

INTRODUCTION

Whitebark pine (*Pinus albicaulis* Engelm.) is in serious decline across its range, largely due to the combined effects of *Cronartium ribicola* J. C. Fisch (an introduced fungal pathogen that causes white pine blister rust), replacement by late successional species, and widespread infestation of mountain pine beetle (*Dendroctonus ponderosae* Hopkins) (Gibson and others 2008, Hoff and others 1980).

C. ribicola was introduced into North America from Europe, possibly as early as 1898 (Benedict 1981, Fins and others 2001). From two initial points of introduction, blister rust has spread to all five-needle pine species in North America, including nearly the full range of whitebark pine populations (McDonald and Hoff 2001). Whitebark pine trees of all sizes are susceptible to infection, but mortality from blister rust is usually < 100 percent. While uninfected trees may be escapes, an alternative hypothesis is that some of them are genetically resistant, as several genetically controlled defense mechanisms against blister rust have been observed in whitebark pine (Hoff and others 2001, Mahalovich and others 2006). But, because the level of native rust resistance is estimated to be low, only a small percentage of trees are predicted to survive to maturity when rust infection is epidemic (Hoff 1994, Kendall and Keane 2001). Selection and breeding for specific defense mechanisms is likely to improve resistance.

Mountain pine beetle, the most destructive insect pest of pine species in western North

America (Gibson 2003), is a native species that ranges from British Columbia and Alberta to northern Mexico. The beetle's recorded hosts include many pine species of western North America. Beetles have killed nearly 6 million high elevation five-needle pines in the last 5 years, with mortality highest in lodgepole pine (Gibson and others 2008). Whitebark pine mortality from mountain pine beetle was reported across almost 500,000 acres in 2007 (Gibson and others 2008). The recent expansion of mountain pine beetle, both northward and upward in elevation, has been attributed largely to warmer than normal temperatures (Bentz and Schen-Langenheim 2007, Carroll and others 2003).

The steep, rocky, and mountainous terrain of the Frank Church River of No Return Wilderness Area (hereafter referred to as the Frank Church) covers approximately 2.3 million acres and includes much of the Salmon River drainage in central Idaho. It is the second largest wilderness area in the conterminous United States.

Whitebark pine is broadly distributed in the Frank Church from approximately 2300 m to timberline, but to date, no extensive studies have been published on the health status of these populations. As a keystone species in these ecosystems, the general condition of whitebark pine populations is of major importance and any significant decline could have broad ecological ramifications. The purpose of our study was to provide baseline information on the health status of whitebark pine populations in the Frank Church. This information will be useful to managers in assessing population trends and

CHAPTER 13.

Health, Reproduction, and Fuels in Whitebark Pine in the Frank Church River of No Return Wilderness Area in Central Idaho (Project INT-F-05-02)

LAUREN FINS¹ AND BEN HOPPUS²

¹Professor of Forestry, College of Natural Resources, University of Idaho, Moscow, ID 83844-1133.

²Forestry Technician, U.S. Department of Agriculture Forest Service, Mt. Adams Ranger Station, Gifford Pinchot National Forest, Trout Lake, WA 98590.

developing management strategies, especially if any remedial action is anticipated. Our objectives were to:

1. Evaluate and establish baseline information on levels of infection and mortality from white pine blister rust and mountain pine beetle in whitebark pine populations in the Frank Church.
2. Establish baseline information on existing stand structures and occurrence of whitebark pine regeneration.
3. Estimate loadings of downed woody material in these populations.

MATERIALS AND METHODS

Between the summers of 2005 and 2008, we established and assessed the conditions of trees in 119 permanent study plots in six whitebark pine populations in the Frank Church. Populations were selected based on landscape level GAP analysis cover type data (Scott and others 2002), as well as on fire history data from the Forest Service, U.S. Department of Agriculture (Gibson and Morgan 2009), but selection was also influenced by accessibility and logistical considerations. The initial study design consisted of a 3 x 3 factorial of plot types (3 habitat classes x 3 burn classes) replicated three times each, i.e., 27 study plots in each study population. We were able to locate sites and establish the full factorial design (nine habitat/burn factor combinations) in three of the six populations, two of which included at least three replications per habitat/burn combination. In the other three populations, where some of the habitat/burn

combinations were either limited or missing, plots were established in the combinations that were available.

Plot locations were selected based on availability of areas with appropriate habitat/burn combinations and accessibility, with a minimum distance of 30 m between plots. However, most of the plots (> 80 percent) were separated from the nearest plot by at least 100 m. Rectangular plots (150 feet x 30 feet = 45.7 m x 9.1 m = 0.04 ha) were established as per Whitebark Pine Ecosystem Foundation guidelines (Tomback and others 2005). Plot elevations ranged between 2290 m and 2930 m, primarily on south aspects, with slopes from 0 to 80 percent, averaging 32 percent. Data were analyzed using a nested model, with stand composition and burn history nested within population. Population, habitat type, and burn history were treated as fixed independent variables.

Data collection generally followed protocols recommended by the Whitebark Pine Ecosystem Foundation, with some additions and minor modifications (Tomback and others 2005). Data on whitebark pines > 1.4 m tall included d.b.h., distance along and away from the transect (midline of the plot), overall health, canopy condition, blister rust status, and mortality. Types and locations of blister rust cankers were recorded, as well as whether the cankers were active or dead. Also recorded were counts of whitebark pines shorter than 1.4 m (considered as the regeneration class), occurrence of blister rust in this group, and whether trees were

shorter or taller than 0.5 m. Plot data also included location, aspect, elevation, and year plots were established, percent composition of other tree species with stems > 1.4 m tall, mountain pine beetle infestation on live whitebark pine trees, and likely agent of mortality on dead mature whitebark pines. Frank Church RONRWA research protocols were followed in establishing and monumenting plots.

RESULTS

Within the 119 permanent plots in the study were 3,529 whitebark pine trees ≥ 1.4 m and 3,950 whitebark pine seedlings/saplings < 1.4 m. Trees in the latter category (the regeneration class) were found in 110 of the 119 plots (92 percent), their numbers ranging from 1 to 160, and averaging 36 stems (< 1.4 m tall) per plot. Of 3,529 larger whitebark pines in our plots, 661 (19 percent) were dead. Of the 661 dead trees, 23 (3 percent) were clearly killed by blister rust, 252 (38 percent) killed by mountain pine beetle, 262 (40 percent) by fire, and 124 (19 percent) by unknown causes. Our values for blister rust and mountain pine beetle mortality are likely conservative since some of the 124 trees killed by unknown causes are likely to have been killed by one of these two factors.

Blister rust infection was present in all six populations. Plots (of trees ≥ 1.4 m tall) averaged 18.6 percent of live trees infected (range 0–64.5 percent). This average is based on 2008 assessments in five populations and the 2006 assessment in one population. Using data

only from the three populations in which all plot types were present, we found a statistically significant difference among the populations in blister rust infection ($p < 0.001$). Blister rust was observed in the regeneration class in only seven of the plots. Differences in blister rust infection levels were not associated with either composition or burn class.

Active mountain pine beetle infestation was observed in 22 of 119 plots (19 percent of plots), ranging from 1.5 percent to 52 percent active infestation within plots. Affected plots averaged 15.7 percent of trees showing active infestations, but across all plots, the overall mean of live trees under attack was only 2.9 percent. Levels of mountain pine attack were significantly different among populations ($p < 0.001$) and among burn classes nested within population ($p < 0.001$). Recently burned plots in the Sleeping Deer Mountain population were attacked at higher levels than other plot types. Habitat type was not associated with differences in levels of mountain pine beetle attack.

The mean litter loading was 4.94 tons/ha (2 tons per acre), ranging from 0 to 22.23 tons/ha (0 to 9 tons per acre); mean duff loading was 38.8 tons per ha (15.7 tons per acre), ranging from 0 to 150.4 tons/ha (0 to 60.9 tons per acre); mean loading of the 0 to 7.62 cm class (0 to 3 inch) fuels was 5.18 tons/ha (2.1 tons per acre), ranging from 0 to 587.86 tons/ha (0 to 23.8 tons per acre); and mean > 7.62-cm class (> 3 inch) fuels was 19.02 tons/ha (7.7 tons per acre), ranging from 0 to 148.94 tons/ha (0 to 60.3 tons per acre).

DISCUSSION

The mean level of white pine blister rust infection on live whitebark pine trees in the Frank Church River of No Return Wilderness Area ($\bar{x}$ = 18.6 percent) is lower than has been reported in other parts of the species' range, but comparable to blister rust levels assessed between 1995 and 1996 near the perimeter of the southern half of the wilderness area, which averaged 17.6 percent infection (Smith and Hoffman 2000). The low level of blister rust infection in regeneration is encouraging, but infection levels are likely to increase over time as the trees live longer, with more opportunities for infection and increased target area (Fins and others 2002).

The mean level of mountain pine beetle attack in live whitebark pine across the six populations was quite low (2.9 percent), and the trees under attack were in small widely scattered clusters. However, since only whitebark pine was inspected for mountain pine beetle in our study, and lodgepole pine was not inspected, the actual numbers of trees attacked per hectare is likely to be much higher. The plots with the highest levels of attack (up to 39 percent of live whitebark pine) were in a recently burned area in the Sleeping Deer Mountain population where as many as 36 whitebark pine trees per hectare were infested and will likely die in the near future.

The mean fuel loads measured on our plots were slightly lower, but comparable to those

found in Fire Group 10 habitats described by Crane and Fischer (1986). Fuels were discontinuous and were influenced by the topography and heterogeneity of the landscape. This lack of uniformity was observed among neighboring plots and across populations.

Before modern fire suppression, fires of mixed severity occurred at intervals ranging from 30 to 300 years (Arno 2001). Stand replacing fires created sites for Clark's nutcracker to cache seeds, and light intensity fires killed understory spruce and fir (Arno and Hoff 1990). But fire exclusion in the last century has altered natural fire cycles where whitebark pine is seral, resulting in successional replacement by more shade tolerant species such as subalpine fir (Arno 1986). Restoring natural fire regimes may be possible and useful in maintaining whitebark pine populations, because the fires would create cache sites for the Clark's nutcracker, and at least some of the cached seeds will have come from trees that survived blister rust infection and which may be genetically resistant (Keane and Arno 2001). Further discussion of results and more detailed descriptions of analyses can be found in Hoppus (2009). Photographs of the study area, a description of habitat types and burn classes, and tables and graphs of results by populations and/or by plot type can be found at www.fs.fed.us/foresthealth/fhm/posters/posters09/posters09.shtml (Fins and others 2009).

CONCLUSION

Although population recruitment was evident in most of these populations (92 percent), the persistence of blister rust, in conjunction with losses to mountain pine beetle and potential habitat shifts due to climate change, suggest the possible future loss of some whitebark pine populations. As a hedge against future losses and a strategy to maintain a broader array of future options, we argue for continued monitoring for changes in health in these populations and the collection and archiving of genetically representative samples, including seed and scion from trees in each population.

Our work provides baseline information on the health and regeneration potential of whitebark pine populations in the Frank Church, and the 119 permanent plots we established can be used to help understand and predict the trajectory of these populations in the future. The levels of blister rust infection in the Frank Church were relatively low in 2008 compared to other areas in the range of whitebark pine, such as in Glacier National Park. However, changes in climate or blister rust wave years could sharply increase infection levels over just a few years.

Given the ecological importance of whitebark pine as a keystone species in these ecosystems, it is imperative that genetic materials, such as seed and/or scion, are collected from these populations and placed in genetic archives for their potential use in ecosystem restoration in the future. Furthermore, the populations should be

monitored regularly to determine the trajectories of their health status over time and action taken as appropriate.

All options intended to reverse the decline of these populations should be considered, including the dissemination of genetically resistant materials, restoration of natural fire regimes, and maintaining and archiving the gene pools of current populations.

CONTACT INFORMATION

Lauren Fins, College of Natural Resources,
University of Idaho, Moscow, ID 83844-1133;
telephone: (208) 882-6834;
e-mail: lfins@uidaho.edu.

Ben Hoppus, Forestry Technician, U.S.
Department of Agriculture Forest Service, Mt.
Adams Ranger Station, Gifford Pinchot National
Forest, Trout Lake, WA 98590;
e-mail: bhoppus@fs.fed.us.

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