Appendix C: Plants

Conifers and Cycads

SPECIES ACCOUNT: *Cupressus goveniana ssp. goveniana* (Gowen cypress)

Species Taxonomic and Listing Information

Listing Status: Threatened; Pacific Southwest (R8) (USFWS, 2015)

Physical Description

A small, sparsely branched coniferous tree, usually 5-7 m tall and 2-4 m wide at the crown. Foliage is scale-like, pale green to yellow-green in color. Seed cones are round, 1-1.5 cm long, woody; they remain closed for many years, typically opening and releasing seeds during fires (NatureServe, 2015).

Taxonomy

In the second edition of the Jepson Manual (Baldwin et al. 2012) what Kartesz (1994 and 1999) treated as Cupressus goveniana ssp. goveniana and C. goveniana ssp. pygmaea are treated as distinct species in the genus Hesperocyparis (NatureServe, 2015).

Historical Range

The historical extent of the two extant stands is not known (USFWS 2008) (NatureServe, 2015).

Current Range

Restricted to two sites approximately 6.4 km apart on the Monterey Peninsula, coastal Monterey County, California. Suitable habitat for this species is very limited in extent (USFWS 2008) (NatureServe, 2015).

Critical Habitat Designated

No;

Life History

Food/Nutrient Resources

Reproductive Strategy

Adult: Sexual (inferred from NatureServe, 2015)

Lifespan

Adult: 85 - 127 years (USFWS, 2012)

Key Resources Needed for Breeding

Adult: Frequent fires, wind (NatureServe, 2015); open canopy (USFWS, 2008); bare mideral soils (USFWS, 2012).

Reproduction Narrative

Adult: Has adaptations typically associated with frequent fires: Female cones are serotinous (only opening to release seeds when exposed to very high heat or fire) and may be produced on a tree as young as four years. Mass synchronized openings of the serotinous cones during fires are likely a critical part of its natural regeneration process (USFWS 2008). Pollination occurs via wind. Recruitment rates are very low in maritime chaparral habitat due to the high vegetation density; this (sub)species will not germinate in shade (USFWS 2008). (NatureServe, 2015). Hesperocyparis goveniana is a long-lived species (85-127 years). H. goveniana also needs light and bare mineral soils for seedling establishment. Recruitment rates for the species are extremely low in maritime chaparral habitat due to the high density of vegetation and the intolerance of this species to germinate in a shaded environment (Doak et al. 2000). Recruitment is higher, but still low, in mixed conifer habitat (Doak et al. 2000) (USFWS, 2012).

Habitat Type

Adult: Terrestrial (NatureServe, 2015)

Habitat Vegetation or Surface Water Classification

Adult: Coniferous forest, pygmy forest, maritime chaparral (NatureServe, 2015)

Dependencies on Specific Environmental Elements

Adult: Natural fire regime (see reproduction narrative)

Geographic or Habitat Restraints or Barriers

Adult: Occurs at 30 - 300 m elevation; restircted to Monterey coast (NatureServe, 2015)

Environmental Specificity

Adult: Very narrow (Natureserve, 2015)

Tolerance Ranges/Thresholds

Adult: Mderate (inferred from USFWS, 2012)

Habitat Narrative

Adult: Closed-cone coniferous forest, pygmy forest, and maritime chaparral habitats; may occur in pure stands or in mixed stands with Monterey pines or Bishop pines (Pinus muricata). An understory of chaparral shrubs is often present. Apparently restricted to shallow Cieneba or podzolic soil types with severely reduced nutrient availability, which precludes establishment by other trees. Occurs at 30 - 300 m elevation. The nevironmental secificity is very narrow; this (sub)species' adaptation to a very specific soil type essentially restricts it to its current distribution on the Monterey coast (USFWS 2008) (NatureServe, 2015). While H. goveniana can grow in a variety of habitats with minor disturbances, it apparently requires mineral soil surfaces and unshaded conditions for successful recruitment (Doak et al. 2000) (USFWS, 2012).

Dispersal/Migration

Dispersal

Adult: Low (USFWS, 2012)

Dispersal/Migration Narrative

Adult: Natural seed dispersal occurs during September and October, although seeds are not light enough to be carried far from the parent plant (Sudworth 1967) (USFWS, 2012). Seeds are dispersed upon mechanical removal from the tree, death of the tree or supporting branch, when heat from fire breaks the cones' resinous seal and allows seeds to escape, or during hot, dry weather (USFWS, 2004).

Population Information and Trends

Population Trends:
Not available

Species Trends:
Stable (NatureServe, 2015)

Resiliency:
Low (inferred from NatureServe, 2015)

Redundancy:
Very low (inferred from NatureServe, 2015)

Number of Populations:
2 (NatureServe, 2015)

Adaptability:
Low (inferred from NatureServe, 2015)

Population Narrative:
In the absence of fire, lack of recruitment becomes a major issue for this (sub)species. During the last glacial era, it is believed that the temperate coastal climate was more favorable for conifers; increasing aridity since then probably contributes to the restriction of this (sub)species to a few coastal sites. The U.S. Fish and Wildlife Service counts only 2 occurrences, one at the Del Monte Forest within the Morse Botanical Reserve, and one at Point Lobos State Reserve (USFWS 2002). The California Natural Diversity Database (2008) reports a third occurrence of less than 10 plants on private land < 1 km south of the Del Monte Forest; it appears that USFWS considers these plants part of the Del Monte Forest stand. Estimated stable by USFWS in April 2003 (NatureServe, 2015).

Threats and Stressors

Stressor: Development (USFWS, 2013)
Exposure:
Response:
Consequence:
Narrative: The species was listed as threatened in 1998 primarily due to habitat loss, fragmentation, and secondary impacts from development of privately owned land contiguous with what is now known as the Huckleberry Hill stand. This threat continues, as proposals for additional development of residential neighborhoods and resort areas are currently under review by the County of Monterey, and will result in the increase of urbanization adjacent to the Del Monte Forest population. Current development proposals will likely continue to encroach on habitat that is already surrounded by urbanized areas. Similar encroachment can be seen around the Point Lobos population, where properties immediately surrounding the Ranch continue to be

developed (K. Barry, pers. obs. 2011b). The proximity of permanent structures and human habitation is likely to impede future management of both populations. With no room for the Del Monte Forest population to expand because of planned development, increasing the size of both populations, as called for in the Recovery Plan, will remain a challenge to the land managers. Existing development allowed on the immediate edge of occupied habitat, such as that along the upper ridge of Huckleberry Hill, constrain active management techniques (such as prescribed burning) that are likely to be necessary in the future (USFWS, 2013).

Stressor: Erosion (USFWS, 2013)
Exposure:
Response:
Consequence:
Narrative: In 2007, observations at the Del Monte Forest stand indicated that activity from hikers and mountain bikers was causing considerable and damaging erosion (C. West, pers. obs. 2007). The Pebble Beach Company acknowledged that they did not have the personnel resources necessary to patrol or enforce trespassing and mountain biking restrictions on these portions of its property (E. Love, pers. comm. 2007). Trails were cut through the surrounding area and even entered the stand itself. Mountain bike jumps and bridges made from cut tree trunks from within the stand were also visible throughout the area (C. West, pers. obs. 2007). Remnants of old, cleared ranch roads total 5.9 percent of the Point Lobos stand (T. Moss in litt. 2011). Removal of anchoring vegetation and poor drainage in these areas have allowed the topsoil to be completely washed away in much of the road system and caused severe erosion in the surrounding habitat (Figure 1) (T. Moss in litt. 2011; C. West, pers. obs. 2007). Continued flushing with surface water during rains prevents soil accumulation and vegetation establishment. Water seeping from the soil layer upslope of one of these exposed areas, or water falling as rain onto an exposed area, is not slowed by soil or vegetation, and therefore accumulates and behaves like a stream. Upon reaching natural drainages, this swift-moving water cuts deep ravines, further removing the limited topsoil and eroding habitat (T. Moss in litt. 2011; K. Barry, pers. obs. 2011a; C. West, pers. obs. 2007) (USFWS, 2013).

Stressor: Invasive species (USFWS, 2013)
Exposure:
Response:
Consequence:
Narrative: At the time of listing, displacement by invasive species was considered a key threat. While this threat remains, the Pebble Beach Company regularly works to control problem species in the Hesperocyparis goveniana stand (E. Love, pers. comm. 2011). In 2011, no pampas grass was observed at either population of H. goveniana, and while considerable numbers of French Broom were present at both occurrences, they were all small, recently-recruited individuals that germinated between removal projects (K. Barry, pers. obs. 2011a). These control efforts are critical to the persistence of H. goveniana, and will need to be continued for the foreseeable future (USFWS, 2013).

Stressor: Disruption of natural fire cycles (USFWS, 2013)
Exposure:
Response:
Consequence:

Narrative: The disruption of natural fire cycles was considered a primary threat at the time of listing and remains a major and continuing threat to this species (V. Yadon in litt. 2002; Jones and Stokes Associates 1996). As succession continues and native chaparral along with nonnative species fill in as understory, the availability of bare soil exposed to direct sun, which this taxon requires for establishment, will be reduced. Fire is necessary to clear the understory and litter layers and remains the best management tool for maintaining Hesperocyparis goveniana habitat (Jones and Stokes Associates 1996). As mentioned previously, development continues to encroach and surround H. goveniana habitat, and the ability to safely and effectively use fire as a management tool has been greatly reduced. Jones and Stokes Associates (1996) found that existing development surrounding the Del Monte Forest likely precludes the use of fire as a management tool. At the time of listing, accumulation of understory litter leading to lower recruitment was not considered a major threat to this species. Currently, recruitment seems to be occurring with frequency only at the Del Monte Forest population, where two fires in the past 60 years (1959, 1987) have cleared ground litter, understory, and canopy in portions of the stand allowing for regeneration of the species (K. Barry, pers. obs. 2011a). This threat will likely continue until the natural disturbance cycles can be restored or until adequate surrogates for natural disturbance factors are identified and implemented (USFWS, 2013).

Stressor: Climate change (USFWS, 2013)
Exposure:
Response:
Consequence:
Narrative: Current climate change predictions for terrestrial areas in the northern hemisphere indicate warmer air temperatures, more intense precipitation events, and increased summer continental drying (Field et al. 1999, Cayan et al. 2005, Intergovernmental Panel on Climate Change (IPCC) 2007). Hesperocyparis goveniana's small and isolated range increases its vulnerability to random fluctuations in annual weather patterns and environmental disturbances such as can be brought about by climate change (USFWS, 2013).

Recovery

Reclassification Criteria:
Not available

Delisting Criteria:
1. Monitoring of the Del Monte Forest population and the Point Lobos population for a minimum of 10 years (or longer if needed) shows long-term reproductive success in both populations. As determined by research, protected habitat must be of adequate size (large enough to support a functioning ecosystem, including areas that support suitable unoccupied habitat for population expansion and fluctuations in distribution) to ensure that ecosystem and community processes and associated species (e.g., hydrologic regime, fire, food webs, fauna, Monterey pine forest communities) are maintained, and that the locations are adequate to provide for population expansion and for colonization of new areas as microhabitat conditions change (USFWS, 2013).

2. Twelve or more years (or possibly as much as one generation) of monitoring have determined that successful recruitment has increased the overall size of both populations. Regeneration success should be measured in terms of abundant natural regeneration (with parental

contributions from many trees for genetic purposes) and measured directly with genetic analysis if possible (USFWS, 2013).

3. A prescribed burn plan is established to improve surrounding habitat to reduce high vegetation cover and promote recruitment, or research has documented an alternative method to burning that is successful in promoting reproduction. Appropriate management to improve the surrounding habitat would need to be successfully implemented. Funds must be available for appropriate long-term management (USFWS, 2013).

4. A seed bank is established at a recognized institution certified by the Center for Plant Conservation (CPC). The seed bank is needed for protection of the species in case of an unforeseen naturally occurring event that would create a lack of reproduction or die-off from disease. Seeds should represent the remaining genetic diversity of the species and the viability (i.e., germination percentage) of the seed collection should be determined (USFWS, 2013).

Recovery Actions:

- Secure and protect existing populations and habitat on private or unprotected lands through willing landowners (USFWS, 2004).
- Manage lands to control or eliminate threats to the plants and their habitat (USFWS, 2004).
- Conduct research to document life history characteristics and plants' responses to vegetation management (USFWS, 2004).
- Survey for additional populations and suitable habitat for reintroduction or reestablishment and establish new populations (USFWS, 2004).
- Develop management strategies and monitor populations to determine effectiveness of management (USFWS, 2004).
- Coordinate recovery actions with other listed species or species of concern (USFWS, 2004).
- Develop and implement a public outreach program (USFWS, 2004).
- Reevaluate recovery criteria and revise recovery plan based on knowledge obtained from research, monitoring, and management (USFWS, 2004).

Conservation Measures and Best Management Practices:

- Experiments should be undertaken to determine the effectiveness of mechanical clearing and controlled burns for the recruitment of young saplings. Any experiments conducted should include a genetic analysis component to determine the diversity of recruitment based on clearing method used (USFWS, 2013).
- Land managers should attempt to reclaim unused road and trail systems within existing stands. Restoration of these areas by replacing topsoil and planting native vegetation to anchor it in place could reduce erosion and increase the amount of suitable habitat within the existing stands. Within the Point Lobos stand, 0.46 hectares (1.1 acres) could be recovered, allowing recruitment to increase population size (USFWS, 2013).
- An effort should be made to determine where trees have been intentionally planted within the naturally occurring populations. For each of these plantings, it should be noted from where the seeds originated. All plantings should be undertaken from stock that derives from, and thus is genetically consistent with, the targeted planting area (USFWS, 2013).
- Genetic analyses should be undertaken to determine the relatedness of the two stands and a seed bank should be created by collecting seed from both stands and many individuals per site (USFWS, 2013).

- Any plans for trail construction or recreational public use at Point Lobos Ranch should only be considered if natural ecological processes within the stand will not be negatively impacted. In addition, no structures, fencing, signage, or other improvements should be installed near or within the stand that could hinder management activities, such as heavy equipment use, or prescribed burns (USFWS, 2013).

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SPECIES ACCOUNT: *Cycas micronesica* (Fadang)

Species Taxonomic and Listing Information

Listing Status: Threatened; 11/02/2015; Pacific Region (R1) (USFWS, 2016)

Physical Description

A cycad tree (USFWS, 2015).

Taxonomy

A member of the cycad family (Cycadaceae) (USFWS, 2015).

Historical Range

Just 10 years ago, Cycas micronesica was ubiquitous on the island of Guam, and similarly common on Rota (USFWS, 2015).

Current Range

It is known from Guam, Rota, and tentatively on Pagan, as well as Palau (politically the independent Republic of Palau) and Yap (geographically part of the Caroline Islands; politically part of the Federated States of Micronesia) (USFWS, 2015).

Critical Habitat Designated

No;

Life History

Food/Nutrient Resources

Reproduction Narrative

Adult: Not available

Habitat Vegetation or Surface Water Classification

Adult: Forest (USFWS, 2015)

Habitat Narrative

Adult: Occurs in the forest ecosystem (Hill et al. 2004, p. 280; Keppel et al. 2008, p. 1,006; Cibrian-Jaramillo et al. 2010, pp. 2,372–2,375; Marler 2013, in litt.) (USFWS, 2015).

Dispersal/Migration

Dispersal/Migration Narrative

Adult: Not available

Population Information and Trends

Population Trends:

Not available

Resiliency:
Moderate (inferred from USFWS, 2015)

Redundancy:
Moderate (inferred from USFWS, 2015)

Number of Populations:
15 - 20 (USFWS, 2015)

Population Size:
900,000 - 950,000 (USFWS, 2015)

Population Narrative:
Currently, there are 15 to 20 occurrences of Cycas micronesica totaling 900,000 to 950,000 individuals on the Micronesian Islands of Guam, Rota, Yap, and Palau (USFWS, 2015).

Threats and Stressors

Stressor: Cycad aulacaspis scale (USFWS, 2015)
Exposure:
Response:
Consequence:
Narrative: Cycas micronesica is currently under attack by a nonnative insect, the cycad aulacaspis scale (Aulacaspis yasumatsui) that is causing rapid mortality of plants at all locations (Marler 2014, in litt.). As of January 2013, C. micronesica mortality reached 92 percent on Guam, and cycads on Rota are experiencing a similar fate (Marler 2013, in litt.). All seedlings of C. micronesica in a study area were observed to die within 9 months of infestation by A. yasumatsui (Marler and Muniappan 2006, p. 3; Marler and Lawrence 2012, p. 233; Western Pacific Tropical Research Center 2012, p. 4; Marler 2013, pers. comm.) (USFWS, 2015).

Stressor: Development, military training, and urbanization (USFWS, 2015)
Exposure:
Response:
Consequence:
Narrative: In their 2015 Final SEIS (http:// guambuildupeis.us/) (see ‘‘Historical and Ongoing Human Impacts,’’ above), the U.S. Department of Navy states that approximately 5,000 Marines will be relocated from Okinawa to Guam, accompanied by approximately 1,300 dependents, with a concurrent introduction of support staff and development of infrastructure, and increased use of resources such as water (Berger et al. 2005, p. 347; JGPO– NavFac, Pacific 2015, p. ES–3). The current preferred alternative sites on Guam for cantonment and live-fire training include the Naval Computer and Telecommunications Station Finegayan and Northwest Field on Andersen AFB, where this species occurs. Further, the Navy is planning jungle training at the Naval Munitions Site (NMS) on Guam, which will require the establishment of foot trails within the southern portion of the NMS due to repeat use during maneuvering training. The inhabited island of Tinian and the uninhabited island of Pagan are planned to be used for military training with live-fire weapons and presence of military personnel. In November 2007, the people of Rota voted to legalize casino gambling to increase tourism, and two development projects have been

proposed. Development around and within forested areas on Rota will also directly impact the forest habitat and individuals of this species (USFWS, 2015).

Stressor: Ungulate activity (USFWS, 2015)
Exposure:
Response:
Consequence:
Narrative: Habitat degradation or destruction by ungulates is a threat to this species. Erosion, resulting from rooting and trampling by pigs, impacts native plant communities by contributing to watershed degradation and alteration of plant nutrient status, as well as causing direct damage to individual plants from landslides (Berger et al. 2005, pp. 42– 44; Vitousek et al. 2009, pp. 3,074– 3,086; Chan-Halbrendt et al. 2010, p. 251; Kessler 2011, pp. 320–324). Several herds of Asiatic water buffalo or carabao roam southern Guam and the Naval Magazine area, and cause damage to the forest and savanna ecosystems that support this species. Philippine deer have caused extensive damage resulting in changes in the forest structure, including erosion, grazing to the point of clearing the entire herbaceous understory, consumption of seeds and seedlings preventing regeneration of native plants and the spread of invasive plant species, and other physical damage (e.g., trunk rubbing) (Schreiner 1997, pp. 179–180; Wiles et al.1999, pp. 193–215; Berger et al. 2005, pp. 36, 45–46, 100; CNMI– SWARS 2010, p. 24; JGPO–NavFac, Pacific 2010b, p. 3–33; SWCA 2011, pp. 35, 42; Harrington et al. 2012, in litt.) (USFWS, 2015).

Stressor: Fire (USFWS, 2015)
Exposure:
Response:
Consequence:
Narrative: Fire is a human-exacerbated threat to native species and native ecosystems throughout the Mariana Islands, particularly on the island of Guam. Wildfires plague forest and savanna areas on Guam every dry season despite the island's humid climate, with at least 80 percent of wildfires resulting from arson (JGPO–NavFac, Pacific 2010b, p. 1–9). Fire can destroy dormant seeds of native species as well as plants themselves, even in steep or inaccessible areas. Successive fires that burn farther and farther into native habitat destroy native plants and remove habitat for native species by altering microclimate conditions to those favorable to alien plants (USFWS, 2015).

Stressor: Climate change (USFWS, 2015)
Exposure:
Response:
Consequence:
Narrative: On a global scale, sea level is rising as a result of thermal expansion of warming ocean water; the melting of ice sheets, glaciers, and ice caps; and the addition of water from terrestrial systems (Climate Institute 2011, in litt.). Sea level rose at an average rate of 0.1 in (3.1 mm) per year between 1961 and 2003 (IPCC AR4 2007, p. 30), with a predicted increase in 2100 of 1.6 to 4.6 ft. (0.5 to 1.4 m) above the 1990 level (Rahmstorf 2007, p. 368). This species occurs close to the coast in the adjacent forest ecosystem at or near sea-level and may be negatively impacted by sea-level rise and coastal inundation due to climate change (USFWS, 2015).

Recovery

Reclassification Criteria:
Not available - this species does not have a recovery plan.

Delisting Criteria:
Not available - this species does not have a recovery plan.

Recovery Actions:
- Not available - this species does not have a recovery plan.

Conservation Measures and Best Management Practices:
- A conservation project on Rota, administered through the Water and Environmental Research Institute of the Western Pacific at the University of Guam, is aimed to analyze the island's hydrology, with the ultimate goal of protection of the Sabana Watershed and Talakhaya Springs (Keel et al. 2007, pp. 5, 22–23). Erosion control, revegetation, and water source preservation conducted as part of this project may provide protection to this species (USFWS, 2015).
- There have been five fenced 1-ac (0.5-ha) exclosures erected on Tinian as of 2013, each planted with 1,000 individuals of mature Cycas micronesica (DON 2014, in litt.). Precautions were taken to ensure plantings had broad genetic representation. Cycads within these exclosures actively managed to ensure health and survival. Funding has been programmed to support this project through 2020. Tinian was selected for these exclosures since the scale does not occur on this island (USFWS, 2015).

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SPECIES ACCOUNT: *Hesperocyparis (=Cupressus) abramsiana* (Santa Cruz cypress)

Species Taxonomic and Listing Information

Listing Status: Reclassified as Threatened (effective 3/21/2016; USFWS, 2016); originally listed as Endangered (effective 02/09/1987); California/Nevada Region (R8)

Physical Description

A tree 1 to 25 meters (3 to 82 feet) in height. The grey bark is fibrous, thin, and broken into vertical strips or plates. Young shoots are 1 to 1.5 millimeters (0.06 inch) in diameter and are cylindric. The scale-like leaves are light bright green. The pollen cones are more or less 4-sided, being 3 to 4 millimeters (0.12 to 0.16 inch) long and 2 millimeters (0.08 inch) in diameter. Each pollen cone has 10 to 16 scales. The seed cones are spherical to widely elliptic. The 8 to 10 brown scales each have a central projection. The seeds are 3 to 5 millimeters (0.12 to 0.14 inch) long and dull brown. The seed has a conspicuous scar where it was attached to the seed cone (Bartel 1993). (USFWS, 1998)

Taxonomy

First collected by Marcus E. Jones in 1881 probably from Bonny Doon in Santa Cruz County. (USFWS, 1998). The taxonomy of cypresses has long been problematic because of their similar appearance (Lanner 1999). Since being listed, Cupressus abramsiana has been included with Gowen cypress by some (Eckenwalder 1993) and maintained a separate species by others (Bartel 1993). An initial genetic study confirmed the close relationship between Cupressus abramsiana and Gowen cypress while still recognizing them as separate taxa (Bartel et al. 2003). In a recent study of members of the cypress family, Little (2006) combined molecular sequence data with data on morphological characteristics to present a revision of the taxonomic relationships among them. In doing so, all North American members of the genus Cupressus, including what was Cupressus abramsiana, were moved to the genus Callitropsis (Little 2006). In 2009, the western cypresses, including Cupressus abramsiana, were segregated from the "true" cypresses and moved to a new genus, Hesperocyparis (Adams et al. 2009). (USFWS, 2009)

Historical Range

See current range/distribution.

Current Range

In California, in San Mateo County and Santa Cruz County, in the Santa Cruz Mountains. No historical distribution beyond these five sites is known. (USFWS, 1998)

Critical Habitat Designated

No;

Life History

Food/Nutrient Resources

Reproductive Strategy

Adult: Sexual (USFWS, 1998)

Lifespan

Adult: 100+ years (USFWS, 2009)

Key Resources Needed for Breeding

Adult: Recent disturbance, 50 - 100 year fire intervals (USFWS, 2009)

Reproduction Narrative

Adult: The average age at which individuals start producing cones is 11 years. The pollen cones produce copious amounts of pollen and the seed cones bear 8 - 10 seeds. Areas that have been recently disturbed through fire or mechanical means can support a high density of saplings. Trees may live in excess of 100 years. The release of seeds and subsequent germination are stimulated by fire. Natural fire cycles were estimated to be between 50 and 100 years (Keeley 1981 in Bartel and Knudsen 1982) (USFWS, 2009). The ovulate cones take 2 years to mature with seeds maturing at 15 to 18 months after pollination (USFWS, 1998).

Habitat Type

Adult: Terrestrial (USFWS, 2009)

Habitat Vegetation or Surface Water Classification

Adult: Coastal chaparral, mixed evergreen forest (USFWS, 2009)

Dependencies on Specific Environmental Elements

Adult: Mediterranean climate (USFWS, 1998)

Geographic or Habitat Restraints or Barriers

Adult: Coastal fog belt (USFWS, 2009); occurs at 1,000 - 2,500 ft. elevation (USFWS, 1998)

Spatial Arrangements of the Population

Adult: Clumped (USFWS, 2009)

Habitat Narrative

Adult: Populations occur as patches within a mosaic of coastal chaparral and mixed evergreen forests. All populations occur on or near dry ridges that are located inland from the coastal fog belt. Soils tend to be sandy or gravelly, and therefore well-drained and porous (USFWS, 2009). The Santa Cruz cypress habitat ranges in elevation from approximately 300 to 760 meters (1,000 to 2,500 feet) in areas having a Mediterranean climate with cool, wet winters and hot, dry summers (USFWS, 1998).

Dispersal/Migration

Dispersal

Adult: Low (USFWS, 1998)

Dispersal/Migration Narrative

Adult: Pollen is windblown (USFWS, 2009). Unlike seeds of other conifers, cypress seeds lack wings that would aid in dispersal; consequently, their natural ability to disperse is limited.

However, squirrels have been observed to facilitate Santa Cruz cypress cone/seed dispersal by chewing off branch tips, causing portions of branches with cones to fall to the ground (M. Hummel, pers. comm. 1993) (USFWS, 1998).

Population Information and Trends

Population Trends:

Not available

Number of Populations:

5 (USFWS, 1998)

Population Size:

> 5,100 (USFWS, 1998)

Adaptability:

Low (inferred from USFWS, 1998)

Population Narrative:

Five populations with a total of5,100+ individuals collectively occupy about 142 hectares (356 acres) along a 24 kilometer (15 mile) range in the Santa Cruz Mountains. Its restricted range means that destruction of even one population by random events would be a severe setback for recovery (USFWS, 1998).

Threats and Stressors

Stressor: Alteration of fire cycles (USFWS, 2009)
Exposure:
Response:
Consequence:
Narrative: The effects of altered fire frequencies on other cypress taxa have been studied. With a shorter fire interval, the species may face "immaturity risk" - widespread mortality of trees prior to their maturity may reduce the reproductive potential of the population (Zedelr 1977, de Gouvenain and Ansary 2006). A "senescence risk" may occur if the fire interval is longer than the lifespan of the trees, resulting in lower population vitality, reduced cone production, and reduced seedling establishment (Ne'eman et al. 1999) (USFWS, 2009).

Stressor: Climate change (USFWS, 2009)
Exposure:
Response:
Consequence:
Narrative: Current climate change predictions for terrestrial areas in the Northern Hemisphere indicate warmer air temperatures, more intense precipitation events, and increased summer continental drying (Field et al. 1999, Cayan et al. 2005, IPCC 2007). It is unknown at this time if climate change in California will affect C. abramsiana (USFWS, 2009).

Recovery

Reclassification Criteria:
Not applicable due to reclassification as Threatened (USFWS, 2016).

Delisting Criteria:
1. When all five populations are assured of long-term reproductive success, with insurance against failure provided by the availability of banked seed (USFWS, 1998).

Recovery Actions:

- 1. Protect Habitat for Populations on Private Land with Permission of the Landowner. Privately-held populations should be secured such that plans to manage the populations can subsequently be developed. The privately-held populations can be secured using several mechanisms: acquisition of property, gifts of easement or fee interest by the property owner, deed restrictions (provided restrictions cannot be changed privately without the knowledge of Federal, State and County agencies), acquisition of property rights (i.e., development rights, timber harvest rights), or permanent resource management easement. (USFWS, 1998)
- 2. Broaden the understanding of the demographics, life history, and ecology of the Santa Cruz cypress. Management of populations of Santa Cruz cypress is perhaps more difficult than protecting the sites. Management guidelines need to address site-specific threats to each of the populations. Before management plans can be developed, a better understanding of the population characteristics, and identification of those factors that may be affecting those characteristics is needed. Although initial demographic data has been collected on three of the five populations (Bracken Brae, Butano Ridge, and Eagle Rock) (Lyons 1988), none has been collected from the other two (Bonny Doon and Majors Creek). A more extensive effort to describe the demographic prof ile of each population should focus on identifying the life history stages critical to maintaining population viability (Schemske et al. 1994). When stands differ in age structure or size-class structure, the reasons for the differences should be identified so that appropriate management actions can be planned (see Task 3). (USFWS ,1998)
- 3. Manage and enhance each population and its habitat. Management of Santa Cruz cypress and its habitat will depend upon information gained from monitoring, threat analyses, and the evaluation of protection alternatives. There may be different management programs for each population. Development of management plans should include all interested and affected groups. The management program selected will require periodic review to ensure that it is effective in protecting the species. (USFWS, 1998).
- 4. Develop educational materials. Educational brochures and other materials (such as video or slide presentation) should be prepared that include discussion of the importance of the species to the region (e.g., legal status), plant identification, plant ecology and related management issues (e.g., use of fire or mechanical methods for species regeneration, recommended landscape species compatible for adjacent development). Separate brochures could be developed to target youth in public schools and an adult audience. (USFWS, 1998)
- 5. Due to the small numbers of individuals and populations of the Santa Cruz cypress, a seed bank should be established to maintain the genetic variability within and between populations as insurance against the possibility of stochastic extinction (extinction due to randomly-occurring events). The U.S. Forest Service Pacific Southwest Research Station in Albany, California has conducted preliminary analyses for genetic variability. (USFWS, 1998)

- Recommendation for Future Action from 2009 5-Year Review: The Service should continue to pursue efforts to secure the long-term conservation of the Bracken Brae population (USFWS, 2009).
- Recommendation for Future Action from 2009 5-Year Review: The Service should work with the California Department of Forestry and Fire Protection to clarify guidelines issued to landowners located adjacent to Cupressus abramsiana populations (USFWS, 2009).
- Recommendation for Future Action from 2009 5-Year Review: The Service should work with the county of Santa Cruz to clarify local protections for C. abramsiana (USFWS, 2009).
- Recommendation for Future Action from 2009 5-Year Review: The Service should work with agency and academic partners to determine rates of recruitment needed to maintain the long-term persistence of all populations (USFWS, 2009).
- Recommendation for Future Action from 2009 5-Year Review: The Service should support research that focuses on management strategies that will enhance the long-term persistence of C. abramsiana. In particular, the Service should participate in the inter-agency working group that has been established to address such issues in the aftermath of the Martin Fire at Bonny Doon Ecological Reserve (USFWS, 2009).
- Recommendation for Future Action from 2009 5-Year Review: The Service should partner with the Center for Plant Conservation and member gardens (e.g., Rancho Santa Ana Botanic Garden, Santa Barbara Botanic Garden, and UC Berkeley Botanic Garden) to ensure that seed representing all Cupressus abramsiana populations are banked at the National Seed Laboratory (USFWS, 2009).

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SPECIES ACCOUNT: *Pinus albicaulis* (Whitebark pine)

Species Taxonomic and Listing Information

Listing Status: Proposed threatened

Physical Description

Pinus albicaulis is a tree that is typically 5 to 20 meters (m) (16 to 66 feet (ft)) tall with a rounded or irregularly spreading crown shape. On higher density conifer sites, P. albicaulis tends to grow as tall, single-stemmed trees, whereas on open, more exposed sites, it tends to have multiple stems (McCaughey and Tomback 2001, pp. 113114). Above tree line, it grows in a krummholz form, with stunted, shrub-like growth caused by high winds and cold temperatures (Arno and Hoff 1989, p. 6). This pine species is monoecious (with both male pollen and female seed cones on the same tree). Its characteristic dark brown to purple seed cones are 5 to 8 centimeters (cm) (2 to 3 inches (in.)) long and grow at the outer ends of upper branches (Hosie 1969, p. 42).

Taxonomy

Pinus albicaulis is a 5-needled conifer species placed in the subgenus Strobus, which also includes other 5-needled white pines. This subgenus is further divided into two sections (Strobus and Parrya), and under section Strobus, into two subsections (Cembrae and Strobi). The traditional taxonomic classifications placed P. albicaulis in the subsection Cembrae with four other Eurasian stone pines (Critchfield and Little 1966, p. 5; Lanner 1990, p. 19). However, recent phylogenetic studies (Liston et al. 1999, 2007; Syring et al. 2005, 2007; as cited in Committee on the Status of Endangered Wildlife in Canada (COSEWIC) 2010, p. 4) showed no difference in monophyly (ancestry) between subsection Cembrae and subsection Strobi and merged them to form subsection Strobus. No taxonomic subspecies or varieties of P. albicaulis are recognized (COSEWIC 2010, p. 6). Based on this taxonomic classification information, we recognize P. albicaulis as a valid species and a listable entity.

Historical Range

The historical distribution of Pinus albicaulis is unknown.

Current Range

Pinus albicaulis occurs in scattered areas of the warm and dry Great Basin but it typically occurs on cold and windy high-elevation or high-latitude sites in western North America. As a result, many stands are geographically isolated (Arno and Hoff 1989, p. 1; Keane et al. 2012, p. 13). Its range extends longitudinally between 107 and 128 degrees west and latitudinally between 27 and 55 degrees north (McCaughey and Schmidt 2001, p. 33). The distribution of P. albicaulis includes coastal and Rocky Mountain ranges that are connected by scattered populations in northeastern Washington and southeastern British Columbia (Arno and Hoff 1990, p. 268; Keane et al. 2012, p. 13). The coastal distribution of P. albicaulis extends from the Bulkley Mountains in British Columbia to the northeastern Olympic Mountains and Cascade Range of Washington and Oregon, to the Kern River of the Sierra Nevada Range of east-central California (Arno and Hoff 1990, p. 268). Isolated stands of P. albicaulis are known from the Blue and Wallowa Mountains in northeastern Oregon and the subalpine and montane zones of mountains in northeastern California, south-central Oregon, and northern Nevada (Arno and Hoff 1990, p. 268; Keane et al. 2012, p. 13). The Rocky Mountain distribution of P. albicaulis ranges from northern British Columbia and Alberta to Idaho, Montana, Wyoming, and Nevada (Arno and Hoff 1990, p. 268;

Keane et al. 2012, p. 13), with extensive stands occurring in the Yellowstone ecosystem (McCaughey and Schmidt 2001, p. 33). The Wind River Range in Wyoming is the eastern most distribution of the species (Arno and Hoff 1990, p. 268; McCaughey and Schmidt 2001, p. 33).

Critical Habitat Designated

No;

Life History

Food/Nutrient Resources

Food Source

Adult: Sunlight

Food/Nutrient Narrative

Adult: Pinis albicaulis obtains its energy from sunlight via photosynthesis.

Reproductive Strategy

Adult: monoecious and masting

Lifespan

Adult: 500 years and sometimes more than 1000 years

Dependency on Other Individuals or Species

Adult: Clarks nutcatcher

Key Resources Needed for Breeding

Adult: No information

Reproduction Narrative

Adult: Pinus albicaulis is a slow-growing, long-lived tree with a life span of up to 500 years and sometimes more than 1,000 years (Arno and Hoff 1989, pp. 56). It is considered a keystone, or foundation species in western North America where it increases biodiversity and contributes to critical ecosystem functions (Tomback et al. 2001, pp. 7 8). As a pioneer or early successional species, it may be the first conifer to become established after disturbance, subsequently stabilizing soils and regulating runoff (Tomback et al. 2001, pp. 1011). At higher elevations, snow drifts around P. albicaulis trees, thereby increasing soil moisture, modifying soil temperatures, and holding soil moisture later into the season (Farnes 1990, p. 303). These higher elevation trees also shade, protect, and slow the progression of snowmelt, essentially reducing spring flooding at lowerelevations. Pinus albicaulis also provides important, highly nutritious seeds for a number of birds and mammals (Tomback et al. 2001, pp. 8, 10). P. albicaulis trees are capable of producing seed cones at 20 to 30 years of age, although large cone crops usually are not produced until 60 to 80 years (Krugman and Jenkinson 1974, as cited in McCaughey and Tomback 2001, p. 109). Therefore, the generation time of P. albicaulis is approximately 60 years (COSEWIC 2010, p. v). P. albicaulis seed predators are numerous and include more than 20 species of vertebrates including Clarks nutcracker (Nucifraga columbiana), pine squirrels (Tamiasciurus spp.), grizzly bears (Ursus arctos), black bears (Ursus americanus), Stellers Jay (Cyanocitta stelleri), and pine grosbeak (Pinicola enucleator) (Lorenz et al. 2008, p. 3). Seed predation plays a major role in P. albicaulis population dynamics, as seed predators largely determine the fate of seeds. However, P. albicaulis has co-evolved with seed predators and has several adaptations, like masting, that has allowed the species to persist despite heavy seed predation (Lorenz et al. 2008, p. 34). Masting is the process by which populations synchronize their seed production and provide varying amounts from year to year. During years with high seed production, typically once every 35 years in P. albicaulis (McCaughey and Tomback 2001, p. 110), seed consumers are satiated, resulting in excess seeds that escape predation (Lorenz et al. 2008, pp. 34).

Habitat Type

Adult: cold and windy high-elevation or high-latitude sites

Dependencies on Specific Environmental Elements

Adult: cool, high-elevation habitats

Geographic or Habitat Restraints or Barriers

Adult: northern habitat is currently impacted by disease (white pine blister rust)

Spatial Arrangements of the Population

Adult: clumped according to suitable habitat

Environmental Specificity

Adult: moderate

Tolerance Ranges/Thresholds

Adult: shade intolerant; tolerates poor soils, steep slopes, and windy exposures

Site Fidelity

Adult: high, individuals do not move

Dependency on Other Individuals or Species for Habitat

Adult: not applicable

Habitat Narrative

Adult: Pinus albicaulis is a hardy conifer that tolerates poor soils, steep slopes, and windy exposures and is found at alpine tree line and subalpine elevations throughout its range (Tomback et al. 2001, pp. 6, 27). It grows under a wide range of precipitation amounts, from about 51 to over 254 cm (20 to 100 in.) per year (Farnes 1990, p. 303). Pinus albicaulis may occur as a climax species, early successional species, or seral (midsuccessional stage) co-dominant associated with other tree species. Although it occurs in pure or nearly pure stands at high elevations, it typically occurs in stands of mixed species in a variety of forest community types.

Dispersal/Migration

Motility/Mobility

Adult: Sessile

Dispersal

Adult: Yes

Dependency on Other Individuals or Species for Dispersal

Adult: Clarks nutcracker

Dispersal/Migration Narrative

Adult: Pinis albicaulis is a tree; individuals do not move. The Clarks nutcracker serves as the main dispersal agent for P. albicaulis by caching seeds in disturbed sites, such as burns. In addition, Clarks nutcrackers can disperse seeds farther than the wind-dispersed seeds of other conifers, thereby facilitating P. albicaulis succession in burned sites over a broad geographic area (McCaughey et al. 1985, Tomback et al. 1990, 1993 in Keane and Parsons 2010, p. 58).

Population Information and Trends

Population Trends:

Declining

Species Trends:

Declining

Resiliency:

moderately low

Representation:

low

Redundancy:

moderately low

Population Growth Rate:

declining greater than 57 percent in the past 100 years

Minimum Viable Population Size:

unknown

Resistance to Disease:

low, but it is starting showing signs of adapting to disease

Adaptability:

low for climate change; some evidence of natural adaptation to white pine blister rust

Population Narrative:

Mortality data collected in multiple studies throughout the range of Pinus albicaulis strongly suggests that the species is in range-wide decline. Although the majority of available data was collected in the last several decades, the decline in P. albicaulis populations likely began sometime following the 1910 introduction of the exotic disease white pine blister rust. Although we do not have a study that quantifies the rate of decline across the entire range, we conclude that the preponderance of data from the studies listed below and elsewhere in this status review provides evidence of a substantial and pervasive decline throughout almost the entire range of the species.In Canada, based on current mortality rates, it is anticipated that Pinus albicaulis will decline by 57 percent within 100 years (COSEWIC 2010, p. 19). The value for this anticipated decline is likely an underestimate, as it assumes current mortality rates remain constant into the foreseeable future. Past trends have shown that mortality rates have been increasing over the last several decades. The range of mortality rates for P. albicaulis in the United States are similar to those in Canada, which suggests that the anticipated rates of decline will be similar.

Threats and Stressors

Stressor: Fire and Fire Suppression
Exposure:
Response:
Consequence:
Narrative: Fire is one of the most important landscape-level disturbance processes within high-elevation Pinus albicaulis forests (Agee 1993, p. 259; Morgan and Murray 2001, p. 238; Spurr and Barnes 1980, p. 422), and has been important to perpetuating early seral (successional stage) P. albicaulis communities (Arno 2001, p. 82; Shoal et al. 2008, p. 20). Without regular disturbance, primarily from fire, these forest communities follow successional pathways that eventually lead to dominance by shade-tolerant conifers such as Abies lasiocarpa, Picea engelmannii, and Tsuga mertensiana, to the exclusion of P. albicaulis (Keane and Parsons 2010, p. 57). When fire is present on the landscape, P. albicaulis has an advantage over its competitors for several reasons (Keane and Parsons 2010, p. 57). The Clarks nutcracker serves as the main dispersal agent for P. albicaulis by caching seeds in disturbed sites, such as burns. Fire creates sites that are suitable for this seed caching behavior and that most importantly contain optimal growing conditions for P. albicaulis (Tomback et al. 2001, p. 13). In addition, Clarks nutcrackers can disperse seeds farther than the wind-dispersed seeds of other conifers, thereby facilitating P. albicaulis succession in burned sites over a broad geographic area (McCaughey et al. 1985, Tomback et al. 1990, 1993 in Keane and Parsons 2010, p. 58). Additionally, P. albicaulis has thicker bark, a thinner crown, and a deeper root system, which allow it to withstand low-intensity fires better than many of its competitors (Arno and Hoff 1990 in Keane and Parsons 2010, p. 58). Historically, fire has been an important factor in maintaining healthy stands of P. albicaulis on the landscape. Fires in the high-elevation ecosystem of Pinus albicaulis can be of low intensity, high intensity, or mixed intensity. These varying intensity levels result in very different impacts to P. albicaulis communities. Low-intensity, surface-level ground fires occur frequently under low-fuel conditions. These fires remove small-diameter, thin-barked seedlings and allow large, mature trees to thrive (Arno 2001, p. 82). Low-intensity fires also reduce fuel loads and competition from fire-susceptible conifers, shrubs, and grasses, thereby opening up spaces necessary for the shade-intolerant P. albicaulis to regenerate and thus maintain prominence in seral communities (Arno 1986 in Keane et al. 1994, p. 215). High-intensity fires occur where high fuel loads, ladder fuels (vegetation below the crown level of forest trees, which allows fire to move from the forest floor to tree crowns), and other compounding conditions result in increased flammability (Agee 1993, p. 258). High-intensity fires, often referred to as stand replacement fires, or crown fires (Agee 1993, p. 16), produce intensive heat, resulting in the removal of all or most of the vegetation from the ground. High-intensity fires begin the process of vegetative succession by opening seed beds that become available for the establishment and development of shade-intolerant species like P. albicaulis. High-intensity

fires are generally less frequent because it takes longer time intervals to build the large fuel accumulations necessary to promote these types of fires (Agee 1993, p. 258). Mixed intensity fires are most common and result in a mosaic of dead trees, live trees, and open sites for regeneration (Arno 1980, p. 460; Keane 2001a, p. 17). In general, historical fire return intervals in P. albicaulis communities have been estimated at between 50 and 300 years (Arno 1980, p. 461). Fire suppression has had unintended negative impacts on Pinus albicaulis populations (Keane 2001a, entire), due to this shift from a natural fire regime to a managed fire regime. Stands once dominated by P. albicaulis have undergone succession to more shade-tolerant conifers (Arno et al. 1993 in Keane et al. 1994, p. 225; Flanagan et al. 1998, p. 307). Once shade-tolerant conifer species become firmly established, the habitat is effectively lost to P. albicaulis until a disturbance like fire once again opens the area for P. albicaulis regeneration. Determining the total amount of P. albicaulis habitat lost to succession rangewide is difficult, as there is seldom a historic baseline for comparison, and the degree of succession is very specific to local conditions (Keane 2011a, pers. comm.). Shade-tolerant conifer species grow more densely than shade intolerant conifer species like P. albicaulis (Minore 1979, p. 3). Denser stands eliminate the open sites that are often used by Clarks nutcracker for seed caching and which are also the sites required to facilitate the regeneration of the shade-intolerant P. albicaulis. Additionally, the growth of more homogeneously structured stands with continuous crowns and increased surface fuels has resulted in fires that are larger and more intense (Keane 2001b, p. 175). Pinus albicaulis cannot withstand high-intensity fires; during such fires, all age and size classes can be killed. However, newly burned areas provide a seedbed for P. albicaulis, and if stands of unburned cone-producing P. albicaulis are nearby (i.e., within the range of Clarks nutcracker caching behavior), Clarks nutcrackers will cache those seeds on the burned site, and regeneration is very likely. However, the introduction of the disease white pine blister rust and the current epidemic of the predatory mountain pine beetle (Dendroctonus ponderosae) have reduced or effectively eliminated P. albicaulis seed sources on a landscape scale. Although there is variation in the degree to which specific stands have been impacted, over the range of P. albicaulis the widespread incidence of poor stand health from disease and predation, coupled with changes in fire regimes, means that regeneration of P. albicaulis following fire is unlikely in many cases (Tomback et al. 2008, p. 20).

Stressor: Fire and Fire Suppression and the Interaction of Other Factors
Exposure:
Response:
Consequence:
Narrative: Environmental changes resulting from climate change are expected to exacerbate the already observed negative effects of fire suppression (i.e., forest succession, increased fire intensity) (see the Climate Change section below). These environmental changes are predicted to increase the number, intensity, and extent of wildfires (Aubry et al. 2008, p. 6; Keane 2001b, p. 175). Already, large increases in wildfire have been documented and are particularly pronounced in Northern Rockies forests, which account for 60 percent of documented increases in large fires (Westerling et al. 2006, p. 941, 943). Some of the increase has been independent of past management activities and, thus, appears to be a direct result of warming trends in the last several decades (Westerling et al. 2006, p. 943). In 2013, fires burned approximately 7,507 ha (18,552 ac) in P. albicaulis stands in the Northern Rockies on USFS lands (Shelley 2014), which is approximately 0.3% of the Northern Rockies range of the species (2,757,580 ha (6,814,128 ac)) (Keane et al. 2012). Fire suppression is also expected to negatively interact with white pine blister rust and mountain pine beetle predation. As forests become denser, individual Pinus albicaulis

are more vulnerable to white pine blister rust and infestation by mountain pine beetle (see Factor C, Disease and Predation). As mortality from white pine blister rust and mountain pine beetle increase, forest succession to more dense stands of shade tolerant conifers is accelerated (Keane 2011a, pers. comm.).

Stressor: Climate Change
Exposure:
Response:
Consequence:
Narrative: Direct habitat loss from climate change is anticipated to occur with current habitats becoming unsuitable for P. albicaulis as temperatures increase and soil moisture availability decreases (Hamman and Wang 2006, p. 2783; Schrag et al. 2007, p. 8; Aitken et al. 2008, p. 103). Habitat loss is expected because (1) temperatures become so warm that they exceed the thermal tolerance of P. albicaulis and the species is unable to survive or (2) warmer temperatures favor other species of conifer that currently cannot compete with P. albicaulis in cold high-elevation habitats. Pinus albicaulis is widely distributed and thus likely has a wide range of tolerance to varying temperatures (Keane 2011c, pers.comm.). Therefore, increasing competition from other species that cannot normally persist in current P. albicaulis habitats is possibly the more probable climate-driven mechanism for habitat loss. Given the anticipated loss of suitable habitat, P. albicaulis persistence will likely be dependent on the species ability to either migrate to new suitable habitats, or adapt to changing conditions (Aitken et al. 2008, p. 95). Historical (paleoecological) evidence indicates that plant species have generally responded to past climate change through migration, and that adaptation to changing climate conditions is less likely to occur (Bradshaw and McNeilly 1991, p. 12; Huntley 1991, p. 19). Adaptation to a change in habitat conditions as a result of a changing climate is even more unlikely for P. albicaulis, given its very long generation time of approximately 60 years (Bradshaw and McNeilly 1991, p. 10). The rate of latitudinal plant migration during past warming and cooling events is estimated to have been on the order of 100 m (328 ft) per year (Aitken et al. 2008, p. 96). Given the current and anticipated rates of global climate change, migration rates will potentially need to be substantially higher than those measured in historic pollen records to sustain the species over time. A migration rate of at least a magnitude higher (1,000 m (3,280 ft)) per year is estimated to be necessary in order for tree species to be capable of tracking suitable habitats under projected warming trends (Malcolm et al. 2002, entire). Latitudinal migration rates on this scale may significantly exceed the migration abilities of many plant species, including P. albicaulis (Malcolm et al. 2002, p. 844845; McKenney et al. 2007, p. 941). Pinus albicaulis may have an advantage in its ability to migrate given that its seeds are dispersed by Clarks nutcracker. As mentioned above, Clarks nutcrackers can disperse seeds farther than the wind-dispersed seeds of other conifers (McCaughey et al. 1985, Tomback et al. 1990, 1993 in Keane and Parsons 2010, p. 58). However, migration of P. albicaulis to the north may be impeded by the disease white pine blister rust, which is currently present at the northern range limits of P. albicaulis (Smith et al. 2008, Figure 1, p. 984; Resler and Tomback 2008, p. 165). Pinus albicaulis already is typically the first species to establish on cold, exposed high-elevation sites, thus the species could potentially migrate higher in elevation to more suitable habitats. Shifts in the optimum elevation for many high-elevation plant species have already been documented under current warming trends (Lenoir et al. 2008, p. 1770). However, elevational migration as a refuge from temperature increase has limits, because eventually, suitable habitat may not be present even on mountaintops due to continuing temperature increases. Climate change is expected to significantly decrease the probability of rangewide persistence of Pinus albicaulis. Projections from an empirically based bioclimatic

model for P. albicaulis showed a rangewide distribution decline of 70 percent and an average elevation loss of 333 m (1,093 ft) for the decade beginning in 2030 (Warwell et al. 2007, p. 2). At the end of the century, less than 3 percent of currently suitable habitat is expected to remain (Warwell et al. 2007, p. 2). Similarly, climate envelope modeling on P. albicaulis distribution in British Columbia estimated a potential decrease of 70 percent of currently suitable habitat by the year 2055 (Hamman and Wang 2006, p. 2783). The area occupied by P. albicaulis in the Greater Yellowstone Ecosystem also is predicted to be significantly reduced with increasing temperature under various climate change scenarios (Schrag et al. 2007, p. 6). P. albicaulis is predicted to be nearly extirpated under a scenario of warming only and warming with a concomitant increase in precipitation (Schrag et al. 2007, p. 7). Climate envelope modeling by the USDA Forest Service using the A2 scenario projects that by 2090, a temperature increase of 9.1 °F (5.1 °C) would cause P. albicaulis suitable climate to contract to the highest elevation areas in the northern Shoshone National Forest and Greater Yellowstone Ecosystem or be extirpated (Rice 2012, p. 31). Loehman and others (2010) modeling study indicated that climate changes may significantly impact P. albicaulis in Glacier National Park through the indirect mechanisms of altered distributions of competing tree species and increased fire frequency and fire size. The above studies all suggest that the area currently occupied by P. albicaulis will be severely reduced in the foreseeable future.

Stressor: Climate Change and the Interaction of Other Factors
Exposure:
Response:
Consequence:
Narrative: In addition to direct habitat loss, Pinus albicaulis is expected to experience decrease in population size from synergistic interactions between habitat changes as a result of climate change and other threat factors including altered fire regimes, disease, and predation. P. albicaulis has evolved with fire, and under many conditions, fire is beneficial to the species (see Fire and Fire Suppression above). However, environmental changes resulting from climate change are expected to alter fire regimes resulting in increased fire intervals, increased fire severity, and habitat loss (Westerling et al. 2006, p. 943). Pinus albicaulis also evolved with the predatory native mountain pine beetle (Dendroctonus ponderosae). However, the life cycle of the mountain pine beetle is temperature dependent, and warming trends have resulted in unprecedented mountain pine beetle epidemics throughout the range of P. albicaulis (the interaction of mountain pine beetle and P. albicaulis is discussed further below under Factor C, Predation) (Logan et al. 2003, p. 130; Logan et al. 2010, p. 896). At epidemic levels, mountain pine beetle outbreaks become stand-replacing events killing 80 to 95 percent of suitable host trees, and in many parts of the P. albicaulis range, those levels of mortality have already been reached (Gibson et al. 2008, p. 10). Even populations of P. albicaulis once considered mostly immune to mountain pine beetle epidemics are now being severely impacted; mountain pine beetles have now moved into areas previously climatically inhospitable for epidemic-level mountain pine beetle population growth (Carroll et al. 2003 in Gibson et al. 2008, p. 4; Raffa et al. 2008, p. 503; Logan et al. 2010, p. 895). Given ongoing and predicted environmental changes resulting from global climate change, we expect the expansion of habitat favorable to mountain pine beetle (and mountain pine epidemics) to continue into the foreseeable future.

Stressor: Disease
Exposure:
Response:

Consequence:

Narrative: White pine blister rust is a disease of 5-needled pines caused by a nonnative fungus, Cronartium ribicola (Geils et al. 2010, p. 153). It was introduced into western North America in 1910 near Vancouver, British Columbia (McDonald and Hoff 2001, p. 198). White pine blister rust initially spread rapidly through maritime and montane environments, which have environmental conditions more conducive to spread of infection, but over several decades, it spread through continental and alpine environments throughout western North America (Geils et al. 2010, p. 163). White pine blister rusts rate and intensity of spread is influenced by microclimate and other factors (described below). Therefore, the incidence of white pine blister rust at stand, landscape, and regional scales varies due to time since introduction and environmental suitability for its development. It continues to spread into areas originally considered less suitable for persistence, and it has become a serious threat, causing severe population losses to several species of western pines, including Pinus albicaulis, P. monticola (western white pine), and P. lambertiana Dougl. (sugar pine) (Schwandt et al. 2010, pp. 226230). Its current known geographic distribution in western North America includes all U.S. States (except Utah, as well as the Great Basin Desert) and British Columbia and Alberta, Canada (Tomback and Achuff 2010, pp. 187, 206). The white pine blister rust fungus has a complex life cycle: It does not spread directly from one tree to another, but alternates between living primary hosts (i.e., 5- needle pines) and alternate hosts. Alternate hosts in western North America are typically woody shrubs in the genus Ribes (gooseberries and currants) but also may include herbaceous species of the genus Pedicularis (lousewort) and the genus Castilleja (paintbrush) (McDonald and Hoff 2001, p. 193; McDonald et al. 2006, p. 73). Ribes is widespread in North America and, while most species are susceptible to white pine blister rust infection, they vary in their susceptibility and capability to support inoculum (spores) that are infective to white pines, depending on factors such as habitat, topographic location, timing, and environment (Zambino 2010, pp. 265268). A widescale Federal program to eradicate Ribes from the landscape was conducted from the 1920s to the 1960s. However, due to the abundance of Ribes shrubs, longevity of Ribes seed in the soil, and other factors, white pine blister rust continued to spread, and pathologists realized that eradication was ineffective in controlling white pine blister rust. White pine blister rust is now pervasive in high-altitude 5-needled pines within most of the western United States (McDonald and Hoff 2001, p. 201). White pine blister rust progresses through five spore stages to complete each generation: two spore stages occur on white pine (Pinus spp.), and three stages occur on an alternate host. The five fungal spore stages require specific temperature and moisture conditions for production, germination, and dissemination. The spreading of spores depends on the distribution of hosts, the microclimate, and the different genotypes of white pine blister rust and hosts (McDonald and Hoff 2001, pp. 193, 202). Local meteorological conditions also may be important factors in infection success, infection periodicity, and disease intensity (Jacobi et al. 2010, p. 41). On white pines, spores enter through openings in the needle surface, or stomates, and move into the twigs, branches, and tree trunk, causing swelling and cankers to form. White pine blister rust attacks seedlings and mature trees, initially damaging upper canopy and cone-bearing branches and restricting nutrient flows; it eventually girdles branches and trunks, leading to the death of branches or the entire tree (Tomback et al. 2001, p. 15, McDonald and Hoff 2001, p. 195). White pine blister rust can kill small trees within 3 years, and even one canker can be lethal. While some infected mature trees can continue to live for decades, their cone-bearing branches typically die, thereby eliminating the seed source required for reproduction (Geils et al . 2010, p. 156). In addition, the inner sapwood moisture decreases, making trees prone to desiccation and secondary attacks by insects (Six and Adams 2007, p. 351). Death to upper branches results in lower or no cone production and a reduced likelihood that seed will be dispersed by Clarks nutcrackers (McKinney and Tomback 2007, p. 1049). Similar to a total loss of cone production, even when cone production is low there could be a loss of regeneration for two reasons: (1) Clarks nutcrackers abandon sites with low seed production and (2) the proportion of seeds taken by predators becomes so high that no seeds remain for regeneration (COSEWIC 2010, p. 25). Each year that an infected tree lives, the white pine blister rust infecting it continues to produce spores, thereby perpetuating and intensifying the disease. A wave, or massive spreading, of new blister rust infections into new areas or intensification from a cumulative buildup in already-infected stands occurs where Ribes shrubs are abundant and when summer weather is favorable to spore production and dispersal. Spores can be produced on pines for many years, and appropriate conditions need to occur only occasionally for white pine blister rust to spread and intensify (Zambino 2010, p. 265). The frequency of wave years depends on various factors, including elevation, geographical region, topography, wind patterns, temperature, and genetic variation in the rust (Kendall and Keane 2001, pp. 222223). Because its abundance is influenced by weather and host populations, white pine blister rust also is affected by climate change. If conditions become moister, white pine blister rust will likely increase; conversely, where conditions become both warmer and drier, it may decrease. Because infection is usually through stomates, whatever affects the stomates affects infection rates (Kliejunas et al. 2009, pp. 1920). Stomates close in drought conditions and open more readily in moist conditions. In general, weather conditions favorable to the intensification of white pine blister rust occur more often in climates with coastal influences than in dry continental climates (Kendall and Keane 2001, p. 223). Due to current climate conditions in western North America, white pine blister rust now infects Pinus albicaulis populations throughout all of its range except for the interior Great Basin (Nevada and adjacent areas) (Tomback and Achuff 2010, Figure 1a, p. 187). However, the small uninfected area in the Great Basin accounts for only 0.4 percent of P. albicaulis distribution in the United States. The incidence of white pine blister rust is highest in the Rocky Mountains of northwestern Montana and northern Idaho, the Olympic and western Cascade Ranges of the United States, the southern Canadian Rocky Mountains, and British Columbias Coastal Mountains (Schwandt et al. 2010, p. 228; Tomback et al. 2001, p. 15).

Stressor: Insect Predation
Exposure:
Response:
Consequence:
Narrative: Pinus albicaulis trees are fed upon by a variety of insects; however, none has had a more widespread impact than the native mountain pine beetle (Dendroctonus ponderosae Hopkins). The mountain pine beetle is recognized as one of the principal sources of P. albicaulis mortality (Raffa and Berryman 1987, p. 234; Arno and Hoff 1989, p. 7). Mountain pine beetles are true predators on P. albicaulis and other western conifers because, to successfully reproduce, the beetles must kill host trees (Logan and Powell 2001, p. 162; Logan et al. 2010, p. 895). Upon locating a suitable host (i.e., large diameter tree with greater resources for brood production success), adult female mountain pine beetles emit pheromones that attract adult males and other adult females to the host tree. This attractant pheromone initiates a synchronized mass attack for the purpose of overcoming the host trees defenses to mountain pine beetle predation. Once a tree has been fully colonized, the beetles produce an anti-aggregation pheromone that signals to incoming beetles to pass on to nearby unoccupied trees. Almost all host trees, even stressed individuals, will mount a chemical defense against these mass attacks. However, given a sufficient number of beetles, even a healthy trees defensive mechanisms can be exhausted (Raffa and Berryman 1987, p. 239). Following the pheromone-mediated mass attack, male and female

mountain pine beetles mate in the phloem (living vascular tissue) under the bark of the host tree. Females subsequently excavate vertical galleries where they lay eggs. Larvae hatched from these eggs feed on the phloem, pupate, and emerge as adults to initiate new mass attacks of nearby suitable trees (Gibson et al. 2008, p. 3). Mountain pine beetle development is directly controlled by temperature. The entire mountain pine beetle life cycle (from egg to adult) can take between 1 and 2 years depending on ambient temperatures. Warmer temperatures promote a more rapid development that facilitates a 1-year life cycle (Amman et al. 1997, p. 4; Gibson et al. 2008, p. 3). Beetle activity in the phloem mechanically girdles the host tree, disrupting nutrient and water transport and ultimately killing the host tree. Additionally, mountain pine beetles carry on their mouthparts symbiotic blue-stain fungi, which are introduced into the host tree. These fungi also inhibit water transport and further assist in killing the host tree (Raffa and Berryman 1987, p. 239; Keane et al. 2010, p. 34). Mountain pine beetles are considered an important component of natural forest disturbance (Raffa et al. 2008, p. 502; Bentz et al. 2010, p. 602). At endemic or natural levels, mountain pine beetle remove relatively small areas of trees, changing stand structure and species composition in localized areas. However, when conditions are favorable, mountain pine beetle populations can erupt to epidemic levels and create stand-replacing events that kill 80 to 95 percent of suitable host trees (Keane et al. 2010, p. 34). Such outbreaks are episodic, can have a magnitude of impact on the structure of western forests greater than wildfire (the other major component of natural forest disturbance), and are often the primary renewal source for mature stands of western pines (Hicke et al. 2006, p. 1; Raffa et al. 2008, pp 502-503). Mountain pine beetle outbreaks typically subside only when suitable host trees are exhausted or temperatures are sufficiently low to kill larvae and adults (Gibson et al. 2008, p. 2). The range of mountain pine beetle completely overlaps with the range of Pinus albicaulis, and mountain pine beetle epidemics affecting P. albicaulis have occurred throughout recorded history (Keane et al. 2010, p. 34). Recent outbreaks occurred in the 1930s, 1940s, and 1970s, and numerous ghost forests of dead P. albicaulis still dot the landscape as a result (Arno and Hoff 1989, p. 7; Ward et al. 2006, p. 8). Despite recorded historical impacts to the species, Pinus albicaulis has not been considered an important host of mountain pine beetle in the past. Unlike the lower elevation sites occupied by mountain pine beetles primary hosts, P. contorta Douglas (lodgepole pine) and P. ponderosae (ponderosa pine), the high-elevation sites occupied by P. albicaulis typically have been climatically inhospitable to mountain pine beetle (Logan and Powell 2001, p. 161). At the low temperatures typical of high-elevation sites, mountain pine beetle mostly experience a 2-year life cycle, which is not favorable to epidemic outbreaks (i.e., eruptive population growth). Warmer temperatures promote a 1-year life cycle, which facilitates the synchronized mass attacks important in overcoming host tree defenses (Logan and Powell 2001, p. 167). However, unlike previous epidemics, the current mountain pine beetle outbreak is having an increasingly significant rangewide impact on Pinus albicaulis (Logan et al. 2003, p. 130; Logan et al. 2010, p. 896). The reported mortality rates of mostly mature trees (i.e. large-diameter trees) can be as high as 96 percent (Gibson et al. 2008, p. 9). In 2007 alone, P. albicaulis trees on almost 202,342 ha (500,000 ac) were killed. At the time this was the highest recorded mountain pine beetle mortality ever reported for P. albicaulis (Gibson et al. 2008, p. 2). The number of acres with mountain pine beetle-killed P. albicaulis trees continues to increase significantly rangewide, and in 2009 P. albicaulis trees on an estimated 809,371 ha (2,000,000 ac) were killed (Service 2010). Aerial survey results from 2003-2013 estimate over 5.3 million P. albicaulis trees were attacked and killed by mountain pine beetle across 351,267 ha (868,000 acres) in the USFS Region 1 (Northern Region) (Shelly 2014, pers. comm.). Trends of environmental effects from climate change have provided the favorable conditions necessary for the current, unprecedented mountain pine beetle epidemic in high-elevation communities across the western United States and Canada (Logan and Powell 2001, p. 167; Logan et al. 2003, p. 130; Raffa et al. 2008, p. 511). Warming trends have resulted in not only intensified mountain pine beetle activity in high-elevation Pinus albicaulis forests, but have resulted in mountain pine beetle range expansion into more northern latitudes and higher elevations (Logan and Powell 2003, p. 131; Carroll et al. 2003 in Gibson et al. 2008, p. 4; Raffa et al. 2008, p. 503; Logan et al. 2010, p. 895). Winter temperatures are now warm enough for winter survival for all mountain pine beetle life stages and for maintenance of the 1-year life cycle that promotes epidemic mountain pine beetle population levels (Buotte 2014, pers. comm.; Bentz and Schen-Langenheim 2007, p. 47; Logan et al. 2010, p. 896). Along with warmer winter conditions, summers have been drier, with droughts occurring through much of the range of P. albicaulis (Bentz et al. 2010, p. 605). Mountain pine beetles frequently target drought-stressed trees, which are more vulnerable to attack as they are less able to mount an effective defense against even less dense mass attacks by mountain pine beetles (Bentz et al. 2010, p. 605). Given ongoing and predicted environmental effects from climate change, we expect the expansion of habitat favorable to mountain pine beetle (and mountain pine epidemics) to continue into the foreseeable future. The current mountain pine beetle epidemic began in the late 1990s and continues to be an important source of mortality for P. albicaulis (Shelly 2014, pers. comm.; Macfarlane et al. 2013, pg. 434; Mahalovich 2013, p. 21). However, we are aware of recent monitoring data that indicates that the current epidemic may be waning in some areas (Shelly 2014, pers. comm.; Olliff et al. 2013, p.245; Hayes 2013, p.3). This reduction in beetle-caused mortality is expected. Significant numbers of P.albicaulis have already been killed leaving less food (live trees) available for mountain pine beetles to continue reproducing at epidemic levels. Despite the apparent reduction of mountain pine beetle-caused mortality in some areas, we expect that mountain pine beetle will remain a threat to P. albicaulis. We anticipate that ongoing warming tr

Stressor: Synergistic Interactions between Disease and Predation
Exposure:
Response:
Consequence:
Narrative: White pine blister rust and mountain pine beetle act both individually and synergistically to threaten Pinus albicaulis rangewide. Mountain pine beetle will preferentially attack P. albicaulis infected with, and weakened by, white pine blister rust (Six and Adams 2007, p. 351; Bokino and Tinker 2012, p. 38). This preference results in increased susceptibility of P. albicaulis to mountain pine beetle-caused mortality. Mountain pine beetles and white pine blister rust also interact in other ways that threaten P. albicaulis regeneration and persistence. Mountain pine beetles preferentially target large mature trees. As a result, large trees are removed from populations, leaving smaller trees for regeneration in a less competitive environment. Unfortunately, white pine blister rust is not selective and infects all age and size classes of P. albicaulis. Thus, in the current environment that contains epidemic levels of mountain pine beetle and a nearly ubiquitous presence of white pine blister rust, P. albicaulis that have escaped mountain pine beetle mortality are still susceptible to white pine blister rust, and the possibility of regeneration following mountain pine beetle epidemics is jeopardized. Conversely, the small percentage of P. albicaulis individuals that are genetically resistant to white pine blister rust, and thus critical to species persistence, are still vulnerable to mountain pine beetle attack. White pine blister rust and mountain pine beetle further impact the probability of Pinus albicaulis regeneration because both act to severely decrease seed cone production. White pine blister rust does this by killing cone-bearing branches, such that even if the tree itself remains alive for some time, seed production is compromised. Mountain pine beetles decrease

seed production by targeting and killing larger trees, which are the main trees that bear cones. A severe reduction in seed production has the potential to limit the effectiveness of the masting strategy employed by P. albicaulis (see Taxonomy and Life History), such that the proportion of seeds taken by seed predators will eventually become too high to allow regeneration (Rapp 2013, p. 1349). Additionally, severe seed reduction disrupts the relationship between P. albicaulis and Clarks nutcracker (Barringer et al. 2012, pg. 10). Clarks nutcrackers eventually abandon P. albicaulis stands when seed production is too low (McKinney et al. 2009, p. 599). Limited research has focused on detecting amounts of Pinus albicaulis regeneration. Most remaining high-elevation P. albicaulis stands in the U.S. Intermountain West that are climax communities have little regeneration (Kendall and Keane 2001b, p. 228). In contrast, new and advanced P. albicaulis regeneration was documented on the majority of plots in southwestern Montana and eastern Oregon, indicating that the Wallowa and Pioneer Mountains sites seem to be more vigorous and to be regenerating better than sites farther north in the Rockies (Larson 2007, pp. 1618). However, there is much P. albicaulis site variability and the regeneration on some of these sites was preceded by a particularly large cone crop in 2006. In addition, as seedlings grow, their increased foliage surface area becomes a larger target for infection by white pine blister rust spores (Tomback et al. 1995, p. 662). Therefore, despite observed regeneration, the level of effective regeneration (i.e., seedlings that actually reach a reproductive age) is questionable given the high incidence of white pine blister rust currently on the landscape. We conclude that P. albicaulis regeneration will generally be less successful in the future than it has been in the past.

Stressor: Inadequate regulations
Exposure:
Response:
Consequence:
Narrative: The existing regulatory mechanisms currently in place throughout the range of Pinus albicaulis are inadequate to reduce or eliminate any of the four main threats to the species identified abovethe loss of habitat from fire suppression and the exacerbating environmental effects of climate change under Factor A, and mortality from white pine blister rust and mountain pine beetle. Therefore, based on our review of the best scientific and commercial information available, we conclude that existing regulatory mechanisms are inadequate to protect P. albicaulis or its habitat.

Recovery

Reclassification Criteria:
Not applicable

Delisting Criteria:
Not applicable

Recovery Actions:

- We support continuing monitoring efforts across several states by BLM for the conservation of Pinus albicaulis. The USFS in Region 1 has conducted aerial monitoring for almost 50 years. The valuable information gathered from these surveys includes determining the approximate location and amount of tree mortality, defoliation, and other non-fire damage for P. albicaulis. We recommend continuing the aerial surveys and continuing reforestation efforts such as planting of seedlings, prescribed burning and fuels treatments on USFS and BLM managed lands and in Canada across the range of P. albicaulis. Various research efforts are ongoing on blister rust and its impacts on P. albicaulis in the U.S. and Canada. Most current management and research focuses on producing P. albicaulis with inherited resistance to white pine blister rust and genetic management. This research may provide important information in conserving P. albicaulis populations in the future.

Conservation Measures and Best Management Practices:

- Most current management and research focuses on producing Pinus albicaulis with inherited resistance to white pine blister rust. Additional research investigates natural regeneration and silvicultural treatments, such as appropriate site selection and preparation, pruning, and thinning in order to protect genetic resources, increase reproduction, reduce blister rust damage, and increase stand volume (Zeglen et al. 2010, p. 361). Genetic management of white pine blister rust is actively conducted for P. albicaulis (Mahalovich 2013). Cone collections are used for blister rust resistance testing, molecular genetics studies, other research, growing compatible rootstock for seed orchards, and gene conservation (Mahalovich 2013). Efforts are underway to coordinate natural regeneration monitoring (Mahalovich 2013). In the USFS Region 1 and the Region 4, 160 ha (397 acres) of P. albicaulis seedlings were planted in 2012 (Mahalovich 2013). As of 2013, approximately 1,453 ha (3,547 acres) P. albicaulis seedlings had been planted among three USFS Regions (Mahalovich 2013).
- The USFS in Region 1 finalized a range-wide restoration strategy for Pinus albicaulis pine forests (Keane et.al. 2012). The objectives are to promote P. albicaulis survival and regeneration for ecological diversity, wildlife, hydrologic and other benefits through the use of planned and unplanned ignitions (Keane et.al. 2012). The strategy contains guiding principles, central tenets for a strategy and assessment criteria for future planning (Keane et.al. 2012). The USFS Region 1 also completed P. albicaulis plantings in 2013 on a small scale and may continue with future reforestation plantings (Shelley 2014, pers. comm.).
- The objectives for modeling efforts for the distribution and extent of Pinus albicaulis are to create a comprehensive set of methods to produce GIS products that map current P. albicaulis extent, current potential range, and suitable regeneration areas, implement these methods to create the GIS map products for the Flathead National Forest, and conduct a field validation of the Flathead National Forest GIS map products (Housman 2014, pers.comm.). These validated single-species map products will guide management decisions for restoration projects, fire management activities, and ESA compliance within the Flathead National Forest with expected report completion in 2014.
- The BLM continues to institute various programs for the conservation of Pinus albicaulis. In Wyoming on Commissary Ridge, the BLM had been removing mountain pine beetle infested trees and thinning subalpine fir and lodgepole pine from around surviving, healthy P. albicaulis, but was not able to continue in 2012 due to high fire danger (Means 2013, pers. comm.). Further conservation projects conducted by BLM in 2012 included a P. albicaulis inventory throughout Idaho, Montana and Wyoming. The BLM in Wyoming conducted aerial and ground surveys of potential P. albicaulis populations during 2012. Initially the BLM had estimated that there were between 1,011 ha (2,500 acres) and 1,416 ha (3,500 acres) of P. albicaulis on BLM lands in Wyoming. The 2012 surveys indicate that there are approximately 4,127 ha (10,200 ac) of confirmed P. albicaulis stands. The BLM has given this new location information to the Greater Yellowstone Resource Inventory Network, so that the whitebark pine stands can be integrated into the Greater Yellowstone whitebark pine monitoring program. This effort will provide baseline and annually updated information on the status of white pine blister rust infection levels and the level of mountain pine beetle infestation within these stands.

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U.S. FISH AND WILDLIFE SERVICE SPECIES ASSESSMENT AND LISTING PRIORITY ASSIGNMENT FORM 04/01/2021

SPECIES ACCOUNT: *Torreya taxifolia* (Florida torreya)

Species Taxonomic and Listing Information

Listing Status: Endangered; 1/23/1984; Southeast Region (Region 4) (USFWS, 2015)

Physical Description

A relatively small, symmetrical evergreen tree (mature trees can reach 18 m in height but none of the plants surviving in the species' natural range are over sapling size). Herbage has a pungent, disagreeable odor when crushed. This is one of the oldest tree species on earth; Taxus taxifolia fossils date back at least 165 million years. It is also one of the few close relatives of the Pacific yew (Taxus brevifolia), the species which produces the cancer-fighting substance taxol. (NatureServe, 2015)

Taxonomy

Distinct species in small but ancient genus. (NatureServe, 2015)

Historical Range

Historically, the distribution of T. taxifolia included the ravine slopes along the eastern side of the Apalachicola River from Bristol (Liberty County), FL to just across the Florida-Georgia state line, north of Chattahoochee, FL (Schwartz et al. 2000a). According to G. Nelson (2010, pers. comm.), no live trees were found in a survey conducted for the Jackson County's EO, therefore the current historical range has declined to just three counties. (NatureServe, 2015)

Current Range

Very narrow endemic, known only from ravines along east bank of the Apalachicola river in the Florida panhandle (Liberty and Gadsten Counties) and adjacent southwestern most Georgia (Seminole and Decatur Counties). Kral reports for Jackson County, Florida but basis unclear; the Florida Natural Areas Inventory has mapped one occurrence in Jackson County, but it is believed extirpated. (NatureServe, 2015)

Critical Habitat Designated

No;

Life History

Food/Nutrient Resources

Food/Nutrient Narrative

<u>Adult:</u> Growth following germination is slow; 8 - 12 year old trees are generally 6 - 8 feet tall (USFWS, 1986).

Reproductive Strategy

<u>Adult:</u> Sexual (inferred from NatureServe, 2015)

Breeding Season

<u>Adult:</u> March or April (NatureServe, 2015)

Key Resources Needed for Breeding

Adult: Wind, a period of warm and then cool temperatures, squirrels (NatureServe, 2015)

Reproduction Narrative

Adult: Torreya trees do not reach reproductive maturity until they are 15-20 years old (USFWS 1986). Reproductive structures appear in March or April (USFWS 1986), and pollen is disseminated by wind (Baker 1985). Seeds mature in September to October of the second year following pollination and, when mature, are often gathered by squirrels (USFWS 1986). Seeds germinate in one to three years (usually two), following a period of warm and then cool temperatures (USFWS 1986) (NatureServe, 2015).

Habitat Type

Adult: Terrestrial (NatureServe, 2015)

Habitat Vegetation or Surface Water Classification

Adult: Hardwood hammock slopes, ravines, and bluffs (NatureServe, 2015)

Dependencies on Specific Environmental Elements

Adult: Uninterrupted seepage, humid microclimate (NatureServe, 2015)

Environmental Specificity

Adult: Narrow (inferred from NatureServe, 2015)

Habitat Narrative

Adult: Rich, dark, sandy loam soils of hardwood hammock slopes, ravines, and bluffs. Usually in steephead ravines (deep cuts made by erosion into coastal plain sediments). The ravines are much cooler and more moist than the land surface above and harbor remnants of the more temperate flora that existed in the region during the Tertiary ice ages. Uninterrupted seepage and a humid microclimate appear to be important characteristics of the habitat (NatureServe, 2015).

Dispersal/Migration

Dispersal

Adult: Low (inferred from NatureServe, 2015).

Dependency on Other Individuals or Species for Dispersal

Adult: Squirrels (NatureServe, 2015)

Dispersal/Migration Narrative

Adult: Pollen is dispersed by wind and seeds are dispersed by squirrels (see reproduction narrative).

Population Information and Trends

Population Trends:

Decline of >90% (NatureServe, 2015)

Species Trends:

Decreasing (USFWS, 2010)

Resiliency:

Very low (inferred from NatureServe, 2015)

Representation:

Very low (inferred from NatureServe, 2015)

Redundancy:

Moderate (inferred from NatureServe, 2015)

Number of Populations:

31 (NatureServe, 2015)

Population Size:

500 - 600 individuals (USFWS, 2010)

Adaptability:

Persists for decades as stump shoots, but does not reach reproducing size before being killed back by fungus. (NatureServe, 2015)

Population Narrative:

Persists for decades as stump shoots, but does not reach reproducing size before being killed back by fungus. This species was historically locally abundant within its restricted range and habitat. There is a documented decline of 98.5% in the 100 years from 1900 to 2000 (Schwartz et al 2000). This species has experienced a long-term decline of >90%. The occupied range of this species is 92 square km. Approximately 29 occurrences have been mapped in Florida and 2 in Georgia. Fungus blight has infected all natural stands and few if any reproducing mature trees remain in the wild (NatureServe, 2015). At present, the Florida torreya population is estimated to be less than a 1,000 individuals (likely there are 500-600 trees; T. Spector 2010, pers. comm.). The species status is decreasing, based on the 2009 Recovery Data Call (USFWS, 2010).

Threats and Stressors

Stressor: Disease and predation (USFWS, 2010)
Exposure:
Response:
Consequence:
Narrative: "The Recovery Plan identified a fungal disease as one of the primary threats responsible for the species' decline. Attempts to isolate the main disease agent had failed. Currently, researchers are still puzzled as to the cause, and research is ongoing to determine or arrest the fungal infestation; therefore, this factor is a threat. Deer browsing affects small trees accounting for 46.5 % of the damage; deer rub was present on more than 50% of the 223 Torreya trees surveyed in 2008-2010. Some of these rubs were extremely severe to the cambium as to break stems or kill trees. Deer rub the main stem and could introduce disease into the vascular cambium. Therefore, stem damage caused by deer rubbing represents a threat to the T. taxifolia populations (USFWS, 2010). "

Stressor: Habitat destruction or modification (USFWS, 2010)
Exposure:
Response:
Consequence:
Narrative: Two factors impacting habitat have been speculated as potential threats: changes in soil chemistry associated with disruption of hydrology when upload topsoils were plowed in the 1950's, and perhaps fire suppression. Forest slope is highly alterred as a resulte of logging practices. Habitat continues to be altered due to logging and the plants in outplanted areas grow but eventually die; habitat alteration is a present threat (USFWS, 2010).

Stressor:
Exposure:
Response:
Consequence:
Narrative:

Recovery

Reclassification Criteria:
Not available

Delisting Criteria:
1. Ensure the preservation and appropriate management of Torreya's native habitat to allow for reintroduction (USFWS, 2010).

2. Produce cultivated plants of Torreya and conduct empirical investigations of methods to control the decline in cultivated plants (USFWS, 2010).

3. Investigate the decline to determine its cause and, if possible to find a cure. Research into the cause of the decline is ongoing. See Recovery action 2 for details (USFWS, 2010).

4. Introduce cultivated plants into secure habitat within its former range See Recovery action 7 for details (USFWS, 2010).

Recovery Actions:
- Protect the existing habitat (USFWS, 2010).
- Control the torreya decline (USFWS, 2010).
- Produce seedlings and cuttings (USFWS, 2010).
- Investigate the ecological requirements, population dynamics, and life history of Florida torreya (USFWS, 2010).
- Establish experimental collections of torreya outside its native habitat (USFWS, 2010).
- Place seed in long-term storage (USFWS, 2010).
- Reestablish Torreya in its native habitat (USFWS, 2010).

Conservation Measures and Best Management Practices:

- Build and maintain enclosures at TSP to protect the plants from deer herbivory and rubbing, and to better assess the impact of browsing on T. taxifolia. One concern is tree fall hitting the enclosure; it takes the enclosure and the Torreya plant. This is an ongoing action implemented by the TSP and now they are testing a narrow enclosure design (USFWS, 2010).
- Conduct surveys for new populations (and potentially for reintroduction) where similar habitat exists. This action can include the use of aerial photographs and/or species distribution modelling methods (e.g., Niche modelling) to initially determine potential sites, with subsequent validation or inspection of the sites for plants (USFWS, 2010).
- A plan should be developed to address guidelines for reintroduction, translocation (and/or managed relocation), and augmentation, a three-step process of planning, implementing and monitoring. Since this species is unlikely to disperse and colonize on its own because current populations are characterized by small individuals that are failing to achieve reproductive maturity, therefore it is a candidate for assistance (USFWS, 2010).

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