Vegetation Response to Prescribed Fire in Douglas-Fir Forests, Olympic National Park

Authors: Fonda, Richard W., and Binney, Elizabeth P.

Source: Northwest Science, 85(1) : 30-40

Published By: Northwest Scientific Association

URL: https://doi.org/10.3955/046.085.0103

Richard W. Fonda[1] and **Elizabeth P. Binney**[2], Biology Department, Western Washington University, Bellingham, Washington 98225

Vegetation Response to Prescribed Fire in Douglas-fir Forests, Olympic National Park

Abstract

Fire exclusion for more than a century in Olympic National Park changed old-growth Douglas-fir forest stand structure and species composition from an open structure of fire-adapted species to a more crowded, complex structure formed by fire-avoiding species. A previous study identified an historical mean fire return interval of 21 yr for these forests in the eastern Olympics prior to fire exclusion, with frequent, small surface fires maintaining an open forest. We tested whether Douglas-fir/salal forests would support low intensity prescribed fires, and monitored community responses for 3 yr post-fire in a randomized complete block ANOVA. Compared to pre-fire values, total fuels, 1000-hr fuels, 1-hr fuels, duff depth, total tree density, tree species density, sapling density, understory cover, and understory frequency of five prominent species were significantly lower one month post-fire. Differences in tree basal area, Douglas-fir sapling density, and western redcedar tree density were not significant after the fire. Lower sapling density was an important result. Salal, a known resprouter, was the only understory species to return to at least 50% of pre-fire cover within 3 yr. This first use of prescribed fire in Olympic National Park demonstrated that Douglas-fir/salal forests would support low intensity surface fire. Although community structure changed significantly immediately after fire, the tree canopy was little affected, and the understory will eventually recover to pre-fire values. The data from this study will contribute to a fire management plan that will incorporate prescribed fire with fire suppression, non-suppressed fires, and other active management to maintain forest health in the eastern Olympic Mountains.

Introduction

For most of the history of the Olympic Peninsula, Washington, fire management was based on fire exclusion. Fire ecology research in the past 40 yr, however, has identified two undesirable results of fire exclusion in communities adapted to periodic fires. The most well-known and visible result is a change in stand structure and species composition, from an open structure formed by fire-adapted species to a more crowded, complex structure formed by fire-avoiding species. A subtle, but perhaps more significant result relative to fire hazard, is an increase in woody fuel loads. Although it is axiomatic that fuels increase in the absence of fire, and decrease with fire, the fuel reduction is fleeting. Charred branches, burned saplings, even canopy trees killed as a result of the fire fall to the ground within a few years, effectively returning the fuel loads to pre-fire conditions (Schwilk et al. 2009). Much research into fuel reduction has demonstrated the immediate post-fire fuel decrease and the longer term post-fire fuel increase (Stephens and Moghaddas 2005, Knapp et al. 2005, Agee and Lolley 2006, Glitzenstein et al. 2006, Keifer et al. 2006, Wayman and North 2007, Youngblood et al. 2008, Stephens et al. 2009, Vaillant et al. 2009).

Recently, middle layer and understory composition relative to restoration treatments have received much attention in the literature. The favored approach to restoration now combines pre-fire thinning of saplings with prescribed fire (Agee and Lehmkuhl 2009, Schwilk et al. 2009, and citations therein). All experimental research investigating this dual approach, however, includes a burn-only treatment, which allows researchers to assess the effects of fire on overstory and understory species.

In general, forest communities in which fire was historically present were little affected by periodic or prescribed fire. The dominant species in the community are endurers (Agee 1993) or maintainers (Tveten and Fonda 1999), well adapted to the fire environment. In ponderosa pine-Douglas-fir (*Pinus ponderosa-Pseudotsuga menziesii*) forests in western North America, most data demonstrate the stability of the community. Overstory trees usually are above the reach of the fire, and resist direct flames at their bases, but saplings in the middle layer are fire sensitive.

---

[1] Author to whom correspondence should be addressed. Email: richfonda@comcast.net

[2] Current address: 1833 Summit Street, Bellingham, Washington 98229

Understory species are top-killed, but shrubs are resprouters, whereas forbs and graminoids are either resprouters or vigorous seeders. The best measure of species stability is frequency (Tveten and Fonda 1999). Cover invariably declines after low intensity fires, particularly for shrubs, which then resprout and re-establish pre-fire cover in 3-9 yr (Armour et al. 1984; Ruha et al. 1996; Busse et al. 2000; Metlen et al. 2004; Keeling et al. 2006; Metlen and Fiedler 2006; Wayman and North 2007; Dodson et al. 2007, 2008; Harrod et al. 2008; Nelson et al. 2008; Schwilk at al. 2009). Even forest understories subjected to high intensity fires demonstrate few deleterious effects (Huisinga et al. 2005).

Agee and Skinner (2005) elucidated four principles of fire resistance in dry forests: 1) reduce surface fuels; 2) increase height to live crown; 3) decrease crown density; and 4) retain big trees of resistant species. Although these principles were applied to fuel reductions, all four functioned in our research in Olympic dry forests because they describe the work accomplished by fire in the forest community, whether natural or prescribed in origin. When fire is part of the landscape, so that fuels, height to live crown, and crown density are reduced, torching and crown fire are virtually unknown. Conversely, when fire is absent from the landscape, all effects are reversed, so that stand-replacing crown fires are likely. Even if a periodic fire tends to burn more intensely than normal, the presence of widely-spaced, fire-resistant tree species in the canopy is further protection against flames reaching the canopy. Every prescribed fire attempts to bring these four principles into play to maintain the fire resistant structure that insures forest longevity. Forests in which fire-resistant conditions function are considered healthy (Quigley 1992, Mutch et al. 1993, Agee and Lehmkuhl 2009, Washington Department of Natural Resources 2011). But, a sequence of several prescribed fires invariably is required to bring these four principles into balance in the forest community.

Shifts in community composition and fuel loads in dry Douglas-fir forests in the eastern Olympic Mountains were identified by Wetzel and Fonda (2000), but that research stopped short of actually setting fires. These are the salient points from Wetzel and Fonda (2000). Based on germination dates and fire release markers in tree rings, dozens of lightning-ignited fires have burned in Douglas-fir forests in the 2500 ha Morse Creek drainage since 1400. Previously, it was accepted that only large, high intensity, high severity fires had burned every 200-300 yr in these forests (Fonda and Bliss 1969). The mean fire return interval (FRI) for the entire drainage was 3 yr; mean FRI for the most useful unit of study, 200 ha lateral tributaries, was 21 yr. This interval is shorter than the 99 yr mean FRI in an 1873 ha tract in the Elwha Valley, in the central Olympics where rainfall is greater than the Deer Park area (Wendel and Zabowski 2010). Nearly all of these fires were small, low intensity, low severity surface fires, especially in the old-growth forests. Only four high intensity, high severity crown fires appeared in the data. As a consequence of the complex fire history, this drainage supports three stand types, based on stand age. DF1 stands are 75-125 yr old; DF2 stands are 150-300 yr old; and DF3 stands are >300 yr old. All are dominated by Douglas-fir, a fire resistor (Agee 1993, Fonda et al. 1998). DF3 stands are the most common community type in the montane zone of the eastern Olympics (Fonda and Bliss 1969), and they are the focus of this research.

DF3 stands are characterized by a matrix of ~300 yr old trees in the canopy, with some trees well over 300 yr old (Figure 1A). The 2-12 cm diameter class contains 40% of the stems, and all stems >50 cm are Douglas-fir. Significantly, ~120 stems/ha are <2 cm diameter, most of which should be killed by surface fire. Western hemlock (*Tsuga heterophylla*) and western redcedar (*Thuja plicata*) are becoming abundant in these stands. Both are shade-tolerant fire-avoiders (Agee 1993, Fonda et al. 1998) in the DF3 stands, especially as saplings. They constitute a noticeable middle layer component in these stands, because of their reproductive effort (Figure 1A). In the absence of fire, these two species could replace Douglas-fir in the canopy, resulting in a change in stand structure and species composition. Conversely, in the presence of fire, these saplings would be killed, the middle layer would be removed, and the forest would be more open.

Understories of DF3 stands are dominated by the low-growing shrubs salal (*Gaultheria shallon*) (Figure 1A) , Oregongrape (*Mahonia nervosa*), and kinnikinnick (*Arctostaphylos uva-ursi*). Stair-step moss (*Hylocomium splendens*) can be extensive under this shrub layer. Pipsissewa (*Chimaphila umbellata*), twin-flower (*Linnaea borealis*), and pipe-cleaner moss (*Rhytidiopsis robusta*) are

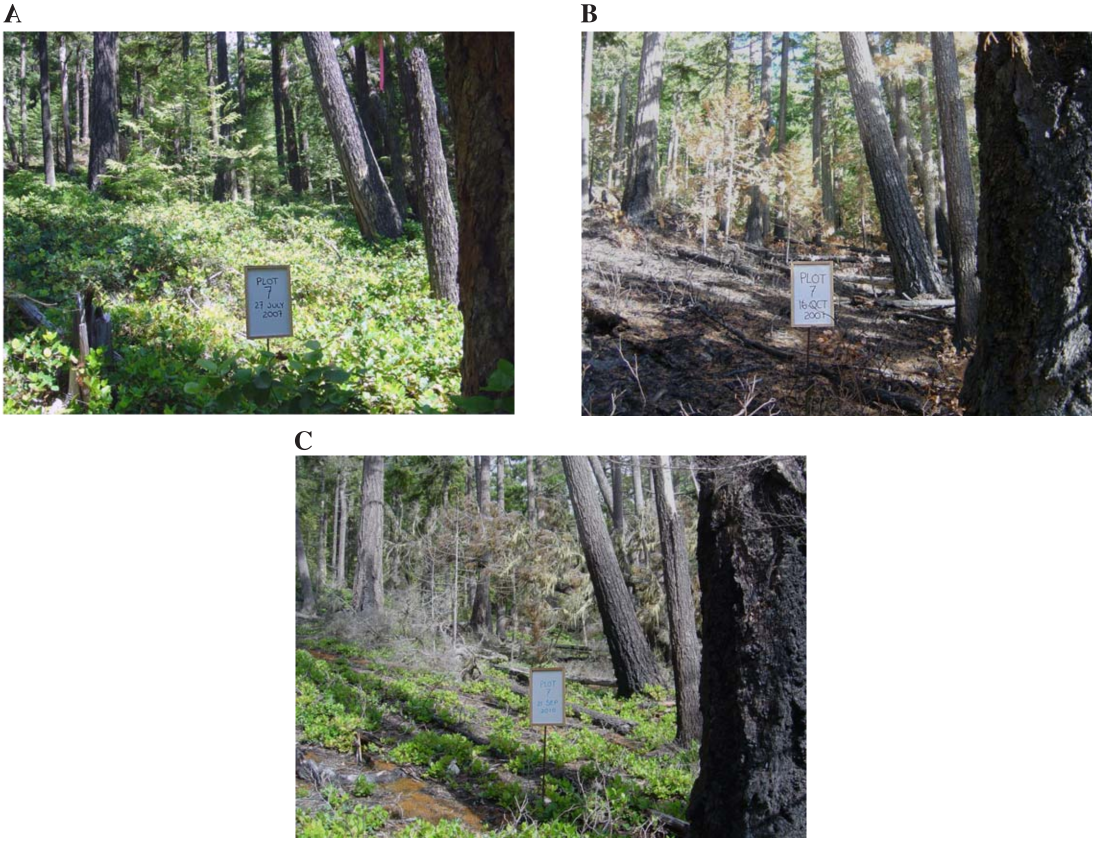

Figure 1. Photo sequence from pre-fire to +3 yr post-fire. A) Pre-fire (July 2007). The community structure is complex, with a canopy tree mix of Douglas-fir, western hemlock, and western redcedar, mid-level saplings, and a ~1 m deep shrub layer dominated by salal over a nearly continuous layer of stair-step moss. B) One month post-fire (October 2007). The salal layer is completely burned, as are the mosses that thrived below the salal; saplings are heavily scorched and will die within a year; the trees are lightly scorched on the lower trunks. Lower branches of ~30 cm dbh canopy trees have been scorched. Coarse woody fuels litter the ground. C) Three years post-fire (September 2010). The structure is simpler, and will be more open when the dead saplings fall. Salal has resprouted well, although far shorter than the pre-fire layer that reached the bottom of the sign board; stair-step moss is absent except in unburned patches. Two western hemlocks from the canopy layer have fallen across the plot, their dead branches visible in the center of the photo. Post-fire fuel loads, especially coarse woody fuels, have returned to pre-fire levels. (For figure in full color, see *Northwest Science* BioOne online site.)

less common. Salal, a known resprouter, is a fire-endurer (Agee 1993). The responses to fire of the 10+ minor species in this forest type are little known.

We applied prescribed fires to five 20 x 20 m plots in DF3 forests, monitored fire behavior, and gathered pre-fire and post-fire data on fuel loads, community structure, and species composition to test the conclusion in Wetzel and Fonda (2000) that 300+ yr old forests dominated by Douglas-fir in the Morse Creek drainage will support low intensity, low severity surface fires that enhance the structure and health of the stand. Specifically, the long-term structure should include light surface fuels, a tree canopy layer of larger, widely spaced Douglas-firs, and ~50% cover of <0.5 m understory shrubs. Furthermore, it should lack a middle layer of tree saplings. These were the first prescribed fires set in Olympic National Park.

We asked four research questions: 1) Will these forests carry fire sufficient to cause community changes for a few years after the fire? 2) Will fire reduce fuel loads by ~50%? 3) Will fire reduce Douglas-fir, western hemlock, and western redcedar reproduction (≤20 cm diameter) by 50-100%? 4) Will fire promote the post-fire growth of understory species, especially salal?

Methods

The research fire was conducted in old-growth (>300 yr) Douglas-fir/salal forests, along the west side of the Deer Park Road in the Maiden Creek drainage of Olympic National Park. The road acted as a fire break, and the firelines were anchored at both ends on the road, thus the burned area is easily visible. Two burn units (lower and upper) were identified ~3 km south of the northern park boundary. Five macroplots, 20 x 20 m (0.04 ha), were located in 2000, two in the lower unit (870 m elevation) and three in the upper unit (900 m elevation). Slope ranged from 14° to 21°. Pre-fire data on fuels, trees, and understory were initially gathered in 2000, but lack of a Park master fire plan caused the fires to be postponed until 2007. Pre-fire data were re-gathered in 2006; there were no significant differences between the 2000 and 2006 data; the 2006 data are used in this article.

The five macroplots were burned by prescription in September, 2007. After a buffer zone was burned around the entire unit, the macroplots were ignited from the bottom of the slope and the fires were allowed to burn uphill (Figure 2). Fuel moisture was <20%, and winds were calm to slight from the west. The small smoke column rose nearly vertically. Fire behavior was typical of slow-moving, low intensity surface fires, carried mainly by the salal layer. Rate of spread was slightly <1 m per minute, flame lengths were <2 m, and the flame zone depth conformed to the depth of the salal layer. Also typical was the patchy nature of the fires, from burning hot in some spots to missing other spots completely. The fires were ignited

Figure 2. Fire behavior ~15 min after the fire was ignited along the left side of the fireline at the bottom of the plot. The flame front has moved uphill (left) to the top of the plot; pockets of fire still burn in the salal layer and on low-hanging branches of a western hemlock sapling. Visible in the left foreground are the burned stems of salal and a western redcedar sapling. The large tree trunks bear little evidence of fire effects. (For figure in full color, see *Northwest Science* BioOne online site.)

at midday, the flame fronts moved through the macroplots in ~30 min, and the fires continued to smolder and creep overnight. Macroplot area blackened by the fires ranged from 50% to 98% (Figure 1B).

Downed woody fuel loadings were estimated by a modified transect microplot method (Brown 1974), using a systematic grid. Post-fire fuel loads were sampled twice, at +1 mo (October 2007) and +22 mo (July 2009).

The basic fuels microplot length was 16 m, located between meters 2 and 18 on a 20-m transect that began at the 3, 6, 9, 12, 15, and 18 m marks on the Y-axis. All downed woody fuels intersecting the 2-m vertical plane were tallied. Duff depth was measured by slicing into the soil with a hatchet at meters 5 and 9 of each transect.

All sound and rotten fuels >7.62 cm diameter (1000-hr fuels) were tallied on the 16-m microplot (96 m total length per macroplot); woody fuels <7.62 diameter were tallied on randomly selected segments of the microplots. Downed woody fuels 0.1 to 2.54 cm diameters (1-hr and 10-hr fuels) were tallied on 2 m segments in each microplot (12 m total length), and fuels 2.55 to 7.62 cm diameter (100-hr fuels) were tallied on 4 m of line (24 m total length). The 2 m and 4 m segments began at the same randomly selected meter within the microplot.

The diameters of all trees >1 cm diameter within the macroplot were measured pre-fire, and post-fire at +1 yr (2008) and +3 yr (2010). Density, population structure, and basal area were derived from these data.

Forty 20 x 50 cm microplots, on which cover and frequency of understory species were gathered, were located by a stratified-random arrangement within each 20 x 20 m macroplot. Each macroplot was stratified by 5-m segments along the X-axis, and one 20-m transect of 10 microplots was selected randomly to start from one of the X-axis meter marks within each segment (1-4; 6-9; 11-14; 16-19). Meter marks 0, 5, 10, 15, and 20 were eliminated from consideration so that we could move within the macroplot with minimal disruption to the understory.

Microplots 1-10 and 21-30 were located at odd-numbered meters on the first and third transects; microplots 11-20 and 31-40 were located at even-number meters on the second and fourth transects. At each microplot point, percent cover for each species, and the amount of litter and bare ground, were estimated to the nearest 1% for pre-fire (2006), and +1 (2008), +2 (2009), and +3 (2010) years post-fire. Frequency estimates were based on occurrence over all 40 microplots. We calculated species richness as number of species per macroplot.

The research was designed as a randomized complete block ANOVA, with a significance level of $P = 0.05$ chosen before the research began. Assumptions for ANOVA were met. Time relative to the fire constituted the treatments (t=3 for fuels and tree analyses; t=4 for understory analyses), and the macroplots constituted the five blocks. Cover percentage values were transformed by arc sine before analysis. Frequency values for the understory species were analyzed using raw occurrence values rather than percentages. Significant differences among the treatments were determined by a Newman-Keuls multiple range test.

Botanical nomenclature follows USDA, NRCS (2010) for vascular plants, and Schofield (1992) for mosses.

Results

Fuels

Total and 1000-hr fuel loads had the same statistical relationships. For both categories, post-fire +1 mo fuel loads were significantly lower than pre-fire or post-fire +22 mo fuel loads, but the differences between pre-fire and post-fire +22 mo were not significant (Table 1). Between the fire in 2007 and the final fuel measurements in 2009, many burned branches, saplings, and snags fell into the plot, so that fuels in these categories increased, creating the overlap in the results of the multiple range test (Table 1). The fire reduced total fuels by 57% and 1000-hr fuels by 59%, with a range

TABLE 1. Mean pre-fire and post-fire fuels (Mg ha^{-1}) and duff depth. Values with the same superscript are not significantly different.

Fuels category	Pre-fire	Post-fire +1 mo	Post-fire + 22 mo
Total fuels	46.8^{a}	20.3^{b}	32.7^{ab}
1000-hr	44.4^{a}	18.4^{b}	29.9^{ab}
100-hr	0.5^{a}	0.4^{a}	0.5^{a}
10-hr	1.3^{a}	1.3^{a}	1.7^{a}
1-hr	0.6^{a}	0.2	0.6^{a}
Duff depth (cm)	5.7	2.5^{a}	1.7^{a}

of 42% to 100% reduction for 1000-hr fuels, one month post-fire.

The 100-hr, 10-hr, and 1-hr fuels mostly did not differ significantly from pre-fire fuels at any time (Table 1). Only 1-hr fuels at post-fire +1 mo were significantly lower than pre-fire and post-fire +22 mo. Pre-fire duff depths were reduced significantly by fire (Table 1).

Canopy and Middle Layers

The typical structure of old-growth Douglas-fir forests in the eastern Olympics and in our study area consists of a well-developed overstory, with tree saplings in a middle layer, and a dense understory of shrubs (Figure 1A). Of the >1500 total trees ha^{-1} in the study area, 54% were ≤20 cm diameter (Table 2). Roughly one-third of Douglas-fir and western hemlock stems were ≤20 cm, but 83% of western redcedars and 100% of western white pines (*Pinus monticola*) were small saplings. These young trees, with a pre-fire estimated mean cover of 31%, were a target of the prescribed fires.

Total stem density was significantly reduced by fire (Table 2; Figure 1B). By 2010, post-fire +3 yr, only 42% of the pre-fire stem density existed in the plots (Figure 1C). Douglas-fir is a fire resister, which these data confirm. The difference between pre-fire and post-fire +1 yr Douglas-fir density was not significant, even among the smaller trees (Table 2). Although <200 Douglas-firs per hectare were killed by fire, mortality among smaller trees was not visible until 2010, so that post-fire +3 yr density was significantly lower than pre-fire density. In contrast, 325 western hemlocks and 260 western redcedars per hectare were killed. Post-fire western hemlock density was significantly lower than pre-fire density, although the total stem density of western redcedar did not differ significantly pre-fire versus post-fire. Nevertheless, saplings of both species decreased significantly because of fire (Table 2; Figure 1C). All of the western white pines on the study site were saplings; they were nearly obliterated by fire. In 2010, estimated mean cover of the middle layer was 6%; three macroplots had <1% cover in the middle layer.

Perhaps the best measure of the abilities of these larger trees to resist fire and remain healthy on the site is reflected by basal area, for which there were no significant differences among pre-fire and post-fire values (Table 2). Most trees removed by fire contributed little to the total and individual species basal area of the plot.

Understory Layer

Cover and frequency of all five species listed in Table 3 decreased significantly as a result of fire (Figure 1B and 1C). Total post-fire plant cover was ~40%, much lower than the pre-fire cover

TABLE 2. Pre-fire and post-fire mean tree density (stems ha^{-1}) and basal area (m^2 ha^{-1}). Values with the same superscript are not significantly different.

Species	Pre-fire	Post-fire + 1 yr	Post-fire + 3 yr
		Density of all stems	
Total	1565	815^{a}	655^{a}
Douglas-fir	545^{a}	420^{a}	355^{a}
Western hemlock	520	250^{a}	195^{a}
Western redcedar	360^{a}	140^{a}	100^{a}
Western white pine	140	5^{a}	5^{a}
		Density of stems ≤20 cm diameter	
Total	845	295^{a}	190^{a}
Douglas-fir	195^{a}	115^{a}	60^{a}
Western hemlock	210	70^{a}	35^{a}
Western redcedar	300	105^{a}	90^{a}
Western white pine	140	5^{a}	5^{a}
		Basal area	
Total	81.70^{a}	69.24^{a}	59.58^{a}
Douglas-fir	49.34^{a}	50.13^{a}	44.87^{a}
Western hemlock	28.74^{a}	13.74^{a}	13.28^{a}
Western redcedar	3.51^{a}	3.53^{a}	1.43^{a}
Western white pine	0.11^{a}	$<0.01^{a}$	$<0.01^{a}$

TABLE 3. Pre-fire and post-fire understory mean percent cover and frequency. Values with the same superscript are not significantly different.

Species	Pre-fire	Post-fire + 1 yr	Post-fire + 2 yr	Post-fire + 3 yr
			Cover	
Salal	53.8	14.2^{b}	23.4^{ab}	28.9^{a}
Stair-step moss	39.7	3.7^{a}	5.0^{a}	7.1^{a}
Oregongrape	3.4	0.5^{a}	0.7^{a}	1.0^{a}
Kinnikinnick	9.1	0.5^{a}	0.6^{a}	0.8^{a}
Pipsissewa	1.0	0.1^{a}	0.2^{a}	0.3^{a}
Litter	50.8	88.3^{a}	74.2^{a}	75.6^{a}
Rock & soil	0	3.1^{a}	2.5^{a}	3.0^{a}
			Frequency	
Salal	92	62	74^{a}	80^{a}
Stair-step moss	85	17^{a}	21^{a}	31
Oregongrape	27	10^{a}	13^{a}	15^{a}
Kinnikinnick	23	11^{a}	8^{a}	5^{a}
Pipsissewa	19	4^{a}	6^{a}	10^{a}
Litter	100^{a}	100^{a}	98^{a}	99^{a}
Rock & soil	0	15^{a}	20^{a}	19^{a}

total of >107%. Complex layering accounted for the high pre-fire cover (Figure 1A), a structure that was rare in post-fire plots (Figure 1C). Salal was the only post-fire species with >10% mean cover, and the only species with >50% frequency.

Although minimal cover values came in the first post-fire year, all species were present after the fire. Species richness did not differ significantly among any treatments. Only salal, the understory dominant of these forests, showed a significant post-fire increase in cover and frequency by year three, although frequency of stair-step moss was significantly higher in post-fire +3 yr compared to post-fire +1 yr and +2 yr (Table 3). Virtually all post-fire salal cover was a result of resprouting (Figure 1C), and only salal returned to >50% of pre-fire cover levels. Stair-step moss succeeded better than either Oregongrape or kinnikinnick. Although Oregongrape and kinnikinnick resprouted on burned sites, they apparently are much less vigorous than salal, and will take longer to repopulate the sites. Indeed, kinnikinnick is the only species that might be struggling in the post-fire environment (Table 3). The remaining species all had significantly lower post-fire frequency for all three measurement periods.

Litter was significantly higher in the post-fire forest (Table 3), a function of needle and twig drop from burned plants. Rock and bare soil often indicate spots where fire burned the entire duff layer, or where tree fall disturbed large soil patches. Although the extent of rock and bare soil was <5%, the values are significant because the pre-fire forest lacked evidence of rock and bare soil. The duff was burned more completely than these data indicate, because much bare soil was covered by post-fire litter.

Discussion

We used prescribed fires to study the community response to fire in the Morse Creek drainage of Olympic National Park, and confirmed the conclusion of Wetzel and Fonda (2000) that older Douglas-fir stands will support low intensity, low severity fires. The data demonstrate that these old-growth forests would carry fire through the understory and middle layer without canopy involvement (Figure 2), and that fuels would be reduced temporarily. Species composition would re-establish quickly, because the dominant species have strategies that allow individual plants to persist on the post-fire site. In sum, prescribed fires would promote the health of these old-growth forests, as suggested by Agee and Skinner (2005), but more than one prescribed fire would be needed, as suggested by Schwilk et al. (2009).

Douglas-fir is a known fire-resister; mature trees were not harmed. Although mature western hemlock and western redcedar trees resisted the fire, saplings and smaller trees of these species were killed (Table 2). Some seedlings of western hemlock appeared in the stands in 2010, but the likelihood of their survival is unknown.

The data from this study confirmed that salal, kinnikinnick, and Oregongrape in particular are fire-endurers, by virtue of resprouting. At this point 3 yr post-fire, however, all understory species must be classified as decreasers. Plants of these species were top-killed by fire, but belowground portions of the plants survived to re-establish aerial portions. Based on the rate of increase in cover and frequency of these species (Table 3), the shrubby understories of the burned plots should exceed 40% and approach pre-fire conditions in the near future. Stair-step moss and pipe-cleaner moss, however, were decimated by fire. They survived only on unburned or lightly burned microsites. It may take considerably longer than 10 yr for them to re-establish their pre-fire cover and frequency.

The prescribed fires reduced total fuels by 57% (Table 1), which slightly exceeded the research objective. The Olympic data lie between reported values for mixed conifer forests in California, which varied from ~90% fuels reductions (Knapp et al. 2005, Stephens and Moghaddas 2005, Keifer et al. 2006, Stephens et al. 2009) to <50% fuels reduction (Wayman and North 2007,Vaillant et al. 2009). Fuel reductions in western ponderosa pine forests average ~50%, although 99% was reported by Keifer et al. (2006). In general, however, reported fuel reductions were <60% (Youngblood et al. 2008, Stephens et al. 2009, Vaillant et al. 2009). The lowest reported value was 23% in the Blue Mountains (Youngblood 2008) and the Klamath Mountains (Vaillant et al. 2009). Even a cool spring burn in a ponderosa pine-Douglas-fir forest in the Cascade Mountains reduced total fuels by 59% (Agee and Lolley 2006). In mixed conifer and pine stands, downed woody fuels and flashy fine fuels were responsible for carrying the fire. In the Olympics, even though they were reduced significantly by fire, woody fuels were less instrumental in carrying the fire than the live shrub layer dominated by salal.

Our fuel sampling method tallied only dead and downed (detached) woody fuels (Brown 1974). A considerable amount of fine fuel, however, was represented by dead and attached salal stems. These attached fuels that contributed greatly to carrying the fire front through the stand were entirely consumed by fire (Figure 1B).

Despite the immediate post-fire fuel reduction, within two years fuel loads in all time lag categories were not significantly different from pre-fire fuel loads (Table 1). This relationship was noted by Schwilk et al.(2009), prompting them to argue that multiple prescribed fires would be required to restore a fuel structure stable with the fire environment.

The vegetation responses in the Olympic forests were consistent with responses in other communities in which Douglas-fir was prominent, mainly ponderosa pine-Douglas-fir forests in the Cascade Mountains (Dodson et al. 2008, Harrod et al. 2008), Blue Mountains (Metlen et al. 2004), or Montana Rockies (Metlen and Fiedler 2006, Dodson et al. 2007). Ponderosa pine dominated the canopy layer of those forests, and the understory tended toward discontinuous shrubs in an herbaceous matrix. In the Olympics, Douglas-fir dominated the tree layer, with western hemlock and western redcedar subordinates. And, salal formed a nearly continuous understory layer.

In these studies, the canopy layer was unaffected by fire. Reducing the density of tree saplings in the middle layer, however, was an important research objective in the Olympics. Total sapling density post-fire +3 yr decreased by 71% compared to pre-fire densities; post-fire densities for saplings of all species were significantly less dense than pre-fire densities (Table 2). Prescribed fire combined with pre-fire thinning significantly reduced sapling density by 50% in the Cascades (Harrod et al. 2008), and by 27% in the Blue Mountains (Metlen et al. 2004). Clearly, one of the most beneficial results of prescribed fire in over-stocked forest stands is a more open middle layer with few tree saplings. Such a structure disrupts vertical fuels, preventing fires from reaching the canopy layer (Agee and Skinner 2005).

Understory response in forests depends on the dominant growth forms. Herbaceous understories respond quickly in the post-fire environment, so that post-fire +3 yr cover seldom is significantly different from pre-fire cover (Metlen et al. 2004, Metlen and Fiedler 2006, Harrod et al. 2008). Shrub cover, however, takes longer to return. We expect these same species to re-establish eventually because of their resprouting abilities. The lack of significant differences among species richness values indicates that the 8-10 species consistently present in this study before and after fire are endurers. No species were lost, no species invaded from outside the plots, because of the fire. Shrub frequency recovered more rapidly in a Sierran mixed conifer forest than in the Olympics, yet Sierran shrub density demonstrated about the same relationships as Olympic shrub cover (Wayman and North 2007). Similarly, shrub density was significantly reduced in the Blue Mountains by 75-80% (Metlen et al. 2004). At least a decade will be required for salal to return to pre-fire cover and frequency values, and for the shrub layer to re-establish the layered complexity (i.e., salal over mosses) that existed before the prescribed fire.

This study contributed data on the role of prescribed fire in Olympic National Park, which should help solidify a fire management plan that will incorporate prescribed fire with fire suppression and non-suppression. For about a decade, the concept of forest health has become an important driver of management decisions about forest management. Although the term forest health is somewhat nebulous, and has many interpretations, the concept was identified to address connections among insect outbreaks, diseases, and wildfire (Quigley 1992, Mutch et al. 1993). Ultimately, management goals were developed to protect forested regions from stand-replacing wildfires. Managers strived to create conditions in forests that emulated natural conditions in which frequent fire was common, insect outbreaks were limited or non-existent, and diseases were easily resisted by the forest community (Washington Department of Natural Resources 2011).

The four principles of fire resistance, and their effects (Agee and Skinner 2005), identify processes that operate in naturally healthy forests. Removing surface fuels and ladder fuels (saplings) decreases the possibility of torching and crown fires. Reducing crown density and maintaining big trees of resistant species increases defenses against insects and disease agents, and greater distances among host trees, so that outbreaks are minimized. Managers today try to create this forest structure with fire and fire surrogates, such as thinning, but it always must be recognized that the historical, presumably healthy, conditions are the target (Agee and Lehmkuhl 2009).

Harrod et al. (2008) and Schwilk et al. (2009) advocated that a series of prescribed fires would be required to return stands to historical conditions. One prescribed fire, such as this one in Olympic National Park, will not restore fire-resistant stand structure. Although forest structure and composition, and fuels, were changed by the Olympic fires, it already is clear that some of the effects were short-lived. Soon, the understory layer will re-establish the pre-fire complexity, and saplings of the canopy trees will grow again to form a middle layer. In time, the forest will cycle back to a stage of diminished forest health, for which insect and disease outbreaks, and stand-replacing fire, will be more likely. Indeed, stand-replacing fires on 200-300 yr cycles have been documented in the Douglas-fir forests of the eastern Olympics (Fonda and Bliss 1969, Wetzel and Fonda 2000).

Olympic National Park has an established fire management plan. Managers will be tasked to integrate various approaches to fire management, so that a complex mosaic of ages and structures exists in the Douglas-fir zone of the eastern Olympics, thereby reducing stand-replacing fires that cover many hectares. Removing ladder fuels in the form of non-resistant tree saplings, and maintaining endurers in healthy structures, are important goals for maintaining forest health in these forests. The data from this research fire will contribute to formulating an on-going prescribed fire program for the eastern Olympic forests.

Acknowledgements

We are pleased to thank Todd Rankin, Larry Nickey, and Jerry Freilich of Olympic National Park, for their cooperation and support. The Olympic fire crew set the prescribed fires. Thanks are due to Mary Jo Case for her help with the field data collection. The manuscript benefitted from the valuable help of Richy Harrod, and three anonymous reviewers provided outstanding critical comments. Western Washington University provided financial support.

Literature Cited

Agee, J. K. 1993. Fire Ecology of Pacific Northwest Forests. Island Press, Covelo, California.

Agee, J. K., and J. F. Lehmkuhl (compilers). 2009. Dry forests of the northeastern Cascades fire and fire surrogates project site, Mission Creek, Okanogan-Wenatchee National Forest. USDA Forest Service Research Paper PNW-RP-577. Pacific Northwest Research Station, Portland, OR.

Agee, J. K., and M. R. Lolley. 2006. Thinning and prescribed fire effects on fuels and potential fire behavior in an eastern Cascades forest, Washington, USA. Forest Ecology 2:142-158.

Agee, J. K., and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211:83-96.

Armour, C. D., S. C. Bunting, and L. F. Neuenschwander. 1984. Fire intensity effects on the understory in ponderosa pine forests. Journal of Range Management 37:44-49.

Brown, J. K. 1974. Handbook for inventorying downed woody material. USDA Forest Service General Technical Report INT-16. Intermountain Forest and Range Experiment Station, Ogden, Utah.

Busse, M. D., S. A. Simon, and G. M. Riegel. 2000. Tree-growth and understory responses to low-severity prescribed burning in thinned *Pinus ponderosa* forests of central Oregon. Forest Science 46:258-268.

Dodson, E. K., K. L. Metlen, and C. E. Fiedler. 2007. Common and uncommon understory species differentially respond to restoration treatments in ponderosa pine/Douglas-fir forests, Montana. Restoration Ecology 15:696-708.

Dodson, E. K., D. W. Peterson, and R. J. Harrod. 2008. Understory vegetation response to thinning and burning restoration treatments in dry conifer forests of the eastern Cascades, USA. Forest Ecology and Management 255:3130-3140.

Fonda, R. W., and L. C. Bliss. 1969. Forest vegetation of the montane and subalpine zones, Olympic National Park, Washington. Ecological Monographs 39:271-301.

Fonda, R. W., L. A. Belanger, and L. L. Burley. 1998. Burning characteristics of western conifer needles. Northwest Science 72:1-9.

Glitzenstein, J. S., D. R. Streng, G. L. Achtemeier, L. P. Naeher, and D. D. Wade. 2006. Fuels and fire behavior in chipped and unchipped plots: implications for land management near the wildland/urban interface. Forest Ecology and Management 236:18-29.

Harrod, R. J., R. W. Fonda, and M. K. McGrath. 2008. Vegetation response to thinning and burning in a ponderosa pine forest, Washington. Northwest Science 82:141-150.

Huisinga, K. D., D. C. Laughlin, P. Z. Fulé, J. D. Springer, and C. M. McGlone. 2005. Effects of an intense prescribed fire on understory vegetation in a mixed conifer forest. Journal of the Torrey Botanical Club 132:590-601.

Keeling, E. G., A. Sala, and T. H. DeLuca. 2006. Effects of fire exclusion on forest structure and composition in unlogged ponderosa pine/Douglas-fir forests. Forest Ecology and Management 237:418-428.

Keifer, M., J. W. van Wagtendonk, and M. Buhler. 2006. Long-term surface fuel accumulation in burned and unburned mixed-conifer forests of the central and southern Sierra Nevada, CA (USA). Fire Ecology 2:53-72.

Knapp, E. E., J. E. Keeley, E. A. Ballenger, and T. J. Brennan. 2005. Fuel reduction and coarse woody debris dynamics with early season and late season prescribed fire in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 208:383-397.

Metlen, K. L., and C. E. Fiedler. 2006. Restoration treatment effects on the understory of ponderosa pine/Douglas-fir forests in western Montana, USA. Forest Ecology and Management 222:355-369.

Metlen, K. L., C. E. Fiedler, and A. Youngblood. 2004. Understory response to fuel reduction treatments in the Blue Mountains of northeastern Oregon. Northwest Science 78:175-185.

Mutch, R. W., S. F. Arno, J. K. Brown, C. E. Carlson, R. D. Ottmar, and J. L. Peterson. 1993. Forest health in the Blue Mountains: a management strategy for fire-adapted ecosystems. USDA Forest Service General Technical Report PNW-GTR-310. Pacific Northwest Research Station, Portland, OR.

Nelson, C. R., C. B. Halpern, and J. K. Agee. 2008. Thinning and burning results in low-level invasion by nonnative plants but neutral effects on natives. Ecological Applications 18:762-770.

Quigley, T. M. 1992. Forest health in the Blue Mountains: social and economic perspectives. USDA Forest Service General Technical Report PNW-GTR-296. Pacific Northwest Research Station, Portland, OR.

Ruha, T. L., J. D. Landsberg, and R. E. Martin. 1996. Influence of fire on understory shrub vegetation in ponderosa pine stands. *In* J. R. Barrow, E. D. McArthur, R. E. Sosebee, and R. J. Tausch (compilers), Proceedings of the Symposium on Shrubland Ecosystem Dynamics in a Changing Environment. USDA Forest Service General Technical Report INT-GTR-338. Intermountain Research Station, Ogden, UT.

Schofield, W. B. 1992. Some Common Mosses of British Columbia. Royal British Columbia Museum, Victoria, British Columbia, Canada.

Schwilk, D. W., J. E. Keeley, E. E. Knapp, J. McIver, J. D. Bailey, C. J. Fettig, C. E. Fiedler, R. J. Harrod, J. J. Moghaddas, K. W. Outcalt, C. N. Skinner, S. L. Stephens, T. A. Waldrop, D. A. Yaussy, and A. Youngblood. 2009. The national fire and fire surrogate study: effects of fuel reduction methods on forest vegetation structure and fuels. Ecological Applications 19:285-304.

Stephens, S. L., and J. J. Moghaddas. 2005. Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. Forest Ecology and Management 215:21-36.

Stephens, S. L., J. J. Moghaddas, C. Edminster, C. E. Fiedler, S. Haase, M. Harrington, J. E. Keeley, E. E. Knapp, J. D. McIver, K. Metlen, C. N. Skinner, and A. Youngblood. 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U. S. forests. Ecological Applications 19:305-320.

Tveten, R. K., and R. W. Fonda. 1999. Fire effects on prairies and oak woodlands on Fort Lewis, Washington. Northwest Science 73:145-158.

USDA, NRCS. 2010. The PLANTS Database. National Plant Data Center, Baton Rouge, LA 70874-4490 USA. Available online at http://plants.usda.gov. (accessed 18 December 2010).

Vaillant, N. M., J. A. Fites-Kaufman, and S. L. Stephens. 2009. Effectiveness of prescribed fire as a fuel treatment in Californian coniferous forests. International Journal of Wildland Fire 18:165-175.

Washington Department of Natural Resources. 2011. Forest Heath and Ecology. Available online at http://www.dnr.wa.gov/ResearchScience/Topics/ForestHealthEcology/Pages/rp_foresthealth.aspx. (accessed 11 January 2011).

Wayman, R. B., and M. North. 2007. Initial response of a mixed-conifer understory plant community to burning and thinning restoration treatments. Forest Ecology and Management 239:32-44.

Wendel, R., and D. Zabowski. 2010. Fire history within the lower Elwha River watershed, Olympic National Park, Washington. Northwest Science 84:88-97.

Wetzel, S. A., and R. W. Fonda. 2000. Fire history of Douglas-fir forests in the Morse Creek drainage of Olympic National Park, Washington. Northwest Science 74:263-279.

Youngblood, A., C. S. Wright, R. D. Ottmar, and J. D. McIver. 2008. Changes in fuel bed characteristics and resulting fire potentials after fuel reduction treatments in dry forests of the Blue Mountains, northeastern Oregon. Forest Ecology and Management 255:3151-3169.

Received 18 October 2010
Accepted for publication 19 January 2011