A PETITION TO LIST THE WHITEBARK PINE, *PINUS ALBICAULIS*, AS AN ENDANGERED SPECIES UNDER THE ENDANGERED SPECIES ACT

December 8, 2008

Acknowledgments: Cover photo from the Forest Service's website: http://www.fs.fed.us. Background material for this petition was largely informed by Tomback, D. F., S. F. Arno and R. E. Keane. 2001. *Whitebark Pine Communities*. Island Press, Washington, D.C. Susan Casey-Lefkowitz, Andrew Wetzler, Niel Lawrence and Louisa Willcox contributed to the drafting and editing of portions of this petition which was authored by Melly Reuling and Sylvia Fallon. Jesse Logan and Diana Tomback provided valuable consultation and review.

Executive summary

Introduction

Whitebark pine (*Pinus albicaulis*) is a keystone species distributed across the high-elevation western United States and southwestern Canada. The species has declined dramatically over the past fifty years as a result of the combined effects of the introduced disease white pine blister rust, a current wide-scale outbreak of mountain pine beetles, and increased competition with other species due in part to fire suppression. Global warming now poses an even more formidable threat by rapidly shifting the range of environmental conditions necessary for this high elevation, slowly-growing species. Climate change will also exacerbate the threats to whitebark pine: with warming temperatures, mountain pine beetle outbreaks will continue unabated, killing mature whitebark pine and complicating the management of blister rust. Predictions of increased severity and frequency of forest fires in the region threaten to further diminish whitebark pine and limit the species' ability to replenish numbers without active human management.

The federal Endangered Species Act ("ESA") requires the protection of a species as "endangered" if it "is in danger of extinction throughout all or a significant portion of its range." 16 U.S.C. § 1532 (12). The current extent of losses to whitebark pine populations and the combined synergistic effects of advancing succession from fire suppression, disease, and pests currently, and under future climate warming scenarios, threaten the continued existence of whitebark pine and qualify it for protection as endangered under the ESA.

Petitioner, the Natural Resources Defense Council, submits this Petition to the Secretary of the Interior and the United States Fish and Wildlife Service ("FWS") requesting formal protection of whitebark pine as an endangered species under the ESA. The ESA requires the Secretary and FWS to determine within 90 days of receiving a petition to list a species as threatened or endangered whether the petition "presents substantial scientific or commercial information indicating that the petitioned action may be warranted." 16 U.S.C. § 1533 (b)(3)(A). Such a determination is to be made solely on the basis of the "best available science." 16 U.S.C. § 1533 (b)(1)(A). Following a positive "90-day" finding, the Secretary and FWS must within one year of receipt of the petition complete a review of the status of the species and publish either a proposed listing rule or a determination that such listing is not warranted. 16 U.S.C. § 1533 (b)(3)(B). The Secretary and FWS then have an additional year to finalize the proposed rule. 16 U.S.C. § 1533 (b)(6)(A). Assuming the Secretary and FWS comply with the statutory timelines of the ESA, whitebark pine must be formally designated as an endangered species within two years of the receipt of this Petition. Critical habitat for whitebark pine must also be designated concurrently with the species' listing as endangered. 16 U.S.C. § 1533 (a)(3)(A).

This Petition describes the natural history and biology of whitebark pine and the current status and distribution of the species. The Petition also reviews the primary threats to the continued existence to the species and explains how existing law and regulations, both domestically and internationally, are inadequate to address these threats and prevent the extinction of whitebark pine. The Petition clearly demonstrates that whitebark pine meets the legal criteria for listing as endangered under the ESA and therefore the species must be protected as such.

Whitebark pine

Whitebark pine (*Pinus albicaulis*) is regarded as a foundation and keystone species for its value in promoting biodiversity, creating locally stable conditions for other species, and for modulating and stabilizing fundamental ecosystem processes (Tomback et al. 2001, Ellison et al. 2005, Tomback and Achuff in press). Whitebark pine colonizes the highest elevations in the most inhospitable sites where it forms treeline ecosystems. Here, the whitebark pine canopies shade snow and thus prolong snowmelt which greatly influences hydrological processes. The trees also stabilize soil, reducing erosion. Its seeds are an important food source to grizzly bears as well as to many small birds and mammals. A variety of species including elk, deer, and grouse also use whitebark pine communities for shelter. As an early colonizer of alpine and mountain meadow vegetation, whitebark pine creates microsites that are then colonized by subalpine fir, *Abies lasiocarpa,* and forest understory plants (Weaver 2001). The tree is a distinctive feature on the high altitude landscape and considered aesthetically pleasing. Whitebark pine is currently endangered by a combination of factors including introduced disease, outbreaks of mountain pine beetles, the direct and indirect effects of climate change and forest management practices.

These threats are summarized below:

White pine blister rust: an introduced pathogen

Whitebark pine is infected nearly rangewide by white pine blister rust, a non-native fungal disease caused by the pathogen *Cronartium ribicola*. Blister rust infection kills tree branches, reducing photosynthesis and cone production; if the disease grows into the trunk or starts in the trunk, it can girdle trees and kill them. Some mature trees with extensive canopy damage from blister rust can persist for years but are reproductively non-functional (Hoff et al. 2001). The accidental introduction of white pine blister rust into Vancouver, British Columbia, on seedlings imported from Europe in 1910 was a watershed event for western forest biodiversity (Mielke 1943, Hummer 2000, McDonald and Hoff 2001). In the past century, the blister rust has steadily spread throughout whitebark pine communities, with the exception of a few remaining interior Great Basin ranges (Kendall and Keane 2001, McDonald and Hoff 2001, Tomback and Achuff In press). The highest infection levels in whitebark pine--75 to 100%--occur in the Intermountain and Northern Rocky Mountain region of their range near the United States and Canadian border (Smith et al. 2008, Tomback and Achuff, in press). The conditions in the high dry environment of the Greater Yellowstone Area were thought to be resistant to the spread of blister rust into these whitebark pine populations (Keane and Morgan 1994). However, infection levels and mortality are now increasing in these areas as well and, by 2006, the infection rate in the Greater Yellowstone Ecosystem was reported to have increased to 20 % (Reinhart et al. 2007).

Mountain pine beetle: a native pest

In many regions, whitebark pine is additionally suffering from infestation by native mountain pine beetles (*Dendroctonus ponderosae*), which feed and reproduce under the bark of the tree, rapidly killing it. A massive outbreak of mountain pine beetle is currently underway in the western United States, and infestations are at record levels within whitebark pine communities in the Rocky Mountains (Keane and Arno 1993, Logan and Powell 2001, USDA 2004, Gibson 2006). Studies indicate that atypically warm temperatures have increased mountain pine beetle invasion by enhancing beetle survival and, in some cases, by shortening their life cycles leading to more frequent reproduction (Logan and Powell 2001, Logan et al. 2003, Gibson 2006). Recent annual aerial surveys have detected a dramatic increase in mountain pine beetle activity in northern Idaho, west-central and southwestern Montana, and the Greater Yellowstone Ecosystem (Kegley et al. 2001, Gibson 2006). Western Canadian forests are also currently experiencing the largest infestation of mountain beetle ever documented for the area, and whitebark pine mortality is widespread (Carroll et al. 2006).

Mountain pine beetle outbreaks at high elevation are rare and historically correlate with periods of warmer temperatures. From 1909 through the 1930s and again from the 1970s to the 1980s, widespread mountain pine beetle outbreaks killed white pine communities throughout the U.S. Rocky Mountains, producing “ghost forests” (Arno and Hoff 1990, and references therein, Wood and Unger 1996, Kendall and Keane 2001). The current outbreak, which is also related to higher temperatures, is predicted to increase to unprecedented levels under future climate warming scenarios (Logan and Powell 2001, Logan et al. 2003).

Climate change

Global warming is generally expected to speed the decline of whitebark pine communities through three mechanisms: (1) warming associated with global climate change will limit the geographical range of whitebark pine while increasing the range of neighboring tree species; (2) pest ranges will shift into new locations that are currently only marginally suitable climatically for infestation; and (3) hotter and drier conditions are predicted to accompany climate change thus increasing the frequency and intensity of stand-replacing wildfires.

Evidence from historic pollen records indicate that migration, and not adaptation, have been the standard response of tree species to climate change in the past (Huntley 1991). However, climate change historically occurred over much longer time scales than those currently predicted and it is therefore unlikely that trees such as whitebark pine could either adapt or migrate quickly enough to keep pace with the rapid change predicted in future climate scenarios (Davis 1989, Romme and Turner 1991, Malcolm et al. 2002, Hamman and Wang 2006, Schrag et al. 2008). The rapid rates of climate change predicted for the coming centuries, and even decades may lead to unprecedented extinction events (Bradshaw and McNeilly 1991). High elevation, slow growing and relatively immobile species like whitebark pine are likely to be the most sensitive to climate changes (Aitken et al. 2008).

Fire regimes and successional replacement of whitebark pine

Finally, fire suppression policy in the northern U.S. Rocky Mountains has led to significant changes in forest communities. While forest succession is a natural process that contributes to changes in the representation of tree species within communities, and can reduce whitebark pine densities, fire suppression practices have facilitated the rate and extent of succession in some whitebark pine forest communities, contributing to their decline. As the frequency of fire has declined, so has the number of suitable openings for seed germination in successional whitebark pine communities (Keane and Morgan 1994). Meanwhile, shade-tolerant species, such as spruce and fir, which are relatively intolerant of fire have thrived, leading to the replacement of whitebark pine (Kendall and Keane 2001, Arno 2001, Keane 2001, Murray et al. 2000). Encroachment by subalpine fir also creates multilayered canopies that, in the event of fire, precipitate higher surface fire intensities and flame lengths (Lasko 1990). This facilitates the conversion of a low to moderate severity mixed fire regime to a stand replacing, crown fire regime (Steele 1960, Loope and Gruell 1973, Arno et al. 1993).

Synergies

All of these factors work together to threaten the continued existence of whitebark pine. In some areas, fire suppression allows whitebark pine to be outcompeted by other species and increases the risk of stand replacing fires. Mountain pine beetles are already expanding their elevational range in response to warmer temperatures and favor trees that are weakened including those infected with blister rust (Six and Adams 2007). While mountain pine beetles target large, mature trees, blister rust infects all age classes of trees in areas with high infection levels leaving all whitebark pine trees vulnerable to infection and death. Finally, climate models predict range shifts for whitebark pine and the mountain pine beetle infestation which will greatly reduce the amount of land occupied by whitebark pine and increase the number of trees infested with beetles. A warming climate in the western U.S. is also predicted to bring an increase in drought conditions and changes to fire regimes. Concurrently, the widespread mortality of trees from pest and fungus infestations reduces the carbon uptake and increases future emissions from the trees' decay further fueling the process of climate change (Kurz et al. 2008). The cumulative effect of these threats qualifies whitebark pine as an endangered species under the U. S. Endangered Species Act.

Notice of Petition

The Natural Resources Defense Council ("NRDC") hereby petitions the Secretary of the Interior, through the United States Fish and Wildlife Service ("FWS"), to list the whitebark pine (*Pinus albicaulis*) as an endangered species and designate critical habitat to ensure its recovery pursuant to Section 4(b) of the Endangered Species Act ("ESA"), 16 U.S.C. § 1533(b), section 553(3) of the Administrative Procedures Act, 5 U.S.C. § 533(e), and 50 C.F.R. § 424.14(a).

NRDC is a national not-for-profit conservation organization with approximately 1-million members and activists. One of NRDC's organizational goals is to further the ESA's purpose and to preserve our national biodiversity. NRDC's members have a direct interest in ensuring the survival and recovery of whitebark pine and in conserving the unique native plant and animal communities on which they rely and which they benefit.

FWS has jurisdiction over this Petition. This Petition sets in motion a specific process, placing definite response requirements on FWS. FWS must issue an initial finding as to whether the Petition "presents substantial scientific or commercial information indicating that the petitioned action may be warranted." 16 U.S.C. § 1533 (b)(3)(A). FWS must make this initial finding "(t)o the maximum extent practicable, within 90 days after receiving the petition." *Id*. A petitioner need not demonstrate that listing is warranted, rather, a petitioner must present on information demonstrating that such a listing *may* be warranted. While NRDC believes that the best available science demonstrates that listing whitebark pine as endangered is in fact warranted, the available information clearly indicates that listing the species may be warranted. As such, FWS must promptly make a positive finding on the Petition and commence a status review as required by 16 U.S.C. § 1533 (b)(3)(B).

Respectfully submitted this 8th day of December, 2008.

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I. Natural History and Biology of Whitebark Pine

Whitebark pine (*Pinus albicaulis*) is an important high elevation species that is long-lived and well adapted to the harsh, inhospitable environment that exists near mountain tops and on exposed ridges (Tomback et al. 2001). On these harsh sites, whitebark pine is often the only tree species that is capable of growing, and it may be the starter of tree islands at treeline (Tomback and Resler 2008). After a disturbance whitebark pine are often the first trees to become established and they provide shade, break the wind, and otherwise mitigate harsh conditions and help facilitate succession, pioneering the way for other tree species to grow. Many wildlife species depend on whitebark pine for food and shelter. The seeds of whitebark pine trees provide a critical food source for the grizzly bear, as well as Clark's nutcracker, pine squirrels and black bears.

A. Taxonomy and Classification

Pines are classified in kingdom Plantae, division Spermatophyta, subdivision Gymnospermae, order Coniferales (Harlow et al.1979, Price et al. 1998). Whitebark pine is one of more than a hundred species of pines. It is classified in the family Pinaceae, genus *Pinus,* subgenus *Strobus* (Critchfield and Little 1966, Price et al. 1998). The Strobus pines include several taxa of white pines that have five needles per fascicle. They are also knows as the haploxylon pines, having a single fibrovascular bundle per needle and as the 'soft' pines, referring to their softer wood. The five-needled pines are taxonomically diverse and have been classified into various subsections. Recent phylogenetic studies of pines based on gene sequences of nuclear ribosomal DNA and chloroploast DNA place whitebark pine in section *Quniquefoliae* and subsection *Strobus* (Liston et al. 1999, Gernandt et al. 2005).

B. Lifecycle

Whitebark pine trees are long lived, with many over 500 years old. Some individuals have been recorded to live over 1200 years (RMTRR 2006). They first reach sexual maturity between 20-30 years of age although large cone crops do not usually occur until the trees are at least 60-80 years old (Day 1967, Krugman and Jenkinson 1974). The reproduction process for whitebark pine may take up to five years to successfully complete including cone production, seed dispersal and seed germination.

The first stage of cone and seed initiation and maturation can take more than two years (Lanner 1990). Seed and pollen initiation occurs around the time of winter bud formation, from mid-July through mid-September depending on elevation, latitude and climate including temperature and precipitation (Schmidt and Lotan 1980). Cone buds overwinter and develop the following year beginning in April. Pines are monoecious with male and female cones on the same tree (Krugman and Jenkinson 1974). Male cones release pollen between May and August (Schmidt and Lotan 1980). The pollen is shed and wind-dispersed with most pollen fertilizing nearby trees; however, long distance dispersal has also been documented (Shuster et al. 1989, Latta and Mitton 1997). Female cone scales open during the pollination period. The female cones remain on the tree for further growth and development while the male cones fall off after pollen release.

The second phase of regeneration involves seed dissemination primarily via the dispersal and caching of seeds by Clark's nutcrackers (See also section E. Whitebark pine dispersal and Clark's nutcracker). Several other species, such as Stellar's jay (*Cyanocitta stelleri*), deer mice (*Peromyscus maniculatus*) and chipmunks (*Tamias* spp.), may also cache whitebark pine seeds and contribute to a small percentage of regeneration. However many birds and small mammals consume and thereby destroy whitebark pine seeds (McCaughey and Tomback 2001). Furthermore, whitebark pine seeds lack a terminal, membranous wing that would allow for wind-dispersal. Therefore, whitebark pine is highly dependent on Clark's nutcrackers for dispersal. A whitebark pine cone contains an average of 75 seeds (Krugman and Jenkinson 1974). Nutcrackers can carry more than 100 seeds in a throat pouch (Tombak 1978). They 'scatter-hoard' the seeds by burying many small caches 2-3 cm deep in the forest soil. The nutcrackers return to these caches throughout the year to consume seeds, but those that are not consumed by nutcrackers or other seed foragers are available to germinate.

After adequate seed production and dispersal by Clark's Nutcrackers, the seeds form a soil seed bank and germination of the seeds may be delayed for one or more years, in part because of underdeveloped embryos at the time of seed dispersal. Whitebark pine shows a soil seed bank strategy that is unique among pines and allows for seedling recruitment in the absence of seed production, seedling recruitment in most years where seeds lie in moist microsites, and high densities of germination and recruitment greater than 2 years after large to moderate cone production (Tomback et al. 2001).

C. Distribution

Whitebark pine's distribution is limited to the high mountains of western North America (Figure 1). While not contiguous, whitebark pine forests extend longitudinally between 107 and 128 degrees west and latitudinally between 37 and 55 degrees north (McCaughey and Schmidt 1990, Ogilvie 1990). Whitebark pine distribution is divided into western and eastern ranges that are connected by an area in southern British Columbia and northeastern Washington where a few isolated stands occur.

The western range extends southward from the coastal ranges of the Bulkley Mountains in northern British Columbia through the Blue and Wallowa Mountains of northeastern Oregon and Cascades, Coastal Ranges, and Sierra Nevada, ending in southeastern California as far south as the Kern River. There are some outliers in the Warner and Siskiyou Mountains in California (Critchfield and Little 1966, Ogilvie 1990) and small isolated stands occur in mountain ranges in northeastern California, south-central Oregon, and northern Nevada (Arno and Hoff 1990).

The eastern range of whitebark pine is distributed throughout the northern Rocky Mountains of British Columbia, Alberta, Idaho, Montana, and Wyoming. The northern extent of the eastern range occurs along the Continental Divide of the Rocky Mountains just below the 54^{th} latitude, northeast of McBride, British Columbia. The southern extent of the eastern range consists of isolated occurrences in small mountain ranges of northeastern Nevada. The Yellowstone ecosystem of Wyoming, Idaho and Montana is home to extensive stands of whitebark pine and contiguous stands extend into the eastern ranges of the Salt River and Wind River mountain ranges of Wyoming (Thompson and Kuijt 1976). The Wind River Range of Wyoming represents the eastern-most extension of the species (Schmidt 1994, McCaughly and Schmidt 2001).

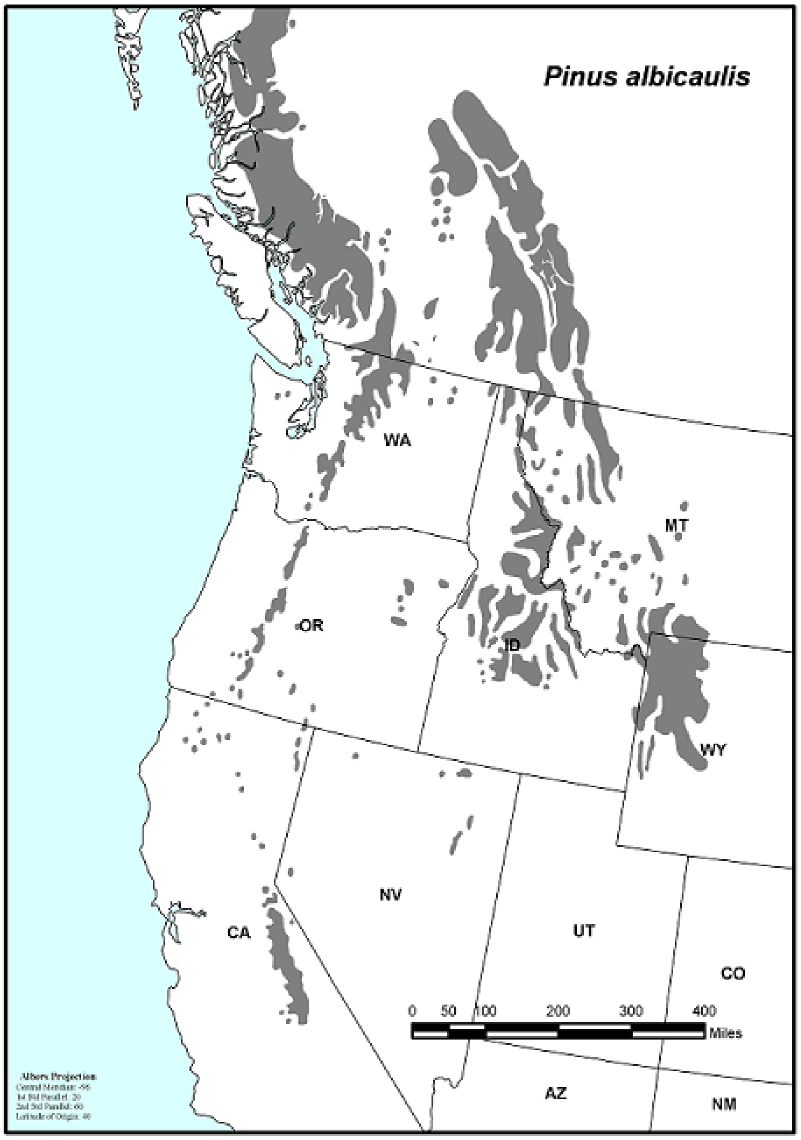

Figure 1. Distribution of whitebark pine (*Pinus albicaulis*). Range map produced by USDA Forest Service, Forest Health Protection Technology Enterprise Team.

Within its geographical range, the upper elevational limits of whitebark pine, and of alpine timberlines, decreases with increasing latitude. Whitebark pine occurs as high as 3,050 to 3,660 meters (10,000 to 12,000 feet) in the Sierra Nevada and 2,590 to 3,200 meters (8500 to 10500 ft) in western Wyoming. It occurs as low as 900 meters (2950 ft) in the northern limits of its range in British Columbia (Arno and Hoff 1990, Ogilvie 1990).

Except perhaps in the snowiest regions, whitebark pine is typically found in the alpine treeline and extends downward in elevation into associations with a variety of other conifers, usually lodgepole pine (*Pinus contorta)*, Englemann spruce (*Picea englemannii)*, subalpine fir (*Abies lasiocarpa)* and mountain hemlock *(Tsuga mertensiana)* in the Rocky mountains, and Sierra lodgepole pine (*Pinus contorta murrayana)* in the Sierra Nevada and the Cascade and Blue Mountains.   Other associated tree species include interior Douglas-fir *(Psuedotusuga menziesii* var. *glauca),* limber pine (*Pinus flexilis*), alpine larch (*Larix lyallii),* Rocky Mountain juniper (*Juniperus scopulorum),* California red fir (*Abies magnifica)*, western white pine (*Pinus monticola*) and foxtail pine (*Pinus balfouriana*) (McCaughey and Schmidt 1990).

About 98% of the range of whitebark pine in the United States is on public lands, including national parks, wilderness areas, and national forests: Whitebark pine communities are found in twenty-five national forests in the northern U.S. Rocky Mountains alone and within all western high elevation national parks except for Rocky Mountain National Park (R. E. Keane, unpublished data, cited in Tomback et al. 2001). In Canada, the majority of whitebark pine also occurs on public lands, both federal and provincial, including extensive protected areas (parks, wilderness areas, ecological reserves).

D. The Whitebark Pine Environment

Whitebark pine, like other stone pines, is adapted to a particular high elevation climate. Below is a summary of the whitebark pine environment with respect to the factors most likely to influence its persistence and health: climate, soil, and substrate.

1. Climate

Although the temperature at tree line is nearly constant over whitebark pine's range, there are many other variables that affect its occurrence, such as photoperiod, wind, ultraviolet radiation, and snow depth (Billings 2000, Koerner 1998).[1]

Temperature: Above the snow, whitebark pine ecosystems experience average max/min temperatures of -5°/-14° C in January (Weaver 1990, 1994). Absolute minimum temperatures are around -34° C. In contrast, organisms under deep snow such as microbes, insects, and mammals, and plant roots on all but windswept sites, function at near 0° C because snow insulates against the loss of heat stored by the soil in the previous summer (Seligman 1939, Sirucek et al. 1999). Below ground there is minimal temperature change year round. For example, soil temperatures at 50 cm in whitebark

[1] The presence of whitebark pine on various sites is described in this Petition by an average condition and condition limits from all the long-term weather stations located in whitebark pine forests between central California and central Alberta (Weaver 1990, 1994).

pine and subalpine fir stands rise from 1°C in mid-May to 9°C in mid-August and fall again to 4°C in October (Sirucek et al. 1999).

Summer temperatures are cool in whitebark pine environments with average maximum/minimum July temperatures around 18°/4° C. Absolute maximums are about 29°C (Weaver 1990, 1994). Summer temperatures increase downslope from an average July maximum of 19°C in the whitebark pine ecosystem to 30°C in the foothills and plains below. The average July maximum of whitebark pine ranges in the middle of the July maximum for the five other stone pines.

Water: Precipitation in whitebark pine environments ranges from 600 to 1,600 mm/year. Over 85 % of this falls in winter and accumulates as 170 to 300 cm of snow (Weaver 1990) which contributes important runoff for streams, rivers and irrigation throughout the spring and summer. Stone pine ecosystems rarely experience soil drought. Soils are saturated in spring and precipitation exceeds evaporation which is less than about 2 mm per month (Weaver 1994a).

Wind and radiation: Winds at high elevation flow faster than at lower elevations because they are not slowed by friction against the ground. When winds meet mountains, a complex of accelerating and slowing forces usually results in some slowing, but with maintenance of the vertical profile (Barry 1992). The average daily solar radiation values in the whitebark pine region are among the lowest in the United States in fall and winter; however, summer values approach those over deserts of the Southwest (Weaver 2001). Summer radiation varies little over the altitudinal gradient (Barry 1973).

2. Soils and Substrates

Soils of the whitebark pine ecosystem are most often classified as cryochrepts, which are cold-climate soils with light colored (leached) surface horizons and minimally developed subsurface horizons (Hansen-Bristow et al. 1990). Typically, high altitude soils have very little clay and therefore they have low water-holding capacity. Any clay that does form is typically exported to lower ecosystems in running water. Clays and silts that do exist in the alpine and whitebark pine zones were probably deposited as loess. Typically, organic carbon contents are high in tundra, decline through the alpine and subalpine zones and rise again in the grasslands. Given this, the water storage capacity of whitebark pine soils, 2.1 centimeter in the top decimeter, is higher than most other high-altitude ecosystems of the northern Rocky Mountains (Weaver 2001).

In addition to soil characteristics, the composition of substrate also has a marked influence on whitebark pine distribution. In particular, the presence or absence of limestone influences whitebark pine distribution. In most of Montana and to the south whitebark pine avoids limestone (Weaver and Dale 1974). However, in wetter areas near and north of the Canadian border whitebark pine appears mostly on limestone (Arno and Weaver 1990). This pattern of avoiding limestone at the core of their range but favoring it in the north is also true of other stone pine species in Europe and Asia. Well drained substrates (e.g. limestone) exclude whitebark pine in dry areas, such as the central Rocky

Mountains. In wetter areas, limestone in the Canadian Rocky Mountains and pumice in the Cascade Range, exclude competing trees by allowing "excess" water to drain off and therefore support whitebark pine.

E. Whitebark pine dispersal and Clark's Nutcrackers

Whitebark pine engages in a mutualistic relationship with its primary disperser, the Clark's nutcracker, (*Nucifraga columbiana,* Family Corvidae; Figure 2). This nearly crow-sized grey, black and white bird lives throughout the coniferous mountain forests of western North America. Clark's nutcrackers use fresh and stored pine seeds year-round and possess several morphological and behavioral adaptations for this life history. The nutcracker bill is long, pointed and sturdy, enabling the bird to open both unripe and ripe whitebark pine cones by severing and ripping off the cone scales (Tomback 1978). This action requires considerable effort by the bird; nutcrackers often remove the cones from some conifers, such as limber pine, and use the crotch of a tree or a rock or log as an anvil to support the cone while they remove cone scales. Pine squirrels (*Tamiasciurus spp.)* compete with the nutcrackers for whitebark pine seeds by cutting and caching the cones in middens for a winter food supply (Tomback 1978, Siepielski and Benkman 2007). While some seeds from these cone-caches may remain available for germination the following spring, the caching of cones (rather than seeds) inhibits the reproductive potential of seeds in the cones thereby limiting the role that squirrels might play as dispersers (Tomback 1982, Hutchins and Lanner 1982).

Figure 2. Clark's nutcracker, *Nucifraga columbiana.* Source: U.S. Fish and Wildlife Service.

Clark's nutcrackers and whitebark pine have a mutually beneficial interaction that has resulted from reciprocal selection pressures (Tomback 1982, Abrahamson 1989). Whitebark pine is actually an obligate or nearly obligate mutualist of Clark's nutcracker

(Tomback and Linhart 1990), which is considered unusual in plant-seed disperser systems (Wheelwright and Orians 1982).

Clark's nutcrackers are the main dispersers of whitebark pine seed; they remove the seeds from the ripe but indehiscent cones of whitebark pine and bury thousands of seeds per bird throughout the forest. Nutcrackers determine the pioneering status of whitebark pine following a disturbance in the forest and influence the population genetic structure of whitebark pine on local and regional scales (Tomback 2001).

Seed dispersal by nutcrackers provides whitebark pine with two major advantages in recolonizing large-scale burns over its wind-dispersed, shade-tolerant competitors, subalpine fir and Engelmann spruce. Nutcrackers have been recorded to disperse whitebark pine seeds up to 12.5 kilometers from the seed source which is much farther than wind-dispersed seeds can usually travel (Tomback 1978). In addition, nutcrackers routinely disperse seeds against prevailing winds, which often limit the dispersal of wind-dependent conifer seed (Tomback et al. 1990). In western Montana, for example, a burn in the Sapphire Range consumed 11,350 acres of pine, fir, and spruce habitat. Seed sources for these trees were on the east side of the burn, opposite of the prevailing winds. A study found that whitebark pine regeneration density in the burn area, a quarter century after the fire, was an order of magnitude greater than the density of either fir or spruce regeneration (Tomback et al. 1993).

F. Whitebark Pine as a foundation and keystone species

Whitebark pine is considered both a foundation and a keystone species, meaning that its presence structures a community by creating locally stable conditions for other species, and by modulating and stabilizing fundamental ecosystem processes and species diversity (Tomback et al. 2001, Ellison et al. 2005). In addition to regulating moisture and soil conditions, for example, whitebark pine facilitates forest succession and provides a critical food source for wildlife.

1. Regulates runoff and reduces soil erosion

Whitebark pine is important in regulating runoff and reducing soil erosion, especially at the highest elevations in the upper subalpine and alpine treeline communities, where whitebark pine is better able to tolerate harsh sites than other forest trees. Most precipitation in whitebark pine communities is in the form of snowpack that melts off between the months of May and July. Soils at high elevation do not have much water holding capacity and as a result 35-60% of the annual precipitation becomes runoff (Farnes 1989). The presence of trees such as whitebark pine, at the highest elevations that conifers grow, slows the progression of snow melt, which reduces spring flooding at lower elevations and results in higher stream flows later in summer. The broad, open canopies of whitebark pine, compared to narrow canopies of other conifers, shade larger surface areas and further reduce the rates of snowmelt. These same traits also reduce soil erosion. Slower, more protracted runoff is less damaging to unstable soils. The roots of whitebark pine trees stabilize soil and take up water, further reducing erosion. Because of the roles that whitebark pine plays in regulating runoff and stabilizing soil, low-elevation aquatic systems benefit from high-quality water (Farnes 1990). More importantly, with protracted run-off the downstream forests and valley bottomlands, which are preferred for pastureland and farming, receive an uninterrupted flow of water throughout the growing season.

2. Facilitates succession

Whitebark pine is also a pioneer species in that it is often the first to colonize an area after disturbance such as fire; this rapid dispersal is the result of seed caching by Clark's nutcrackers. Its early presence then provides microenvironments of shade, moisture and shelter from wind that facilitates the establishment of other conifers and understory vegetation (e.g. Lanner 1980). Several traits contribute to whitebark pine's ability to colonize disturbed areas. Whitebark pine seedlings are exceptionally hardy and can withstand drought and exposed sites more than other conifers (Arno 1986, Tomback 1986, 1994). After a fire, the environmental conditions are extremely harsh with charred soil that is directly exposed to solar radiation. The high soil surface temperatures that result can kill many seedlings in their first year. Whitebark pine seedlings, however, are hardy enough to tolerate these types of harsh conditions (Tomback et al. 1993, Tomback 1994, Tomback et al. 2001). Additionally, as mentioned above, Clark's nutcrackers cache whitebark pine seeds in open areas, thereby providing whitebark pine with an advantage over wind dispersed seeds in disturbed areas. Newly germinated whitebark pine seedlings also develop deep taproots quickly, allowing them to become readily established (Arno and Hoff 1990, McCaughey and Tomback 2001). In addition to creating a more hospitable setting for other seedlings to become established, once whitebark pine has colonized a region, they may further act as "nurse" trees by facilitating the survival and growth of other conifer species. For example, subalpine fir trees growing on harsh subalpine sites without a neighboring whitebark pine were found to experience significantly lower growth rates than fir trees growing near whitebark pine (Callaway 1998). Thus, due to a combination of unique traits, whitebark pine is often the first species to pioneer disturbed areas becoming established quickly and facilitating the growth of other vegetation in the area.

3. Food for wildlife

Whitebark pine is a critical food source not only for Clark's nutcracker, but for many wildlife species including other birds, small mammals as well as bears. In particular, whitebark pine seeds have several features that make them a valuable bear food. They are large and therefore more energetically rewarding, and the nutrients are less perishable compared to other bear food. Whitebark pine seeds weigh an average of 180 milligrams per seed, compared to 3-13 milligrams for the seeds of species such as subalpine fir, Englemann spruce, and lodgepole pine (McCaughey et al. 1986). They are also a rich source of dietary fat. For example, the average 30% to 50 % of fat content in pine seeds is only exceeded in the Yellowstone ecosystem by the adult army cutworm moth, certain ant colonies and meat of ungulates in the fall after they have accumulated body fat through the summer months (Craighead et al. 1982, Lanner and Gilbert 1994, Mattson et al. 1999). Fat from pine seeds is efficiently converted to adipose tissue, or body fat, and promotes survival and reproduction of female grizzly bears that rely on adipose reserves not only to hibernate, but also to support lactation (Hellgren 1998).

Bears obtain pine seeds almost exclusively from cone caches made by squirrels because whitebark pine cones rarely spontaneously fall from trees before the seeds are consumed or harvested by nutcrackers. Red squirrels perform a critical function for bears by harvesting cones and burying them in large concentrated caches or "larder hordes" contained in squirrel middens. Bears search out these caches, excavate the whitebark pine cones, and then carefully break away the cone scales with their claws and extract the seeds with their tongues (Kendall 1983). Use of the caches by bears typically does not occur until late August or early September and persists until the bears den, usually in late October or early November. If any cones are left in the squirrel middens, bears will resume consuming them after hibernation in June until the seeds are depleted (Kendall 1983, Mattson and Reinhart 1994).

Whitebark pine seeds are consumed in large quantities by brown and grizzly bears wherever their geographical ranges overlap. Grizzly bears consume the greatest number of whitebark pine seeds in the continental portion of their range which includes the Greater Yellowstone Area and on the eastern front of the Montana Rocky Mountains (Mattson et al. 1999). In fact, grizzly bears in the Yellowstone area obtain one-quarter to two-thirds of their net digested energy from pine seeds depending on the relative abundance of pine seeds and alternative high-quality foods such as meat from ungulates or trout (Mattson et al. 1999). When grizzly bears feed on pine seeds in this area, they feed on virtually nothing else (Mattson and Reinhart 1994).

There is a strong relationship between whitebark pine seed crop size and grizzly bear demographics and survival in the Greater Yellowstone Area. Adult female grizzlies that eat more pine seeds have more surviving cubs than females who eat fewer pine seeds (Mattson 2000). When pine seed crops are large, bears are found in whitebark pine stands – far from most humans. When seed crops are small, bears tend to forage for alternative natural foods such as clover or yampa roots that occur at lower elevations closer to human facilities (Mattson et al. 1992). During the years when pine seeds are scarce, conflicts with humans escalate dramatically as does the death rate among bears (Blanchard 1990, Mattson et al. 1992, Blanchard and Knight 1995). As a result, during the years when Yellowstone's grizzly bears are intensively using pine seeds, the population increases, whereas during the years when they are not, the population declines (Mattson et al. 1992, Mattson 1998, Pease and Mattson 1999). In this way the availability of whitebark pine seeds is closely linked to the survival of grizzly bears in the Greater Yellowstone Area.

II. Status of Whitebark pine

The patterns and trends of whitebark pine decline are documented in the second half of this Petition according to the threats that qualify the species for endangered species protections. Several provincial, national and international organizations have assessed the status of whitebark pine either globally or regionally. Those assessments are presented here.

Whitebark pine is classified as vulnerable by the International Union for the Conservation of Nature (IUCN) Redlist meaning it is "facing a high risk of extinction in the wild in the medium-term future" due to a combination of threats including habitat loss/degradation, land management, invasive alien species and pathogens/parasites (Conifer Specialist Group 1998).

NatureServe, a non-profit conservation organization, is a leading source for information about rare and endangered species. NatureServe maintains a list of threatened and endangered organisms worldwide that it ranks according to the species rarity and severity of threats. NatureServe assigns whitebark pine a global rank of G4, meaning the species is considered "uncommon but not rare with some cause for long-term concern due to declines or other factors." At a subnational scale, for the state of Wyoming, NatureServe assigns whitebark pine a rank of S3 meaning it is "(v)ulnerable in the nation or state/province due to a restricted range, relatively few populations (often 80 or fewer), recent and widespread declines, or other factors making it vulnerable to extirpation" (NatureServe 2008).

Several subgroups within Alberta's government have assessed the status of whitebark pine. The Alberta Natural Heritage Information Centre (ANHIC) ranks the species as S2 defined as "6 to 20 or fewer occurrences or with many individuals in fewer locations. May be especially vulnerable to extirpation because of some factor of its biology" (Gould 2006). Alberta Sustainable Resources Development maintains a list called "The General Status of Alberta Wild Species." Whitebark pine is classified on this list as "May be at Risk" defined as "Any species that may be at risk of extinction or extirpation, and is therefore a candidate for detailed risk assessment" (Alberta Sustainable Resource Development 2007). In 2007 the status report, *Status of the Whitebark Pine in Alberta,* was produced and will be used to "assess the status of the species in Alberta and to guide development of prospective management and conservation strategies" (Alberta Sustainable Resource Development 2007).

In 2008 the British Columbia Ministry of Environment's Conservation Data Center adjusted the conservation ranking of whitebark pine, adding it to the *blue-list,* a list of special conservation concern in British Columbia. This species was uplisted due to a "severe negative long-term trend expected from mountain pine beetle infections, the white pine blister rust epidemics, climatic warming trends, and successional replacement" (Campbell 2008).

Finally, the Western Washington Fish and Wildlife Office of the U.S. Fish and Wildlife Service lists whitebark pine as a "species of concern"(WWFWO 2007).

III. Whitebark pine meets the definition of an endangered species under the Endangered Species Act

The Endangered Species Act, 16 U.S.C. § 1531-1534, allows for the protection of any species of fish, wildlife or plant. Whitebark pine, *Pinus albicaulis*, is a plant. Whitebark pine was officially named and described by Engelmann in 1863. Its taxonomy and validity as a species is uncontested (Flora of North America Editorial Committee 1993). It therefore qualifies as a "species" under the ESA. 16 U.S.C. §1531. Petitioners seek protection for the species throughout its range.

The U. S. Fish and Wildlife Service ("FWS") is required to list a species for protection if it is in danger of extinction or threatened by possible extinction in all or a significant portion of its range, 16 U.S.C. 1533 (a)(1). In making this determination, the FWS must rely "solely on the best scientific and commercial data available" and analyze the species' status in light of five statutory listing factors:

A) the present or threatened destruction, modification, or curtailment of its habitat or range;
B) over utilization for commercial, recreational, scientific or educational purposes;
C) disease and predation
D) the inadequacy of existing regulatory mechanisms;
E) other natural or manmade factors affecting its continued existence

16 U.S.C. § 1533(a)(1)(A)-(E); 50C.F.R. § 424.11(c)(1)-(5).

Whitebark pine is endangered by four of the five listing factors: present modification of its habitat, disease and predation, the inadequacy of existing regulatory mechanisms, and other natural or manmade factors.

Whitebark pine communities have declined dramatically over the past fifty years throughout most of their range due to a combination of advancing succession from fire suppression which has resulted in widespread replacement of whitebark pine by other species, the introduced fungus white pine blister rust (*Cronartium ribicola)*, and a large-scale outbreak of mountain pine beetle infestation (*Dendroctonus ponderosae)*. Global warming, which exacerbates some of these problems, also poses a threat to the distributional limits of this species. Finally, there are currently no regulatory mechanisms to slow, halt, or reverse mortality of whitebark pine.

As described below, the threats to whitebark pine are multifaceted, current and compounding. The species has already suffered significant, widespread losses, including ecological extinction within several regions in the northern Rocky Mountains and

Intermountain region. "Ecological extinction" means that a species is reduced to the point that "its effects on the other species in its community are negligible" (Primack 2002). Urgent action, including listing under the ESA, is needed now to ensure that whitebark pine does not become extinct. As described below, whitebark pine meets the definition of an endangered species under the ESA.

IV. Criteria for Listing a Species as Endangered or Threatened under the Endangered Species Act

A. Modification of whitebark pine habitat

One of the threats to whitebark pine is the continued alteration of its high alpine habitat as the result of fire suppression policies in the United States and Canada. In the absence of fire, whitebark pine forests experience succession – a process by which early colonizing, 'pioneer,' species such as whitebark pine are slowly displaced by competing species. While succession is a natural process, fire disrupts this process by creating openings for the repeated establishment of early colonizers like whitebark pine thereby leaving a mosaic of both early and late successional forests across the landscape. The expansive scope of fire suppression policies in the United States has inhibited this mosaic of early and late successional sub-alpine forests allowing much of the area occupied by whitebark pine to transition to late succession thus favoring the exclusion of whitebark pine for other more shade tolerant species.

Fire is an essential element for whitebark pine communities. Whitebark pine is well adapted to frequent, light to moderate burns which create openings for the pine to regenerate, but also has historically experienced infrequent stand-replacing burns in many of its seral communities (Arno 2001, Larson 2005). High-intensity, stand-replacing fires in dense subalpine fir-spruce forests often allow whitebark pine to become established as a result of seed caching by the Clark's nutcracker. After establishment these seral whitebark pine communities are perpetuated by low-intensity fires that kill understory fir and spruce (Arno and Hoff 1990.) Not only are there now fewer regeneration sites for whitebark pine, but fire suppression and natural late seral advance has led to an increase of competing, shade tolerant species growing under the whitebark pine.

The relatively low severity and low return interval of fires in whitebark pine communities is primarily a result of low fuel loads and duff depths in the whitebark pine forests. Whitebark pine communities on favorable sites depend on occasional fire and other disturbance for renewal. Whitebark pine regenerates early after disturbance. Because whitebark pine seeds are transported and cached by Clark's nutcrackers (*Nucifraga columbiana),* whitebark pine often becomes established in disturbed areas before or simultaneously with wind-dispersed species (Tomback et al. 1990, 1993, 2001). In addition, the whitebark pine seedlings are hardier and can survive better than wind-dispersed conifers such as spruce and fir. Clark's nutcrackers typically cache whitebark pine seeds 1 to 3 kilometers from harvest sites; however, they are capable of carrying whitebark pine seeds up to 22 kilometers which is 100 times farther than most wind-dispersed seeds of spruce and fir travel (Tomback and Linhart 1990) giving whitebark pine an advantage as a colonizing species when low severity scattered native fire regimes occur (Tomback et al. 1990).

In the past century, however, fire suppression policies have altered this natural pattern. In general, fire suppression has resulted in widespread successional advancement leading to an abundance of understory vegetation. This vegetation allows shade-tolerant species, such as sub-alpine fir, that are more tolerant of fire than whitebark pine, to encroach and out-compete whitebark pine in areas where fire suppression policies are in place (Arno and Hoff 1989, Arno 2001). As a consequence, fir species have come to dominate much of the upper subalpine forest, as well as the understories of many seral (early successional and shade intolerant) whitebark pine communities throughout the tree's range (Keane and Morgan 1994).

In the Bitterroot Mountains of west-central Montana, for example, subalpine fir increased from 10 % to 42 % during 1900 to 1995 above 2,100 meters elevation, while whitebark pine decreased from 32 % to 17 % (Hartwell 1997). High elevation landscapes in the Big Hole Range of Montana and Idaho were composed of lodgepole and whitebark pine communities from the mid 1700s to the early 1900s, but they are now becoming dominated by subalpine fir and spruce (Murray 1996). In high altitude forests of Washington and Oregon, advanced forest succession has allowed mountain hemlock (*Tsuga mertensiana*) to dominate at the expense of seral whitebark pine communities (Arno and Hoff 1990).

Encroachment by subalpine fir can also increase the intensity of fires by creating multilayered canopies that have low crowns and higher crown bulk densities because of higher leaf areas. As a general matter, higher biomass combined with slow decomposition rates at a high altitude can result in fuel accumulations that precipitate higher surface fire intensities and flame lengths (Lasko 1990). The higher flame lengths coupled with lower and thicker subalpine fir crowns hasten the transition of a surface fire to a crown fire. Crown fires tend to be larger, more intense and more difficult to control than surface fires (Keane et al. 1996) and can cause increased mortality in even relatively fire tolerant species, such as whitebark pine. Therefore, the long term consequence of fire suppression in the whitebark pine ecosystem is the successional replacement and the conversion of a low to moderate severity mixed fire regime to a stand replacing, crown fire regime (Steele 1960, Loope and Gruell 1973, Arno et al. 1993). It should be noted, however, that while these more intense fires may pose a risk to whitebark pine trees, the return of a more natural pattern of frequent to occasional forest fires is likely to have a beneficial effect on whitebark pine communities as a whole, assuming that a healthy seed source is available for whitebark pine regeneration.

Finally, in addition to reducing its ability to compete with fir species, fire suppression induced overgrowth can inhibit whitebark pine seed germination. Overgrowth allows less light and fewer nutrients to reach the forest floor, so the success and growth of cached seeds is impaired (Keane and Morgan 1994).

B. Disease and predation

Whitebark pine is currently being devastated by the combination of an introduced fungal disease, white pine blister rust, and a native pest, the mountain pine beetle.

1. Blister Rust: An Introduced Pathogen

White pine blister rust is a disease that affects the five-needled white pines, a group that includes whitebark pine, and totals nine species in the U.S. and Canada, and additional species in Mexico. In infected trees, the disease forms sporulating cankers, which can cut off the flow of water and nutrients, severely damaging or killing the tree. The disease is caused by an introduced fungus. All native North American five-needled white pines are highly susceptible to the fungus, and blister rust has made a dramatic impact on white pine forest ecosystems, including both commercial and non-commercial timber species, since its introduction to the continent from the 1890's.

a. History of the Invasion of Blister Rust

White pine blister rust was first observed in 1854 by H. A. Dietrich in the Baltic provinces of Russia on several types of *Ribes*, a genus of flowering plants including many types of currants, and on introduced eastern white pine (*Pinus strobus)* (Spaulding 1911). Dietrich did not recognize that the fungus was the same on both *Ribes* and pines. The realization that the rusts on the *Ribes* and the pine were a single species was not confirmed until 1888 when cross-inoculation tests were performed (Spaulding 1911). By 1900, blister rust had spread throughout northern Europe where susceptible pine stands were intermingled with patches of cultivated European black currant (*Ribes nigrum).* The abundance of both hosts in close proximity provided an ideal environment for rapid spread of the fungus, and the destructive abilities of blister rust were quickly determined.

Spaulding (1911) listed some of the disaster statements that several northern European foresters made during this time, which demonstrate how quickly the rust could destroy its pine host. In 1854, Dietrich observed that blister rust killed most eastern white pine in a few years in the Baltic provinces of Russia. In Finland in 1876, foresters found that new infections in thirty-year-old eastern white pine caused rapid mortality. In Denmark in 1883, it was reported that blister rust caused more damage than any other bark-inhabiting rust. By 1901, infection was so severe on eastern white pines in Belgium that foresters considered it imprudent to continue managing the pine species.

When a North American market developed for seedlings, European tree nurseries began to grow additional eastern white pine seedlings for export (Spaulding 1922). European pathologists had been warning foresters since the 1890's to prevent the transportation of seedlings around Europe and especially to North America without thorough inspection for the disease. Infections of the stem of the seedlings were easily identified and it was assumed that these inspections were sufficient to control the spread of the disease. However, a major problem was undiscovered at that time. The original site of infection on pines is not actually on the stem, but on the secondary needles, primary leaves and cotyledons. The discovery that yellow spots observed on the needles were the first sign of blister rust and that stem infections were not visible until a year or more after the original infection was made much later (Clinton and McCormick 1919). Many seedlings infected with blister rust were labeled “uninfected” and shipped to North America.

In North America in the early 1900’s, foresters, nurserymen and pathologists were all well aware of blister rust and were on the lookout for infections on imported seedlings (Spaulding 1922). Blister rust was found in several places on the East Coast, and an unsuccessful campaign to eradicate *Ribes* and infected trees was organized. In the fall of 1921, blister rust was found in Vancouver, British Columbia on European black currant (Mielke 1943). Across the border in Washington State the rust was found on European black currant and on two planted white pines in a Mt. Vernon nursery. In 1922, Canadian officials inspected all white pine imported between 1904 and 1914 and found infected white pines that had been imported in 1910. Blister rust was first documented on whitebark pine in North America in 1926 in the coast range of British Columbia and it subsequently spread to northern Idaho stands of whitebark pine by 1938 (Childs et al. 1938).

The fungus moved rapidly. Back-dating of canker age (Lachmund 1933) on western white pine showed the fungus spread southward almost to the Columbia River by 1920, into southern Oregon by 1929, and halfway down the Sierra Nevada range of California by 1941. It spread eastward on western white pine into Idaho by 1923 and on into northwestern Montana by 1927. The rust reached whitebark pine growing on the continental divide in Glacier National Park in 1939 (Mielke 1943). The current extent of blister rust includes nearly the full range of whitebark pine (Figure 5; McDonald and Hoff 2001, Tomback and Achuff in press).

b. Life Cycle and Biology

White pine blister rust is a disease caused by the fungus, *Cronartium ribicola*, which has a complex life cycle that requires two hosts, a five-needled white pine and, most commonly, a currant or gooseberry plant of the genus *Ribes*. More recently, Indian paint brush (*Castillija spp.)* and a lousewort (*Pedicularis spp.*) have been identified to be alternate hosts as well (McDonald et al. 2006). All species of white pine are susceptible to infection at all ages, although a small number of resistant trees do exist. Bingham (1983) estimated that only 1 in 10,000 western white pine trees showed resistance in high infection areas; however, a survey conducted between 2002-2004 found some level of rust resistance in 48% of 108 seed sources sampled across the US range of whitebark pine (Mahalovich et al. 2006). Seedlings and young trees are often more easily infected and die more quickly as a result of infection.

Generally, white pine blister rust spores germinate on the plant surface and grow into the pine through the stomatal openings in the needles or through a wound. Once inside the pine needle, the fungus grows down into the twig and the branch and ultimately to the main stem of the tree. The fungus kills the tree’s cambium, which transports nutrients, causing a canker to form which can girdle the stem preventing water and nutrients from passing through the canker area; as a result, the distal portion of the twig, branch or stem dies (Figure 3). If the canker forms on the main stem, it will cause topkill which prevents cone and seed production, and often ultimately kills the tree.

An infected branch will often develop a characteristic swelling and after a year or more, the rust forms spores that are contained in blister-like sacks that erupt through the bark of the twig or stem. When the blisters rupture they release bright orange colored spores which are carried by wind and infect the alternate host, *Ribes* (most commonly gooseberry or currant plants) in the spring. These spores then mature during the summer months on the underside of *Ribes* leaves and give rise to the final basidiosproes which germinate under conditions of high humidity and become windborne again infecting new pine trees (McDonald and Hoff 2001).

Blister rust is a mild annual disease occurring on the leaves of *Ribes*, but the *Ribes* shrubs drop their leaves in the fall. Therefore, rust infection of *Ribes* requires a source of infection each spring from the infected pines. The fungus usually continues to grow in whitebark pine until the tree dies or, on rare occasions, overcomes the infection. Large trees may have dozens of separate infections (cankers) from a year of favorable conditions for basidiospore formation, or from exposure to rust over multiple years. Each year an infected tree lives, it has the potential to produce spores that infect *Ribes* and thereby perpetuate the disease (Kendall and Keane 2001).

Figure 3. White pine blister rust cankers (*Cronartium ribicola)*. Source: US Forest Service.

Where *Ribes* shrubs are abundant, and when summer weather allows many cycles of multiplication of spores, high levels of blister rust can accumulate. If a large portion of these spores mature and reach the pine needles, the result is a “wave” of new infections in pines. During wave years, rust infection spreads into new areas and intensifies in previously infected stands (Kendall and Keane 2001, McDonald and Hoff 2001). The frequency of wave years of infection in whitebark pine ecosystems is highly variable depending on elevation, geographical region, topographical setting and wind patterns. While the colder temperatures of higher elevations tend to inhibit spore development, these areas generally have higher precipitation and humidity which favors the spread of blister rust (Kendall and Keane 2001)

It can take years for the disease to kill a large tree but the blister rust can kill small trees within just a few years. Smaller trees, seedlings, and saplings are killed more quickly because there is a shorter distance for the fungal hyphae to travel from the infected needles to the main stem, and there is a smaller cambium circumference to girdle in the main stem and branches. Most infected seedlings die within three years (Hoff and Hagle 1990, Tomback et al. 1995, Kendall and Keane 2001). Red needles on mature trees are referred to as "flags" and usually indicate that the tree is infected with blister rust (Figure 4). Typically these trees remain alive for decades after becoming infected with the fungus and act as a host for the disease. Cone production typically ceases long before the tree actually dies as the tree usually dies from the top, where cones are produced, down. Trees are weakened by loss of photosynthetic biomass, as branches die. Blister rust thus threatens multiple aspects of the regeneration process by not only reducing available seed but also by causing seedling mortality.

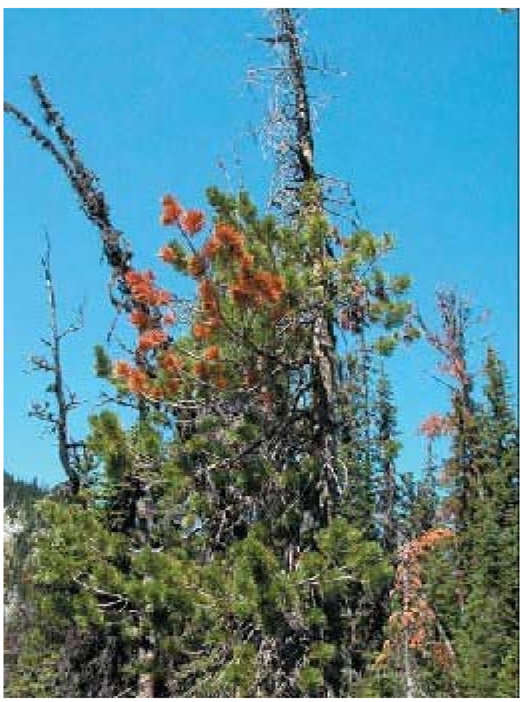

Figure 4. Red needles indicative of blister rust infection. Source: USDA Forest Service.

c. Blister Rust and Whitebark Pine Decline

White pine blister rust has led to extreme declines in whitebark pine throughout its range. Blister rust has been detected almost everywhere whitebark pine has been examined, with the exception of interior Great Basin ranges. Site specific surveys provide an overview of blister rust infections throughout the whitebark pine range. Below is a summary of published regional infection assessments. The most recent compilation of blister rust infection surveys is presented in Figure 5.

i. Rocky Mountains

The northern Rocky Mountains generally show a high level of blister rust infection and extensive whitebark pine mortality. The highest mortality from blister rust in the Rocky Mountains is in northwestern Montana, northern Idaho and the southern Canadian Rockies, where a quarter to half of all whitebark pine trees were already dead by the late 1990s (Keane et al. 1994, Kendall et al. 1996a, Stuart-Smith 1998). Of the remaining live trees in this region 80 to 100 % were infected with blister rust and will eventually die (Kendall et al. 1996a, Campbell 1998, Stuart-Smith1998, Smith et al. 2008).

Similarly, for a survey of the Bob Marshall Wilderness Complex in Montana, field results show that 83% of the 2,503 sampled whitebark pine trees were infected with blister rust in 1994 (Keane et al. 1994). Two hundred miles farther south in the Greater Yellowstone Ecosystem the mortality rate was reported at 10 % (Kendall et al. 1996). A 2004 study found that blister rust was found on 71 % of transects performed indicating the disease is widespread and expanding. Of 1,012 trees examined on these transects, 22.8 % were found to be infected with blister rust (Shanahan et al. 2005). By 2007, the overall infection rate in the Greater Yellowstone Ecosystem was reported at 20% (Reinhart et al. 2007), although the infection levels on the western and northern sides of the GYE are considerably higher.

In the Targhee National Forest on the west side of the Greater Yellowstone Ecosystem blister rust is present in all but one of 21 stands sampled and on average 49 % of the trees sampled were infected. On many sites between Monida Pass and Targhee Pass 70 to 100% of the trees are infected (Kendall 1994, Smith and Hoffman 2000). In 2006 four areas in the south and east of the Greater Yellowstone Ecosystem were surveyed by Bockino (2007) who found "roughly one half of the whitebark pine sampled in these areas were dead, 70% had been attacked by mountain pine beetle and 85% had at least one blister rust symptom." Over all, blister rust infection is intensifying in the subalpine zone of the Greater Yellowstone Area and this trend is expected to continue, especially in light of possible favorable conditions being created by climate change (see also "Range shifts of disease and pathogens" section C.2.).

Finally, in the moist northern half of Idaho, whitebark pine mortality and rust infection level are higher than the drier south-central mountains of Idaho. In the Salmon and Selway Rivers region, 50 % of the trees larger than 10 cm in diameter are dead and blister rust is present in 34 % of the stands sampled (J.T. Hogg, unpublished data, Craighead wildlife-wildlands Institute, Missoula, Montana, in Tomback 2001). Spore loads are so heavy that small seedlings are being infected, impacting regeneration in large burns (Tomback et al. 1995). Ground surveys conducted in the Selkirk Mountains in 2000 documented a loss of 45 to 82 % of the whitebark pine to mountain pine beetle and from 33 to 87 % of the remaining trees had blister rust (Kegley et al. 2001).

ii. United States Coastal Ranges

The presence of blister rust is also a significant source of mortality and decline in whitebark pine populations in the northwest United States. In 2000, the presence of whitebark pine was surveyed along the Pacific Crest National Scenic Trail on the Umpqua National Forest where it occurred on 76 % of the transects. Forty-six percent of the trees examined were infected by blister rust, and 10 % were dead. Two-thirds of the mortality was due to white pine blister rust (Goheen et al. 2002). In a 2005 regional summary of the "Status of Whitebark Pine on National Forest Lands in Washington and Oregon," sixty-nine sites across nine National Forests and one National Park were surveyed. Blister Rust incidence ranged from 0.0 to 100 % with a mean of 41.5 %. Overall mortality ranged from 0.0 % to 89.4 % with a mean of 33.4%. The highest mortality from blister rust was observed in the Mt Hood National Forest with 47% dead trees (Shoal et al. 2005).

Using field data of blister rust infection to create a spatially explicit model, scientists now calculate a 94% chance that whitebark pine within Mt. Rainier National Park will approach extinction in approximately 150 years (Ettle and Cottone 2004). Since whitebark pine trees live to 500 years on suitable sites, the decline is predicted to occur in less then one generation (Arno and Hoff 1990). Additionally, the massive decline and possible extinction of whitebark pine in this area could occur even more rapidly since cone production decreases before trees actually die (DelPrato 1999).

In the Sierra Nevada Mountains of California, blister rust infection in whitebark pine ranges widely. A 2002 survey of white pine blister rust on high elevation white pines in California established 44 whitebark pine survey plots and found that statewide, blister rust occurred on 11.7% of the plots and infection rates ranged from 0 – 71% (Maloney et al. 2002). In the Bald Mountains, north of Reno, Nevada and near the summit of Mt. Rose, west of Reno, more than 50 % of the whitebark pine trees sampled were infected with rust (Smith and Hoffman 2000).

iii. British Columbia and Alberta

Blister rust infection levels in Canada are also quite high. In 2004, the status of whitebark pine in British Columbia was summarized in a report presented at the 16th International Conference on Ecological Restoration (Smith et al. 2008). The report was based on 2003 surveys, conducted to identify blister rust in whitebark pine between Glacier National Park, Montana, and McBride, British Columbia. Active blister rust cankers were found in 88% of the plots, and overall 95% of the plots were infected with active and inactive cankers. Nineteen percent of whitebark pine trees were dead form all causes (Smith et al. 2008). In the northern Canadian Rocky Mountains (Jasper National Park and adjacent areas of British Columbia) Smith et al. (2008) found the overall infection in live trees to be 44% (range 4-88) in 26 stands sampled. In the central Rockies the rate of infection was 20% (range 0-73) in 26 stands sampled. In the southern Rockies (south of Banff National Park to and including Glacier National Park, Montana) the rate of infection was 62% (range 0-100) in 95 stands sampled.

Whitebark pine mortality in Alberta was measured in eight plots in Waterton Lakes National Park that were established by Kendall (2003) in 1996 and re-measured in 2003. Mortality was found to have increased from 26% to 61% in seven years (Smith et al.2008). This represents a yearly mortality rate over twice (5% compared to 2.1%) the rate reported by Keane and Arno (1993) in western Montana.

Blister rust alone is reducing whitebark pine populations at an alarming rate. For the past century the disease has steadily spread throughout the range of whitebark pine and now the incidence of mortality is steadily increasing in all areas sampled. Early predictions that the disease would be stopped by more severe climates in the extremes of whitebark pine range have proven false. Blister rust has spread to nearly the full range of whitebark pine and, as illustrated below, the entire range of all associated North American white pines is either occupied or threatened (Tomback et al. 2001).

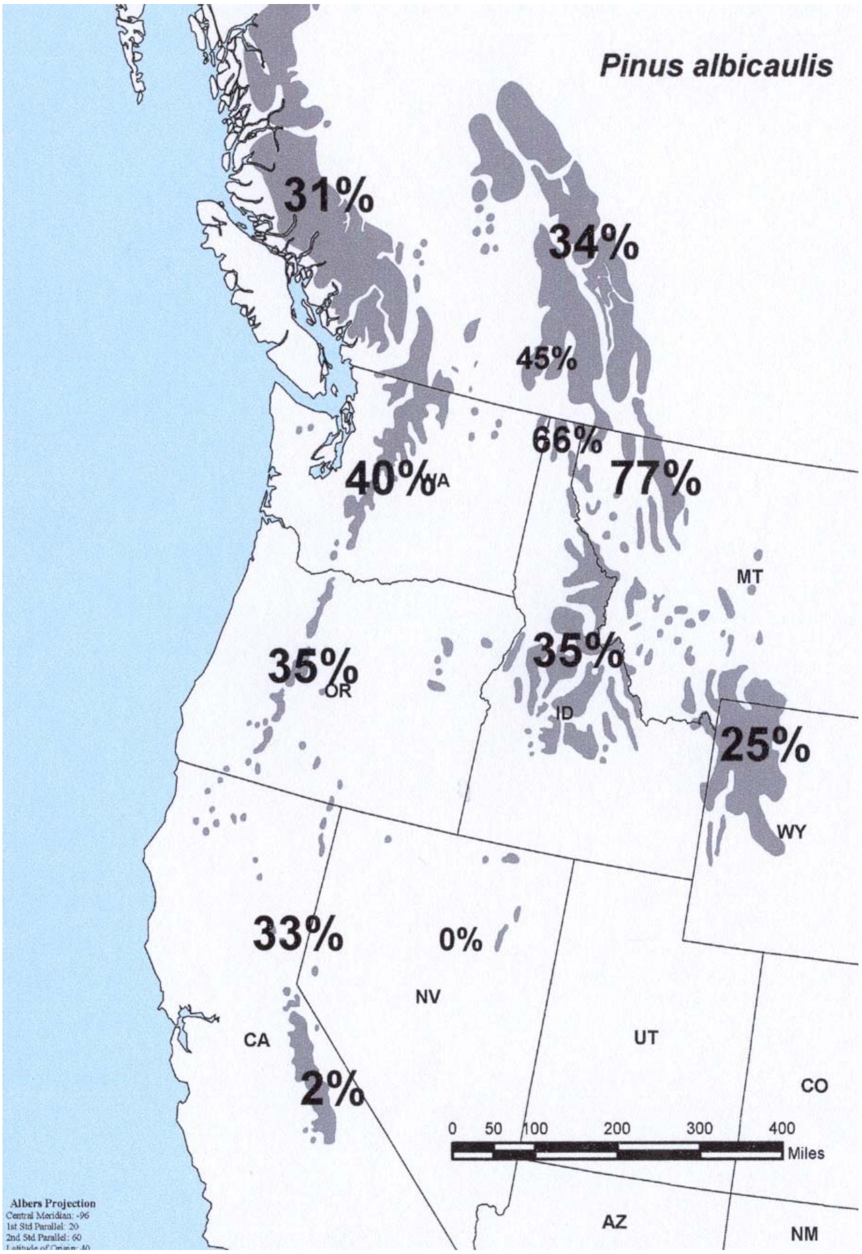

Figure 5. Distribution of whitebark pine and regional infection levels of white pine blister rust (percent of living trees infected). Range map produced by USDA Forest Service, Forest Health Protection Technology Enterprise Team. Regional blister rust values compiled from independent assessments by government agencies in the U.S. and Canada. Data available from D. F. Tomback.

2. Mountain Pine Beetle: A Native Pest

Whitebark pine forests are also suffering heavy mortality from a native pest, the mountain pine beetle (*Dendroctonus ponderosae*). While, historically, climatic conditions in high elevation whitebark pine habitats have prevented sustained mountain pine beetle outbreaks, today anthropogenic global warming appears to be allowing outbreak populations to expand into these previously inhospitable areas (Logan and Powell 2001). The result is an alarming intensification of beetle activity in vulnerable whitebark pine forests. (See also "Range shifts of disease and pathogens" section C.2.)

Mountain pine beetles usually hit larger trees and weakened trees first. In the Umpqua National Forest in the Cascade Mountains, for example, mountain pine beetles alone accounted for 13 % of the whitebark pine mortality. Another 18% of dead whitebark pines showed evidence of both mountain pine beetle and white pine blister rust (Goheen et al. 2002). The fact that beetles are commonly found on more trees with blister rust than without suggests that blister rust infection likely facilitates the spread of the pine beetles (Keane et al. 1994, Kendall and Keane 2001). During an epidemic, however, all mature trees in a stand can become victims and, in contrast to blister rust, mountain pine beetle does not primarily kill seedlings and young trees. The combination of blister rust with a mountain pine beetle outbreak can thus be particularly damaging since (1) the beetles can kill the few rust - resistant mature trees; (2) the rust will kill any seedlings that remain or become established after a beetle invasion; (3) beetles will favor trees that have been weakened by blister rust; and (4) both the pest and fungus impact mature trees and therefore seed production (Keane et al. 1994, Kendall and Keane 2001).

a. Life Cycle and Biology

The mountain pine beetle (*Dendroctonus ponderosae*) is a native insect that has coevolved as an important ecological component of western pine forests (Figure 6). Beetle kill is usually a natural disturbance process that benefits forest ecosystems by opening the canopy and providing snags for cavity nesting birds.

Figure 6. Mountain pine beetle, *Dendroctonus ponderosa.* Source: USDA Forest Service.

The beetle develops through four stages: egg, larva, pupa, and adult. Except for a short period in the summer when adults may emerge and fly to new trees, all stages are spent under the bark of infested trees (Amman et al. 1990). While the life cycle of the beetle is typically one year, this varies with elevation and temperature. For example, at high elevations where summer temperatures are cool, two years may be required to complete the beetle's life cycle. By contrast, two generations have been recorded in one year in warmer climates, such as those found in low-elevation sugar pines (*Pinus lambertiana*) in California (Amman et al. 1990).

Under the bark, female beetles construct straight, vertical egg galleries. Packed with boring dust, these galleries are mostly in the phloem, or inner bark, although they slightly score the sapwood. The galleries range from 4 to 48 inches (10 to 122 cm) long, averaging about 10 inches (25 cm). Females lay tiny, pearl-white eggs in niches along the sides of the galleries, usually during the summer and early fall. The eggs hatch in 10 to 14 days, although they may take longer during cool weather. Sometimes eggs are also laid in late spring by females that survived the winter. These surviving females may either reemerge and re-attack trees or merely extend their egg galleries (Amman et al. 1990, Logan and Powell 2001).

Once hatched, the legless larvae are white with brown heads. The larval stage lasts for about 10 months over the winter. The larval broods feed on the phloem and construct their own galleries that extend at right angles to the egg galleries. When mature, larvae excavate oval cells in which they turn into pupae. By July, the pupae usually have been transformed into adults.

Adults feed within the bark before they emerge; when several feeding chambers coalesce, adults occur in groups. One or more beetles will then make an exit hole from which several adults will emerge. Within one or two days after emerging, the beetles will attack other trees and the cycle begins again (Amman et al. 1990). Unmated female beetles making the first attacks release chemicals called aggregating pheromones. These pheromones attract males and other females until a mass attack overcomes the tree. Adjacent trees are then infested.

The trees respond to pine beetle attacks by increasing their resin output in order to discourage or kill the beetles, but attacking beetles carry with them the spores of blue-staining fungi on their bodies and in a special structure on their heads. The fungi blocks the trees resin response, aiding the beetles in overcoming the tree. As the fungi develop and spread throughout the sapwood, they also interrupt the flow of water to the crown of the tree. Usually within two weeks of attack, the trees are overwhelmed as the phloem layer is damaged enough to cut off the flow of water and nutrients. In the end, the trees starve to death, and the damage can be easily seen even from the air in the form of reddened needles. For this reason, the foliage of entire groves of attacked trees will appear reddish after an outbreak (Figure 7).

Figure 7. Mountain pine mortality from mountain pine beetle in Yellowstone National Park. Source: USDA Forest Service

Unlike white pine blister rust, bark beetles do not primarily affect the viability of young seedlings or saplings. Instead, beetles prefer dense stands of older and larger trees, which are often the primary cone producing trees within high elevation sites. When these trees die, reduced seed production can impact regeneration capabilities and wildlife that depend on those seeds as a vital food source. Moreover, mature trees that have already been infected with blister rust and are compromised are easier for mountain pine beetles to infect (Keane et al. 1994, Kendall and Keane 2001, Six and Adams 2007).

b. Regional mountain pine beetle infestations

As with data on blister rust, bark beetle infections are monitored at local to regional scales. Below we present information on local surveys by region of mountain pine beetle outbreaks in whitebark pine. An assessment of the death of whitebark pine within its US range due to the combined effects of blister rust and mountain pine beetle is presented as a time series, aerial detection survey from 1999-2007 in Figure 9a and b.

i. Yellowstone Ecosystem

For the decade prior to 2000, only minor beetle-caused mortality was recorded in most of the Greater Yellowstone Ecosystem (Gibson 2006). Although aerial detection survey data for Greater Yellowstone have only been recorded during the past 4-5 years, "on the ground" plot surveys documenting beetle activity between the years 2000-2004 are considered representative of mortality throughout the area (Gibson 2006). In twenty plots near Avalanche Peak in Yellowstone National Park, whitebark pine killed by mountain pine beetle within the past 2-3 years averaged 96 trees per acre or 80% of the whitebark pine over 5 inches in diameter (breast height) (Gibson 2006). Near Lightening Lake in Gallatin National Forest, mortality in ten plots for the past 3 years averaged 162 whitebark pine per acre, or 74% of the whitebark over 5 inches. In 2005, aerial detection survey data documented the highest recorded level of whitebark pine mortality in the Greater Yellowstone region. In Yellowstone National Park and surrounding forests, there are approximately 1,064,600 acres of whitebark pine-dominated forests. About 171,200 or 16% of those contain mountain pine beetle caused mortality. Nearly 720,000 "faders," or trees that were attacked the previous year, were also recorded in that year alone (Gibson 2006).

ii. Selkirk Mountains of Northern Idaho

Whitebark pine stands in the Selkirk Mountains, Idaho Panhandle National Forests, have been steadily declining for several years due to white pine blister rust and mountain pine beetle (Kegley et al. 2004). Within the past few years, the mountain pine beetle outbreak has significantly increased the rate and amount of mortality. Ground surveys conducted in three areas in the fall of 2000 documented a loss of 45-82% of the whitebark pine primarily due to mountain pine beetle. From 33%-87% of the remaining green trees in those areas were infected with blister rust symptoms (Kegley et al. 2001).

iii. Washington and Oregon

A 2000 survey of whitebark pine at Crater Lake National Park found 20% infected with blister rust. The authors predict a 46% loss of the park's whitebark pine by 2050 (Murray and Rasmussen 2000). In a 2005 regional summary of the "Status of Whitebark Pine on National Forest Lands in Washington and Oregon," 69 sites across nine national forests and one national park were surveyed. Over 10,500 individual whitebark pine trees were observed and the mountain pine beetle incidence ranged from 0.0% to 34.3% with a mean of 4.5%. Overall mortality, which combines deaths from mountain pine beetle and blister rust, ranged from 0.0% to 89.4% with a mean of 33.4%. While mountain pine beetle mortality has been lower along the coastal, maritime region of whitebark pine's range, infection levels have nonetheless increased across the tree's entire U.S. range within the last several years (Figure 8; Schwandt 2006).

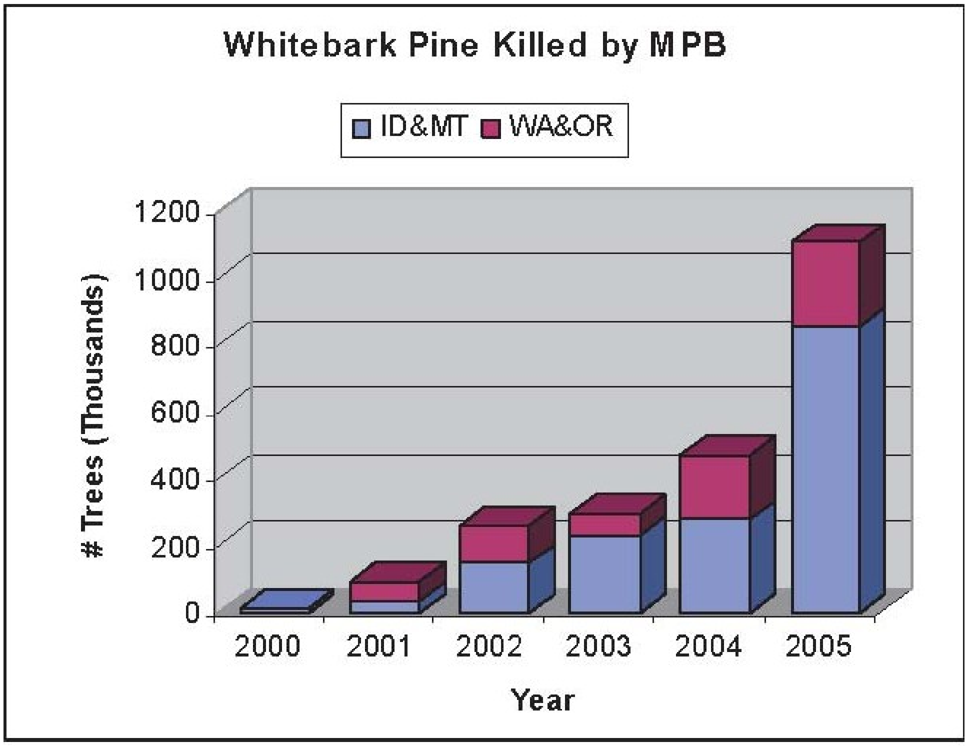

Figure 8. Whitebark pine mortality recorded during annual aerial surveys in Idaho, Montana, Washington and Oregon (Schwandt 2006).

Aerial detection survey (ADS) is a tool for documenting dead and dying trees; however, the precise cause of mortality can be difficult to establish without on the ground confirmation. ADS data from 2000-2005 across Washington, Oregon, Idaho and Montana combined shows an increase in the number of whitebark pine deaths presumably due primarily to mountain pine beetle outbreaks though some of these deaths may also be due to blister rust (Figure 8; Schwandt 2006). In another time series comparison, ADS data for whitebark pine mortality across its entire U.S. range was evaluated every year since 1999. Figures 9a and b graphically represent the cumulative mortality of whitebark pine over a 9 year period (1999-2007). The cumulative mortality is primarily due to a combination of blister rust and mountain pine beetle, but includes death from any cause.

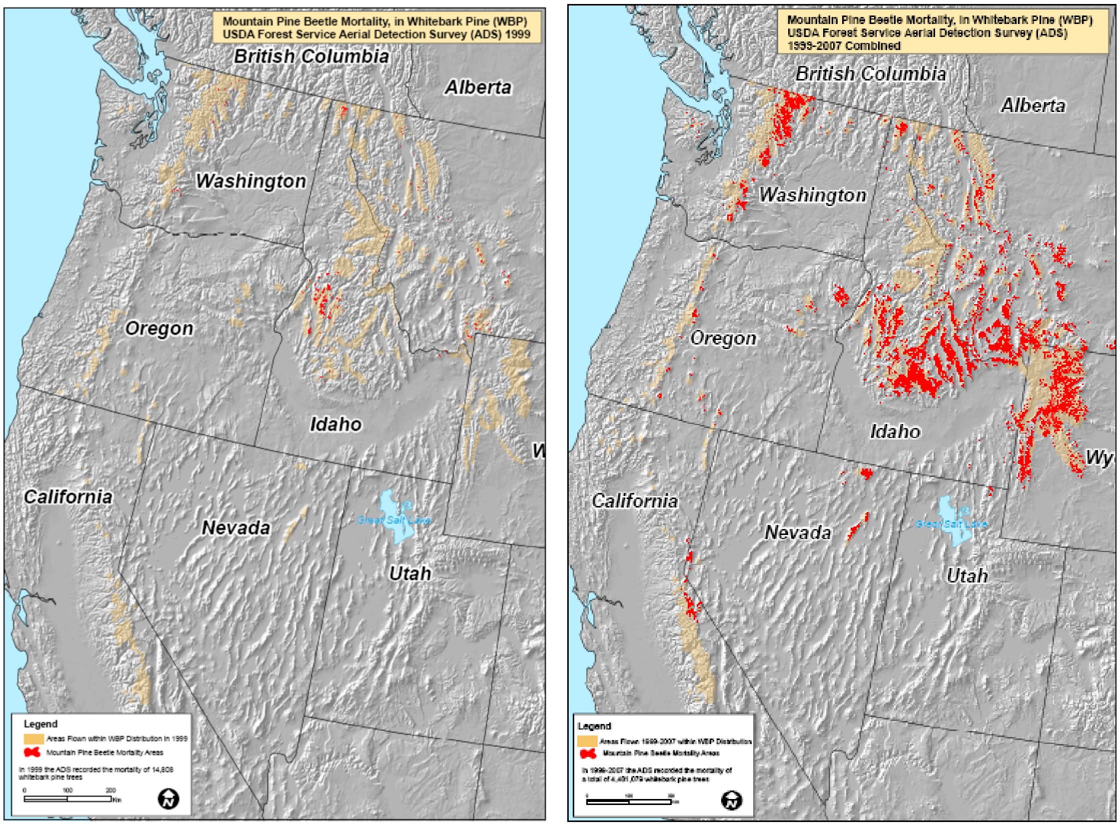

Figure 9 a and b. Aerial detection survey data of whitebark pine mortality in 1999 (a) and combined deaths from 1999 through 2007 (b). Data from USDA Forest Service.

iv. British Columbia and Alberta

Lodgepole pine forests in British Columbia are currently experiencing the largest outbreak of mountain pine beetle in recorded history (Carroll et al. 2006). Mountain pine beetle populations and subsequent damage have also steadily increased in the Canadian Rocky Mountains of Alberta. Considering the extent of the current outbreak in low elevation forests, we can expect that a massive decline of whitebark pine in British Columbia is imminent (Cambell et al. 2008) Infestations have been aided by a series of warm winters and extensive availability of susceptible mature pine forests. Several thousand trees were infested in Banff National Park in 2002. The mountain pine beetle spread eastward into the Canmore area in 2001 and several hundred trees were infested in 2002. Since 1999, mountain pine beetle infestations in the southern Wilmore Wilderness area have steadily grown and several patches of infested trees exist in the western part of Jasper National Park (Langor 2003). Thus, although a survey of losses of whitebark pine in British Columbia in 2002 proved minor (Zeglan 2002), there is significant data indicating that pine beetle outbreaks are rapidly expanding in Canada.

3. Conclusion

Whitebark pine is currently afflicted by both an introduced fungus and an epidemic outbreak of mountain pine beetles. White pine blister rust, which was introduced to the northwestern United States over a century ago, has slowly established a strong and pervasive presence across nearly the entire range of whitebark pine with infection levels steadily rising and currently ranging up to a regional mean of 77%. Mountain pine beetles are a native pest, but have reached population levels now believed to be unprecedented due to a combination of favorable conditions. Fire suppression policies and advancing succession have led to a more continuous forest landscape that allows the beetles to progress across undisturbed swaths of land. Additionally, the mountain pine beetles have expanded their range into higher elevation ecosystems due to warmer climate conditions (see also "Range shifts of pathogens and pests" section C.2). Finally, blister rust infections weaken the defense systems of whitebark pine making them more vulnerable to mountain pine beetle invasions. The extent of these invasions varies regionally, but affects up to 80% or more of whitebark pine trees in some areas, such as parts of the Greater Yellowstone Ecosystem. White pine blister rust and mountain pine beetles together are decimating whitebark pine communities. As detailed in the following section, climate change in the future is predicted to intensify the effects of blister rust and mountain pine beetle in whitebark pine ecosystems.

C. Other natural or manmade factors affecting its continued existence

Climate change now poses one of the most significant threats to whitebark pine. Not only are whitebark pine populations on the decline from blister rust and mountain pine beetle, climate change will reduce the bioclimatically-suitable range for whitebark pine in the U.S. Human-induced changes to the earth's natural greenhouse gasses are projected to result in global warming of 1.1 ° C to 6.4 ° C in the 21st century (IPCC 2007). If the change is beyond 1.5-2.5 °, significant ecological impacts are predicted to occur, making climate change fundamental to the discussion of ecological consequences for the earth's habitats (IPCC 2007). The rate of climate change predicted to occur over the next century is an order of magnitude several times greater than the average since the last glacial maximum. Therefore, it is likely that many species will not be capable of successfully tracking and adapting to such changes through migration to more suitable habitats (Davis and Botkin 1985, Malcolm et al. 2002, Aitken et al. 2008). In addition, human-caused fragmentation of our current landscapes will inhibit and in some cases prevent species migration to more suitable regions as a response to change (Bartlein et al.1997).

Species that will be most vulnerable to climate change will be large, long lived species that have very specialized habitats and limited mobility such as trees (Malcolm et al. 2002, Lenoir et al. 2008). Furthermore, trees that have late sexual maturity and occur at high elevations, such as whitebark pine, are least likely to be able to adapt or migrate in response to climate change (Aitken et al. 2008). According to the U.S. Forest Service, global warming alone may result in an over 90% reduction in the U.S. distribution of whitebark pine by 2100 (Warwell et al. 2007). Additionally, because of its synergistic effects on pine beetle infestation and the spread of blister rust, global warming also will exacerbate these other significant threats to the species. Finally, global warming is likely to be accompanied by changes to local fire regimes in whitebark pine habitat that could also negatively impact the species.

1. Range Shifts of Whitebark Pine

The response of vegetation to global warming is likely to be complicated with varying outcomes; however, it is often described as a general shifting of species' ranges northward, combined with upward elevational movement toward currently cooler areas, and recent studies are beginning to document such patterns (Aitken et al. 2008, Lenoir et al. 2008). The specific response of whitebark pine to climate change has been modeled under a variety of scenarios that all predict dramatic range reduction.

Romme and Turner (1991) applied several models of climate change that included a 1-5ºC rise in temperature and varying levels of moisture and found that the zone of whitebark pine in the Yellowstone ecosystem will rise at least 500 meters. Since the area of a band decreases when it is pushed up a cone, such a warming scenario would cause the area of whitebark forest to shrink by 90 %. Another study that applied the Canadian Climate Center general circulation model (CCC GCM) to the Yellowstone region found that whitebark pine was the tree species that would be most affected by climate change and would demonstrate a significant range retraction (Bartlein et al. 1995). A similar study concluded that whitebark pine will be reduced to less than 10% of its current range in Yellowstone National Park (Mattson and Reinhart 1994).

More recently, Hamman and Wang (2006) predicted the climate response of forest communities in British Columbia and found that whitebark pine is expected to lose habitat faster than it is able to gain habitat and is therefore predicted to "rapidly decline in frequency." Their model predicts a 98% decline by the year 2085. Schrag et al. (2008) studied the distribution of a number of treeline conifers in Yellowstone National Park and found that temperature and temperature-related variables were most influential on whitebark pine. Their models predicted that whitebark pine is likely to disappear from the park under certain climate scenarios. Additionally, because these models do not also incorporate the effects of blister rust or mountain pine beetles, the authors point out that the predicted mortality rates for whitebark pine are likely to be much higher than predicted by their climate models alone.

Finally, the USDA Forest Service has developed a bioclimatic model for predicting the occurrence of whitebark pine from its climate profile. Results indicate a rapid and large scale decline of the species' climate profile with estimates that whitebark pine will diminish to less than 3% of its current U.S. distribution by the end of the century (Warwell et al. 2007; Figure 10).

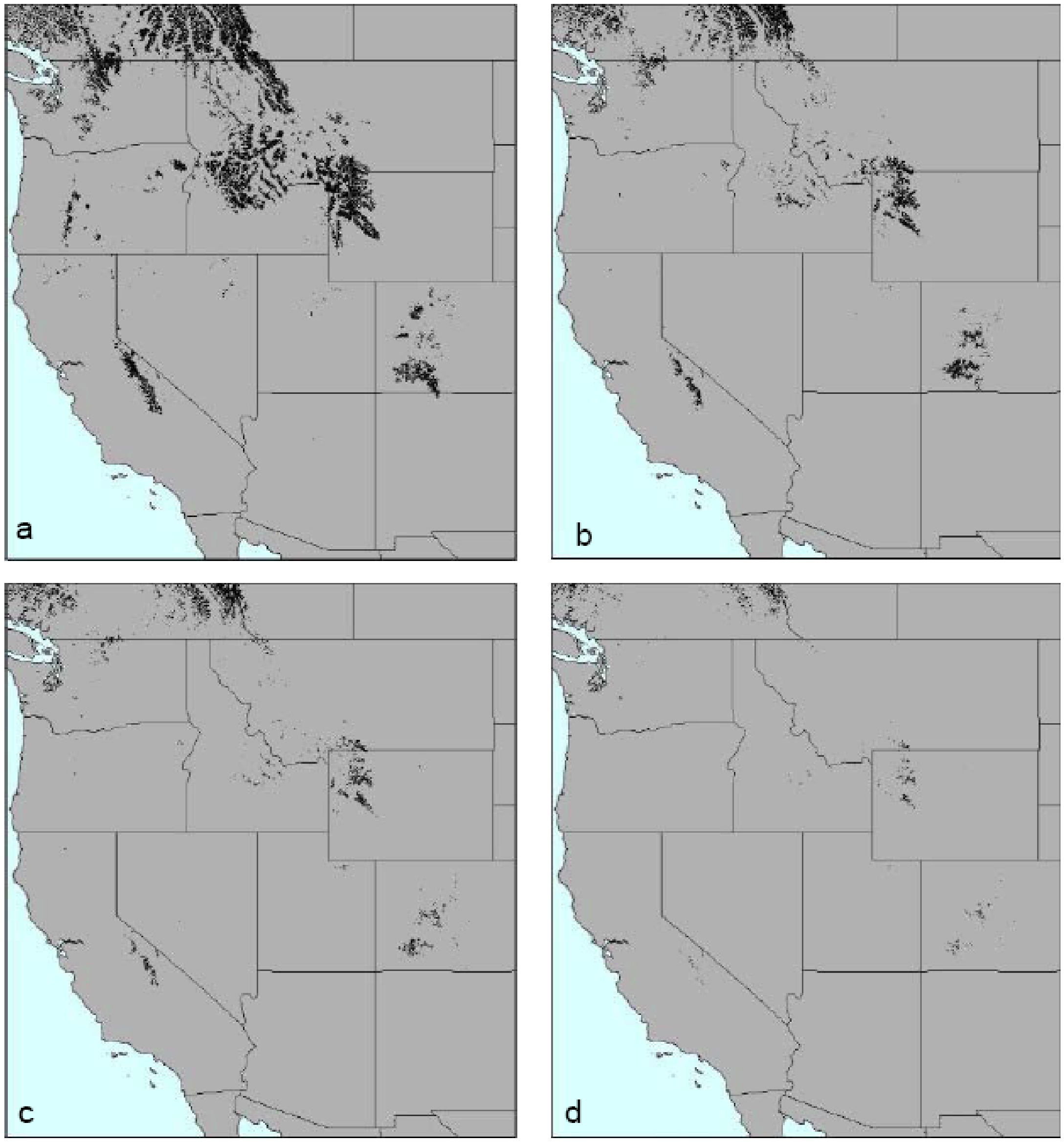

Figure 10. Modeled bioclimate profile of whitebark pine, *Pinus albicaulis,* for the present (a) and predicted climate for decades 2030 (b), 2060 (c) and 2090 (d) under climate change scenario using an average of Hadley and CCC GCM scenarios of 1% per year increase GGa. Black indicates location of pixels receiving ≥ 50% proportion of votes in favor of being within the climate profile. From Warwell et al. 2007.

Moreover, blister rust is currently present at the northern range limits for *P. albicaulis*, as well as at treeline in *P. albicaulis* (e.g., Smith et al. 2008; Resler and Tomback 2008). Damage and mortality caused by blister rust is likely to inhibit northern and altitudinal migration of whitebark pine as the climate warms (e.g., Tomback and Resler 2007). All evidence points to the conclusion that under current climate change scenarios, whitebark pine will be severely reduced if not eliminated via range changes and displacement.

2. Range Shifts of Pathogens and Pests

Climate change is also expected to facilitate the expansion of pathogens and pests. Some possible results of climate change include (1) altered dispersal, reproductive, or developmental processes of pathogens; (2) increased pathogen virulence or growth of host populations; and (3) increased pathogen predation on host species by mediating pathogen competition with symbiotic organisms that protect plants against pathogens (Cammell and Knight 1992, Manning and Tiedemann 1995, Koteen 2002). In the case of whitebark pine, climate models predict an increased incidence of both white pine blister rust and mountain pine beetles.

a. Blister Rust and Climate Change

Historically, biologists thought that whitebark pine populations occurring in the extreme climates of the Greater Yellowstone Ecosystem (GYE) of northwest Wyoming would be protected from blister rust fungus due to the unfavorable climate. It was hoped that these "protected" populations would prevent general whitebark pine population collapse (Carlson 1978, Arno 1986). In 1967, a survey of the Greater Yellowstone Ecosystem performed by Brown and Graham, although not extensive, found low blister rust levels of 1.1% on average, and the highest incidence of infection anywhere in the area was 2.3% (Brown and Graham 1967). Surveys conducted in 1995, reported mean estimated infection levels that had increased several fold to 5% in Yellowstone and to 15% in Grand Teton National Park (Kendall 1995). Infection levels further increased to a reported 20% in the Greater Yellowstone Ecosystem by 2007 (Reinhart 2007). Blister rust infection is continuing to intensify in the subalpine zone of the Greater Yellowstone Area. Much of this increase in infection is due to the cumulative buildup of blister rust spores associated with increasing levels of pine infection (Kendall and Keane 2001). However, the increase may also be due to favorable conditions being created by climate change.

Currently, the persistence of snowpack into the summer at high elevations delays the emergence of *Ribes* leaves, thereby slowing the rust cycle. Similarly, early frost in the fall may lead to early *Ribes* leaf senescence, reducing the time available for completion of the blister rust cycle (Koteen 2002). Climate change is predicted to cause earlier snowmelt and/or a later frost which would allow more time for the rust cycle to occur. Additionally, changes in the frequency or persistence of rainfall patterns may lead to the heightened presence of blister rust among whitebark pine populations (Koteen 2002).

Koteen (2002) performed a sensitivity analysis by varying the monthly mean and interannual variability in precipitation in a model for the Greater Yellowstone Ecosystem. These variables were chosen because an increase in these statistics matches the direction of those predicted by most general circulation models of climate change for the region. Any increase in the summer or early autumn precipitation would be expected to increase both the occurrence of years in which all the climatic conditions for blister rust cycle are met and the frequency of blister rust events within an individual season, contributing to disease intensification (Koteen 2002). More frequent, extreme precipitation events would provide more opportunities for widespread dispersion of blister rust. On the ground, when the climatic conditions for the spread of blister rust are met continually over the course of a summer, the stages of blister rust development may cycle through numerous generations, increasing the intensity with each cycle and producing large-scale episodes of blister rust spread (Koteen 2002). These conditions combined with the slow and steady buildup of blister rust in the region over the past decades has likely set the stage for larger transmission events in the future.

While the modeling efforts related to white pine blister rust and climate change described here are limited to the Greater Yellowstone Ecosystem, the general principles have broad applicability across the range of whitebark pine. For example, earlier snow melt or later frosts are a likely range-wide consequence of global warming though specific precipitation patterns may vary by location (Koteen 2002). Nonetheless, even if the conditions described by Koteen (2002) allow for the proliferation of blister rust at a limited, local scale, this proliferation will increase the overall amount of blister rust that is available to spread throughout the largely continuous range of whitebark pine. Therefore, climate change is predicted to increase the infection levels of whitebark pine due to blister rust.

b. Mountain Pine Beetle Infestation and Climate Change

Although historically the mountain pine beetle occasionally attacked and killed individual whitebark pine trees, outbreaks of the beetle in whitebark pines have been unusual, largely because the climate in which the trees are found is too cold for the beetles to reproduce quickly enough to reach outbreak levels. The extreme winter conditions also caused significant mortality for the beetles (Logan and Powell 2001, Amman 1990). However, with warming temperatures due to climate change, whitebark pine forests are now experiencing an ever-increasing, widespread attack of mountain pine beetles. Although whitebark is adapted to a harsh, high-elevation environment, it is poorly adapted to dealing with an onslaught of the insects and pathogens that occur in more benign environments (Logan and Powell 2001, Dunn and Crutchfield 2006).

A few historical outbreaks of mountain pine beetle have been recorded in the United States. A large proportion of mature whitebark pine were killed during the 1930s and 1940s and again in the 1970s and 80s when mountain pine beetle outbreaks spread from low elevation lodgepole pine forests upward into whitebark pine forests (Ciesla and Furniss, 1975). The outbreaks of the 1930s were accompanied by a decade of temperature averages that were 2° warmer than the average temperature for the entire century (1895-1998) (Logan and Powel 2001). This temperature change is the amount predicted to result in a shift from a semivoltine to univoltine life cycle and synchronous emergence – the two conditions necessary to produce massive outbreaks (Logan and Powell 2001). These outbreaks created many whitebark pine “ghost” forests (Arno and Hoff 1990, Perkins and Swetnam 1996). For example, in the Big Hole Range, straddling the Idaho-Montana border, whitebark pine basal area decreased measurably from 1928 to 1932 as a result of the mountain pine beetle outbreak (Murray 1996).

Using the events of the 1930s to model the current distribution and spread of mountain pine beetles at higher elevations, a long-term study of whitebark pine forest was established at 10,000 ft on Railroad Ridge, in the White Cloud Mountains of Central Idaho where high-resolution weather monitoring systems have been maintained since 1996 (Logan and Bentz 1999, Logan and Powell 2001). Based on climate models, the researchers predicted that mountain pine beetle adaptive seasonality would cause higher elevation outbreaks at the Railroad Ridge study site by mid-century. However, outbreak populations began to be expressed in 2003, and by 2005 a full scale mountain pine beetle epidemic was underway in the study area (Logan and Powell 2005). Logan and Powell (2005) attribute the earlier than predicted infestation to the nature of climate change which is expressed more intensely at high elevations and latitudes than the global average, and to the ecology of mountain pine beetle operating as an invasive species in an ecosystem that has not co-evolved with epidemic pine beetle disturbance. Modeling results indicate that the impacts of mountain pine beetle outbreaks will intensify as climate continues to warm (Logan and Powell 2001, Logan et al. 2003)

Not only did an increase in temperature predict shorter life cycles and higher elevation shifts, but the models also predicted a northerly latitudinal shift of mountain pine beetles. A temperature increase of 2.5 ° C predicted a corresponding shift of 7 ° north in latitude (Logan and Powell 2001). By 2003 this shift had occurred and invading populations of mountain pine beetle are now widespread in Alberta. In fact, the largest recorded mountain pine beetle outbreak in British Columbia (currently involving 13 million ha.) continues to expand in previously unoccupied lodgepole pine forests to the north, and previously unattacked jack pine (*Pinus banksiana*) forest which connects the pine habitat of the entire North American continent (Carroll et al. 2006).

While mountain pine beetles are a native pest to the western forests of North America, they have been rare in cold, high elevation ecosystems. Historical outbreaks have coincided with increased temperatures that allow the beetles to expand their range, rapidly reproduce and emerge synchronously leading to devastating infestations. While predicted by climate modeling, the current outbreaks in whitebark pine habitat have occurred earlier than expected and are far reaching. High elevation and northerly habitat that was previously inhospitable to the beetles is predicted to continue becoming susceptible to the spreading infestations as the climate continues to warm. Global warming, therefore, poses an additional threat to whitebark pine by facilitating the spread of its primary disease organisms – white pine blister rust and the mountain pine beetle (Logan 2008).

3. Changes in fire regimes

While the extent to which climate change is related to forest fire frequency and intensity remains a subject of great debate, there is significant evidence to suggest that global warming may well result in changes to fire patterns in western North America. A 2006 study found a dramatic and sudden increase in large wildfire activity in the western US in the mid-1980s closely associated with increased spring and summer temperatures and an earlier spring snow melt (Westerling et al. 2006). By examining climate and fire data over a 34 year period, the authors found that spring and summer temperatures in the west were 1.5º F warmer from 1987-2004 than they were from 1970-1987. Related to these increased temperatures was an increase in the length of the fire season, a fourfold increase in the number of fires, a fivefold increase in the time required to put out the fires and greater than a sixfold increase in the area that was burned. A series of other studies have also linked warmer temperatures in Canada to recent increases in wildfires (Duffy et al. 2005, Kasischke and Turetsky 2006, Flannigan et al. 2005, Gillett et al. 2004).

These studies are consistent with trends that have documented a 70% increase in the number of acres burned by wildfires in the U. S. since 1980 compared to the period between 1920-1980 (IPCC 2007). One of the complications with identifying climate as the definitive cause of increased fire frequency and intensity is the confounding effect of forest management including fire suppression over the last century. As discussed above, one result of fire suppression is an increase in flammable vegetation, or fuel levels, so that when fires do occur they may be larger and more severe than before fire suppression (Keane et al. 2002).

In addition, trends toward greater storminess are predicted to raise the number of lightning ignitions that may occur over a season (Davis 1998, Clark 1990, Franklin et al. 1992). Indirect effects of fire regimes may include lengthening of the fire season, elevated CO_2 concentration, and higher autocorrelation of daily temperature leading to prolonged droughts (Flannigan and Wagner 1990, Balling 1996).

Climate change may also influence the flammability and accumulation of forest fuels. Rising CO_2 slows decomposition rates of the litter of some species by increasing their carbon to nitrogen ratios, therefore prolonging the time for matter to decay on the forest floor. These processes also alter forest structure and chemistry which affect flammability characteristics (Koteen 2002).

While whitebark pine communities need occasional, moderate fires, future changes to regional fire regimes are expected to result in more frequent and intense fires. Combined with encroachment by other species from successional replacement, fires in whitebark pine forests would be predicted to result in large, stand replacing fires. When subalpine fir is present, for example, the developing fir understory can serve as a fuel ladder that facilitates the spread of flames to the forest canopy (Bradley et al. 1992). Therefore, changes in fire regimes under climate change scenarios are likely to further threaten whitebark pine.

D. Inadequacy of existing regulatory mechanisms

There are few, if any, regulatory mechanisms in place to protect whitebark pine from the triple threat that it faces: global warming, blister rust, and pine beetles, much less the synergistic effects of all three of these threats acting in combination. Currently there are no mechanisms to effectively control greenhouse gas emissions – the driver of global warming. As the U.S. Fish and Wildlife Service itself recognized recently when it listed the polar bear as a threatened species, while "there are some existing regulatory mechanisms to address anthropogenic causes of climate change…these mechanisms are not expected to be effective in counteracting the worldwide growth of GHG (greenhouse gas) emissions within the foreseeable future." Determination of Threatened Status for the Polar Bear (*Ursus maritimus*) Throughout Its Range, 73 Fed. Reg. 28212, 28241 (May 15, 2008). Nor do any mechanisms exist in the United States or Canada to adequately address other threats to the species.

1. The United States

Existing forest management law in the United States provides few regulatory standards through which to mandate the conservation of whitebark pine. Nearly all whitebark pine occurs on public lands and therefore is subject to the National Forest Management Act ("NFMA"), 16 U.S.C. §§ 1600, *et seq.*, and the Healthy Forest Restoration Act ("HFRA"), 16 U.S.C. §§ 6501, *et seq.*.

The HFRA does not contain any provisions which mandate the conservation of whitebark pine specifically, or diverse forests more generally. While the HFRA does provide for information gathering and silviculture assessments of pine beetle infestations, 16 U.S.C. §§ 6551 - 6556, these provisions carry little regulatory weight and do not impose any enforceable conservation requirements on the United States Forest Service.

By contrast, NFMA contains a mandate that the Forest Service adopt guidelines for the management of national forests that "provide for diversity of plant and animal communities" and, in particular, ensure "steps to be taken to preserve the diversity of tree species." 16 U.S.C. §1604(g)(3)(B). Unfortunately, this provision has resulted in no enforceable mandates to preserve whitebark pine.

In fact, the Forest Service recently adopted new regulations that govern national forest planning under NFMA which substantially reduces the Forest Service's obligation to conserve tree diversity in its forest management planning process. *See* National Forest System Land Management Planning," 73 Fed. Reg. 21468 (April 21, 2008) ("Planning Regulations"). In its Planning Regulations, the Forest Service explicitly supersedes previous regulations that "established requirements for plant and animal diversity," including maintaining the diversity of tree species. 73 Fed. Reg. at 21470. The Forest Service's new regulations not only do not contain these mandates, but also reject the use of any "standards" (which can be used to restrict the use of management practices such as clearcutting, forest thinning, and prescribed burns) in its Planning Regulations. 73 Fed. Reg. at 21474. Instead, the Forest Service chooses to rely on its internal "directives" as the "primary basis" for forest management. 73 Fed. Reg. at 21478. The Forest Service has not issued any directives mandating or prescribing for the conservation of whitebark pine (U. S. Forest Service 2008). Regardless, and unlike protections under the ESA, the Forest Service asserts that any such directives would be unenforceable in federal court. Id. (citing Western Radio Services Co. v. Espy, 79 F.3d 896 (9th Cir. 1996)).

It is also worth noting that global warming (which the Forest Service itself acknowledges is an "important" forest management issue) is a problem well beyond the capacity of individual forest managers, or the Forest Service itself, to effectively address. Indeed, rejecting its previous planning regulations that mandated maintaining a diversity of plant and wildlife communities, the Forest Service argues that it is technically impossible for it to be responsible for maintaining species diversity on national forests "given that the cause of the decline of some species is outside the Agency's control." 73 Fed. Reg. at 21472, 2176.

It should be noted that the Forest Service has undertaken some efforts to conserve whitebark pine. In 2003, for example, the Forest Service published a report called, "Managing for healthy white pine ecosystems in the United State to reduce the impacts of white pine blister rust" (Samman et al. 2003). In 2005, the Forest Service detailed a member of its staff to "assess the health of whitebark pine across its range and to develop a conservation and restoration plan for FHP activities related to whitebark pine" (Schwandt 2005). This effort resulted in the publication of a report dated August, 2006, which details some of the threats to the species discussed in this Petition (Schwandt 2006). In 2006, a whitebark pine restoration fund was created by the Washington, DC, office of Forest Health Protection, but the available funding is inadequate for the scale of the problem and suffers from year to year uncertainties. However, in general, efforts to date are not part of any national mandate, are haphazard, uncoordinated between regions, and suffer from limited funding. In short, existing regulatory mechanisms and efforts on whitebark pine cannot be expected to adequately deal with existing threats to the species.

2. Canada

Canadian law also provides few protections to whitebark pine. Most whitebark pine in Canada is found on provincial lands in British Columbia and Alberta and in national parks. In general, Canada, British Columbia and Alberta do not have regulatory mechanisms in place specifically to protect whitebark pine from the triple threat that it faces: global warming, blister rust, and pine beetles, much less the effects of all three of these threats acting in combination.

At the federal level, there are no laws protecting whitebark pine and its habitat or adequate regulation of the threats to whitebark pine. Whitebark pine is not listed under the Committee of the Status of Endangered Wildlife in Canada (COSEWIC) or protected under the Canadian Species at Risk Act (SARA). However, Parks Canada, which has jurisdiction over national parks land in Canada and significant areas of whitebark pine habitat (e.g., Waterton, Banff and Jasper National Parks in Alberta), has expressed concern over the health of the whitebark pine, and seems to be taking voluntary action (Parks Canada 2005). For example, Waterton Lakes National Park is evaluating the use of prescribed fires or artificially creating small open areas to revitalize whitebark pine. Waterton Lakes National Park is also undertaking a whitebark pine ecosystem restoration research program to refine management practices to restore what Parks Canada calls a "key threatened species" (Parks Canada 2005). Standing alone, however, this action is unlikely to be sufficient to adequately conserve the species.

In Alberta, in 2005, the government identified whitebark pine as a species that "may be at risk" meaning that it would receive evaluation (Alberta Sustainable Resource Development 2005). The government identified the threats as extensive infection and subsequent mortality from white pine blister rust throughout its narrow Alberta range; additional mortality and threats from mountain pine beetle; and limited threats to habitat from logging and oil and gas activities. As early as 2003 and as recently as June 2008, the Alberta government publicly acknowledged that threats from the mountain pine beetle are real and need to become part of the provincial management strategy (Alberta Sustainable Resource Development 2003, 2008). The laws in Alberta do not meet the protection needs of the whitebark pine providing only for general protections of natural resources. The Alberta Forest and Prairie Protection Act does provide that the Minister may carry out control measures to prevent tree pest infestations and may initiate fire for control purposes (Alberta Forest and Prairie Protection Act – RSA 2000).

In 2008 the British Columbia Ministry of Environment's Conservation Data Center adjusted the conservation ranking of whitebark pine, adding it to the *blue-list,* a list of special conservation concern in British Columbia. This species was uplisted due to a "severe negative long-term trend expected from mountain pine beetle infections, the white pine blister rust epidemics, climatic warming trends, and successional replacement" (B.C. Conservation Data Centre 2008). The British Columbia forest laws, however, do not include any protections for whitebark pine. The Forest and Range Practices Act (SBC 2002, Chapter 69) does include a section on control of insects and diseases that allows the Minister to require the owner or lease-holder of the land to submit a proposal for insect or disease control. However, it is unlikely that this would apply to whitebark pine habitat which tends to grow at altitudes outside of normal harvesting areas. The Act also allows the government to declare an area a forest health emergency management area. For whitebark pine found in parks in British Columbia, the British Columbia Park Act (RSBC 1996, Chapter 244) applies. It has a general provision that natural resources are protected from "damage and disturbance."

In short, there are no effective mechanisms in place to control global warming pollution. Moreover, given the lack of any national mandates to conserve whitebark pine and the (at best) scattered attempts by regional national forests in the United States, and some Provinces in Canada, to restore whitebark pine populations, it is clear that the threats to whitebark pine are exacerbated by the presence of inadequate regulatory mechanisms.

V. Conclusion: Threats Work Together to Cause Unprecedented Declines

Whitebark pine has been experiencing steady declines over its entire range due largely to a combination of forest succession in part from fire suppression practices, infection from the introduced fungus white pine blister rust, and an unprecedented outbreak of mountain pine beetles. Global warming now poses an additional threat to this high-elevation species. For example, the impact of beetles and blister rust is increasing at northern latitudes and high elevations thereby limiting the ability of whitebark pine to respond to warming temperatures through adaptation or migration. Individually, each of these threats has diminished whitebark pine communities in large portions of the species' range. Working in combination, these elements interact synergistically to threaten the very existence of whitebark pine.

Fire suppression allows for the encroachment of shade tolerant species that compete with whitebark pine and decrease available openings on the ground that facilitate nutcracker seed caching and support productive whitebark pine tree growth. Fire suppression additionally allows for a build up of fuels that can cause more devastating, stand-replacing fires which are likely to become more frequent under global warming.

Blister rust can infect whitebark pine trees of all ages, including seedlings. Mountain pine beetles, on the other hand, need larger, adult trees to support their invasion. They favor, however, trees that have already been weakened by blister rust. Warmer temperatures, which lengthen the growing season, and changes in precipitation patterns caused by climate change exacerbate whitebark pine infections by both the fungus and the beetle.

Finally, global warming threatens to severely reduce the suitable bioclimatic range for whitebark pine communities and other high elevation ecosystems, although the outbreaks of mountain pine beetle illustrate that the ecological effects of climate change are already in progress. The increased tree mortality due to a combination of the above factors also means that less atmospheric carbon is absorbed and in fact additional carbon may be released as the trees decay thereby feeding the cycle of global warming. Without adequate regulatory mechanisms to prevent or even slow the loss of whitebark pine, the species must be listed as endangered under the Endangered Species Act.

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