Rocky Mountains Cooperative Ecosystem Studies Unit (RM-CESU)
RM-CESU Cooperative Agreement Number: H1200090004 (IMR)

Project Final Report, September 14, 2011

TITLE OF PROJECT: Regeneration Status and Dynamics of Rare Ponderosa Pine (*Pinus Ponderosa*) Stands in Western Rocky Mountain National Park

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INTRODUCTION

The goal of this research was to provide a case study on the implications of recent climate change for adaptation planning to future climate change in Rocky Mountain National Park (ROMO). While there are several good sources that provide general recommendations for adaptation planning for climate change (e.g. Millar *et al.* 2007; Keane *et al.* 2009), actual case studies are rare and it is not clear that the recommendations are viable options following the impacts of recent climate change. Moreover, local case studies can provide valuable opportunities to develop science-based, site-specific management plans and frameworks. The current case study is focused on the ecological consequences of the recent mountain pine beetle (MPB; *Dendroctonus ponderosae*) outbreak, which has been at least partially attributed to climate change (Bentz *et al.* 2009), in a locally rare population of ponderosa pine (PIPO; *Pinus ponderosa*) in the North Inlet of western ROMO.

While the recent MPB outbreak has reshaped an extensive area of lodgepole pine (*Pinus contorta*) forest cover in western ROMO, in the context of ecosystem adaptation to climate change, the implications of the outbreak for the locally rare population of PIPO is potentially of greater significance. Based on projected climate change in the region (IPCC, 2007), species distribution models (SDMs) suggest that the Colorado River headwaters area (including western ROMO), where PIPO is currently rare, will become suitable for PIPO by 2030 (Rehfeldt *et al.* 2006). However, the general criticism of SDMs, that species will not be able to disperse to new areas of suitable climate because of physical barriers to dispersal and the relatively rapid rate of change, is likely applicable to PIPO migration in this situation. Specifically, even though PIPO is common on the east slope of the Northern Colorado Front Range (NCFR), the physical barrier created by the continental divide combined with the short seed-dispersal distances of PIPO (Barrett 1979) suggest that dispersal to the Colorado River headwaters will be difficult. In contrast to dispersal, the existing population of PIPO in the North Inlet could serve as a local source population for range expansion as climate warms. Consequently, the North Inlet PIPO population could play a critical role for ecosystem adaptation to climate change.

While there are no clear management blueprints to achieve ecosystem adaptation to climate change, the best management practices are likely conservative hedge-betting plans, including the restoration of systems to within historic range of variability (HRV) conditions (Millar *et al.* 2007; Keane *et al.* 2009). HRV conditions might not be indicative of forest conditions under future climate; however, a restored landscape will maximize the potential of forests to adapt to changes in climate and disturbance regimes (Keane *et al.* 2009). Moreover, restoration will avert potential compounded impacts from the interaction of climate change influences on disturbances with the ecological legacies of ecosystem management practices such as fire exclusion. One potential issue with adopting the "restore to HRV conditions" management strategy is that recent climate change influences, including influences on recent patterns of wildfires (Westerling *et al.* 2006*)* and insect outbreaks (Bentz *et al.* 2009), may have eliminated the option to restore systems. Specifically, forest ecosystem restoration plans often rely on an overabundance of trees in different age and size classes that can be manipulated to achieve desired conditions. It is possible that recent events (e.g. wildfires and insect outbreaks) may have killed trees critical for restoration. For the North Inlet PIPO population it is not clear that the stand was outside of HRV conditions prior to the recent MPB outbreak, if pre-outbreak conditions influenced the outbreak, or if the recent outbreak has altered management options, including the option to restore the stand if needed.

The critical ecosystem management questions for the North Inlet PIPO population addressed in this study include:

Q1: What are the drivers of stand dynamics and did 20th century fire exclusion practices create conditions outside of the HRV?

Q2: If the site was outside of HRV conditions did this amplify the impacts of the recent MPB outbreak, and would pre-outbreak restoration to HRV conditions have mitigated these impacts?

Q3: Did the outbreak alter ecosystem management options, and given the current status of PIPO, what are current management options?

STUDY AREA

The PIPO population in the North Inlet is spread over approximately 50 ha on a rocky south-facing slope of the southwest edge of the valley (Fig. 1). Site elevation ranges from 2600-2800m and slope steepness ranges from 5 to 30. ° In addition to PIPO, Douglas-fir (*Pseudotsuga menziesii*), lodgepole pine (*Pinus contorta*), limber pine (*Pinus flexilus*), aspen (*Populus tremuloides*), Engelmann spruce (Picea engelmannii) and subalpine fir (Abies lasiocarpa) are present at the site. The last major fire event in the valley was in 1851 (Sibold *et al.* 2006) and it is possible that regional MPB outbreaks in the 1930s and 1970s occurred within the site but there is no obvious evidence of these outbreaks (i.e. extensive dead boles with MPB galleries in advanced stages of decay). Many Douglas-fir at the site have a "skeleton tree" appearance (many dead branches) indicating that they have been defoliated in the past by western spruce budworm (WSB; *Choristoneura occidentalis*). WSB attacks Douglas-fir and kills many seedlings and saplings and some canopy trees in outbreaks lasting from one to several decades (Swetnam and Lynch, 1989; Hadley and Veblen 1993). The most recent outbreak in the region occurred in the 1970s and 80s and tree-ring based reconstructions of outbreaks in the upper montane zone of the east slope of the NCFR demonstrate that outbreaks occurred every 30-50 years over the past few centuries (Swetnam and Lynch 1989).

METHODS

I documented the current status of PIPO in the North Inlet and used dendroecological methods to reconstruct PIPO establishment dates to compare with the history of fires, MPB and WSB outbreaks at the site. Because of the lack of visual evidence of past MPB outbreaks, I reconstructed WSB outbreaks in order to test if canopy openings associated with insect outbreaks provide an opportunity for PIPO establishment. While WSB outbreaks are a different disturbance than MPB outbreaks, most notably in that they have different host species, outbreaks of both insects are similar in that they result in prolonged periods of canopy openings but do not create bare mineral soil conditions associated with fire. Thus, WSB outbreaks are not an ideal proxy for MPB outbreaks but they can provide some indication of relationships between the two processes (insect outbreaks and PIPO establishment) and the potential for establishment of new PIPO in relation to the recent MPB outbreak.

Field methods

I systematically searched for and sampled all PIPO (live and dead) within the North Inlet study area. For all PIPO I recorded the diameter at breast height (dbh), status (live or dead), and location using a Global Positioning System (GPS). To reconstruct establishment dates I removed a tree core from the base of all live and dead PIPO trees and saplings that were large enough to sample non-destructively, angling the borer slightly downward to attempt to intercept the tree pith at the root/shoot collar. I cored trees until I obtained a sample that appeared to be within *ca.* five years of the pith. For trees that were too rotten to core at the base I extracted a core from higher up on the tree and noted the coring height. For seedlings that were too small to core nondestructively, I counted branch whorls to estimate age.

To reconstruct disturbance history I collected wedge samples from fire-scarred trees, and cores from Douglas-fir. I cut a partial cross-section from the base of scarred trees (Arno and Sneck 1977) with the goal of sampling trees spread throughout the site to verify fire extents. To reconstruct past WSB outbreaks I collected two increment core samples at breast height from Douglas-fir trees spread throughout the site. I targeted larger-diameter trees that had a skeleton-tree appearance, which indicates previous defoliation. The locations of all wedge and core samples were recorded using a GPS. I did not collect additional samples to reconstruct MPB outbreaks because the core samples collected from PIPO can be used to reconstruct these events. Specifically, coincident death dates of PIPO would indicate past MPB outbreaks.

Lab methods

I processed all samples according to established dendrochronological methods (Stokes and Smiley 1968). Annual rings on all core and wedge samples were counted and visually cross-dated with a master chronology (Monarch Lake; Graybill). Marker rings were identified from previous chronologies in the area (Sibold *et al.* 2007). Difficult to date samples were measured and quantitatively cross-dated using the program COFECHA (Holmes 1983). To refine establishment dates for ponderosa pine cores that did not intercept the tree pith, I used Duncan's (1989) method to estimate for up to 10 missing rings from the tree center.

Analytical methods

Pulses of ponderosa pine regeneration were graphed and visually compared against fires, and insect outbreaks (e.g. Villalba and Veblen 1997). Fire dates were identified from fire scars and defined as scarring at least two trees in the stand. To identify past MPB outbreaks I looked for coincident pulses of ponderosa pine mortality. To reconstruct WSB outbreaks I used the program OUTBREAK (Holmes and Swetnam 1996), which compares ring widths for Douglas-fir against ring widths for a non-host species (Swetnam 1986). For a non-host chronology I created a robust multi-site chronology from upper montane PIPO sites in the NCFR. Because relationships between climate conditions and ring widths for Douglas-fir and PIPO are similar, prolonged (>7 years) growth suppressions in Douglas-fir that do not correspond to suppressions in PIPO can be interpreted as resulting from WSB defoliation of Douglas-fir.

RESULTS

Disturbance history

I collected a total of 19 wedge samples from dead ponderosa pine and lodgepole pine spread throughout the site to reconstruct fire history. Individual trees recorded between one and three fire events. Only three fire years, 1666, 1732, and 1851, were recorded at the site and based on the spatial distribution of trees recording each fire, all three fires appear to have spread throughout the site. Based on PIPO age structure (presented below) it is likely that a mid-1500s fire burned through the site as well, but I did not find fire scar evidence of this older fire. Based on the fire scar record, the mean fire return interval (MFRI) was 92 years with a range from 66 to 119 years. When a 1550 fire date is included the MFRI increases slightly to 100 years. Approximately 150 years elapsed between the last fire event (1851) and the start of the MPB outbreak.

I did not detect any evidence of past MPB outbreaks at the site (represented by coincident PIPO mortality) outside of the current outbreak; however, WSB outbreaks were common over the last few centuries. I reconstructed a total of nine WSB outbreaks between

1600 and present (Fig. 2). Individual outbreaks ranged in severity, with *ca.* 25% to 80% of sampled trees recording individual outbreaks, and duration from *ca.* 10-30 years (Fig. 2).

Ponderosa pine demographics

I recorded a total of 467 PIPO individuals spread throughout the North Inlet study site (Fig. 1). PIPO establishment dates in the North Inlet were temporally clustered in pulses lasting from 30 to 60 years (Fig. 3). Pulses of establishment occurred in 1560-1610, 1680-1710, 1740-1770, 1850-1910, and 2000-present. While the 1560-1610 pulse predates the disturbance history record for the site, the 1680-1710, 1740-1770 and 1850-1910 pulses follow the three widespread fire events in 1666, 1732, and 1851 respectively. Based on fire-PIPO establishment relationships, it is likely that the 1550-1610 PIPO establishment pulse is in relation to a *ca.* 1550 fire. Establishment outside of post-fire periods was rare and there was only slight evidence that insect outbreaks (WSB) created establishment opportunities. PIPO establishment corresponding to insect outbreaks occurred in the early 1900s, a period that also corresponds to the post-1851 fire period, and in the last decade, which corresponds to the recent MPB outbreak.

Current status of ponderosa pine

The majority of PIPO in the North Inlet were killed in the recent MPB outbreak and surviving PIPO are predominantly smaller-diameter trees, saplings or seedlings. Of the 467 PIPO recorded at the site 378 (81%) are dead, with the majority (98%) dying within the last decade. PIPO death dates from the recent MPB outbreak ranged from 2000 to 2008 with a peak in mortality from 2004-2006 (Fig. 4). MPB preferentially killed larger diameter trees and few individuals >30cm dbh survived the outbreak (Fig. 5). The majority of surviving trees were <20cm dbh. The general positive relationship between PIPO age and diameter means that older PIPO tended to have larger diameters and died in the outbreak (Fig. 5).

DISCUSSION

Q1: What are the drivers of stand dynamics and did 20th century fire exclusion practices create conditions outside of the HRV?

My results indicate that ponderosa pine dynamics in the North Inlet are dominated by the influences of wildfires. Over the last several centuries, fires were slightly more frequent (MFRI: *ca.* 100 years) in the low-elevation south-facing site than in the surrounding subalpine forests, which had a MFRI of 156 years (Sibold *et al.* 2006). The period between the last fire event at the site (1851) and the start of the MPB outbreak (ca. 2000) was *ca.* 150 years, approximately 50 years longer than the HRV MFRI and 30 years longer than the longest previous interval at the site. While this time interval without fire is not a dramatic departure from HRV return intervals, it can be argued that fire exclusion did result in an unnatural fire interval at the North Inlet site.

PIPO age structure in the North Inlet clearly shows that prior to the recent MPB outbreak, fire was the primary influence on regeneration dynamics. Many PIPO survived fires, suggesting that fires at the site were mixed severity, and following fires PIPO established in pulses that lasted from two to several decades (Fig. 3). Establishment pulses were likely in response to the opening of the forest canopy and bare mineral soil, which is preferred by PIPO. Establishment outside of post-fire periods was limited and the general lack of regeneration coincident with WSB outbreaks (Fig. 2 and 3) implies that the current MPB outbreak might not create an opportunity for new establishment. While pre-20th century WSB outbreak-PIPO establishment relationships may be obscured by the influence of wildfires (e.g. 1830-40s outbreak and 1851 fire), the lack of a pulse of PIPO regeneration during the 1960s-1980s high-severity WSB outbreak suggests that insect outbreaks do not create opportunities for establishment. Nevertheless, there are a significant number (45) of seedlings presently at the site. Seedling ages indicate that many seedlings likely established in response to canopy openings from MPB-caused mortality; however, it is not known if these seedlings will successfully recruit to the canopy.

Q2: If the site was outside of HRV conditions did this amplify the impacts of the recent MPB outbreak, and would pre-outbreak restoration to HRV conditions have mitigated these impacts?

The ecological legacies of fire exclusion practices likely amplified PIPO mortality from the recent MPB outbreak in the North Inlet. At the time of the outbreak, PIPO age and size structures reflected cohorts that established following fires in the mid 1500s, 1666, 1732, and 1851 (Fig. 3). Consequently, the population was dominated by relatively older (>100 years) larger-diameter trees (> 25 cm; Fig. 3 and 5) that would have been able to withstand wildfire but were highly susceptible to MPB attack (Schmid and Mata 1996). Thus, the stand characteristics resulting from the abnormally long time since fire exacerbated the consequences of the MPB outbreak.

The elevated mortality resulting from the interaction of the climate change driven MPB outbreak with the ecological legacies of fire exclusion provides valuable lessons for ecosystem management planning for forest ecosystem adaptation to climate change. In retrospect, the general hedge-betting strategy of restoring stands to within HRV conditions (Millar *et al.* 2007; Keane *et al.* 2009) very likely would have mitigated the impacts of the outbreak. A mixed-severity wildfire, natural or prescribed, in the decades prior to the outbreak would have removed some old, large-diameter, MPB-susceptible trees and initiated a new cohort of ponderosa pine that would have been less susceptible to MPB attack.

Q3: Did the outbreak alter ecosystem management options, and given the current status of PIPO, what are current management options?

The recent MPB outbreak in the North Inlet site dramatically changed ecosystem management options. The current age and size structure of the stand is dominated by small-diameter trees, seedlings and saplings (Fig. 3 and 5) that are not well suited to survive fire. As a result, the role of fire has changed from a restoration tool to an existential threat for the North Inlet PIPO population. In addition to the likelihood that fire would result in high mortality, it is not certain that sufficient seed trees required for post-fire regeneration would survive to repopulate the site.

Given the current status of PIPO in the North Inlet and the likelihood that climate conditions in the upper Colorado River region will become more suitable for the species (Rehfeldt *et al.* 2006), several possible management options exist. Ecosystem management options ranging from hands-off to intensive, proactive responses include: 1) monitoring PIPO status at the site, 2) a fire exclusion policy to protect existing individuals, 3) collection and storage of seeds from the site to guard against the loss of potentially unique genetics in case of a catastrophic event, 4) small-scale strategic prescribed burns in areas without surviving ponderosa pine with the goals of reducing fuels and the likelihood for catastrophic fire, and creating opportunities for new PIPO establishment, and 5) assisted migration of PIPO to similar low-elevation south-facing sites in western ROMO to establish satellite PIPO populations. The assisted migration option would not only facilitate PIPO range expansion if climate change progresses as projected, but would also protect against local extinction from a catastrophic event. The establishment of satellite populations would be resource intensive and would require the collection of seeds at the North Inlet site, site preparation with prescribed fire to create areas of bare mineral soil, and removing competing lodgepole pine seedlings.

CONCLUSION

The recent climate change driven MPB outbreak in the North Inlet PIPO population clearly illustrates the challenges that recent climate change is creating for ecosystem management planning for adaptation to climate change. The changing role of fire at the North Inlet site, from a restoration tool to a threat, stresses the importance of adaptive management in the face of climate change. Moreover, this case study demonstrates that while HRV concepts and information are still useful for ecosystem management planning, recent and future climate change dictate that HRV information is coupled with current information on the status of species, stands and landscapes.

Even though the MFRI in the North Inlet was two to three times longer than those reported for similar sites in the upper montane zone on the east slope of the NCFR (Veblen *et al.* 2000), similarities in fire-PIPO establishment relationships (Ehle and Baker 2003; Sherriff and Veblen 2006) suggest that some of the conclusions in this study are applicable to PIPO management in eastern ROMO. While it is too late to change pre-MPB outbreak conditions in east ROMO, based on these results it is clear that following the outbreak, ROMO will need to evaluate the status of populations and reassess the potential for fire to restore the ecosystem to desired conditions. In contrast to western ROMO, the extensive areas of PIPO in eastern ROMO might provide opportunities to experiment with post-MPB restoration prescriptions without the risk associated with the small population in the North Inlet. Results from treatments in eastern ROMO would undoubtedly provide valuable information for management planning in the North Inlet.

One important topic that was not studied or discussed in this project is the need to view ROMO management decisions for PIPO in the North Inlet in the context of the broader upper Colorado River landscape. In particular, two aspects of the surrounding forested landscape are directly applicable to PIPO management in western ROMO. The first is that one other known PIPO population exists in the area and other similar sites might exist. The other documented population is south of ROMO close to Monarch Lake in the Indian Peaks Wilderness. The site and population in the Indian Peaks Wilderness appear to be similar to the North Inlet and collaborative cross-boundary ecosystem management could enhance PIPO conservation and potential adaptation to climate change. The second aspect is that based on conversations with private landowners in the area, people are planting PIPO in areas adjacent to the western boundary of ROMO to reforest areas in the wake of the recent MPB outbreak. The motive for planting PIPO is based on the perception that the species is better suited for recent climate conditions in the area. Thus, while ecosystem managers and scientists engage in serious debate about assisted migration, private landowners are inadvertently assisting migration of PIPO in the upper Colorado River headwaters, likely without understanding the potential ecological consequences of their actions for species migration or the preservation of local genetics.

References:

Arno, S.F. & Sneck, K.M. 1977. A method for determining fire history in coniferous forests of the mountain west. US Department of Agriculture Forest Service General Technical Report INT-42, Ogden, UT.

Barrett, James W. 1979. Silviculture of ponderosa pine in the Pacific Northwest: the state of our knowledge. USDA Forest Service, General Technical Report PNW-97. Pacific Northwest Forest and Range Experiment Station, Portland, OR. 106 p.

Bentz, B.J., C.D. Allen, M. Ayres, E. Berg, A. Carroll, M. Hansen, J. Hicke, L. Joyce, J. Logan, W. MacFarlane, J. MacMahon, S. Munson, J. Negron, T. Paine, J. Powell, K. Raffa, J. Régnière, M. Reid, W. Romme, S. Seybold, D. Six, D. Tomback, J. Vandygriff, T. Veblen, M. White, J. Witcosky, and D. Wood. 2009. Bark Beetle Outbreaks in Western North America: Causes and Consequences. University of Utah Press.

Duncan, R.P. 1989. An evaluation of errors in tree age esti- mates based on increment cores in Kahikatea. New Zealand Natural Sciences, 16, 31.

Ehle D.S., and W.L. Baker. 2003. Disturbance and stand dynamics in ponderosa pine forests in Rocky Mountain National Park, USA. *Ecological Monographs* 73:543–66.

Graybill, D.A. International tree-ring data bank. IGBP PAGES/ World Data Center for Paleoclimatology, NOAA/NGDC Paleoclimatology Program, Boulder, CO.

Hadley, K. and Veblen, T. 1993. Stand response to western spruce budworm and Douglas-fir bark beetle outbreaks, Colorado Front Range. Can. J. For. Res. 23(3): 479–491.

Holmes, R.L. 1983. Computer-assisted quality control in tree- ring dating and measurement. Tree-Ring Bulletin, 43, 69.

Holmes, R.L., and Swetnam, T. 1996. Detecting outbreaks of spruce budworm and tussock moth in annual tree-ring growth, and distinguishing between the insect species. Laboratory of Tree-Ring Research, University of Arizona, Tucson, Ariz.

Millar, C.I., N.L. Stephenson, and S.L. Stephens. 2007. Climate change and forests of the future: Managing in the face of uncertainty. Ecological Applications, 17:2145-2151.

Rehfeldt, G.E., N.L. Crookston, M.V. Warwell, and J.S. Evans. 2006. Empirical analyses of plant-climate relationships for the western United States. International Journal of Plant Science, 167:1123-1150.

Sherriff R.L., and T.T. Veblen. 2006. Ecological effects of changes in fire regimes in Pinus ponderosa ecosystems in the Colorado Front Range. *Journal of Vegetation Science* 17:705–718.

Sibold, J.S., T.T. Veblen and M.E. Gonzalez. 2006. Spatial and temporal variation in historic fire regimes in subalpine forests across the Colorado Front Range in Rocky Mountain National Park. *Journal of Biogeography* 32, 631-647.

Schmid, J. M., and S. A. Mata. 1996. Natural variability of specific forest insect populations and their associated effects in Colorado. General Technical Report RM-275. USDA Forest Service, Fort Collins, Colorado, USA.

Stokes, M.A. & Smiley, T.L. 1968. An introduction to tree-ring dating. University of Chicago Press, Chicago, IL.

Swetnam T.W. 1986. Radial growth losses in Douglas-fir and white fir caused by western spruce budworm in northern New Mexico: 1700-1983. USDA Forest Service SW Region Forest Pest Management Report 86-2.

Swetnam, T. W., and A. M. Lynch, 1993. Multi-century, regional-scale patterns of western spruce budworm history. Ecological Monographs 63(4):399-424).

Veblen T.T., T. Kitzberger, and J. Donnegan. 2000. Climatic and human influences on fire regimes in ponderosa pine forests in the Colorado Front Range. *Ecological Applications* 10:1178–95.

Villalba, R., and T.T. Veblen. 1997. Regional patterns of tree population age structures in northern Patagonia: climate and disturbance influences. Journal of Ecology 85:113-124.

Westerling, A.L., H.G. Hidalgo, D.R. Cayan, and T.W. Swetnam. 2006.Warming and Earlier Spring Increase Western U.S. Forest Wildfire Activity. Science 313: 940-943.

Figure 1. The spatial distribution of live and dead ponderosa pine in the North Inlet of Rocky Mountain National Park, Colorado.

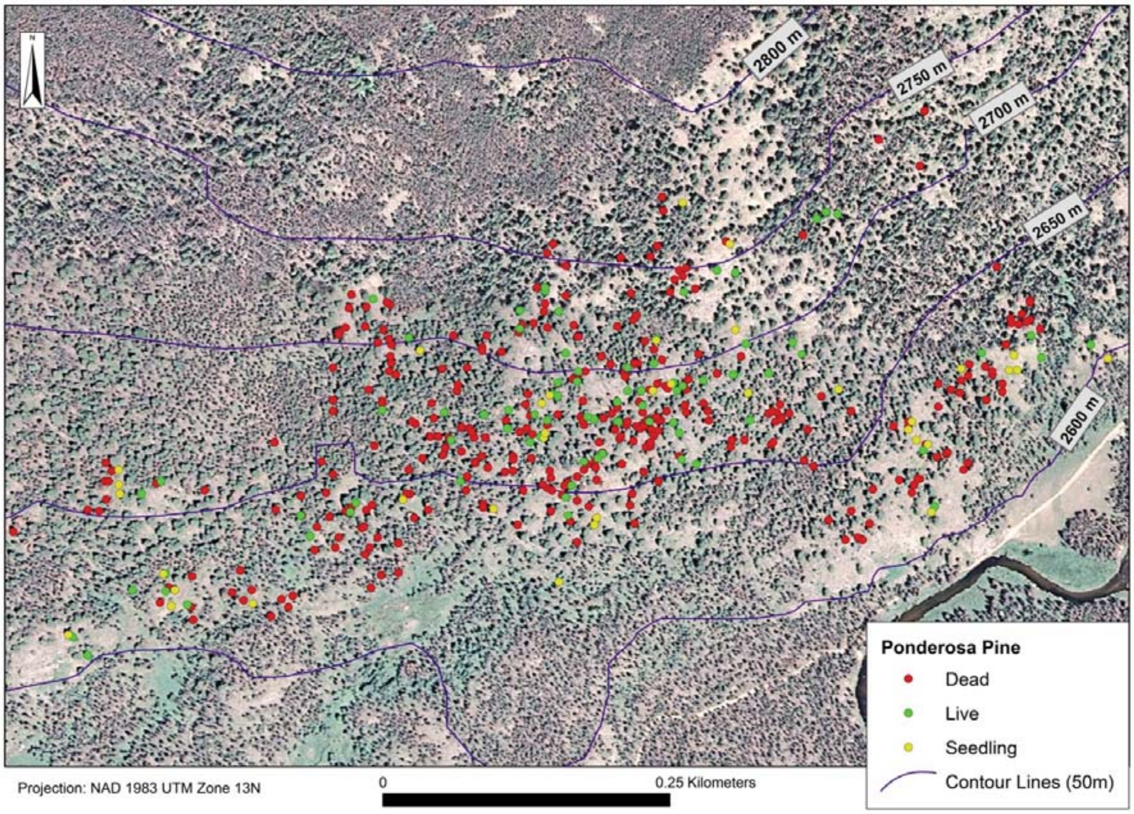

Fig. 2 Western spruce budworm outbreaks in the North Inlet as recorded by growth suppressions on Douglas-fir.

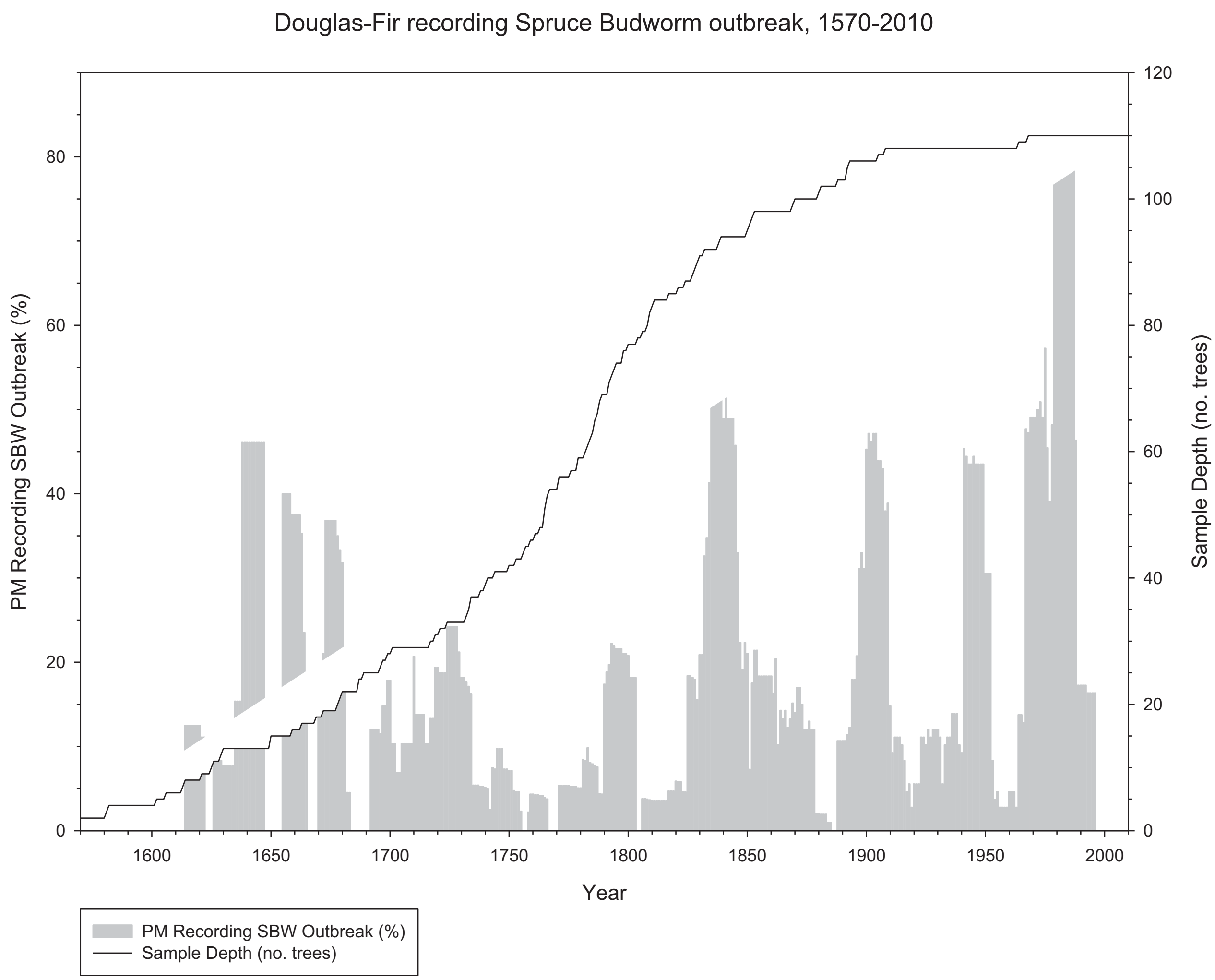

Figure 3. Establishment dates of ponderosa pine in relation to fires.