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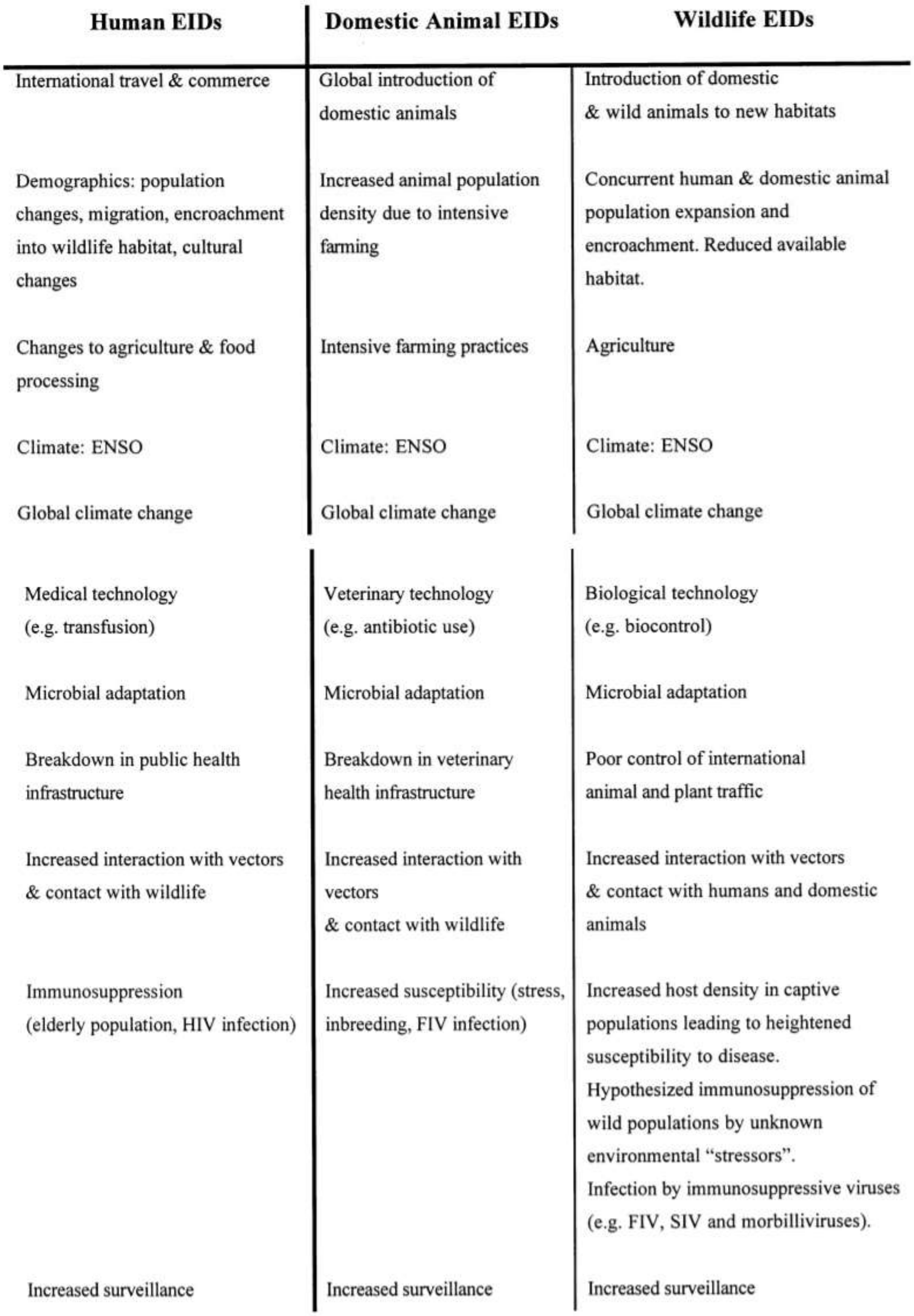
Pests and pathogens are a significant stressor to the health of forests in the Pacific Northwest. In the past, pests such as native bark bugs that feed on trees have been seen as a part of the wildlife system. However, traditional policies have been to not interfere with these processes. Fungal and animal pathogens were likewise considered a part of that system. However, recent studies have shown that diseases are often anthropogenic in nature, whether by their introduction into a non-native environment or by lowering resilience by their vectors, which allows for more widespread infection (Buttke et al., 2021).

Diseases have three main components to it: a host, an agent, and an environment. The stable balance between disease and host that leads to a healthy ecosystem can quickly succumb to deleterious disease infestation if any of these components are shifted (Daszak et al., 2001). Land uses such as logging—which often lowers forest biodiversity to the same age and tree species across an entire sector—and habitat fragmentation are both anthropogenic factors that have disturbed ecosystems. Disturbed ecosystems shift the environmental component of disease transmission, making them more prone to infectious diseases than others.

Climate change directly affects pest and pathogen transmission by causing changes in insect and fungal physiology in response to temperature, precipitation, weather patterns, or other factors, and indirectly through its impact on hosts and the environment (Halofsky et al., 2022). The health of a host plant can often determine the ability of infestations to grow. Trees under stress through climatic changes such as increased wildfires and drought are more prone to bark beetles or root rot infections. Drought weakens the tree's resilience, and the physiological changes trees go through in response to drought can act as a lure for herbivorous insects due to the high concentrations of nitrogen compounds and sugar in the newer plant tissue (Vose et al., 2019).

The introduction of foreign pests is a large contributing factor to the concern of pests and pathogens. One study has shown that 19 of the 70 major insect pests in U.S. forests are exotic insects that have been anthropogenically introduced past barriers such as the ocean (Liebhold et al., 1995). The causes of Emerging Infectious Diseases (EIDs) are numerous and often overlap in social, commercial, and environmental sectors, as can be seen in Figure 1 (Daszak et al., 2001).

The purpose of this paper is to illuminate insects and fungal infections as two main points of concern for the health of the forest ecosystem as climate change continues to affect the Pacific Northwest. A case study for each pathogen will be included to emphasize the current effects these pathogens have on forestry in the Pacific Northwest, followed by possible options for adaptation and mitigation in these specific scenarios.



Human EIDs	Domestic Animal EIDs	Wildlife EIDs
International travel & commerce	Global introduction of domestic animals	Introduction of domestic & wild animals to new habitats
Demographics: population changes, migration, encroachment into wildlife habitat, cultural changes	Increased animal population density due to intensive farming	Concurrent human & domestic animal population expansion and encroachment. Reduced available habitat.
Changes to agriculture & food processing	Intensive farming practices	Agriculture
Climate: ENSO	Climate: ENSO	Climate: ENSO
Global climate change	Global climate change	Global climate change
Medical technology (e.g. transfusion)	Veterinary technology (e.g. antibiotic use)	Biological technology (e.g. biocontrol)
Microbial adaptation	Microbial adaptation	Microbial adaptation
Breakdown in public health infrastructure	Breakdown in veterinary health infrastructure	Poor control of international animal and plant traffic
Increased interaction with vectors & contact with wildlife	Increased interaction with vectors & contact with wildlife	Increased interaction with vectors & contact with humans and domestic animals
Immunosuppression (elderly population, HIV infection)	Increased susceptibility (stress, inbreeding, FIV infection)	Increased host density in captive populations leading to heightened susceptibility to disease. Hypothesized immunosuppression of wild populations by unknown environmental "stressors". Infection by immunosuppressive viruses (e.g. FIV, SIV and morbilliviruses).
Increased surveillance	Increased surveillance	Increased surveillance

Figure 1. Common underlying causes of human, domestic animal, and wildlife Emerging Infectious Diseases. There is a high overlap between the categories, demonstrating an interdisciplinary link between social, economic, and environmental factors. Figure taken from Daszak, 2001.

Insects

Climate change offers extreme difficulty in predicting future trends for insect infestation because many occurrences are not uniform or generalizable across several systems. Temperature has effects on insects and their environments in multiple ways, some which are beneficial to the growth of insect populations and some which decrease the viability of survival. Warmer temperatures increase insect consumption, development, movement, and range (Halofsky et al., 2022). Decreased time for reproduction of the spruce beetle *(Dendroctonus rufipennis* Kirby) has contributed to damage to spruce forests in northwestern North America (Robinet et al., 2010). Colder temperatures act as a limiting factor toward range distribution for many species. In temperate zones such as the Pacific Northwest, low lethal temperatures limit the range of insects such as the spruce beetle. Since 2000, increasing temperatures have led to outbreaks at northern latitudes (Robinet et al., 2010). This development is problematic because the ecosystems of northern latitudes are often not adapted to the presence of these new pathogens, often lacking biological resilience in the vectors or predatory mitigation.

Case Study: Mountain Pine Beetle (*Dendroctonus ponderosae* Hopkins)

Insect infestations are more prevalent than wildfires in western North America. The two most prevalent native insect infestations are mountain pine beetle (*Dendroctonus ponderosae* Hopkins) and western spruce budworm (*Choristoneura freeman* Razowski) (Meigs et al., 2015). An investigation into the mountain pine beetle (MPB) provides the opportunity for establishing the current state of affairs in insect infestation as well as a chance to look into adaptation and mitigation policies that have been and could be implemented to deal with this ongoing issue.

MPB is a native species to western North America conifer forests, specializing in host species such as lodgepole pine and ponderosa pine; however, most species of pine native to North America are suitable hosts (Sambaraju et al., 2021). Its range spans across much of western North America; from southern British Columbia, Canada, east to South Dakota and south to Baja, California, Mexico, and New Mexico, pictured in Figure 2. Its range appears to be expanding northward and eastward in response to the changing climate (Audley et al., 2020). It is characterized as an aggressive species due to the mass numbers of beetles that swarm a tree at a single time, their efficient feeding causing tree death within a year (Sambaraju et al., 2021). They generally feed on trees that are weakened or stressed, whether by fire, drought, or previous pathogen influence such as root rot; however, MPB is one of the bark beetles that are capable of killing healthy trees through pheromone-mediated mass attack (Bleiker et al., 2014). Their host is identified using visual and semiochemical-based attraction to host trees. The process of swarming begins when a female selects and bores into the bark of a host tree and constructs galleries in the phloem and outer sapwood. Release of aggregation pheromones attracts female and male conspecifics, the latter of which contributes to colonization by releasing a synergizing pheromone (Sambaraju et al., 2021).

The density of the mass attack determines the ability for MPB to kill healthy trees. At low densities MPB acts as an endemic species that is not harmful to its environment, instead acting as a disturbance agent that contributes to the removal of diseased and stressed trees. This in turn helps with ecosystem processes such as nutrient cycling and forest succession (Sambaraju et al., 2021). However, when densities of MPB increase in response to rising temperatures, beetles become capable of colonizing larger, healthier trees. These trees

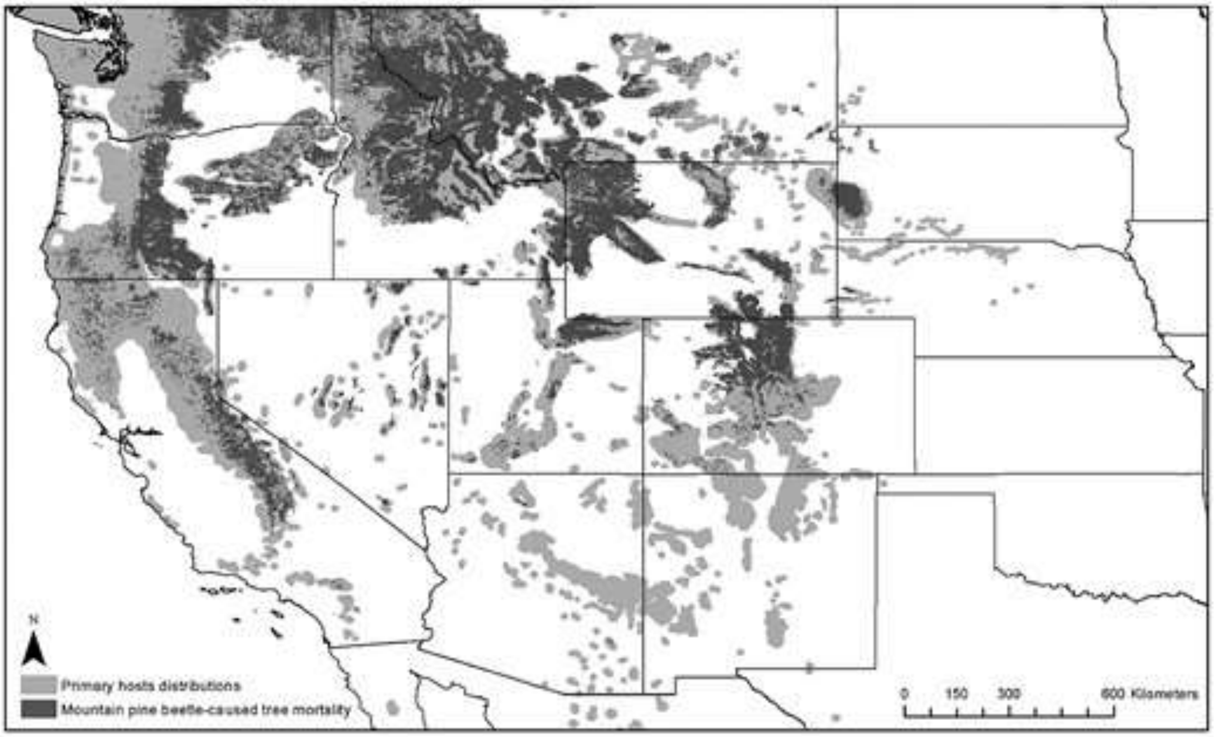


Figure 2. Areas (dark gray impacted by the mountain pine beetle in the western United States during 2001-2011. Data provided by the USDA Forest Service based on aerial survey. Light gray represents the distribution of MPB's primary hosts. Figure taken from Audley et al., 2020.

contain more phloem, allowing for greater reproductive success, creating a positive feedback loop that leads to epidemics as healthy tree defenses fail to defend against the large mass attacks. Once an epidemic has become widespread it often only declines due to lack of hosts capable of sustaining the large population size or due to extremely unfavorable weather (Bleiker et al., 2014).

MPB colonization includes the introduction of ophiostomatoid fungi that act as mutualistic partners in the colonization of the tree. MPB carries the fungal spores on their body surface and introduces it into the phloem of the tree (Audley et al., 2020). The fungal spores germinate and block water transport from soil to canopy in the tree, causing drought related tree mortality. It has been suggested that fungi—rather than phloem feeding by MPB—may be the primary cause of mortality for beetle-infested lodgepole pine (Hubbard et al., 2013).

Figure 3a. Pitch tubes, resin, and boring dust pictured on a lodgepole pine tree. Image from Sambaraju et al., 2021.

An MPB infestation on a tree can be identified by the presence of "pitch tubes": beetle entrance holes lined with tree resin and boring dust, pictured in Figure 3a. These infestations kill their host tree after approximately a year (Sambaraju et al., 2021). Epidemics become visible about one year after tree death, discerned as a landscape of reddened trees, pictured in Figure 3b. These trees

Figure 3b. Reddened trees due to MPB infestation. Photo taken from U.S. Forest Service. Credit to Whitney Cranshaw, Colorado State University, Bugwood.org

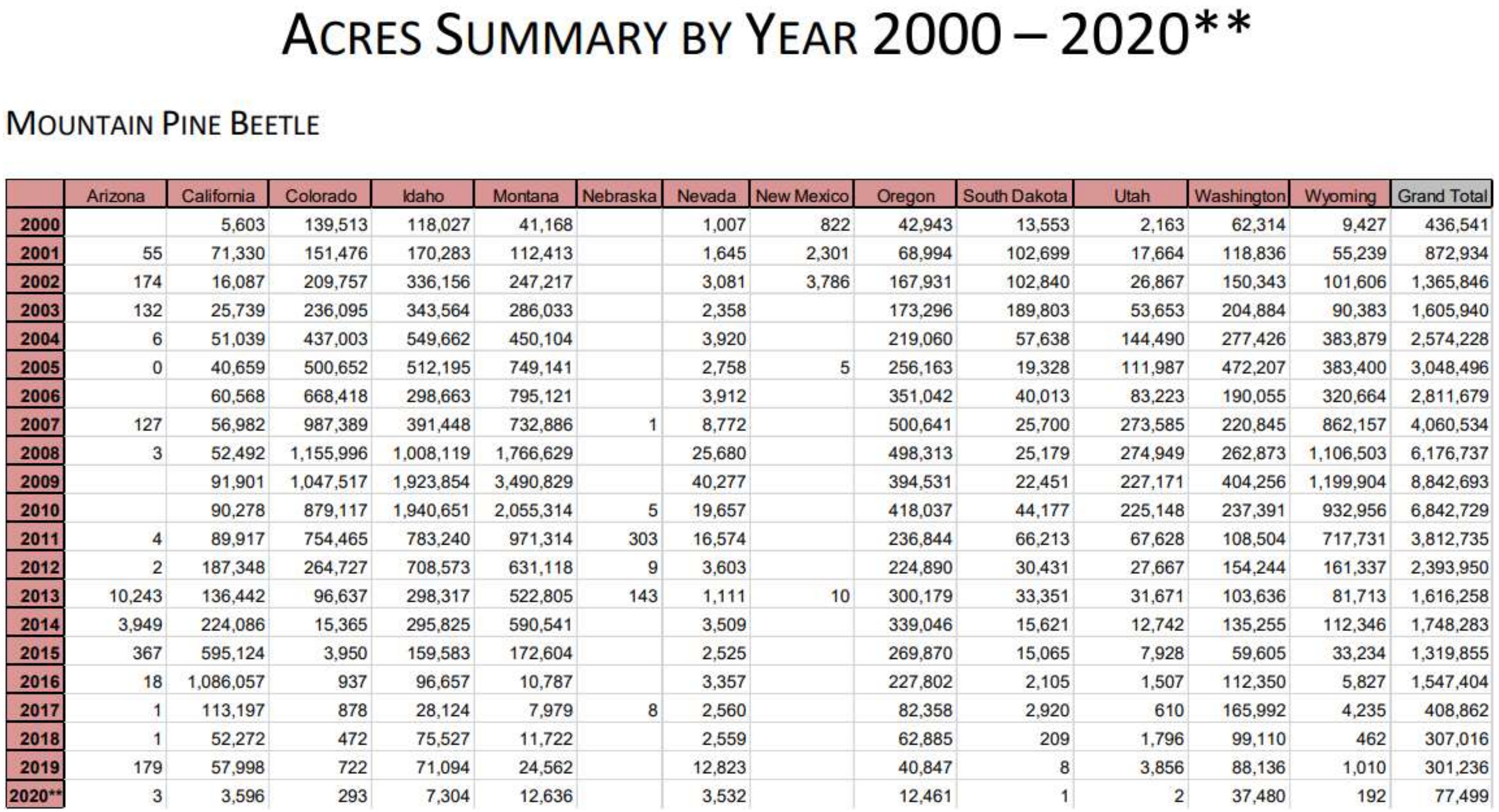
eventually turn gray. Using this visual identification, aerial overview surveys have allowed for major MPB infestations to be identified and mapped.

The first recorded outbreak of MPB occurred in 1895 in the Black Hills of South Dakota. The main concern at the time was the loss of lumber caused by the infestation, prompting the first attempt at direct control strategies. Direct control includes the use of fire, insecticides, semiochemicals, sanitation harvests, or a combination thereof in order to address current infestations for the short term (Fettig et al., 2014). Direct control has the detriment of merely treating the symptoms of the issue, rather than taking preventative measures. Furthermore, it is often expensive and thus dictated by economic viability rather than ecological need. The effectiveness of these treatments varies from case to case, depending on various factors such as proximity to untreated infestations (Fettig et al., 2014).

On the western coast, two main MPB outbreaks have occurred in the last 40 years, the first in 1980-1981 (Sambaraju et al., 2021). Since 2000, approximately 26.7M acres of forest in the western U.S. have experienced tree mortality from MPB (United States Department of Agriculture - Forest Service, 2021). The distribution by state is visible in Figure 4. In the Intermountain West (Idaho, Montana, Utah, and Wyoming) initial growth of populations began in 2004, peaked in 2007, and returned to endemic levels in 2011 (Audley et al., 2020). In Montana, the 2000s outbreak resulted in over 9 million acres of tree mortality, beginning in 1999 and lasting until 2015. The greatest peak was in 2008 and 2009. Because tree mortality is viewed a year after initial infestation, the actual increase in MPB numbers to epidemic proportions occurred in 2007. Increased activity of MPB correlates with abnormally dry conditions occurring in Montana around 2000 and intensifying into extreme drought levels in

2004-2006 (Lestina et al., 2019). This epidemic demonstrated a high correlation between drought—a common symptom of climate change—and MPB infestations.

Though monetary loss from logging was the primary concern for the initial direct responses to MPB infestations, the widespread ecological detriment of outbreaks became an essential factor to policy planning when dealing with MPB (Fettig et al., 2014). It was recognized that infested forests provide more loss than simply economic; aesthetic value is lost, wildfires become more severe, microbial decomposition of dead trees after outbreaks causes forests to transfer from carbon sinks to carbon sources (Sambaraju et al., 2021). The concept of indirect control developed in the wake of this understanding. In contrast to direct control, indirect control is preventative, and consists of actions such as thinning, prescribed burning, and/or alterations of age classes and species composition. These actions seek to block one of the two requirements for MPB outbreak: (1) several years of favorable weather and (2) abundance of susceptible host trees. The first is infeasible to block—therefore indirect actions such as clearing away trees through different methods has been an oft employed strategy (Fettig et al., 2014). However, the most feasible and long-term preventative action that can be taken to combat MPB is increasing the resiliency of forests.



ACRES SUMMARY BY YEAR 2000 – 2020**

MOUNTAIN PINE BEETLE

	Arizona	California	Colorado	Idaho	Montana	Nebraska	Nevada	New Mexico	Oregon	South Dakota	Utah	Washington	Wyoming	Grand Total
2000		5,603	139,513	118,027	41,168		1,007	822	42,943	13,553	2,163	62,314	9,427	436,541
2001	55	71,330	151,476	170,283	112,413		1,645	2,301	68,994	102,699	17,664	118,836	55,239	872,934
2002	174	16,087	209,757	336,156	247,217		3,081	3,786	167,931	102,840	26,867	150,343	101,606	1,365,846
2003	132	25,739	236,095	343,564	286,033		2,358		173,296	189,803	53,653	204,884	90,383	1,605,940
2004	6	51,039	437,003	549,662	450,104		3,920		219,060	57,638	144,490	277,426	383,879	2,574,228
2005	0	40,659	500,652	512,195	749,141		2,758	5	256,163	19,328	111,987	472,207	383,400	3,048,496
2006		60,568	668,418	298,663	795,121		3,912		351,042	40,013	83,223	190,055	320,664	2,811,679
2007	127	56,982	987,389	391,448	732,886	1	8,772		500,641	25,700	273,585	220,845	862,157	4,060,534
2008	3	52,492	1,155,996	1,008,119	1,766,629		25,680		498,313	25,179	274,949	262,873	1,106,503	6,176,737
2009		91,901	1,047,517	1,923,854	3,490,829		40,277		394,531	22,451	227,171	404,256	1,199,904	8,842,693
2010		90,278	879,117	1,940,651	2,055,314	5	19,657		418,037	44,177	225,148	237,391	932,956	6,842,729
2011	4	89,917	754,465	783,240	971,314	303	16,574		236,844	66,213	67,628	108,504	717,731	3,812,735
2012	2	187,348	264,727	708,573	631,118	9	3,603		224,890	30,431	27,667	154,244	161,337	2,393,950
2013	10,243	136,442	96,637	298,317	522,805	143	1,111	10	300,179	33,351	31,671	103,636	81,713	1,616,258
2014	3,949	224,086	15,365	295,825	590,541		3,509		339,046	15,621	12,742	135,255	112,346	1,748,283
2015	367	595,124	3,950	159,583	172,604		2,525		269,870	15,065	7,928	59,605	33,234	1,319,855
2016	18	1,086,057	937	96,657	10,787		3,357		227,802	2,105	1,507	112,350	5,827	1,547,404
2017	1	113,197	878	28,124	7,979	8	2,560		82,358	2,920	610	165,992	4,235	408,862
2018	1	52,272	472	75,527	11,722		2,559		62,885	209	1,796	99,110	462	307,016
2019	179	57,998	722	71,094	24,562		12,823		40,847	8	3,856	88,136	1,010	301,236
2020**	3	3,596	293	7,304	12,636		3,532		12,461	1	2	37,480	192	77,499

Figure 4. Acres of forest affected by MPB are divided by state and year. The Pacific Northwest (Washington, Oregon, and Idaho) have experienced high numbers of infection, but lower than that of the Northern US (Montana) or Colorado. The figure is taken from United States Department of Agriculture - Forest Service.

Fungus

Like with insect infections, the vulnerabilities experienced by trees due to climate change (such as increased exposure to drought and wildfires) decreases the resilience of vegetation to fungal diseases. The extent that fungal diseases are directly affected by climate change varies by species. Swiss needle cast (*Phaeocryptopus gaeumannii* (Rohde) Pilát) and white pine blister rust (*Cronartium ribicola* A. Dietr.) are fungal infections that are directly affected by climate change due to their reliance on climate-influenced environmental conditions such as temperature and precipitation. Other species such as laminated root rot and dwarf mistletoe are indirectly associated and not as affected by climate change (Halofsky et. al., 2022).

Case Study: White Pine Blister Rust

White pine blister rust (here-on called only "blister rust") is caused by the fungus *Cronartium ribicola*. Unlike the mountain pine beetle, which is a native insect that has grown into an infestation due to overpopulation and range-spread, blister rust is an exotic fungus that originated in Asia, became established in Europe in the 18th century, and was introduced into North America around 1900 (Maloy, 2001). It was introduced to the eastern U.S. as early as 1897, the western states around 1910, and spread inland to the Greater Yellowstone Ecosystem by the 1940s. The range of observations as of 2019 can be seen in Figure 5.

As its name suggests, white pine blister rust affects all North American white pine species. In the western U.S. blister rust affects whitebark pine, limber pine, and western white pine

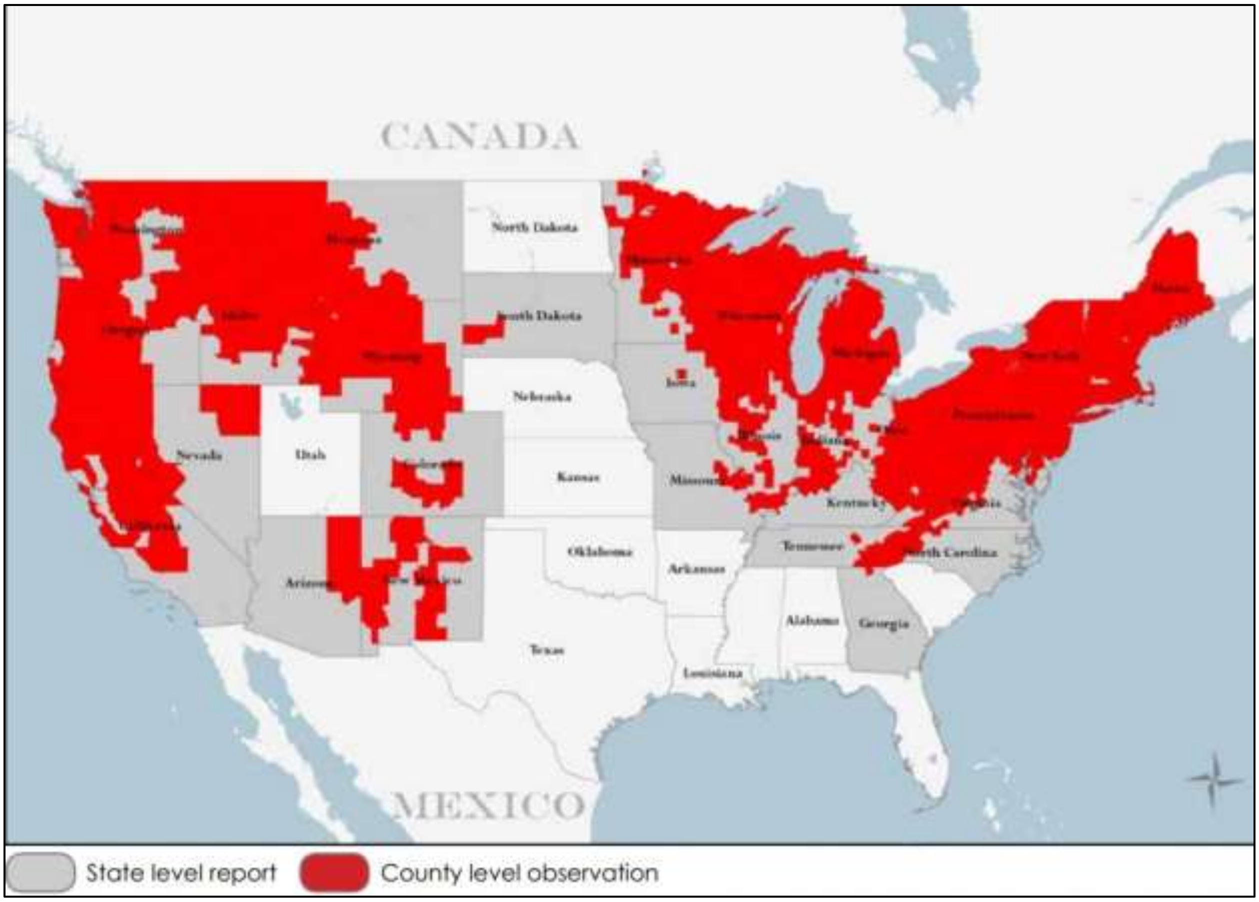


Figure 5. Reported observations of white pine blister rust in the United States. Created 27 March 2019 by USDA Forest Service.

(Thoma et al., 2019). Since the 1900s, the impact of blister rust on commercial North American white pines has been the focus of studies; however, by the mid-1980s conservation practices turned toward the management of ecosystems over species and focus expanded to encompass the effects of blister rust on whitebark pine (Schoettle, 2004). Concerns toward whitebark pine populations continue to rise due to its identity as a keystone and foundation species in many high elevation ecosystems throughout the western United States and Canada (Bockino and Tinker, 2012). None of the white pines in North America have significant genetic resistance to blister rust, leading to heavily infected areas succumbing to complete mortality.

The forest communities that replace these mortality zones are less resilient to epidemics by native pests and pathogens (Schoettle, 2004).

The life cycle of blister rust requires two obligate hosts. One is the white pine. The other is often the currant or gooseberry species (*Ribes* ssp.), though it can infect some species of lousewort (*Pedicularis* spp.) and Indian paintbrush (*Castilleja* spp.). Unlike the white pine, the first host for blister rust is a deciduous tree that is capable of shedding the blister rust infection each year, thus not falling prey to the mortality rate experienced by the white pine. In the spring, aeciospores are released from infected areas of the white pines. They use wind transport to eventually infect a *Ribes* species. In the late summer or early fall, basidiospores are

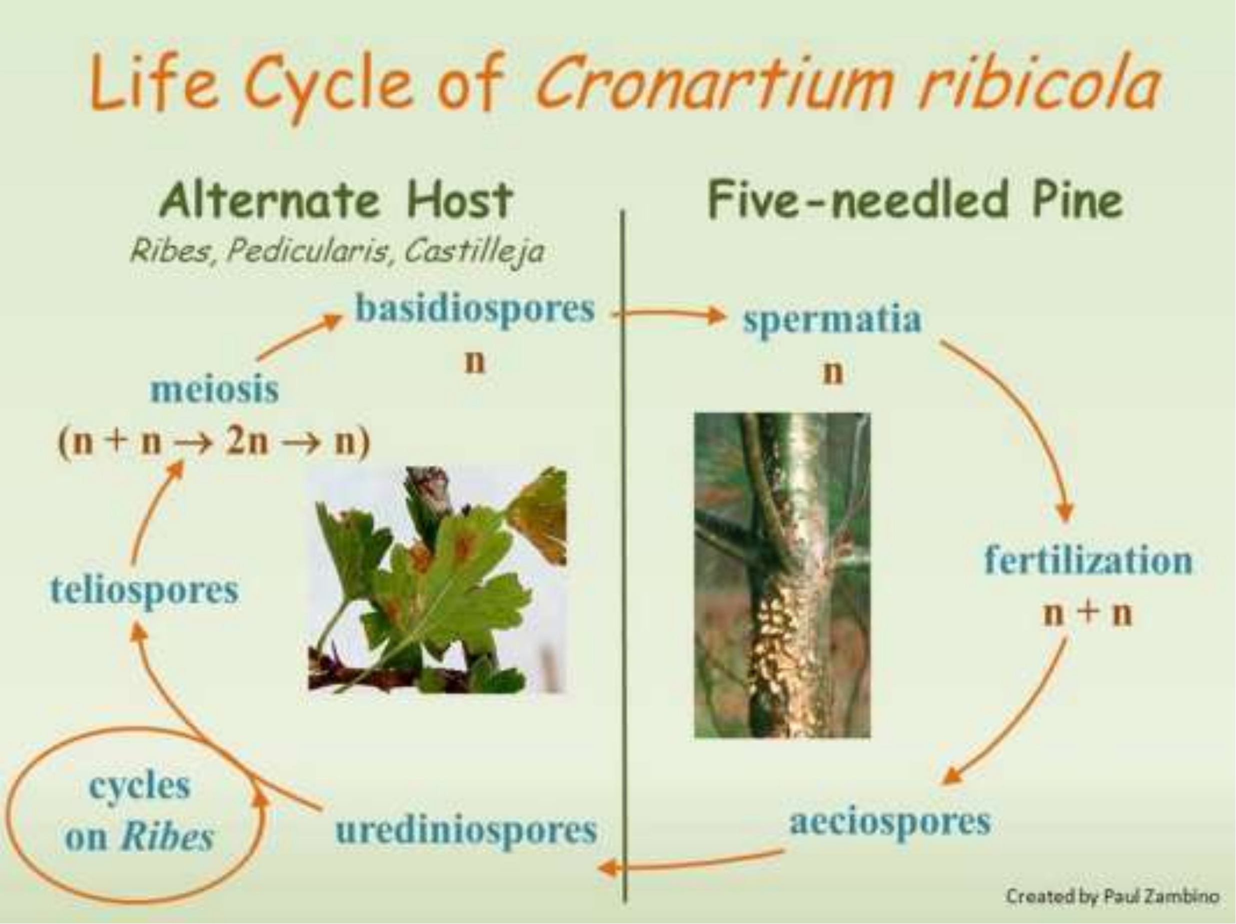


Figure 6. The life cycle of white pine blister rust. Graphic taken from Schwandt, 2013).

released. The spores enter white pines through stomatal openings in the pine needles and the fungus grows into the twig (Schoettle, 2004). This process is illustrated in a simplified schematic in Figure 6.

Infections favor one-year old foliage (50%) and are minimal on foliage older than three years. In the following spring, the rust causes small yellow needle spots to form, and as it advances into the branch (2-inches per year), it kills the tissue it encounters. Stem cankers form once the fungus reaches the main stem. These cankers eventually girdle the stem and cause top kill or mortality (pictured in Figure 7). In the late summer, water droplets appear on the canker containing spermatia that insects transmit to other cankers in fertilization. Yellow-orange blister-like aecia appear on the canker in the spring (pictured in Figure 8), containing the aeciospores that infect the alternate host (Schwandt, 2013). Infections can lead to tree mortality, but top kill can result in a dramatic reduction in reproduction as well due to the loss of cone-bearing branches (Bockino and Tinker, 2012).

← **Figure 7**. Top kill from blister rust. Image taken from Schwandt, 2013.

Figure 8. Titular blister-like aecia forming on canker. Image taken from Schwandt, 2013.

Whitebark pine—a species threatened by blister rust, MPB, and climate change—is vital for high elevation ecosystems by regulating soil development, facilitating plant succession, providing carbon storage, and capturing and retaining snow (Bockino and Tinker, 2012). This snow retention is beneficial for riparian habitats by increasing the quantity and duration of summer runoff, feeding streams further into the growing season. Their ability to tolerate stress and persist on climatically harsh sites with nutrient-deficient soils makes them well-suited to living in alpine environments other conifer species cannot. Whitebark pines—along with other species of white pine that live in subalpine environments—are capable of acting as nurse trees for less hardy species (Tomback and Achuff, 2010). Furthermore, whitebark pines act as an important food source for many animal species, their seeds acting as a nutritious food for granivorous birds, mammals, and insects. Whitebark pine seeds are an important food item for the grizzly bear (a threatened species) and black bear (Tomback and Achuff, 2010). Many sites within the northwestern U.S. and southwestern Canada are found with 50-100% of their whitebark populations infected with blister rust (Tomback and Achuff, 2010). The high infection rate is a threat not only to the whitebark species, but to every species that relies on the foundation species.

In the past, more than $100 million were spent on attempts (quarantines, antibiotics, wide-scale *Ribes* eradication) to control blister rust in the western United States (Schwandt et al., 2013). The management strategy with the most promise appears to be tree improvement in terms of genetic and biological resilience to infection and mortality. Genetically resilient white pines can be bred off site secure from blister rust infection. Once they have grown established and more resistant to blister rust, they can be replanted in the desired restored area.

Conclusion

Pests and pathogens are a hidden menace within forests of the Pacific Northwest. Without constant monitoring an infection can spread throughout an entire area without external signs until it is too late to save the host. It is also notoriously difficult to combat; the contributing factor to their spread being an increase in temperature that cannot be directly controlled and their extent being large and impossible to completely irradicate, even if small areas can be cleared. Furthermore, the agents behind epidemic diseases are not always invading species that can be completely irradicated. Native insects and fungi such as the mountain pine beetle contribute to the forest ecosystem when they are in endemic numbers and their complete removal may pave the way to worse infections. Adaptation and mitigation factors can include human interference practices such as controlled wildfires or clear-cutting, but the most important step that can be made is to prevent the lowering of tree resilience in other facets. Maintaining biodiversity within a forest prevents the accumulation of too many hosts within an area, strengthens healthy symbiotic interactions among vegetation and wildlife, and allows for trees' defensive measures to remain strong against epidemic infection. The importance of the forests within the Pacific Northwest cannot be understated. Protecting forests – and facilitating tree defenses – is vital to the continuing sustainability of the Pacific Northwest.

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