). Piñon Ips beetle-caused tree mortality has affected approximately 13,360 acres of the piñon-juniper forest (approximately 11 percent) since 2002. Piñon Ips populations have been most active on the South Unit of the Duchesne Ranger District. Figure 16 shows the acres of piñon Ips beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015 and the locations of unaffected piñon-juniper stands.

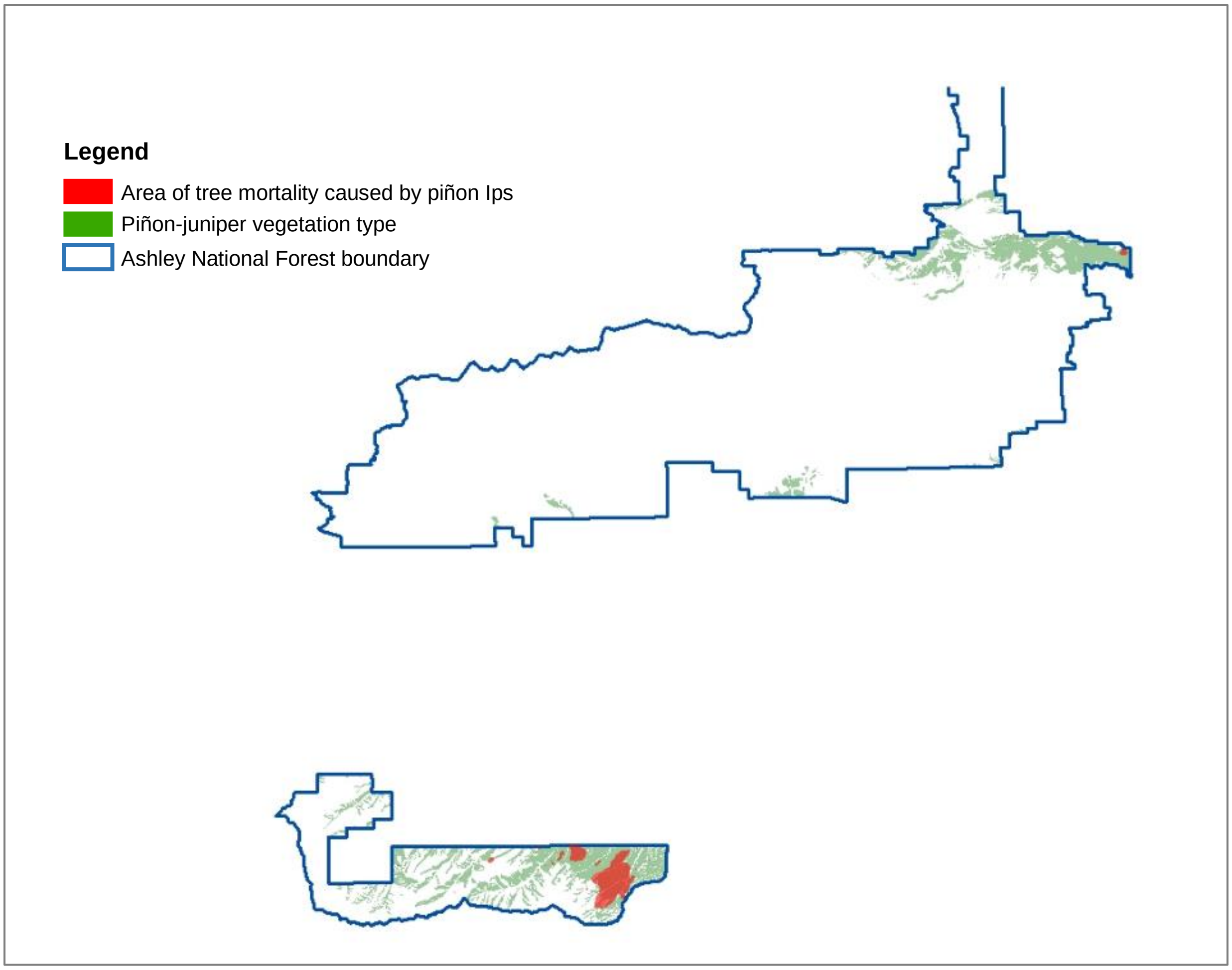


Figure 16. Acres of piñon Ips beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015

Aspen dieback and decline: Aspen dieback and decline has been noted since the 1970s, but recent widespread mortality has led to increasing interest in the status of aspen forests. The increases in damage are occurring along with changes in aspen ecosystems, due to climate changes. Dieback refers to stem mortality where there are sufficient saplings for replacement and suckering for recruitment. Decline refers to the gradual loss of healthy stems without replacement by sprouts, which can lead to loss of the clone. When mapped, it is difficult for aerial observers to distinguish between the two conditions. Forest staff investigated some areas coded as decline and found aspen sprouting. For this reason, dieback and decline will be summarized together in this assessment.

Aspen and seral aspen types total 152,712 acres on the Ashley National Forest, of which 117,198 acres are typed as seral aspen and 35,514 acres are typed as persistent. Acres of dieback and decline during years 1997 through 2015 total approximately 8,600 acres in the Ashley's typed aspen. Dieback and decline occurred in two percent of the area typed as seral aspen and in 17 percent of the area typed as persistent aspen. The agents involved in dieback and decline vary depending on location. These agents tend to be a complex of bark beetles, wood boring beetles, and canker diseases in Utah (Guyon and Hoffman 2011). In addition to insects and disease, senescence can also play a role in these forests as the agents causing periodic episodes of mortality that can trigger regeneration (Kurzel et al. 2007).

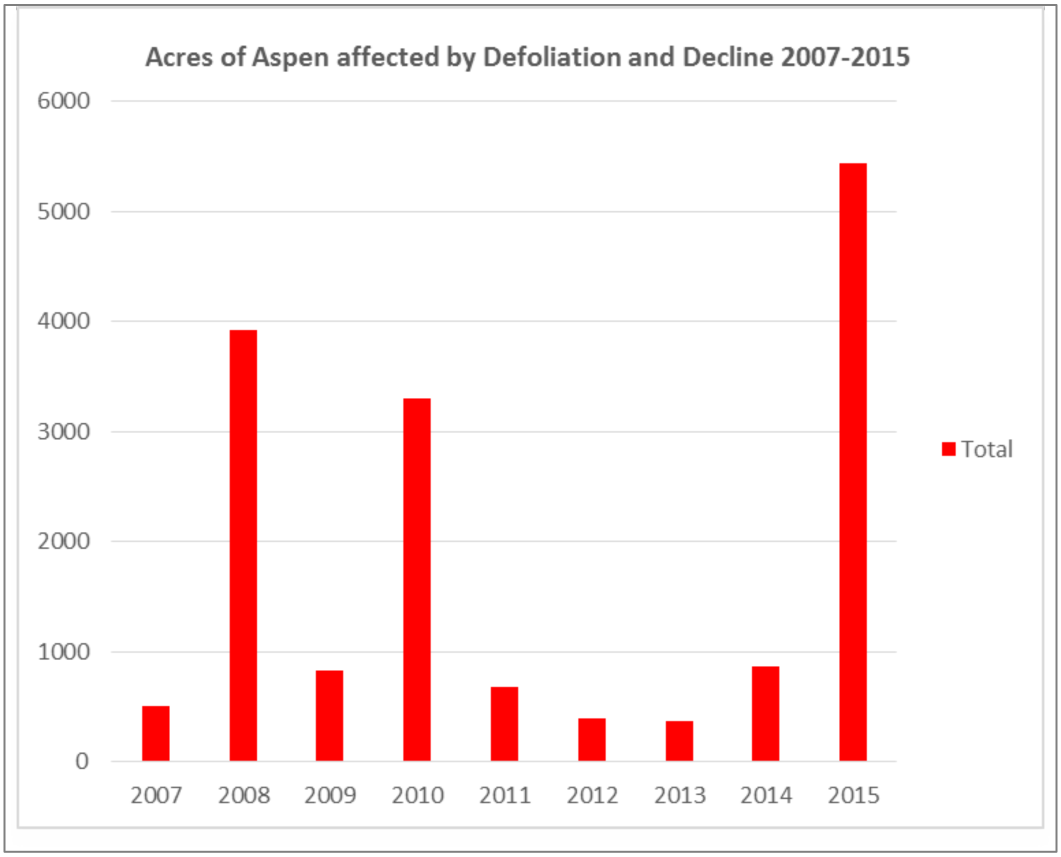


Figure 17. Trend in aspen dieback, defoliation, and decline by year from 2007 to 2015. No damage was detected in years 1997 through 2006.

Figure 17 shows the trend in aspen dieback, defoliation, and decline by year from 2007 to 2015; no damage was detected from 1997 through 2006. Increasing symptoms of aspen dieback and decline were recorded following a drought in 2001 through 2004. These symptoms peaked in 2008 through 2010. Subsequent symptoms and aspen mortality rates had returned to near pre-drought levels until 2015. Figure 18 shows the area of aspen dieback, defoliation, and decline on the Ashley National Forest from 1997 to 2015 and the locations of unaffected aspen stands.

Area of mapped aspen dieback and decline reportedly increased in 2015, totaling 5,436 acres. In 2015, however, the foliar disease Marssonina was especially prevalent on the Ashley National Forest. Marssonina is a defoliation event. Defoliation caused by Marssonina is associated with a moist period right after aspen leaf-out and can trigger a dieback event when there are two or more years of repeated damage. Forest Health Protection staff observed wood borer activity on the Red Cloud Loop of Vernal District in 2015. However, many reports of dieback were incidents of defoliation, likely caused by Marssonina. Forest staff and members of the public reported the leaf condition on the eastern end of the Ashley National Forest near Davenport Draw and Bowden Draw. Without ground surveys, it is unknown how much of the aerially mapped dieback and decline in 2015 was actually defoliation caused by Marssonina.

Of the areas mapped by aerial observers in 2015, large polygons reported as dieback and decline were concentrated near Yellowstone River and the southern boundary of the Vernal and Flaming Gorge Districts. Dieback and decline were also detected near Left Fork Canyon and Fork Ridge.

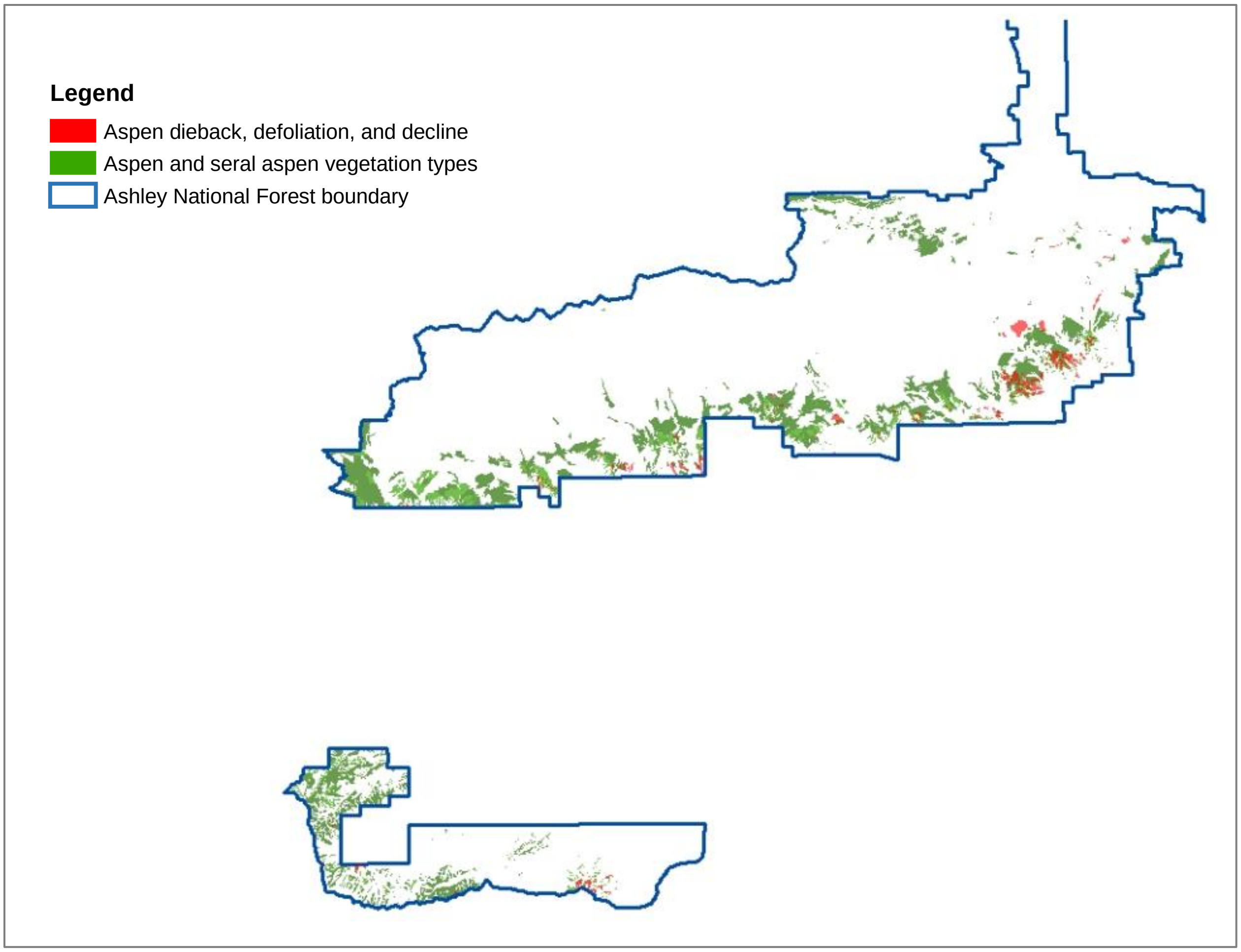


Figure 18. Acres of aspen dieback, defoliation, and decline on the Ashley National Forest from 1997 to 2015

Spruce Beetle: On the Ashley National Forest, spruce beetle affects the mixed conifer (spruce, lodgepole pine, subalpine fir) and spruce forest types. The extent of forested area typed as mixed conifer[14] is 358,000 acres, while that typed as Engelmann spruce totals 144,500 acres. Spruce-beetle-caused mortality has recently affected approximately 66,400 acres of the spruce-fir vegetation types (approximately 13 percent), much of which occurs in the High Uintas Wilderness. Spruce-beetle-caused tree mortality is modifying stand and age-class structure. The mortality is also altering the amount, composition, and arrangement of living and dead biomass in both the mixed-conifer and spruce communities that are affected.

In 2015, mapped spruce-beetle-caused tree-mortality continued to increase across the Ashley National Forest. On the Roosevelt District, there was a two-fold increase in spruce-beetle-affected acres and tree mortality from the prior year: 2015 (46,068 trees; 10,348 acres) compared to 2014 (12,410 trees; 5,200 acres). Aerial detection surveyors mapped 5 to 14 pockets of spruce mortality across the majority of the Roosevelt District. On the Duchesne District, spruce beetle-affected areas also increased: 2015 (132,945 trees; 22,072 acres) compared to 2014 (113,382 trees; 18,734 acres) showing high densities of spruce mortality throughout the district. The Flaming Gorge and Vernal Districts had no spruce beetle detections

[14] Forested area totals on mixed conifer include the seral aspen types

in 2014, but in 2015, surveys detected 184 affected acres on Flaming Gorge and 1,078 acres on Vernal District.

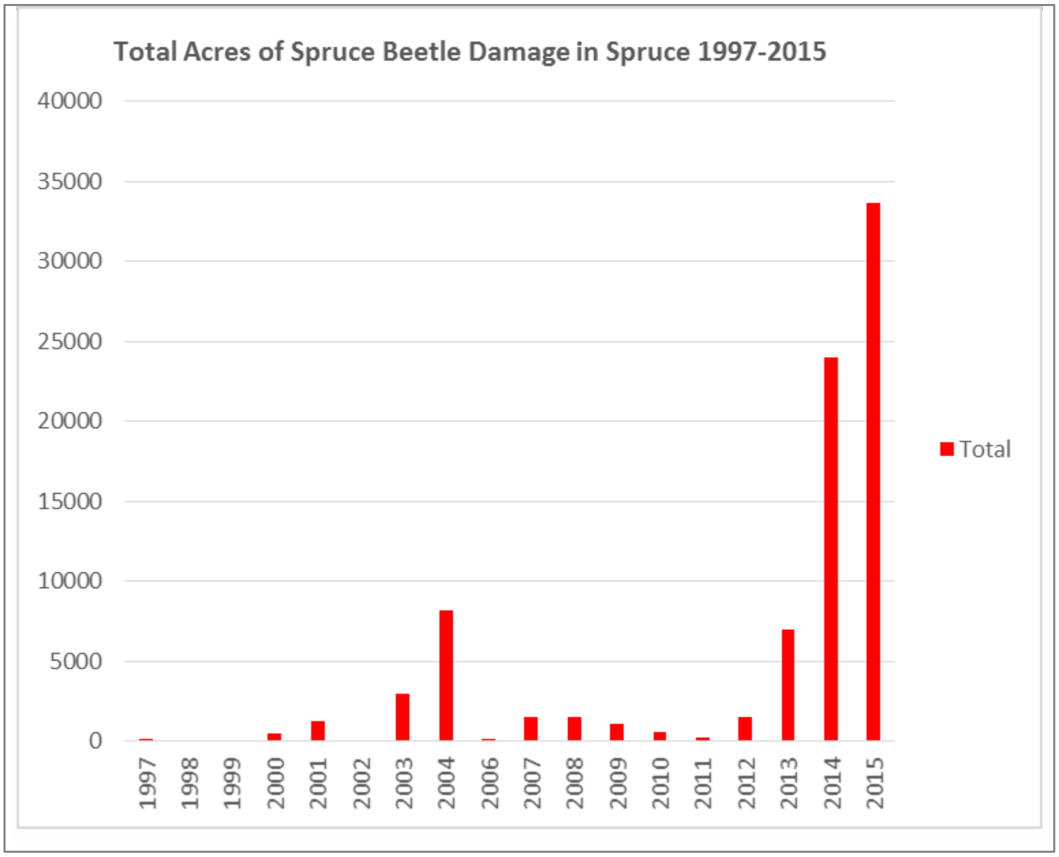


Figure 19. Trend in acres of spruce forest affected by spruce beetle from 1997 to 2015

Figure 19 shows the trend in acres of spruce forest affected by spruce beetle from 1997 to 2015. An on-going outbreak of spruce beetles are killing susceptible Engelmann spruce trees, causing mortality in up to 90 percent of host trees. Spruce beetle-caused mortality has already affected approximately 66,400 acres of the spruce-fir vegetation types (approximately 13 percent). Much of the affected area occurs in the High Uintas Wilderness. The epidemic originated on the west end of the Ashley National Forest and is spreading east. Not shown in the graph is the acreage of spruce beetle damage in 2016, which totals approximately 45,850 acres. Mortality is expected to continue to the east due to availability of susceptible spruce hosts. Figure 20 shows the area of spruce beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015 and the locations of unaffected spruce stands.

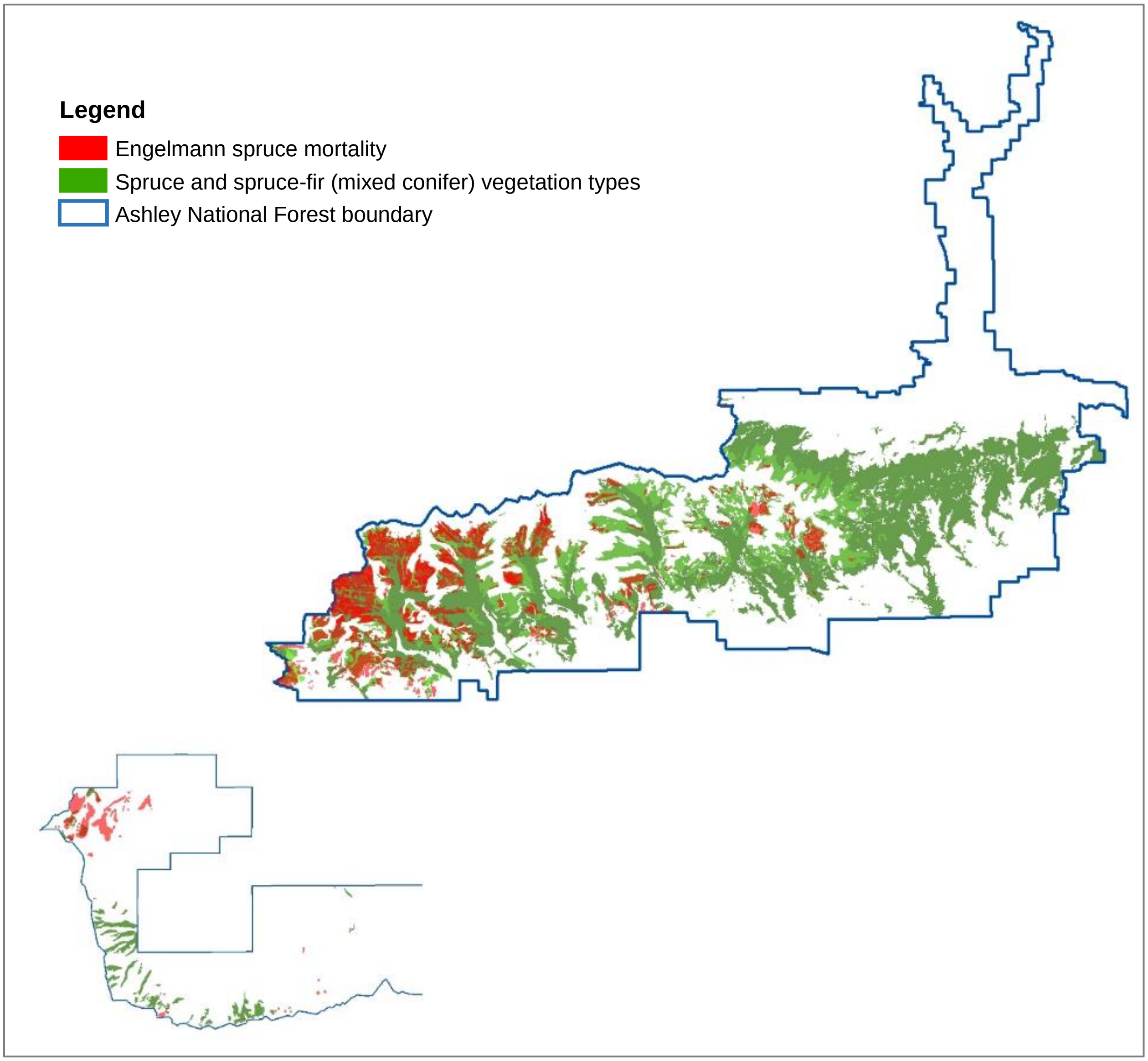


Figure 20. Acres of spruce beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015 (South Unit is not the same scale)

Climate Related Risks and Trends

Climatic variability and change can alter patterns of disease distribution and abundance through (1) direct effects on development and survival of a pathogen, (2) physiological changes in tree defenses, and (3) indirect effects on abundance of natural enemies, mutualists and competitors, (4) interactions with other disturbance agents such as fire, and insects (Malesky et al, In Review). Climate change will affect pathogens, hosts, and their interaction. Changes in these interactions may become the most substantial drivers of future disease outbreaks. Climate change will have the largest impact on diseases that take advantage of a weakened host. Some diseases may be considered “threshold diseases”, that is, they are damaging but only under certain climatic conditions. One of the key triggers for this type of disease is the onset of drought stress (Malesky et al. in review).

Warmer temperatures associated with climate projections are believed to impact (and frequently amplify) insect species’ population dynamics directly through effects on survival, generation time, fecundity, and dispersal. Large and consistent decreases in snowpack have been observed throughout the western United States between 1955 and 2015 (EPA 2015). Insect species generally have relatively short life cycles, high reproductive capacity, and high degree of mobility. Because of this, the physiological responses to warming temperatures can produce large and rapid effects on species population dynamics (Stange and Ayres 2010). Mid to high latitudinal insect populations are anticipated to benefit from climate change through more rapid development and increased survival. Insect species' mortality may decrease with warmer winter temperatures, thereby leading to poleward range expansions (Stange and Ayres 2010).

Large mortality events in forests are normally associated with the occurrence of several stressors (Allen and others 2010; McDowell and others 2016). The interactions among disturbances working over various spatial and temporal scales define the nature of forested landscapes (Jenkins and others 2008). Changes in drought intensity and frequency, for example, have the potential to alter populations and impacts of tree-damaging forest insects and pathogens (Ayres and Lombradero 2000, Dale and others 2001, Weed and others 2013). In addition, bark beetle-caused tree mortality in conifer forests affects the quantity and quality of forest fuels (Jenkins et al 2008). Dissimilar and extemporized interactions make it challenging to predict the effect of multiple stressors and whether threshold-type responses may take place (McDowell and others 2016).

Fire

Wildland fire is a primary disturbance agent that historically and currently drives ecosystem change at the landscape-scale (Hessburg and others 2015). Across the Ashley National Forest, fire has influenced vegetative patterns, composition, structure, age, and development of both individual stands and the larger landscape. In many areas in the West, existing disturbance regimes are markedly altered from natural disturbance regimes (Bassman and others 2015). Today's landscape patterns are largely a byproduct of the cumulative effects of human activities and altered disturbance regimes.

Forests on the Ashley National Forest have undergone changes during the 20th century. Evidence for these changes is based on repeat photography, fire-scar analyses, forest stand reconstructions, and pollen and charcoal studies. Among the changes are substantial increases in the density of trees, and landscape-scale continuity of dense fuels. Today, in place of an open understory, conditions reflect brush, downed timber and many young trees. In some stands, fires that start on the ground can spread quickly and then climb through the branches of small trees, which create a "ladder" to the larger trees in the forest canopy.

Data presented in this section quantifies current and historical indicators and stressors that are linked to wildland fire within the Ashley National Forests. The forest vegetation types associated with Utah fire groups and nonforested vegetation types were assessed to quantify their departure from historical fire regime groups across the Ashley National Forest. In addition, the stressors and indicators for existing conditions are described to provide information on ecosystem function and contribution to ecosystem function.

Description of the Natural Range of Variation and Influences of Drivers and Stressors

The Ashley National Forest is representative of broader regional trends in the Rocky Mountains. Fire exclusion and suppression has caused significant changes in forest vegetation and stand conditions. In return, these deviations in stand structure and species composition have altered natural disturbance regimes with significant long-term consequences (Byler and Hagle 2000, figure 21).

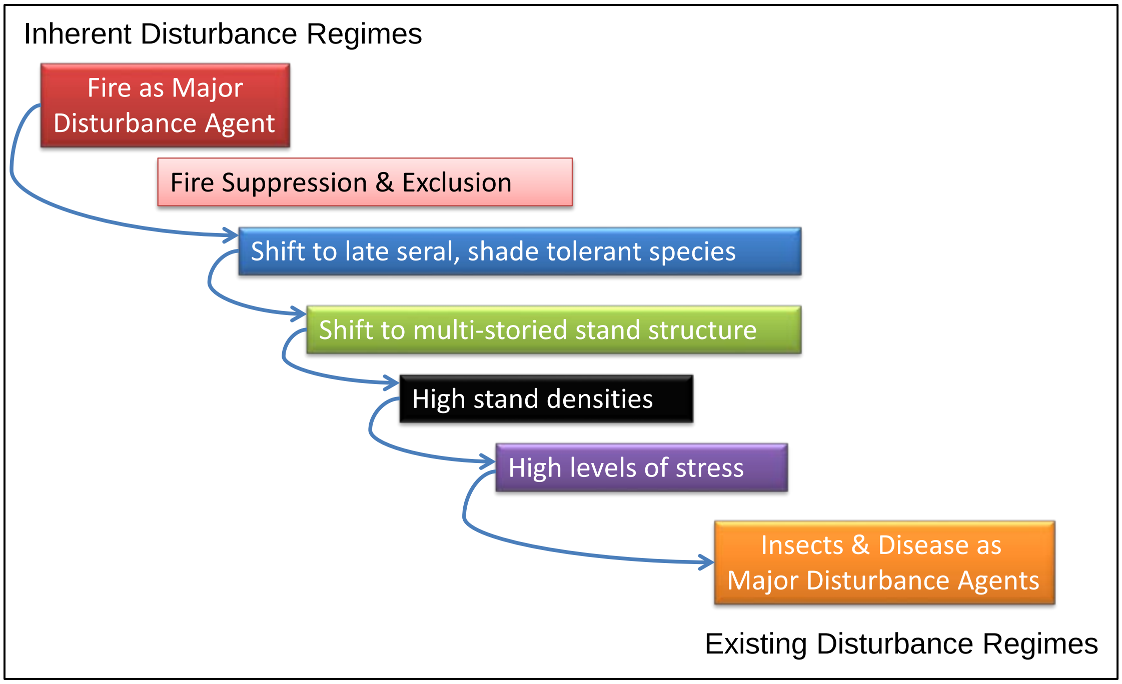


Figure 21. Changes to natural disturbance regimes resulting from fire exclusion in the Northern and Southern Rockies

Fire Regimes

A fire regime describes the frequency, predictability, and severity of fire in an ecosystem. On the Ashley National Forest, wildland fire has influenced patterns of species composition, structure, and age. Wildland fire has also influenced succession of both individual stands and the larger landscape. The distribution of forest and rangeland vegetation in the analysis area occurred due to low, mixed and high severity fire regimes (Bradley and others 1992).

Drought cycles and fuel availability have considerable influence on fire regimes. Pre-settlement wildland fires generally burned through the early summer season, until extinguished by monsoonal precipitation. In the settlement period before 1941, wildland fire suppression was often not successful. Suppression efforts resulted in fires burning thousands, even up to tens of thousands of acres. Fire suppression efforts since then have altered natural fire regimes and reduced the number of forested acres burned each year (Arno 1996a). The combination of fire suppression, fire exclusion, and natural disturbance processes has allowed fuels to accumulate in unmanaged forested stands. For many vegetation types, the current fire regimes have moderate to high departure from natural fire regimes. These current fire regimes are responsible for higher tree densities and fuel accumulations that support wildfires with uncharacteristically severe effects (Arno 1996b).

A generalized system of classifying fire regimes, given the wide range possible, is to define fire severity categories of high, moderate, and low. Low-severity fire regimes typically had frequent, low-intensity fires. High-severity fire regimes had infrequent, but stand-replacing fires. Moderate-severity fire regimes, also called mixed severity, had complex combinations of high-, low-, and moderate-severity fires (Agee 1996). The LANDFIRE classification system defines the five natural fire regime groups. These groups are based on the average number of years between fires (fire frequency), combined with characteristic fire severity reflecting percent replacement of dominant overstory vegetation (table 26) (National Interagency Fuels Fire and Technology Transfer System 2010).

Table 26. Fire regime group descriptions from the LANDFIRE classification system

Group	Frequency (years)	Severity	Severity description
I	0 – 35	Low / mixed	Generally low-severity fires replacing less than 25 percent of the dominant overstory vegetation; can include mixed-severity fires that replace up to 75 percent of the overstory.
II	0 – 35	Replacement	High-severity fires replacing greater than 75 percent of the dominant overstory vegetation.
III	35 – 200	Mixed / low	Generally mixed-severity; can also include low severity fires.
IV	35 – 200	Replacement	High-severity fires.
V	200 plus	Replacement / any severity	Generally replacement severity; can include any severity type in this frequency range.

Current Status and Trends

To better understand the historical fire disturbances within each of the vegetation types, LANDFIRE fire regime groups were used to characterize the severity and frequency of fires within forested and non-forested vegetation types (table 27). Natural fire regimes are usually defined in a historical sense that are typically restricted to the pre-1900s. It is recognized that indigenous cultures affected landscapes with fire before 1900, but we cannot, in most cases, separate out the human component of natural fire regimes from the effects of indigenous groups before 1900.

Table 27. Fire regime group (FRG) within forested and nonforested vegetation types by percent of area

Vegetation Types	FRG I	FRG II	FRG III	FRG IV	FRG V	Other*	Total % Area
Ponderosa pine	63.96%	0.13%	10.55%	21.62%	3.65%	0.11%	100%
Lodgepole pine	21.65%	1.93%	4.01%	34.17%	36.95%	1.29%	100%
Douglas-fir	65.24%	0.17%	16.54%	7.92%	8.01%	2.13%	100%
Mixed conifer	18.24%	0.80%	4.02%	15.45%	59.09%	2.40%	100%
Engelmann spruce	2.73%	2.34%	1.52%	0.72%	86.05%	6.64%	100%
Miscellaneous	30.09%	3.45%	22.93%	15.21%	21.92%	6.40%	100%
Seral aspen	59.85%	0.76%	4.81%	14.75%	19.00%	0.83%	100%
Persistent aspen	79.91%	0.42%	6.55%	8.18%	4.41%	0.53%	100%
Sagebrush	23.78%	0.72%	11.87%	56.41%	4.84%	2.38%	100%
Pinyon juniper	18.81%	0.02%	32.05%	33.80%	2.53%	12.79%	100%
Desert shrub	0.25%	0.01%	2.65%	79.59%	10.89%	6.61%	100%

*Other includes water, barren ground, or sparse vegetation

Vegetation Condition Class

Vegetation condition class is a measure of departure from reference (pre-settlement, natural, or historical) ecological conditions that typically results in alterations of native ecosystem components (see table 28 for vegetation condition class descriptions). Vegetation condition class is used to assess the ecological departure from the natural fire regimes for each of the existing forest and non-forest vegetation types (table 29). One or more of the following activities may have caused departures: fire suppression, timber harvesting, livestock grazing, introduction and establishment of exotic plant species, introduced insects or diseases, or other management activities (National Interagency Fuels Fire and Technology Transfer System 2010). For landscapes that were moderately or severely departed, those acres can provide the foundation for how many acres potentially need restoration. In addition to vegetation condition class, ecosystem attributes could also contribute to departure from historical ecological conditions. These attributes include species composition, structural stage, stand age, canopy closure and fuel loadings

Table 28. Vegetation condition class descriptions from the LANDFIRE classification system

Vegetation condition class	Description
1. Vegetation Condition Class I.A	Very Low, Vegetation Departure 0-16%
2. Vegetation Condition Class I.B	Low to Moderate, Vegetation Departure 17-33%
3. Vegetation Condition Class II.A	Moderate to Low, Vegetation Departure 34-50%
4. Vegetation Condition Class II.B	Moderate to High, Vegetation Departure 51-66%
5. Vegetation Condition Class III.A	High, Vegetation Departure 67-83%
6. Vegetation Condition Class III.B	Very High, Vegetation Departure 84-100%
Other	Water, snow and ice, non-burnable urban, burnable urban, barren, sparsely vegetated, burnable agriculture

Table 29. Vegetation condition class (VCC) in forested and nonforested vegetation types by percent of area

Vegetation Types	VCC IA	VCC IB	VCC IIA	VCC IIB	VCC IIIA	VCC IIIB	Other[1]	Total % Area
Ponderosa pine	0.01%	6.76%	48.38%	25.98%	16.40%	0.12%	2.35%	100%
Lodgepole pine	0.19%	0.12%	43.50%	46.21%	5.29%	1.92%	2.77%	100%
Douglas-fir	0.03%	11.04%	41.81%	37.05%	7.34%	0.17%	2.56%	100%
Mixed conifer	0.32%	0.62%	65.36%	28.53%	1.63%	0.82%	2.72%	100%
Engelmann spruce	1.85%	0.18%	84.94%	3.73%	0.05%	2.43%	6.83%	100%
Miscellaneous[2]	1.27%	1.73%	55.05%	22.30%	6.05%	3.46%	10.15%	100%
Seral aspen	0.05%	0.68%	35.02%	50.04%	12.09%	0.75%	1.38%	100%
Persistent aspen	0.04%	0.69%	16.28%	55.47%	25.53%	0.42%	1.58%	100%
Sagebrush	0.63%	5.47%	57.14%	24.55%	7.79%	0.72%	3.70%	100%
Pinyon juniper	0.79%	20.40%	42.16%	21.37%	1.94%	0.02%	13.32%	100%
Desert shrub	5.99%	5.19%	3.38%	78.51%	0.05%	0.01%	6.87%	100%

[1]Other includes water, barren ground, or sparse vegetation

[2]Miscellaneous forest vegetation types includes subalpine fir, blue spruce, 5-needle pines, riparian forest

Fire Groups and Forested/Non-Forested Vegetation Types

Some habitat types represent later seral stages, or vegetation succession in the absence of significant disturbance. Wildland fire can reset vegetation communities to an earlier seral state and, in some locations, fire may prevent vegetation from reaching a late seral condition. Likewise, an early-mid seral state can be perpetuated by the presence of wildland fire.

The classification of Utah fire groups are used to further describe the role of wildland fire and the influence it has on the representation of forested vegetation types on the Ashley National Forest. Fire groups are based on the presence of forest vegetation that make up individual habitat types. Additionally, the 12 fire groups are assigned based on the biological response of the tree species to disturbance and shade (Bradley and others 1992). The exception is fire group zero, which is used to describe grassland, riparian, and rock-scree communities. See Table 5 for fire group descriptions of selected vegetation types on the Ashley National Forest.

Table 30. Utah fire group descriptions

Fire Group	Vegetation Types	Brief Description and Role of Fire
0	Alpine/Rock Miscellaneous	Miscellaneous sites that are dominated by deciduous trees, shrubs, or herbaceous meadows. In addition, scree, rock outcrops, and unburnable areas, such as lakes and large bodies of water, fall in this fire group. These fire groups were not analyzed for fire regime groups and vegetation condition class.
1	Pinyon juniper	Prior to European settlement, these sites burned more frequently, current estimates demonstrate longer frequencies between fires. Stand-replacing, high severity fires occurring every 30 to 200 years (fire regime group IV) are most common in the pinyon/juniper type. Large fires, such as the Mustang Ridge Fire of 2002, consumed close to 15,000 acres of pinyon/juniper communities, leaving only small pockets to provide seed sources. The potential increase in fire severity and the change in the overall fire return interval for this vegetation types generates a higher percent classified as vegetation condition class IIB. Fuel conditions vary greatly depending on the canopy closure of pinyon-juniper. More open stands consist of grasses and shrubs while closed stands have less surface fuels. Fuel loadings are usually less the two tons/acre with variable fuel heights. Fire spread and flame lengths vary greatly depending the condition of the stands and the wind speeds. Livestock grazing will reduce fire behavior characteristics. Exotics such as cheatgrass may significantly increase fire spread rates.
3	Ponderosa pine	Stands in pre-settlement times were typically open and experienced frequent, low severity fire occurrence. In properly functioning stands the majority of ground fuels are perennial grasses, small diameter branch wood, and needle cast. (Bradley and others 1992). With fire exclusion the predominant historical fire regime group I has been altered and an unnatural buildup of litter and downed woody material has increased. As a result, overstocking and crowding occurs increasing the presence of more shade tolerant species. Many of these areas have moderate to high departure and are classified as vegetation condition class IIA, IIB, and IIIA. In his technical fire management report on ponderosa pine on the Flaming Gorge Ranger District, Palmer determined the mean fire return interval to be 21 years (Palmer 1992). This study represents about 16,000 acres of ponderosa pine between Dowd Mountain and Gorge Creek, where 13,205 acres have been treated between 1992 and 2016, with 2,305 acres having already lapsed the average return interval.
4	Ponderosa pine Douglas-fir	Ponderosa pine and understory grasses are often important constituents of this forest type. These sites often burned with high frequency and low severity fires. Without frequent fires, Douglas-fir will become the dominant species increasing fire severity. Historically, wildfires burned frequently with low to moderate severity in the dry Douglas-fir vegetation types. Many of these areas favored the development of ponderosa pine or on drier sites the establishment of shrub and grasses. With the absence of fire, the density of trees along with an increase in ladder fuels promote higher severity fires. Historically, a majority of these vegetation types had frequent low to mixed severity fires falling within fire regime group I, and with the absence of fire, a high percent of these types are classified as vegetation condition class IIA and IIB. The Douglas-fir vegetation type fits into Utah fire groups 4 and 5, with 53 percent being fire group 4: drier Douglas-fir habitat. Surface fuels are typically light, consisting of small branch wood, with occasional grasses and shrubs. Fire behavior is characterized by low spread rates and flame lengths. Only under extreme fire weather conditions, can crown fire be initiated.

Fire Group	Vegetation Types	Brief Description and Role of Fire
5	Douglas-fir Lodgepole pine	Thirty-three percent of the area falls within the cool or moist Douglas-fir habitat types. These forests are dominated by Douglas-fir and lodgepole pine that historically had mix of stand replacement to light surface fire based on the topographic position, structure, and fuel loading. Under moist Douglas-fir types a considerable increase in dead downed fuels has occurred with heavier surface fuel loadings. The dense overstory create ladder fuel conditions that increase the potential for crown fires to occur. Therefore, the spread rates and flame lengths can increase substantially (Bradley and others 1992). Stand replacement fire was common in those areas dominated by lodgepole pine and aspen, while Douglas-fir generally burned more frequently with lower severity.
7	Persistent and Seral aspen	Utah fire group classifies aspen stands in relation to fuels into five types that are a mixture of forbs and shrubs. The frequency of fire may differ between even and uneven aged aspen stands. Persistent and seral aspen have a high percent of the area classified as fire regime group I. During the last century, fire suppression and grazing have altered the fire frequency and potential severity in this type creating a higher percent of the area falling into vegetation condition class IIB and IIIA. Persistent Aspen: Weather conditions seldom occur where entire stands of persistent aspen may experience high fire severity. Fuel conditions in these types have low fuel loading. Fire behavior is characterized by slow rates of spread and low flame lengths. Seral Aspen: The seral aspen vegetative group is perhaps the most complex due to the multiple species in which it interacts. In early successional stages aspen is dominant but eventually gives way to conifer species that are typically more shade tolerant or out compete the aspen for nutrients. Viable aspen clones may remain on site for numerous years, but will eventually die out without disturbance. Fuels generally consist of leaf litter and conifer needles that are characterized by light fuel loading. Ground fire are generally slow-burning with low flame lengths. Some fire may encounter an occasional “jackpot” or heavy fuel concentration that can flare up. Under increased conifer encroachment and severe weather conditions, these fuels can pose fire hazards.
8	Lodgepole pine, Mixed conifer	Lodgepole pine can be a seral or climax species. Fire either perpetuates lodgepole or renews it. Fire frequencies and severities are highly variable, with ranges from less than 50 years to more than 300 plus years. Well known as a high severity, stand replacement species with a fire return interval of 35 to 200 years, lodgepole pine is typified as a fire regime group IV and V species. Currently a majority of the lodgepole pine falls within vegetation condition class IIA and IIB, with a moderate to low departure. The greatest amount of down dead woody debris can be found within this vegetation type, mainly due to the Rocky Mountain pine beetle outbreak in the late 1970s and early 1980s. The majority of the trees killed in this epidemic have since toppled and await treatment through natural fire or prescribed fire. More recently, within the past 5-8 years, additional beetle activity has added to that mortality that will further increase fuel loading within the next several years. Average fuel loads are 15 to 18 tons per acre with maximum loads being much higher, depending on the impacts of insects such as mountain pine beetle. Typically, most of this loading is in the large-fuel category. The nature of fuels changes over time in lodgepole pine stands, with many stands having a high degree of variability depending on the age class and disturbance that has affected the stands. Fires in Rocky Mountain lodgepole pine stands are highly variable ranging from a surface fire with low fire spread rates and flame lengths, consuming litter and duff, or high-severity, stand-replacing crown fires. (Bradley and others 1992).

Fire Group	Vegetation Types	Brief Description and Role of Fire
10	Mixed conifer Miscellaneous[1]	These types are not in cold or moist landscapes. Subalpine fir and Engelmann spruce are the climax species. This group contains the heaviest downed woody fuel loads, particularly where lodgepole pine is the seral species. Fires historically burned in mosaic patterns and are thought to be less frequent than those in the drier fire groups. Where lodgepole pine or aspen occurred, higher frequency of fires favored long-term dominance by these species. Under a natural fire regime, these types are similar to stands dominated by Engelmann spruce and subalpine fir that promote highly destructive stand-destroying fires. As a result, they are defined by fire regime groups I, II, and V. Due to the lack of disturbance via wildfire or fuels treatment, the majority of the mixed conifer falls into vegetation condition classes IIA and IIB. Historical loadings in the mixed conifer type were probably no more than one fourth to one-third of present-day loadings. Fuel loads are higher than in lower elevation montane stands, and the fuel beds tend to be irregular and have large amounts of needle litter accumulating under the narrow crowned trees (Fischer and Bradley 1987). Under high or extreme fire weather conditions, fire behavior characteristics exhibit increased rates of spread with passive or active crown fire and long range spotting.
11	Mixed conifer Engelmann spruce	Subalpine forest habitat types generally found adjacent to riparian areas, on moist benches, or as stands associated with late-melting, high-elevation snowbanks are included in this group. Engelmann spruce is often a persistent seral or climax co-dominate with subalpine fir and is best represented by this fire group. Fuels consist of large diameter dead downed logs. Fires are generally less frequent, however the severity of fires are much greater due to the longer intervals and high fuel loads. Fire frequencies are usually greater than 300 years—fire regime group V. In addition, a high percent of the Engelmann spruce has a moderate-low departure and is classified as vegetation condition class IIA.
12	Engelmann spruce Miscellaneous[1]	These types are generally above 10,000 feet and climax subalpine fir and Engelmann spruce are usually the only seral species. Fire is generally infrequent, occurring primarily in drier autumn periods. With cooler, moisture conditions prevailing, this species has a fire return interval that exceeds 200 years and severity that most commonly is stand replacing. As a result, a majority of this vegetation type falls within fire regime group V. In addition, a high percent of the Engelmann spruce has a moderate-low departure and is classified as vegetation condition class IIA. Engelmann spruce is a subalpine species that thrives in cold, moist, high elevation conditions that are generally too harsh for other species (Bradley and others 1992). The fuel structure in stands dominated by Engelmann spruce and subalpine fir promotes highly destructive stand-destroying fires. Fuel loads are higher than in lower elevation montane stands, and the fuel beds tend to be irregular and have large amounts of needle litter accumulating under the narrow crowned trees (Fischer and Bradley 1987). Fire behavior is generally characterized by low spread rates and flames lengths under normal conditions, but under extreme fire weather, the chance of higher intensity crowns may occur.
None[2]	Desert shrub	Desert shrub is not a Utah fire group. This vegetation type encompasses many individual plant species adapted to arid environments. The most notable species include Wyoming big sagebrush, saltbush, shadscale, spiny hopsage, desert shrub, winterfat, greasewood, gray molly, and bud sagebrush. These vegetation types are best represented by fire regime group IV. Some species, such as greasewood for example, tend to have a greater fuel bed continuity that produces a stand replacement severity characteristic when burned. Other species tend to grow in clumps that are broken by bare soil and tend not to carry fire as well. Vegetation condition class IIB best represents the desert shrub with many of these vegetation types not having experienced any recorded disturbance in modern record, although the introduction of cheatgrass may favor more frequent fire. Fuels consist of discontinuous scattered shrub component with live and dead shrub twigs and foliage. Fuel loading is generally light with a majority of the fuels less than one foot tall. Without significant fire weather conditions, fire spreads are generally low, with low flame lengths. Under more continuous fuel beds, fire spread and flame increases greatly.

Fire Group	Vegetation Types	Brief Description and Role of Fire
None[2]	Sagebrush	Sagebrush is not a stand-alone Utah fire group. However, it is an important subspecies to many other habitat groups such as Utah fire group 1, 3, and 4. The most abundant of the brush types on the Ashley National Forest is mountain big sagebrush. Additional species include: Wyoming big sagebrush, basin big sagebrush, spiked big sagebrush, black sagebrush, fringe sagebrush, and silver sagebrush. A majority of this vegetation type is characterized by fire regime groups I, III, and IV and vegetation condition classes IIA and IIB. The fire regime group depends largely on the species of sagebrush, its association with other vegetation types, and the horizontal fuel continuity. For example, Wyoming and black sagebrush thrive in areas unfavorable to grass production. Therefore, they have longer fire return intervals and higher severity fires. Furthermore, mountain big sagebrush has a high percent of the area that is highly variable with regard to fire regime groups and vegetation condition class. Fuel conditions are linked closely with the various sagebrush species. Both live herbaceous and live woody fuels can be associated with within these types. The fuel loading also varies depending the productivity of the site and can range from less than one ton/acre to several tons/acre. Fire spread rates also vary greatly and are highly dependent on fuel continuity and increased wind speeds. Where exotics such as cheatgrass has replaced these communities, fire spread rates can greatly increase.

[1]Miscellaneous forest vegetation types includes subalpine fir, blue spruce, 5-needle pines, riparian forest

[2] Nonforested types that do not have a Utah fire group classification

Topography and Weather Patterns

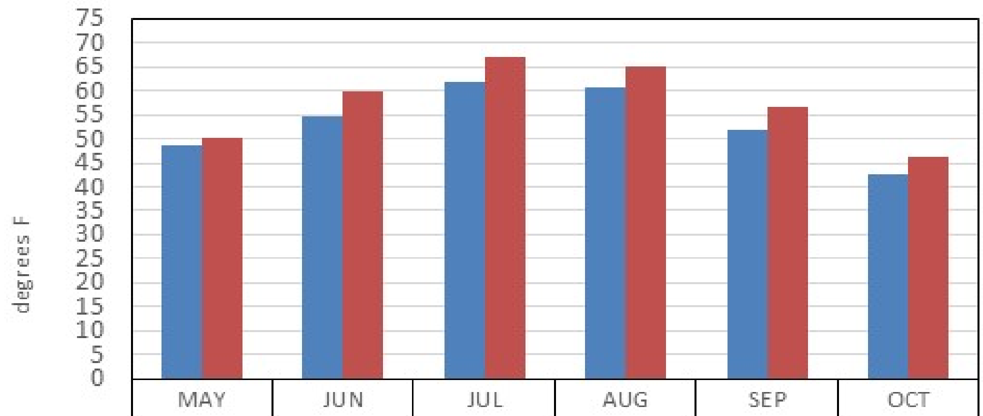
Topography and weather patterns are essential inputs to determine fire behavior for a given landscape. Aspect, elevation and topographic features have an impact on moisture profiles across the landscape that directly affect vegetation types and resultant fuel types (Malesky and others 2016). In addition, slope and aspect affect fire spread and surface fuel moistures. The following geographical areas encompass the Ashley National Forest: the east-west range of the Uinta Mountains, the Wyoming Basin, and the Tavaputs Plateau. Elevations range from 5,480 feet at the eastern edge of the Flaming Gorge Ranger District along the Green River, to 13,528 feet at Kings Peak on the Duchesne-Roosevelt Ranger District. King's Peak is the highest mountain in the State of Utah.

The Ashley National Forest has steep canyons and high mountain peaks, glaciated basins, and large open meadow areas. Important areas on the Ashley include the Sheep Creek Geological loop, the High Uintas Wilderness, the Flaming Gorge National Recreation Area, and the Uinta Mountains. The Uinta is a major east-west oriented mountain range in North America (Erskine 2013). In addition, an aggregation of geographic and topographic features influence the fire season weather patterns that occur across the Ashley National Forest.

Typical fire season (June to September) weather patterns are influenced by high-pressure systems that establish over the Great Basin area in May or June. The stationary high-pressure influence will normally last until mid-July. The high pressure will begin to wander from the center of the Great Basin, which will open the door to tropical moisture from the south and west. The shift will start the monsoon season. Prior to mid-July, thunderstorms are often of the dry variety or only moderately wet. As the monsoon season gets into full swing (normally in late July and August), the daily afternoon thunderstorms will be significantly wetter, especially at the higher elevations (7,000 feet or greater). The monsoon influence will begin to lessen in late August and early September. By late September and early October, the weather pattern usually dries out (Erskine 2013).

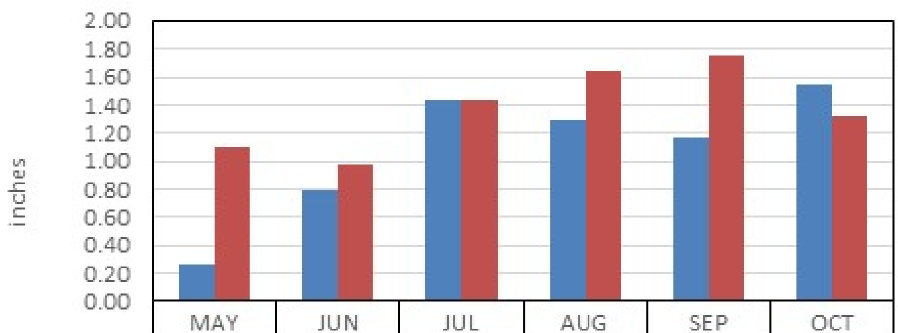
To better understand the influence of local topography and geography on weather, weather information was obtained from http://famweb.nwcg.gov/weatherfirecd/ for the Yellowstone remote automated weather station for the time period from 1970 through 2014. The computer program, Fire Family Plus Version 4.1

(USDA Forest Service 2004), was used to compare historical and current fire weather parameters associated with temperature, wind, and precipitation. Based on this analysis, the average temperatures and precipitation since 1970 have slightly increased, while winds have changed to a predominant south-southwest direction. In addition, these weather parameters are analyzed to determine trends associated with fire danger, fire behavior characteristics, and overall fire preparedness staffing. See Figure 22 through figure 24 for temperature, wind, and precipitation comparisons.



	MAY	JUN	JUL	AUG	SEP	OCT
1970-1984	48.8	54.6	61.9	60.7	51.7	42.6
1985-2014	50.4	59.7	67.1	64.9	56.5	46.3

Figure 22. Average monthly temperature from May to October, 1970-2014



	MAY	JUN	JUL	AUG	SEP	OCT
1970-1984 Avg total precip	0.26	0.79	1.44	1.30	1.17	1.54
1985-2014 Avg total precip	1.09	0.98	1.44	1.65	1.76	1.32

Figure 23. Average monthly precipitation from May to October, 1970-2014

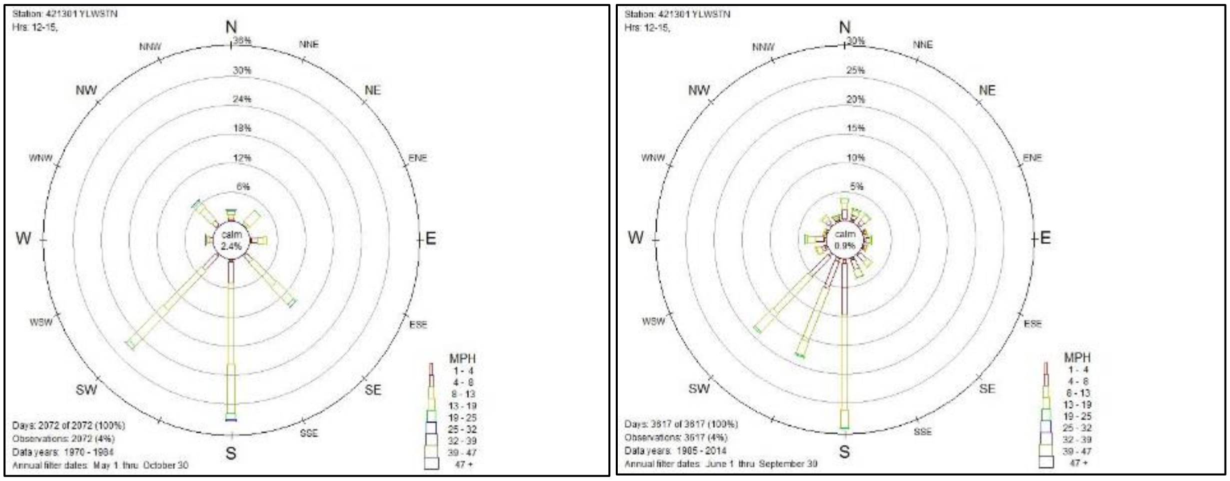


Figure 24. Average monthly wind direction and speed from May to October, 1970-2014

Climate Related Risks and Trends

With or without precipitation changes, temperature increases can result in decreased snow depth, alter timing/rate of snowmelt, lengthen or alter the timing of the growing season, and affect soil moisture levels. Climate changes will also affect disturbance processes in ecosystems, including fire frequency and intensity and insects and disease (Malesky and others 2016). Increasing air temperatures are expected to change the frequency, severity, and extent of wildfires. Large wildfires that have occurred during a warmer climatic period during the past two decades indicate a future in which wildfire is an increasingly dominant feature of western landscapes (Vose and others 2016).

With an increasing temperatures over the last several decades, there has been an increase in the number of years of drought. Drought has a clear correlation to the biotic and abiotic conditions in forested and rangeland vegetation types and increases the potential for large fires (Vose and others 2016). Although some of these interactions are predictable, they can be difficult to quantify. By analyzing the energy release component and Palmer drought severity index, a correlation of recent drought conditions to an increase in large fires on Ashley National Forest can be inferred.

Fire Danger Index Energy Release Component and Climatology

Regional fire potential is correlated to regional climate and stand conditions. The association of weather and fire behavior to any significant change in climate will affect the frequency and severity of conditions suitable for the ignition and spread of fires. In addition, each type of fuel has characteristic physical and chemical properties that affect flammability, and these properties vary with climate and weather (Brown 2000). Because drought influences fire directly through fuel moisture, and indirectly through biological and ecological effects on vegetation, the potential fire hazard can be quantified by both drought indices and fire behavior metrics. Interpretation of these metrics is complicated because not all vegetation types respond the same to drought in terms of fuel availability and flammability. However, the probability of ignition increases in most fuels when fuel moisture is low. Fuels can burn under different conditions in different ecosystems. However, even short-term drought generally increases wildland fire risk through its effects on fuel moisture, and thus on probability of ignition and spread rate (Vose and others 2016).

An indicator that is most often used to describe the fuel conditions associated with climatology changes throughout a fire season is the fire danger index energy release component for fuel model G. The energy release component is a number related to the available energy (British thermal unit) per unit area (square foot) within the flaming front at the head of a fire (Bradshaw and others 1983). An energy release component index, as used in the U.S. National Fire Danger Rating System (NFDRS), provides an approximation of dryness based on estimates of fuel moisture in a fuel model G (Andrews and others 2003). Thus, the larger the energy release component value, the "hotter" and potentially more severe the fire. Values typically range from 0 to 100, though they can be higher depending on weather extremes and fuel model (Brown and others 2004). Specifically, the energy release component is used to describe fire danger trends because it is sensitive to wetting rains that change fuel conditions. The energy release component calculation is also affected by fuel loadings in different size classes. Fuel model G, which includes both live and dead fuels, has a significant portion of the fuel load driven by large dead fuels. These large fuel moistures (also called 1000-hour fuels) are driven by weather conditions during the previous 1.5 months, which is the time it takes to mostly equilibrate to constant ambient conditions (Fosberg and others 1981). Because the energy release component varies across different ecosystems, the raw values are commonly converted to percentiles to indicate departure from average conditions (Riley and others 2013).

An analysis using Fire Family Plus 4.1 was conducted by comparing the changes in energy release component values for the Yellowstone remote automated weather station that are representative of weather and fuel conditions within the Ashley National Forest. In addition, the number of large fires over 1,000 acres was used to realize the significant changes that weather and fuel conditions have on the promoting large fires. Both the 90 and 97 percentile energy release component conditions were used to determine the difference for a May 1 through October 31 period from 1970 to 1984, compared to 1985 to 2014. The increase in large fire activity within the Ashley National Forest has a direct correlation to the histrionic increase in energy release component for this period.

Table 31. Energy release component (ERC) data comparison 1970 to 2014 from the Yellowstone remote automated weather station (kcfast and Fire Family Plus v4.1)

Year Range	90th percentile ERC	97 percentile ERC	Fires > 1,000 acres
1970-1984	60	69	2
1985-2014	80	89	12

Palmer Drought Severity Index and Climatology

Another tool that is used to assess periods of drought is the Palmer drought severity index. The index is commonly used to monitor regional climatic conditions. The Palmer drought severity index is reasonably successful at quantifying long term drought within a region by using temperature data and a physical water balance model. The index can capture the basic effect of global warming on drought through changes in potential evapotranspiration. As a long term measure, single precipitation events seldom drastically affect the Palmer drought severity index. Lastly, the Palmer drought severity index is based on the relative dryness or wetness of an area, so that areas with differing water budgets may be compared (Dai 2016).

In the mid- to late-20th century, relationships between area burned and climate parallel those in the fire history record. From 1980 forward, the area burned on Federal lands can be related to the monthly Palmer drought severity index, and the sign and magnitude of the relationships were consistent with reconstructed fire histories (Westerling and others 2003). The index has some weakness and variability associated with mountainous terrain and the correlation of area burned. On the Ashley National Forest, there is an indication of large fire growth for the later part of the month of June and early July (NOAA 2016) (figure 25 below).

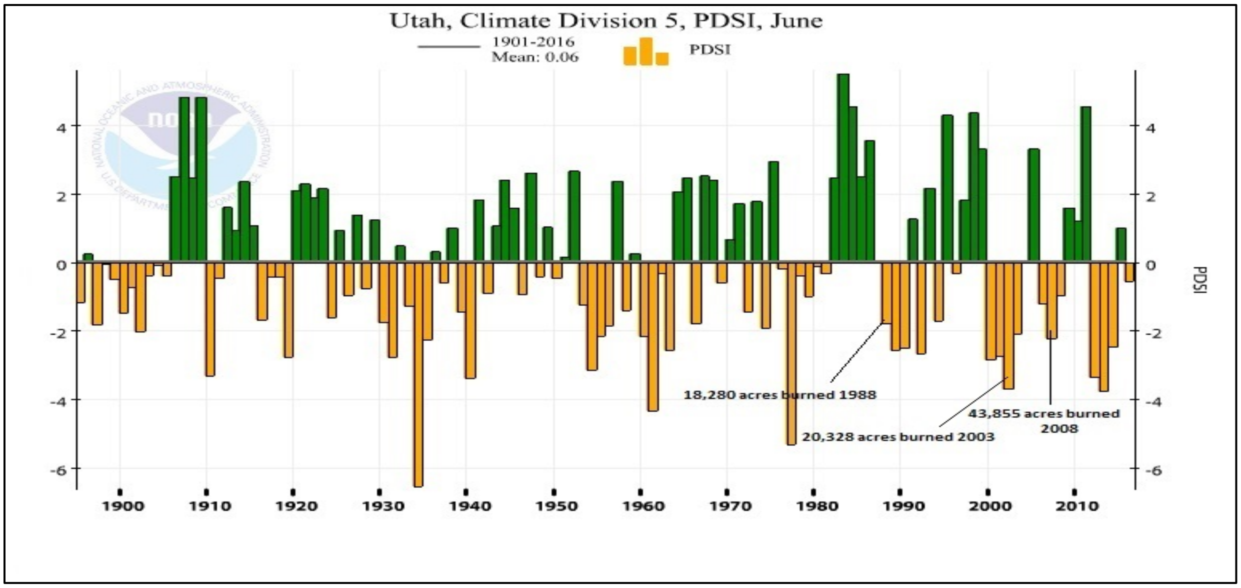


Figure 25. Palmer drought severity index from 1901 to 2016 and large fire growth for Utah Climate Division 5, Northern Mountains

Forest Fuels and Fire Behavior

The greatest effect of fire suppression and exclusion, in unison with other natural disturbance processes, has allowed biomass to accumulate in most unmanaged timber stands. The bulk of the biomass is in the form of dead standing and downed trees and shrubs - as well as live shade-tolerant true firs, spruce, lodgepole pine, and Douglas-fir. The combination of dead fuel and continuous live vegetation—from the forest floor to the upper forest canopy—creates a complex of fuel that, when ignited under severe fire conditions, has a higher occurrence of crown fire.

The vegetation conditions that are represented by the standard fire behavior fuel models on the Ashley National Forest are highly variable from the high desert to the alpine zone. Grasses and shrubs of the desert transition into pinyon-juniper and ponderosa pine forests at the mid mountain elevations. This transition, in turn, gives way to grasses and shrubs of the high mountain elevations. Aspen occurs along the mid elevations of the Ashley National Forest. Aspen transitions into mixed aspen-conifer, followed by conifer forests. The conifer forests are comprised primarily of lodgepole pine, with a mix of subalpine fir, Douglas-fir, and Engelmann spruce. At the highest elevations, krummholz fir borders grasses, forbs, and rock talus slopes above timberline.

In order to calculate how fires may burn within the vegetation types, an estimate of the fuel bed is used to define fuel models. A representative fuel model can be used to quantify fire behavior characteristics and the potential effects on vegetation. Fuels are made up of the various components of vegetation, live and dead, that occur on a site. These components include litter and duff layers, the dead-downed woody material, grasses and forbs, shrubs, regeneration, and timber. Various combinations of these components define the major fuel groups of grass, shrub, timber, and slash. The differences in fire behavior among these groups are related to the fuel load and its distribution among the fuel particle size classes. In addition to surface fuels, crown fuels are described by canopy bulk density (the foliage contained per unit crown volume), canopy base height (the average height from the ground to the lowest living foliage), and canopy fuel load (the volume of canopy fuel load) (Scott and Burgan 2005).

Table 32 through table 37 show the current surface fire behavior fuel models across the Ashley National Forest (Landfire 2012, Scott and Burgan 2005).

Table 32. Acres and percent of the non-burnable fire behavior fuel model across the Ashley National Forest

Fuel Model Number	Total Acres	Percent Total Area
91, 92, 93 98, 99	251,518	18 %

Table 33. Acres and percent of the grass fire behavior fuel model across the Ashley National Forest

Fuel Model Number	Total Acres	Percent Total Area
101	51,823	4 %
102	49,301	4 %
104	3,443	<1 %

Table 34. Acres and percent of the grass/shrub fire behavior fuel model across the Ashley National Forest

Fuel Model Number	Total Acres	Percent Total Area
121	106,184	8 %
122	165,152	12 %

Table 35. Acres and percent of the shrub fire behavior fuel model across the Ashley National Forest

Fuel Model Number	Total Acres	Percent Total Area
141	7,824	1 %
142	18,481	1 %
143	16	<1 %
145	33,306	2 %
147	7,777	1 %

Table 36. Acres and percent of the timber understory fire behavior fuel model across the Ashley National Forest

Fuel Model Number	Total Acres	Percent Total Area
161	79,638	6 %
162	245	<1 %
165	352,929	25 %

Table 37. Acres and percent of the timber litter fire behavior fuel model across the Ashley National Forest

Fuel Model Number	Total Acres	Percent Total Area
181	1,396	<1 %
182	206	<1 %
183	264,223	19 %
185	853	<1 %
186	278	<1 %
188	4,937	<1 %
189	1	<1 %

Recent Fire History and Trends

By determining the statistical trends of a fire starts, the likelihood of existing and future fire starts can interpolated. The total number of acres burned has increased considerably over the last three decades. There was an abrupt transition in the mid-1980s, from a regime of infrequent large wildfires that were generally short duration, to one with much more frequent larger fire events. Reduced winter precipitation, early spring snowmelt and warmer dry seasons have played a role in this shift. An increase of large wildfires greater than 1,000 acres is particularly robust in lower to mid-elevation forests, where these forests have missed one or more fire return intervals over the last 100 years. This area consists of dry forest types such as ponderosa pine, Douglas-fir and pinyon pine, where fire exclusion has created a departure from the natural fire regimes (Westerling and others 2006).

Over the last three decades, many areas have also experienced mountain pine beetle infestations. Those stands that have significant beetle infestations will continue to change the fuel profile and foliar moisture content overtime. These infestations are creating conditions that are potentially more susceptible to higher intensity wildfires (Cleetus and Mulik 2014).

While large fires have burned a significant number of acres across the Ashley National Forest, they are generally rare, with less than one percent of these fires burning greater than 1,000 acres. Due to the strong influence of the monsoon weather, the fire season is determined by its occurrence, or lack of thereof. Usually the first lighting starts are in late May and account for sixty-eight percent of the fires. Due to vegetation becoming green and less flammable, 77 percent of the fires are usually less than ¼ of an acre and are easily managed. The potential for larger fires (i.e., greater than 100 acres) usually occurs between late-June and mid-July. As the monsoons get into full swing by mid to late July, all fires are less than 100 acres. As the monsoon influence subsides in the fall, fires usually remain relatively small and manageable, due to the shorter days and reduced fire danger (Erskine 2013). Figures 6, 7, 8, and 9 describe the fire statistics for the Ashley National Forest since 1970.

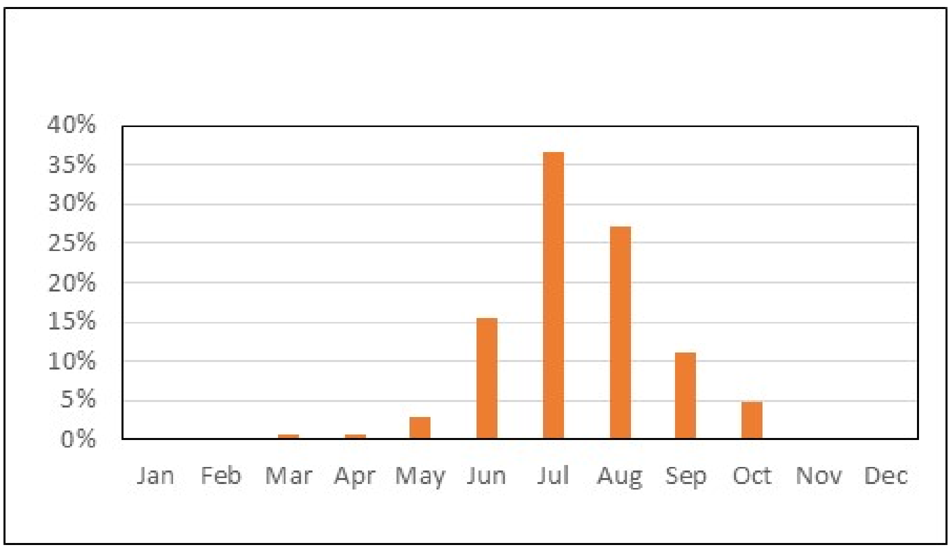


Figure 26. Ashley National Forest percent of fire by each month from 1970 to 2014

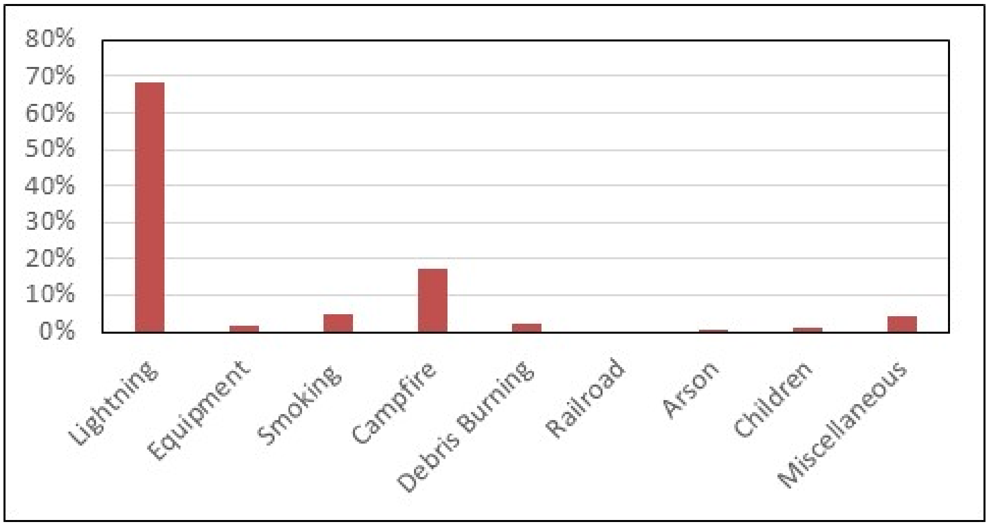


Figure 27. Ashley National Forest percent of fires by cause class

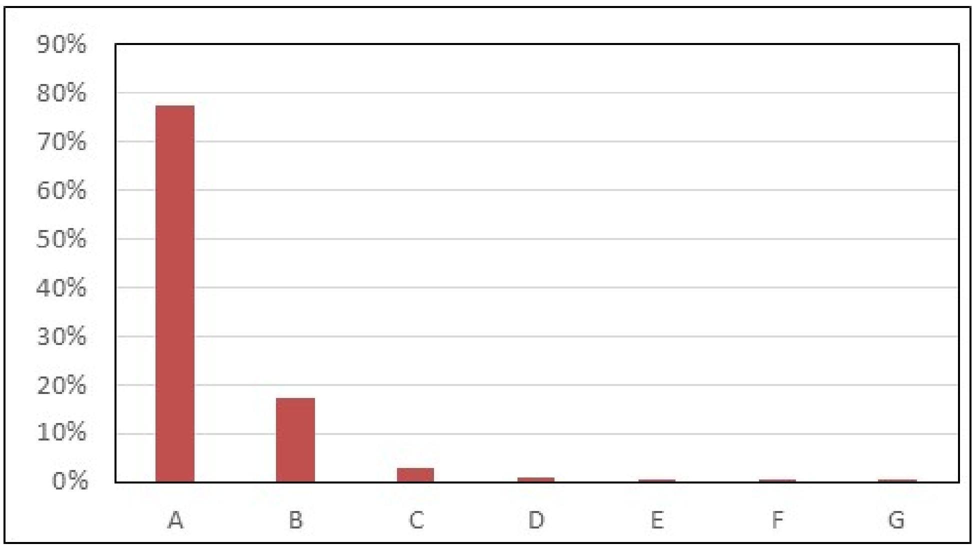


Figure 28. Ashley National Forest percent of fires by size class from 1970 to 2014

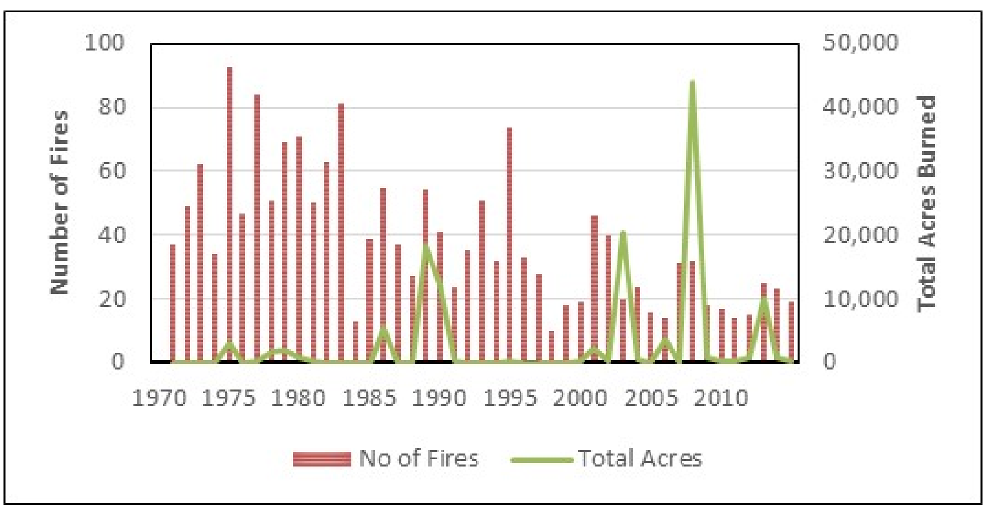


Figure 29. Ashley National Forest number of fires and total acres burned from 1970 to 2014

Literature Cited

Agee, J.K. 1996. Fire regimes and approaches for determining fire history. Pp. 12-13 In: Hardy, C.C.; Arno, S.F., ed., The use of fire in forest restoration. Ogden, Utah: USDA Forest Service, Intermountain Research Station. pp.

Agee, J. K. 1998. The landscape ecology of western forest fire regimes. Northwest Sci. 72, 24–34.

Allen, C.D.; Breshears, D.D. 1998. Drought-induced shift of a forest–woodland ecotone: rapid landscape response to climate variation. Proceedings National Academy of Sciences USA. 95: 14839–14842. Available online at http://www.pnas.org/content/95/25/14839.full.pdf+html; last accessed January 12, 2009.

Anderson, J. E.; Holte, K. E. 1981. Vegetation development over 25 years without grazing on sagebrush-dominated rangeland in southeastern Idaho. J. Range Manage. 34: 25-29.

Anderson, W. E. and Scherzinger, R. J. 1975. Improving Quality of Winter Forage for Elk by Cattle Grazing. Journal of Range Management 28 (2): 120-125.

Andrews, P.L.; Loftsgaarden, D.O.; Bradshaw, L.S. 2003. Evaluation of fire danger rating indexes using logistic regression and percentile analysis. International Journal of Wildland Fire 12: 213–226.

Arendt, P.A., W.L. Baker. 2013. Northern Colorado Plateau piñon-juniper woodland decline over the past century. Ecosphere, Volume 4(8), Article 103, http://dx.doi.org/10.1890/ES13-00081.1 Arno, S. F. 1980. Forest fire history in the northern Rockies. Journal of Forestry. 78(8): 460-465.

Arno, S.F. 1996a. The concept: Restoring ecological structure and process in ponderosa pine forests. Pp. 37-38/ In: Hardy, C.C.; Arno, S.F., ed., The use of fire in forest restoration. Ogden, Utah: USDA Forest Service, Intermountain Experiment Station.

Arno, S.F. 1996b. The seminal role of fire in ecosystem management -impetus for this publication. Pp. 3-5 In: Hardy, C.C.; Arno, S.F., ed., The use of fire in forest restoration. Ogden, Utah: USDA Forest Service, Intermountain Experiment Station.

Arno, S.F.; Parsons, D.J.; Keane, R.E. 2000. Mixed-severity fire regimes in the northern Rocky Mountains: Consequences of fire exclusion and options for the future. In: Wilderness science in a time of change conference. USDA For. Serv. RMRS-P-15-Vol-5, pp. 225-232.

Baker, W. L. 1983. Alpine vegetation of Wheeler Peak, New Mexico, U.S.A. Arctic and Alpine Research. 15: 223-240.

Bassman, J.H.; Gillespie, B.; Fisher, C. 2015. Disturbance factors driving ecological pattern and process in the Blackfoot Swan Landscape Restoration Project Area. Bigfork, Montana: USDA Forest Service, Northern Region, Regional Office. 45 pp.

Barker, J. R.; McKell, C. M. 1983. Habitat differences between basin and Wyoming big sagebrush in contiguous populations. J. Range Manage. 36: 450-454.

Bartos, D. L. 2001. Landscape dynamics of aspen and conifer forests. In: Shepperd, W.D., Binkley, D., Bartos, D.L., Stohlgren, T.J., Eskew, L.G. (Eds.), Sustaining Aspen in Western Landscapes: Symposium Proceedings, U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experimental Station. RMRS-P-18, pp. 5-14.

Bartos, D. L., Mueggler, W. F. 1981. Early succession in aspen communities following fire in Western Wyoming. Journal of Range Management. 34: 315-18.

Bates, J. D.; Rhodes, E. C.; Davies, K. W.; Sharp, R. 2009. Post fire succession in big sagebrush steppe with livestock grazing. Rangeland Ecology and Management 62: 98-110.

Behle, W.H., Perry, M.L. 1975. Utah birds: check-list, seasonal and ecological occurrence charts and guides to bird finding. University of Utah, Museum of Natural History, Salt Lake City, UT. 144 p.

Bentz, B.J.; A.S. Munson. 2000. Spruce beetle population suppression in northern Utah. Western Journal of Applied Forestry, 15, 122-128.

Bentz, B.J.; Régnière, J.; Fettig, C.J.; Hansen, E.M.; Hayes, J.L.; Hicke, J.A.; Kelsey, R.G.; Negrón, J.F.; Seybold, S.J. 2010. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. BioScience, 60(8), pp.602-613.

Bentz, B.J.; Jönsson, A.M. 2015. Modeling bark beetle responses to climate change. Chapter 13, In Vega, F. and R. Hofstetter eds., Bark Beetles: Biology and Ecology of Native and Invasive Species. Elsevier.

Billings, W. D. 1973. Arctic and alpine vegetation: similarities, differences, and susceptibility to disturbances. Bio Science. 23: 697-704.

Binkley, D., T. Sisk, C. Chambers, J. Springer, and W. Block. 2007. The Role of Old-Growth Forests in Frequent-Fire Landscapes. *Ecology and Society* **12**(2): 18. [online] URL: http://www.ecologyandsociety.org/vol12/iss2/art18/

Bistryski, J.R. 1996. Decision memo: Roosevelt/Duchesne District Prescribed Burns. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Duchesne Ranger District, Duchesne, UT. 6 p.

Bistryski, J.R. 2000. Decision memo: Roosevelt/Duchesne District Prescribed Burns. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Duchesne Ranger District, Duchesne, UT. 12 p.

Blaisdell, J. P.; Holmgren, R. C. 1984. Managing Intermountain rangelands--salt desert shrub ranges. Gen. Tech. Rep. INT-163. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 52 p.

Boyd, C. S.; Svejcar, T. T. 2011. The influence on plant removal on succession in Wyoming big sagebrush. J. of Arid Environments. 75: 734-741.

Bradley, Anne F., Noste, Nonan V., Fischer, Willian C. 1992. Fire Ecology of Forests and woodlands in Utah. Gen. Tech. Rep. INT-287. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 128 p.

Bradshaw, L.S.; Deeming, J.E.; Burgan, R.E.; Cohen, J.D. 1983. The 1978 National Fire-Danger Rating System: technical documentation. General Technical Report GTR-INT-169. USDA Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah pp.

Briton, C. M.; Clark, R. G. 1984. Effects of fire on sagebrush and bitterbrush In: Sanders, K.; Durham, J. eds. Rangeland fire effects: a symposium; (xx date of sym) U.S. Department of Interior, Bureau of Land Management. 22-26.

Brown, J. K., DeByle, N. V. 1987. Fire damage, mortality and suckering in aspen. Canadian Journal Forest Res. 17: 1100-1109.

Brown, J.K. 2000. Wildland fire in ecosystems: effect of fire on flora, Introduction and Fire Regimes. General Technical Report RMRS-GTR-42-VOL-2. 1-7 pp.

Brown, G. D. 2006. An alpine plant community classification for the Uinta Mountains, Utah. Ogden, UT: U.S. Forest Service, Intermountain Region, Ashley and Wasatch-Cache National Forests. 140 p.

Brown, R.T.; Agee, J.K.; Franklin, J.F. 2004. Forest restoration and fire: Principles in the context of place. Conservation Biology 18(4): 903-912.

Bryant, J. P.; Scheinberg, E. 1970. Vegetation and frost activity in an alpine fellfield on the summit of Plateau Mountain, Alberta. Canadian J. of Bot. 751-771.

Bunting, S. C. 1984. Fires in sagebrush-grass ecosystems: successional changes. In: Sanders, K.; Durham, J. eds. Rangeland fire effects: a symposium; U.S. Department of Interior, Bureau of Land Management. 7-11.

Bunting, S. C. 1985. Fire in sagebrush-grass ecosystems: successional changes. In: Sanders, K.; Durham, J. eds. Rangeland fire effects--a symposium: 1984 November 27-29; Boise, ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office. 7-11.

Bunting, S.C.; Kingery, J.L; Strand, E. 1999. Effects of succession on species richness of the western juniper woodland/sagebrush steppe mosaic. In: S.B. Monsen, and R. Stevens, comps. Proceedings: ecology and management of pinyon/juniper communities within the Interior West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Byler, J.W.; Hagle, S.K. 2000. Succession functions of pathogens and insects. Ecoregion Sections M332a and M333d in northern Idaho and western Montana. Forest Health Protection Report No. 00-09. USDA Forest Service, Northern Region, Ogden, UT. 45 pp

Chadde, S.W., J.S. Shelly, R.J. Bursik, R.K. Moseley, A.G. Evenden, M. Mantas, F. Rabe, and B. Heidel. 1998. Peatlands on National Forests of the Northern Rocky Mountains: ecology and conservation. Gen. Tech. Rep. RMRS-GTR-11. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Chambers, J.C.; McArthur, E.D.; Monson, S.B.; Meyer, S.E.; Shaw, N.L.; Tausch, R.J.; Blank, R.R.; Bunting, S.; Miller, R.R.; Pellant, M.; Roundy, B.A.; Walker, S.C.; Whittaker, A. 2005. Sagebrush steppe and pinyon-juniper ecosystems: effects of changing fire regimes, increased fuel loads, and invasive species. Joint Fire Sciences Report. Project #00-1-1-03.

Christenson, G.E. 1997. Investigation of Erosional Ravines in Dry Fork Canyon, Utah. Technical Report for 1997 Applied Geology Program: Report Investigation 236. Utah Geological Survey.

Clary, W.P. 1971. Effects of Utah juniper removal on herbage yields from Springerville soils. Journal of Range Management 24: 373-378.

Cleetus, R.; Mulik, R. 2014. Playing with Fire, How Climate Change and Development Patterns are contributing to the soaring costs of western wildfires.

http://www.ucsusa.org/global_warming/science_and_impacts/impacts/climate-change-development-patterns-wildfire-costs.html#.VUkn0ZgcSfA. (Accessed: 5 May 2015).

Cleland, D.T., P.E. Avers, W.H. McNab, M.E. Jensen, R.G. Bailey, T. King, and W.E. Russell. 1997. National hierarchical framework of ecological units. In, Boyce, M.S.; Haney, A., eds. Ecosystem management applications for sustainable forest and wildlife resources. Yale University Press, New Haven, CT. pp. 181-200.

Cooper, S. V.; Lesica, P.; Page-Dumroese, D. 1997. Plant community classification for alpine vegetation on the Beaverhead National Forest, Montana. Gen. Tech. Rep. INT-GTR-362. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 61 p.

Covington, W. W.; Moore, M.M. 1994. Post settlement changes in natural fire regimes and forest structure: ecological restoration of old-growth ponderosa pine forests. J. Sustain. For. 2, 153-181.

Cox, C. F. 1933. Alpine succession on James Peak, Colorado. Ecol. Monog. 3: 299-372.

Crane, K.K, Mosley, J.C., Mosley, T.K, Frost, R.A, Smith, M.A., Fuller W.L., and Tess, M.W. 2016. Rangeland Ecology & Management, 69(4):319-325.

Dai, A. 2016. The Climate Guide: Palmer Drought Severity Index (PDSI). https://climatedataguide.ucar.edu/climate-data/palmer-drought-severity-index-pds (Accessed 12/23/2106).

Daniels, L.D. 2004. Climate and fire: A case study of the Cariboo forest, British Columbia. In: Taylor, L., Zelnik, J., Cadwallader, S., Hughes, B. (Eds.), Proceedings of the mixed severity fire regimes: Ecology and management conference. pp. 235-246.

Davies, K. W.; Bates, J. D.; Miller, R. F. 2006. Vegetation characteristics across part of the Wyoming big sagebrush alliance. Rangeland Ecology and Management 59: 667-575.

Davies, K. W.; Bates, J. D. 2010. Vegetation characteristics of mountain and Wyoming big sagebrush plant communities in the northern Great Basin. Rangeland Ecology and Management 63: 461-466.

DeByle, N.V. 1985. Wildlife. In: Aspen: ecology and management in the western United States, DeByle, N.V., Winokur, R.P., editors. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. pp. 135-152.

DeByle, N.V. 1985b. Water and Watershed. In: Aspen: ecology and management in the western United States, DeByle, N.V., Winokur, R.P., editors. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. pp. 153-160.

DeByle, N.V., Winokur, R.P., editors. 1985. Vegetation regeneration. In: Aspen: ecology and management in the western United States, Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. 283 p.

Douglas, G. W.; Bliss, L. C. 1977. Alpine and high subalpine plant communities of the north Cascades Range, Washington and British Columbia. Ecological Monographs 47: 113-150.

Dwire, Kate. 2016. Condition, natural range of variability, and key ecosystem characteristics of groundwater dependent ecosystems on the Ashley National Forest: springs and fens. Chapter 2.

Unpublished report on file at: U.S. Department of Agriculture, Forest Service, Intermountain Region. 44 pp.

Dymerski, Alan D.; Anhold, J. A.; Munson, A. S. 2001. "Spruce beetle (Dendroctonus rufipennis) outbreak in Engelmann spruce (Picea engelmannii) in central Utah, 1986–1998," Western North American Naturalist: Vol. 61 : No. 1 , Article 3. Available at: http://scholarsarchive.byu.edu/wnan/vol61/iss1/3

Eisenhart, K.S. 2004. Historic range of variability of pinyon-juniper woodlands on the Uncompahgre Plateau, western Colorado. Boulder, CO: University of Colorado, Geography Department.

Erskine, I. 2013. Ashley National Forest Fire Management Plan. Unpublished paper on file at: Ashley NF, Vernal UT.pp.

Everett, R.L. 1987. Plant response to fire in the pinyon/juniper zone. In: R.L. Everett, comp. Proceedings – Pinyon/juniper conference; 1986 January 13-16; Reno, NV. Gen. Tech. Rep. INT-215. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.

Fiedler, C.E., R.R. Becker, and S.A. Haglund. 1988. *Preliminary guidelines for uneven-aged silvicultural prescriptions in ponderosa pine.* Pp. 235-241 in Symp. Proc. Ponderosa pine: The species and its management. Baumgartner, D.M. and J.E. Lotan (eds). Washington State Univ., Pullman, WA. First four pages available on-line at http://www.for.gov.bc.ca/hfd/efp/Uneven_Aged/toc/sec05_05b.pdf.

Fiedler, C.E., Arno, S.F, Harrington, M.G. 1995. *Flexible Silvicultural and Prescribed Burning Approaches for Improving Health of Ponderosa Pine Forests.* RM-GTR-278 Pp. 69-74 in Conference on Adaptive Ecosystem Restoration and Management in Flagstaff, Arizona. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Fiedler, C. E., P. Friederici, and M. Petruncio. 2007. Monitoring Old-Growth in Frequent-Fire Forests. *Ecology and Society* 12(2): 22. [online] URL: http://www.ecologyandsociety.org/vol12/iss2/art22/

Fiedler, C. E., P. Friederici, M. Petruncio, C. Denton, and W. D. Hacker. 2007. Managing for Old-Growth in Frequent-Fire Forests. *Ecology and Society* 12(2): 20. [online] URL: http://www.ecologyandsociety.org/vol12/iss2/art20/

Fischer, W.C.; Bradley, A.F. 1987. Fire ecology of western Montana forest habitat types. General Technical Report INT-223. USDA Forest Service, Intermountain Research Station, Ogden, UT. 95 pp.

Fosberg, M.A.; Rothermel, R.C.; Andrews, P.L. 1981. Moisture content calculations for 1000-hour timelag fuels. Forest Science 27: 19-26.

Frisina, M. R. 1992. Elk Habitat Use Within a Rest-Rotation Grazing System. Rangelands 14 (2): 93-96.

Fryer, Janet L. 2009. Artemisia nova. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: http://www.fs.fed.us/database/feis/plants/shrub/artnov/all.html. [2016, November 3].

Giraud, R.E. 2006. Investigation of the 2005 Uinta Canyon Snowmelt Debris Flows, Duchesne County, Utah. Technical Report 06-03. Utah Geological Survey.

Giunta, Andrew D. 2016. Douglas-fir Beetle Mediated Changes to Fuel Complexes, Foliar Moisture Content and Terpenes in Interior Douglas-Fir Forests of the Central Rocky Mountains. Utah State University. *All Graduate Theses and Dissertations.* Graduate Studies at DigitalCommons@USU Paper 4614.

Goodrich, S.K. 2001. Classification and capabilities of woody sagebrush communities of western North America with emphasis on Greater Sage-grouse habitat. In: Shaw, Nancy L.; Pellant, Mike; Monsen, Stephen B., comps. 2005. Sage grouse habitat restoration symposium proceedings; 2001 June 4–7; Boise, ID. Proceedings RMRS-P-38. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. pp. 17-37.

Goodrich, S. K. 2004. Alpine plant communities: factors to consider. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor’s Office, Vernal, UT.

Goodrich, S.K. 2005. Cedar Mountain Allotment, Flaming Gorge District, Ashley National Forest: Condition and trend. Unpublished. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor’s Office, Vernal, UT. 19 p.

Goodrich, S.K. 2005. Condition and trend, Wyoming big sagebrush communities. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor’s Office, Vernal, UT.

Goodrich, S.K. 2005b. Rock Springs Allotment, Flaming Gorge District, Ashley National Forest: Condition and trend. Unpublished. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor’s Office, Vernal, UT. 16 p.

Goodrich, S.K. 2005c. Sugar Loaf Allotment, Flaming Gorge District, Ashley National Forest: Condition and trend. Unpublished. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor’s Office, Vernal, UT. 8 p.

Goodrich, S.K. 2005d. Condition and trend, Wyoming big sagebrush communities. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor’s Office, Vernal, UT.

Goodrich, S.K. 2005e. Vegetation of the Uinta Mountains and its Relationship with Geology and Geomorphology. In Dehler, C.M., Pederson, J.L., Sprinkel, D.A., and Kowallis, B.J., editors, Uinta Mountain geology: Utah Geological Association Publication 33. Ashley National Forest, Vernal, Ut.

Goodrich, S. K. 2006. Alpine plant communities: Uinta Mountains. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor’s Office, Vernal, UT.

Goodrich, S. K. 2006b. Alpine distribution and abundance of pocket gophers in the Uinta Mountains in context of livestock grazing and other factors. Unpublished. On file at: Ashley National Forest, Supervisor’s Office, Vernal, UT. 10 p.

Goodrich, S.K. 2012. Wildland plants of the Uinta and lower Green River Basins. Unpublished. On file: On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor’s Office, Vernal, UT. 361 p.

Goodrich, S. K., Cameron, A. 2011. Pocket gophers and plant community dynamics. Unpublished. On file at: Ashley National Forest, Supervisor's Office, Vernal, UT. 27 p.

Goodrich, S.K., Huber, A.A. 2001. Mountain big sagebrush communities on the Bishop Conglomerate in eastern Uinta Mountains. In: Shrubland ecosystem genetics and biodiversity. McArthur, E.D., Fairbanks, D.J., comps. Proceedings: 2000 June 13-15, Provo, UT. Proc. RMRS-P-21. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT. pp. 336-343.

Goodrich, S. K., Huber, A. A. 2005. Fall Creek allotment: rangeland condition and trend. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Huber, A.A. 2007. Fire and mountain big sagebrush and aspen communities on the South Face Landtype Association of the Uinta Mountains. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Huber, A.A. 2008. Pole Creek allotment: trend and condition. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S. K., Huber, A. A. 2009. Ottoson allotment: condition and trend. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S. K., Huber, A. A. 2009b. Oweep allotment: vegetation condition and trend. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S. K., Huber, A. A. 2009c. Tungsten allotment: condition and trend. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S. K., Huber, A. A. 2009d. Uinta recreation stock allotment: condition and trend of vegetation and ground cover. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Huber, A.A. 2011. Neola North Fire 2007: post burn vegetation monitoring. Ashley National Forest. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S. K., Huber, A. A. 2014. Painter Basin allotment: condition and trend. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Huber, A.A. 2015. Assessment of aspen sustainability, Ashley National Forest and Evanston-Mountain View Districts, Uinta-Wasatch-Cache National Forest. Unpublished. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Huber, A.A. 2015c. Assessment of condition and trend of livestock allotments on the Ashley National Forest. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Huber, A.A. 2015b. Aspen trend south slope Uinta Mountains east. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Huber, A.A., Munroe, B. 2008. Trend of mountain big sagebrush canopy cover and ground cover on burned sites, of Uinta Mountains and West Tavaputs Plateau. In: Proceedings - shrubland under fire: disturbance and recovery in a changing world; 2006 June 6-8; Cedar City, UT. Kitchen, S.G., Pendleton, R.L., Monaco, T.A., Vernon, J., comps. Proc. RMRS-P-52. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO. pp. 151-160.

Goodrich, S.K., Huber, A.A., Munroe, B. 2005. Sage trends following fire, herbicides, and drought, of Uinta Mountains and West Tavaputs Plateau. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S. K., Huber, A. A., Zobell, R. 2005. Part 2, Uinta bollie LTA and alpine moraine LTA, alpine condition and trend, Uinta Mountains. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Zobell, A. 2009. Halogeton displacement of Gardner saltbush in the lower Green River Basin, Wyoming. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Zobell, A. 2011. Trend of Gardner Saltbush and Halogeton in the Lower Green River Basin, Wyoming. In Monaco, Thomas and others comps. 2011. Proceedings – Threats to Shrubland Ecosystem Integrity; 2010 ay 18-20; Logan, UT. Natural Resources and Environmental Issues, Volume XVII. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan Utah, USA.

Goodrich, S.K., Zobell, A. 2012. Lucerne Reference Area. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Zobell, A. 2013. Cedar Mountain Allotment: Condition and trend. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Zobell, A. 2013b. Rock Springs Allotment: Condition and trend. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Zobell, A. 2013c. Sugar Loaf Allotment: Condition and trend. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.K., Zobell, A. 2016. Canopy Cover and Ground Cover in Wyoming Big Sagebrush Communities in the Lucerne Reference Area. Unpublished. U.S. Department of Agriculture, Forest Service, Ashley National Forest, Vernal, UT. 17 p.

Goodrich, S.K., Zobell, A. 2016. Canopy Cover and Ground Cover in Wyoming Big Sagebrush Communities in the Lucerne Reference Area. Unpublished. U.S. Department of Agriculture, Forest Service, Ashley National Forest, Vernal, UT. 17 p.

Goodrich, S. K., Huber, A. A., Zobell, R. 2015. Trend of alpine willows, Uinta Mountains. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Goodrich, S.; McArthur, E. D.; Winward, A. H. 1999. Sagebrush ecotones and average annual precipitation. In: McArthur and others comps. 1999. 88-94.

Goodrich, S.K., McArthur, E.D., Winward, A.H. 1999b. Sagebrush ecotones and average precipitation. In: Proceedings: Shrubland Ecotones. McArthur, E.D., Ostler, W.K., Wambolt, C.L., comps. Proceedings: 1998 August 12-14, Ephraim, UT. Proc. RMRS-P-11. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT. pp. 88-94.

Goodrich, S.K., Nelson, D., Gale, N. 1999. Some features of Wyoming big sagebrush communities on gravel pediments of the Green River in Daggett County, Utah. In: Proceedings: Shrubland Ecotones. McArthur, E.D., Ostler, W.K., Wambolt, C.L., comps. Proceedings: 1998 August 12-14, Ephraim, UT. Proc. RMRS-P-11. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT. pp. 159-167.

Goodrich, S., and D. Rooks. 1999. Control of weeds at a pinyon-juniper site by seeding grasses. In: S.B. Monsen, and R. Stevens comps. 1999. Proceedings: ecology and management of pinyon/juniper communities within the Interior West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Goodrich, S. K., Zobell, R., Huber, A. A., Cameron, A., 2015b. Pocket gophers in alpine and sub-alpine communities, Uinta Mountains. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Graham, E. H. 1937. Botanical studies in the Uinta Basin of Utah and Colorado. Annals of the Carnegie Museum. 26: 232 p.

Gray, S.T., S.T. Jackson, and J.L. Betancourt. 2004. Tree-Ring Based Reconstructions of Interannual to Decadal Scale Precipitation Variability for Northeastern Utah Since 1226 A.D. Journal of the American Water Resources Association (JAWRA) 40(4):94f7-960.

Gruell, G.E. 1999. Historical and modern roles of fire in pinyon/juniper. In: S.B. Monsen, and R. Stevens comps. Proceedings: ecology and management of pinyon/juniper communities within the Interior West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Guyon, J. 2013. Dwarf Mistletoe Management after a Mountain Pine Beetle Outbreak for the Flaming Gorge Ranger District, Ashley National Forest. OFO-TR-13-21. Prepared by John Guyon, Plant Pathologist, State & Private Forestry, Forest Health Protection, Ogden Field Office, Utah

Harper, K.T., and J.N. Davis. 1999. Biotic, edaphic, and other factors influencing pinyon-juniper distribution in the Great Basin. In: S.B. Monsen, and R. Stevens, comps. Proceedings: ecology and management of pinyon-juniper communities within the Interior West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Hawksworth FG and Johnson DW. 1989. Biology and management of dwarf mistletoe in lodgepole pine in the Rocky Mountains. Gen. Tech. Rep. RM-169. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 38 p.

Herron, David. 2007. 2007 Field Report: Uinta Canyon Slope Stability. Unpublished. Internal Paper. Ashley National Forest.

Herron, D. 2016. Personal Communication.

Hessburg, P.F.; Churchill, D.J.; Larson, A.J. [and others]. 2015. Restoring fire-prone Inland Pacific landscapes: seven core principles. Landscape Ecology

Hitchcock, L.E. 1910. Reconnaissance report. Ashley National Forest Season of 1910. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region, Regional Office Historic Files.

Hood, S.M. 2010. *Mitigating old tree mortality in long-unburned, fire-dependent forests: a synthesis.* Gen. Tech. Rep. RMRS-GTR-238. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 71 p.

Howard, Janet L. 1999. Artemisia tridentata subsp. wyomingensis. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: http://www.fs.fed.us/database/feis/plants/shrub/arttriw/all.html. [2016, December 1].

Huber, Allen. 2016. Rare and unique habitat: filtering criteria. Unpublished report on file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest. 8 pp.

Huber, A. A. 2016. Specialist report for vegetation resources for Fall Creek, Ottoson, Oweep, Painter Basin and Tungsten Allotments. Unpublished, with appendices. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Huber, A. A. 2016b. Species richness of select alpine communities, Uinta Mountains. Unpublished. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT. 2 p.

Huber, A.A., Goodrich, S.K. 2011. Community dynamics and structural stages of pinyon-juniper/shrub communities on the Ashley National Forest. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Huber, A.A., Goodrich, S.K. 2015. Aspen trend West Tavaputs Plateau, Ashley National Forest. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Huber, A.A., Goodrich, S.K. 2015b. Aspen trend south slope Uinta Mountains west. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Huber, A.A., Goodrich, S.K. 2016. Pathways of aspen re-initiation and persistence on the Ashley National Forest. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Huber, A.A., Goodrich, S.K., Christensen, R., Thacker, E. 2010. Anthro Mountain wildlife enhancement habitat projects, Ashley National Forest. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

Huber, A., S. Goodrich, and K. Anderson. 1999. Diversity with successional status in the pinyon-juniper/mountain mahogany/bluebunch wheatgrass community type near Dutch John, Utah. In: S.B. Monsen, and R. Stevens comps. Proceedings: ecology and management of pinyon-juniper communities within the Interior West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Heyerdahl, E.; Brown, E.; Kitchen, S. G.; Weber, M. H. 2011. Multicentury fire in forest histories at nineteen sites in Utah and eastern Nevada. U. S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Gen. Tech. Rep. RMRS-GTR-261WWW. 254 p.

Hylland, Michael E. editor. 1996. Guidelines for Evaluating Landslide Hazards in Utah: Utah Geological Survey Circular 92. Utah Department of Natural Resources.

Intermountain Society of American Foresters. 2013. Management of Pinyon-Juniper "Woodland" Ecosystems. Position Statement. February 15, 2013.

Jacobs, B.F., W.H. Romme, C.D. Allen. 2008. Mapping "Old" vs. "Young" Piñon-Juniper Stands with a Predictive Topo-Climatic Model. Ecological Applications, 18(7), pp. 1627-1641. Ecological Society of America

Jenkins, M.J., E.G. Heberteson, S.A. Munson. 2014. Spruce Beetle Biology, Ecology and Management in the Rocky Mountains: An Addendum to Spruce Beetle in the Rockies. *Forests* 2014, *5*, 21-71; doi:10.3390/f5010021.

Jones, J.R., DeByle, N.V.. 1985. Fire. In: Aspen: ecology and management in the western United States, DeByle, N.V., Winokur, R.P., editors. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. pp. 77-81.

Johnson, C. 2000. Climate and environment. In: B. Loosle, and C. Johnson, eds. Dutch John excavations: seasonal occupations on the north slope of the Uinta Mountains. Heritage Report 1-01/2000. Vernal, UT: U.S. Department of Agriculture Forest Service, Intermountain Region, Ashley National Forest.

Johnson, Kathleen A. 2000. Artemisia tridentata subsp. vaseyana. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: http://www.fs.fed.us/database/feis/plants/shrub/arttriv/all.html. [2016, December 1].

Johnson, P. L.; Billings, W. D. 1962. The alpine vegetation of the Beartooth Plateau in relation to cryopedogenic processes and patterns. Ecol. Monog. 32: 105-135.

Kindschy, R. R. 1994. Pristine vegetation of the Jordan Crater kipukas: 1978-91. In: Monsen, S. B.; Kitchen, S. G., comps. Proceedings--ecology and management of annual rangelands; 1992 May 18-21; Boise, ID. Gen. Tech. Rep. INT-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 85-88.

Kirkaldie, J.R. 2009. Decision memo: Twelve Hundred Dollar Ridge Prescribed Burn Project. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Duchesne Ranger District, Duchesne, UT. 9 p.

Kirkaldie, J.R. 2009b. Decision memo: Anthro Mountain - Antelope Prescribed Burn Project. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Duchesne Ranger District, Duchesne, UT. 13 p.

Kirkaldie, J.R. 2009c. Decision memo: Flat, Fossil, Horse, and Tub Ridge Prescribed Burn Project. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Duchesne Ranger District, Duchesne, UT. 9 p.

Kitchen, S.G., McArthur, E.D. 2007. Big and black sagebrush landscapes. In: Fire ecology and management of the major ecosystems of southern Utah. Hood, Sharon M.; Miller, Melanie, editors. Gen. Tech. Rep. RMRS-GTR-202. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. pp. 73-95.

Knapp, P.A. (1996) Cheatgrass (*Bromus tectorum L*.) dominance in the Great Basin Desert: History, persistence, and influences to human activities. Global Environmental Change 6:37-52.

Koch, Peter. 1996. Lodgepole Pine in North America. Part 1: Background. Forest Products Society. Madison, Wisconsin.

Komarkova, V. 1980. Classification and ordination in the Indian Peaks area, Colorado Rocky Mountains. Vegetation 42:149-163.

Koniak, S., and R.L. Everett. 1982. Seed reserves in soil of successional stages of pinyon woodlands. American Midland Naturalist 108: 295-303.

Kowallis, B.J.; Bradfield, T.D. 2005. Landslide Hazards Along the Southern Flank of the Uinta Mountains. In Uinta Mountain Geology: Utah Geological Association Publication 33. Utah Geological Association.

Kurzel, B.P., Veblen, T.T., Kulakowski, D. 2007. A typology of stand structure and dynamics of quaking aspen in northwestern Colorado. Forest Ecology and Management 252 (2007), pp. 176-190.

Laabs, Benjamin J. C. and Carson, Eric C. 2005. Glacial Geology of the Southern Uinta Mountains. In: Dehler, C. M.; Pederson, J.L.; Sprinkel, D. A.; and Kowallis, B J. editors, Uinta Mountain Geology: Utah Geological Association Publication 33. 2005. Utah Geological Association.

Landfire. 2012. Landscape File used for FlamMap. U.S. Department of Agriculture and U.S. Department of the Interior. http://www.landfire.gov/fuel.php. (Accessed November 2016).

Lesica, P.; Cooper, S. V.; Kudray, G. 2007. Recovery of big sagebrush following fire in southwest Montana. Rangeland Ecology and Management. 60: 261-269.

Lewis, M. E. 1970. Alpine rangelands of the Uinta Mountains. Ogden, UT: U.S. Forest Service, Intermountain Region, Ashley and Wasatch National Forests. 75 p.

Loftin, S.R. 1999. Initial response of soil and understory vegetation to a simulated fuelwood cut of a pinyon-juniper woodland in the Santa Fe National Forest. In: S.B. Monsen, and R. Stevens comps. Proceedings: ecology and management of pinyon-juniper communities within the Interior

West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Long, J.N. and Mock, K., 2012. Changing perspectives on regeneration ecology and genetic diversity in western quaking aspen: implications for silviculture. *Canadian Journal of Forest Research*, *42*(12), pp.2011-2021.

Lotan, James E., and David A. Perry. 1983. Ecology and regeneration of lodgepole pine. U.S. Department of Agriculture, Agriculture Handbook 606. Washington, DC. 51 p.

Malesky, D.M., Bentz, B.J., Brown, G., Chappell, L., DeRose, J., Guyon, J., Jorgensen, C.L., Lowrey, L., Lynch, A., Matyjasik, M., McMillin, J., Mercado, J., Morris, J., Negrón, J., Padgett, W., Progar, R., Randall, C. In review (2016). USDA Forest Service, Ecological Disturbances, In: The Intermountain Adaptive Partnership Climate Change Assessment.

Marquiss, R.; Lang, R. 1959. Vegetational composition and ground cover of two natural relict areas and their associated grazed areas in the Red Desert of Wyoming. J. Range Mange. 12: 104-109.

Marr, J. W. 1961. Ecosystems of the east slope of the Front Range in Colorado. University of Colorado Studies, Series in Biology. 8: 1-134.

Mauk, R.L., Henderson, J.A. 1984. Coniferous Forest Habitat Types of Northern Utah. GTR INT-170. Intermountain Forest and Range Experiment Station. Ogden , Utah.

McDaniel, K. C.; Torell, L. A.; Ochoa, C. G. 2005. Wyoming big sagebrush recovery and understory response with Tebuthiuron control. Rangeland Ecology and Management. 58: 65-76.

Mensing, S.; Livingston, S.; Baker, P. 2006. Long-term fire history in Great Basin sagebrush reconstructed from macroscopic charcoal in spring sediments, Newark Valley, Nevada. Western North American Naturalists. 66: 64-77.

Miller, R.F., and P.E. Wigand. 1994. Holocene changes in semiarid pinyon-juniper woodlands. Bioscience 44:465–474. Available online at http://www.jstor.org/stable/ pdfplus/1312298.pdf; last accessed January 12, 2009.

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

Miller, R.; Svejcar, T.; Rose, J. 1999. Conversion of shrub steppe to juniper woodland. In: Monsen, S. B.; Stevens, R. comps. 1999. Proceedings: ecology and management of pinyon-juniper communities within the Interior West; 1997 September 15-18; Provo, UT: Proc. RMRS-P9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 385-390.

Miller, R.F., J.D. Bates, T.J. Svejcar, F.B. Pierson, L.E. Eddleman. 2005. Biology, ecology and management of western juniper. Technical Bulletin 152. Oregon State University., Agricultural Exp. Stn. (June 2005). Available online at http://juniper.oregonstate.edu/bibliography/documents/phpQ65pOk_tb152.pdf; last accessed January 12, 2009.

Miller, R.F., R.J. Tausch, E.D. McArthur, D.D. Johnson, S.C. Sanderson. 2008. Age structure and expansion of piñon-juniper woodlands: a regional perspective in the Intermountain West. Res. Pap. RMRS-RP-69. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky

Mountain Research Station. 15 p. Available online at http://www.fs.fed.us/rm/pubs/rmrs_rp069.pdf; last accessed June 7, 2017.

Mock C J, Brunelle-Daines, AR (1999) A modern analogue of western United States summer palaeoclimate at 6000 years before present. The Holocene 9:541-545.

Mueggler, W. F. 1988. Aspen community types of the Intermountain Region. Gen. Tech. Report INT-250. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 135 p.

Mueggler, W.F., Campbell, R.B. 1986. Aspen community types of Utah. Research Paper INT-362. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 69 p.

Munroe, J. S. 2003. Estimates of little ice age climate inferred through historical rephotography, northern Uinta Mountains, U.S.A. Arctic, Antarctic, and Alpine Research, University of Colorado. Vol. 35, No. 4 (Nov. 2003), pp. 489-498.

Munroe, Jeffrey S.; Laabs, Benjamin J.C.; Pederson, Joel L.; Carson, Eric C. 2005. From Cirques to Canyon Cutting: New Quaternary Research in the Uinta Mountains. In: Pederson, J. and Dehler C. M. editors, Interior Western United States: Geological Society of America Field Guide 6. 2005. Geological Society of America.

Naillon, D., K. Memmott, and S.B. Monsen. 1999. A comparison of understory species at three densities in a pinyon/juniper woodland. In S.B. Monsen, and R. Stevens comps. Proceedings: ecology and management of pinyon/juniper communities within the Interior West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

National Interagency Fuels Fire and Technology Transfer System. 2010. Interagency Fire Regime Condition Class Guidebook, Version 3.0. https://www.frames.gov/partner-sites/frcc/frcc-guidebook-and-forms/. (Accessed: 7 December 2015).

Nature Conservancy. 1995. Rock Creek special interest botanical area: a proposal. Unpublished report on file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest. 29 pp.

Nelson, D. L.; Harper, K. T.; Boyer, K. C.; et al. 1989. Wildland shrub dieoffs in Utah: an approach to understanding the cause. In: Wallace, A.; McArthur, E. D.; Haferkamp, M. R., comps.. Proceedings--symposium on shrub ecophysiology and biotechnology; 1987 June 30-July 2; Logan, UT. Gen. Tech. Rep. INT-256. Ogden, UT: U. S. Department of Agriculture, Forest Service, Intermountain research Station. 119-135.

NOAA. 2016. Climate at a Glance: U.S. Time Series, Palmer Drought Severity Index (PDSI). http://www.ncdc.noaa.gov/cag/. (Accessed: December 2, 2016).

NWCG. 2006. Appendix B. Fire Line Handbook. Boise, ID. 124 pp.

NWCG. 2014. Wildland Fire Incident Management Field Guide. Operations Group, Boise, ID. 160 pp.

O’Brien, RA.; Tymcio, R. P. 1997. Forest resources of the Ashley National Forest. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 6 p.

Ogle, K.; DuMond, V. 1997. Historic vegetation on National Forest Lands in the Intermountain Region. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region. 129 p.

Olsen, W.K., Schmid, J.M., and S.A. Mata. 1996. Stand characteristics associated with mountain pine beetle infestations in ponderosa pine. For. Sci. 42(3): 310-327.

Osburn, W. S. Jr. 1958. Ecology of winter snow-free areas of the alpine tundra of Niwot Ridge, Boulder County, Colorado. Boulder, CO: University of Colorado. Thesis 77 p.

Padgett, W., Reeves, M., Tart, D., Chambers, J., Howell, C., Kitchen, S. Manning, M., Proctor, J. 2016. Chapter 7: Effects of Climate Change on Non-Forested Vegetation Types in the Intermountain Region. Draft. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region. 76 p.

Palmer, C.E. 1993. Returning Fire to Ponderosa Pine Stand. Vernal, Utah: Technical Fire Management pp. Utah, D. 2016. Utah Wildfire Risk Assessment. https://utahwildfirerisk.utah.gov/. (Accessed: December 13,).

Pederson, J. L.; Dehler, C.M. editors. 2005. Interior Western United States, Field Guide 6. The Geological Society of America.

Peters, E. F.; Bunting, S. C. 1994 Fire conditions and occurrence of annual grasses on the Snake River Plain. In: Monsen, S. B.; Kitchen, S. G., comps. Proceedings--ecology and management of annual rangelands; 1992 May 18-21; Boise, ID. Gen. Tech. Rep. INT-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 31-36.

Perry, D.A., Hessburg, P.F., Skinner, C.N., Spies, T.A., Stephens, S.L., Taylor, A.H., Franklin, J.F., McComb, B., Riegel, G., 2011. The Ecology of mixed severity fire regimes in Washington, Oregon, and North California. For. Ecol. Manag. 262(5), 703-717.

Perryman, B. L.; Olson, R. A.; Petersburg, S.; Naumann, T. 2002. Vegetation response to prescribed fire in Dinosaur National Monument. Western North American Nat. 62: 414-422.

Poulsen, C. L., S.C. Walker, and R. Stevens. 1999. Soil seed banking in pinyon/juniper areas with differing levels of tree cover, understory, density and composition. In: S.B. Monsen, and R. Stevens comps. Proceedings: ecology and management of pinyon/juniper communities within the Interior West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Powell, D.C. 1988. Aspen community types of the Pike and San Isabel National Forests in south-central Colorado. U.S. Department of Agriculture, Forest Service, Rocky Mountain Region. R2-Ecol-88-01. 254 p.

Price, B. 1983. Flood Damage Survey Report for the Ashley National Forest. Internal Paper, unpublished USDA, Forest Service, Ashley National Forest.

Rehfeldt, G.E.; Crookston, N.L.; Saenz-Romero, C.; Campbell, E.M. 2012. North American vegetation model for land-use planning in a changing climate: a solution to large classification problems. Ecological Applications. 22: 119–141.

Riley, K.L.; Abatzoglou, J.T.; Grenfell, I.C. [and others]. 2013. The relationship of large fire occurrence with drought and fire danger indices in the western USA, 1984–2008: the role of temporal scale. International Journal of Wildland Fire 22: 894–909.

Rice, Janine; Bardsley, Tim; Gomben, Pete; Bambrough, Dustin; Weems, Stacey; Huber, Allen; Joyce, Linda A. 2015. Assessment of Aspen Ecosystem Vulnerability to Climate Change for the Uinta-Wasatch-Cache and Ashley National Forests, Utah. Gen. Tech. Rep. RMRS-GTR-XXX. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 66 p.

Rogers, P. 1996. Disturbance ecology and forest management: a review of the literature general technical report. INT-GTR-336. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT.

Romme, W.H., J. Clement, J.A. Hicke, D. Kulakoswki, L.H. MacDonald, T. Schoennagel, and T.T. Veblen. 2006. Recent forest insect outbreaks and fire risk in Colorado forests: a brief synthesis of relevant research. Colorado Forest Restoration Institute, Colorado State University, Fort Collins, CO. Available online at http://www.cfri.colostate.edu/docs/cfri_insect.pdf; last accessed March 23, 2009

Romme, W.H., W.C. Allen, J. Bailey, W. Baker, B. Bestelmeyer, P. Brown, K. Eisenhart, L. Floyd-Hanna, D. Huffman, B. Jacobs, R. Miller, E. Muldavin, T. Swetnam, R. Tausch, and P. Weisberg. 2007. Historical and modern disturbance regimes of piñon-juniper vegetation in the western U.S. Available online at http://www.cfri.colostate.edu/docs/P-J_disturbance_regimes_short%20synthesis_5-07.pdf; last accessed January 12, 2009.

Romme, W. H., Floyd, M. L., Hanna, D. 2009. Alpine ecosystems. In: Historical Range of Variability and Current Landscape Condition Analysis: South Central Highlands Section, Southwestern Colorado and Northwestern New Mexico. Colorado Forest Restoration Institute, Colorado State University, U.S. Forest Service, Region 2. pp. 202-204.

Rowlands, P. G.; Brian, N. J. 2001. Fishtail Mesa; a vegetation resurvey of a relict area in Grand Canyon National Park, Arizona. Western North American Nat. 61: 159-181.

Sankey, T. T.; Moffet, C.; Weber, K. 2008. Postfire recovery of sagebrush communities: Assessment using SPOT 5 and very large-scale aerial imagery. Rangeland Ecology and Management 61: 598-604.

Schier, G.A. 1975. Deterioration of aspen clones in the middle Rocky Mountains. Department of Agriculture, Forest Service Research Paper. INT-170. Ogden, UT. 14 p.

Schier, G.A., Jones, J.R., Winokur, R.P. 1985. Vegetation regeneration. In: Aspen: ecology and management in the western United States, DeByle, N.V., Winokur, R.P., editors. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. pp. 29-33.

Schmid J.M., and Frye, R. H. 1977. Spruce Beetle in the Rockies. Rocky Mountain Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture Fort Collins, Colorado, 80521 General Technical Report RM-49.

Scott, J.H.; Burgan, R.E. 2005. Standard fire behavior fuel models: A comprehensive set for use with Rothermel's surface fire spread model. General Technical Report RMRS-GTR-153. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 72 pp.

Short, H.L., W. Evans, and E.L. Boeker. 1977. The use of natural and modified pinyon pinyon-juniper woodlands by deer and elk. Journal of Wildland Management 41: 543-559.

Smith, A.E., Smith, F.W. 2005. Twenty-year change in aspen dominance in pure aspen and mixed conifer/aspen stands on the Uncompahgre Plateau, Colorado, USA. Forest Ecology and Management 213:338-348.

Stanton, M. L.; Rejmdnek, M.; Galen, C. 1994. Changes in vegetation and soil fertility along a predictable snowmelt gradient in the Mosquito Range, Colorado, U. S. A. Arctic and Alpine Research 26: 364-374.

Stewart, S.S. 1911. Report on timber survey of Ashley National Forest. Field season 1909. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region. Regional Office Historic Files.

Stoecker, R. 1976. Pocket gopher distribution in relation to snow in the alpine tundra. In: Steinhoff, H. W.; Ives, J. D. eds. 1976. Ecological impacts of snowpack augmentation in the San Juan Mountains, Colorado. Final Report, San Juan Ecology Project. Fort Collins, CO: Colorado State University: 281-287.

Sturges, D.L., Nelson, D.L. 1986. Snow depth and incidence of snowmold disease on mountain big sagebrush. In: Proceedings – Symposium on the biology of Artemisia and Chrysothamnus. McArthur, E.D., Welsh, B.L., comps. 1984 July 9-13, Provo, UT. Gen. Tech. Rep. INT-200. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT. pp. 215-221.

Svejcar, T. 1999. Implications of weedy species in management and restoration of pinyon and juniper woodlands. In: S.B. Monsen, and R. Stevens comps. 1999. Proceedings: ecology and management of pinyon/juniper communities within the Interior West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station

Tart. D. L. 1996. Big sagebrush plant associations of the Pinedale Ranger District. Pinedale, WY: Bridger-Teton National Forest. 97 p.

Tausch, R.J. 1999. Historic pinon and juniper woodland development. In: S.B. Monsen, and R. Stevens comps. 1999. Proceedings: ecology and management of pinyon/juniper communitites within the Interior West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Tausch, R.J., and R.S. Nowak. 1999. Fifty years of ecotone change between shrub and tree dominance in the Jack Springs pinyon research natural area. In: E.D. McArthur, W.K. Ostler, and C.L Wamboldt, comps. Proceedings: shrubland ecotones; 1998 August 12-14, Ephraim, UT. Proc. RMRS-P-11. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Tausch, R. J., C.L. Nowak, and S.A. Mensing. 2004. Climate change and associated vegetation dynamics during the Holocene: The paleoecological record. In: Chambers, J. C., and J.R. Miller, eds. Chapter 2 – Great basin riparian ecosystems. Washington: Island Press.

Tausch, R.J., and S.M. Hood. 2007. Pinyon/juniper woodlands. In: S.M. Hood, and M. Miller, eds. Fire ecology and management of the major ecosystems of southern Utah. Gen. Tech. Rep. RMRS-GTR-202. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rock Mountain Research Station.

Thorn, C. E. 1982. Gopher disturbance its variability by Braun-Blanquet vegetation units in the Niwot Ridge alpine tundra zone, Colorado Front Range, U.S.A. Arctic and Alpine Research. 14: 45-51.

Tress J., and J.M. Klopatek. 1987. Successional changes in community structure of pinyon-juniper woodlands in north central Arizona. In: R. Everett comp. Proceedings – pinyon-juniper conference; 1986 January 13-15; Reno, NV. Proc. Gen. Tech. Rep. Int-215. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Experiment Station.

USDA Forest Service, Monitoring Studies Inventory. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Vernal, UT. 2060 files.

USDA Forest Service. 1965. Preliminary Evaluation of Flood Frequencies Potentials, Distributions, and Magnitude for Sheep Creek Canyon, Ashley National Forest. Internal Paper, unpublished. USDA, Forest Service, Ashley National Forest.

USDA Forest Service. 1998. Sub-Regional Assessment of Properly Functioning Conditions for Areas Encompassing the National Forests of Northern Utah. Version 1.0 May 1998. Pages UM – 13-22.

USDA Forest Service. 1990. Burns, R.M., and B. H. Honkala (comps.). Silvics of North America. Forest Service Handbook 654. Pgs. 675 -877. Online: http://na.fs.fed.us/spfo/pubs/silvics_manual/table_of_contents.htm

USDA Forest Service. 2005. Land Systems Inventory. Ashley National Forest. Inventory maintained by Forest Ecologist (S. Goodrich) and Forest Soil Scientist (D. Koerner). Original inventory: Andrew E. Godfrey. 1976. Description of Landtypes and Landtype Associations on the Ashley National Forest. Unpublished, USDA Forest Service.

USDA Forest Service. 2006. Green River Landtype Association. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

USDA Forest Service. 2009. Ashley National Forest Ecosystem Diversity Evaluation Report. Draft. USDA, Forest Service, Ashley National Forest.

USDA Forest Service. 2009. *Regional Assessment of Properly Functioning Condition*. Version 2008. Maintained by Clinton K. Williams. Region 4. Latest version available online at http://fsweb.r4.fs.fed.us/unit/vm/ecology

USDA Forest Service. 2011. North Flank Vegetation Management Project Decision Notice and FONSI. Signed by Marcia Pfleiderer, Flaming Gorge District Ranger, Ashley National Forest, Vernal, UT

USDA, Forest Service, Ashley National Forest. 2012. Record of Decision: South Unit Oil and Gas Development Final Environmental Impact Statement: Duchesne Ranger District, Ashley National Forest, Duchesne County, Utah USDA, Forest Service.

USDA Forest Service. 2013. Ponderosa pine maintenance and fuel reduction project. Flaming Gorge Ranger District. Ashley National Forest. Decision Notice signed by Rowdy Muir on May 8th, 2013.

USDA Forest Service. 2013a. Stand Exam data collected on the Half Whiskey Moon Lookout Forest Restoration Project of the Greendale Plateau LTA. Flaming Gorge Ranger District. Ashley National Forest.Dat

USDA Forest Service. 2015a. Mosby Mountain and Lake Mountain Allotments. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

USDA Forest Service. 2015b. Yellowstone Allotment. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor's Office, Vernal, UT.

USDA 2015. Vegetation Prescription for forest and woodlands. Ponderosa prescribed burn—Red Canyon Unit. Prepared by Colette Webb, Silviculturist. Ashley National Forest, Vernal, UT

USDA Forest Service. 2016a. Sagebrush natural range of variation. Online: www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5434335.

USDA Forest Service. 2016b. Intermountain adaptation partnership: vulnerability assessment summaries. Unpublished report on file at: U.S. Department of Agriculture, Forest Service, Intermountain Region. 116 pp.

USDA Forest Service. 2016c. Ashley National Forest landtype association GIS data. J:\fsfiles\gis\lib\ashley\natural\lsi_lta. On file at: Ashley National Forest, Vernal UT.

Utah, D. 2016. Utah Wildfire Risk Assessment. https://utahwildfirerisk.utah.gov/. (Accessed: December 13,).

USDA 2017. Vegetation Prescription for forest and woodlands. Ponderosa prescribed burn—Dowd Mountain Unit. Prepared by Steve Overton, TEAMS Silviculturist. Ashley National Forest, Vernal, UT

Utah Division of Wildlife Resources. 2012. Elk herd unit management plan: elk herd unit #9, south slope. Department of Natural Resources, Utah Division of Wildlife Resources, Vernal, UT. 8 p.

Utah Division of Wildlife Resources. 2012b. Elk herd unit management plan: elk herd unit #11, Nine Mile. Department of Natural Resources, Utah Division of Wildlife Resources, Vernal, UT. 8 p.

Utah Forest Restoration Working Group. 2010. Guidelines for aspen restoration on the national forests of Utah. Western Aspen Alliance, Utah State University, Logan, UT. 48 p.

Utah Reclamation Mitigation and Conservation Commission. 2016. High Mountain Lakes Stabilization Project. Available online at: http://www.mitigationcommission.gov/hmls/hmls_home.htm.

van der Kamp, B.J., Hawksworth, F.G., 1985. Damage control of the major diseases of lodgepole pine. In: Baumgartner, Krebill, D.M., Arnott, J.T., Weetman, G.F. _Eds.., Lodgepole Pine: The Species and Its Management, Symposium Proceedings. Washington State University Cooperative Extension Service, Pullman, WA, USA, pp. 125–132.

Vavra, M. 2005. Livestock Grazing and Wildlife: Developing Compatibilities. Rangeland Ecology & Management 58 (2): 128-134.

Volland, Leonard A. 1985. Ecological classification of lodgepole pine in the United States. In: D.M. Baumgartner, R.G. Krebill, J.T. Arnott, and G.F. Weetman, comps. Symposium proceedings: lodgepole pine, the species and its management; 1984 May 8-10; Spokane, WA; and repeated May 14-16; Vancouver, British Columbia, Canada. Pullman, WA: Washington State University.

Vose, J.M.; S., C.J.; Luce, C.H.; T., P.-W. 2016. Effects of Drought on Forests and Rangelands in the United States: A Comprehensive Science Synthesis. General Tech Report WO-93b. US Department of Agriculture, USFS, Washington D.C. 302 pp.

Walker, D. A.; Halfpenny, J.C.; Walker, M. D.; Wessman, C. A. 1993. Long-term studies of snow-vegetation interactions. Bioscience. 43: 287-301.

Wengert, E.M, Donnelly, D.M., Markstrom, D.C., Worth, H.E. 1985. Wood utilization. In: Aspen: ecology and management in the western United States, DeByle, N.V., Winokur, R.P., editors. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. pp. 169-180.

West, N.E. 1988. Intermountain deserts, shrub steppes, and woodlands. In: M.G. Barbour, and W.D. Billings, eds. North American terrestrial vegetation. New York, NY: Cambridge University Press.

West, N. E. 2000. Synecology and disturbance regimes of sagebrush steppe ecosystems. In: Entwistle, P. G. et al. comps. Proceedings: Sagebrush steppe ecosystems symposium; (xxxxdate); Boise, ID. Publication No. BLM/ID/PT-001001+150. Boise, ID: U.S. Department of Interior, Bureau of Land Management. 15-26.

West, N. E.; Yorks, T. P. 2006. Long-term interactions of climate, productivity, species richness, and growth form in relictual sagebrush steppe plant communities. Western North American Naturalists 66: 502-526.

Westerling, A.L.; Gershunov, A.; Brown, T.J. [and others]. 2003. Climate and Wildfire in the Western United States. Science.

Westerling, A.L.; Hidalog, H.G.; Cayan, D.R.; Swetnam, T.W. 2006. Warming and earlier spring increases western US Forest wildfire activity. Science 313: 940-943.

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

Willard, B. E. 1979. Plant sociology of alpine tundra, Trail Ridge, Rocky Mountain National Park, Colorado. Colorado School of Mines Quarterly. 74: 119 p.

Winward, A. H. 1983. Using sagebrush ecology in wildland management. In: Johnson, K. L., editor. Proceedings of the first Utah shrub ecology workshop; 1981 Sept. 9-10; Ephraim, UT. Logan, UT: Utah State University, College of Natural Resources: 15-19.

Winward, A. H. 1991. A renewed commitment to management of sagebrush grasslands. In: Miller, R. F., editor. Management of sagebrush steppe. Special Rep. 880. Corvallis, OR: Oregon State University Agricultural Experiment Station. 2-7.

Woods, B. A.; Rachlow, J. L.; Bunting, S. C.; Johnson, T. R.; Brocking, K. 2013. Managing high-elevation sagebrush steppe: Do conifer encroachment and prescribed fire affect habitat for pygmy rabbits. Rangeland Ecology and Management. 66: 462-471.

Worrall, J. J., Marchetti, S. B., Egeland, L., Mask, R. A., Eager, T., Howell, B. 2010. Effects and etiology of sudden aspen decline in southwestern Colorado, USA. Forest Ecology and Management 260 (2010) 638-648.

Zobell, R., Goodrich, S.K. 2015. Aspen trend north slope, Uinta Mountains. Powerpoint. On file at: U.S. Department of Agriculture, Forest Service, Ashley National Forest, Supervisor’s Office, Vernal, UT.

Appendices

Appendix A. Description of Lodgepole Pine Structure

Structure stages are similar on the Greendale Plateau and Round Park landtype associations.

Early-seral, post replacement

Lodgepole pine seedlings establish in great abundance the year of, or the year after disturbance (Studies 18-41, 44-19B). Ross sedge and a few other seedbank species and species with wind borne seeds generally increase rapidly. Ross sedge is a common seedbank species in early post burn and post-harvest sites. It is often lacking or widely scattered in dense stands of lodgepole pine that are older than about 100 years. It is found in abundance a year or two following fire and timber harvest. This indicates this plant builds up seedbanks in early-seral and mid-seral stages when conditions are favorable for this species. It declines under increasing shade, and perhaps other factors of competition, as trees increase in size.

Lodgepole pine seedlings and saplings dominate usually within 5 to 10 years. Where seral aspen is present, aspen sprouts can often quickly become the aspect dominant. Sprouts of three to four feet (0.9 to 1.2 meters) high are possible in the first or second year following disturbance.

Mid-seral, closed canopy

Pole-sized or somewhat larger trees with pruned stems and short crowns dominate this stage (Study 19-38A). This stage has likely persisted, in many cases, until the next stand-replacing fire. Due to intense competition for sunlight, water, and other resources, the understory is limited to few, often widely scattered, herbaceous plants. Where aspen is present, and depending on density, it might still dominate in this stage. However, lodgepole pine can be expected to overtop the aspen in this or the following stage.

Mid-seral, open canopy

In many cases open canopy stands do not develop for a long time, and it is likely open stands will not develop before the next stand-replacing fire. Where this stage does develop, a thinning of trees usually occurs. This thinning allows for the diameter of lodgepole pine to increase. Often trees persisting into this stage have larger crowns than the ones that did not persist. Generally, the understory remains suppressed in this stage, except where aspen is present.

Late-seral, open canopy

Lodgepole pine larger than eight inches (20 centimeters) in diameter at breast height dominates, and there might be some response by understory species to the opening of the canopy. At this stage, stands become highly vulnerable to mountain pine beetle epidemics (Study 30-13C), and depending on the degree of openness of the lodgepole pine canopy, aspen might persist in a vigorous state.

Late-seral, closed canopy

Mature lodgepole pine dominates this stage. Aspen can be expected to lose vigor, die back, and will eventually be purged from the stand. The understory is usually limited to a few species capable of persisting in dense shade. Depending on the density of stems, lodgepole pine can persist for 100 years or more in dense pole stands on this landtype association.

Appendix B. Description of Douglas-fir Structure

Five seral stages are described below for Douglas-fir. These descriptions are based on observed burns within the Avintaquin Canyon landtype association (Study 64-42C). Generally, dense stands of Douglas-fir in this landtype association are more likely to develop on northerly aspects. Density and growth form of Douglas-fir on southerly aspects are such that the mid-seral, closed canopy stage does not develop, and later stages apparently develop without going through much of a thinning process.

Early-seral, post replacement

Herbaceous species and sprouting shrubs dominate for a few to several years following fire. Ninebark is a locally common to dominant shrub. Where aspen was present prior to burning, it forms moderate to dense stands of sprouts in the early-seral stage. The density of these aspen sprouts depends on the abundance of aspen prior to the burn.

Mid-seral, closed canopy

Saplings and poles of Douglas-fir dominate. Young aspen dominates patches where aspen sprouts were numerous. Early-seral herbaceous species and shrubs decline, shade tolerant understory species increase, and the self-pruning of lower limbs begin.

Mid-seral, open canopy

Self-thinning of poles and self-pruning of lower limbs due to shading from adjacent trees begins. Pruning advances until live limbs persist in only the upper half of the stem. Even-aged stands of trees dominate.

Late-seral, open canopy

The diameter at breast height of surviving trees increases, and large limbs develop. Crowns change from narrow spire-shapes to broad and sometimes asymmetrical shapes. Recruitment of shade-tolerant seedlings and saplings begins, and stands of mixed ages develop.

Late-seral, closed canopy

Mixed aged stands dominate in this stage. Older and larger trees have thick bark, large limbs, and asymmetrical crowns. Accumulation of woody debris continues, and there is an increase in standing dead trees of larger diameter. Large woody debris increases and stands dominated by large trees become highly vulnerable to bark beetle epidemics as observed on this landtype association in 2005.

Appendix C. Description of Mixed Conifer Structure

Seral stages as described on the Trout Slope (TS), Alpine Moraine (AM), and Uinta Bollies (UB) landtype associations.

Early-seral, post replacement

Alpine Moraine: Herbaceous species dominate the early-seral communities. Ross sedge, which is a seedbank species, is a common early-seral species in coniferous forests of the Uinta Mountains. At higher elevations where Engelmann spruce dominates, herbaceous species dominate for few to several decades before spruce regeneration dominates the community. At lower elevations where lodgepole pine is common, herbaceous species dominate for a few years, after which, lodgepole pine seedlings and saplings dominate.

Trout Slope: At very early stages in mixed conifer stands, herbaceous species dominate. Shrub dominance of early-seral communities is rather uncommon in this landtype association. In relatively few cases, snowbush ceanothus forms a dense shrub layer. Ross sedge, which is a strong seedbank species, is often abundant in drier sites. In moist areas, bluejoint forms dense patches. This shade tolerant species is highly capable of persisting in open areas as well as in the shade and increases with the removal of shade following fire and harvest. At somewhat higher elevations, and especially in areas adjacent to or near subalpine meadows, tufted hairgrass and other meadow species dominate the herbaceous layer in the early years of succession.

Of the three tree species, lodgepole pine commonly dominates early-seral, post-replacement stands. This is especially so at lower elevations. At higher elevations, Engelmann spruce becomes more important in early-seral stands. Subalpine fir seems to be much more common on soils with heavy clay horizons derived from the shale of the Uinta Mountain Group. It is relatively common for the density of tree seedlings to vary with the dominance of early-seral herbaceous species. However, a number of factors are likely to influence stocking rate, and density of seedlings is not always predictable by herbaceous dominance. By comparison, sites dominated by Ross sedge often have greater density of tree seedlings than areas dominated by bluejoint. Meadow-like conditions have persisted for a decade or more in areas occupied by tufted hairgrass and other meadow species. Dense pole stands are more likely to develop in Ross sedge areas than in bluejoint or tufted hairgrass areas.

Uinta Bollie: The early-seral, post replacement seral stage is dominated by forbs and a few sprouting shrubs. Plants can be expected to be scattered and of low-growth form.

Mid-seral, closed canopy

Alpine Moraine: At higher elevations, mid-seral closed-canopy conditions might take 200 years or more to develop. This usually includes trees of various ages and sizes, with smaller trees filling gaps between the larger trees. At lower elevations, dense stands of even-aged lodgepole pine quickly dominate. These stands persist into the mid-seral, open-canopy stage.

Trout Slope: Lodgepole pine commonly dominates this seral stage, especially at lower elevations. Stand density varies considerably with dense pole stands forming in places. However, dense pole stands on the Trout Slope landtype association are not as common or extensive as they are on the PP4 and PP5 landtypes of the Parks Plateau landtype association or the Greendale Plateau landtype association. Engelmann spruce and subalpine fir species are commonly absent or lightly represented in denser stands of lodgepole pine. These species typically increase in abundance with elevation and other factors.

Uinta Bollie: This stage can be expected to take between 200 to 300 years or longer to develop and, due to high elevation and rocky conditions, closed canopy might not be within the capability of some of the more forested sites that are very rocky. Engelmann spruce and, less commonly, subalpine fir are the expected dominate species.

Mid-seral, open canopy

Alpine Moraine: At higher elevations, larger trees dominate with most of the previous openings occupied by larger trees. Some smaller trees persist in places and recruitment of shade tolerant seedlings continues. At lower elevations, lodgepole pine has developed the typical pruned trunk growth form with saplings of spruce and fir increasing in the understory in places. In other places, lodgepole pine continues as the only tree species.

Grouse whortleberry often increases in the understory in this stage. Other species of understory shrubs are quite limited in most forest stands throughout the landtype association. Associated understory species are generally short (less than 20 inches or 51 centimeters tall) and vertical structure of these forests is commonly limited to a low layer of herbaceous species, grouse whortleberry, and that provided by trees. Where mature lodgepole pine is dominant, pruned stems dominate the structure in the lower ½ to ¾ of the tree layer. Above the bare stems, live crowns usually provide the bulk of the forest cover. Where Engelmann spruce and subalpine fir dominate, live branches often extend to near ground level. In some places, subalpine fir provides a low shrub-like layer.

Trout Slope: This seral stage is more likely to develop in areas of lighter stocking rates of lodgepole pine at lower elevations. Cover or abundance of Engelmann spruce increases with elevation. In this seral stage, lodgepole pine is trending toward maturity and, depending on tree density; the stems of lodgepole pine are pruned. However, in open stands, seedlings of lodgepole pine are still establishing and older trees still have an open growth form. The limbs of Engelmann spruce and subalpine firs either extend to the ground level, or are very close to the ground.

Uinta Bollie: This stage is dependent on an abundant recruitment of seedlings that result in dense tree stocking. This process appears to be outside the capability of some areas of the Uinta Bollie landtype association. Low density of tree recruitment is common for this area; however, rather dense stands of Engelmann spruce set in rock do develop in some places. This process likely takes hundreds of years.

Late-seral, open canopy

Alpine Moraine: Depending on elevation and other factors, Engelmann spruce or lodgepole pine dominates the overstory in this stage. In comparatively few areas, subalpine fir dominates. Seedlings and saplings are mostly Engelmann spruce and subalpine fir. Seedlings and saplings contribute to multiple layers of canopy. Cover of herbaceous species decreases and grouse whortleberry increases.

Trout Slope: Depending on elevation and other factors, Engelmann spruce or lodgepole pine dominates the overstory in this stage. In comparatively few areas, subalpine fir dominates. Seedlings and saplings are mostly Engelmann spruce and subalpine fir, and contribute to the multiple layers of the canopy. Typically, in this stage, cover of herbaceous species decreases and grouse whortleberry increases.

Uinta Bollie: The description of mid-seral, open canopy is also applicable to this stage of forest development.

Late-seral, closed canopy

Alpine Moraine: At higher elevations, Engelmann spruce often dominates with an understory of grouse whortleberry. At somewhat lower elevations Engelmann, spruce and subalpine fir species dominate usually with some large lodgepole pine trees present. Engelmann spruce and subalpine fir produce seedlings in their own shade, and these seedlings appear capable of long-term persistence in the absence of disturbance. On the quartz-rich sandstones of the Uinta Mountain Group, it is common for lodgepole pine seedlings to establish in openings created as the older trees die.

Trout Slope: A mixture of Engelmann spruce, subalpine fir, and lodgepole pine is common in this seral stage with the great majority of lodgepole pine being mature and Engelmann spruce and subalpine fir represented by mature and other age-classes. Some pruning of limbs of mature Engelmann spruce is usually evident, and to a lesser extent, this is so for subalpine fir. Subalpine fir is sometimes represented by decadent and dead trees while Engelmann spruce remains vigorous.

On quartz-rich sandstones of the Uinta Mountain Group, subalpine fir sometimes forms a shrub-like layer that appears to persist for many years without trending toward maturity or dominance. In areas underlain by dense clays derived from Red Pine Shale, subalpine fir seems to trend toward maturity in greater abundance and more rapidly than on the sandstone areas. The understory is commonly dominated by grouse whortleberry, but other low-growing species of dwarf bilberry and whortleberry are also locally common to dominant.

Uinta Bollie: Under the most productive conditions, there might be somewhat of a closure of canopy as trees reach full stature and crown development occurs in this stage. However, the spacing of trees in the rocky high-elevation stands of this landtype association is often sparse enough that “closed canopy” is not likely an accurate description of these forests. However, rather dense stands of Engelmann spruce set in rock do develop in some places. This process likely takes hundreds of years, and may still be in progress since the last glacial event.