CSIRO PUBLISHING

International Journal of Wildland Fire **2016**, *25*, 633–645
http://dx.doi.org/10.1071/WF14223

Forest fuels and potential fire behaviour 12 years after variable-retention harvest in lodgepole pine

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Abstract. Variable-retention harvesting in lodgepole pine offers an alternative to conventional, even-aged management. This harvesting technique promotes structural complexity and age-class diversity in residual stands and promotes resilience to disturbance. We examined fuel loads and potential fire behaviour 12 years after two modes of variable-retention harvesting (dispersed and aggregated retention patterns) crossed by post-harvest prescribed fire (burned or unburned) in central Montana. Results characterise 12-year post-treatment fuel loads. We found greater fuel load reduction in treated than untreated stands, namely in the 10- and 100-h classes ($P = 0.002$ and 0.049 respectively). Reductions in 1-h ($P < 0.001$), 10-h ($P = 0.008$) and 1000-h ($P = 0.014$) classes were greater in magnitude for unburned than burned treatments. Fire behaviour modelling incorporated the regenerating seedling cohort into the surface fuel complex. Our analysis indicates greater surface fireline intensity in treated than untreated stands ($P < 0.001$), and in unburned over burned stands ($P = 0.001$) in dry, windy weather. Although potential fire behaviour in treated stands is predicted to be more erratic, within-stand structural variability reduces probability of crown fire spread. Overall, results illustrate trade-offs between potential fire attributes that should be acknowledged with variable-retention harvesting.

Additional keywords: custom fire behaviour fuel models, fuel accumulation, fuel treatments, Little Belt Mountains, multiaged silviculture, northern Rocky Mountains, Tenderfoot Creek Experimental Forest.

Received 20 December 2014, accepted 28 January 2016, published online 20 April 2016

Introduction

Contemporary silviculturists and ecologists advocate that inter- and intrastand structural diversity can promote long-term landscape and ecosystem resilience to a suite of disturbances (Drever *et al.* 2006; Puettmann *et al.* 2009; Keyes *et al.* 2014). This capacity for resilience is especially important because changes in climate are projected to alter the temperature and precipitation drivers that impact critical disturbances, including wildland fire and bark beetles (Chapman *et al.* 2012). Recent decades in public land management host numerous ecosystem stewardship treatments designed to increase forest resilience (e.g. Schultz *et al.* 2012). In the United States, the Forest Service aims to 'restore, sustain, and enhance the Nation's forests' (USDA 2007); thus, management places an emphasis on ensuring critical ecological processes will persist following wildland fire, insect epidemics or climate-related disturbances. Yet silvicultural treatments designed to enhance resilience by promoting structural and age-class diversity are rarely applied to lodgepole pine (*Pinus contorta* Douglas ex Louden var. *latifolia* Engelm. ex S. Watson).

Lodgepole pine (LP) has the most extensive range of any conifer in western North America. It is the dominant forest cover over ~26 million ha. LP is typically considered a shade-intolerant and fire-adapted pioneer species that often regenerates naturally as dense, even-aged stands (Lotan and Critchfield 1990). As such, traditional silvicultural systems in LP-dominated forests aim to produce continuous canopied stands and are the epitome of even-aged management (Schmidt and Alexander 1985). Though even-aged management of LP mimics age distributions arising from one of its most common disturbance agents (i.e. stand-replacing fire), mixed-severity disturbances are also common and often result in multi-aged LP stands (Arno 1980; Agee 1993; Kollenberg and O'Hara 1999; Axelson *et al.* 2010). This indicates other silvicultural alternatives can mimic live structures created by natural disturbances to enhance forest resilience.

Multi-aged management of LP forests can improve both structural complexity and age-class diversity to a degree that supports variable light infiltration, cohort regeneration, wildlife forage and tree vigour (Schmidt and Alexander 1985; O'Hara

2014). Furthermore, multi-aged silviculture can complement spatially expansive even-aged regeneration systems such as clear-cutting to promote heterogeneous stand and landscape conditions resilient to primary disturbances (i.e. bark beetle and wildland fire) (O'Hara 1998; Axelson *et al.* 2010; Johnson *et al.* 2014; Keyes *et al.* 2014).

One flexible silvicultural tool for multi-aged management is the variable-retention harvest (VRH) (Franklin *et al.* 1997; Gustafsson *et al.* 2012). This tree-harvesting approach enables managers to emulate the spatial, structural and age complexity historically maintained in natural forests mosaicked by a suite of disturbances. However, little is known of the long-term effects of implementing these treatments, as VRHs are not currently part of a formal silvicultural management system (e.g. as outlined in Smith *et al.* 1997). Critical evaluation is required to determine the impacts of this multi-aged management approach on post-treatment disturbance processes. In particular, we need to know if this fairly new strategy alters fuel conditions that drive the potential for stand-replacing wildfire. Treatments may exacerbate fire behaviour by increasing near-surface wind speed owing to reduced stem density, increasing dead surface fuel loads as treatments relocate crown fuels to the ground and by promoting the ingrowth of natural regeneration and ladder fuels into the surface fuel complex (Keyes and Varner 2006).

In the present study, we examined the effects of an experimental VRH in Rocky Mountain LP on surface woody debris accumulation and simulated fire behaviour. This experiment was established within the Tenderfoot Creek Experimental Forest (TCEF) in central Montana and was specifically designed to initiate two-aged stands. The VRH resulted in two forest structural patterns, according to spatially aggregated or dispersed overstorey tree retention targets, and subsequently half of each of the harvested stands was burned.

Our first research question was: how do harvest pattern and use of prescribed fire influence downed woody debris (DWD) dynamics? We quantified post-treatment fuel loadings 2–4 and 12 years post-harvest to address this query. Second, we asked if these treatments increase or decrease the potential for crown fire in residual overstories. We simulated potential fire using data-driven custom fire behaviour models at multiple plots within stands to investigate the variability of potential fire behaviour 12 years after VRH. The fire behaviour predictions we present provide an integral assessment of this multi-aged management strategy, and the relative findings are relevant where VRHs are implemented by forest managers in, but not exclusive to, LP forest types.

Methods

Study site

The TCEF is a 3693-ha watershed in the Little Belt Mountains, within the Lewis and Clark National Forest in central Montana. Elevation ranges from 1840 to 2421 m above sea level. The forest is dominated by LP, forming nearly pure even-aged and two-aged stands. Associated overstorey species are Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) and subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.). Associated shrub species are grouse whortleberry (*Vaccinium scoparium* Leiberg ex Coville) and thinleaf huckleberry (*V. globulare* Douglas ex Torr.). Soils are typified by loamy skeletal, mixed Typic Cryochrepts, and clayey, mixed Aquic Cryoboralfs (Adams *et al.* 2008).

Climate in the study area is generally continental, though is also influenced by the Pacific maritime climate along the Continental Divide. Annual precipitation is 880 mm, ranging from 594 to 1050 mm across the elevation gradient. The majority of the precipitation occurs in the form of snow from November to May. Typical mean temperatures range from −9°C in January to 17°C in July, with freezing temperatures possible throughout the year. The average plant growing season is estimated to be between 30 and 75 days (Adams *et al.* 2008).

Fire history reconstruction revealed a characteristic mixed-severity fire regime in the study area (Hardy *et al.* 2006). For the period of 1580 to 1992, mean fire return interval was 38 years, with large, severe fires occurring infrequently, and low- to mixed-severity fires occurring between large, severe fire events.

Experimental design and sampling

Treatments were installed in 16 units, split among two subwatersheds of Tenderfoot Creek (McCaughey *et al.* 2006). Units in two adjacent subwatersheds were established as untreated reserves (hereafter, 'controls'). The VRH prescription called for 50% basal area retention and created two stand structure types: aggregated, where residual overstorey was distributed in a clumped spatial pattern; and dispersed, where residual overstorey was primarily distributed at an even spacing (Fig. 1). Half of the units were broadcast-burned with low-intensity fire (though severity was greater than anticipated; see Hood *et al.* 2012). Burned treatments are labelled 'B' in tables and figures, whereas unburned treatments are labelled 'U'. Thus, there were a total of

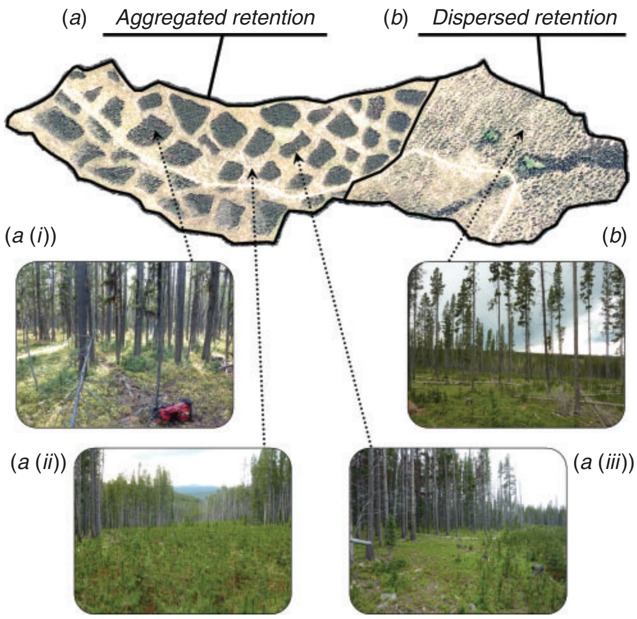

Fig. 1. Photo-diagram of variable-retention harvest structural conditions: aerial perspective and typical stand profiles of points within aggregated (*a*), and dispersed retention (*b*) management units. Note that the natural consequence of the aggregated retention is three distinct within-stand structural elements: (*a* (*i*)) retained patch interior, (*a* (*ii*)) clearing, and (*a* (*iii*)) the patch-to-clearing edge interface.

16 treatment units: two replications of burn × harvest treatment per subwatershed. Harvesting took place in 1999 and 2000; stands selected to be treated with prescribed fire were burned in 2002 or 2003. Pre- and post-harvest stand conditions have been documented in detail as a restoration guide (Hood *et al.* 2012).

Surface fuels and live tree characteristics were sampled after treatment. A planar-intercept sampling method (Brown 1974) was used to estimate DWD following treatment completion (2002–04, as reported in Hardy *et al.* 2006), and then again in 2012. A total of 281 sample points were systematically located throughout stands, whereupon a set of two randomly oriented, perpendicular transects were established. To avoid trampling, transects were offset 3.28 m from sample points. On each transect, the following fuel characteristics were measured: 1000-h fuels (diameter >7.62 cm) on 19.8-m sections; 100-h fuels (2.54 < diameter < 7.62 cm) on 3-m sections; and 1-h (diameter <0.64 cm) and 10-h fuels (0.64 < diameter < 2.54 cm) on 1.8-m sections. Combined litter and duff depths were measured at two points along each transect.

We measured live tree characteristics at a subset (180 plots) of the surface fuel points using nested, fixed-area plots in 2011. Overstorey trees were sampled using 0.04-ha circular plots, wherein we recorded diameter and species for all live stems greater than 10.16-cm diameter at breast height (DBH). Height was predicted from tree DBH using a local DBH-to-height regression equation (C. Keyes, unpubl. data). Seedlings (DBH < 10.16 cm) were tallied on 0.001-ha circular subplots according to species and height.

Downed woody debris

Non-rotten woody loadings were calculated per time-lag size class (Brown 1974). Litter depth was assumed to be one-third of the total litter-duff depth measured; litter load was calculated at the rate of 4.41 Mg ha^{-1} cm^{-1} (D. Lutes, pers. comm.). Duff load calculation followed an equation developed for LP–subalpine fir forests of the Eastern Cascades (Woodard and Martin 1980). Estimates were averaged by sample point and compared with reference conditions (Brown and Bevins 1986; Baker 2009; Fuel Characteristic Classification System (FCCS), Ottmar *et al.* 2007).

As stated in our first objective, we contrasted the relative effects of treatments on DWD. Our four statistical null hypotheses were: (H1) there is no difference between treatment and control surface fuel loads; (H2) there is no difference between aggregated and dispersed retention surface fuel loads; (H3) there is no difference between burned and unburned surface fuel loads; and (H4) there is no treatment interaction (retention pattern × burn status) effect on surface fuel loads. We modelled current fuel loads by fuel class to test these hypotheses as mutually orthogonal linear contrasts. Fitted linear mixed-effects models had the form:

$$y_{ijkl} = \mu + B_i + \varepsilon_{(1)i} + R_j + \varepsilon_{(2)ij} + T_k + \varepsilon_{(3)ijkl}$$

where y_{ijkl} is the load in a given fuel class (i.e. 1-h, 10-h, 100-h, sound 1000-h, litter + duff load, or total dead surface fuel load) on plot l; μ is the grand mean load in the fuel class; B_i is the random effect of the ith subwatershed block ($i = 1,2$); R_j is the random effect of the jth treatment replicate within a block ($j = 1,2$); T_k is the fixed effect of the kth treatment level ($k = 1,2,3,4,5$; four treatments plus control); $\varepsilon_{(1)i} \sim N\left(0, \sigma^2_{\varepsilon(1)}\right)$, $\varepsilon_{(2)ij} \sim N\left(0, \sigma^2_{\varepsilon(2)}\right)$, and $\varepsilon_{(3)ijkl} \sim N\left(0, \sigma^2_{\varepsilon(3)}\right)$ are independent. Models were fitted using *R* statistical software (R Core Team 2013) and the package nlme (Pinheiro *et al.* 2013) using a constant variance function structure to account for treatment heteroscedasticity where appropriate. We examined normal quantile plots and correlations between predicted and observed values for model validation.

To better understand fuel dynamics resulting from harvesting and burning treatments, we calculated the change in loadings by fuel class between measurements. Net load was linearly annualised to account for slight differences in intermeasurement period length. We modelled net annual fuel load (ΔAFL) by fuel class to determine the effect of treatment on load accumulation. ΔAFL responses were modelled and linear contrasts were analysed using the procedure described above.

Potential fire behaviour

A noteworthy problem with typical application of fire behaviour models is a reliance on default fire behaviour fuel models (FMs), which often vary substantially from *in situ* fuel conditions. Customised model inputs are more appropriate where data are available (Varner and Keyes 2009). Furthermore, if silviculturists are interested in creating and managing for complex structural attributes, it is not appropriate to focus solely on stand-level mean values or coarsely averaged fuel loading to characterise potential fire behaviour (Agee and Lolley 2006). In the same fashion, default FMs that do not adequately represent highly variable fuel characteristics in the field may lead to fire behaviour simulations that are insufficient to accurately contrast heterogeneous stand conditions.

We used BehavePlus (v. 5.0.5; Andrews *et al.* 2008) and First Order Fire Effects Model (FOFEM v. 6.0; Reinhardt *et al.* 1997) to characterise potential fire behaviour resulting from the applied harvest and burning treatment combinations. We created customised FMs for each measured plot, electing to use unvalidated but data-driven FMs over default FMs that poorly matched plot-level characteristics. Surface fire was modelled using BehavePlus, fire intensity was adjusted based on parallel FOFEM modelling, and then potential for crown fire was modelled using BehavePlus. We used this model routine to better account for the wide array of DWD present in these novel fuelbeds.

The present study's FMs were informed by DWD loadings, biomass of observed regeneration, live fuel loads derived from the Fire and Fuels Extension of the Forest Vegetation Simulator (FVS-FFE; Beukema *et al.* 1997), and the most similar standard FMs (Anderson 1982; Scott and Burgan 2005). Table 1 shows the inputs used to develop FMs. Because the FMs were not field-validated, we focussed our interpretation of simulated fire behaviour on relative differences between treatment classes rather than absolute values.

We used four predefined fire weather (wind and fuel moisture) conditions in this analysis for comparative purposes (Scott and Burgan 2005; Table 2). Overstorey canopy characteristics were calculated from sample tree data according to FVS algorithms. We calculated live herb load using FVS; live woody understorey loads were calculated using FVS shrub load plus tree regeneration load (Brown 1978). Surface-wind adjustment factors ranged from 0.1 to 0.4 per overstorey canopy cover (Rothermel 1983).

Table 1. Custom fire behaviour fuel model assignment coefficients calculated or assumed for fire behaviour simulations within BehavePlus and FOFEM

Fuel models were assigned to each measured plot. SAV, surface area-to-volume ratio; FVS-FFE, Fire and Fuels Extension of the Forest Vegetation Simulator; BTU, British thermal unit

Characteristics	Metric units	Imperial units	Value – metric (imperial)	Derivation
1-h fuel load	Mg ha^{-1}	tons acre^{-1}	[plot-specific]	Calculated after Brown (1974)
10-h fuel load	Mg ha^{-1}	tons acre^{-1}	[plot-specific]	Calculated after Brown (1974)
100-h fuel load	Mg ha^{-1}	tons acre^{-1}	[plot-specific]	Calculated after Brown (1974)
Live herbaceous fuel load	Mg ha^{-1}	tons acre^{-1}	[plot-specific]	FFE-FVS FUELOUT herb load
Live woody fuel load	Mg ha^{-1}	tons acre^{-1}	[plot-specific]	FFE-FVS FUELOUT shrub load + calculated seedling load (foliage + half of 1-h branch load; Brown (1978))
Fuel model type	–	–	'static'	–
1-h dead SAV	cm^{-1}	foot^{-1}	60 960 (2000)	Anderson (1982), Scott and Burgan (2005)
Live herbaceous SAV	cm^{-1}	foot^{-1}	50 292 (1650)	Compromise between Anderson (1982), Scott and Burgan (2005)
Live woody SAV	cm^{-1}	foot^{-1}	47 244 (1550)	Compromise between Anderson (1982), Scott and Burgan (2005)
Fuel bed depth	cm	feet	[plot-specific]	Seedling density-modified seedling height
Moisture of extinction	%	%	[plot-specific]	Modified by overstorey canopy cover, as reflected in Anderson 1982; Scott and Burgan (2005)
Dead heat content	J kg^{-1}	BTU lb^{-1}	18 607 978 (8000)	Anderson (1982), Scott and Burgan (2005)
Live heat content	J kg^{-1}	BTU lb^{-1}	18 607 978 (8000)	Anderson (1982), Scott and Burgan (2005)

Table 2. Live and dead fuel moistures and wind scenarios modelled using BehavePlus and FOFEM

Fuel moistures are based on Scott and Burgan (2005)

	Moisture content (%)						
Scenario name	1-h fuels	10-h fuels[A]	100-h fuels	Live herbaceous fuels[D]	Live woody fuels[D]	Canopy foliar	6.1-m wind speed (km h^{-1})
Dry–low wind	3	4[B]	5	60	90	100	16.1
Dry–high wind	3	4[B]	5	60	90	100	40.2
Moist–low wind	12	13[C]	14	60	90	100	16.1
Moist–high wind	12	13[C]	14	60	90	100	40.2

[A]Plots with canopy cover >50% were assigned 3.25% greater 10-h moisture, per Rothermel (1983).
[B]Based on Scott and Burgan (2005) dead fuel moisture scenario D1.
[C]Based on Scott and Burgan (2005) dead fuel moisture scenario D4.
[D]Based on Scott and Burgan (2005) live fuel moisture scenario L2.

We report a suite of potential fire behaviour metrics across the four weather scenarios. We tested for differences in mean surface fireline intensity across treatments using the linear model framework outlined above. The potential for crown fire was assessed by examining the variability of critical surface fire flame length for canopy ignition, critical fire rate of spread for sustained canopy fire, and transition ratio (predicted flame length divided by critical flame length). Transition ratio was modelled in the same fashion as fireline intensity.

Finally, we generated heat release profiles for each plot based on fuel availability and plot environmental conditions. Whereas heat release at time 0 is indicative of frontal flaming and fire spread, subsequent flaming and smouldering has substantial effects on biota and post-fire fuel loads. We assessed within and among treatment variability both visually and with general descriptive statistics.

Results

Fuel characteristics

The grand mean of total stand-level dead surface fuels across treatments and controls (12 years post-harvest, and 9–10 years post-burn) was 81.59 Mg ha^{-1} (average of treatment-level values reported in Table 3). Total dead surface fuel loads ranged from 48.96 Mg ha^{-1} in one aggregated burned stand to 124.79 Mg ha^{-1} in a control stand (see treatment means in Table 3). Model residual standard errors (in Mg ha^{-1}) were as follows: 0.19 for 1-h current fuel load, 1.50 for 10-h, 3.14 for 100-h, 20.02 for 1000-h, 29.28 for duff and litter, and 40.09 for total dead surface fuel. Squared predicted-to-observed correlations ranged from 0.084 for the 1-h model to 0.32 for the 1000-h model, but fixed-effects only contributed up to 0.09 to the squared correlations. Our model contrasts show that total dead surface fuel loads in treated stands were no different than untreated stands (statistical hypothesis H1; Table 4), though there is mild to strong evidence for differences in the 10-, 100- and 1000-h fuel classes. Dispersed retention treatments were associated with greater 1000-h fuel load than aggregated treatments, but less 1-h load (H2). The contrasts also indicate greater loading in unburned than burned stands, except for in the 1000-h fuel class where the opposite case holds (H3). Interaction between the main effects was evident only in the 10-h fuels (H4).

Average annual change in fuel load (ΔAFL) for individual stands varied from −0.07 to 0.00 Mg ha^{-1} year^{-1} within the 1-h

Table 3. Treatment means and standard errors of downed woody debris by fuels class, 12 years after variable-retention harvest and 9–10 years after prescribed fire in Tenderfoot Creek Experimental Forest

Three sets of no-treatment reference means are provided for comparison. The Brown and Bevins (1986) lodgepole pine fuelbed means were developed using an average of four sites across Idaho, Montana and Wyoming. The means presented by Baker (2009) were derived using the Fuel Characteristic Classification System (FCCS; Ottmar *et al.* 2007) for lodgepole pine fuelbeds across the entirety of the United States Rocky Mountain range. The reference means in the final row of the table were calculated using FCCS given the typical overstorey condition of the present study's control units. B, burned treatments; U, unburned treatments

	Litter and duff	1-h	10-h	100-h	Sound 1000-h	Total
			($Mg\ ha^{-1}$)			
Control	48.16 (5.43)	0.24 (0.03)	1.69 (0.27)	3.64 (1.21)	34.07 (7.73)	87.83 (11.15)
Aggregated: B	40.35 (5.31)	0.22 (0.03)	1.96 (0.26)	3.88 (1.24)	25.54 (7.69)	72.07 (11.05)
Aggregated: U	49.13 (5.19)	0.25 (0.03)	2.41 (0.25)	5.92 (1.28)	22.73 (7.65)	80.46 (10.94)
Dispersed: B	36.20 (5.43)	0.15 (0.03)	1.38 (0.27)	4.41 (1.22)	36.40 (7.73)	78.67 (11.16)
Dispersed: U	49.57 (4.98)	0.23 (0.03)	2.72 (0.24)	6.02 (1.29)	30.51 (7.58)	88.91 (10.76)
Brown and Bevins (1986)	1.26 (litter only)	0.40	1.50	4.34	–	–
Baker (2009)	35.4	1.1	6.0	7.6	25.9	76.0
FCCS	32.45	0.90	4.93	6.28	21.30	33.40

Table 4. Estimated linear contrasts of dead surface fuel loads ($Mg\ ha^{-1}$) from mixed-effects models 12 years after variable-retention harvest and 9–10 years after prescribed fire in Tenderfoot Creek Experimental Forest

Response variables were modelled first at the year 2012, and second as net annual fuel load (ΔAFL) evaluated over post-treatment years

	1-h		10-h		100-h		Sound 1000-h+		Litter and duff		Total dead surface fuel	
Contrast test	Estimate	*P* value	Estimate	*P* value	Estimate	*P* value	Estimate	*P* value	Estimate	*P* value	Estimate	*P* value
Year 2012												
Grand mean	0.218	<0.001	2.033	<0.001	4.773	<0.001	29.851	<0.001	44.681	<0.001	81.589	<0.001
Control – treated	0.027	0.362	−0.427	0.070	−1.415	0.005	5.274	0.093	4.347	0.343	7.796	0.214
Aggregated – dispersed	0.050	0.045	0.134	0.500	−0.317	0.554	−9.317	0.001	1.856	0.632	−7.527	0.157
Burned – unburned	−0.056	0.025	−0.899	<0.001	−1.824	0.001	4.351	0.103	−11.075	0.005	−9.315	0.082
Treatment interaction	0.025	0.310	0.440	0.028	−0.211	0.695	−1.544	0.561	2.301	0.553	0.928	0.861
Net annual fuel load												
Grand mean	−0.042	<0.001	−0.125	<0.001	−0.041	0.237	0.472	<0.001	−1.096	0.002	−0.838	0.015
Control – treated	0.012	0.065	0.148	<0.001	0.166	0.009	−0.374	0.106	1.312	0.008	1.270	0.037
Aggregated – dispersed	0.005	0.379	0.067	0.064	0.074	0.165	−0.225	0.248	−0.067	0.872	−0.157	0.758
Burned – unburned	0.028	<0.001	0.094	0.010	0.042	0.432	0.714	<0.001	−0.261	0.530	0.599	0.243
Treatment interaction	0.000	0.990	0.038	0.301	0.027	0.614	0.011	0.957	0.017	0.968	0.098	0.848

component and increased with fuel size to −0.69 to 1.77 $Mg\ ha^{-1}\ year^{-1}$ within the 1000-h component (means of treatment-level values reported in Fig. 2). Total dead surface fuel load was most influenced by the change in combined litter and duff load, which ranged from −3.13 to 0.90 $Mg\ ha^{-1}\ year^{-1}$ in individual stands (see treatment means in Fig. 2). Model residual standard errors ($Mg\ ha^{-1}\ year^{-1}$) were: 0.04 for 1-h ΔAFL, 0.23 for 10-h, 0.40 for 100-h, 1.47 for 1000-h, 3.14 for duff and litter, and 3.88 for total dead surface fuel. Squared predicted-to-observed correlations ranged from 0.03 for the total fuel model to 0.17 for the 1-h model; fixed-effects contributed 0.02 to 0.10 to the squared correlations. Tests on estimated contrasts confirmed that ΔAFLs are significantly different from zero in the 1-h (−), 10-h (−), 1000-h (+), litter and duff (−) and total (−) fuel classes (Table 4). Treated stand ΔAFL was lower than untreated stands except for in the 1000-h class, indicating that fuels less than 7.62 cm as well as litter and duff have been more rapidly accumulating in control stands. There was some weak evidence that 10-h ΔAFL in aggregated treatments was greater than those in the dispersed treatments. Burned stands had significantly greater ΔAFL than unburned in the 1-, 10- and 1000-h classes, which highlights that both fine and coarse woody debris fall from the canopy to the surface in the years after burning.

Means of live fuel characteristics indicate that stands exhibit distinct structural variability 12 years after harvest (Table 5). As controls were not harvested or burned, overstorey density and basal area were greatest in untreated stands. Despite identical basal area targets in the dispersed and aggregated retention prescriptions, residual stem density and basal area were two-thirds to one-half less in the dispersed stands than aggregated stands. Estimated canopy bulk densities follow accordingly; at the plot level, they range from 0 to 0.20 $kg\ m^{-3}$ (see treatment-level means in Table 5). Herb loads were inversely related to overstorey cover; as calculated by FVS, these values range from 0.16 to 0.40 $Mg\ ha^{-1}$ per plot. Owing to dense patches of regeneration, live understorey woody loads likewise had an inverse relationship with overstorey cover that ranged from 0.05 to 4.49 $Mg\ ha^{-1}$ per plot (treatment means in Table 5).

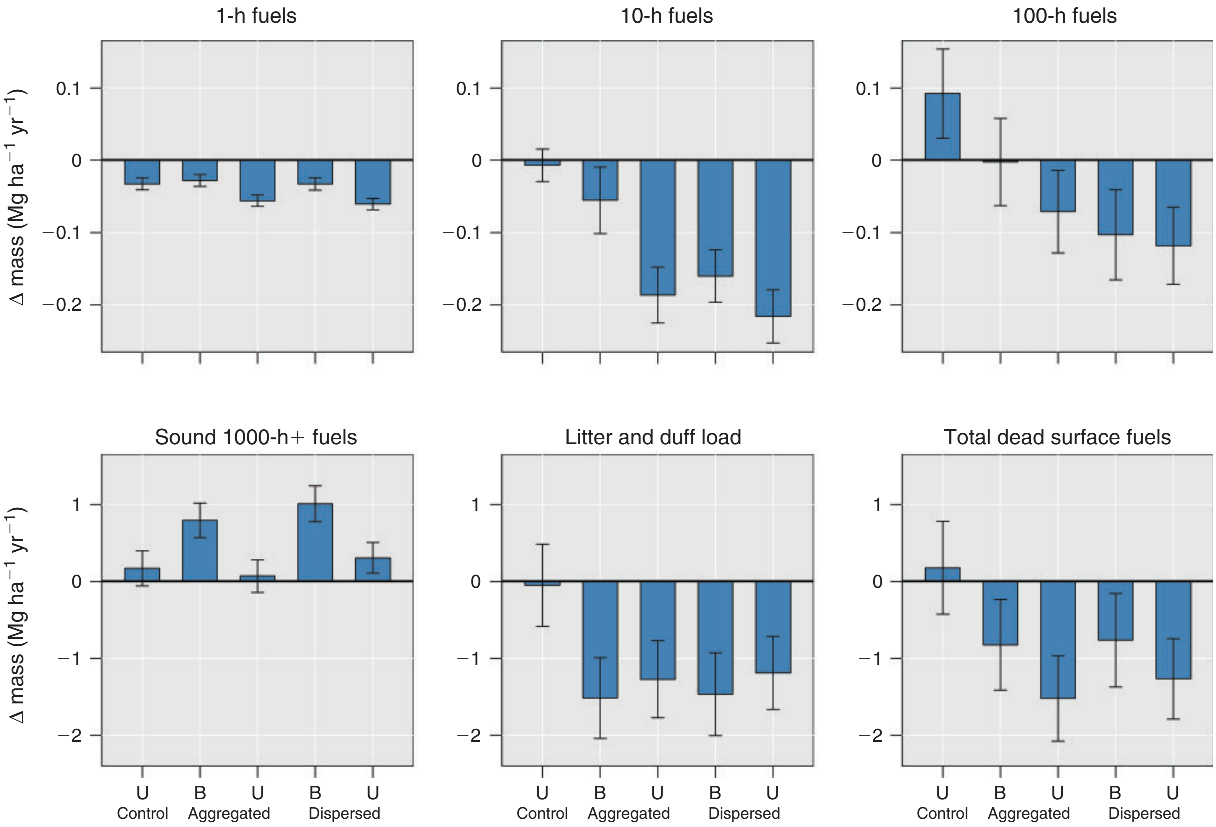

Fig. 2. Net annual fuel load (ΔAFL) of surface-fuel components in 12 years after variable-retention harvest and 9–10 years after prescribed fire in Tenderfoot Creek Experimental Forest.

Table 5. Live vegetation characteristics (mean and standard error) 12 years after variable-retention harvest and 9–10 years after prescribed fire in Tenderfoot Creek Experimental Forest

BA, stand basal area; QMD, quadratic mean diameter; CBD, canopy bulk density; CBH, canopy base height. Understorey herb and woody load represent the aboveground plant biomass that contributes to surface fire spread

Treatment	Overstorey						Understorey	
	Stem density (trees ha^{-1})	BA (m^2 ha^{-1})	QMD (cm)	Top height (m)	CBD (kg m^{-3})	CBH (m)	Herb load (Mg ha^{-1})	Woody load (Mg ha^{-1})
Control	745 (94)	30.2 (1.8)	19.4 (0.3)	23.3 (0.9)	0.118 (0.011)	7.1 (0.7)	0.55 (0.04)	0.76 (0.16)
Aggregated: B	497 (88)	18.4 (3.2)	15.0 (0.9)	17.8 (1.7)	0.069 (0.014)	6.3 (0.6)	0.70 (0.04)	1.36 (0.22)
Aggregated: U	364 (150)	11.3 (2.8)	13.5 (0.8)	15.3 (1.7)	0.042 (0.009)	5.0 (0.7)	0.74 (0.03)	2.37 (0.50)
Dispersed: B	108 (32)	4.9 (1.1)	12.8 (2.8)	16.8 (3.4)	0.016 (0.003)	5.5 (0.8)	0.85 (0.01)	2.17 (0.17)
Dispersed: U	194 (25)	8.9 (1.0)	18.0 (0.6)	22.7 (1.1)	0.030 (0.003)	6.0 (0.7)	0.80 (0.02)	2.10 (0.48)

Potential fire behaviour

We simulated fire on all plots separately using BehavePlus and FOFEM, under each of the four moisture and wind scenarios (Table 2). Within each treatment × scenario combination, simulated fireline spread rates, flame lengths and intensities were heavily right-skewed. These were greatest in the dry–high wind scenario, where pooled intensities averaged 693 kW m^{-1} (range: 0.0–9686.0) from 1.57-m flame lengths (range: 0.00–5.28 m). In the moist–low wind scenario, intensities averaged 87 kW m^{-1} (range: 0.0–1696.0), given an average flame length of 0.31 m (range: 0.00–2.37 m).

Levene's variance homogeneity test on fireline intensity in the each scenario (pooled within treatments) had P values less than 0.001 ($F_{4,116}$ ranged from 5.82 to 8.65), indicating non-constant variance. Fireline intensity was modelled with treatment-level variances specified. Surface fireline intensity model residual standard errors were 354.8 for dry–low wind, 9991.9 for dry–high wind, 154.3 for moist–low wind, and 420.3 for moist–high wind. Squared predicted-to-observed correlations ranged from 0.14 for the moist–high-wind model to 0.21 for the dry–low-wind model; fixed-effects contributed 0.13 to 0.19 to the squared correlations. Unharvested stands are

Table 6. Estimated linear contrasts of potential fire characteristics from mixed-effects models 12 years after variable-retention harvest and 9–10 years after prescribed fire in Tenderfoot Creek Experimental Forest

Response variables were surface fireline intensity and crown fire transition ratio

	Dry–low wind		Dry–high wind		Moist–low wind		Moist–high wind	
Contrast	Estimate	*P* value	Estimate	*P* value	Estimate	*P* value	Estimate	*P* value
Surface fireline intensity (kW m^{-1})								
Grand mean – 0	265.4	<0.001	742.9	<0.001	93.7	<0.001	235.8	<0.001
Control – treated	−240.4	<0.001	−675.9	<0.001	−87.1	<0.001	−220.5	<0.001
Aggregated – dispersed	−114.6	0.126	−319.0	0.166	−39.3	0.251	−103.2	0.284
Burned – unburned	−270.4	<0.001	−784.0	0.001	−121.5	<0.001	−313.0	0.001
Treatment interaction	29.8	0.689	101.7	0.658	33.3	0.330	98.2	0.308
Transition ratio								
Grand mean – 0	0.119	<0.001	0.327	<0.001	0.053	<0.001	0.135	<0.001
Control – treated	−0.089	<0.001	−0.243	<0.001	−0.032	0.031	−0.085	0.120
Aggregated – dispersed	−0.047	0.201	−0.122	0.252	−0.015	0.398	−0.046	0.445
Burned – unburned	−0.124	0.001	−0.378	<0.001	−0.068	<0.001	−0.176	0.004
Treatment interaction	−0.003	0.942	0.007	0.951	0.006	0.749	0.017	0.770

predicted to have significantly lower mean fireline intensities than harvested stands in all scenarios (Table 6). There was insufficient statistical evidence to identify a difference between mean predicted fireline intensities among the cutting patterns in any scenario. Predicted fireline intensities in unburned stands are significantly greater than in burned stands.

Where residual overstorey trees were present on plots, the calculated critical flame lengths to ignite crowns were similar across plots (Fig. 3*a*). The median critical flame length in the control stands was 3.46 m (range: 1.52–4.67 m). Medians ranged from 2.74 to 3.51 m in the treated stands; the minimum and maximum critical flame lengths, averaged across treatments, were 1.71 and 4.64 m respectively. Visual inspection of within-treatment distributions suggests medians were slightly lower in the unburned treatments. Much more variability was exhibited among treatments in the critical crown rate of spread (Fig. 3*b*). Critical rates of spread in the untreated units had the lowest median (0.34 m s^{-1}) and smallest range (0.23–1.20 m s^{-1}). Medians for the burned and unburned aggregated treatments were 1.6 and 3.6 times greater than that of the control respectively. In the dispersed retention units, medians were 7.7 and 4.9 times greater than the control. Maximum critical rate of spread was limited to 3.0 m s^{-1} because BehavePlus's minimum input value for canopy bulk density is 0.016 kg m^{-3}. Plots with zero residual overstorey represented a minimum of 6% (in the unburned dispersed) and a maximum of 35% (burned dispersed) of plots measured within treatments. In these 'no-tree' plots, critical flame length and critical crown rate of spread could not be calculated as there were no overstorey trees to ignite.

In the dry–high-wind scenario, 'conditional' crown fire (per Scott and Reinhardt 2001) was predicted on 81.5% of the plots in untreated stands. In contrast, 34.3 and 16.7% of aggregated retention plots (burned and unburned respectively) were predicted to have conditional crown fire. No plots in the dispersed treatment were predicted to have conditional crown fire. No active crown fire was predicted. Proportion of plots predicted to have conditional crown fire in the moist–high-wind scenario were 73–83% lower than the dry scenario, and no conditional crown fire was predicted for the low-wind scenarios.

We modelled crown fire transition ratio as a quantitative measure of fire ascension into crowns. Levene's variance homogeneity test on transition ratio (pooled within treatments) had a *P* value of less than 0.01 ($F_{4,116}$ ranged from 3.54 to 7.98) for all but the wet–high-wind scenario. Thus, even with a median-centred test, there is strong evidence that variability in transition ratio is not constant across treatment groups. Like the surface fireline intensity models, we modelled transition ratio with treatment-level variances specified. Harvested stands were predicted to have a greater mean transition ratio than control stands in all but the moist–high-wind scenario (Table 6). There appeared to be no effect of retention pattern on transition ratio, but unburned stands had greater mean ratios than burned stands, regardless of scenario.

In a plot of transition ratio (Fig. 4), we observe with greater detail the relative susceptibility of plots to crown fire initiation (torching) in the dry–high-wind scenario. Fig. 4 illustrates more unstable fire behaviour (points above 1.0 on the *y* axis) is associated with low overstorey densities, and also highlights the variability in transition ratio within and across treatments. The control and burned aggregated plots are most tightly clustered in a low susceptibility range (medians = 0.02 and 0.03, and third quartiles = 0.05 and 0.18 respectively), although both treatments still result in some torching. Plots in burned and unburned treatments exhibited 1.9 and 13.5 times greater variance from zero than control plots respectively. The greatest transition ratios across all treatment levels tended to be to the left of the maximum overstorey threshold retained by the dispersed cutting method, i.e. 600 trees ha^{-1}. Thus, even though dense clumps in the control and aggregated treatments exhibited the greatest canopy bulk densities (CBDs) and lowest crowning indices, predicted surface fireline intensities were much lower than the crown fire initiation thresholds in clumps with at least 600 trees ha^{-1}.

We characterised post-frontal burning by generating heat release response profiles (Fig. 5). The ratio of variance of heat

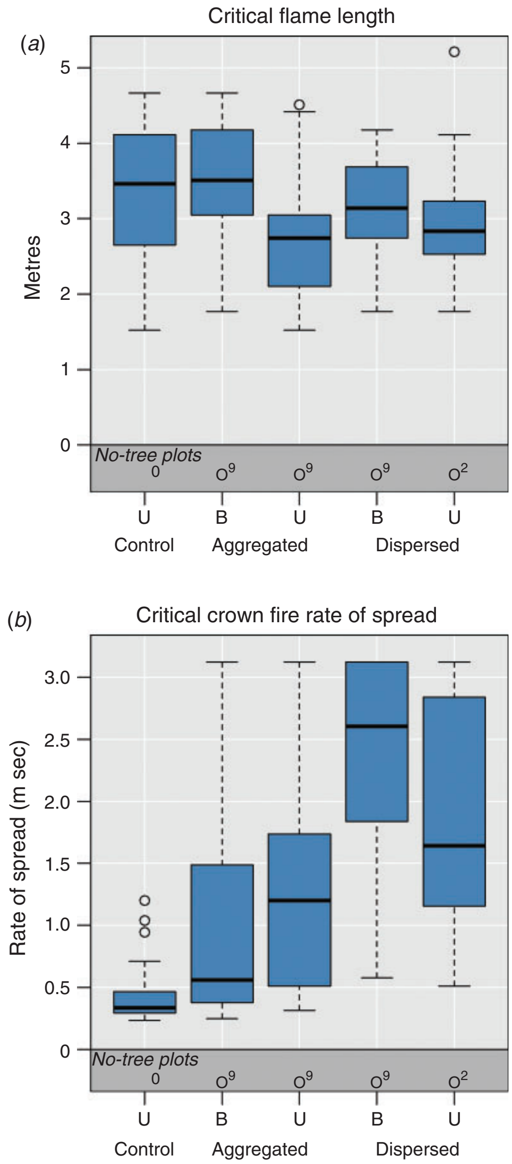

Fig. 3. Critical flame length to initiate crown fire (*a*), and critical crown fire rate of spread (*b*), 12 years after variable-retention harvest and 9–10 years after prescribed fire in Tenderfoot Creek Experimental Forest. Critical flame length is defined as the surface fire flame length necessary for fire to transition into tree crowns. Critical crown fire rate of spread is the rate necessary for fire to perpetuate in the canopy.

release from $t = 20$ to 2 min was 0.19 in control, 0.10 and 0.09 in aggregated burned and unburned respectively, and 0.13 and 0.11 in dispersed burned and unburned plots respectively. The ratio of median heat release at these times showcased similar relative values. Though heat release medians and variation tended to decay less rapidly in controls than treated units, median heat release values in controls were 2.4 to 4.3 times lower than treated units at $t = 2$ min. Median biomass consumption associated with the heat release curves was greatest in the dispersed burned treatment (61.7 Mg ha^{-1}) and lowest in the aggregated burned treatment (39.1 Mg ha^{-1}).

Discussion

Control stands had lower 1-h loading, but quite similar 10- and 100-h loading to the average condition identified by a study of four 'typical' cool, moist LP sites across Idaho, Montana and Wyoming (Brown and Bevins 1986; Table 3). Fine fuel (1-, 10- and 100-h) loadings generated by FCCS for the typical TCEF stand condition and those presented by Baker (2009) (generated by FCCS for a regional LP stand condition) were higher than our study site; 1000-h and litter and duff loads were greater at our site than either set of FCCS-generated values. These may conflict because FCCS values apply to a broader ecoregion (stretching from northern Idaho down to Colorado and New Mexico) than typified by the present study's site or the northern Rocky Mountain stands characterised in Brown and Bevins (1986).

In addition to addressing the effects of wind and dead activity fuels on potential fire behaviour, our study incorporates natural regeneration loads that resulted from treatment. However, we did not measure height of advance regeneration, and therefore potential fire behaviour in control stands may be underestimated. Our calculations indicated that some surface fuelbeds (<2 m) were more influenced by seedling biomass than by DWD, live herbaceous load or shrub load (compare Table 5 with published live woody loads in Anderson (1982), Scott and Burgan (2005)). Incorporating seedling-based fuel loads requires customisation of fire behaviour FMs but is necessary for a comprehensive evaluation of silvicultural or fuels-reduction treatments on potential fire behaviour.

Application of the present study's VRH and burn treatments in other LP forests may result in similar fuels dynamics, but potential fire behaviour may be quite different from these predictions. For instance, a stand representative of FCCS-identified fuelbed characteristics (Table 3) will result in more rapid predicted surface fire spread and unstable behaviour than presented predictions, owing to increased 1- and 10-h fine surface fuels. Care must be taken in inference and extrapolation of the potential fire behaviour predictions because this study's fire behaviour FMs have not been field-validated.

Directly modelling fire effects (i.e. tree mortality) was beyond the scope of our study. Our analytical framework assumes that crown fire initiation and spread are the ultimate concern for the manager, although we present heat release and biomass consumption results for better characterisation of fire behaviour. Such an additional analysis would be useful given sensitivity of trees in our study site to even a low-intensity fire.

Treated vs untreated

The tests on estimated contrasts in this study revealed first that there was no difference in total dead surface fuel loads between treated and untreated stands 12 years after harvest (Table 4). This conclusion suggests that activity fuels from harvesting and burning were no different than adjacent natural fuelbed aggradations. This is at least partially due to the study's harvest and burn prescriptions, which aimed to minimise activity residues. By whole-tree yarding to a centralised landing, fuel from non-merchantable materials such as tree branches and tops did not overload the surface fuel complex as a typical cut-to-length operation might do.

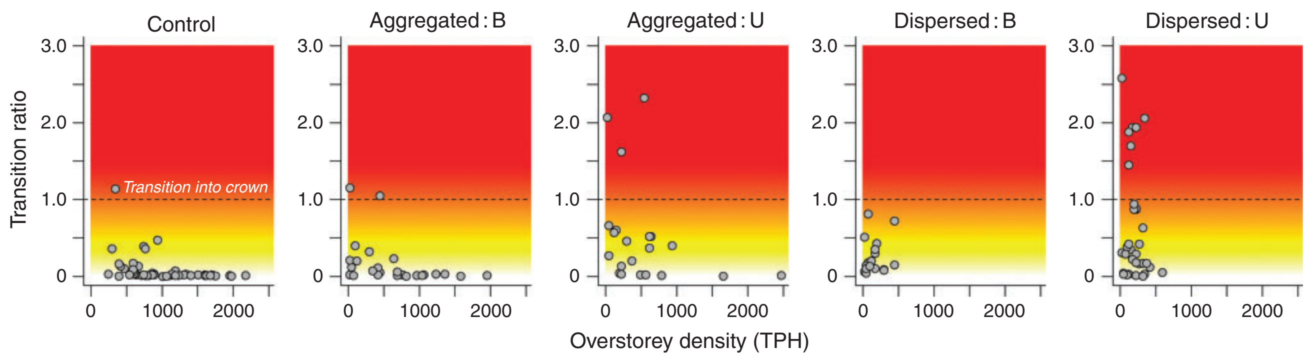

Fig. 4. Transition ratios (predicted to critical surface fire flame length needed to ignite overstorey crowns) in the dry–high-wind scenario (see Table 2), 12 years after variable-retention harvest and 9–10 years after prescribed fire in Tenderfoot Creek Experimental Forest. Points represent plot-level ratios from four experimental units.

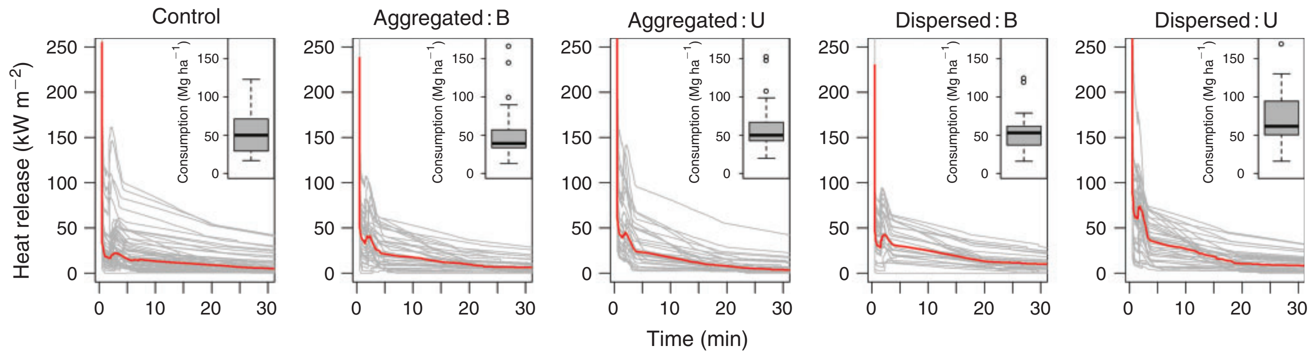

Fig. 5. Predicted heat-release profile (truncated) in the dry–high-wind scenario (see Table 2), 12 years after variable-retention harvest and 9–10 years after prescribed fire in Tenderfoot Creek Experimental Forest. Light (grey) lines represent plot-level responses from four experimental units. Dark (red) lines represent median heat release from pooled responses. Inset box-and-whisker plot represents biomass consumed from predicted flaming and smouldering from pooled responses. (For colour figure, see online version available at http://www.publish.csiro.au/nid/17.htm.)

Second, testing revealed that the annual change in 10- and 100-h fuel load components differs because of treatment (only weak evidence for 1-h fuels). Treated stand ΔAFLs were 22.5 and 1.8 times less than the untreated stands, for respective 10- and 100-h fuels. This suggests that initial activity fuels may have been slightly higher among the treated stands, but accumulation rate has decreased owing to overstorey removal. As surface fire spread is predominately influenced by 1- and 10-h timelag-class fuels, dead fuel loadings would have been conducive to carrying a surface fire immediately post treatment. Twelve years later, dead surface fuel connectivity has been influenced by the reduction of fine fuels. In fact, we observed several plots where either 1- or 10-h fuels were not found (9.25% of plots), which will continue to hinder surface fire spread where mature trees were removed. Also, fine woody debris (i.e. less than 7.62 cm) decomposition rate may have increased owing to particle fragmentation and forest floor insolation. Increased decomposition would suggest that the post-treatment environment increased microorganism activity on the forest floor. This hypothesis addresses the reduction of small woody fuel loads, yet historical chronosequence and process-based experiments arrive at contrasting conclusions regarding post-harvest surface fuel decay (Yanai *et al.* 2003). Regardless of the mechanism, these rates may continue until the regenerating cohort enters into a crown competition growth phase.

Modelled fire behaviour confirmed that potential surface fire flaming front intensities are influenced by the treatment at TCEF, particularly in low moisture conditions (Table 6, Fig. 4). Model results suggest greater fireline intensities in treated stands, which is consistent with other post-treatment fire behaviour studies in the western USA (e.g. Agee and Lolley 2006). This result was expected because of the increased live surface fuel load and within-stand wind penetration after partial overstorey removal. BehavePlus predicted that 'conditional' crown fire was possible in each of the stand types given 40-km h^{-1} wind scenarios, but most prevalent in untreated stands. Furthermore, median values indicate that treatments raised critical crown fire rate of spread overall. These results imply that a variety of LP stand configurations support sufficient canopy bulk density to carry crown fire given abnormally strong winds, but the VRH treatments evaluated can play a role in reducing that probability. However, this is further complicated by surface fire behaviour because the relative potential for crown fire initiation (transition ratio) increased by treatment in all four weather scenarios (Table 6). We acknowledge there are trade-offs between reduced potential for crown fire spread and increased potential for canopy ignition, both of which are largely driven by wind dynamics. Considering fine-scale resolution of intrastand wind conditions may be very useful to increase stand resistance to crown fire in the treatment design phase.

The heat release profiles we generated highlight the wide range of variability of post-frontal burning within and among treatments. We identified two key differences in heat profiles and associated consumed biomass between treatments and controls. First, the median of control plots decayed more rapidly than medians in the treated units. The median heat release in the control remained below 23 kW m^{-2} shortly after 2 min, whereas the same heat flux threshold was reached in ~4 min in aggregated and 11 min in dispersed units. Second, biomass consumption medians were more or less similar across treatments, but the few plots that approached or surpassed 150 Mg ha^{-1} of consumption were in the treated units. These plots reflect greater stockpiles of large woody debris that can profoundly impact subsurface heating. Our predictions highlight that although quantity of biomass consumption may vary only slightly, differences in the quality (e.g. time-lag class) of consumed materials may result in more adverse fire effects from the post-flaming front in treated units, particularly in the dispersed retention.

Burned vs unburned

Despite the seemingly detrimental differences in ΔAFL rates due to prescribed fire, 12-year post-treatment total dead surface fuel loads were generally greater in unburned treatments. Fine woody debris (1-, 10- and 100-h fuel classes) were highly influenced by burning (Table 4). Burn treatments resulted in 23 to 35% lower loads in these classes, but treatment interaction suggests burn effect was greater in the dispersed retention treatment for 10-h fuels. Although burning resulted in lower 1- and 10-h loads after 12 years, ΔAFLs in unburned stands were 48 and 46% lower than burned stands. It is clear that burning plays a very influential role in the immediate removal of fine woody debris, but delayed recruitment of fuels from fire-killed trees added fuels to this pool, reducing the effect that the mechanical treatment had in increasing decomposition rates. Burning was also associated with lower current litter and duff loads (22% less than unburned stands), but the rate of change over the measurement period was not significantly different from zero (Table 4). As for the largest fuel class, current burned and unburned 1000-h fuel loads were no different. Recruitment rate (ΔAFL) of 1000-h fuels was notably greater in burned than unburned stands, however. Recruitment of 1000-h fuels was greater in burned stands because of fire-induced tree mortality and subsequent translocation of fuels to the surface fuel complex. It is likely that tree mortality was driven by both first-order and second-order fire effects, but we were unable to quantify the relative rates of occurrence in this study. See Hardy *et al.* (2006) and Hood *et al.* (2012) for further assessment of fire-induced mortality in the study area.

In review, fuels less than 2.54 cm in diameter (1- and 10-h classes) and fuels greater than 7.62 cm (1000-h class) tended to stockpile more rapidly after burning. These results suggest that the structural benefit of fuels reduction in burn treatments was curtailed by post-treatment recruitment of woody fuels from the fire-damaged stand. Yet current fine woody debris loading in burned stands is still less than that of unburned stands. If prescribed burn severity was greater than we observed, then the recruitment of fine and large woody debris might have profound impacts on future fire effects. The burning prescription for these stands was for low-intensity fire, but the applied fire was more intense than anticipated, resulting in greater overstorey mortality and subsequent fuel accumulation. Future surface fire in these stands may again result in higher fire intensity than expected, but also greater soil heating and overstorey mortality because of large fuel loads from past mortality.

Our models confirmed that unburned-stand surface fireline intensity would be greater than in burned stands (Table 6) because treatments were designed to minimise post-treatment surface fuel loading. This supports that the burn treatment adequately decreases surface fireline intensity to reduce transition from surface to crown fire. We note that some of the difference in transition ratio due to burning treatment may be due to the fact that stands were burned 2–3 years after harvest, thus setting back the development of natural regeneration fuel loads. More conditional crown fire was predicted in burn treatments, but we believe this may be driven by tree density more than the burning treatment.

Aggregated vs dispersed

Only 1- and 1000-h fuels differed in the current surface fuel profile by retention pattern. We observed 21% lower 1-h and 39% greater 1000-h fuel loads in dispersed treatments. Current 1000-h load was high in dispersed stands because of a windthrow event immediately after harvesting and before sampling (Hood *et al.* 2012). Clump structures in the aggregated treatment drastically improved stem stability, as windthrow in these treatments was limited to clump edges. We did not observe significantly greater recruitment by retention pattern although we expected it. More trees in the dispersed treatment were directly exposed to prescribed fire (which influenced mortality), whereas interiors of clumps in the aggregated treatment had poor fire coverage because of moisture conditions (Fig. 6). Nevertheless, 1000-h fuel recruitment was similar between treatments because when fire did kill aggregated trees, it killed many of them.

Our analysis partly elucidated potential fire behaviour differences between retention patterns. Although contrast tests between retention patterns revealed no differences in mean effects, the Levene's test of variance homogeneity indicates greater variability in aggregated treatments. This emphasises that predicted fire behaviour based on averaged data from pooled plots masks important treatment differences identified among plots (see also Harrington *et al.* 2007), particularly where treatments were designed for structural irregularity. Aggregated treatment stands were defined by clumps and openings. Interiors of the residual tree clumps tended to have predicted fire behaviour akin to the untreated controls, i.e. low surface fire spread rate and low transition ratio. Openings where all overstorey trees were removed had the greatest surface fire spread rate and flame length owing to increased open wind speed. Clump edges were predicted to have fire behaviour most similar to stands in the dispersed treatment (Fig. 6). Where stands are designed for structural diversity, measures of central tendency (mean, median) of stand condition are clearly insufficient to assess the scope of potential fire behaviour. Greater resolution of within-stand variability and appropriate replication will aid the development

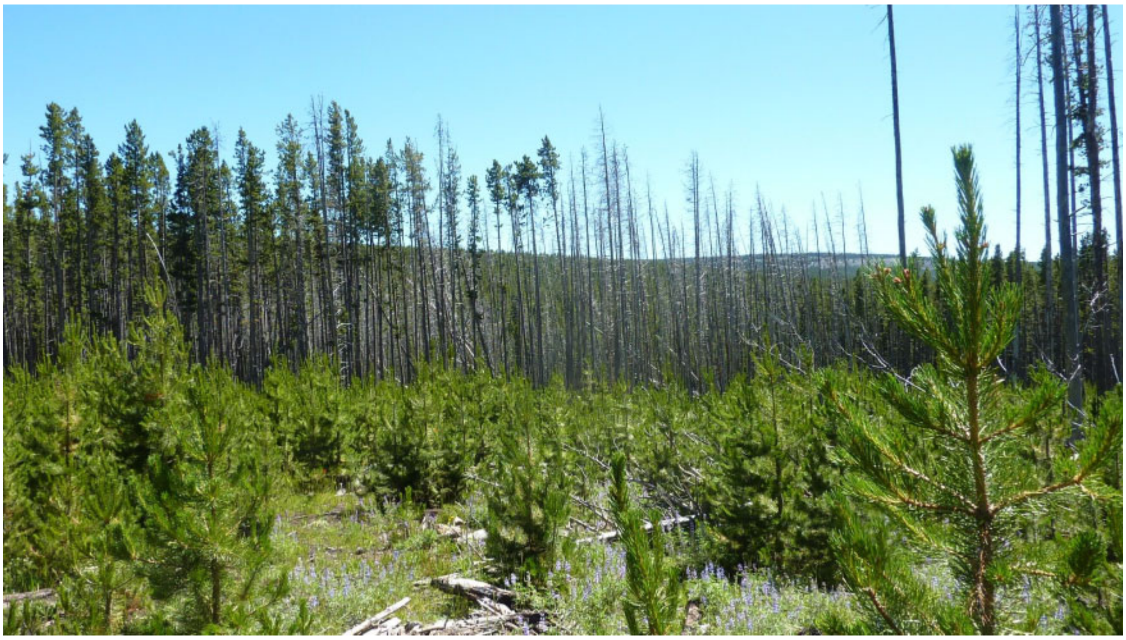

Fig. 6. Burned aggregated retention unit in the Tenderfoot Creek Experimental Forest, 12 years after prescribed fire. Retention clump (beyond the regeneration in the open foreground) shows fire-caused overstorey mortality on the downslope edge of the group. (Photo: C. Keyes)

of within-stand potential fire behaviour distributions after aggregated VRHs.

Structural complexity and disturbance

We found that nearly all untreated plots, less than half of aggregated treatment plots, and very few of dispersed treatment plots had low critical crown rate of spread thresholds (<0.75 m s^{-1}; Fig. 3). As removal of 50% of the stand basal area was the treatment prescription, it is no surprise that many of the plots in treated stands had low to no CBD values, and thus high critical crown rates of spread. Not evident in Fig. 3 is the spatial discontinuity inherent among clumps of trees within aggregated treatment stands. Aggregated retention *stands* are likely more resistant to crown fire than Fig. 3 indicates because the spatially discontinuous pattern of the retention layout reduces crown fire contagion. At this stage in stand development, aggregated clump disconnectedness is a major driver of structural resilience to fire.

Homogeneous, even-aged LP forests are often highly susceptible to severe and widespread disturbance events, but structural diversity and resilience can be improved by creating multi-aged stands (Safranyik and Carroll 2006; Axelson *et al.* 2010). At TCEF, VRH techniques increased stand complexity by reducing overstorey densities and promoting a new cohort. These treated stands also reduced the amount of forested area susceptible to mortality caused by mountain pine beetle (*Dendroctonus pondersae* Hopkins), because these insects generally cannot amplify populations to epidemic levels within LP trees less than 20 cm DBH or in stands younger than 80 years old (Safranyik and Carroll 2006; Axelson *et al.* 2010). Furthermore, the structural complexity created likely increased resistance to beetle attack in the retained overstorey portions of the aggregated retention stands. This residual structure has been shown beneficial in a forest patch-cutting experiment in Wyoming, where tree mortality caused by biotic agents (including mountain pine beetle) was reduced within overstorey retention groups similar to those created at TCEF (Johnson *et al.* 2014).

We also expect improved resistance to some biotic and abiotic disturbances in the dispersed retention stands, because their structure is similar to shaded fuel-breaks designed to hinder stand-replacing crown fires (Agee *et al.* 2000) and thinning treatments implemented to reduce stand susceptibility to mountain pine beetle attack (Whitehead and Russo 2005; Whitehead *et al.* 2007). However, dispersed retention treatments may exacerbate wind- and snow-related tree mortality, as observed in partial cutting of old-growth LP stands in the central Rocky Mountains (Alexander 1966). In general, flexible saplings are resilient to windstorms and heavy snow loads, whereas windthrow can be common in mature trees (Johnson 1987). Substantial windthrow was observed in multiple dispersed retention plots at the current study site following harvesting (Hood *et al.* 2012). As such, aggregated retention may be preferred over the dispersed stand structure when converting to multiple cohorts in locations prone to high wind speeds or snow damage. This is especially true in dense, previously unthinned stands with high height-to-diameter ratios.

We suggest that the VRH treatments implemented at TCEF can effectively improve forest heterogeneity in such a manner as to mitigate stand-level susceptibility to severe biotic and abiotic disturbances. However, these treatments increase within-stand variability in surface fireline intensity and crown fire initiation ratio after 12 years. We believe it is critical to acknowledge the trade-offs in overstorey retention structure (i.e. for stand growth and disturbance susceptibility) when using VRH to create multi-aged stands.

Conclusion

This study provides much-needed insight into the change in fuel loadings for 12-year fuel dynamics after variable-retention harvests. Our results suggest that operational efforts to reduce fuel loading were countered by post-treatment mortality. We observed lower accumulation of fine woody debris due to treatment, but burning greatly increased large woody debris accumulation. Our potential fire analysis shows that that averaged fuel and fire behaviour metrics are insufficient to characterise the scope of potential fire behaviour in highly irregular stands. Treatments increased likelihood of crown ignition because of increased live surface fuels and subcanopy wind penetration. However, critical crown fire spread rates generally indicated higher wind speeds needed in treated vs untreated to facilitate crown fire spread.

Acknowledgements

This was a study of the Applied Forest Management Program at the University of Montana, a research and demonstration unit of the Montana Forest and Conservation Experiment Station. Support for this study came from the USDA Forest Service, Forest Health Protection, Special Technology Development Program (grant R1–2011–03). The study was made possible with contributions by Forest Health Protection, Northern Region, USDA Forest Service, and the Rocky Mountain Research Station (RMRS), USDA Forest Service. We are indebted to Tom Perry (University of Montana College of Forestry and Conservation), who provided integral technical leadership; Helen Smith (RMRS), gracious provider of historic data; and Duncan Lutes (RMRS), who administered helpful technical guidance.

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