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Restoring forest resilience: From reference spatial patterns to silvicultural prescriptions and monitoring

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ABSTRACT

Stand-level spatial pattern influences key aspects of resilience and ecosystem function such as disturbance behavior, regeneration, snow retention, and habitat quality in frequent-fire pine and mixed-conifer forests. Reference sites, from both pre-settlement era reconstructions and contemporary forests with active fire regimes, indicate that frequent-fire forests are complex mosaics of individual trees, tree clumps, and openings. There is a broad scientific consensus that restoration treatments should seek to restore this mosaic pattern in order to restore resilience and maintain ecosystem function. Yet, methods to explicitly incorporate spatial reference information into restoration treatments are not widely used. In addition, targets from reference conditions must be critically evaluated in light of climate change. We used a spatial clump identification algorithm to quantify reference patterns based on a specified inter-tree distance that defines when trees form clumps. We used climatic water balance parameters, downscaled climate projections, and plant associations to assess our historical reference sites in the context of projected future climate and identify climate analog reference conditions. Spatial reference information was incorporated into a novel approach to prescription development, tree marking, and monitoring based on viewing stand structure and pattern in terms of individuals, clumps, and openings (ICO) in a mixed-conifer forest restoration case study. We compared the results from the ICO approach with simulations of traditional basal area and spacing-based thinning prescriptions in terms of agreement with reference conditions and functional aspects of resilience. The ICO method resulted in a distribution of tree clumps and openings within the range of reference patterns, while the basal area and spacing approaches resulted in uniform patterns inconsistent with known reference conditions. Susceptibility to insect mortality was lower in basal area and spacing prescriptions, but openings and corresponding opportunities for regeneration and *in situ* climate adaptation were fewer. Operationally, the method struck a balance between providing clear targets for spatial pattern directly linked to reference conditions, sufficient flexibility to achieve other restoration objectives, and implementation efficiency. The need to track pattern targets during implementation and provide immediate feedback to marking crews was a key lesson. The ICO method, especially when used in combination with climate analog reference targets, offers a practical approach to restoring spatial patterns that are likely to enhance resilience and climate adaptation.

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1. Introduction

Increasing ecological resilience has become a central objective in management of public forestlands due to the combined effects of past management and projected climate change (Baron et al., 2009; Joyce et al., 2009). Ecological resilience (hereafter, resilience) includes the capacity to persist through and re-organize after disturbance, adapt to shifting environmental conditions, and maintain basic ecosystem structure and function over time (Walker et al., 2004). There is increasing evidence that spatial heterogeneity at multiple scales, in addition to forest structure and composition, is a critical component of ecosystem resilience (Levin, 1998; Moritz et al., 2011; North et al., 2009; Stephens et al., 2008). General

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frameworks that incorporate ecologically-based guidelines for spatial pattern have been developed for some forest ecosystems (Carey, 2003; Franklin and Johnson, 2012; Hessburg et al., 2004, 1999; Franklin et al., 2007; Mitchell et al., 2006). A major remaining task is the identification of spatial pattern targets for specific ecosystems that are empirically linked to resilience, climate adaptation, and desired ecological functions (Puettmann et al., 2009). A related challenge is translating such targets into operationally-efficient prescriptions and monitoring protocols (North and Sherlock, 2012; O'Hara et al., 2012).

In frequent-fire pine and mixed-conifer forests in western North America (hereafter, *dry forests*), pre-settlement era forests are commonly used as reference systems to inform treatment targets (Larson and Churchill, 2012). Contemporary dry forests with minimally altered or restored fire regimes are increasingly being used as reference sites as well (Lydersen and North, 2012; Stephens et al., 2008; Taylor, 2010). At the stand level, tree patterns in these forests are commonly characterized by an uneven-aged mosaic of individual trees, clumps ranging from 2 to more than 20 trees, and openings (Kaufmann et al., 2007; Larson and Churchill, 2012). Such mosaics persisted for centuries in a dynamic system of fine-scale, gap-phase replacement driven primarily by frequent fire and insect mortality (Agee, 1993; Cooper, 1960). Patch size and within-patch heterogeneity varied temporally at individual sites, and across different biophysical environments (Kaufmann et al., 2007). Infrequent moderate to high-severity disturbances did occasionally reset these stand-level patterns as well (Arno et al., 1995; Hessburg et al., 2007). Here, we refer to stands as patches embedded in a hierarchy of landscape organization; smaller than sub-watersheds and larger than the largest tree clumps and openings (Urban et al., 1987). Historical patch size distributions of stands in dry forests ranged from 1 to 10^4 ha (Perry et al., 2011), different from the traditional view of stands as 10–50 ha management units.

The fine-scale mosaic pattern is thought to be a key factor underpinning the resilience of dry forest ecosystems (Allen et al., 2002; Binkley et al., 2007; Stephens et al., 2010). Irregular tree patterns, large openings, and resulting variation in surface fuels inhibit the spread of crown fire and perpetuate variable post-fire patterns (Beaty and Taylor, 2007; Pimont et al., 2011; Stephens et al., 2008), analogous to strategic placement of fuel treatments at larger spatial scales (Finney et al., 2007). Heterogeneous stand structures impede the buildup of epidemic insect outbreaks by disrupting pheromone plumes and breaking up the continuity of susceptible species, as well as age and size classes (Fettig et al., 2007).

Similarly, openings create barriers to the spread of dwarf mistletoes and fungal pathogens (Goheen and Hansen, 1993; Hawksworth et al., 1996; Shaw et al., 2005). Likewise, openings and frequent disturbances facilitate periodic tree regeneration in dry forests (Boyden et al., 2005; Sánchez Meador et al., 2009), which is thought to be partly responsible for high levels of local genetic diversity of trees (Linhart et al., 1981; Hamrick et al., 1989). Snow retention is highest where canopy openings are large enough reduce canopy interception, but small enough to be shaded and protected from wind (Varhola et al., 2010). In addition, the contrasting light, moisture, and soil nutrient environments in heterogeneous stands increase understory plant abundance and diversity (Dodson et al., 2008; Moore et al., 2006).

There is a growing scientific consensus that to increase resilience, mechanical and prescribed-fire treatments should seek to restore the range of mosaic patterns found in reference stands (Allen et al., 2002; Franklin et al., 2008; Hessburg et al., 2005; Moore et al., 1999; North et al., 2009; Perry et al., 2011), in addition to retaining large and old fire-tolerant trees and following other resilience principles (Table 1). Widespread adoption of prescription approaches based on spatial reference information has been slow, however, despite numerous operational-level research studies (e.g. Graham et al., 2007; Knapp et al., 2012; Lynch et al., 2000; USFS, 2008; Waltz et al., 2003).

The fundamental challenge facing managers is the mismatch between the grain and variation of pattern found in dry forests and the tools commonly used to quantify and manage them. Most silvicultural methods are based on stand-average density metrics originally designed to create homogenous stands (Puettmann et al., 2009). Modifying these approaches to manage for within-stand spatial variability requires re-conceptualizing "stands" as mosaics of variably sized canopy patches (Puettmann et al., 2009). This shift—and the associated changes in mensuration tools, operational methods, and contractual mechanisms—can be initially complex and time consuming (Knapp et al., 2012; North and Sherlock, 2012). Thus, many managers continue to employ stand-average basal area or spacing-based approaches (e.g. Powell, 2010). There is concern that these approaches create evenly spaced stands inconsistent with ecologically important fine-scale processes, and may have unintended negative effects to wildlife and disturbance behavior. The tradeoffs between the effort of transitioning to new approaches and gains in ecological functionality and resilience are unknown, however.

Projected changes in climate and related shifts in disturbance behavior present an additional challenge to the use of historical

Table 1
Comparison of three stand-level treatment approaches in dry-mixed conifer forests: fuels treatments and hazard reduction, restoration of pre-settlement or current reference conditions, and climate adaptation/resilience management. Treatment targets for different strategies may vary considerably between the three different approaches.

Recommended strategies	Fuels treatments[a]	Restoration[b]	Resilience/ adaptation[c]
1. Reduce surface and ladder fuels; increase crown base height	x	x	x
2. Reduce and maintain lower tree densities; decrease crown bulk density.	x	x	x
3. Increase relative composition of fire and drought tolerant species	x	x	x
4. Increase mean tree diameter and individual tree vigor by generally retaining larger trees with healthy crowns	x	x	x
5. Conserve existing species and genetic diversity, including pre-settlement trees		x	x
6. Restore horizontal spatial heterogeneity of forest structure, including openings where early-seral species can establish		x	x
7. Re-introduce fire to reduce fuel loads, stimulate understory species, and maintain desired fuel beds	x	x	x
8. Reduce and/or maintain appropriate levels of pathogens, insects, and other disturbances in order to create decadence, mortality, and interactions with fire that lead to regeneration of new tree cohorts and diverse understories		x	x
9. Replant desired native species, especially after high severity disturbances			x
10. Plant new genotypes and/or species			x
11. Monitor key processes such as mortality, regeneration, growth, fuel accumulation, and new species colonization to inform future management	x	x	x

[a] Agee and Skinner (2005), Graham et al. (2004), and Peterson et al. (2005).
[b] Allen et al. (2002), Covington et al. (1997) and Franklin and Johnson (2012).
[c] Chmura et al. (2011), Peterson et al. (2011), Spies et al. (2010) and Stephens et al. (2010). These include resistance, resilience, response, and realignment options.

reference conditions (Franklin et al., 1991; Peterson et al., 2011; Williams et al., 2007). Pre-settlement-era dry forests developed from pattern-process linkages that persisted through centuries of frequent disturbances and climatic fluctuation and thus serve as a general guide for increasing resilience (Fule, 2008; Keane et al., 2009; Moritz et al., 2011). Restoration approaches based on reference conditions follow almost all of the strategies recommended for climate adaptation (Table 1). Restoring historical spatial pattern, composition, and density for a specific site, however, may not ensure resilience. Targets derived from reference conditions must be critically evaluated, and potentially modified, based on projected future climates and ecological knowledge to provide operative targets for restoration (Keane et al., 2009; Spies et al., 2010; Stephens et al., 2010). Climate analog reference sites offer a constructive approach to evaluating and using historical reference conditions. A climate analog site is one that developed in a climate similar to the projected future climate of the treatment site (Gärnter et al., 2008). Climate analogs provide an empirical basis for integrating climate adaptation with ecosystem restoration and can provide ecologically based targets for resilience, response, or realignment adaptation strategies (Stephens et al., 2010).

In this paper we address two central challenges of dry forest restoration: managing for resilient spatial patterns consistent with reference sites, and the use reference conditions in conjunction with climate change adaptation. We employ a spatial clump identification algorithm to quantify reference patterns in terms of individual trees and clumps (Plotkin et al., 2002). We assess our historical reference sites in the context of projected future climate using climatic water balance parameters (annual Actual Evapotranspiration and Deficit), plant associations, and downscaled climate model outputs. We then introduce a novel method to incorporate reference spatial pattern targets into silvicultural prescriptions and tree marking guidelines, which we term the ICO method (for individuals, clumps, openings), in an operational case study. We evaluate the practical utility of the ICO method, and compare it with simulated basal area and spacing-based prescriptions in terms of alignment with reference conditions and effects on several functional aspects of resilience. We hypothesized that: (1) the pattern created using the ICO method would fall within the range of variation, or envelope, of past and climate analog reference patterns; and (2) basal area and (3) spacing-based prescriptions would be outside of these envelopes.

2. Methods

2.1. Background and study area

In 2008, managers from the Okanogan-Wenatchee National Forest (OWNF) selected a 30 ha harvest unit within the Wildcat Timber Sale to test stand-level implementation of the OWNF Restoration Strategy (Gaines et al., 2010). The project area is located within the Tieton River drainage in the eastern Cascade Mountains, 30 miles west of Yakima, Washington and immediately north of Rimrock Reservoir. The site lies on a south-facing aspect with slopes of 10–45%, at an elevation of 975–1060 m. Soils are derived from volcanic ash deposits layered over basalt colluvium (NRCS, 2009), and are deep (>150 cm), well drained, and moderately productive (100 year site index, 29 m) (Barret, 1978). Modeled precipitation from 1971–2000 averaged 1103 mm annually, falling mostly as snow between November and April (ClimateWNA, 2012). Mean January and July temperatures were −2.1 °C and 15.7 °C.

Forest structure, composition, and management history in the project area is representative of dry-forest biophysical environments common throughout the Eastern Cascades of Washington and Oregon. Tree species consist of ponderosa pine (*Pinus ponderosa*) and interior Douglas-fir (*Pseudotsuga menziesi* var. *glauca*) with a minor component of grand fir (*Abies grandis*). The dominant plant association is Douglas-fir/shiny-leaf spirea (*Spirea betulifolia*))/pinegrass (*Calamagrostis rubescens*) (Lillybridge et al., 1995). Members of the Yakama Nation extensively utilized the area prior to Euro-American settlement in the 1880s. Intensive grazing by sheep and cattle occurred from the 1880s until the early 1900s (USFS, 1998), followed by logging in the 1930s and 1970s that removed almost all large-diameter ponderosa pines and spaced out residual overstory trees. Fire exclusion and suppression began at the turn of the century and has continued to the present. Current forest structure is dominated by trees established between 1890 and 1930. A pre-settlement multi-age structure with the oldest trees greater than 250 years was inferred from field ring counts of stumps. Pre-settlement disturbance was likely dominated by low and mixed severity fire with a return interval of 5–20 years (USFS, 1998), as well as native insect and pathogenic disturbances (Hessburg et al., 1994).

2.2. Tree clump identification

To characterize within-stand tree patterns, we use a clump identification algorithm from Plotkin et al. (2002). The algorithm partitions a stem map of tree locations into clumps at a specified inter-tree distance (t), measured from tree center to tree center. Trees are members of the same clump if they are within distance t of at least one other tree in the clump. Trees with no neighbors within distance t are termed individuals. This method has been previously used to identify tree clumps in dry forests (Abella and Denton, 2009; Sánchez Meador et al., 2011) and to compare silvicultural treatments with patterns from reference old-growth forests (Larson and Churchill, 2008; Larson et al., 2012).

We elected not to use any edge correction method following Plotkin et al. (2002). Yamada and Rogerson (2003) explored the effects of edge correction methods and concluded that there is no advantage in using corrections when the goal is to describe an observed pattern. However, the use of edge correction is important when testing for departure from a null pattern with point pattern statistics. Sánchez Meador et al. (2011) developed an edge correction technique for the clump identification method, where clumps that have trees within distance t of the edge are removed. We implemented this correction for comparison.

2.3. Climate analog reference conditions

The pre-settlement era reference conditions used to guide prescription development were based on a 100 m × 100 m stem-map plot installed within the Wildcat project area as well as 32, 50 m × 100 m plots from Harrod et al. (1999), (hereafter, Harrod sites). Harrod et al. (1999) is the only spatially explicit reconstruction of pre-settlement conditions from the eastern Cascade Range of Washington. The Harrod sites reside approximately 85 km northeast of the harvest unit, and display similar biophysical conditions, forest types, and pre-settlement era disturbance regimes; however, approximately half of the 32 plots are in plant associations that are warmer and/or drier than the plant association in the Wildcat unit (Lillybridge et al., 1995).

Our rationale for using the two reference datasets was to compare the reference plot from the Wildcat site against a known range of variation of structure and pattern. This allowed us to determine that the Wildcat site was not an outlier and also compare its pattern and structure with warmer and drier sites. We followed the reconstruction protocols used by Harrod et al. (1999) for the Wildcat reference plot. All live trees, stumps, snags, and logs that were estimated to have originated before 1865 were

considered "historical" and mapped. The plot was subjectively located in an area with relatively homogenous slopes, aspects, and soil conditions that were typical of the project area, where stumps, snags, and logs from the pre-settlement forest were well preserved. The slope and aspect of the plot location was compared in GIS with the overall project area to ensure it was not anomalous.

To assess whether the Harrod locations were suitable climate analog reference sites, we compared current climate (1971–2000) from the Harrod sites to the projected climate of the Wildcat site for three future time periods (2011–2040, 2041–2070, 2071–2100). We used current and projected temperature and precipitation from the Climate WNA database (ClimateWNA, 2012). The Climate WNA methodology downscales projections from general circulation models (GCMs) by combining PRISM downscaling methods (Daly et al., 2008) for 800 m grid cells with local lapse rates to account for elevation differences (Wang et al., 2012). We used an ensemble of 15 general circulation models (GCMs) to capture the variability in GCM predictions and create an envelope of future climatic conditions (see Appendix for a list of the 15 GCMs). We chose the A1B and A2 emissions scenarios (IPCC, 2007) as they reflect the most likely scenarios given current trends (Rogelj et al., 2009).

To compare the climate regimes of the sites, we used temperature, precipitation, and soil water-holding capacity to calculate annual actual evapotranspiration (AET) and annual climatic water deficit (Deficit), the difference between potential evapotranspiration (PET) and AET. AET and Deficit have been shown to be good predictors of species presence/absence and growth rates (Littell et al., 2008; McKenzie et al., 2003; Stephenson, 1998, 1990); and have been used to project shifts in species distributions and disturbance regimes (Littell et al., 2010; Lutz et al., 2010; Shafer et al., 2001). We used a Thornthwaite-type model for PET (Lutz et al., 2010; Thornthwaite et al., 1957) and a soil water balance model from Dingman (2002) based on soil water-holding capacity in the top 150 cm of the soil profile (NRCS, 2009).

We added a multiplier to the PET equation for solar radiation similar to the approach used by Stephenson (1998) and Lutz et al. (2010) to account for differences introduced by aspect, slope, and latitude. We calculated solar gain at each Harrod plot center in ArcMAP v10 (ESRI, 2012). We then normalized PET using the ratio of solar gain of each plot to the average value for the study area, which was derived from a 90 m × 90 m grid of points distributed across the 6500 ha study area. Plant associations from Lillybridge et al. (1995) were used to assess the ecological significance of relative AET and Deficit and evaluate which of the Harrod sites and respective plant associations represented appropriate climate analogs for the Wildcat site. Plant associations were used as indicators of site potential in terms of productivity (site index), carrying capacity (SDI levels for full stocking), and species viability (whether sites can support Douglas-fir and grand fir); not as stationary communities (Daubenmire, 1976). After assessing the AET and Deficit of plant associations, we combined similar associations to more easily compare the current and projected climates of the Wildcat site against the current climate of Harrod sites. Plant associations were combined into four groups based on defining tree species, proximity of average AET and Deficit values, and overlap of ranges of Deficit values. The four groups were PIPO High, PSME High; PSME Moderate, and PSME Low.

2.4. Prescription development using the ICO method

Translation of the reference spatial patterns into prescription guidelines was done by using the Plotkin et al. (2002) clump algorithm to derive clump size distributions for the reference plots. This distribution summarizes the percent of trees arranged as individuals and in clumps of different sizes at a specified inter-tree distance (Table 2). We selected a single inter-tree distance threshold of 6 m to define tree clumps based on the observed distance at which most mature ponderosa pine trees (>120 years) in our study area display interlocking crowns and form patches of continuous canopy (Graham et al., 2007). Crown radius data from the US Forest Service Forest Inventory and Analysis database for similar plant associations on the OWNF were consistent with distances between 5 m and 7 m. The 6 m distance represented the 33rd percentile of crown diameter for ponderosa pine. Abella and Denton (2009) and Sánchez Meador et al. (2011) used a similar process to derive a clump distance and used 6 and 5.3–6.6 m, respectively. We then generated cumulative clump size distributions using the 6 m threshold to compare the Wildcat and Harrod sites and to assess differences in clump size distributions among plant associations. Visual inspection of these distributions revealed no major difference in pattern between the associations, and the distribution from the Wildcat reference site lay in the middle third of the Harrod distributions.

Table 2
Proportional clump size distribution for 1 ha Wildcat reference plot.

t	Cluster size (# of trees)									
	1	2	3	4	5	6	7	8	9	10
Percent of total trees										
1 m	96	4	0	0	0	0	0	0	0	0
2 m	67	21	12	0	0	0	0	0	0	0
3 m	60	17	6	0	0	0	0	17	0	0
4 m	58	8	6	8	0	0	0	0	19	0
5 m	52	12	6	8	0	0	0	0	0	21
6 m	*42*	*17*	*12*	*8*	*0*	*0*	*0*	*0*	*0*	*21*
7 m	35	17	19	8	0	0	0	0	0	21
8 m	35	17	19	8	0	0	0	0	0	21
9 m	33	17	19	0	10	0	0	0	0	21
10 m	31	17	19	0	0	12	0	0	0	21
Percent of total trees with edge correction										
6 m	32	18	9	12	0	0	0	0	0	29
Percent of total plot basal area										
6 m	51	17	11	4	0	0	0	0	0	17
Mean dbh of trees (cm)										
6 m	90	81	77	57	0	0	0	0	0	73

t is the inter-tree distance.
6 m values were selected to guide prescription development.

The next step in prescription development was to determine the average leave tree density and pattern to be approximated in tree marking. We chose 100 trees ha^{-1} (tph) based on the desired range of 50–75 tph in dry, old forest structure described in the OWNF restoration strategy (Gaines et al., 2010) and allowance for post-treatment mortality from prescribed fire, inter-tree competition, and ongoing insect and pathogen disturbance. We then multiplied the proportion of trees each clump size from the Wildcat reference pattern by the target of 100 tph to generate the target number of tph in each clump size.

To facilitate marking and tracking of clumps in the field, we grouped clump sizes together into three bins: individual trees (clump size = 1), 2–4 tree clumps, and 5+ tree clumps. We divided the target number of trees in each bin by the average number of trees for that bin (3 for 2–4 tree clumps and 6 for 5+ tree clumps) to derive the target number of clumps per hectare for each bin (Table 3). To ensure that large clumps would exist in the future, the target for 5+ tree clumps was increased by 30% because we expected more density-dependent bark beetle mortality in large clumps vs. individuals. Bin size was based on observed functional differences between clump sizes: for example, 5+ clumps typically contain "interior trees" that are more susceptible to competitive stress and insect related mortality (Olsen et al., 1996); smaller clumps do not. Moreover, understory shading and micro-climatic

Table 3
Derivation of prescription targets for clumps from proportional clump size distribution of reference plot at 6 m.

	Clump size (# of trees)		
	1	2–4	5+
Target trees per hectare is 100			
Percent of trees in clumps in reference plot (%)	42	37	21
Rx target for trees per ha	42	37	21
Rx target for clumps per ha	42	12	5[a]

[a] The target for 5+ tree clumps was increased by 30% to hedge against higher anticipated rates of mortality in large clumps vs. individual trees.

effects begin occurring in larger clumps (~5+ trees), which can affect understory development, wildlife use, and fire behavior (Dodd et al., 2006; Ma et al., 2010). Individual trees have faster growth rates and distinct crown architecture which can affect bark thickness, fire and insect resistance, resin response to wounding, and sensitivity to climatic fluctuation (Carnwath et al., 2012).

The purpose of the prescription was not to replicate the clump size distribution of the reference stand on every hectare but to promote a mosaic pattern of individual trees, clumps, and openings within the envelope of historic conditions. Consequently, the per hectare targets for individuals and clumps were not rigid targets but approximate averages to be obtained over the entire unit. The marking guidelines also included instructions to retain all trees >110 years old, generally thin from below, favor ponderosa pine, and leave trees with live crown ratios >40% (see Appendix for complete marking guidelines). The overall goal was to work with existing stand conditions to achieve the approximate individual tree and clump targets, while meeting the other criteria. We assumed that leaving over 50% of the target density in clumps would result in the desired range of opening sizes and thus did not provide explicit instructions to create openings. Leave tree marking was used with crews tallying the number of residual individuals and clumps by bin size. The results were tallied after the marking was completed, so no adjustments were made during marking. This treatment was called the ICO treatment (individuals, clumps, and openings).

2.5. Monitoring and simulated treatment alternatives

To monitor and evaluate the implementation of the prescription, we installed a 2 ha plot (141 m × 141 m) prior to harvest, in which all trees >15 cm dbh were mapped. The 15 cm cutoff was based on a commercial thinning minimum diameter threshold. We derived clump size distributions at 6 m and determined that too few large clumps were retained. This presented an opportunity for adaptive learning through simulation of an adaptive management marking scenario. In this scenario, we adapted the initial mark to better achieve the 5+ tree clump targets, and create larger openings that better reflected conditions of the reference plot. This simulated marking scenario was labeled the adaptive management (AM) treatment and was intended to be an example of how tallying clump totals during marking could have been used to make adjustments to the original ICO treatment.

We also simulated basal area (BA) and spacing-based (Space) prescriptions to compare our method with these more standard thinning approaches. The BA simulated marking scenario sought to achieve a uniform target basal area throughout the 2 ha monitoring plot, with a ±50% allowance for natural openings and denser areas. The target basal area was the same as that retained on the 2 ha plot in the actual marking. Variable radius plots were simulated on a 20 m × 20 m square grid within the 2 ha using a basal area factor of 4.59. The same leave tree criteria used in the ICO treatment, minus the clump targets, were followed on each plot until the basal area target was met. Ponderosa pine was selected for retention over Douglas-fir if dwarf mistletoe (*Arceuthobium douglasii*) infections were present, or if the pines were similar in diameter (within 5 cm dbh). The BA simulation was implemented in sequential strips within the stem map, similar to how a marking crew ordinarily moves through a stand.

The third simulated marking scenario (Space) followed a spacing-based prescription where all trees within 6.7 m of a larger tree were cut, while retaining all ponderosa pine >50 cm dbh. The distance of 6.7 m was calculated to achieve the same tph as the ICO treatment. This approach, called Designation by Description (D × D), is used in some contemporary dry, mixed conifer forests in the Pacific Northwest and northern Rockies (Wynsma and Keyes, 2010). The BA and Space treatments resulted in post-treatment basal areas (16.9 and 17.5 $m^2\ ha^{-1}$) and stocking levels (30% and 31% of maximum SDI) similar to other dry forest fuel reduction and restoration treatments conducted in the region (Larson et al., 2012; Powell, 1999; Prichard et al., 2010; Youngblood et al., 2006).

2.6. Comparing treatment alternatives and reference plots

We used a combination of global and local point pattern analysis methods to provide a robust evaluation (*sensu* Perry et al., 2006) of whether the field ICO treatment and the three simulated treatments fell within the envelope of spatial patterns found in the Wildcat and Harrod reference plots. The pair correlation function, $g(t)$, was selected as the primary global pattern statistic (Wiegand and Moloney, 2004). The $g(t)$ function is a non-cumulative form of Ripley's $K(t)$ (Ripley, 1988), and is calculated by taking the derivative of the kernel smoothed $K(t)$ distribution (Wiegand and Moloney, 2004). The $g(t)$ function avoids the problem of "cumulative effects" associated with $K(t)$ or $L(t)$, where patterns at short t values can lead to incorrect assessments of clumping, randomness, or inhibition at longer t values (Fortin and Dale, 2005; Perry et al., 2006; Illian et al., 2008). We generated reference pattern envelopes from the Harrod plots using the $g(t)$ function at a range of t distances from 0 to 12.5 m, or 1/4 of the distance of the shortest length of the plots (Diggle, 2003). Results of the $g(t)$ function from the Wildcat reference site and four treatments were then plotted against the reference envelopes to visually assess departure.

To test our hypotheses of no differences between the reference and treatment patterns, we tested all patterns against a null hypothesis of complete spatial randomness (CSR) across a specified range of t values. This allowed us to infer statistical differences even though patterns were not directly tested against each other (Fortin and Dale, 2005; Perry et al., 2006; Illian et al., 2008). We first tested for overall departure from CSR at 0–12.5 m using the Cramer–von Mises goodness-of-fit (GoF) test from Loosmore and Ford (2006) with $g(t)$. Values of the test pattern were ranked against the $g(t)$ values of 1000 simulated CSR patterns based on the squared difference from the mean of the CSR patterns. The null hypothesis, no departure from CSR, was tested with a one tailed test using the rank of the test pattern with critical value of 0.05. We then inferred clustering, randomness, or inhibition (even spacing) at specific inter-tree distances based on where $g(t)$ values of the test pattern were above, within, or below the 95th percentile of 1000 CSR patterns respectively (Wiegand and Moloney, 2004). We applied an isotropic (Ripley, 1988) edge correction to all $g(t)$ tests based on an analysis of edge correction methodologies by Yamada and Rogerson (2003).

We plotted clump size distributions at 6 m to visually assess departure of the treatment patterns from reference pattern envelopes, and also used a modified version of the Loosmore and Ford (2006) GoF test. To quantify departure, the Harrod plots were used as the null pattern distribution in place of simulated CSR patterns (Illian et al., 2008). Ranks were based on the sum of differences

between the test pattern and the mean cumulative proportion of trees in 1 to 18 tree clumps; 18 was the largest observed clump size. Ranks were converted to percentiles with the 100th percentile representing the pattern with the highest proportion of trees as individuals and in small clumps, while the lowest percentile indicated the pattern with the greatest proportion in large clumps. This test was also done at 5 m and 7 m to assess how using a different threshold distance for defining clumps would affect the results.

As the Harrod et al. (1999) plots were 0.5 ha (100 × 50 m), we partitioned the 1 ha Wildcat reference plot and the 2 ha treatment plot into three and six 100 m × 50 m subplots, respectively, using a moving window with 10–20 m of overlap. This subsampled local variation in pattern across the treatment plots and provided comparisons at the same spatial extent. We also plotted the Harrod clump size distributions by plant association grouping to assess relationships between AET/Deficit and levels of clumping. Finally, we derived clump size distributions at 2 m and 10 m distances to compare patterns at multiple distances.

2.7. Evaluating functional attributes of resilience

To assess the potential implications for climate adaptation of the different treatment alternatives, we quantified several functional attributes of resilience. First, species composition and diameter distributions were compared to determine the extent to which treatments favored drought- and fire-tolerant species and larger size classes (Table 1). Second, potential susceptibility to mountain pine beetle (MPB; *Dendroctonus ponderosae*) was compared by characterizing inter-tree competition using Stand Density Index (SDI). We derived critical MPB mortality thresholds for the different plant association groupings using methods from Cochran (1994), with data from Lillybridge et al. (1995), and results from other empirical studies (Negron and Popp, 2004; Olsen et al., 1996). SDI was calculated around each individual ponderosa pine using an 8 m radius following methods of Negron and Popp (2004). SDI is commonly used to quantify density relative to maximum carrying capacity (Reineke, 1933) and is an adequate predictor of BA increment (Contreras et al., 2011) and bark beetle vulnerability (Fettig et al., 2007).

A third aspect of resilience we quantified was the amount and size of openings. Delineating and quantifying individual gaps in forests with low canopy cover is challenging due to irregular gap shapes and lack of clear boundaries. Instead, we used the empty space function $F(t)$ to quantify openings in terms of distance from the nearest tree, or distance from the nearest gap edge. The $F(t)$ function generates a grid of cells, set to 1 m in our case, and then derives the distance from the center of each grid cell to its nearest tree. We created histograms of binned $F(t)$ distances for all the grid cells, using 3 m bins. This quantified the proportion of the total plot area in the different bin sizes. A graphical plot of this function was also created to visually assess the spatial distribution of cells in the 3 m bins across each treatment. We defined large openings as those with distances $\geqslant$15 m from the nearest tree or gap edge, based on minimum light requirements for ponderosa pine growth and associated gap sizes (Gersonde et al., 2004; York et al., 2004). Openings of this width are also likely to impede the spread of active or independent crown fire (Peterson et al., 2005).

All analyses were implemented in R v.2.8.1.and made use of functions in the *spatstat* package (Baddeley and Turner, 2005). The R code for the Plotkin et al. (2002) algorithm is available upon request and instructions for implementing it in ArcMAP v10 are provided in the Appendix.

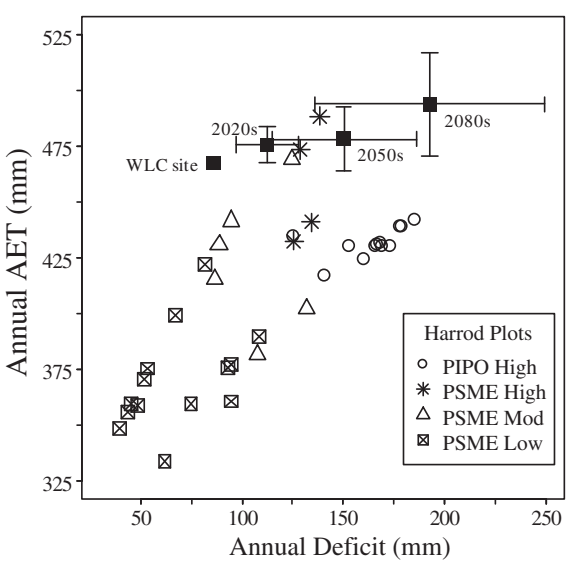

Fig. 1. Actual evapotranspiration (AET) and climatic water deficit (Deficit) for the contemporary climate (1971–2000) for Harrod sites, and current and projected future climates for the Wildcat site. Climate projections include A2 and A1B emissions scenarios from 15 GCMs. See Stephenson (1998) for a detailed explanation of the relationship between AET and Deficit. Plant association groupings are: PIPO High: *Pinus ponderosa/Purshia tridentata/Agropyron spicatum;* PSME High *Pseudotsuga menziesii/Purshia-tridentata/Agropyron spicatum;* PSME Moderate: *P. menziesii/Symphoricarpos albus/Calamagrostis rubescens*, and *P. menziesii/Calamagrostis rubescens*, PSME Low: *P. menziesii/Symphoricarpos oreophilus*, *P. menziesii/Spirea-betulifolia/Calamagrostis rubescens*, and *P. menziesii/Carex geyer*.

3. Results

3.1. Climate analog reference conditions

Annual AET and Deficit values at the Harrod sites were generally consistent with the productivity and moisture gradients of the respective plant associations described in Lillybridge et al. (1995). Ponderosa pine associations had the highest Deficit followed by the Douglas-fir/bitterbrush/bunchgrass (*Purshia-tridentata/Agropyron-spicatum*; PSME-High). Ponderosa pine associations are found on low productivity sites, too droughty for Douglas-fir, while the PSME-High association is found at the dry end of the Douglas-fir ecotone (Lillybridge et al., 1995). The PSME-Moderate and PSME-Low groups contained plant associations found on cooler and moister sites, with generally higher productivity and carrying capacity (Fig. 1).

The present-day Deficit value for the Wildcat site fell well within the range of the same plant association on the Harrod sites (Douglas-fir/shiny-leaf spirea-pinegrass) although AET values were higher. Deficit projections for the Wildcat site for the 2011–2040 period were within AET-Deficit levels for the PSME-Moderate group. By the 2041–70 period, the Wildcat AET-Deficit projections were similar to the ponderosa pine and PSME-High groupings. At the end of the 21st century, only the lowest Wildcat Deficit projections were similar to current ponderosa pine sites. The variability of the AET and Deficit projections increased through time due to increasing difference between the A2 and A1B emissions scenarios and the GCM projections in the second half of the century (IPCC, 2007).

3.2. Prescription development, implementation, and adaptive management

After initial training period, the marking crew implemented the prescription efficiently and created a mosaic pattern of clumps and openings (Fig. 2). The proportion of trees left in 2–4 tree clumps

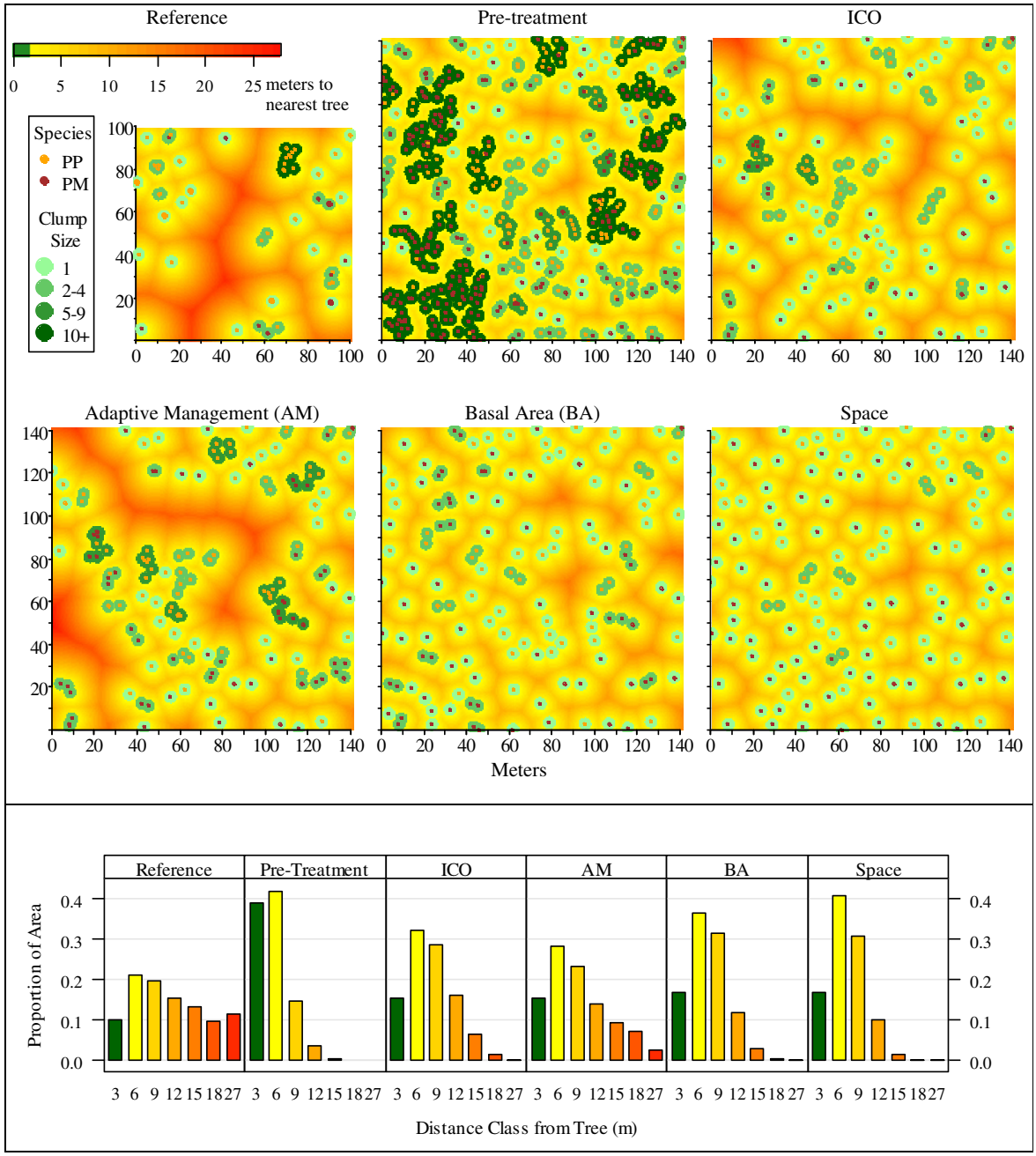

Fig. 2. Upper panel displays tree spatial patterns for reference plot (100 m × 100 m), pre-treatment, and 4 treatment alternatives in the 2 ha monitoring plot (141 m × 141 m). ICO: individuals, clumps, and openings. PP: *Pinus ponderosa*. PM: *Pseudtosuga menziesii*. Projected crown radii are 3 m to indicate interlocking crowns and clump formation. Larger clump sizes are shown in darker green colors. Background coloration indicates the distance to nearest tree or gap edge from the centers of a 1 m cell grid – a graphical representation of the empty space function. The areas colored dark orange in plots are areas that are approximately 15 m from the nearest tree or gap edge. An opening with a relatively circular center of dark orange, such as in the Pre-treatment plot, has an approximate area of 0.07 ha (the area of a circle with a radius of 15 m). The long, sinuous opening the AM pattern, for example, has dark red in the middle indicating that it is roughly 50 m wide (2 × 25 m distance to the gap edge). The lower panel displays histograms that show the proportion of area in 3 m distance to nearest tree bins with corresponding colors. Bin labels are maximum values (e.g. 6 m = 3.01–6 m). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

closely approximated the target; 34–37% respectively, however, an insufficient number of trees were left in 5 + tree clump (4% vs. the target of 21%). Results from the 2 ha monitoring plot were similar (Fig. 3), but 10% of trees were left in 5+ tree clumps. Through discussions with the marking crew, we learned that the crew had difficulty deciding when to thin from below and when to leave smaller trees to form larger clumps. In addition, we had not defined the upper range of clump size for the 5+ tree clumps. Consequently, the crew left mostly 5–6 tree clumps and no clumps with more than eight trees. Most importantly, tracking results were not tallied until after marking was completed, so bias was not detected early in the implementation process when corrections could have been made. There was also some confusion as to whether stream buffer zones (no-entry "skips") should count as large clumps. Finally, our assumption that large openings would occur as a result of creating clumps did not prove correct (Fig. 2). Crews were uncomfortable with creating large openings, especially when it required marking large trees for removal.

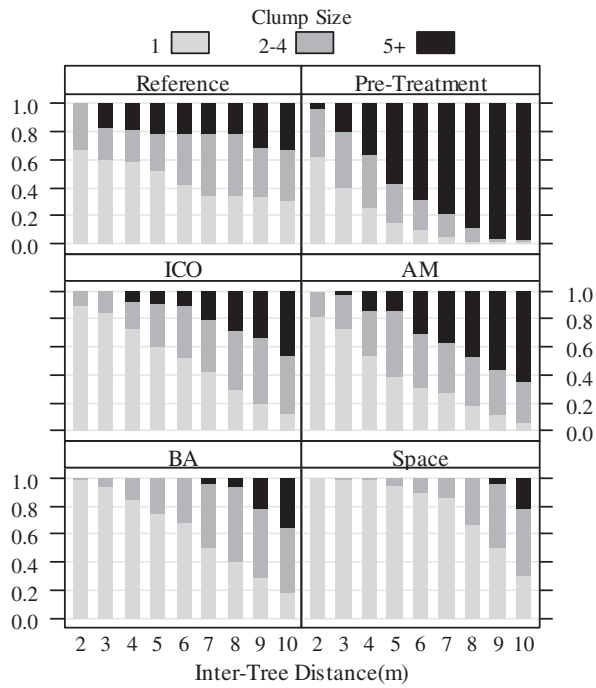

Fig. 3. Proportion of trees in different clump size bins across different inter-tree distances in the 1 ha reference plot and pre-treatment and four treatment alternatives on the 2 ha monitoring plot: the actual treatment (ICO), adaptive management (AM), basal area (BA), and spacing based (Space). Clump size is defined as the number of trees comprising the clump. Six meters was used to define tree clumps in subsequent analyses (Fig. 5).

3.3. Treatment alternative and reference pattern envelopes

We accepted the hypothesis of no difference between the ICO treatment pattern and those of Harrod sites, as well as the hypotheses that the BA and Space treatment patterns were different than the Harrod sites. Most Harrod plots (23 out of 32) showed statistically significant deviance from complete spatial randomness (CSR) in the GoF test, and were clustered patterns. The remaining 9 plots displayed no difference from CSR, and none were uniform (see Appendix). The ICO treatment was also different from CSR and clustered, but was close to being a random pattern ($p = 0.048$) (Table 4, Fig. 4). Both the Wildcat reference site and the simulated AM treatment were different than CSR and clearly clustered ($p = 0.001$ and $p = 0.002$ respectively). The two patterns also followed a similar pattern of clustering and randomness at short (<6 m), intermediate (6–12 m), and long intermediate distances (>12 m). The BA and Space treatments displayed significant deviance from CSR and were uniform, although the BA pattern was close to random ($p = 0.047$). Compared directly to the Harrod $g(t)$ pattern envelope as opposed to a CSR envelope, the BA and Space treatments were near the boundary, or outside, of the Harrod envelope (Fig. 4B).

Comparison of the cumulative clump size distributions and the modified GoF rankings confirmed the results of the GoF tests (Table 4 & Fig. 5). Of the treatment patterns, the AM treatment was the most clumped, and was on the high end of clumping relative to the Harrod envelope (40th percentile). Approximately 1/3rd of the trees in this treatment were individuals, 1/3rd were in small clumps (2–4 trees), and 1/3rd were in medium clumps (5–9) (Fig. 3). It displayed high variation in the subsample of 0.5 ha plots laid out across the 2 ha plot (Figs. 2 and 5). The ICO treatment was less clumped relative to the Harrod envelope (65th percentile), with 52% of trees as individual trees, 40% in small clumps, and 8% in medium clumps; and more modest variation in the 0.5 ha subsamples. In the BA treatment, 70% of the trees were individuals trees with the remainder in small clumps. The pattern had little variation in the 0.5 ha subsamples, and was near the low clumping boundary of the Harrod envelope (95th percentile). The Space treatment created a pattern outside the Harrod envelope (100th percentile) with 90% of the trees as individuals, and no variation in the 0.5 ha subsamples. The percentile rankings of the four treatments varied somewhat using clump size distributions based on 5 and 7 m distances, but results were generally the same (Table 4).

Finally, the cumulative clump size distributions of the Harrod plots were plotted by AET/Deficit plant association group to assess

Table 4
Summary information for 1 ha reference plot and different patterns on 2 ha monitoring plot.

	Plot/pattern					
	Reference	Pre-treat	ICO	AM	BA	Space
Density and size						
Trees per ha	48	221	64	70	67	65
Mean dbh (cm)	77.5	40.4	55.6	53.1	54.5	56.6
Basal area (m^2/ha)	24.9	32.7	16.7	16.9	16.9	17.5
% Max stocking[1]	40%	64%	30%	31%	30%	31%
Species composition						
PIPO (%BA)	74%	35%	48%	49%	42%	39%
PSME (%BA)	26%	65%	52%	51%	58%	61%
Pattern type	Clumped	Clumped	Clumped	Clumped	Uniform	Uniform
PCF results						
p-Value	0.001	0.001	0.048	0.002	0.048	0.008
Distances (m)	0.1–6.7 and 9.8–12.5	0.1–9.3 and 14.0–15.1	0.3–3.7	0.1–5.8 and 6.7–10.2	0.1–2.2 and 3.5–6.4 and 11.2–12.5	0.1–7.7
Harrod percentile rank						
5 m	22%		68%	49%	95%	97%
6 m	35%		65%	41%	95%	100%
7 m	46%		49%	39%	81%	100%

ICO (individuals, clumps, and openings) is actual field treatment. AM is adaptive management, BA is basal area. Percent of maximum stocking is calculated in Stand Density Index with a maximum value of 800. PIPO: *Pinus ponderosa*. PSME: *Pseudtosuga menziesii*. PCF (pair correlation function) *p*-values are derived from a Monte Carlo goodness of fit test to test for significant difference from complete spatial randomness (CSR) from 0–12.5 m. Distances are the inter tree distances at which the pattern differs from the maximum or minimum envelope of 1000 CSR patterns. Harrod percentile ranks are the percentile rank of the pattern relative to the 32 Harrod patterns; lower values are more clumped and higher values are more uniform.

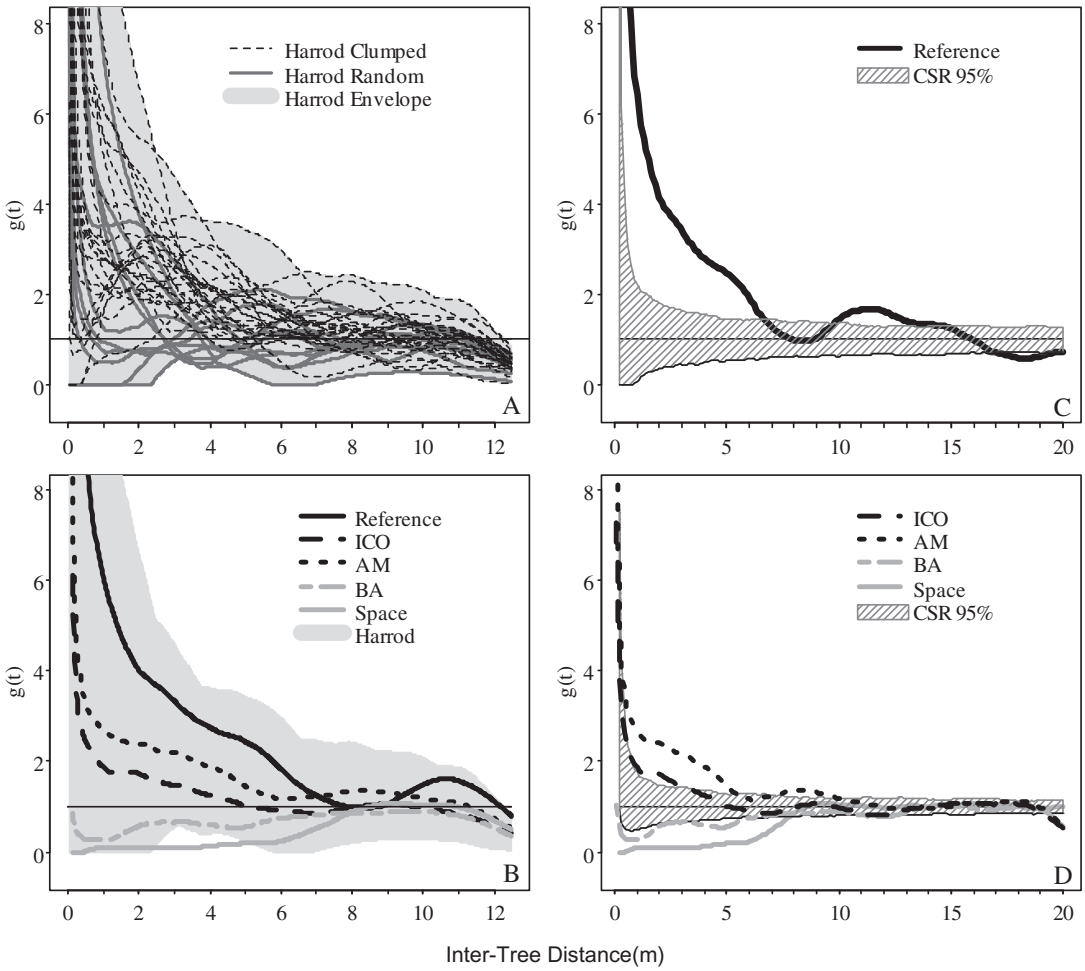

Fig. 4. Point pattern statistics for study plots and Harrod et al. (1999) plots. Panels A and B compare envelopes of the pair correlation function of 32 Harrod et al. (1999) plots with the reference plot, field treatment (ICO), and 3 simulated treatments: adaptive management (AM), basal area (BA), and spacing based (Space). The majority (72%) of the Harrod plots display clustered tree patterns, while the remainder are random (panel A) based on a Monte Carlo goodness of fit test which evaluated patterns from 1 m to 12.5 m. Panels C and D show pair correlation functions of study plots with 95th percentile CSR envelopes. Distances at which curves are within CSR envelopes indicate spatially random tree patterns, while curves above or below CSR envelopes indicate clustered or uniform patterns, respectively. The CSR envelope of reference plot (panel C) is larger due to its lower point density.

the effect of AET/Deficit on pattern (Fig. 5). Patterns dominated by medium to large clumps (10+ trees), as well as patterns with mostly individual trees and small clumps, were found in all of the plant association groupings. The ponderosa pine grouping had only one medium clump pattern, however, with the rest containing predominantly individuals and small clumps (2–4 trees).

3.4. Functional attributes of resilience

By design, density and average diameter measures were similar among the ICO and three simulated treatments (Table 4). A major difference was the greater retention of small and mid-size pine in the ICO and AM treatments compared to the BA and Space treatments (Fig. 6). In the AM treatment, removing additional large trees (>65 cm dbh) was necessary to create openings. This resulted in retention of 7.5 vs. 10 large (>65 cm dbh) Douglas-fir per ha and 7.5 vs. 8 large pine per ha in the AM vs. ICO treatments, respectively. Large trees were also cut in the BA and Space treatments, but to achieve uniform spacing. The proportion of pine basal area was 10% higher in the ICO and AM treatments compared with the BA and Space treatments, but was still 25% less than the reference plot (Table 4).

The BA and Space treatments resulted in zero pine susceptible to mountain pine beetle mortality based on the critical SDI thresholds from the Wildcat site (Fig. 7). If the lower SDI thresholds from the High-Deficit PIPO and PSME plant associations on the Harrod sites are considered, approximately 25% of the pine were susceptible (the extent of the boxplot above the threshold lines in Fig. 7). In the ICO and AM treatments, approximately 50% and 66% of the pine were susceptible, respectively, using the lower SDI thresholds. Trees in the Harrod plots, pooled by plant association grouping, and the Wildcat reference site showed a wider range of inter-tree competitive conditions than any of the treatments. In all groupings, 30–45% of the pines were above their site specific critical SDI thresholds (Fig. 7). Extreme SDI values (2000+) caused by large clumps of large diameter trees were not uncommon in the reference stands.

The spatially explicit plots and histograms of the $F(t)$ function showed differences in the total area, size, and shape of openings among the patterns (Fig. 2). Large openings, defined as those with the opening center $\geqslant$15 m from a tree or gap edge, were generally sinuous, ill-defined, and lacked distinct boundaries. The shape and approximate area of large openings can be estimated by examining the background color surfaces of the stem maps in Fig. 2 Approximately 21% of the area in the Wildcat reference plot was >15 m

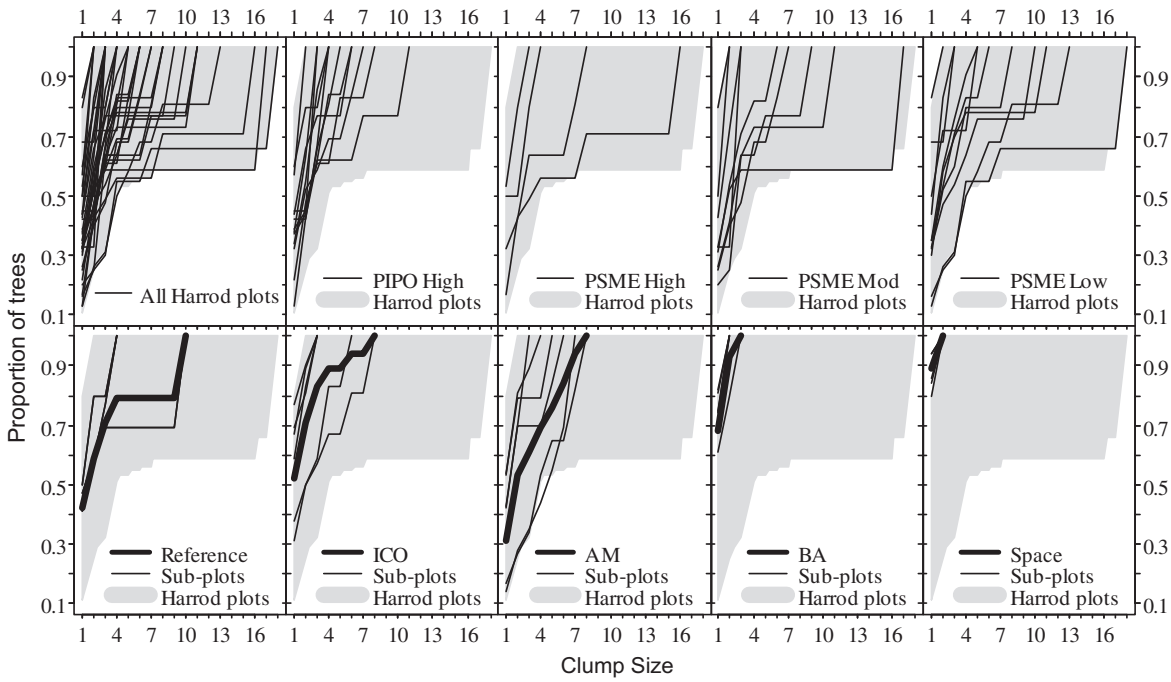

Fig. 5. Comparison of the cumulative clump size distributions for reference and treatment plots against shaded envelope of Harrod et al. (1999) plots. Top row compares plots from Harrod et al. (1999) by plant association groupings from Fig. 1. Bottom row displays the reference plot, and four treatment alternatives: actual treatment (ICO), adaptive management (AM), basal area (BA), and spacing based (Space). In bottom row, bold lines show the distributions of the whole plots. Narrow lines are 0.5 ha sub-samples taken across each large plot using a 0.5 ha (100 m × 50 m) moving window. Clump size is the number of trees in a clump and is based on a 6 m inter-tree distance.

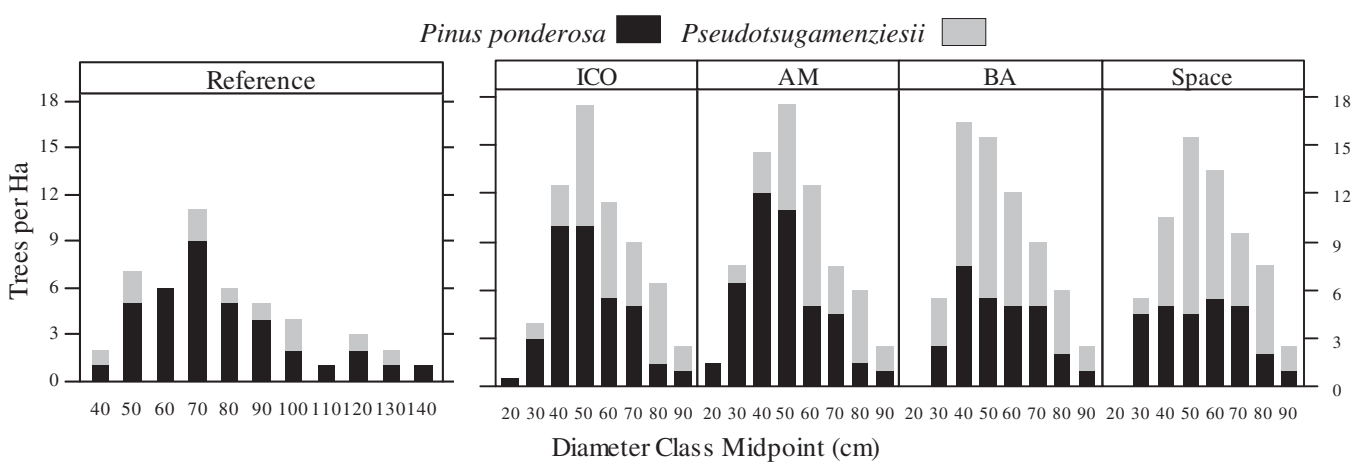

Fig. 6. Diameter distributions of reference plot, ICO treatment, and three simulated treatment alternatives: adaptive management (AM), basal area (BA), and spacing based (Space). Only leave trees are shown.

from a tree or gap edge, while the AM and ICO treatments displayed 10% and 2%, respectively. The BA and Space treatments did not contain any area >15 m from a gap edge.

4. Discussion

4.1. Prescription development, implementation, and adaptive management

The ICO method proved to be a useful framework for conceptualizing stands as mosaics of individuals, clumps and openings. Spatially assessing variation in tree density in terms of individuals and clumps instead of basal area or spacing required training, but resulted in a more intuitive and concrete way for the marking crew to understand and create the desired heterogeneity. Use of a single distance based on crown interlock was necessary to clearly define tree clumps and make implementation operationally tractable. There is no single correct crown overlap distance (Sánchez Meador et al., 2011), however, and choosing a 5 m or 7 m distance would have resulted in somewhat different target proportions for clumps and individuals. Selecting a 5, 6, or 7 m distance would have resulted in highly similar patterns, as the proportion of trees in small, medium, and large clumps increased in a roughly linear fashion with inter-tree distances from 4 m to 7 m (Fig. 3).

The insufficient number of large clumps and openings in the ICO mark demonstrated the need for monitoring and adaptive learning when transitioning to new approaches (Knapp et al., 2012). In subsequent restoration projects, we have added a bin for larger clumps and now use four: individual trees, small clumps (2–4

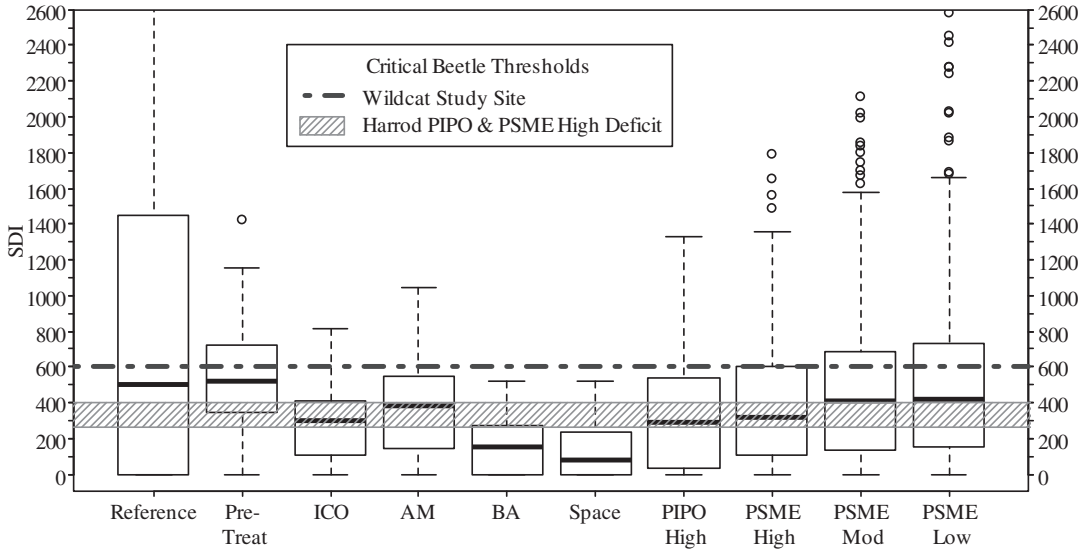

Fig. 7. Boxplot showing distribution of Stand Density Index values for 8 m radius plots placed around all ponderosa pine trees for reference plot; four treatment alternatives; and plots from Harrod et al. (1999) grouped by plant association/water balance deficit level from Fig. 1. Black lines in center of boxes are medians, boxes show the interquartile range (25–75%) and whiskers show the 10–90% range. Critical threshold values are for mountain pine beetle mortality. SDI values are in metric units; to convert to English units divide by 2.47. The proportion of trees above the thresholds for a plot (as shown by the intersection of the respective boxplot and the threshold line) is the proportion of trees considered at risk to mountain pine beetle mortality.

trees), medium clumps (5–9 trees) and large clumps (10–20 trees). We also created a simple tracking system for both leave and cut tree marking that provides real time feedback during marking. Crews are now consistently meeting the clump targets in ongoing projects based on results from this tracking system. Once crews are familiar with the methods, we are finding that it takes a similar amount of time to implement as more traditional marking approaches. More details on implementing the method are available from the corresponding author.

Another lesson learned was the need to provide explicit direction for the creation of large openings. Retaining large, fire tolerant trees is a key principle of dry forest restoration and increasing resilience (Agee and Skinner, 2005; Hessburg and Agee, 2003; Taylor and Skinner, 2003), and removal of pre-settlement era trees (old trees) is not necessary to restore pattern (Franklin and Johnson, 2012). However, we concur with others that rigid diameter limits, with no flexibility for young trees or fire intolerant species, can conflict with restoration of spatial pattern and other objectives (Abella et al., 2006; North et al., 2007). Other factors to consider when creating openings are edaphic conditions such as shallow soils and disturbance processes, including prescribed fire. Refining methodologies to quantify openings in reference patterns is also needed to provide more explicit guidelines for openings.

The ICO method is one approach to implementing variable density (Carey, 2003; O'Hara et al., 2012) or free thinning (Tappeiner et al., 2007). A key challenge for any approach that seeks to create within-stand heterogeneity is to balance the needs for (1) concrete, ecologically based targets for spatial pattern that can be objectively monitored, (2) sufficient flexibility to achieve other structural and compositional objectives (e.g., Table 1), and (3) operational simplicity and efficiency (North and Sherlock, 2012; O'Hara et al., 2012). In designing the ICO method, we explicitly sought to balance these needs by combining elements of other approaches that use spatial information from dry forest reference conditions. The direct, transparent link to reference conditions is similar to the methods of Covington et al. (1997) and Waltz et al. (2003), who used the locations of pre-settlement stumps, snags, and downed logs to guide spatial patterns of tree retention. However, the ICO method is not based off the pre-settlement locations of clumps and thus does not require that evidence of pre-settlement pattern be present. Our approach seeks a middle ground between direct reliance on pre-settlement structures and more flexible "free selection" methods (Lynch et al., 2000; Graham et al., 2007; Jain et al., 2008; Mitchell et al., 2006), which typically lack concrete targets for spatial pattern, require more judgment to implement, and can result in stands that are overly clumped relative to reference conditions (North et al., 2007). Our method is similar to approaches that prescribe a target distribution of basal area or SDI and instruct marking crews to create patches of different densities while incorporating other leave tree criteria (Harrod et al., 1999; USFS, 2008). Similar to Knapp et al. (2012), we have found that identifying and tracking tree clumps is more intuitive and efficient than using basal area. Finally, the ICO method is conceptually similar to several uneven-age, multi-cohort management approaches (e.g. Bailey and Covington, 2002; O'Hara, 1996), and would benefit from more rigorous quantification of growing space allocation using tools from these approaches.

No prescription approach is optimal for all stand conditions, operational systems, and objectives. The ICO method works best in even-age stands simplified by past management. It may not be the right tool in stands with serious forest health issues, poor live crown ratios, or where a major shift in species composition is needed. In stands with large numbers of live pre-settlement era trees, simply retaining these trees can restore spatial pattern (Larson et al., 2012; Lutz et al., 2012), although the ICO method can be useful for managing younger cohorts in such stands.

Another challenge is obtaining stem map data. Stem maps are not mandatory to implement the method, however. Regional reference datasets exist for most areas of the interior western US (Larson and Churchill, 2012). An effort is underway to quantify regional reference pattern envelopes through a meta-analysis of all existing reference stem maps and make the information available to managers in a user friendly format. Additional 3–10 ha stem maps will be needed to ensure that existing datasets indeed capture the range of variation in pattern across the dry forest landscape. Use of remote sensing tools in areas with minimally altered or intact fire regimes to derive reference clump and opening size distributions over larger spatial extents is needed to examine variation in pattern relative to shifts in biophysical conditions, as well as to ensure that the upper range of clump and opening sizes

is captured. Field methods to sample clump and opening size distributions without a full stem map are also needed. Finally, further examination of the edge effects on clump size distributions is warranted.

4.2. Climate analog reference conditions

Climate analog reference conditions offer a way of utilizing information from pre-settlement era conditions while factoring in climate change projections (Gärnter et al., 2008; Keane et al., 2009). By modeling the climate envelopes of plant associations instead of species presence/absence, we were able to show that future projected AET-Deficit of our study site may be within the range of the Harrod sites, at least through the middle of the century. This suggests that Douglas-fir and ponderosa pine will likely be viable species in this area for multiple decades, although productivity and carrying capacity may decline. Other studies have found similar results using climate envelope models (Littell et al., 2010; Rehfeldt et al., 2006), as well as mechanistic models (Coops and Waring, 2011; Griesbauer et al., 2011). Deficit is also a major driver of the two primary disturbance agents in this ecosystem: fire occurrence and severity (Littell et al., 2009; Westerling et al., 2006) and insect related mortality (Littell et al., 2010).

As forest structure and spatial pattern at this scale are shaped by the interaction of vegetation types, fine-scale biophysical variables, disturbances, and climate (Moritz et al., 2011), we assumed that the Harrod sites on moderate and high Deficit plant associations (Fig. 1) were reasonable climate analog reference sites. The plots on these plant associations encompassed almost the full range of the spatial patterns found in the Harrod reference envelope (Fig. 5), whereas the plots on ponderosa pine associations contained predominantly individual trees and small clumps, with only one large clump of 11 trees. This suggests that moisture and site productivity limited pattern only on the harshest Harrod sites, which are currently at the forest and shrub-steppe ecotone (Lillybridge et al., 1995). Abella and Denton (2009) also found reduced clumping on soils with lower moisture and nitrogen levels in ponderosa pine forests in Northern Arizona and attributed it to limitations on tree establishment. Based on the observed relationships between plant association groupings, clump size distributions, and AET-Deficit relations, the full Harrod pattern envelope is a reasonable reference pattern envelope for the Wildcat site through the first climate projection period (2010–2040). After that, the ability of the site to support large clumps may be diminished.

As with all climate change projections, our conclusions must be interpreted and applied in the context of the associated model uncertainty. Climate change models and projected ecological effects are best interpreted at watershed (10,000+ ha) or larger scales (Peterson et al., 2011). Downscaling climate data to the stand scale introduces additional error, especially in complex terrain with strong orographic and microsite effects, such as in our study area (Lundquist and Cayan, 2007). Our method does not factor in localized and fine-scale weather patterns such as cold air drainage and pooling (Daly et al., 2010). Furthermore, changes in other climate variables such as growing season timing and length and minimum winter temperatures may result in additional physiological stress and maladaptation issues (Chmura et al., 2011; Griesbauer et al., 2011). Climate envelope approaches such as ours rely on the assumption that current species ranges are constrained primarily by climate, a top-down spatial control, and that past and future climates will feature similar climate envelopes for each species.

Despite these limitations, use of climate analog reference conditions translates climate projections into site-specific information for managers that is based on empirical relationships between the predicted climate, potential species viability, and site potential. The fact that AET and Deficit captured differences among plant associations reasonably well confirm that they are good composite variables for the components of the biophysical environment that affect plant distributions (Stephenson, 1998). Perhaps most importantly, the climate analog approach facilitates a forward-looking forest restoration paradigm that is not based on recreating historical conditions, but seeks to utilize historical information to restore pattern-process relationships appropriate to specific climatic conditions.

While recommended, we stress that climate analog reference conditions are not required to use the ICO approach. The ICO method is a way to incorporate spatial reference information into prescriptions and marking guidelines, while the climate analog approach is a framework for selecting climate-appropriate reference conditions. Dry forest treatments based on historical reference conditions alone are generally a positive step towards climate adaptation and resilience (Keane et al., 2009; Spies et al., 2010), and spatial patterns can be adjusted in future entries as the climate changes.

4.3. Treatment alternatives and reference pattern envelopes

The BA and Space simulated treatments resulted in patterns that differed from the Harrod reference envelopes and the AM and ICO treatments in two key ways. First, the BA and Space patterns were uniform *and* had relatively high densities (tph). Many of the Harrod patterns (30%) were random and the rest clustered, similar to other reconstruction studies (Abella and Denton, 2009; Binkley et al., 2008; North et al., 2007; Youngblood et al., 2004). A few were close to being statistically uniform, but these had much lower densities than the 64–70 tph of the treatment alternatives (see Appendix). The higher densities, combined with more regular spacing, explain why the BA and Space patterns are statistically different from the Harrod patterns using the GoF test, but within part or all of the Harrod envelopes for the $g(t)$ and clump size distributions. As density increases, the "additional" trees are more likely to be found in larger clumps rather than spaced out evenly across the stand. This is an intuitive result given that regeneration in dry forests is typically clumped and often associated with patchy mortality of overstory trees (Sánchez Meador et al., 2009). The BA and Space patterns also lacked of variation in the 0.5 ha subsamples across the 2 ha plot (Fig. 5). Regularly spaced, individual trees are indeed a component of reference patterns. However, the Harrod plots, along with other studies (Abella and Denton, 2009; Youngblood et al., 2004), suggest that regularly spaced trees rarely cover more than 0.5 ha of contiguous area at densities >50 tph.

It is clear that applied over entire stands, or especially multiple contiguous stands, stand-average basal area and spacing based prescriptions do not restore the range of patterns that existed when frequent fire occurred. Over time, prescribed fire, natural disturbances, and post-treatment stand development will cause mortality, openings, regeneration, and new clump formation (Stephens et al., 2009; Waltz et al., 2003). These processes will likely nudge the BA and Space patterns into the reference envelope over time. However, it may take a long time for natural processes to create openings, and clumps of overstory trees will take many decades to re-form if eliminated during treatments. Basal area and spacing approaches can be modified to leave a greater range of density (e.g. USFS, 2008), but explicitly prescribing the creation of openings and retention of large clumps in such prescriptions may be necessary to achieve the desired pattern.

4.4. Managing for resilience

The primary objective of the treatment was not to recreate the pre-settlement forest but to enhance resilience. Tree patterns in the AM treatment displayed a steeper "gradient of resistance" to different disturbance agents across the plot (*sensu* Moritz et al.,

2011), similar to many of the Harrod plots. At one end of this gradient are "fences" of high resistance that may slow or stop the spread of high intensity disturbance, such as openings or areas of widely spaced trees. At the other end of the gradient are patches of potential instability that can lead to decadence, mortality, and subsequent regeneration of early-seral species (Odion and Sarr, 2007). Examples include high density clumps that are well above critical beetle thresholds, mistletoe patches, or multistory tree clumps conducive to torching. The steeper gradient of resistance in the AM pattern compared to the other patterns predisposes it to higher variation in disturbance effects and post-disturbance development pathways (Parisien et al., 2010; Thaxton and Platt, 2006; Waltz et al., 2003). In theory, patterns with more fences and corridors are less susceptible to high severity disturbances that reset entire stands (North et al., 2009; Stephens et al., 2010) and initiate non-linear shifts to new system states (Messier and Puettmann, 2011; Paine et al., 1998). As the pattern of past disturbances influences future distances (Collins et al., 2009; Peterson, 2002), the mosaic will theoretically be more likely to replenish itself and maintain its basic pattern elements over time through fine-scale, gap phase replacement even as the climate changes. Larger scale climate drivers and disturbances will occasionally override the effects of stand level patterns, however (Turner, 2010).

Tree patterns in the BA and Space treatments contain fewer ponderosa pines susceptible to mortality from mountain pine beetle attack due to more uniform spacing and a lower proportion of pine to Douglas-fir. Higher clumping levels in the AM treatment would likely sustain higher endemic levels of bark beetle mortality, especially during drought periods (Olsen et al., 1996) and where warming trends lead to non-linear changes in beetle life histories and survivorship (Bentz et al., 2010). However, the fact that 30–45% of pines were above beetle thresholds in the Harrod plots demonstrates that many old trees are able to persist in high density clumps, embedded in a mosaic of individuals and openings, for multiple centuries. High severity insect outbreaks did occur historically (Weaver, 1961).

From a climate adaptation standpoint, ongoing tree mortality from endemic levels of insects and pathogens is an important element of resilience and system adaptation. Periodic mortality and subsequent regeneration pulses likely select for genotypes adapted to a wider range of climate regimes than regeneration that establishes quickly after a single severe disturbance event, thereby taping into the high level of local genetic variability in ponderosa pine and Douglas-fir (Hamrick et al., 1989; Linhart et al., 1981). The larger openings in the AM pattern and potential for "time release mortality" of clumps and subsequent regeneration may lead to higher levels of establishment and faster growth of seedlings (Boyden et al., 2005; Fajardo et al., 2006; Sánchez Meador et al., 2009; York et al., 2004). Regeneration will thus more quickly attain size classes capable of reproduction, which may facilitate a gradual *in situ* shift to Douglas-fir and pine genotypes adapted to new climates (Chmura et al., 2011). More uniform treatments may result in less mortality in the short run, but if implemented over large areas may lead to landscapes that have less capacity to adapt to new climates.

There is no single, optimal stand-level approach to maximizing resilience and adaptive capacity for all future conditions. As the climate changes, shifts in the interactions among the vegetation, disturbances, and the climate are likely to give rise to evolving communities and pattern-process relationships (McKenzie et al., 2009; Williams et al., 2007). A resilience strategy that maximally conserves options is prudent. Thus, it is sensible to vary patterns and structure types between stands, thereby varying risk levels and functional tradeoffs among different organisms and processes, as well as economic and social factors. Applying any single treatment type over a large area will increase risks of unintended consequences. Selection of specific strategies and targets for individual stands first requires planning over larger spatial extents in order to link stand-level prescriptions to larger scale resilience strategies (Gaines et al., 2010; Keane et al., 2009; Peterson et al., 2011; Spies et al., 2010). In many cases, disparate stands may need to be blended together into larger patches to restore resilient patch size distributions (Hessburg et al., 2004; Moritz et al., 2011). In other cases, stands may need to be broken up into smaller patches. Stand-level reference envelopes can be used to accomplish this, while also providing ecologically based targets for within-stand heterogeneity. For example, reference patterns that contain high levels of clumping and openings can be used to guide prescriptions for some stands and less clumped patterns used for others. Varying overall density and species composition targets, guided by biophysical setting (North et al., 2009), will also enhance resilience.

5. Conclusion

Managing dry forest landscapes to increase resilience in the face of changing climate and fire regimes poses an immense challenge. Considerable research confirms that dry forest landscapes were heterogeneous at multiple scales prior to the period of fire suppression, and that this heterogeneity was an important driver of the resilience of these systems. This research identified methods that (1) quantified a range of spatial reference patterns to provide concrete targets for stand-level restoration treatments, (2) translated these ranges into operationally efficient marking guidelines, and (3) facilitated monitoring and evaluation of treatment effectiveness. Furthermore, we developed a tractable method to critically evaluate historical reference targets in light of climate change based on empirical correlations between pattern and biophysical conditions, expressed through site water balance and plant association groups. Climate envelope modeling can indicate a site's projected future water balance, thereby allowing managers to select treatment targets from reference sites that are analogous to the projected future conditions of the treatment site. But with or without a climate analog assessment, restoring mosaics of individual trees, clumps, and openings can make future forests more resilient by re-establishing pattern-process linkages. Minimizing treatments that result in spatial homogeneity over large areas can avoid conditions that are likely more susceptible to large-scale, high severity disturbance events. The ICO method provides a quantitative framework to vary pattern both within and among stands that is directly related to reference conditions and can be tailored to the current and future conditions of specific stand. Although initially challenging, managing for spatial heterogeneity can be done.

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Appendix. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2012.11.007.

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Appendices to:

Restoring forest resilience: From reference spatial patterns to silvicultural prescriptions and monitoring

A. Marking Guidelines for Wildcat Unit 44, Okanagon-Wenatchee National Forest:

Spacing:

- Leave an average of 100 tph (40 tpa)
- Leave an average of 12 clumps/ha w/2-4 trees, and 2 with 5+ trees. Clumps have trees w/in 6 m (20 ft) of another tree.
- Leave 42 tph (18 tpa) as individuals with average spacing about 9m (30 ft) with a range of 6-12 m.

Leave Tree Criteria:

- Retain all old trees, established before about 1900. Note, that is younger than Van Pelt (2008) rating greater than 6 for PP and 7 for DF
- Around old PP, remove 100yr age class DF for 1-2 driplines—OK to keep 1-2 large/vigorous DF occasionally-use judgement.
- Thin from below removing mostly trees <21 inch to meet tree density/LCR objectives. Removal of trees >63cm (25") isn't expected. Maybe on east end as needed to prevent mistletoe spread to the west. Remove DF w/LCR <40 (Can go to <35 for clumping,check growth). Retain occasional understory w/LCR > 40 (+- 2/ac)
- Remove 100 yr PP w/LCR <30% or with Van Pelt fig.69 form C or D.

Insect and Disease**:**

- Retain only old trees that are mistletoe infected and isolate them as clumps or individuals with a 40-50 foot host-free (80-100%) buffer beginning at the last visible sign of infection.
- Keep DF mistletoe on east end from moving into unit.

Snags/Down Wood & Size**:**

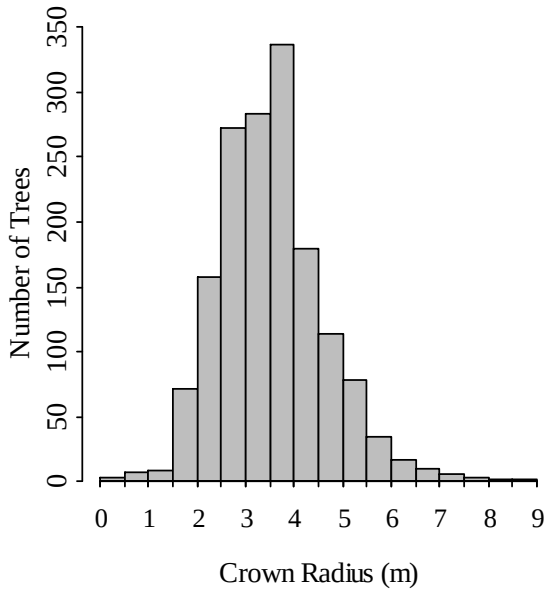
- Protect existing decay class 3 or less snags > 20 inches with a no-cut buffer.
- Retain at least 1-2 of the 15-19 and all > 20 inch trees with dead, forked, or broken tops

Wildlife**:**

- Retain about 15% of stand in no-entry complex patches—counting no-cut riparian areas and patch in SW corner wet area.

B. Crown radii

Frequency distribution of crown radii for trees between 39 and 139 cm dbh. Data are from the USFS Forest Inventory and Analysis database using plots in the eastern Washington Cascades, n = 1584 trees. Bins are 0.5m width.



C. General Circulation Models (GCMs)

List of GCM's used to derive climate variables for both A2 and A1B emissions scenarios. Detailed information on each GCM is available at the IPCC website: http://www.ipcc-data.org/ar4/gcm_data.html. Downscaled model projections for these GCM's were downloaded from Climate WNA: http://www.genetics.forestry.ubc.ca/cfcg/ClimateWNA/ClimateWNA.html.

BCCR:BCM2
CCCMA:CGCM3
CNRM:CM3
CSIRO:MK3
GFDL:CM2
GFD: CM2_1
NASA:GISS-EH
INM:CM3
IPSL:CM4
NIES:MIROC3_2
ONS:ECHO-G
MPIM:ECHAM5
MRI:CGCM2_3_2
NCAR:CCSM3
UKMO:HADCM3

D. Harrod Reference Plot Summary Information:

Summary density metrics, clump size distributions, and pattern type of Harrod et al. (1999) reference plots. Plant association groupings are based on calculated water balance deficits for each plot. SDI is Stand Density index and is in metric units; to convert to English units, divide by 2.47. Pattern types and p-values are derived from a Monte Carlo goodness of fit test to test for significant difference from complete spatial randomness from 0-12.5m using the pair correlation function.

Plot ID	Plant Association Grouping	Trees Ha^{-1}	Mean DBH (cm)	BA m^2/ Ha	SDI	Proportion of trees by Clump size (# of trees)				Pattern Type	P-Value
						1	2-4	5-9	10-20		
CD1	PIPO High	48	66.3	17	256	0.42	0.58	0	0	Clustered	0.005
CD7	PIPO High	58	46.4	11	181	0.34	0.66	0	0	Clustered	0.015
CD8	PIPO High	92	46.5	17	276	0.22	0.39	0.39	0	Clustered	0.005
CD9	PIPO High	28	54.1	7	112	0.57	0.43	0	0	Random	0.245
CD14	PIPO High	48	62.1	16	236	0.38	0.63	0	0	Clustered	0.020
HD1	PIPO High	82	60.4	26	385	0.37	0.44	0.19	0	Clustered	0.005
HD2	PIPO High	72	43.0	12	193	0.39	0.45	0.17	0	Clustered	0.005
HD3	PIPO High	22	71.4	10	138	0.45	0.55	0	0	Clustered	0.020
HD4	PIPO High	40	53.8	9	148	0.6	0.4	0	0	Random	0.130
HD5	PIPO High	94	51.0	21	330	0.13	0.49	0.15	0.23	Clustered	0.005
WD1	PIPO High	54	73.3	26	361	0.44	0.33	0.22	0	Clustered	0.020
HD6	PSME High	60	57.4	17	264	0.53	0.47	0	0	Random	0.235
HD7	PSME High	82	50.3	18	279	0.32	0.32	0.37	0	Clustered	0.010
HD8	PSME High	108	52.4	25	387	0.17	0.39	0.15	0.3	Clustered	0.005
HD9	PSME High	40	57.0	11	171	0.5	0.5	0	0	Random	0.090
CD3	PSME Mod	80	63.7	29	426	0.25	0.43	0.33	0	Clustered	0.005
CD4	PSME Mod	78	54.3	19	295	0.26	0.38	0.36	0	Clustered	0.010
CD6	PSME Mod	24	67.6	9	137	0.5	0.5	0	0	Random	0.115
CD13	PSME Mod	80	63.0	28	408	0.2	0.53	0	0.28	Clustered	0.005
WM1	PSME Mod	36	65.5	13	187	0.33	0.67	0	0	Clustered	0.035
WM2	PSME Mod	42	65.3	15	222	0.43	0.58	0	0	Random	0.075
WM3	PSME Mod	20	75.2	10	137	0.8	0.2	0	0	Random	0.560
WM4	PSME Mod	70	61.6	23	338	0.31	0.51	0.17	0	Clustered	0.005
CD2	PSME Low	68	70.6	28	406	0.35	0.48	0.18	0	Clustered	0.005
CD5	PSME Low	84	50.6	18	292	0.33	0.31	0.12	0.24	Clustered	0.005
CD10	PSME Low	36	69.5	15	216	0.5	0.22	0.28	0	Clustered	0.005
CD11	PSME Low	24	54.8	6	98	0.83	0.17	0	0	Random	0.870
CD12	PSME Low	38	76.1	18	260	0.68	0.32	0	0	Clustered	0.040
WD2	PSME Low	126	58.5	37	557	0.13	0.37	0.31	0.21	Clustered	0.005
WD3	PSME Low	100	57.4	28	429	0.3	0.48	0	0.22	Clustered	0.020
WD4	PSME Low	80	59.3	24	359	0.32	0.48	0.2	0	Clustered	0.020
WD5	PSME Low	128	52.8	30	466	0.16	0.39	0.11	0.34	Clustered	0.005

E. Implementing Plotkin et al. (2002) clump identification algorithm in ArcMAP 10.

1. Compile the stem map data in a text file with columns for x and y coordinates and any desired tree attributes (e.g., dbh, species, crown radii). Import this dataset using the ADD DATA => ADD XY tabs in the FILE menu. Convert to a shapefile.

2. Use the Buffer tool (in the Proximity toolset within the Analysis toolbox) to create a buffer of distance t/2, one half the intertree distance, around each point. This quantity t/2 is meant to approximate the crown radius of a "typical" overstory tree. Set the Dissolve Type option to ALL, which dissolves overlapping buffers, creating a reduced set of spatially non-overlapping polygons stored as a multipart polygon feature.

3. Apply the Multipart to Singlepart tool (in the Features toolset within the Data Management toolbox) to the output from step 2. This step assigns a unique ID to each polygon. The output of this step is a set of multiple polygon features that represent the tree clusters present at a given intertree distance, t.

4. Use the Intersect tool (located in the Overlay toolset within the Analysis toolbox), intersect the output of step 3 with the original point feature shapefile (the stem map data in step 1). This produces a table listing associations between individual trees and the unique clusters they form at the intertree distance, t.

5. The attribute table of the shapefile produced from step 4 can be summarized in terms of the cluster size distribution, number of single trees, etc. Sanchez Meador et al. (2011) provide some useful examples of how clump attributes can be summarized (area, mean dbh, species composition, etc.).

6. Repeat steps 1-5 across a range of d (e.g. 1-20 m) to explore how the number and attributes of tree clumps and single trees vary as a function of distance (t).

Note: The method described here can be modified to use measured or modeled crown radii for each tree in place of t/2 in Step 2.