# Ashley National Forest Assessment

## Insects and Disease Report

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**for:**
Ashley National Forest

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## Introduction

Native forest insects and diseases are an integral part of forest ecosystems. Current ecological theories postulate that 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 by shifting mosaics of diverse communities and habitats across a landscape. Natural forest disturbance regimes have changed due to anthropogenic influences. Influences, such as climate, fire suppression/exclusion, recreation, mining, timber management (policy, markets, availability of infrastructure), and grazing, impact the availability of susceptible host type on the landscapes we manage. Multiple use management can (directly and indirectly) impact forest structure and composition (Fettig et al. 2014). Regardless of anthropogenic influences and management direction, however, natural disturbance regimes remain a 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/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, to a varying extent, can affect important ecosystem processes. These processes can include the allocation of water and nutrients within 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 (structure, type, and species diversity), so do their associated insect and disease communities and the subsequent likelihood of serious impacts.

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

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.

The insects and diseases in table 1 through table 7 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 tables. 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 1. 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)<br>*Malacosoma disstria* (Hubner) | Aspen, willow, cherry, cottonwood, mountain mahogany, oak, alder, and birch |

*Meets criteria listed above

**Table 2. 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 3. Native bark beetles and host species on the Ashley National Forest**

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

**Table 4. 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 5. 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 6. 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 7. 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

# Indicators

Indicator species are insects and diseases that have significantly impacted critical ecosystem services associated with their host/vegetation type. The measure of the effect (or scale) is in acres affected (i.e. area of mortality of host trees within the vegetation type) by a particular insect or disease.

# Scale

Given suitable stand conditions and susceptible landscapes, endemic populations of eruptive bark beetles can achieve exponential growth, affecting 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[1], the area per disturbance event per time period, and the percentage of area within a cover type

[1] Patches refer to an area of vegetation that is relatively homogeneous internally and differs from surrounding elements (Helms 1998). Here, "patches" refers to the mapped damage (polygons) in vegetation due to a particular damage agent.

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 through 2015 (19 years).

# Existing Conditions of the Indicators

## Dwarf Mistletoes

Older ground surveys found 58 percent of lodgepole pine stands on the Ashley National Forest, and 8 percent of the ponderosa pine infected with dwarf mistletoes (Hoffman 1978). More recent but spatially coarse data, from sampling completed by the Forest Inventory and Analysis group, 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 (https://www.fia.fs.fed.us/tools-data/).The largest impact of dwarf mistletoes on their host tree is growth reduction, but they 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 and/or drying scenarios (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[2] and scattered trees of low vigor and poor health. Where there is an abundance of trees recently wind thrown, 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 (i.e. fire, wind throw, and avalanche) have led to exponentially expanded populations (Kegley, 2011).

The extent of forested area on the Ashley 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 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 the Ashley National Forest boundary), and Greendale Junction on the Flaming Gorge Ranger District.

## 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. The mortality has altered the amount, composition, and arrangement of living and dead biomass in both the pine and mixed conifer communities that were affected.

[2] Blowdown refers to a tree or trees felled or broken off by wind—*synonym* windfall, wind throw (Helms 1998)

Mountain pine beetle affects lodgepole pine, ponderosa pine, limber pine, and bristlecone pine on the Ashley. The amount[3] of forested area typed as 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 on the Ashley totals 45,770 acres. For limber pine and bristlecone, the Ashley has a total of 40 acres of mapped stands of limber pine forest type and no mapped bristlecone pine stands. The incidence of limber pine, however, is widespread. Limber pine on the Ashley occurs as a minor species in other forest types. Limber pine is estimated to occur on 287,500 acres of forested land on the Ashley, as part of the limber pine and subalpine fir habitat series (O'Brien and Tymcio 1997). Bristlecone pine also occurs as a minor species, but is confined to a few isolated locations on the western Tavaputs Plateau of the South Unit of the Ashley National Forest (Goodrich & Neese 1986); the acreage that 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 (i.e. approximately 60 percent of area affected) of lodgepole pine forests and mixed conifer forests in this time period. In ponderosa pine, mountain pine beetle has killed trees on about 16,175 acres (i.e. 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 (1 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 National Forest 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 forestwide 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, indicate a decrease to 200 affected acres where mountain pine beetle had caused an estimated two trees per acre of mortality in limber pine[4]. 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.

## Piñon *Ips*

Piñon pine is part of the piñon-juniper woodland vegetation type and totals 122,383 acres on the Ashley. 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 kill 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).

Piñon *Ips* beetle-caused tree mortality 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.

## 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 in conjunction

[3] Acreage totals in this paragraph include the seral aspen types to the lodgepole, mixed conifer, and ponderosa types
[4] Findings have not yet been confirmed or summarized by Forest Health Protection staff.

with changes in aspen ecosystems due to changes in climate. 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. Ashley National 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, 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 on the Ashley. 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, but 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[5] can also play a role in these forests, as the agents causing periodic episodes of mortality that can trigger regeneration (Kurzel et al. 2007).

Areas 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. Marssonina is neither dieback nor decline, but a defoliation event. Defoliation caused by Marssonina is associated with a moist period right after aspen leaf-out (open leaf buds) and can trigger a dieback event when there are two or more years of repeated damage. Although Forest Health Protection staff observed wood borer activity on the Red Cloud Loop of Vernal District in 2015, 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 Forest Boundary (Vernal and Flaming Gorge Districts). Dieback and decline were also detected near Left Fork Canyon and Fork Ridge.

## Spruce Beetle

On the Ashley, spruce beetle affects the mixed conifer (spruce, lodgepole pine, subalpine fir) and spruce forest types. The extent of forested area on the Ashley that typed as mixed conifer[6] 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 2015 (46,068 trees, 10,348 acres), compared to 2014 (12,410 trees, 5,200 acres). Aerial detection surveyors mapped five to14 tree 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

[5] Senescence refers to the life phase of an organism or a part of the organism that precedes natural death, usually involving a decreased ability to repair damage and degradation (Helms 1998)

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

mortality throughout the district. The Flaming Gorge and Vernal Districts had no spruce beetle detections in 2014, but in 2015, surveys detected 184 affected acres on Flaming Gorge and 1,078 acres on Vernal District.

# Trends

Figure 1 shows the acreage of 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.

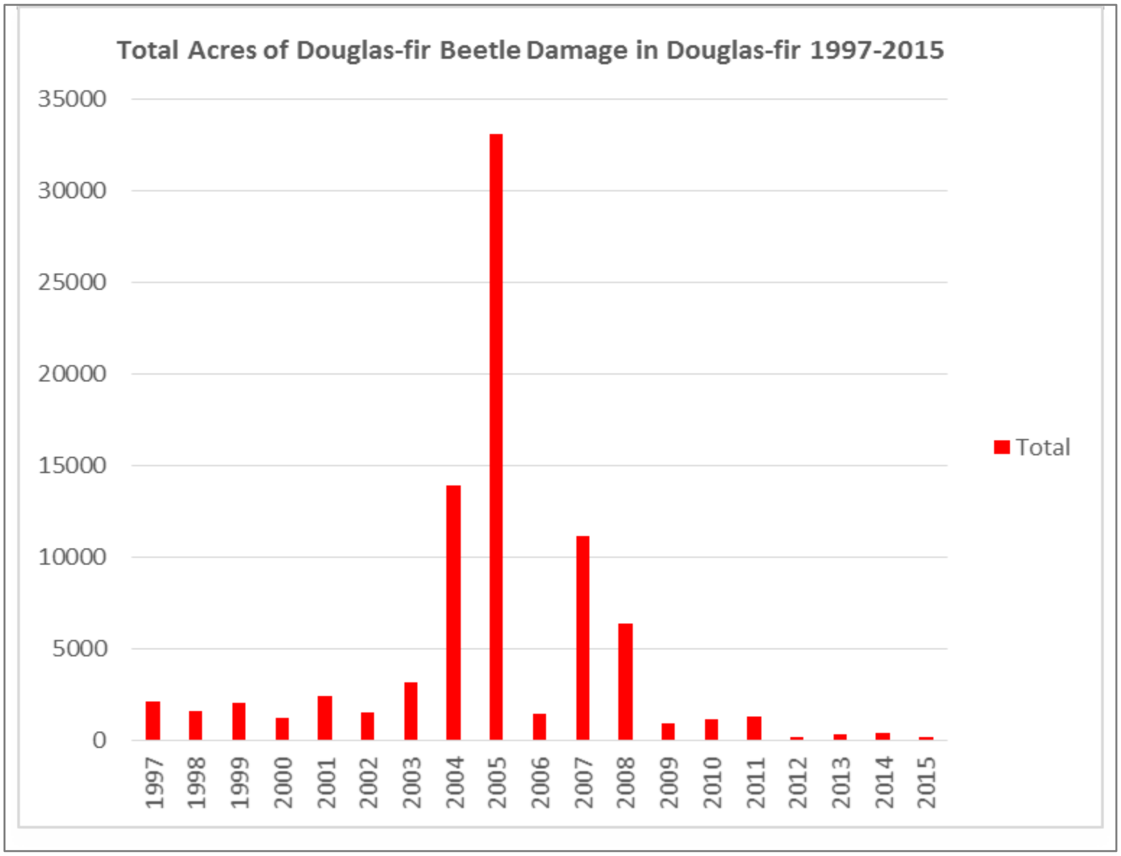


**Figure 1. Trend in Douglas-fir beetle-caused tree mortality, by year, from 1997 to 2015**

Figure 2 shows the area of Douglas-fir beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015 and the remaining locations of unaffected stands within the host type. Mapped acreages 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.

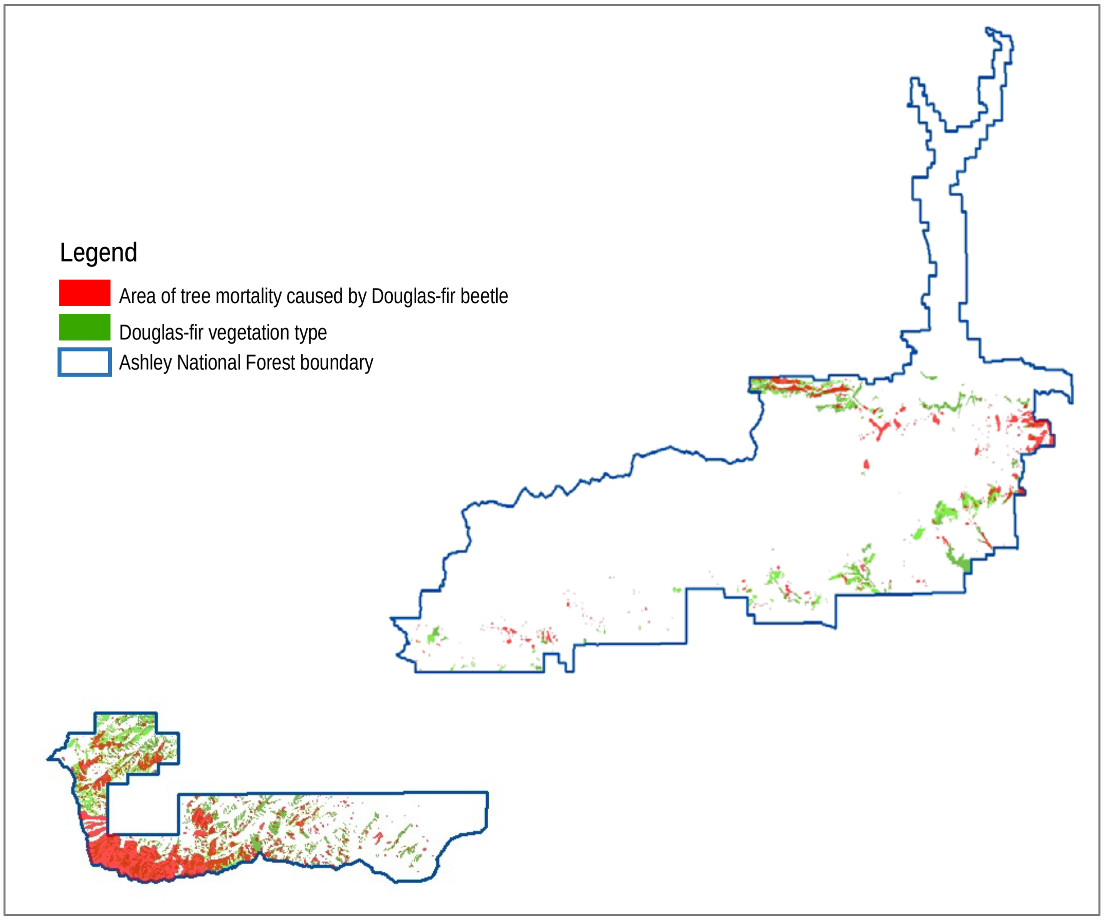


**Figure 2. Douglas-fir beetle-caused tree mortality (shown in red) on the Ashley National Forest from 1997 to 2015. Green areas are the Douglas-fir vegetation type. The South Unit is not necessarily at the same scale.**

Figure 3 shows the acreage of mountain pine beetle-caused lodgepole pine mortality by year since 1997. 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 1999. 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. 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 4 shows the area of mountain pine beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015 and the remaining locations of unaffected stands within the host type on the east and west Uintas of the Ashley. Not displayed in the map are approximately 4,700 acres typed as mixed conifer on the South Unit of the Ashley, of which approximately eight acres were reported as having mountain pine beetle-caused tree mortality

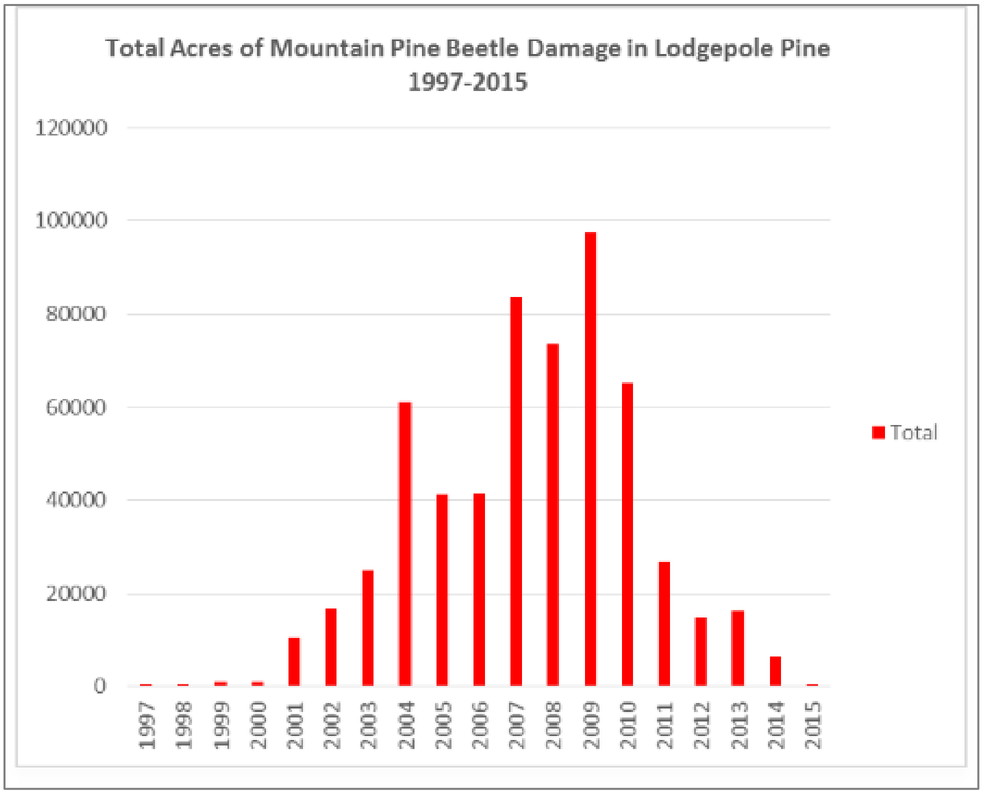


**Figure 3. Trend in mountain pine beetle-caused lodgepole pine mortality from 1997 to 2015**

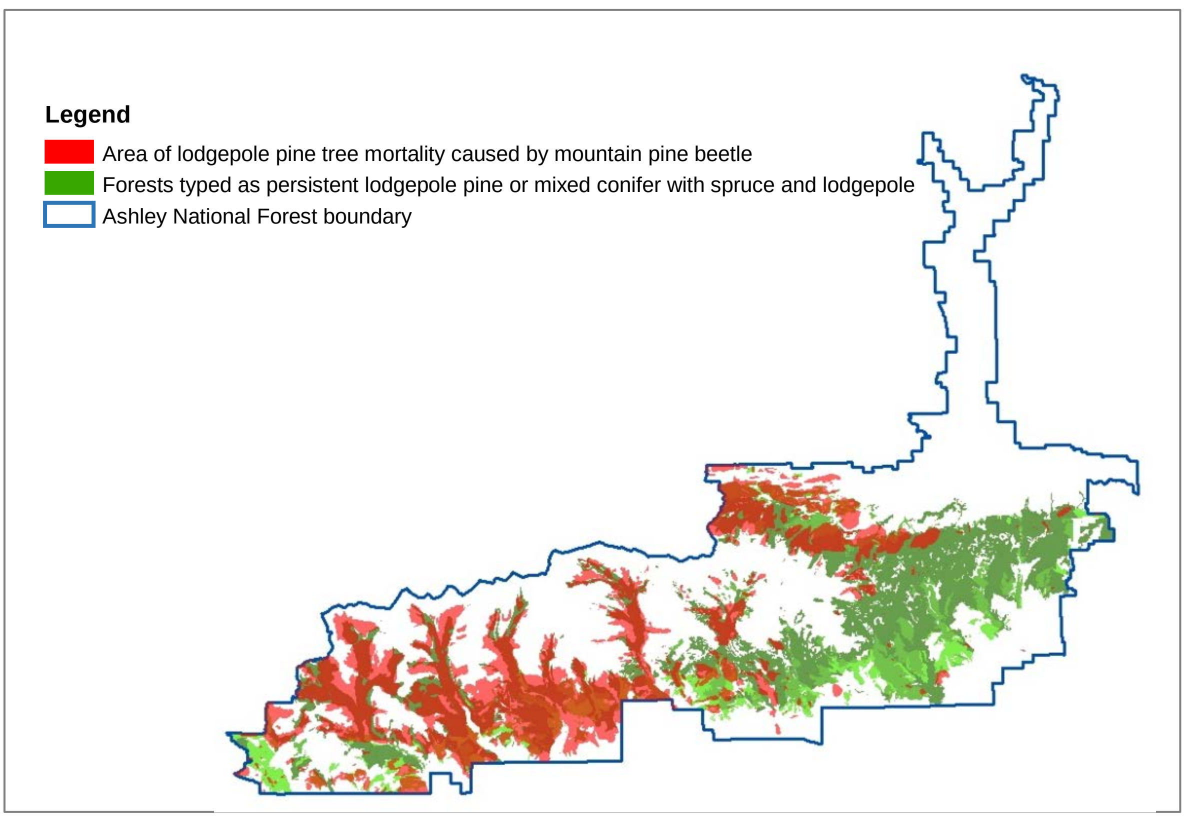


**Figure 4. Acres of mountain pine beetle-caused tree mortality in lodgepole pine on the Ashley National Forest from 1997 to 2015**

Figure 5 shows the acreage of mountain pine beetle-caused ponderosa pine mortality by year since 1997 (no mortality was detected in years 1997 and 1998). 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 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.

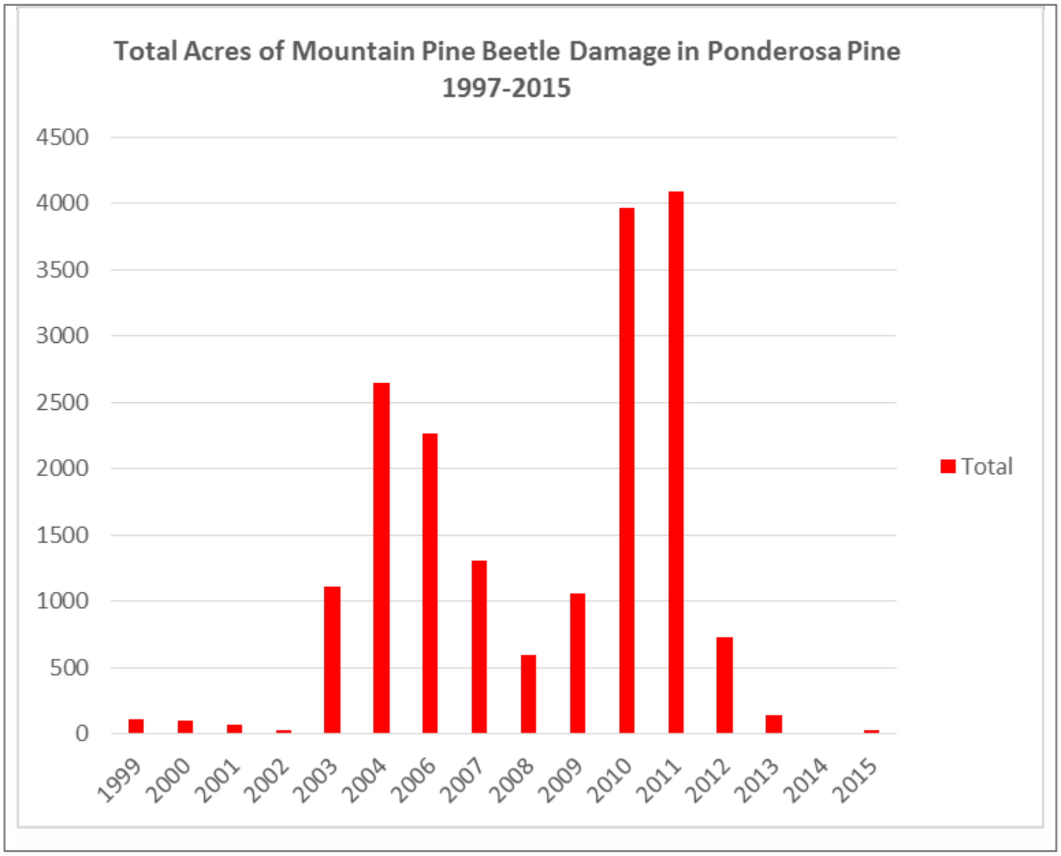


**Figure 5. Trend in mountain pine beetle-caused ponderosa pine mortality by year since 1997 (no mortality was detected in years 1997 and 1998)**

Figure 6 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 stands within the host type 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.

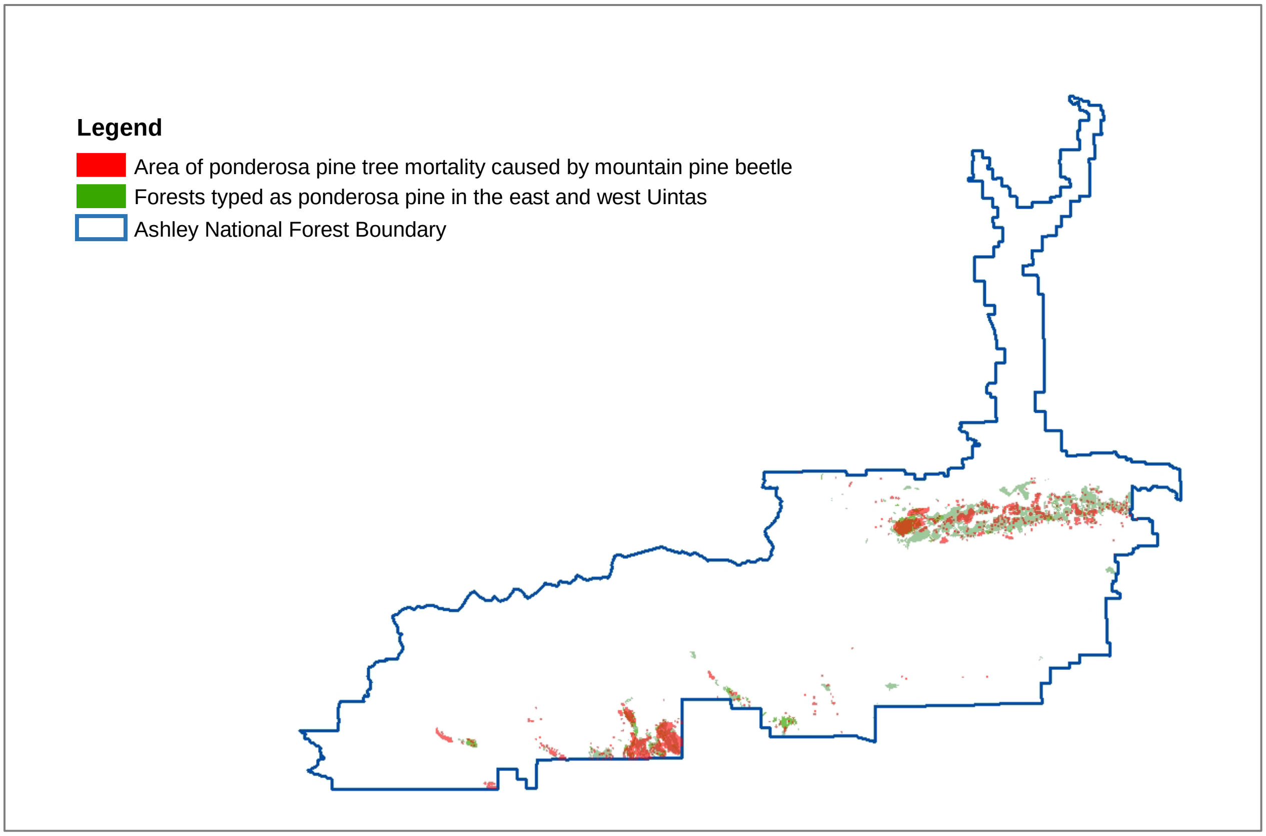


**Figure 6. Acres of mountain pine beetle-caused tree mortality in ponderosa pine on the Ashley National Forest from 1997 to 2015**

Figure 7 shows the acreage of 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.

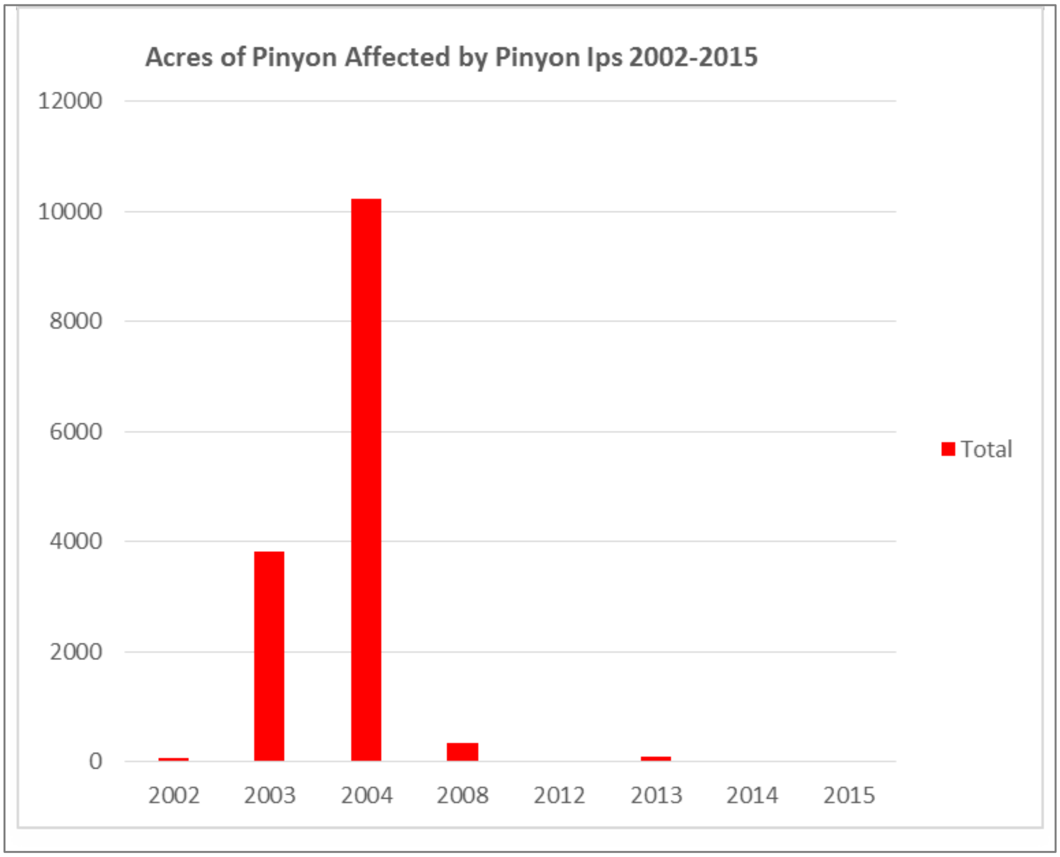


**Figure 7. Trend in piñon Ips beetle-caused piñon pine mortality by year since 1997 (no mortality was detected in years 1997 through 2001)**

Figure 8 shows the area of piñon *Ips* beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015 and the remaining locations of unaffected stands within the host type.

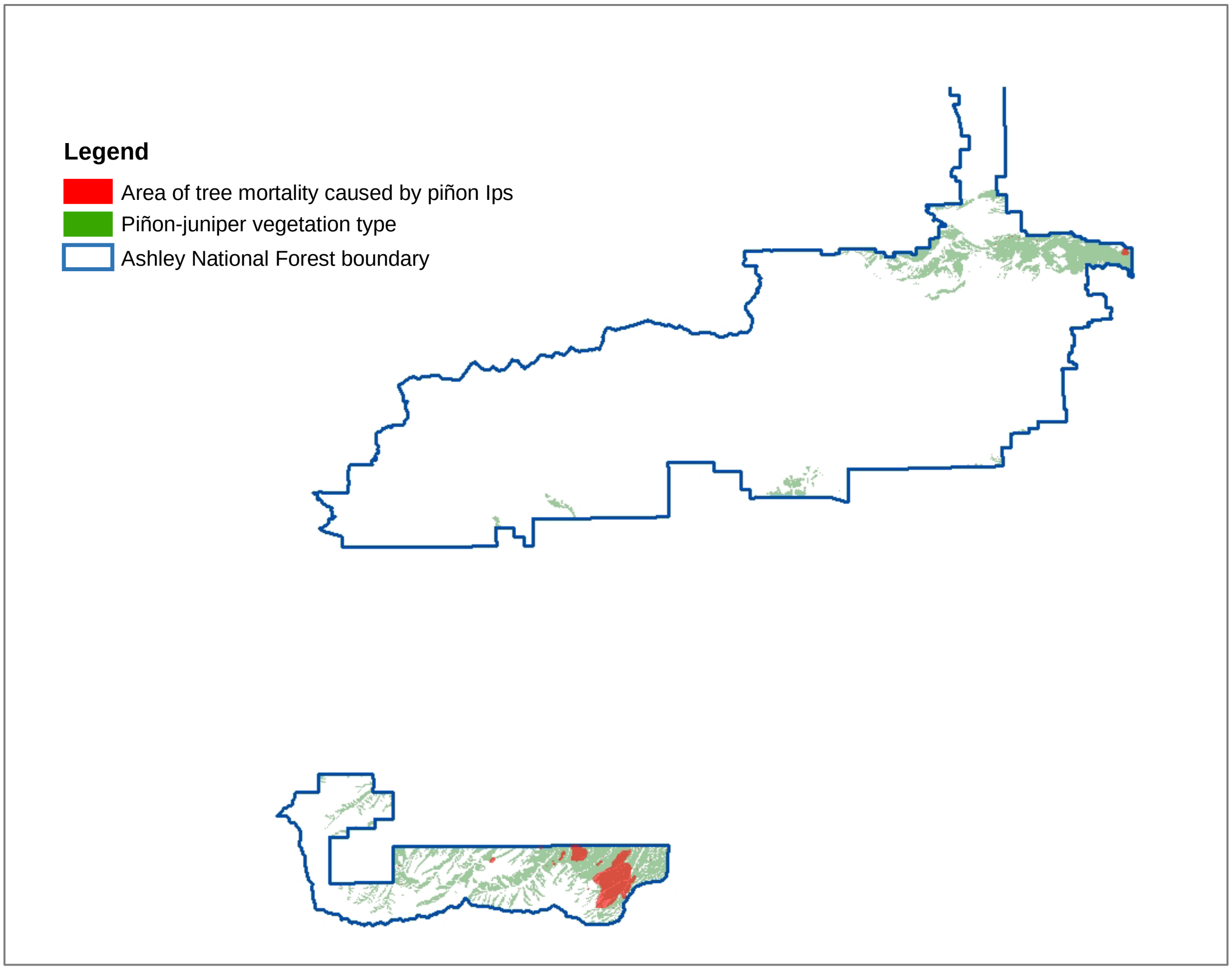


**Figure 8. Acres of piñon Ips beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015**

Figure 9 shows the acreage of dieback, defoliation, and decline by year since 1997 (no damage was detected in years 1997 through 2006). Increasing symptoms of aspen dieback and decline were recorded following a drought in 2001 to 2004. These symptoms peaked in 2008-2010. Subsequent symptoms and aspen mortality rates had returned to near pre-drought levels until 2015. It is unknown how much of the aerially mapped dieback and decline in 2015 was in fact defoliation caused by Marssonina, a condition associated with a moist period right after aspen leaf-out.

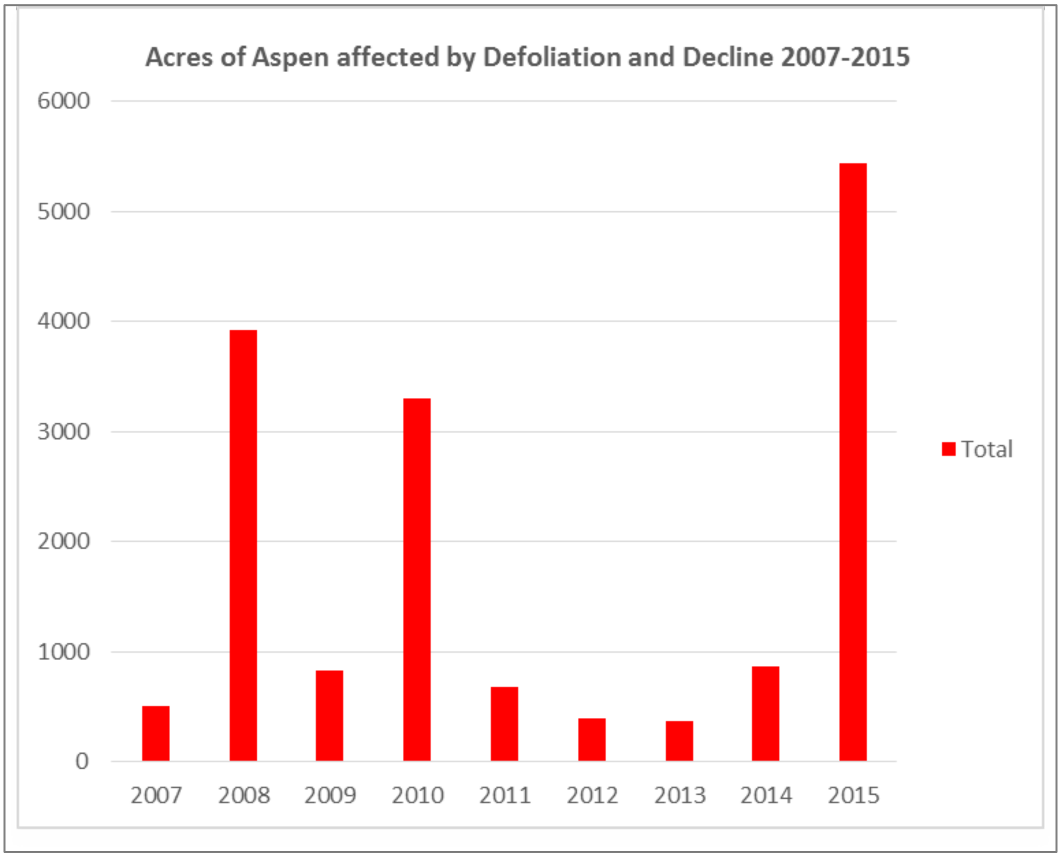


**Figure 9. Trend in aspen dieback, defoliation, and decline by year from 2007 to 2015. No damage was detected in years 1997 through 2006.**

Figure 10 shows the area of aspen dieback, defoliation, and decline on the Ashley National Forest from 1997 to 2015 and the remaining locations of unaffected stands within the host type.

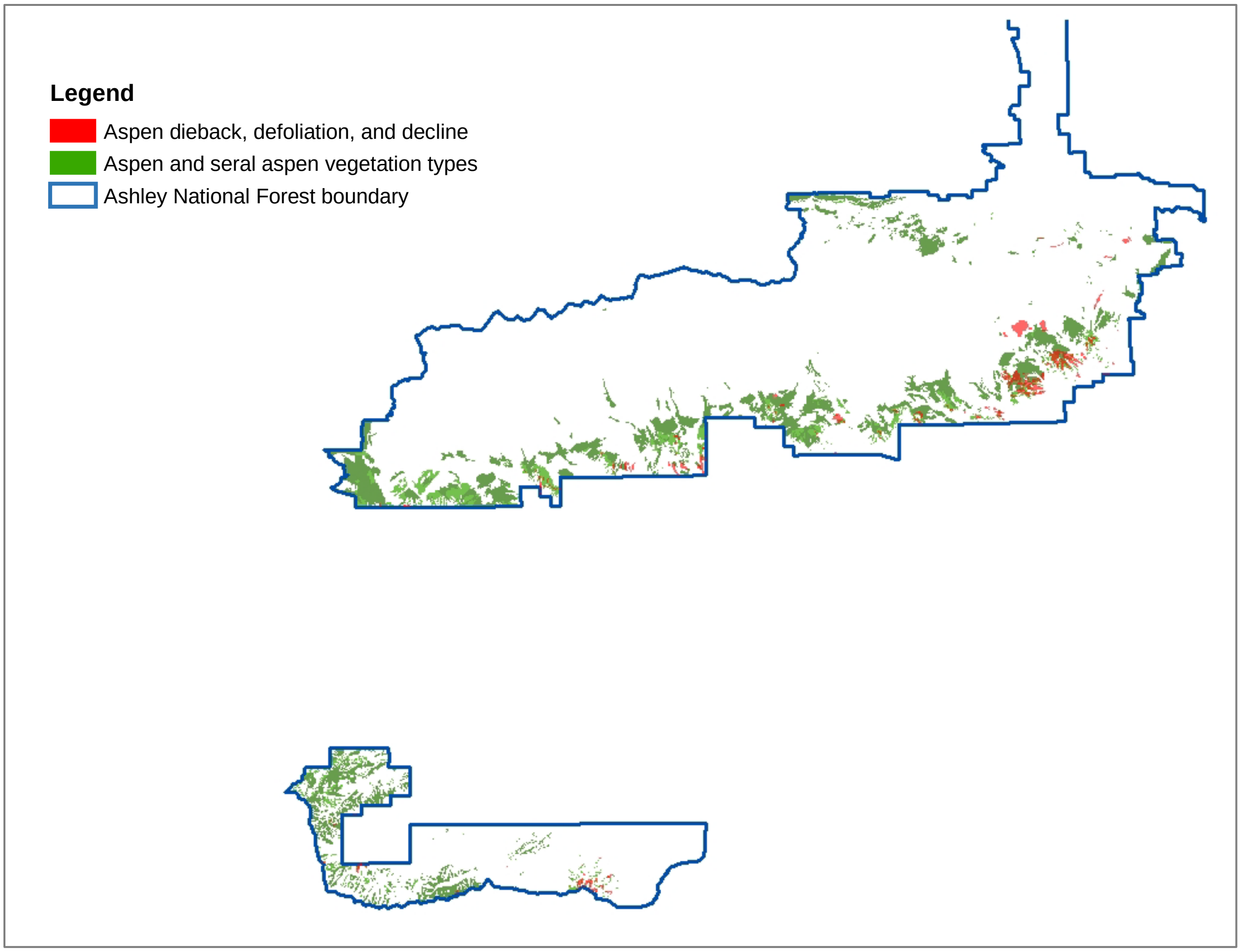


**Figure 10. Acres of aspen dieback, defoliation, and decline on the Ashley National Forest from 1997 to 2015**

Figure 11 shows the annual acreages of forest affected by spruce beetle through 2015. An on-going outbreak of spruce beetles are killing susceptible Engelmann spruce trees, causing extensive (up to 90% of host trees) mortality. 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 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.

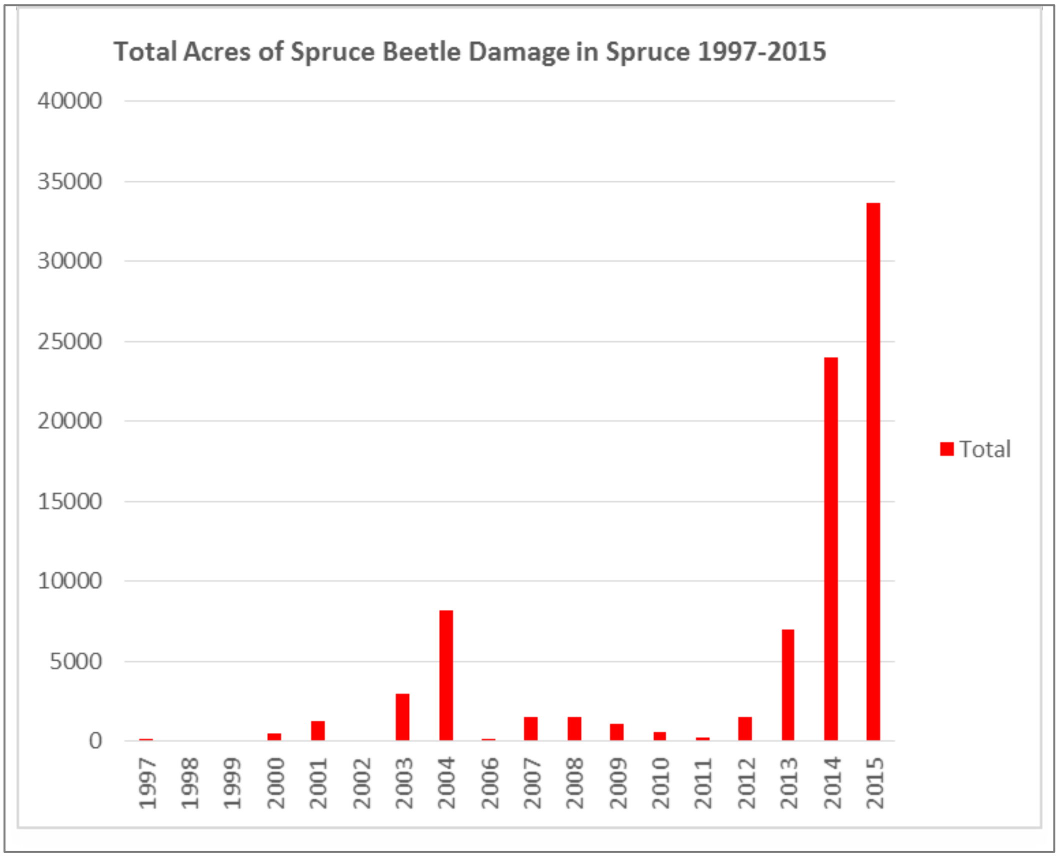


**Figure 11. Trend in acres of spruce forest affected by spruce beetle from 1997 to 2015**

Figure 12 shows the area of spruce beetle-caused tree mortality on the Ashley National Forest since 1997 and the remaining locations of unaffected stands within the host type.

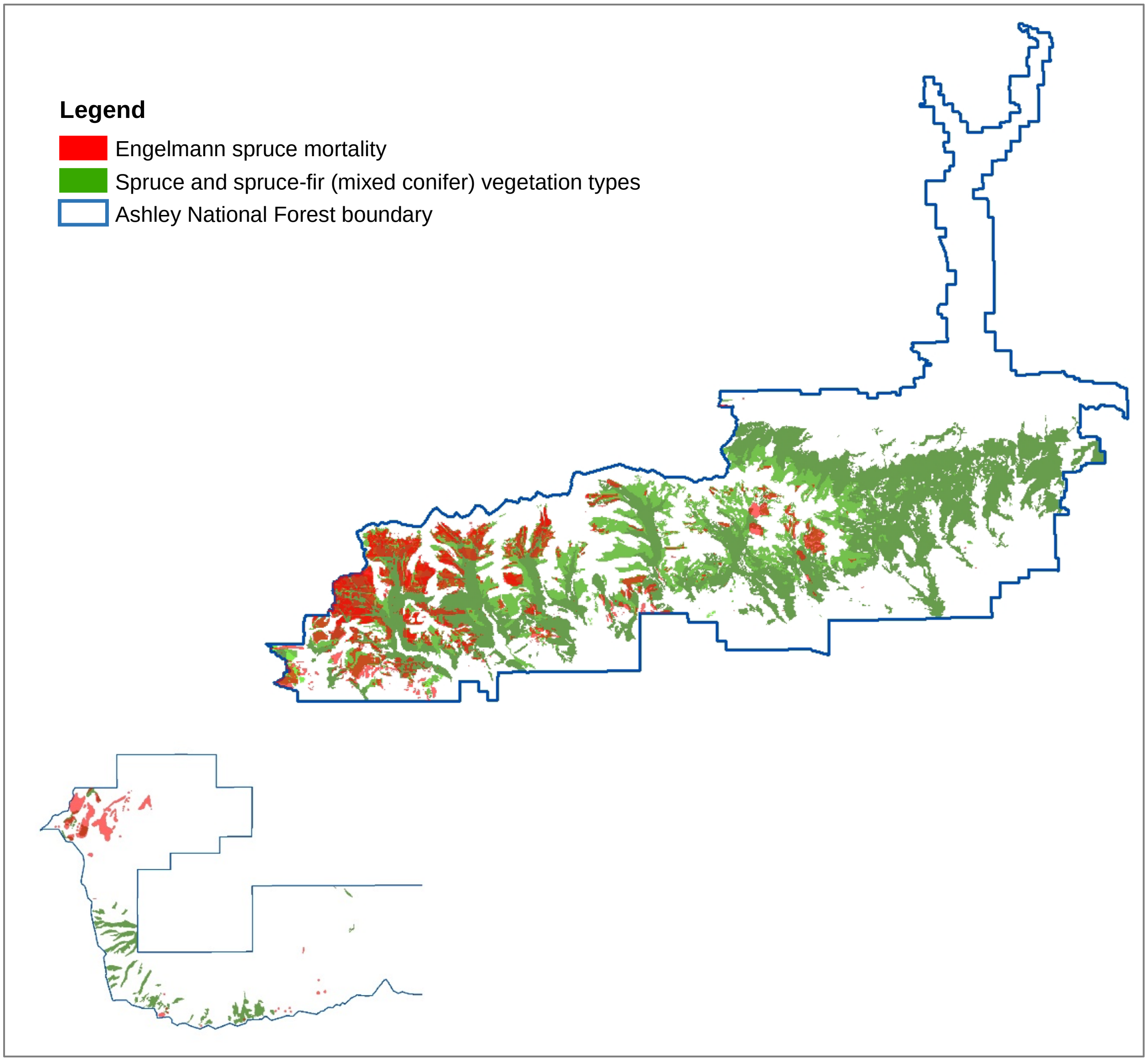


**Figure 12. Acres of spruce beetle-caused tree mortality on the Ashley National Forest from 1997 to 2015 (South Unit is not the same scale)**

# Resources Affected

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 alters 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).

# Management Tools

Known opportunities to reduce risk and adapt to system drivers, stressors, and the influence of climate change in regard to insects and diseases are listed in this segment. Although natural disturbance regimes remain a part of the landscapes we manage and will continue to impact these ecosystems, sound and ecologically appropriate management strategies can build resistance, resilience, or both to likely inevitable outbreaks (Fettig et al. 2014; DeRose 2014).

Prevention treatments (or indirect control) are generally associated with silvicultural[7] practices to modify stand and host type/vegetation conditions, so they are no longer favorable to insect and disease agents. Prevention treatments are most effective before insects or diseases reach unmanageable, or outbreak levels. If insect or disease frequency builds to damaging levels, suppression strategies (or direct control) may be implemented.

Thinning (density management) is the preferred strategy for bark beetle management in western forests (Fettig et al. 2007, Goyer et al. 1998). Thinning effectively reduces a particular host resource base that supports bark beetle population increase, reduces host competition for water and nutrients, and disrupts the effectiveness of insect pheromone communication. The resulting higher temperatures in thinned stands also reduce beetle survival and alter attack behavior of the insect (Schowalter et al. 1992, Amman et al. 1988, Schmid and Frye 1977, Sartwell and Stevens 1975).

Suppression or sanitation treatments (direct control) can reduce insect populations and levels of disease, particularly if treatments encompass the infested or affected area and are timed appropriately. Suppression treatments to protect high value trees or sites, including the use of insecticide or pheromones, are limited in scope due to costs, access and resource objectives. Environmental concerns and treatment costs often limit suppression alternatives to smaller affected sites. Vegetative treatments, such as sanitation, that do not modify stand conditions conducive to insects or disease, often result in short term benefits.

Restoration activities following outbreaks may mitigate vegetative change or loss and promote the growth of desired species.

## Douglas-fir Beetle

### Prevention (Indirect Control)

Commercial thinning can improve tree vigor and reduce susceptibility of attack by altering stand conditions (Kegley 2011). Stand characteristics contributing to high susceptibility of infestation include (Weatherby and Their 1993):

- moist habitats favoring Douglas fir growth;
- stocking greater than 250 square feet per acre;
- Douglas-fir species composition greater than 50 percent in the canopy;
- average stand age greater than 120 years; and
- average diameter of Douglas-fir greater than 14 inches.

[7] Silviculture is the art and science of controlling the establishment, growth, composition, and quality of forests and woodlands to meet the diverse needs and values of society on a sustainable basis (Helms 1998)

## Suppression (Direct Control)

Prompt salvage of wind thrown, fire damaged, or defoliated Douglas-fir and prompt removal or disposal of logs, large slash, and cull[8] material can eliminate potential brood sites. Felled trap trees and attractant pheromone baited trees are used to lure beetles into logs and trees that will be removed and processed. Anti-aggregation pheromones can be used to protect high-value trees and stands or prevent beetle colonization of wind thrown trees (Kegley 2011).

# Mountain Pine Beetle

## Lodgepole Pine

### Prevention (Indirect Control)

Density management can improve tree vigor and reduce susceptibility of attack by altering stand conditions (Gibson 2004, Anhold et al. 1996). Stand characteristics contributing to high susceptibility of infestation include:

- trees greater than eight inches in diameter at breast height;
- stand densities exceeding 120 square feet of basal area per acre; and
- relative density from 20 to 35 percent of the maximum stand density index (140 to 245 for lodgepole pine)

It is worth noting that prevention treatments should be used in accordance with insect activity. Host abundance, mountain pine beetle presence, proximity to mountain pine beetle-caused mortality, and population status should be considered before implementing prevention techniques. Mountain pine beetle in a lodgepole pine rating system (Gibson 2004, Shore and Safranyik 1992) has three distinct facets: susceptibility index, beetle pressure index, and a risk index. The beetle pressure and susceptibility indices are used to calculate risk (Gibson 2001).

### Suppression

Salvage and sanitation cutting can work to suppress losses associated with mountain pine beetle, but effectiveness is largely based on appropriate timing. Using a bio pesticide (synthetic anti-aggregation pheromone like verbenone or SPLAT® Verb) has successfully prevented mountain pine beetle attacks on lodgepole pine (Fettig et al. 2015). Insecticide treatments (application of a pyrethroid or carbaryl for lodgepole or ponderosa pine) as a preventative treatment has been successful as an individual tree treatment. This treatment has proven successful even in heavy pressure outbreak populations for high-value ponderosa and lodgepole pine trees.

An increase in insect populations can overwhelm some suppression techniques (such as pheromones). Where infested trees occur in an area that is treated using suppression, and salvage (removal) is not an option, successful methods for killing insects beneath the bark of an infested tree include: burning, submersing in water, or mechanically de-barking infested tree material.

[8] Any item of production, e.g., trees, logs, lumber, or seedlings, rejected because it does not meet certain specifications of usability or grade (Helms 1998)

### Restoration

This treatment often includes planting and sanitation in remaining stands. The treatment may also require chemical treatment of invasive plants.

## Ponderosa Pine

For ponderosa pine and its susceptibility to mountain pine beetle attack, tree diameter is not as significant as tree densities (Olsen et al. 1996). Ponderosa pine stands most susceptible to mountain pine beetle attack have (Stevens et al. 1980):

- single storied stands;
- trees larger than 12 inches in diameter at breast height; and
- stand densities exceeding 150 square feet of basal area per acre.

### Prevention (Indirect Control)

Thinning stands of ponderosa pine (80 feet$^2$/acre) will prevent or minimize beetle-caused mortality (Schmid et al. 1994). Patch cutting in ponderosa pine stands creates a mosaic of age and size classes, which reduces the acreage that will be highly susceptible to beetles at one time (Amman et al. 1990, Amman and Logan 1998).

Prescribed burning that maintains ponderosa pine's historical nonlethal fire regime will also help keep densities in check. Fire periodically thins the understory when the fire return interval is kept to about five to 45 years, consistent with the species fire regime (USDA 2009). The effects of fire may also 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 attack. Many studies have found [significantly] increased resin production after burning" (Hood 2010).

### Suppression (Direct Control)

Under certain population conditions, bio pesticides (synthetic anti-aggregation pheromones) like verbenone have been successful in preventing mountain pine beetle from attacking single trees. Insecticide treatments (application of a pyrethroid or carbaryl) before trees become infested can also protect individual high-value ponderosa pine trees.

### Sanitation (Direct Control)

As with lodgepole pine, salvage and sanitation cutting can work to suppress losses associated with mountain pine beetles. Resulting slash must be property disposed of, (piled and burned or lopped and scattered) to prevent population buildups of pine engraver beetles (*Ips* spp.) (Kegley 1997).

# Piñon *Ips*

Drought predisposes mature piñon pine trees to piñon Ips attacks. The mechanisms of drought impacts on insect resistance of piñon pine trees are variable (Gaylord et al. 2013). Piñon *Ips*-caused tree mortality has been correlated to stand density and dwarf mistletoe, *Arceuthobium divaricatum* (Engelmann) infection, among other factors. Studies have shown that piñon Ips-killed trees are larger in diameter, and have higher mistletoe infection levels and smaller crown ratios. Piñon pine stand density index can be a good predictor of the likelihood of infestation at the plot level (Negrón and Wilson 2003, Fettig et al. 2007, Page 2008). Based on research by Negron and Wilson (2003), stands dominated by piñon pine, that are susceptible to piñon *Ips* attack, have stand density indices greater than 5.6 percent of maximum. This is particularly true when high levels of beetle activity are present in the general area. Those stands with a higher ratio of piñon-to-juniper also tend to be more susceptible to *Ips*-caused mortality (Negrón and Wilson 2003, Page 2008).

Scheduling potentially disruptive activities, such as thinning and sanitation, in piñon-juniper stands during the cooler winter months can reduce the risk of tree mortality. Activities should be as nondisruptive as possible. But working in cooler weather (roughly late November to early March), while the insects are inactive, allows managers time to sanitize stands (Eager 2008).

### Prevention (Indirect Control)

Thinning piñon-juniper woodlands appears to have a significant effect on the understory vegetation and the subsequent use of the woodlands by wildlife. In general, heavy thinning (basal density of 30 square feet per acre) results in a more abundant and diverse understory, especially in areas with little or no understory prior to thinning (Albert et al. 2004, Page 2008).

### Suppression and Sanitation (Direct Control)

Burning slash destroys potential brood sites. However, avoid scorching standing trees as this makes them more attractive to numerous species of bark and wood-boring insects. During sanitation treatments to remove infested trees, lop residual slash material greater than three inches into smaller pieces (less than 2 feet) and scatter it into openings or pile and burn slash (Eager 2008). Preventative insecticide treatments are available to protect high-value individual trees (Fettig et al. 2013).

## Aspen Dieback and Decline

Aspen management should emphasize identifying the functional types present and use this information to select the appropriate management strategy (Rogers and others 2014, see Table 3). Aspen ecology and disturbance patterns are complex and the functional aspen type should drive decisions about aspen status and management. For example, stable (nearly pure) aspen is much less conducive to wildfire or prescribed burning. Regeneration in these forests can be dependent on gaps created by continuous, low-level tree mortality.

The aspen functional types on the Ashley National Forests include:

1. seral and montane;
2. stable and Colorado Plateau;
3. stable and elevation aspect limited; and
4. stable and terrain isolated.

The most common types on the Ashley National Forest are seral montane and Colorado Plateau types. Seral montane systems should emphasize regeneration efforts that promote a healthy forest by restoring disturbance, particularly if past management has favored conifers. Colorado Plateau types should focus on maintaining or restoring structural diversity. Both types are sensitive to grazing pressure and may experience increased insect and disease damage under drought stress. Aspen management, particularly regeneration treatments, should involve monitoring and controlling browsing pressure before and after treatment (Rogers and others 2014).

## Spruce Beetle

Factors that instigate spruce beetle outbreaks include extended periods of seasonally warm temperatures, combined with drought; an abundance of fresh, downed host material; or both (DeRose et al. 2012).

Susceptible stand conditions must exist, however, to support population increases and spread. Stand conditions most conducive for population expansion include (Schmid and Frye 1977):

- basal area greater than 150 square feet per acre;
- average stand diameter of live spruce greater than 16 inches in diameter at breast height; and
- greater than 65 percent spruce in the canopy.

Hansen et al. 2010, however, suggests this assessment could be improved by considering stand density index[9] – by control of growing stock through initial spacing or subsequent thinning to meet specific management objectives, or spruce basal area rather than total basal area and the density of spruce stems >11 inches or more in diameter at breast height rather than the average diameter of spruce greater than 16 inches in diameter at breast height. Additionally, larger landscapes of susceptible host type are more conducive to outbreaks. Stand disturbance, such as windthrow and fire, contribute to population growth as the affected hosts provide an ideal habitat for population increase (Jenkins et al. 2014).

## Prevention (Indirect Control)

To reduce the long-term susceptibility of uninfested stands to spruce beetle, Alexander (1986), based on studies compiled by Schmid and Frye (1977), recommended density management (Jenkins and others 2014, Hansen and others 2010). Density management includes: partial cutting to remove the larger, over mature spruce leaving spruce diameters less than 10 inches in diameter at breast height, decreasing mean stand basal areas to 80 square feet per acre, and reducing the spruce component to 65 percent or less (Schmid and Frye 1977).

Prevention (indirect control) treatments that result in a multi-storied (variable age class) stand of spruce will mitigate the effects of a spruce beetle outbreak. Uneven-aged prescriptions for spruce stands will not prevent losses caused by the spruce beetle but will mitigate the effects of an outbreak in a treated area. An uneven-aged prescription may provide an average diameter at breast height of 19 centimeters (7.5 inches), a stand density index of 35 percent of maximum, and a basal area of 30 square meters per hectare (130 square feet per acre). Shaw describes the method most commonly used to derive a stand density index, and shows that the variables used to derive stand density index in the case of uneven-aged stands must be carefully selected. The summation method should be used to calculate stand density index if the management objectives are uneven-aged or irregular stand structures.

## Suppression (Direct Control)

Preventive spraying before trees become infested can protect individual high-value trees or vegetation in high value sites (recreation areas). Sanitation treatments, combined with trap trees and the use of baited funnel traps, can be effective when timed appropriately and with thorough follow up treatments such as removal, burning, de-barking, or chipping, to destroy developing brood (Bentz and Munson 2000). However, there are limitations associated with the spatial scale of treatments designed to suppress a local outbreak. The treatments must encompass the infested area to be effective.

Sanitation treatments to remove infested trees within outbreak areas may be effective depending on stand structure. Multi-storied stands composed of various size and age class spruce are a candidate for this type of suppression strategy. Removing the larger diameter hosts (more than 10 inches in diameter at breast

[9] Stand density index is a widely used measure that expresses relative stand density in terms of the relationship of a number of trees to stand quadratic mean diameter; or, any index that expresses relative stand density based on a comparison of measured stand values with some standard condition (Helms 1998). Stand density can be expressed as basal area (the cross-sectional area of a single stem) per unit of land area.

height), and leaving the smaller diameter trees on site, will promote survival of the residual smaller diameter class trees. Sanitation treatments to reduce spruce basal area during an outbreak, in stands composed of even-aged larger diameter hosts, will not mitigate the effects of an outbreak population of this insect. However, salvage and sanitation operations can offer site preparation benefits for restoration (planting) post-outbreak.

## Dwarf Mistletoes

The commonly cited characteristics of dwarf mistletoes that facilitate management include (Hawksworth and Wiens 1996):

- Dwarf mistletoes are obligate parasites. Without the presence of a host plant, the mistletoe dies.
- Dwarf mistletoes are generally host specific, or nearly so. Immune or lightly susceptible species can be used in management in several ways. These ways include screening of spread, buffers, or as replacements for seriously infected stands.
- The life cycle of dwarf mistletoes is long compared to other forest pests. This means that amount of dwarf mistletoe infection increases slowly, and intensification within a tree is also slow, about one dwarf mistletoe rating class per decade in lodgepole pine.
- Spread is spatially limited. Most dwarf mistletoe seeds fall within 60 feet from a tall single tree. In most cases, due to spacing and screening, spread is even less. In even aged stands, spread is only 1 to 2 square feet per year due to screening by existing trees.
- Infection is easy to detect, and quantification is fairly standardized, using a nearly universally accepted rating system (Hawksworth 1977).

One of the principle goals of all dwarf mistletoe management should always be protecting young, susceptible trees from an infected overstory. Seedlings and saplings are severely damaged by infection of only few dwarf mistletoe plants; 50 percent mortality rates and severe growth loss can be expected if trees are infected young (Geils et al. 2002). Therefore, a primary goal should be creating space between regeneration (and areas planned for regeneration) and an infected overstory. This space can be accomplished by creating buffers of space, planting non-host trees, or using existing space created by roads, natural openings or other barriers. This is applicable to both overstory removal and clear felling treatments.

Removal of all infected trees before planting or regeneration treatments is advisable (Hawksworth and Johnson 1989). Clear felling and overstory removal treatments are commonly prescribed for dwarf mistletoe management. If used, long narrow treatment units, as well as units with a high level of interface with surrounding infected stands, (Hawksworth and Johnson 1989) should be avoided. The proportion of a treated area affected by mistletoe from the border decreases as the treated area becomes larger. To minimize the amount of dwarf mistletoe re-invasion of a stand, 20 acres is considered a minimum and 40 acres is recommended. As patches increase beyond 40 acres, the advantage of increasing size becomes less. Irregularly shaped or long, narrow patches must be larger to have a similar area protected.

Seed tree and shelterwood treatments are not generally prescribed in areas heavily infected with dwarf mistletoe. However, if used, mistletoe-free or lightly infected trees should be selected for retention, as well as using non-host trees whenever possible (Hawksworth and Johnson 1989). In using the seed-tree method (cutting of all trees except for a few desirable individuals that provide seed for the next generation) or shelterwood method (cutting of all trees in a series of two or more over a relatively short period of time, establishing even-aged reproduction under the partial shelter of the previous generation), residual overstory trees must be removed before the subsequent regeneration is out of its grace period.

The grace period refers to the amount of time that seedlings can generally be considered safe from infection. Because of their small size as targets and their short exposure to inoculum, this grace period lasts until the seedlings are about 10 years old or three feet tall, whichever comes first. Infection of smaller or younger trees does occur, but it is generally rare. This grace period gives some time before infected overstories must be treated after regeneration is established.

Sanitation, the removal or killing of infected trees to protect other trees, is important in many kinds of stands at various developmental stages. “Sanitation cutting” (or simply sanitation) has been distinguished from “sanitation thinning.” Sanitation cutting is the attempted removal of all visibly infected trees, though it usually is also combined with thinning goals. In sanitation thinning, the emphasis is on spacing, and only the most severely diseased trees may be removed. Because of the long period between infection and appearance of symptoms (latency), and difficulty of locating every visibly infected tree, multiple entries (five to 10 years apart) are usually necessary to sanitize a stand or minimize disease severity.

Thinning is usually not recommended for stands with average dwarf mistletoe rating of three or higher (Hawksworth 1977). Thinning in dwarf mistletoe stands is always problematic, and complete sanitation of dwarf mistletoe infection is rarely possible. Thinning within infested areas tends to stimulate the remaining mistletoe (Conklin and Fairweather 2010), but thinning still has several other benefits and can reduce dwarf mistletoe incidence if properly conducted.

Pruning of large brooms, which are generally in the lower crown, can allow trees to recover vigor and substantially prolong their life. It is most often used in developed recreation sites. Pruning can also be used for sanitation (sanitation pruning). In this case the removals are intended to reduce the population and impact of dwarf mistletoes, as well as future inoculum.

Prescribed fire may be applicable for dwarf mistletoe management in some settings, because infection centers (i.e. stand areas concentrated with infection) tend to torch and dwarf mistletoe on lower branches (those areas on the tree most likely to be infected) is often killed or pruned by scorch (Conklin and Geils 2008). A managed fire, however, often seeks to create a lower intensity mosaic pattern that is easier to control. Such fire may increase the prevalence of dwarf mistletoes by creating interfaces between infected residual stands and post-fire regeneration.

# Stressor Accumulation

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 (number of eggs in the first stage of the life cycle produced by an individual) and dispersal. Large and consistent decreases in snowpack have been observed throughout the western United States between 1955 and 2015 (EPA 2015). Because insect species, in general, have relatively short life cycles, high reproductive capacity, and high degree of mobility, 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).

# Identify Any Data Gaps

Data provided by aerial detection surveys 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.

Aerial insect and disease detection surveys 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, such as mountain pine beetle, spruce beetle, and Douglas-fir beetle, were not available in time to place into this report. 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 Ashley forest levels to help inform forest staff on the Ashley.

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