



REVIEW

Current health issues and management strategies for white pines in the western United States and Canada

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Summary

The introduced pathogen *Cronartium ribicola*, cause of white pine blister rust, has spread across much of western North America and established known infestations within all but one species of white pine endemic to western Canada and the United States. Blister rust damage to severely diseased trees reduces reproduction and survival. Severe losses in white pine populations have resulted in site conversions to other species and seriously impacted resource values for timber, wildlife, watershed, recreation, aesthetic and other ecosystem services. In addition to blister rust, other major forest health threats and challenges to sustaining or restoring white pine populations are infestations of other pathogens, insects, fire, management practices that favour other tree species, and climatic change. Recent, large-scale outbreaks of mountain pine beetle have raised concerns for the viability of some white pine populations. In the 1960s, forest disease management for western white pine and sugar pine shifted from *Ribes* eradication to planting seedlings selected for better survival and resistance to blister rust. Seed orchards for producing improved white pines have been established, but deployment of that improved stock is hampered by a lack of planting opportunities. The inheritance and mechanisms of resistance are best known for western white pine and sugar pine; but new work is extending an understanding of genetics to all the western species of white pine. Current management efforts are focused on locating and protecting individual trees resistant to blister rust and assessing their disease resistance and other adaptive traits. In response to the threats from blister rust, the strategic goal is to sustain or restore viable white pine populations in western forest ecosystems. The four action components of the strategy are: (1) conserve genetic resistance to *C. ribicola*; (2) reduce the risk of adverse impact in stands currently uninfested; (3) restore and maintain white pines where blister rust is causing impacts and (4) assess and monitor the health and management of white pines. Successful implementation requires long-term support for coordinated efforts of management and research agencies, forest industry and an informed public.

1 White pine blister rust extent and severity since 1960

The establishment and early spread of *Cronartium ribicola* J.C. Fisch. in Rabh., cause of white pine blister rust, is summarized in the introductory article by GEILS et al. (2010). We describe here the extent and severity of blister rust in western North America (western Canada and United States) after 1960, when management emphasis shifted from *Ribes* eradication to genetic resistance. The known geographic distribution of blister rust on white pines continues to increase and now includes both western Canadian provinces and all the western states except Utah (Fig. 1). In the West, *C. ribicola* occurs from central British Columbia and Alberta, south through California, across the mountain states to western South Dakota. Within this region, blister rust infests native populations of eight taxa of white pine (*Pinus*, subgenus *Strobus*, subsections *Balfourianae*, *Cembrae* and *Strobi*). Great Basin bristlecone pine (*Pinus longaeva* D.K. Bailey) is the only western species of white pine not known to be infested in nature, although it is susceptible (HOFF

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Fig. 1. Distribution of white pine blister rust in western North America in 2009. The general distribution of *Cronartium ribicola* is mapped over the natural distribution of white pines in western Canada and United States [adapted from LITTLE 1971: Conifers and important hardwoods. Atlas of trees of the United States. Misc. Publ. 1146. Washington, DC: U.S. Department of Agriculture, Forest Service. (Online). Available: <http://esp.cr.usgs.gov/data/atlas/little/>]. The white pines mapped here are *Pinus albicaulis*, *P. aristata*, *P. balfouriana*, *P. lambertiana*, *P. longaeva*, *P. monticola* and *P. flexilis* (including var. *reflexa*).

et al. 1980). The many species of *Ribes* (currants and gooseberries) in the West are also susceptible to rust infection, as are several species of *Castilleja* (paintbrush) and *Pedicularis* (lousewort). These telial hosts vary by species and population in their susceptibility to infection and capability to support inoculum infective to white pines (ZAMBINO 2010).

1.1 Western white pine

Cronartium ribicola occurs throughout the distribution of western white pine, *Pinus monticola* Dougl. ex D. Don (TOMBACK and ACHUFF 2010; Fig. 2b). White pine blister rust

is nearly ubiquitous across the Inland Northwest (northern Idaho and adjacent areas). In this region, western white pine had been the dominant tree species on more than two million hectares of commercial timberland; however, because of site conversion to other conifer species, it is now dominant on less than 5% of those lands (FINS et al. 2002). Blister rust infestations are usually severe in the Inland Northwest and most areas in the Pacific Northwest (western British Columbia, Washington and Oregon).

The incidence of blister rust (percentage of trees infected) varies widely across the distribution of western white pine in the Pacific Southwest (California and western Nevada). KLIEJUNAS and DUNLAP (2007) reported on an intensive survey to detect blister rust and quantify its incidence in white pine populations throughout California. In northwest California, about half of examined trees were infected; in the central Sierra Nevada, about one-fifth were infected; and at the southern limits of western white pine, blister rust was present, but few trees were infected. A low incidence of blister rust (~3%) occurs in the western white pine of Sequoia–Kings Canyon National Parks in east-central California (DURISCOE and DURISCOE 2002). Several factors generally correlate with a trend of decreasing rust incidence in western white pine, including the recency of blister rust entry and the climatic constraints of warmth and aridity on its development.

1.2 Sugar pine

Although *C. ribicola* is a damaging and potentially lethal pathogen to sugar pine (*Pinus lambertiana* Dougl.), the severity of white pine blister rust infestation varies across the host distribution (TOMBACK and ACHUFF 2010; Fig. 2a). Blister rust entered northern California from southern Oregon about 1930 (BEDWELL 1936; WAGENER and MIELKE 1936). It did not spread to the southern Sierra Nevada until 1964 when unusually favourable meteorological conditions permitted significant local intensification and regional spread (a wave year) (KINLOCH and DULITZ 1990). Blister rust intensified and spread into other locations in the later wave years of 1967, 1976, and 1983 (KLIEJUNAS 1982, 1984). DENITTO (1987) reported that 28 of 29 plantations in the northern and central Sierra Nevada were infested by 1986 and more than two-thirds of the trees within infested plantations were infected. Although environmental conditions on favourable sites are occasionally suitable for infection (SMITH 1996), blister rust has not been detected in the Transverse and Peninsula Ranges of southern California or in the Sierra San Pedro Mártir of Baja, Mexico (CALIFORNIA FOREST PEST COUNCIL 2004).

1.3 Whitebark pine

With noted exceptions, *C. ribicola* occurs throughout the distribution of whitebark pine (*Pinus albicaulis* Engelm.) (TOMBACK and ACHUFF 2010; Fig. 1a). White pine blister rust incidence varies by area and is highest in the cool, northern Rocky Mountains (Montana and adjacent areas) and least in the warm, dry Great Basin (Nevada and adjacent areas). But even within a severely infested region such as the Pacific Northwest, blister rust incidence at a plot-level (groups of several to tens of trees) ranges from 100 to 0% (SCHWANDT 2006; WARD et al. 2006). Such differences indicate variations in time since introduction and environmental suitability for its development (hazard). For infestations many decades old, regional incidence reveals the average hazard over a large area. At smaller, stand and landscape scales, hazard changes as natural processes and management affect microclimate and vegetation, including host abundance, distribution and population genetics.

Various authors have used different sampling methods to assess the amount and effects of white pine blister rust. Although exact comparisons are not possible, general trends can be inferred. ZEGLEN (2002) reported that average blister rust incidence was 40% for British Columbia and that incidence was slightly lower in the Coastal Mountains than in the

Canadian Rocky Mountains. In addition, in the Canadian Rocky Mountains, SMITH et al. (2008) found that blister rust incidence and whitebark pine mortality were higher west of the Continental Divide (maritime climate) than east of the Divide (continental climate), but incidence and mortality were increasing on the east side. KEGLEY et al. (2004) observed that plot-level incidence in northern Idaho ranged from 50 to 96% for mature whitebark pine and 32 to 92% for immature whitebark pine. In a 1998 survey along Oregon's Cascade Crest, GOHEEN et al. (2002) observed that about half of the white pine were infected. In the Blue Mountains of northeastern Oregon, ERICKSON et al. (2007) examined three, isolated, high-elevation populations of whitebark pine (4000 trees sampled); blister rust incidence was 27, 64 and 73%, respectively. Recent surveys in California (KLIEJUNAS and DUNLAP 2007; MALONEY and DUNLAP 2007) found 13% of whitebark pine trees infected on 41% of 44 examined plots. SMITH and HOFFMAN (2000) surveyed across the southern distribution of whitebark pine; they found blister rust only in some mountain ranges. Failure to detect the rust, however, does not necessarily indicate an area is not infested. The Nevada Jarbidge Mountains on the northern boundary of the Great Basin were apparently infested around 1977; but presence of blister rust was not confirmed until 2002 (VOGLER and CHARLET 2004).

1.4 Limber pine

Limber pine (*Pinus flexilis* James) has a very wide geographic distribution (TOMBACK and ACHUFF 2010; Fig. 2a); *C. ribicola* occurs throughout all but its southern limits in California (KLIEJUNAS and DUNLAP 2007), Nevada and Utah. White pine blister rust was first discovered on Alberta limber pine in 1952 (GAUTREAU 1963) and is now so common in the province that it threatens the viability of some limber pine populations (ALBERTA SUSTAINABLE RESOURCE DEVELOPMENT and ALBERTA CONSERVATION ASSOCIATION 2007a). Numerous reports have tracked the spread of blister rust south and east since 1960 (KREBILL 1964; BROWN 1970); it extended across Wyoming by 1978 (BROWN 1978b). Blister rust incidence is highly variable among the Rocky Mountain ranges of Wyoming and Colorado. It is common (~30% infected) in the Laramie Mountains of southeastern Wyoming and rare (<1%) in the nearby Sierra Madre (KEARNS and JACOBI 2007). In recent decades, blister rust has been reported for numerous, disjunct locations, including South Dakota (LUNDQUIST et al. 1992), North Dakota (DRAPER and WALLA 1993), western Nevada (SMITH et al. 2000), northern Colorado (JOHNSON and JACOBI 2000), southern Colorado (BLODGETT and SULLIVAN 2004) and south-central Idaho (P. Wolken, personal communication). In contrast, none of the 14 plots with limber pine in the California detection survey were infested (KLIEJUNAS and DUNLAP 2007).

1.5 Southwestern white pine

Cronartium ribicola has infested southwestern white pine (*Pinus flexilis* James var. *reflexa* Engelm. syn. *P. strobiformis* Engelm.) for the past several decades, and it continues to spread and intensify (Fig. 1) within the numerous, disjunct, host populations of the Southwest (Arizona, New Mexico) (see TOMBACK and ACHUFF 2010; Fig. 2a). White pine blister rust was first discovered in 1990 on southwestern white pine in south-central New Mexico (HAWKSWORTH 1990); at the time, the infestation was about 20 years old and extended at very low incidence throughout the Sacramento Mountains (GEILS 2000; FRANK et al. 2008). Owing to large populations of susceptible hosts (white pines and *Ribes*) and early arrival of frequent, summer rains, the Sacramento Mountains provide an especially suitable environment for blister rust (FRANK et al. 2008). CONKLIN (2004) documented increases in rust incidence and host damage on 14 plots representing a range of infestation ages and stand conditions in the Sacramento Mountains. He indicated that high levels of rust incidence and host mortality were usually associated with high-hazard sites (GEILS

et al. 1999) near large populations of *Ribes*. The first infestations in the Southwest outside the Sacramento Mountains were in adjacent mountain ranges; but blister rust has been discovered since 2003 at numerous sites in northern and western New Mexico and western Arizona (FRANK et al. 2008; D. Conklin, B. Geils and M. Fairweather, personal communication; Fig. 1).

Several species of white pine occur in Mexico (GEILS et al. 2010: section 4.4), including southwestern white pine (TOMBACK and ACHUFF 2010: Fig. 2a); but no infestations of *C. ribicola* are known to us. The capability of *C. ribicola* for long-distance dispersal and proximity of susceptible hosts (white pine and *Ribes*) in the Southwest and northern Mexico, however, suggests that white pines and *Ribes* in Mexico are at risk of infestation by blister rust.

1.6 Foxtail pine

Two distinct populations of foxtail pine (*Pinus balfouriana* Grev. and Balf.) occur in California (TOMBACK and ACHUFF 2010; Fig. 1b). KLIEJUNAS and DUNLAP (2007) reported that *C. ribicola* occurred in the northern (Klamath) population on all plots and 15% of examined foxtail pine were infected. No white pine blister rust was observed on any foxtail pine in the southern (Sierra Nevada) population; we know of no reports of blister rust in this population.

1.7 Rocky Mountain bristlecone pine

The first report of *C. ribicola* on native Rocky Mountain bristlecone pine (*Pinus aristata* Engelm.) was in 2003, when an infected tree was discovered in the Colorado Sangre de Cristo Mountains (BLODGETT and SULLIVAN 2004). BURNS (2006) reported on surveys to delimit the extent of white pine blister rust in southern Colorado and estimate its incidence. Although bristlecone pine was common and widely distributed in the Sangre de Cristo Mountains, infected trees were only found near Mosca Pass. Bristlecone pine along Mosca Creek were commonly infected, incidence ~30%; bristlecone pine on adjacent, upland slope were rarely infected, incidence ~2%.

1.8 Great Basin bristlecone pine

Great Basin bristlecone pine is the only western species of white pine not known to be infested by *C. ribicola* in nature (KINLOCH 2003), although the species is susceptible (HOFF et al. 1980). KLIEJUNAS and DUNLAP (2007) searched for white pine blister rust in the California White Mountains but found no blister rust. SMITH and HOFFMAN (2000) included three plots with Great Basin bristlecone pine in their survey but found no blister rust on this species. Although many, remote populations of bristlecone pine have not been surveyed, numerous populations have been examined but no blister rust was found (GEILS et al. 2003; B. Geils, personal communication).

2 Current health issues other than white pine blister rust

In western North America, *C. ribicola* is a serious threat; but it is only one factor affecting the health of white pines. Other agents damaging to white pines are native insects and pathogens such as bark beetles, root diseases, dwarf mistletoes and foliage-disease fungi. Fire and fire suppression also have a profound influence on the viability of white pine populations. The severity of the impacts of these agents and disturbances are markedly affected by climate, especially by prolonged drought and change in temperature or moisture regimes (KLIEJUNAS et al. 2009). Because many white pine populations are

threatened by multiple and often interacting factors, management to sustain or restore white pine confronts numerous forest health issues.

2.1 Mountain pine beetle

Mountain pine beetle (*Dendroctonus ponderosae* Hopkins) is a native North American bark beetle that attacks ponderosa pine (*Pinus ponderosa* Dougl. ex Laws. and Laws.), lodgepole pine (*P. contorta* Dougl. ex Loudon), and many species of white pine (FURNISS and CAROLIN 1977). Other *Dendroctonus* species infesting white pines are *D. adjunctus* Blandford, which infrequently erupts into short outbreaks in the Southwest, and the widely distributed *D. valens* LeConte, which is usually only endemic in weak and dying trees. Although other genera of bark and twig beetles attack white pines, mountain pine beetle is the most important because it kills large, mature trees. The behaviour of mountain pine beetle as preferring stressed or healthy host trees and its population dynamics as chronic or irruptive varies by region and host species. Outbreaks occur when populations quickly increase on abundant, vulnerable hosts, such as ponderosa pine in a montane forest or lodgepole pine in a subalpine forest. These outbreaks are a natural, episodic occurrence in western North America; historically, they were the primary renewal process for mature stands (WOOD and UNGER 1996; SAMMAN and LOGAN 2000; TAYLOR and CARROLL 2004).

Outbreaks of mountain pine beetle have occurred repeatedly in white pine forests. PERKINS and SWETNAM (1996) dated two older outbreaks in Idaho whitebark pine to 1730 and 1887. A mountain pine beetle outbreak that peaked in western Wyoming during the 1930s spread from whitebark pine stands into lodgepole pine stands (GIBSON 1939; EVENDEN 1944). Several outbreaks in the 1970s and 1980s originated in lodgepole pine stands, moved into higher elevations, and killed many whitebark pine (BARTOS and GIBSON 1990; KIPFMUELLER et al. 2002). In western Canada, mountain pine beetle outbreaks from 1974 to 1984 occurred chiefly in stands of lodgepole pine but also killed many whitebark and limber pines (ALBERTA FORESTRY LANDS and WILDLIFE 1986; ACHUFF et al. 2002). Long, severe outbreaks create high-elevation 'ghost forests' (CIESLA and FURNESS 1975; LOGAN and POWELL 2001; KIPFMUELLER et al. 2002).

In recent years, mountain pine beetle activity again surged in vulnerable stands across western North America, possibly triggered by drought (TAYLOR et al. 2006) and mild winters (LOGAN and POWELL 2001). GIBSON et al. (2008) summarized results of aerial detection surveys from 1998 to 2007 for high-elevation areas with mortality to whitebark pine or limber pine attributed to mountain pine beetle. For whitebark pine, the detected area increased from 3000 ha in 1998 to 190 000 ha in 2007; for limber pine, the area increased from 450 ha in 1998 to 50 000 ha in 2007. GIBSON et al. (2008) also reported that beetle-attributed mortality of foxtail and bristlecone pines occurred in small areas for short durations.

Mountain pine beetle activity in mixed-species stands with either western white pine or sugar pine has proved difficult to detect by aerial survey; but mountain pine beetle interacting with other factors has contributed to scattered, infrequent mortality. MALONEY et al. (2008) compared restoration treatments of prescribed fire and thinning in an old-growth, Sierra Nevada forest. Sugar pine mortality from bark beetle attack was higher on large trees in burn treatments, although fire produced little apparent damage to the attacked trees. Mountain pine beetle is a significant mortality agent of medium- to large-size sugar pine in southwestern Oregon and is commonly associated with competition-related stress and blister rust, but not particularly with drought (D. Goheen, personal communication). Mountain pine beetles appear able to locate widely-scattered, low-vigour sugar pine and kill a large proportion of them, even in preference to other host species. For western white pine in northern Idaho, KULHAVY et al. (1984) also found an interaction between a predisposing tree condition (root disease and/or blister rust infection) and subsequent lethal attack by mountain pine beetle.

Accumulated mortality of mature whitebark pine resulting from multiple mountain pine beetle outbreaks varied among high-elevation stands in Idaho and Montana. SCHWANDT et al. (2009) quantified mortality as the observed percentage of trees that were standing dead for 23 stands from three areas in Wyoming, Montana and Idaho. Except for one stand, mortality was very noticeable—at least one-fourth of mature whitebark pine trees were counted as dead in 1930 or 2008. For six stands in the Yellowstone area, cumulative mortality ranged from 20 to 78% (average 48%); for stands in the northern Rocky Mountains or central Idaho, mortality ranged from 60 to 95% (average 80%). Other surveys reported mortality in terms of the density of beetle-killed whitebark pines as >200 dead trees per hectare in the Yellowstone area, >200 dead trees per hectare in central Idaho, and >400 dead trees per hectare in southwest Montana (GIBSON 2004, 2006; KEGLEY and GIBSON 2004).

Although outbreaks of mountain pine beetle had not previously been a concern for whitebark pine in British Columbia, they are now considered a serious threat. CAMPBELL and ANTOS (2000) and ZEGLEN (2002) found little evidence in their surveys from 1995 to 2000 that mountain pine beetle significantly contributed to whitebark pine mortality. Since then, however, regional outbreaks of mountain pine beetle developed in lodgepole pine stands of central British Columbia and expanded into jack pine stands (*P. banksiana* Lamb.) of adjacent Alberta (BRITISH COLUMBIA, 2007). The affected area has extended further north than previous outbreaks and encompassed much of the distribution of white pine species (CAMPBELL and CARROLL 2007).

White pine populations have persisted for eons on landscapes in which mountain pine beetles occasionally eliminated nearly all the mature trees within an outbreak area. Past outbreaks declined when food sources (mature trees) were depleted (KIPFMUELLER et al. 2002; GIBSON et al. 2008). These outbreaks were typically followed by large, severe fires which prepared seed beds for the regeneration that over time renewed the white pine forest. The addition of a severe infestation of *C. ribicola*, however, could detrimentally disrupt this cycle of outbreak, fire and renewal (ARNO 2001; TOMBACK and ACHUFF 2010).

The ecological and genetic interactions are varied and multiple among mountain pine beetle, blister rust and white pine (symbionts, different organisms in an intimate association). The beetles display alternative behaviours of attraction, avoidance or indifference towards rust-infested white pines; the behaviour may differ with the severity of the infestation or the stage of the beetle outbreak (see KEGLEY et al. 2004; GIBSON et al. 2008). White pine trees are disease-free or less severely infested because they are either resistant or they are susceptible but have escaped infection (inheritable resistance may or may not be subject to natural selection). Interactions among the symbionts at one generation can affect population size and genetics in the following generations. For example, the condition of a host white pine influences the brood-productivity of a beetle pair (fitness); and beetle selection behaviour (attraction, avoidance, or indifference) affects frequency of resistance (if present) to blister rust in the white pine. According to one scenario, beetles kill more of the resistant trees; the remaining susceptible trees produce a greater proportion of seedlings; but blister rust and associated organisms kill the susceptible seedlings; white pine seed dispersers fail to restock the site; over generations, white pine disappears from the stand. However, beetles sometimes prefer rust-infested trees (SIX and ADAMS 2007); so, alternative outcomes could occur. Consequently, no single scenario is sufficient to characterize the complex, co-evolutionary network of white pines and their biotic associates (see THOMPSON 2005).

2.2 Root diseases

Several pathogens cause root disease and/or butt decay in white pines, including—*Armillaria ostoyae* (Romagn.) Herink, *Heterobasidion annosum* (Fr.:Fr.) Bref., *Leptographium wageneri* (W.B. Kendr.) M.J. Wingf., *Phaeolus schweinitzii* (Fr.) Pat., *Phellinus*

sulphurascens Pilát [syn. *Phellinus weirii* (Murr.) Gilb.], and *Perenniporia subacida* (Peck) Donk (HUBERT 1950; HEPTING 1971; HOFF and HAGLE 1990; JACOBS and WINGFIELD 2001). However, white pines are considered generally more tolerant to root diseases than true firs (*Abies*) or Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco]. Before introduction of *C. ribicola* into the Inland Northwest, root disease killed the more root-disease-susceptible conifers in mixed-species stands and thereby created openings for white pines to release and regenerate (HAIG et al. 1941; NEUENSCHWANDER et al. 1999). After introduction, rust-caused mortality and accelerated harvest led to significant loss of white pines. These disturbances, along with fire suppression (reducing opportunities for white pine regeneration), converted millions of hectares to poorly stocked stands of species highly susceptible to root disease (HAGLE et al. 2000). Besides conversion to root-disease-susceptible species, mortality and harvest also generated an abundance of snags and stumps, a food-base that might have allowed root disease fungi to assume a new, dominant role in the development of formerly white pine forests (BYLER et al. 1990). By generating canopy gaps, root disease encourages regeneration of *Ribes* and spread of blister rust (HUNT 2000). In areas with actively aggressive *Armillaria*, young white pines are susceptible to root disease (S. Hagle, personal communication).

2.3 Dwarf mistletoes

Dwarf mistletoes (*Arceuthobium*) are native parasitic plants that attack all sizes of trees, reducing host growth and survival (HAWKSWORTH and WIENS 1996). Several species of dwarf mistletoe are specific pathogens of white pines in the western states—*Arceuthobium apachecum* Hawksw. and Wiens and *A. blumeri* A. Nels. on southwestern white pine; *A. californicum* Hawksw. and Wiens on sugar pine; *A. cyanocarpum* (A. Nels. ex Rydb.) Coulter and A. Nels. on several white pine species; and *A. monticola* Hawksw., Wiens and Nickrent commonly on western white pine. These and other species of dwarf mistletoes that infrequently infect white pines (see MATHIASSEN and HAWKSWORTH 1980) are generally found only within a portion the host distribution and in most stands have a minor impact. *Arceuthobium cyanocarpum*, however, is an important but highly localized mortality agent of limber pine in the Rocky Mountains and of whitebark pine on Mt Shasta, California (MATHIASSEN and HAWKSWORTH 1988; TAYLOR and MATHIASSEN 1999).

2.4 Foliage diseases

Dothistroma septosporum (teleomorph: *Mycosphaerella pini*) causes a needle blight in many pine species (COBB and MILLER 1968; FUNK 1985; BARNES et al. 2008). This defoliator was implicated in a serious decline of young limber pine in eastern Montana (TAYLOR and WALLA 1999); JACKSON and LOCKMAN (2003) reported on the amount and causes of mortality associated with this decline. Cumulative mortality on a series of plots monitored from 1996 to 2002 was 42%; severe defoliation by *D. septosporum* was determined to be the principal cause of mortality for 80% of dead limber pine. Although present on these plots, *C. ribicola* was a minor contributor to mortality. Other epidemics of *D. septosporum* have occurred in western white pine (SHAW and LEAPHART 1960) and other pine species (JAMES 1981), but significant mortality in natural stands is unusual and usually confined to small trees (PETERSON 1981).

2.5 Cone and seed insects

Cone and seed insects occasionally cause major losses to white pines in production orchards (HEDLIN et al. 1980; CORDELL et al. 1989). The western conifer-seed bug (*Leptoglossus occidentalis* Heidemann) reduced the number of filled seed by >70% and

reduced seed yield per cone fivefold (CONNELLY and SCHOWALTER 1991). SHEA et al. (1983) reported that cone beetles (*Conophthorus ponderosae* Hopkins) killed >90% of a 1979 crop. To avoid these losses, western white pine orchards are now routinely sprayed for cone and seed insects. KEEN (1958) found 50–80% of limber pine cones were destroyed by cone beetles; NEBEKER (1970) observed that coneworms (*Dioryctria abietivorella* Grote) infested up to 40% of cones. In some years, coneworms, seed bugs and cone beetles cause significant losses to whitebark pine seed crops throughout the Pacific Northwest (KEGLEY et al. 2001b).

2.6 Wildlife

Many species of wildlife have a significant affect on the growth and regeneration of white pines (GILL 1992). Damage from black bear (*Ursus americanus*), elk (*Cervus elaphus*) or deer (*Odocoileus*) is a problem for white pine silviculture in some areas. Bears girdle small trees to obtain nutrients in the bark (NOLTE 2003). Pruned western white pines in northern Idaho and British Columbia are especially subject to bear damage (MOLNAR and McMINN 1960). Elk and deer rub their antlers on pruned trees near game trails (SCHNEPF and SCHWANDT 2006); they also browse seedlings (ZEGLEN et al. 2010). Small mammals and birds are important in the seed dispersal and predation of some white pine species. The mutual dependence of Clark's nutcracker (*Nucifraga columbiana*) and whitebark pine is thoroughly reviewed by TOMBACK and ACHUFF (2010). Nutcrackers usually benefit pines by dispersing their seeds; but where cone crops are limited, nutcrackers may either eat all the seeds or vacate an area without caching any seeds, thereby contributing to pine regeneration failure.

2.7 Fire

Fire is both friend and foe to white pines. Historically, wildfires recycled overmature stands and created regenerative opportunities favouring white pines over other species. However, fires also hinder white pine regeneration by overheating soils (MCDOWELL 2007), consuming small populations, destroying rust-resistant trees, and stimulating regeneration of *Ribes*. On the contrary, fire suppression impacts the long-term persistence of seral populations of white pine. Fire suppression favours regeneration of more shade-tolerant species (such as grand fir, *Abies grandis* Dougl. ex D. Don) that eventually out-compete and replace seral western white pine and sugar pine. For example, western white pine was the dominant species in 1930 on 30% of a forested region in northern Idaho (NEUENSCHWANDER et al. 1999). Over the next 50 years of fire suppression, other management practices, diseases, and plant succession, the area of western white pine dominance shrank to less than 5%; and the portion where Douglas-fir and grand fir were the dominant species expanded from less than 20% to more than 45% (SAMMAN et al. 2003). But, fire suppression in other areas allowed limber pine to expand, at least temporally, into dry shrublands and grasslands. Limber pine on these sites, however, are vulnerable to drought-related mortality (SCHUSTER et al. 1995) and are occasionally but severely damaged by otherwise minor foliage diseases (TAYLOR and SCHWANDT 1998).

2.8 Climate change

Climatic changes over the next century could significantly affect white pine populations both directly and through influences on bark beetles, blister rust, and other pathogens (CAMPBELL and CARROLL 2007; KLIEJUNAS et al. 2009). These complex interactions present serious complications for maintaining the distribution and importance of white pines in western North America.

A warmer climate could be especially detrimental to whitebark pine. In a warmer climate, the species' fundamental habitat would shift to cooler sites at higher elevations and higher latitudes. Whitebark pine could persist on the landscape—if such habitats existed, if sufficiently rapid migration were possible, and if the species were sufficiently capable of adapting. WARWELL et al. (2007) used a conservative model of climate change and found a greatly reduced area of suitable, future habitat for whitebark pine. The likelihood of sustaining whitebark pine, even in suitable habitats, is further diminished if populations are small (owing to random events, the Allee effect, see SCHERM et al. 2006).

An important constraint on mountain pine beetle outbreaks in whitebark pine is climate, especially the frequency of severely cold winters and brief warm summers (CAMPBELL and CARROLL 2007). In warmer winters, more brood could survive; and, with a sufficiently long season, two generations per year could be produced (BENTZ and SCHEN-LANGENHEIM 2007; GIBSON et al. 2008). Although mountain pine beetle already produces two generations per year in sugar pine, the consequence of doubling the reproductive potential in a subalpine forest of whitebark pine rather than a montane forest of sugar pine is not readily apparent.

Changes in timing and duration of warmth and moisture could have major influences on the epidemiology of *C. ribicola*. By some climate scenarios, summers are drier in the Rocky Mountains but wetter in the Southwest (BARTLEIN et al. 1997; KLIEJUNAS et al. 2009). As infection of white pine by blister rust requires a cool, moisture-saturated environment, conditions suitable for infection in some regions might be restricted to fewer wet periods in spring or early summer but extended in other regions. Early season infection of pine is common for infestations in the coastal regions of British Columbia (HUNT 2005). In the southern Sierra Nevada, infection of white pine normally coincides with spring rains and summer thunderstorms (KINLOCH and DULITZ 1990). A sufficiently long and cool winter is required before *Ribes* break dormancy and become susceptible telial hosts (see ZAMBINO 2010). Climatic change may also have a major effect on other pathogens such as those causing foliage disease (KLIEJUNAS et al. 2009). For example, if increased warmth was accompanied by increased moisture, defoliation by *Dothistroma* may lead to widespread mortality, as occurs to lodgepole pine in British Columbia (WOODS et al. 2005).

3 Strategies

The management of western North American forests and approach to control of *C. ribicola* was significantly realigned in the 1960s (KETCHAM et al. 1968). Strategy planners realized that *Ribes* eradication was not only costly but that numerous other disturbances in addition to white pine blister rust caused forest losses. In response, harvesting white pine was accelerated; white pine planting was discontinued; and silviculture switched to favour other species. Owing to progress in genetics research (BINGHAM et al. 1960), operational, tree-breeding programmes began providing improved planting stock better able to survive blister rust (KING et al. 2010). Forty years later, there are new challenges, new opportunities, and a new look at the strategic goals and approaches.

SAMMAN et al. (2003) concluded that without effective intervention, blister rust would continue to spread and adversely impact white pine ecosystems through host mortality, reduced genetic diversity, and impaired function. On U.S. federal lands, species conservation was mandated by the *National Forest Management Act of 1976* and the *Healthy Forest Restoration Act of 2003*. In addition, the *National Fire Plan* called for maintaining or restoring ecosystem health to forests threatened by fire.

The *National Strategy and Implementation Plan for Invasive Species* (RIES et al. 2004) characterized four, general, programme stages: 1) prevention; 2) early detection and rapid response; 3) control and management; and 4) rehabilitation and restoration. This plan also identified important, strategic elements of partnerships, science, communication and

education, and organization for success. Use of certain intervention actions in specific circumstances are constrained by operational and administrative factors such as cost-effectiveness and protection of wilderness values. These stages, elements, and constraints are incorporated into a strategy adapted for white pine blister in western North America. In response to the threats from blister rust, the strategic goal is to sustain or restore viable white pine populations in western forest ecosystems. The action components of the strategy are to:

- Conserve genetic resistance to *C. ribicola*.
- Reduce the risk of adverse impact in stands currently uninfested.
- Restore and maintain white pines where blister rust is causing impacts.
- Assess and monitor the health and management of white pines.

3.1 Conserve genetic resistance to *Cronartium ribicola*

The first and most fundamental component of the strategy for managing *C. ribicola* is to conserve host genetic resistance (SAMMAN et al. 2003; SCHWANDT 2006; SCHOETTLE and SNIEZKO 2007). KING et al. (2010) reviews traditional genetic approaches and programmes for managing white pine blister rust. New, molecular tools are rapidly expanding our understanding of the genetics, mechanisms, and evolution of resistance and virulence in the blister rust pathosystem (RICHARDSON et al. 2010). Management actions for promoting conservation of genetic resistance are to select, breed, and deploy resistant planting stock and to protect sources of resistance genes from loss by fire, competition, insects, pathogens, and other disturbances (OSTRY et al. 2010; ZEGLIN et al. 2010).

3.1.1 Identify and protect resistant trees

Many national and provincial forests have genetics programmes to identify and protect selected trees resistant to blister rust. Selected trees that pass genetic screening may either be included in breeding orchards or used *in situ* to provide seeds. MAHALOVICH and DICKERSON (2004) documented procedures for identifying phenotypically resistant trees. Trees with no or few blister rust cankers found among many, heavily infected trees are assumed to have been challenged by the rust and might possess inheritable traits of resistance or tolerance (putatively resistant). Parent trees with desirable genes are monitored for a cone crop and, if needed, protected from mountain pine beetle, wildfire, and seed predators.

3.1.2 Establish screening and breeding programmes

Regional and provincial genetics programmes screen, breed, store, and propagate white pine planting stock (seed and seedlings). Genetic screening determines the heritability of adaptive traits in the progeny. Along with associated research projects, the programmes insure deployed stock is well-adapted to the site, possesses effective resistance, and provides genetic diversity resilient in a changing environment. Specific programme activities: 1) determine and establish seed collection zones; 2) collect seed from putatively resistant trees; 3) screen progeny for resistance; 4) establish seed orchards or designate natural seed production areas; 5) conduct crosses to concentrate resistance mechanisms; 6) collect, process, and supply seed and seedlings for reforestation; and 7) monitor realized-gain trials and operational plantings for long-term performance (SAMMAN and KITZMILLER 1996; KITZMILLER and DUNLAP 2004; MAHALOVICH and DICKERSON 2004).

Genetics programmes for western white pine and sugar pine were established prior to 1960 and for other species more recently (KING et al. 2010). Programmes are administered at genetic resource centers in California, Oregon, Idaho, and British Columbia. Different programmes have pursued various approaches to provide genetic resistance and diversity. The Idaho programme focuses on partial resistance for greater host survival (tolerance) and reduced selection pressure that might favour a virulent rust (durability). The California programme combines R-gene resistance for immunity with partial resistance for protection against a virulent race. Other programmes use mixed strategies. Although a greater reforestation effort for western white pine and sugar pine could provide ecological benefits, current planting opportunities are more limiting than the availability of seed with improved resistance. A programme for whitebark pine has begun screening from some seed zones. Other species have received less attention, but preliminary screening indicates they have sufficient resistance to justify a selective breeding programme (VOGLER et al. 2006).

3.1.3 Protect trees with known resistance

Individual trees prospectively providing resistant seed are valuable genetic resources. For these trees, extraordinary protections are to: 1) cage cones to prevent foraging by squirrels and birds; 2) apply insecticides in seed orchards to minimize cone and seed damage and *in situ* to prevent bark beetle attack; and 3) thin and clear adjacent vegetation to reduce fire risk. Select high-elevation white pines are often located on remote or wilderness sites; so, they are especially difficult to protect from seed predators, bark beetles, and fires. Some equipment, even visible tags, are prohibited within some wilderness areas; and permitted activities vary by administrative unit. High-risk trees in accessible areas may warrant protection with insecticides or pheromones, but operational procedures require further development (KEGLEY and GIBSON 2004). Lodgepole pine are removed near white pine seed orchards in British Columbia to reduce the threat from mountain pine beetles (R. Hunt, personal communication). Trees identified as genetic resources are candidates for designated protection from wildfire (see http://www.fs.fed.us/rm/wfdss_ravar).

3.2 Reduce the risk of adverse impact in stands currently uninfested

Although continued spread of white pine blister rust could be inevitable, many white pine populations can escape serious impacts owing to environmental conditions and proactive interventions. *Cronartium ribicola* is an adaptable pathogen with substantial capacities for reproduction and dispersal. But, white pines can regenerate and persist in environments of aridity, heat, or lacking telial hosts that are unfavourable for blister rust. White pines can tolerate infection and white pine populations can increase the frequency of genes for resistance. In areas where blister rust has not yet adversely impacted white pine, proactive interventions are: 1) anticipate and delay spread, 2) detect new infestations early and respond rapidly, 3) promote ecosystem resilience to an invasive infestation, and 4) assist with white pine conservation, management, and protection.

3.2.1 Anticipate and delay spread

Because *C. ribicola* requires transport and establishment of a perennial infestation for successful invasion, its spread can be anticipated and delayed. The natural spread of blister rust into an uninfested landscape can be modeled as long-distance, aerial dispersal of aeciospores (FRANK et al. 2008). Hazard mapping based on climatic and vegetation features provides an early warning of where the first, natural infestations are likely to establish (see OSTRY et al. 2010; ZEGLEN et al. 2010). The failure of past quarantines in preventing

anthropogenic spread (GEILS et al. 2010) points to the importance of education, regulation, and inspection to discourage and reduce transplanting of infected nursery stock or wildlings. Anthropogenic spread is likely to introduce blister rust to sites other than native stands such as parks which provide a favourable environment for establishment.

3.2.2 Detect infestations and respond rapidly

Experience demonstrates that, even in warm, dry regions of generally low hazard, blister rust infestations can develop on favourable sites such as along streams. During the one or two decades between the first infection and the initial detection, blister rust can intensify on numerous hosts over a considerable area (see ZAMBINO 2010). Early assessment identifies the threats and the values at risk for the newly infested landscape. Useful information includes: 1) regional hazard and likelihood of additional spread from and to other landscapes; 2) the extent and distribution of high-hazard sites and highly susceptible hosts capable of supporting significant inoculum production; 3) stage and rate of intensification of the epidemic; 4) resource and ecosystem values threatened both locally and beyond; and 5) costs and constraints on control and management options. If an infestation was localized, re-introductions were unlikely, and constraints were not prohibitive, then early eradication could be justified by the resource values and ecosystem services to be protected.

3.2.3 Promote ecosystem resilience

In environments less favourable for blister rust, many stands have so far escaped infestation but remain threatened by serious impacts in the future. SCHOETTLE and SNIETKO (2007) proposed proactive intervention to encourage natural regeneration for increasing the age and size-class diversity of white pine populations. Although greater diversity may not prevent a blister rust epidemic, it may increase population resilience and mitigate long-term impacts.

Several observations support the assertion that promoting natural regeneration of white pines would beneficially affect the population response to a new blister rust epidemic (SCHOETTLE 2004b). High-elevation white pines typically display delayed, infrequent, and sparse reproduction offset by a long lifespan (TOMBACK and ACHUFF 2010). Small, immature white pines are highly susceptible to rust infection and even a single canker can be lethal (GEILS et al. 2010). Although the effective target size presented by mature white pines appears to be large, this klendusic factor for infection (see WICKER and SHAW 1967) is offset by survival factors of age-related resistance, escape, and tolerance that allow these trees to persist within infested stands for many decades. And, as white pines are only infected by basidiospores produced on *Ribes* (or other telial hosts), *C. ribicola* is not contagious from pine to pine. An increased density of large-crown, mature trees could lower the infection rate by shading out *Ribes* and screening spores. The premise of proactive intervention is that more young white pines would increase genetic diversity and the likelihood of retaining rare genes for resistance. Then, when blister rust arrives, it is more likely that some resistant trees would survive, reproduce, and increase the frequency of resistance genes. On accessible sites, seeding or planting stock from a genetic improvement programme could further enhance evolution for resistance. The cautions for relying on white pine regeneration are that coevolutionary interactions are complex—*C. ribicola* also evolves; resistance and virulence can impose their own costs on genetic fitness (ZAMBINO 2010); and change happens. Although the specific outcome of encouraging white pine regeneration is unknowable, managing for diversity at the landscape, community, and species levels remains a useful approach for promoting ecosystem resilience.

3.2.4 Assist with conservation, management, and protection

Cronartium ribicola is an invasive pathogen capable of spread to white pine populations in non-infested regions in the Southwest, Great Basin, and Mexico. Experience and knowledge gained from working with blister rust epidemics can assist with the conservation, management, and protection of white pines in regions threatened but not yet infested. Cooperation can help with developing local programmes and tools for threat assessment, early detection, site-hazard modelling, response and rehabilitation planning, genetic conservation, tree improvement, and silvicultural techniques for disease control and mitigation. Areas where white pines can grow without blister rust (e.g., lacking significant populations of telial hosts) can serve as *ex-situ* conservation refugia where natural or artificial genetic diversity is maintained.

3.3 Restore and maintain white pines in ecosystems where blister rust is causing impacts

Where *C. ribicola* causes unacceptable damage, mortality, and loss of reproduction within intensively managed stands, silvicultural treatments are available to regenerate stands and control disease by removing blister rust-infected branches, trees, and *Ribes*. OSTRY et al. (2010) and ZEGLEN et al. (2010) provide detailed reviews of these treatments. Here, we briefly identify the primary tactics, their role in a strategy for mitigating rust impacts, and constraints or cautions to their use.

3.3.1 Regenerate

After genetically resistant individuals are identified and improved planting stock is available, this material only contributes to sustaining or restoring white pine populations if it is deployed successfully and performs adequately. Resistant planting stock is used where a natural seed source with a satisfactory level of resistance is insufficient for regeneration (HOFF et al. 2001). Specific decisions on deploying resistant material consider management objectives, local epidemic history, availability of planting stock, and site hazard. The planting goals are to provide short-term, adequate stocking and survival and to maintain or even enhance the population for long-term genetic adaptability and resilience.

MULLER (2002) and SCHWANDT and FERGUSON (2003) assessed the performance of improved stock in more than 60 plantations of western white pine in northern Idaho. Their surveys revealed blister rust incidence ranged from 0 to 90% and could significantly change over just a few years. Nonetheless, in plantations where incidence was high, genetically improved trees outperformed natural regeneration in terms of infection and survival.

Canada and the United States have diverged in their use of planting in harvested stands. ZEGLEN et al. (2010) describe British Columbia guidelines that recommend planting western white pine after harvest. In the United States, however, there has been a significant reduction in harvesting western white pine and sugar pine and consequently a decline in planting. In 1991, a planting peak of western white pine was reached at 7505 ha; in 2001, only 1434 ha were planted—a five fold reduction that persisted in subsequent years (Schwandt, unpublished data). With fewer areas harvested and planted, there is less opportunity to deploy resistant white pine. Although reserves are accumulating, seeds are perishable.

Only minimal reforestation planting of other white pine species has occurred. Although Rocky Mountain bristlecone pine and southwestern white pine are planted as ornamental landscaping trees, they are minor species in reforestation. Whitebark pine and limber pine are planted in some forest areas; trials have begun for testing the efficacy of planting whitebark pine seeds and developing procedures for planting in remote areas (SCHWANDT et al. 2007).

3.3.2 Prune and thin

Silvicultural pruning, often accompanied with thinning, reduces the number of blister rust-infected stems and may affect re-infection and host mortality (SCHNEPF and SCHWANDT 2006; ZEGLEN et al. 2010). Pruning either sanitizes high-value trees by removing infections or prevents future infection and loss by reducing the potential of lethal infection. Pruning is not used for sugar pine which is commonly infected high in the crown, but it is used for western white pine where infections are usually in the lower crown. Pruning is unnecessary in stands protected by R-gene resistance (see KING et al. 2010). SCHWANDT et al. (1994) and SCHWANDT and MARSDEN (2002) assessed the interactions of pruning and thinning; for the situations they examined, pruning or pruning and thinning provided long-term, stand-level benefits to stand-volume yield and log value (size and quality).

Although thinning can mitigate blister rust impacts and reduce the risks of fire and beetle attack, it could compromise other tactics. Thinning decreases self-pruning of lower branches and thereby increases the frequency of lethal infection (HUNGERFORD et al. 1982; SCHWANDT et al. 1994). Thinning that sufficiently opens the canopy might stimulate *Ribes* growth or reduce the screening of dispersing spores (ZAMBINO 2010).

3.3.3 Burn

The ecological relations of fire in the blister rust pathosystem are complex. Fire affects the regeneration and development of white pine stands (TOMBACK and ACHUFF 2010); many white pine species regenerate after fire prepares the site by reducing competition and exposing mineral soil (KEANE and ARNO 2001). Opportunities for regeneration are produced by prescribed fires and naturally-ignited fires. Not all fires, however, even prescribed burns, provide a net benefit for white pine regeneration and blister rust control. A burn that generates the seed bed for white pine may also favour reproduction and growth of *Ribes* and other telial hosts (ZAMBINO 2010).

3.3.4 Remove *Ribes*

Because *C. ribicola* requires *Ribes* (or other telia host) to complete its life cycle, *Ribes* eradication was the principal strategy for controlling blister rust and protecting white pine timber crops for many decades (GILS et al. 2010). Although eradication may have been practical where *Ribes* could be easily and inexpensively removed with few ecological impacts and demonstrable silvicultural benefit, *Ribes* eradication in western North America today is problematic (HUMMER and DALE 2010; ZAMBINO 2010). Although *Ribes* control would not prevent every infection, it could augment other silvicultural tools in certain situations and high-value trees, such as ornamental plantings, nurseries, and seed orchards. ZAMBINO (2010) describes conditions for *Ribes* reproduction, and ZEGLEN et al. (2010) identifies techniques for *Ribes* removal. A constraint to eradication is that certain *Ribes* species are either protected themselves or serve as important food sources for wildlife species (ZAMBINO 2010). Various statutes and policies restrict *Ribes* removal (*Endangered Species Act of 1973*, <http://www.fws.gov/endangered/esaall.pdf>; *National Environmental Policy Act of 1969*; and *National Forest Management Act of 1976*, <http://www.fs.fed.us/emc/nfma/>).

3.4 Assess and monitor the health and management of white pines

Several management and research agencies are responsible for monitoring and assessing conditions and trends in white pine populations and the management to sustain or restore those populations. Monitoring detects infestations, tracks disease progress and stand

development, documents distributions and trends, and provides data to assess management effectiveness and to test hypotheses. In addition to forest inventory and analysis plots (<http://www.fia.fs.fed.us>), forest health monitoring plots are established for most white pine species within at least a portion of their distribution. Recent surveys have emphasized high-elevation white pines (TOMBACK et al. 2005a; SHOAL and AUBRY 2006; KEARNS and JACOBI 2007; KLIEJUNAS and DUNLAP 2007). Because procedures vary, communication and data archiving are important to provide current assessments and identify trends. LOCKMAN and DENITTO (2006) developed and distributed a west-wide catalog of surveys for limber pine and whitebark pine. While monitoring and assessment have traditionally tracked only population numbers, technologies are emerging that can provide the genetic information useful for conservation and management (see RICHARDSON et al. 2010).

4 Knowledge gaps and management challenges

We have learned much about *C. ribicola* since its introduction nearly 100 years ago. However, there is much we still do not know about white pine blister rust, its interactions with various hosts and biotic associates, and its behaviour in different environments. We do not know how much *C. ribicola* varies in virulence from eastern to western North America and the potential for races with greater virulence, fitness, and aggressiveness. We have much to learn about the genetics and evolution of resistance, growing and outplanting white pines, and the effects of climatic changes on the blister rust pathosystem.

Surveys in the 1960s and 1970s (MACGREGOR 1969; BYLER and PARMETER 1979) suggested that the relatively hostile environment for *C. ribicola* in the California Sierra Nevada would confine epidemics to erratic but manageable infestations. Blister rust, however, continued to spread and intensify into what had been thought unfavourable sites. Informed analysis of additional data across scales from the regional to the microsite could provide better descriptions and hypotheses for the hazard and development of rust epidemics in a wide variety of environments.

The strategy to reduce impacts from blister rust depends on the availability of planting stock screened and bred for disease resistance and the opportunity to plant it. Even for long-studied western white pine and sugar pine, our understanding of the mechanisms and inheritance of resistance and how blister rust defeats it is relatively limited in comparison to agronomic crops. For the high-elevation white pines, we are only beginning to collect basic data on the genetics, genomics, and proteomics of resistance or other adaptive traits. Knowledge of coevolutionary interactions is very sparse for the many organisms involved in the pathosystem.

The challenge is to advance science and its application from traditional genetics based on surviving phenotypes to molecular genetics based on understanding gene expression and pathogenesis. There is also much progress to be made in various operational, cultural practices. Although procedures are well developed for growing and screening seedlings of western white pine and sugar pine for resistance, initial experience with other species suggests that additional modifications would be helpful for these white pines. Production of genetically improved stock might be accelerated and costs reduced if procedures for grafting resistant stock were developed.

Presently, the chief constraint to increasing white pine survival and resistance through artificial selection in the United States is slow deployment owing to a lack of suitable planting sites. The opportunity for white pine reforestation is usually created by the silvicultural treatments of harvesting and prescribed burning. In recent years, fewer areas have been treated, fewer sites prepared, and fewer trees planted. Between 1976 and 1996, resistance-improved stock of western white pine was planted over 100 000 ha in the Inland Northwest, about 5% of the species' historical distribution in the region (FINS et al. 2002). At the current rate of planting, however, another 60 years would be required to plant

another 5%. As fewer stands are regenerated under silvicultural prescription, wildfire and other natural events are expected to become more common. These also can create conditions suitable for white pine regeneration; given adequate resources and planning, these sites could be reforested with resistance-improved stock. Whether a site was created and regenerated naturally or by treatment, monitoring is important to document long-term gains and to identify where thinning and pruning would be beneficial.

In western Canada, active forest harvesting creates many opportunities for planting western white pine; the supply of genetically improved stock is adequate to meet demand. Because of the threat of blister rust, however, many foresters are reluctant to plant white pines. Especially for the interior districts, resistance is sufficient that the risk from blister rust need not be prohibitive when used with appropriate silvicultural practices. Education as information and demonstrations of successful white pine forestry could lead to increased white pine reforestation.

Many populations of high-elevation white pines present special management challenges owing to constraints on access and permitted treatments. At remote locations or within wilderness areas, silvicultural practices that require frequent or intrusive action, such as caging cones or cutting trees, may be impractical or conflict with wilderness values. Alternative, resistance-deployment techniques that could be developed for these sites include direct seeding and grafting. Monitoring is important for tracking population conditions and assessing operational effectiveness.

Management and research programmes for sustaining white pines and mitigating the impacts of blister rust have been long-established for western white pine and sugar pine and recently for whitebark pine and other white pines (FINS et al. 2002; KITZMILLER and DUNLAP 2004; SCHWANDT 2006; AUBRY et al. 2008; BURNS et al. 2008; CONKLIN et al. 2009). Gene conservation programmes are in planning (TKACZ 2008). Success of these programmes depends on many factors and cooperation from numerous managers, researchers, and stakeholders coordinating their work and support across management levels and regions. Factors critical for successful management by learning and adapting include: (1) a sound scientific foundation; (2) commitment of adequate and sustained resources and (3) good planning, communications, implementation and monitoring. Although white pine ecosystems of western North America are seriously threatened by various biotic and environmental risks, we have presented a broad strategy using genetic and silvicultural techniques to guide management, research and stakeholders for confronting the complex, challenging and changing problems in sustaining healthy forests.

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