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Understory Vegetation Responses to Initial Thinning of Douglas-fir Plantations Undergoing Conversion to Uneven-Age Management

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ABSTRACT

Since the mid-1990's, several studies have been undertaken to evaluate thinning treatments designed to promote development of complex forest structure in order to enhance ecological functioning and biological diversity. Most of these studies have focused on forest responses to one or possibly two thinning entries. In contrast, this study, the Uneven-aged Management Project (UAMP), is evaluating alternative silvicultural regimes for converting young Douglas-fir stands to mixed-species, uneven-aged condition. Conversion is to be achieved through regulation of stocking with repeated thinnings over an indefinite period.

The relatively low intensity thinning treatments applied as a first entry in the conversion process had little impact on the abundance, size, or diversity of understory vegetation. Disturbance resulted in short-term decreases in understory vegetation cover, particularly tall shrubs. However, within five years of treatment, understory vegetation abundance returned to approximate pretreatment condition. Regardless of treatment, shrubs and ferns dominated the understory with coverage that was two-three times that of forbs and grasses. Species richness averaged near 12 species per 0.1-ha plot before treatment and five years post treatment. Community composition was dominated by a few very abundant species regardless of treatment. Post-thinning increases in conifer regeneration were consistent with the density of underplanting. Substantial increases in tree regeneration occurred in the light thinning treatment due to an increased seedling density of the shade-tolerant bigleaf maple. The persistence of planted and natural seedlings of less shade-tolerant species remains to be determined.

The general lack of understory vegetation response to the thinning treatments was likely due to the inherent resistance and resilience of the plant communities to disturbance, as well as the low intensity of disturbance attributable to the treatments. Over time, repeated frequent thinning entries may challenge this apparent community stability.

Keywords: thinning, uneven-aged management, regeneration, structural diversity, species diversity, douglas-fir.

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INTRODUCTION

Thinning and uneven-aged management in douglas-fir

The Northwest Forest Plan established a management goal to enhance the development of late-successional characteristics in young second-growth forests on a large portion of federal lands in western Oregon and Washington. Currently there are several hundred thousand hectares of 40-to-80 year-old plantation forests on federal land (Cissel et al. 2006). These predominantly Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests have developed following large-scale disturbances associated with stand-replacing fires in the late-1800s and early-1900s and subsequent regeneration harvests in the 1950s through the 1980s. They were managed for timber production as a primary objective. Slash and other woody debris was removed prior to reforestation and subsequently competing vegetation was controlled and stands were pre-commercially thinned to enhance survival and growth of conifer regeneration. As a result of these silvicultural practices, relatively uniform, single-species high density stands have developed. These young forests typically lack the legacy features and the structural and biological complexities characteristic of late-successional forests and are poorly suited to supporting the northern spotted owl (*Strix occidentalis cuarina* (Merriam)), and other wildlife species (Carey 1995, Lindermayer and Franklin 2002).

Without intervention, these young Douglas-fir forests may remain in the stem exclusion phase of stand development (Oliver and Larson 1996) for an extended period (Andrews et al. 2005) and therefore delay the onset of mortality process required to create canopy openings conducive to development of understory vegetation and large, complex tree crowns associated with late-successional forests (Franklin and Van Pelt 2004). Retrospective analyses have indicated that many of the existing old-aged stands tended to be less dense during their early stages of stand development and therefore individual trees grew faster, maintained deeper crowns, and developed larger limbs (Tappeiner et al. 1997). Existing old-growth stands contain larger amounts of large woody debris than is present in current young forests. The less dense, more heterogeneous late-successional forests tended to develop multiple cohorts of trees lending vertical connectivity between understory shrub layers and the predominant tree canopy. Finally, natural disturbances and subsequent stand dynamics resulted in a small proportion of trees that developed into dominants extending above the general canopy. Combined, these features of late-successional forests represent a more structurally diverse ecosystem than current young-growth Douglas-fir forests and they provide a suite of ecological and social values that differs from that of young forests.

A variety of silvicultural treatments may be undertaken in young Douglas-fir forests to enhance development of compositional and structural features associated with older-aged forests. Use of thinning to respace young stands is being widely explored both by forest managers and researchers (DeBell et al. 1997, Tappeiner et al. 2002). Variable retention thinning coupled with gap creation and leave islands have has been used to increase spatial heterogeneity within stands (Cissel et al. 2006). Mechanical treatments and biological agents have been explored for creating snags. Understory planting of conifers and hardwoods has been used to establish additional cohorts of trees (Chan et al. 2006, Maas-Hebner et al. 2005).

Although many studies have been undertaken to evaluate thinning treatments designed to promote forest diversity, uneven-aged management, which maintains continuous forest cover and diverse forest structure over time, has not been extensively investigated in the Pacific Northwest, particularly in young Douglas-fir forests (Emmingham 1998, Emmingham et al. 2007). One early trial conducted in the 1930s concluded that selective cutting (individual tree and small group selection) applied to mature Douglas-fir was a failure (Isaac 1956), as regeneration of Douglas-fir was not adequately attained and there was a loss of growth and volume production in the residual stand, much of this loss associated with damage (Curtis 1998, Isaac 1956). From a retrospective assessment of 50-80 year-old Douglas-fir stands managed by selection thinning, Miller and Emmingham (2001) observed that although the volume growth of the residual stand was less than that for a more typical low thinning, growth per unit growing stock was similar and in some cases the residual overstory density was low enough to result in regeneration of the moderately shade-intolerant Douglas-fir.

With an increased emphasis on ecological function and biodiversity as management objectives on some forested lands, responses of understory vegetation to overstory thinning has become a recent focus of research (Muir et al. 2004; Thomas et al. 1999; Thysell and Carey; Wilson and Puettmann 2007). One common tenet is that increased diversity of structure and composition of plant communities will lead to increases in faunal components of biodiversity (Lindh and Muir 2004). Wildlife respond to habitat features at several hierarchical scales reflecting their respective ranging, foraging, sheltering and social traits (Wilson and Puttmann 2007). Thinning in forest stands alters the physical environment in terms of canopy openness, and therefore visibility, access, and cover (Carey 1996), as well as light transmittance and energy balance. In addition to direct impacts on habitat suitability, these changes in physical site and associated resource availabilities may be drivers in the development of understory plant communities important to providing food, cover or nesting. A variety of bird and mammal fauna in the Pacific Northwest have been shown to require a variety of canopy and understory vegetation conditions with preferences differing among species (Hayes et. al. 1997, Hagar et.al. 2004). As a result of increased structural and compositional variation at the stand scale, forests may provide a greater range of functions for a greater number of organisms.

Objectives

The objective of this study was to determine the efficacy of three alternative thinning treatments for increasing the abundance and diversity of understory vegetation in 30-40 year-old, even-aged, Douglas-fir plantations at mid-elevation on the west-slope of the Cascade Range, central Oregon.

Specifically, the following questions related to stand development occurring during the five-year period following the initial thinning entry in the conversion process are addressed:

1) Did the different thinning treatments result in substantially different residual stand densities and canopy covers?

2) Were there differential responses to the thinning treatments in the abundance and diversity of understory vegetation?

3) Did the thinning treatments result in initiation of a new cohort of trees?

METHODS

Study location

The study was conducted on the H.J. Andrews Experimental Forest, a 6,350 ha 6th-field watershed on the Willamette National Forest, near Blue River, Oregon. The H.J. Andrews is located within the western Cascades physiographic province (Franklin and Dyrness 1988). Spanning elevations from 440 to 1090 m, the H.J. Andrews is forested primarily by Douglas-fir, western hemlock (***Tsuga heterophylla*** (Raf.) Sarg.) and western red cedar (***Thuja plicata*** Donn ***ex*** D. Don) at lower elevations and transitions to a mix of noble fir (***Abies procera*** Rehd.), Pacific silver fir (***Abies amabilis*** (Dougl.) Forbes), Douglas-fir and western hemlock at higher elevations. Climate is maritime with mild, wet winters and warm, dry summers. Average precipitation totals about 2,300 mm but only about 6% of that occurs in the summer months.

Sixteen stands were selected from all stands on the H.J. Andrews that were 30-40 years of age, a minimum of 8 ha in area, a mid-point elevation less than 915 m, and having no other ongoing experimental activity. The experimental stands actually ranged from 7 to 14 ha in area, 540 to 920 m elevation, 13 to 45 percent slope, 208 to 272 cm annual precipitation and 26.5 to 40.2 m site index (King 1962). Soils were generally gravelly loams or gravelly clay loams of the McKenzie River, Frissell or Carpenter series. Although Douglas-fir dominated, western hemlock, western redcedar and Pacific yew (***Taxus brevifolia*** Nutt.) co-occurred in lesser abundance. Predominant plant associations, based on potential natural vegetation (McCain and Diaz 2002; Hawk 2005) included TSHE/RHMA/GASH (western hemlock/rhododendron/salal), TSHE/RHMA/MANE2-GASH (western hemlock/rhododendron/Oregon grape-salal) and TSHE/POMU (western hemlock/sword fern). In total, the study encompassed about 140 ha of approximately 611 ha of similarly aged Douglas-fir stands on the experimental forest.

The stands originated following clearcut harvest between 1950 and 1960. Two to five years following harvest and prescribed burning, the units were either aerial seeded, planted, or seeded and planted with Douglas-fir. Most of the stands were precommercially thinned in the late 1960s and early 1970s.

Experimental treatments

The experiment consisted of three thinning treatments, each applied as the first entry in a prescribed multiple entry conversion regime, and an unthinned reference treatment. The three thinning treatments included a light thinning, a light thinning with small gaps, and a heavy thinning. Due to initial variation in average tree size and density among stands, all prescriptions were based on a relativized measure of density – Relative Density (RD, Curtis 1982) – calculated for trees 12.7 cm d.b.h. and larger. The three thinning treatments are summarized below:

Light thinning consisted of a uniform thinning of stands from below toward a target density of RD 4.3 (RD 30 imperial). Stands were underplanted with approximately 550 seedlings per hectare using an even mixture of Douglas-fir, western hemlock, western redcedar, and western white pine (*Pinus monticola* Dougl.). When stands reach RD 7.2 (RD 50 imperial), they will be rethinned to RD 4.3 and thinning will occur in all diameter classes. Re-entry intervals of about 10 years are expected.

The group selection treatment consisted of creating 0.05 to 0.1 ha gaps over 10 percent of the stand and thinning the remainder toward a target density of RD 4.3 as in the light thinning. Stands in this treatment were underplanted throughout as described for the light thinning. Subsequent entries will create additional gaps, 10 percent of the area with each entry and will be triggered by the attainment of RD 7.2 in the thinned matrix of the stands. With each entry, expected to be at approximately ten year intervals, gap size will increase as a function stand height.

The heavy thinning treatment consisted of a thinning from below toward a target density of RD 2.9 (RD 20 imperial) and underplanting. A second thinning is prescribed when the combined RD of the residual overstory and the newly established regeneration cohort reaches RD 6.1 (RD 40 imperial). The second thinning will decrease the RD of the combined cohorts to RD 2.9. Subsequent entries, expected to be at 15 to 20 year intervals, will be performed to maintain the combined RD of all cohorts between minimum and maximum thresholds of 2.9 and 6.1, respectively.

Although the three treatment prescriptions were formulated to achieve the common goals of increasing structural diversity and enhanced understory vegetation development, they differ in expectations for conifer regeneration. It was expected that the light thinning combined with understory planting would result in nearly continuous recruitment of additional tree cohorts of both planted and natural origin with a shift in species composition to the more shade tolerant western hemlock. The group selection treatment was envisioned to provide areas of canopy opening adequate to recruit the intermediate shade tolerant Douglas-fir, although the gaps themselves may become competitive environments for tree regeneration if they become occupied by shrubs. The heavy thinning was envisioned as an intermediate condition in which more persistent opening of the canopy may provide an opportunity for recruitment of both intermediate and tolerant conifers while limiting the potential for large increases in competing understory shrubs.

The 16 stands were grouped into four blocks of four stands each. The four experimental treatments were assigned at random to the stands within each block. Thus the experimental design was that of a randomized complete block. Stands were harvested between April 1 and December 1, 2000 predominantly by cable yarding systems, although there was a small portion of the study where tractor yarding was done. Underplanting occurred in the spring of 2002.

Vegetation measurements

Vegetation abundance and composition were measured using a nested sampling design. Breast height diameter (d.b.h.) of all live trees ≥5 cm d.b.h. and snags ≥10 cm d.b.h. were measured on a 0.1-ha circular plot. Percent cover by tall shrubs and small trees (<5 cm d.b.h.) was measured by species along two 14.5-m planer transects per 0.1-ha plot. For low shrubs, forbs, ferns and grasses, percent cover by species was ocularly estimated from 0.1-m^2 (0.2 m by 0.5 m) subplots at eight equally spaced locations along each of the two 14.5-m transects. Tree seedlings and saplings (stems <5.0 cm d.b.h.) within two 21.75-m^2 (1.5 m by 14.5 m) strip plots per 0.1-ha plot were tallied by species and height class (0.1 to 0.2 m, 0.21 to 0.5 m, 0.51 to 1.0 m, 1.01 to 2.0 m, 2.01 to 3.0 m, >3.0 m). Canopy cover was estimated using a moosehorn densiometer at plot center and at four points located 10.5 m radial distance from plot center in the cardinal directions.

From five to 13 permanent vegetation sample plots were established in each stand. The target sampling intensity for the unthinned, light thinning and heavy thinning treatments was seven percent of the stand area; for the group selection treatment it was 15 percent of the stand area. Pretreatment measurements were made in the summer of 1997 or 1998, and post-treatment measurements were made in the summers of 2001, 2003, and 2005.

Statistical analyses

Overstory tree, canopy cover and understory vegetation data were analyzed by mixed model analysis of variance. Thinning treatment was a fixed effect and block was a random effect. Pretreatment data were analyzed independent of post-treatment data. Post-treatment data were analyzed as repeated measures with block nested within treatment being the repeated subject and using autoregressive or unstructured covariance as dictated by individual data sets. Given the low number of replications in the operational treatment implementation, a probability threshold of $p=0.1$ was used to determine statistical significance.

Vegetation abundance response to treatment was analyzed by four growth-form strata (small trees, tall shrubs, low shrubs, and forbs, ferns and grasses). Seedling and sapling density data were collapsed from six height classes to four (0.1 to 0.2 m, 0.21 to 0.5 m, 0.5 to 2.0 m, and >2.0 m) and were log transformed prior to analysis. Among repeat measurement events, there were apparent crew-biased differences in ocular cover estimates for low shrubs, forbs, ferns and grasses derived from the 0.1-m^2 subplots. Therefore, for analyses of these data, mean percent cover for each treatment was relativized to that of the unthinned control. Specifically, for each repeat measurement and block, the differences in mean covers between each thinning treatment and the unthinned control were calculated and those differences were used as data for the repeated measures analysis of variance. In effect this relaxed the broad assumption that all crews had the same accuracy of ocular estimation; rather, it assumed that in any given year of measurement and for any block, the accuracy of estimation was the same for all treatments

Understory vegetation diversity response to treatment was assessed for the combined forb, fern, grass, low shrub, tall shrub and small tree lifeforms. Due to identification uncertainties, some putative species of the same genus were collectively recognized as a representing a single taxonomic entity (eg. two putative species of the genus ***Galium***). For convenience, we include these aggregated taxa in our use of the term species. Diversity was quantified as the total number of species (N_0) and the effective numbers of species (N_1,N_2), the latter two being abundance weighted measures interpretable as the number of abundant and very abundant species present, respectively (Hill 1973, Ludwig and Reynolds 1988). Evenness, the degree to which species comprising a community are equally represented, was estimated as the ratio of the numbers of very abundant to abundant species (N_2/N_1; Hill 1973). Indices of species diversity and evenness were analyzed by repeated measures analysis of variance as described above.

The relationship between overstory canopy cover and relative density by live trees >12.7 cm d.b.h. was modeled by fitting a second-order polynomial regression to stand averaged data from all treatments and measurement events. Degree of association between overstory cover and tree seedling density for each measurement period was estimated by Pearson correlation.

Analyses of variance, regression modeling, and correlation analyses were performed using Proc Mixed, SAS© v. 9.0 (SAS Institute, Inc., Carey, NC).

RESULTS

Overstory

Prior to treatment, stand density, basal area and average diameter breast height ranged among treatment units from 490 to 915 tph, 29 to 40 m^2/ha, and 21 to 29 cm, respectively. Stands in the unthinned treatment had slightly greater average pretreatment density and lesser average tree diameter than stands receiving the three thinning treatments (table 1) however, differences among treatment means were not statistically significant. Relative density (Curtis 1982) calculated on a metric basis ranged among the sixteen stands from 5.7 to 7.6 (RD 40 to 53 imperial). Pretreatment relative density averaged across stands within treatments (table 1) did not differ among treatments.

Initial post-treatment measurements were made in summer, 2001, the first growing season following harvest. Density, basal area and relative density of the unthinned stands increased slightly from pretreatment (table 1). Basal area of the light thinning, group selection and heavy thinning treatment were decreased by harvest an average of 22, 21, and 33 percent, respectively. Although the residual

Table 1. Mean (standard deviation) pretreatment (1998) and posttreatment (2001) stand conditions for trees greater than 12.7 cm dbh for each experimental treatment.

Treatment		DBH	Basal area	Density	Curtis (1982) Relative density
		Centimeters	Square meters per hectare	Trees per hectare	
Unthinned	Pretreatment	25	33	682	6.6
		(3.8)	(1.8)	(190)	(0.55)
	Posttreatment	25	37	710	7.2
		(4.0)	(1.5)	(198)	(0.55)
Group selection	Pretreatment	29	38	550	6.9
		(0.9)	(2.3)	(64)	(0.51)
	Posttreatment	31	30	391	5.4
		(1.4)	(2.9)	(20)	(0.43)
Heavy Thinning	Pretreatment	26	33	566	6.3
		(1.4)	(3.3)	(45)	(0.50)
	Posttreatment	29	22	311	4.0
		(1.8)	(4.3)	(52)	(0.71)
Light thinning	Pretreatment	27	36	578	6.8
		(2.1)	(2.6)	(72)	(0.43)
	Posttreatment	30	28	371	5.0
		(3.7)	(2.8)	(70)	(0.48)

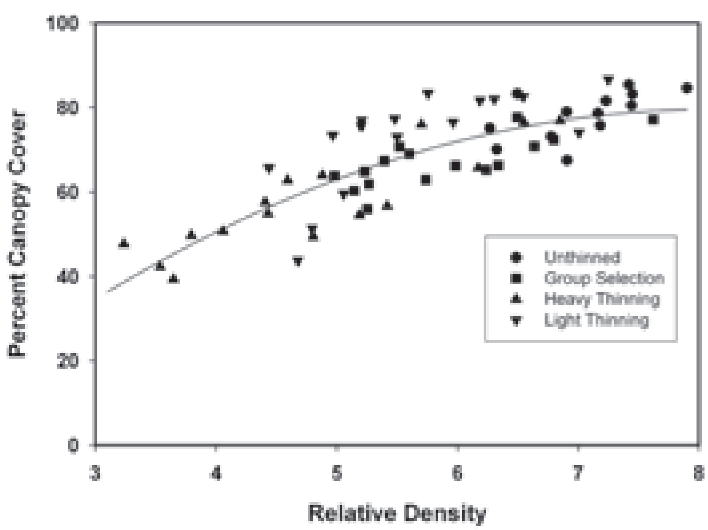

Figure 1—Percent canopy cover as a function of relative density.

relative density and stocking of the heavy thinning treatment was less than that of the light thinning and group selection treatments as intended, the residual relative densities exceeded prescribed targets for all three treatments (table 1). The negligible post-treatment difference in stocking between the light thinning and group selection treatments is not entirely surprising given that small groups were prescribed to cover ten percent of the group selection stands, with the remainder thinned similar to the light thinning treatment. The variation in pre-treatment stand conditions would make such precise treatment differences difficult to achieve in practice.

Canopy cover

Pretreatment mean percent overstory canopy cover ranged from 66 to 87 for the sixteen stands. Initially following harvest, mean percent cover for the group selection, light thinning and heavy thinning treatments was 63, 62, and 48, respectively, while percent cover of unthinned stands averaged 77. Based on stand averaged values, percent canopy cover was positively correlated with RD throughout the1998 to 2005 monitoring period (fig. 1). Based on a second-order polynomial regression, RD accounted for 68 percent of the variation in percent canopy cover ($p<0.0001$).

From 2001 to 2005 canopy cover increased on average 1.1, 2.9 and 2.0 percent per year for the group selection, light thinning, and heavy thinning treatments, respectively. In contrast, changes in canopy cover in unthinned stands were not detected. Although, the group selection and light thinning treatments had similar

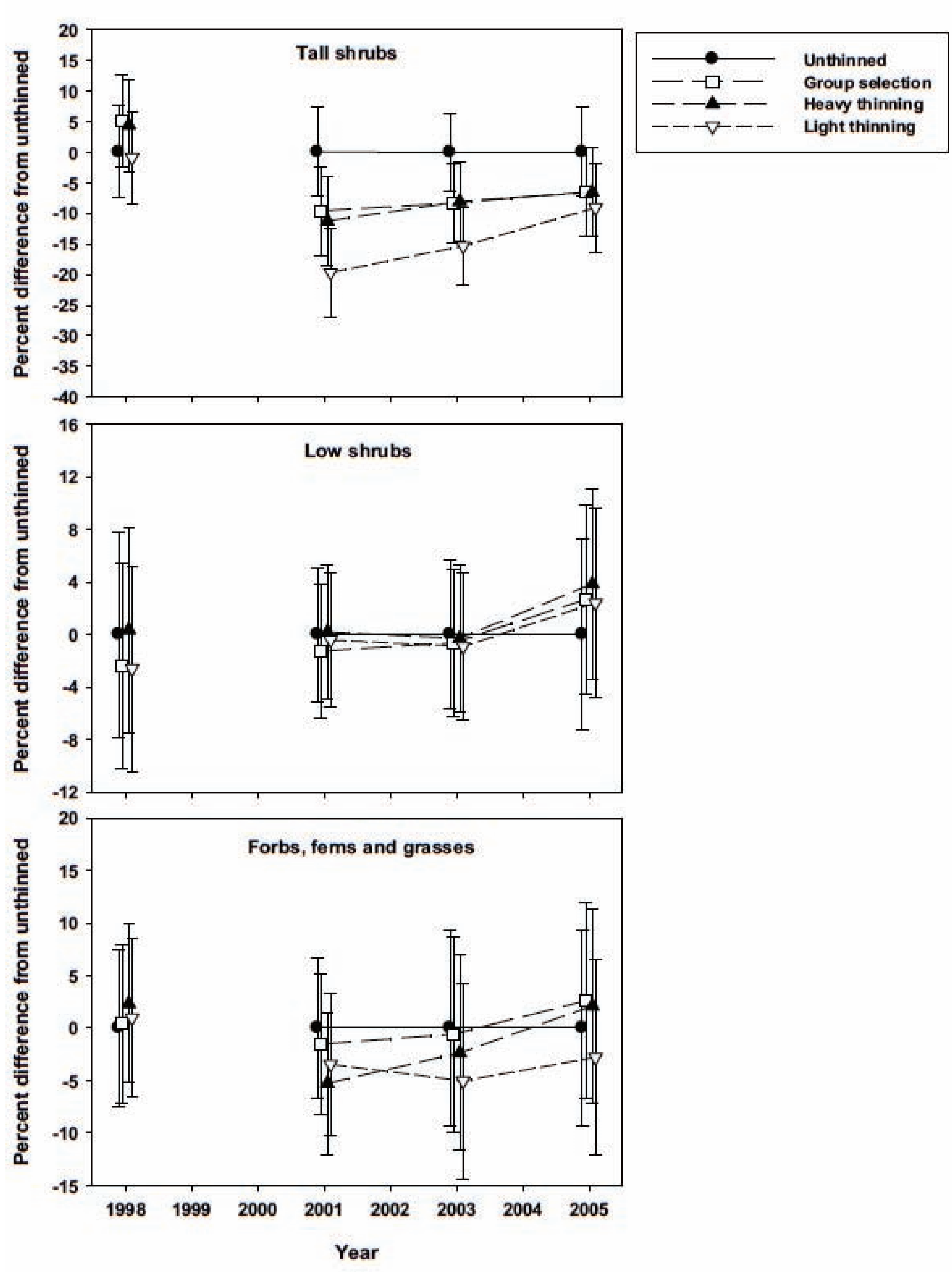

Figure 2 — Cover of tall shrubs, low shrubs, and forbs, ferns and grass vegetation strata by treatment expressed as a difference from the unthinned treatment. Error bars represent one standard error about the mean of n=four replications.

average residual densities and stocking, the lesser rate of canopy expansion in the group selection treatments is likely the effect of the 0.05- to 0.1-ha group openings, which are included in the stand averaged canopy cover estimates.

Understory vegetation

Over the study, mean cover by tall shrubs in the unthinned treatment averaged 36.9 percent. Relative to the unthinned treatment, percent cover by tall shrub species was decreased approximately 11 percent in the group selection and heavy thinning treatments and by 21 percent in the light thinning treatment at the first post-treatment measurement (fig.2). By 2005, significant growth (p=0.058) resulted in treated stands having tall shrub cover that was approximately four to nine percent less than the unthinned treatment suggesting a substantial recovery over the four-year period. Differences among thinning treatments in the rates of recovery or overall relative abundance were not statistically significant.

In contrast to tall shrubs, cover by low shrub species was unchanged by the harvest activity as there was no apparent cover decrease relative to the unthinned stand (fig. 2). Over the study low shrub cover in unthinned stands averaged 17.9 percent. Between 2003 and 2005 there was an apparent three to five percent increase in low shrub cover by all of the thinned treatments, but the increase was not was not statistically significant.

Forbs, ferns, and grasses provided an average of 19.8 percent cover in the unthinned treatment. Cover by these taxa was decreased slightly during harvest as treatment means for the group selection, light, and heavy thinning treatments were approximately two, four, and seven percent less than that of the unthinned treatment in 2001 (fig.2). Subsequently, cover in the thinning treatments increased five to ten percent (p=0.002) between 2001 and 2005 resulting in little difference in cover between thinned and unthinned stands.

The extent to which understory vegetation was dominated by the shrub and fern component is further evident from species-level estimates of relative cover and frequency of occurrence. Shrub species that occurred with both high

Table 2. Percent relative cover (RC) and frequency (Freq) for abundant or common tree, shrub, fern and herb species prior to treatment and five-years post-treatment. Relative cover is the percent cover of a species relative to total vegetation cover averaged over those plots on which a species occurred. Frequency is the percentage of all plots on which the species occurred.

		Pre-treatment (1997-1998)								Five years post-treatment (2005)							
		Unthinned		Group		Heavy Thin		Light Thin		Unthinned		Group		Heavy Thin		Light Thin	
Species	Life-form	RC	Freq	RC	Freq	RC	Freq	RC	Freq	RC	Freq	RC	Freq	RC	Freq	RC	Freq
Douglas-fir	Tree	5.5	9.1	6.2	12.5	6.4	19.4	7.4	9.1	3.1	13.6	2.8	7.1	16.8	2.8	1.4	4.2
Pacific yew	Tree	7.7	45.5	11.4	19.6	8.0	19.4	8.8	59.1	14.6	27.3	5.9	8.9	1.4	5.6	9.1	54.2
Western redcedar	Tree	4.4	27.3	7.1	33.9	7.1	11.1	12.3	54.5	6.1	50.0	9.5	19.6	3.9	8.3	10.9	50.0
Western hemlock	Tree	9.3	50.0	8.8	48.2	6.9	52.8	6.4	54.5	12.4	59.1	9.4	42.9	11.8	38.9	12.2	50.0
Oregon grape	Shrub	18.1	86.4	13.8	76.8	10.9	72.2	18.3	95.5	15.0	90.9	15.1	83.9	11.4	72.2	19.8	95.8
Pacific blackberry	Shrub	3.4	86.4	5.6	92.9	10.0	94.4	4.7	90.9	1.6	86.4	6.5	94.6	12.3	97.2	4.3	87.5
Red huckleberry	Shrub	3.4	36.4	4.0	28.6	3.3	38.9	6.9	72.7	3.6	31.8	2.0	21.4	2.9	30.6	6.7	54.2
Rhodendron	Shrub	17.2	36.4	14.0	44.6	15.2	30.6	9.5	50.0	17.5	36.4	13.2	48.2	14.1	25.0	9.8	45.8
Salal	Shrub	11.6	54.5	11.8	60.7	11.9	63.9	6.8	90.9	9.9	54.5	12.2	66.1	13.6	61.1	9.2	75.0
Vine Maple	Shrub	31.8	86.4	35.2	94.6	36.0	80.6	25.2	81.8	31.3	90.9	27.1	82.1	32.7	75.0	23.0	87.5
Sword fern	Fern	16.7	90.9	20.3	82.1	18.2	83.3	17.2	100.0	20.1	90.9	25.9	87.5	20.3	83.3	17.7	100
Bedstraw	Herb	1.2	45.5	0.9	58.9	0.6	61.1	0.6	45.5	0.5	40.9	1.2	67.9	0.6	61.1	1.5	70.8
Goldthread	Herb	0.7	36.4	0.7	23.2	0.7	19.4	1.4	72.7	0.4	36.4	1.0	26.8	1.0	30.6	1.1	83.3
Oregon oxalis	Herb	19.8	27.3	5.9	16.1	0.3	11.1	2.1	9.1	13.2	31.8	8.5	19.6	3.1	5.6	4.1	8.3
Redwoods violet	Herb	0.2	27.3	0.9	25.0	1.8	50.0	1.3	54.5	0.4	22.7	1.1	33.9	1.7	47.2	0.9	62.5
Wall-lettuce	Herb	0.0	0.0	0.6	7.1	0.2	2.8	0.1	4.5	0.0	0.0	0.9	30.4	0.5	22.2	0.9	20.8
Western starflower	Herb	0.9	18.2	0.5	32.1	1.8	36.1	2.4	4.5	0.4	45.5	0.7	58.9	1.3	58.3	0.3	45.8
Western twinflower	Herb	3.3	50.0	3.2	64.3	7.0	58.3	3.2	81.8	1.8	59.1	4.5	66.1	8.9	69.4	2.4	87.5

relative cover (10 to 35%) and frequency (>50%) were Oregongrape (*Mahonia nervosa* Pursh), salal (*Gaultheria shallon* Pursh), and vine maple (*Acer circinatum* Pursh) (table 2). Sword fern (*Polystichum munitum* (Kauf.)Presl.) occurred at relative densities in the range of 15 to 25% at frequencies in excess of 80%. Pacific blackberry (*Rubus ursinus* Cham. & Schlecht.) also occurred at high frequency but provided low relative cover. In contrast, on an individual species basis few herb species occurred at relative covers greater than 5%, although there were species such as goldthread (*Coptis lanciniata* Gray), western twinflower (*Linnea borealis* Torr.), redwoods violet (*Viola sempervirens* Greene), and bedstraw (*Gallium spp*.) that occurred at frequencies in excess of 50% for some treatments (table 2). Although an increase in herbaceous cover might have been expected as a result of harvest-related disturbance, only the early-seral associate wall-lettuce (*Mycelis muralis* (L.)Dumort), demonstrated an apparent increase in frequency (table 2).

Species diversity

Prior to thinning, species richness (N_0) averaged 11.7 species per 0.1-ha plot when averaged across all treatments. In 2001, following treatment, the mean number of species declined somewhat, particularly for the light thinning and heavy thinning treatments (fig. 3). However, species richness increased significantly (p=0.0014) between 2001 and 2005 to return to pretreatment levels (fig. 3). In contrast to richness, the evenness component of diversity did not differ among treatments or vary over time with mean evenness index values being approximately 0.72 to 0.78 (fig. 3). These values indicate a relatively large contribution to community composition by a few species present in high abundance.

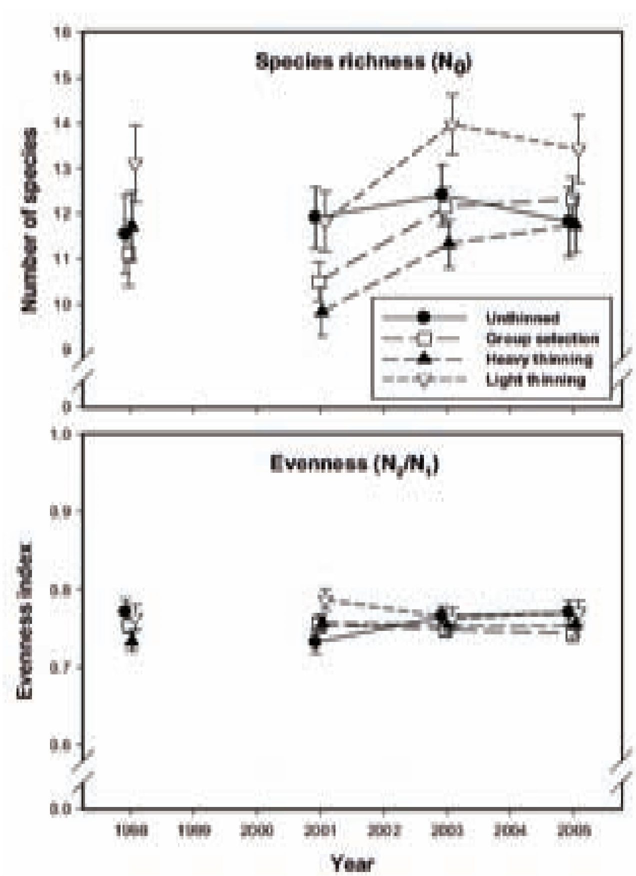

Figure 3— Species richness and evenness by treatment and year. Richness is the mean number of species observed per 0.1-ha plot. Evenness is a measure of the extent to which all species on a plot contribute to total cover. Evenness index values near 0 indicate equal contributions by all species; values near 1 indicate dominance by a few species. Error bars represent one standard error about the mean of n=four replications.

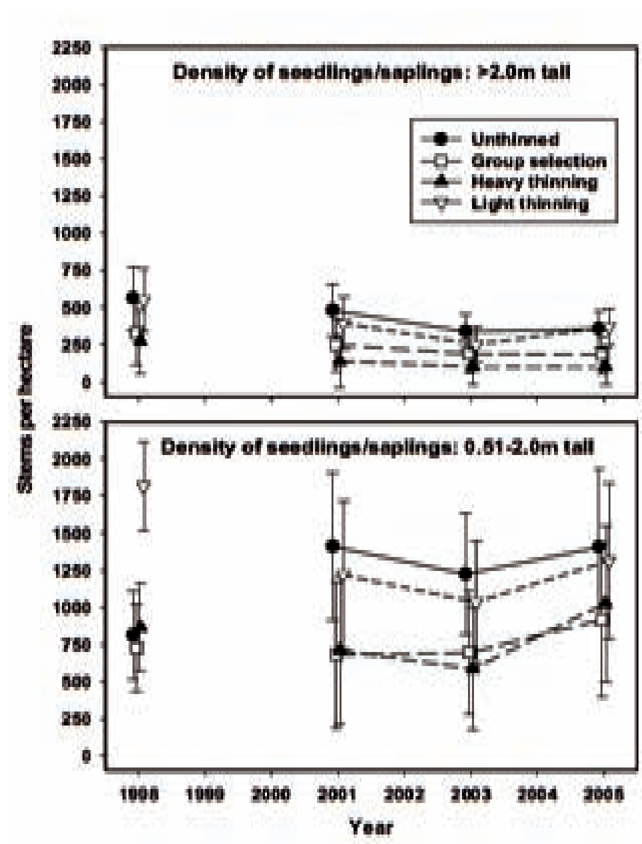

Figure 4 – Density of seedlings and saplings greater than 0.5 m tall. Error bars represent one standard error about the mean of n=four replications.

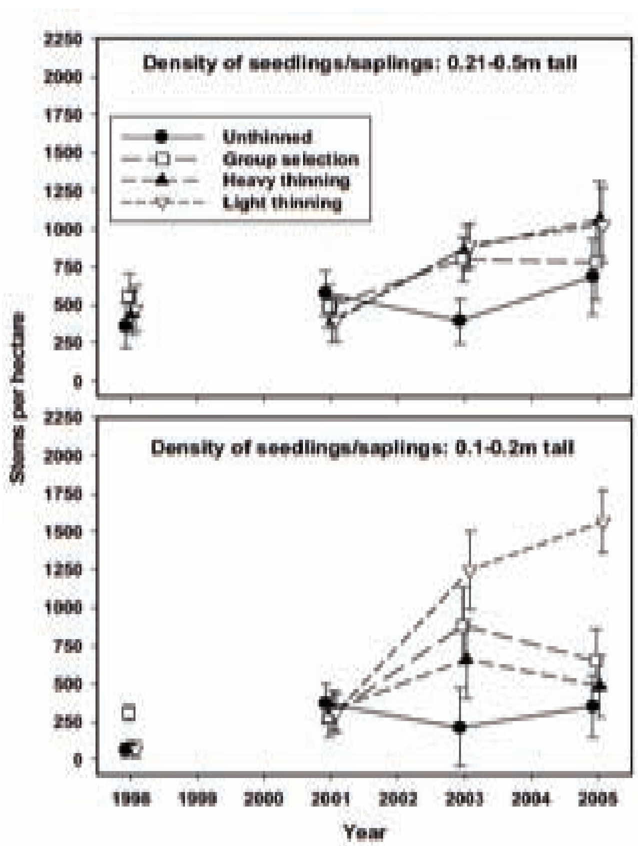

Figure 5— Density of seedlings and saplings from 0.1 to 0.5 m tall. Error bars represent one standard error about the mean of n=four replications.

Tree regeneration

The density of large saplings (trees >2 m tall and <5 cm d.b.h.), averaged over all treatments tended to decline (p=0.060) from about 315 tph to 250 tph over the study. Of the three thinning treatments, only the light thinning demonstrated a tendency for increased large sapling density between 2001 and 2005, as this was the only thinning treatment that demonstrated an obvious decrease in sapling density initially following harvest (fig. 4). This response was consistent with tall shrub cover which also was decreased by harvest activities to the greatest extent in the light thinning regime.

Prior to treatment, densities of saplings 0.51 to 2.0 m tall were more than twice as abundant in the light thinning treatment than in the group selection, heavy thinning and unthinned treatments (p=0.015). However, following harvest the abundance of saplings was diminished to amounts similar to the other treatments (fig. 4). Furthermore, there was a large variation about the treatment mean values and differences among treatments or over time were not detected.

Densities of seedlings in the 0.1 to 0.2 m and the 0.21 to 0.5 m height classes tended to increase between 2001 and 2005, with the extent of increase differing among treatments (time by treatment interactions, p=0.0004 and p=0.0006 for the two respective classes). A minimum increase in seedling density of approximately 500 tph is expected for the three thinning treatments given that the stands were underplanted in 2002 and seedling stock would be nearly 0.2 m tall when planted. Although there was no effort to explicitly monitor planted seedlings or to distinguish planted seedlings from natural regeneration, the expected contribution of underplanted conifers to seedling density is consistent with the data in figure 5.

In addition to underplanted seedlings, there was also evident a large increase in the density of 0.1 to 0.2 m seedlings in the lightly thinned treatment (fig. 5). Although the data are not shown here, the increase in natural seedling density for that treatment was due predominantly to the occurrence of bigleaf maple (*Acer macrophyllum* Pursh).

In general the range of average abundances by year and treatment of western hemlock (650 to 1200 tph) and western red cedar (115 to 800 tph) also indicated recruitment of natural regeneration, however, the occurrence was variable and treatment effects were not statistically evident. While Douglas-fir was represented in all of the seedling and sapling height classes, its abundance (90 to 320 tph) was low relative to the more shade tolerant western redcedar and western hemlock. Among tree species with densities significantly correlated to canopy cover or relative density, western hemlock, western redcedar, and Pacific yew were positively related and Douglas-fir was negatively related, as expected given existing knowledge of species shade tolerance.

DISCUSSION

Of the three questions addressed, the most certain result was that the thinning prescriptions resulted in a variety stand conditions with a distinct separation in terms of density and canopy cover occurring between the unthinned stands and the heavily thinned stands. However, it is important to note that the thinning treatments imposed in this study are of relatively low intensity in terms of both the proportion of basal area removed and the size of gaps created in the group selection treatment (see Figure 3, Poage and Anderson 2007). For example, a study conducted in western Oregon on the Siuslaw National Forest evaluated thinning treatments having basal area removals of 51 to 86% and residual Curtis' relative densities ranging from 1.2 to 4.0 (Chan et al. 2006). The BLM Density Management Studies, also being conducted in western Oregon examined thinning treatments that initially removed 35 to 65% of the basal area in a thinned matrix and incorporated 0.1 to 0.25 ha leave islands and gaps over as much as 40% of the stand area (Cissell et al. 2006).

Although, the treatments resulted in discernable differences in density and cover, particularly between the unthinned and heavy thinning treatments, there was little evidence of substantial alterations of understory shrub and herbaceous vegetation. This lack of strong understory vegetation response in terms of composition, abundance, or size is consistent with several studies of thinning in Douglas-fir. In a recent review of seven operational-scale silviculture experiments, Wilson and Puettmann (2007) report that percent cover by shrubs and percent cover by herbs, one to seven years following thinning showed little difference across a wide range of residual basal area. In contrast, the retrospective studies of Baily et al. (1998) and Suzuki and Hayes (2003) showed in some cases large increases in shrub cover ten to 30 years after thinning. Short-term vegetation responses likely reflect the direct influences of disturbance, such as harvest damage, and transitory shifts in resource availability, whereas a longer period is required for altered canopy structure to have an impact on understory vegetation (Lindh and Muir 2004; Thomas et al. 1999; Wilson and Puettmann 2007).

Halpern et al. (2005), studying young Douglas-fir forests in the western Cascades of Washington, observed that the initial understory vegetation response to overstory removal with green tree retention was an increase in within stand variation in abundance. They hypothesized that this resulted from disturbance associated with harvest and that the influence of canopy structure on understory vegetation composition and structure would be manifest over a longer period of time. However, they also observed that the amount of disturbance was greater with lower levels of green-tree retention (15 percent versus 40 percent retention) and with aggregated retention than with dispersed retention, which contrasts with our observations for greater reductions in cover, particularly for tall shrubs, occurring in the light thinning treatment. These differences were likely due to the overall greater intensity of harvest treatments considered by Halpern et al. However, it may be speculated that logging systems also had a role; light, evenly dispersed thinning may have had less concentration of cable yarding activity. Although fewer trees were removed relative to the heavy thin and group selection treatments, lateral skidding in the light thin treatment may have been more dispersed. As a result the light thin treatment may have impacted a greater proportion of the area.

The question of whether these treatments enhance recruitment of a new cohort of conifer trees is still unclear. Where understory conifers were present in abundance prior to treatment, the density was decreased in harvest. The small post-treatment increase in Douglas-fir seedlings in treated stands can not be attributed to either natural or planted origins because the sampling protocol did not distinguish origin of seedlings. Recent studies by Maas-Hebner et al. (2005) and Chan et al. (2006) have demonstrated in thinned stands of the highly productive Coast Range forests of Oregon that underplanted Douglas-fir and western hemlock will have high rates of survival through eight years following thinning and planting across a broad range of residual stand densities. However, they observed that underplanted western hemlock seedlings generally grew faster and developed better height:diameter ratios than did underplanted Douglas-fir seedlings at all residual densities. Given the relatively high residual overstory densities and the relative abundance of regenerating shade-tolerant conifers and hardwoods in this study, the initial results suggest that recruitment of Douglas-fir regeneration will occur in association with other tree species, rather than as a species-dominant cohort.

The rapid recovery of diversity following disturbance, as indicated in this study by the diversity metrics of Hill (1975), is a response consistent with other studies in managed forests in the western hemlock zone of western Washington and Oregon (Halpern 1989, Halpern and Spies 2005; Thomas et al. 1999; Thysell and Carey 2001). The numbers of species, particularly the total (N_0) are somewhat lower than that reported for stands of similar age in the western Cascades of Washington, but are within the range observed (Halpern and Spies 1995). The rhododendron-salal and rhododendron-Oregon grape plant associations comprise the potential natural vegetation community in 14 of the 16 stands (Hawk 2005). Of six plant associations assessed within the H.J. Andrews, these two plant associations ranked first and second both in resistance to disturbance and in resilience to disturbance (Halpern 1988). The predominance of a shrub component that may confer the attributes of resilience and resistance, may also limit the development of an abundant and diverse herbaceous stratum (Halpern and Spies, 1995; Huffman et al. 1994, Deal 2001, Deal and Tappeiner 2002).

CONCLUSIONS

The treatments imposed in this study produced a range of stand structure conditions. Although stand conditions differed among treatments, the initial understory vegetation responses were limited to a short term decrease in the abundance of tall shrub and saplings greater than one meter height. This decrease was greatest in the lightly thinned unit, possibly because of a greater dispersion of yarding disturbance in the predominantly cable-logged harvest. The shrub-dominated plant communities observed in this study are characteristically resistant and resilient to disturbance, and are not likely to demonstrate significant, long term shifts in abundance or composition with a single harvest entry within the range of treatment intensity evaluated. The long term implications of repeated entry at relatively frequent ten to 20 year intervals, as prescribed, may challenge the resistance and resilience of these understory communities. Although planted conifers have survived through five-years following thinning treatment, the persistence and growth of planted seedlings, particularly the intermediate shade-intolerant Douglas-fir is yet to be determined, To address these long-tern questions, the experimental prescriptions should be followed and the stands monitored for several decades into the future

ACKNOWLEDGMENTS

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Sustaining Northern Red Oak Forests: Managing Oak From Regeneration to Canopy Dominance in Mature Stands

Daniel C. Dey, Gary W. Miller, and John M. Kabrick

ABSTRACT

Across the range of northern red oak, managers have problems sustaining current stocking of northern red oak in forests. Oak species are adapted to frequent stand disturbances that reduce the abundance of shade tolerant competitors and control fast-growing pioneer species. A widely recommended approach to regenerating northern red oak is to develop relatively large advance reproduction by manipulating stand structure to provide adequate light to oak seedlings in the forest understory. One approach to develop large oak advance reproduction is to use the shelterwood method with woody competition control. Herbicides, mechanical cutting and fire can be used to manage competing woody vegetation. Underplanting shelterwoods with oak seedlings can be used to supplement natural reproduction. Where landowners desire to maintain some presence of mature forests at all times, alternative approaches such as the group selection method or shelterwood with long-term retention of a low density overstory may be used to promote oak regeneration and recruitment into the overstory. Once stand regeneration has been initiated, maintaining the desired level of oak stocking in dominant and codominant crown positions can be accomplished through crop tree thinning in young stands. Sustaining oak stocking requires a regime of managed disturbances that begin before stand initiation and extend through the first 20 to 30 years of stand development.

Keywords: northern red oak, silviculture, ecology, regeneration, thinning, stand development.

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INTRODUCTION

Northern red oak (*Quercus rubra* L.) is highly valued for its timber and acorn production. It is widely distributed throughout eastern North America occurring in a variety of ecoregions (Sander 1990). Northern red oak occurs on over 16.9 million ha in the eastern United States (Miles 2007), and successional replacement of this species across the landscape would be a major ecological and economic loss. Its current abundance and distribution is linked with the history of fire and land use during the past 200± years (Abrams 1992, Dey 2002, Dey and Guyette 2000, Pyne 1982, Whitney 1994, Williams 1989). Historic disturbances were important in the expansion and dominance of oaks across a diversity of ecological land types throughout eastern North America (Nowacki and Abrams 1992, Hicks 1998). Modern oak forests originated when logging, fuelwood cutting, charcoal production, woods burning, grazing, and agriculture occurred extensively and frequently in the eastern region, completing a progression toward increasingly greater oak dominance that began when Native Americans brought fire to the region (Pyne 1982, Cronon 1983, Williams 1989, Nowacki and others 1990, Abrams 1992, Whitney 1994, Abrams and McCay 1996).

However, disturbance regimes that once favored oak species have changed. Now, oaks are being displaced by shade tolerant tree species as forest succession progresses in response to small-scale, gap-type disturbances, or by shade tolerant or intolerant tree species following catastrophic natural disturbances or timber harvesting in the absence of frequent fire (Abrams and Nowacki 1992, Hix and Lorimer 1991, Johnson and others 2002, Lorimer 1993, Pallardy and others 1988). Loss of oak species is greatest on high quality sites where oak is competing with high densities of shade tolerant species and, in particular, fast growing species such as yellow-poplar (*Liriodendron tulipifera* L.) (Loftis 1983, Beck and Hooper 1986, Schuler and Miller 1995, Weigel and Johnson 2000, Weigel and Parker 1997). Sustaining current oak stocking levels in mature forests is a common management objective and challenge to forest managers. Loftis (2004) described the two "laws" of oak silviculture as being: (1) the presence of competitive sources of oak reproduction and; (2) disturbances that provide timely and adequate release for those sources. The purpose of this paper is to review the literature on northern red oak physiology, ecology and silviculture and synthesize it to give insight into how to regenerate mature forests and manage forest development to sustain northern red oak stocking.

OAK REGENERATION ECOLOGY AND MANAGEMENT

Sources of Oak Regeneration

It is important to consider the sources of oak reproduction when developing a prescription to regenerate northern red oak. Most oak species regenerate from new seedlings (stem and root system same age), seedling sprouts (stem age younger than root age), and sprouts from the stumps of overstory trees cut during harvesting (Johnson 1993). Oak reproduction present before the regeneration harvest is collectively referred to as advance reproduction, and the ability of trees, from seedlings to mature individuals, to contribute to stand regeneration through sexual or vegetative reproduction is known as regeneration potential (Johnson and others 2002). The goal then is to increase the regeneration potential of northern red oak sources of reproduction to maximize success in oak regeneration after harvesting.

Oak seedling and advance reproduction

Northern red oak produces good to excellent seed crops every two to five years (Sander 1990), and oak regeneration failures are not normally related to matters of seed production or germination (Lorimer 1993). During years of bumper seed crops, northern red oak-dominated forests can produce more than 600,000 acorns per hectare (Auchmoody and others 1993). Even in the low light of mature eastern hardwood forests, which often is below five percent of full sunlight (Lockhart and others 2000, Lorimer and others 1994, Miller and others 2004, Motsinger 2006, Parker and Dey in press), northern red oak acorns provide the energy necessary for first year seedling growth. Subsequent growth is usually hampered by low understory light, and within 10 years of germination most oak seedlings are dead (i.e., < 10 percent survival) (fig. 1) (Beck 1970, Crow 1992, Loftis 1988, 1990a). Any survivors are small in size.

Small (e.g., less than 1.0 cm basal diameter) northern red oak seedlings and seedling sprouts have low regeneration potential, (i.e., low probabilities of assuming dominant or codominant crown positions in the regenerating stand (Johnson and others 2002, Loftis 1990b)). The long-term dominance probability for red oak reproduction decreases

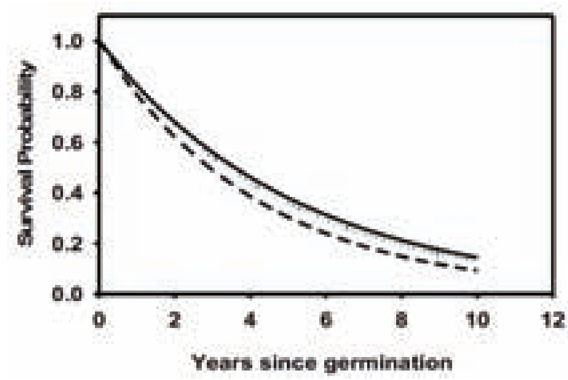

Figure 1— Survival of northern red oak seedlings in the understory of mature mesic forests in the southern Applachians and in northeastern Wisconsin (adapted from Beck 1970 (dashed line), Crow 1992 (dotted line), Loftis 1988 (solid line)).

with increasing site quality because they are out-competed by more rapidly growing species such as yellow-poplar and black cherry (*Prunus serotina* Ehrh.) (Loftis 1990b, Tift and Fajvan 1999, Weigel 1999, Weigel and Johnson 2000). Whether northern red oak reproduction is established naturally or artificially, its regeneration potential increases with increasing initial seedling size (fig. 2) (Johnson 1984, Loftis 1990b, Spetich and others 2002). The key to successful regeneration of northern red oak is to have sufficient numbers of large advance reproduction and potential oak stump sprouts to provide adequate oak stocking at the end of the regeneration period.

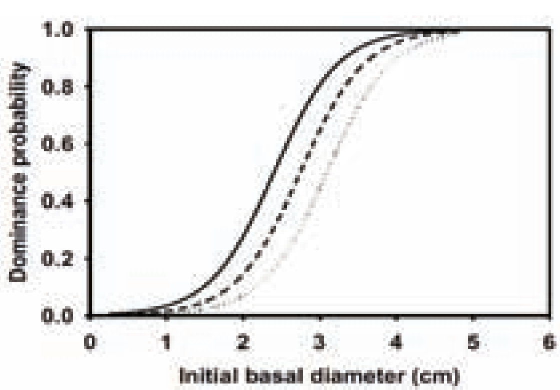

Figure 2— Dominance probability for free-to-grow northern red oak advance reproduction in relation to site quality eight years after clearcutting in the southern Appalachians (adapted from Loftis 1990b). Northern red oak site index 21 m (solid line), 24 m (dashed line), and 27 m (dotted line).

Oak stump sprouts

Another important source of oak reproduction is oak stump sprouts. They are the fastest-growing and most competitive source of oak reproduction (Johnson and others 2002). However, not all oak stems cut in timber harvesting produce sprouts. Thus, current levels of oak stocking in mature forests cannot be sustained by relying on stump sprout regeneration alone, hence the need for and importance of oak advance reproduction. The probability that a northern red oak stump produces a live sprout at the end of the first growing season is highest for smaller diameter trees (e.g., parent tree d.b.h. 10 to 30 cm), decreases with tree age, and increases with increasing site quality (fig. 3)

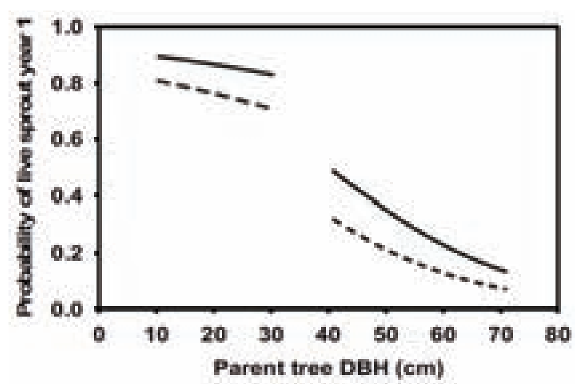

Figure 3— Probability that northern red oak stumps produce a live sprout in relation to initial d.b.h. of the parent tree, tree age and site quality one year after clearcutting in southern Indiana (Weigel and Johnson 1998b). Site index 18 m (dashed line) and 23 m (solid line).

(Weigel and Johnson 1998b). Larger trees have thicker bark that probably reduces sprouting by physically inhibiting development of vegetative buds. Also, older trees have reduced ability to produce sprouts due to physiological senescence. Trees on high quality sites may have improved vigor that initially enhances sprouting capacity.

Longer-term dominance probabilities were reported for northern red oak stump sprouts growing for 15 years in southern Indiana clearcuts by Weigel and others (2006). They defined dominance probability based on the crown class of the most dominant stem in the oak sprout clump. Weigel considered northern red oak sprouts successful, or "dominant" if they were in the dominant or codominant crown class. Northern red oak stump sprout dominance probabilities at stand age 15 decreased with increasing parent tree d.b.h. and age (fig. 4). Further, they estimated

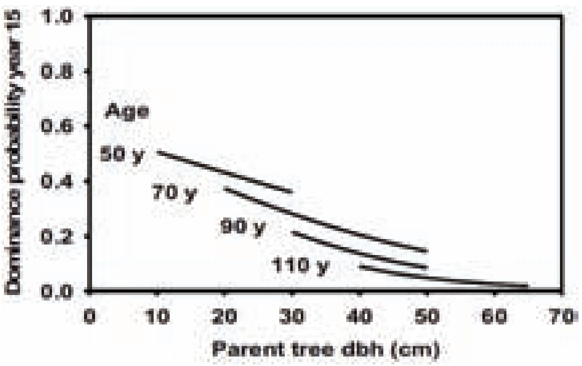

Figure 4— Northern red oak stump sprout dominance probabilities 15 years after clearcutting in southern Indiana (Weigel and others 2006).

conditional dominance probabilities of northern red oak stump sprouts and found that the longer oak stump sprouts persisted in regenerating stands the greater was the likelihood that they would remain dominant at stand age 15 (fig. 5). For northern red oak sprouts that were alive one year after harvesting, the probability they were dominant at age

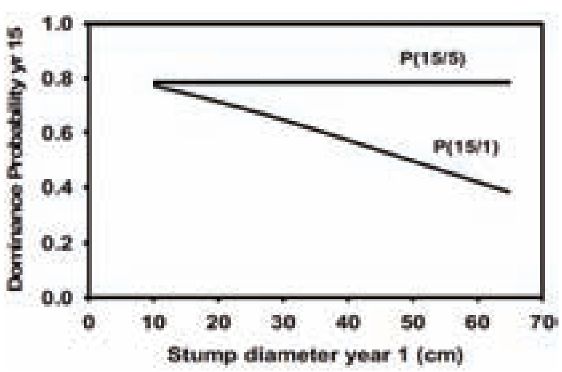

Figure 5— Northern red oak stump sprout conditional probabilities of dominance at stand age 15 in southern Indiana clearcuts – site index 18 m (Weigel and others 2006). Conditional probabilities at stand age 15 are estimated for northern red oak stump sprouts given: (a) they were alive in year 1 (P(15/1)), and (b) given they were alive in year 5 (P(15/5)).

15 decreased with increasing diameter of the parent stump. A sprout on a 10 cm diameter stump had an 80 percent likelihood of persisting as a dominant at age 15, whereas a sprout on a 60 cm diameter stump had only a 40 percent chance of remaining dominant. By stand age 5, if a northern red oak stump sprout was alive, it had a high probability of being dominant at age 15 regardless of the initial size of the parent tree. The less competitive northern red oak stump sprouts are eliminated by stand age 5.

Importance of Light

A primary cause of oak regeneration failure is low light levels (i.e., < five percent of full sunlight) in the understory of fully-stocked, mature forests; light so low that it is not sufficient to meet the respiratory demands and is prohibitive to the development of large oak advance reproduction (Abrams 1992, Gardiner and Yeiser 2006, Hodges and Gardiner 1993, Johnson and others 2002, Lorimer 1993, Lorimer and others 1994, Miller and others 2004). Heavy-shaded forest understories resulted from increasing forest structure due to changes in historical disturbance regimes. Over the past 50 years, the development of shade tolerant understory canopies has followed fire suppression, elimination of woods grazing by livestock, and changes in timber harvest practices (Lorimer 1993).

Northern red oak is intermediate in shade tolerance (Sander 1990). In many forest situations there is insufficient light to support increases in northern red oak seedling root and shoot biomass. Often, understory light is below the light compensation point for northern red oak, which is two to five percent of full sunlight (Gottschalk 1987, Hanson and others 1987). Light saturation of photosynthesis in northern red oak occurs between 20 to 40 percent of full sunlight (Ashton and Berlyn 1994, McGraw and others 1990, Teskey and Shresta 1985). Positive shoot growth in northern red oak seedlings requires more than 20 percent of full sunlight, and seedling diameter and height growth increases with increasing light levels up to 50 to 70 percent of full sunlight (Gottschalk 1994, Musselman and Gatherum 1969, Phares 1971). The physiology of northern red oak natural advance reproduction and planted bareroot seedlings is similar in forest understories. Both types of reproduction respond equally well to increases in light intensity and resource availability following shelterwood harvesting, midstory canopy removal, and other practices that reduce forest stocking and density (Parker and Dey in press).

MANAGING LIGHT IN FORESTS

Midstory Removal

Increasing light for oak advance reproduction in forest understories is accomplished by timber harvesting (e.g., clearcut, shelterwood, group selection), removing midstory or ground-level woody vegetation, or combinations of these two approaches. The density of midstory and understory woody vegetation can be reduced by mechanical cutting, herbicide application or prescribed burning. Stems of any size can be cut with saw or ax. Herbicides can be applied by spraying foliage, basal stem, or cut stump, or by stem injection depending on tree size and chemical used. When properly done, herbicides have the greatest potential to permanently eliminate most woody competitors with one treatment (Miller 1993, Quicke and others 1996, Groninger and others 1998, Kochenderfer and others 2001). Care must be taken in applying herbicides because oaks are susceptible to any chemicals used to control their hardwood competitors. Prescribed burning, which is commonly done in the spring as relatively low intensity surface fires, can reliably girdle or kill trees less than 12 cm d.b.h.; larger trees are able to survive these fires intact, with only minor stem wounding (Barnes and Van Lear 1998, Dey and Hartman 2005, Reich and others 1990, Waldrop and Lloyd 1991). Cutting larger trees followed by burning in subsequent years is one way to reduce stand density and increase light because the shoots of small diameter stump sprouts are very susceptible to low intensity fires. Spring surface fires can kill most acorns that are not buried under the wet duff or in mineral soil, and cause high (e.g., 70 percent) mortality of oak seedlings less than three years old (Auchmoody and Smith 1993, Brose and Van Lear 2004, Johnson 1974). Therefore, burning should be delayed for several years following a good acorn crop.

Whether mechanically cut or killed by fire, stems of smaller-sized (10 to 20 cm d.b.h.) hardwoods often resprout and may need to be controlled again in the future. After harvesting, sprout growth can be great in the high light environment of clearcuts, but increasing amounts of residual overstory trees suppress growth. Hence, there is a negative relationship between overstory shade, hardwood sprout growth, and growth of other competing woody vegetation. The extent of this influence depends on the specie's shade tolerance. On good- to high-quality sites in the southern Appalachians, Loftis (1990b) recommended maintaining an intact main forest canopy while reducing stand basal area by thinning from below to control fast-growing, shade intolerant yellow-poplar while favoring northern red oak advance reproduction. He cautioned that reductions in stand basal area should be decreased (e.g., from 40 percent to 30 percent) as site quality increases (from 21 m to 27 m site index for northern red oak). The challenge is to reduce overstory stocking to increase light to oak while not stimulating the growth of competitors.

Increasing understory light levels by midstory removal has been shown to significantly improve the survival and growth of northern red oak advance reproduction. Lorimer and others (1994) were able to increase understory light from one percent of full sunlight in undisturbed stands to seven to nine percent by reducing the density of the understory woody canopy in Wisconsin mesic oak-northern hardwood forests. They cut the stems of woody species that were taller than 1.5 m and sprayed the stumps with Tordon 101R. This level of light increased the survival and growth of planted northern red oak and white oak seedlings. In the central Appalachians of West Virginia, Miller and others (2004) reported that survival and growth of natural northern red oak advance reproduction was significantly increased after a 22 percent reduction in stand basal area, which was done by thinning from below. They found that understory light was increased from 2 to 12 percent of full sunlight when they applied herbicide by cut stump or stem injection to all tree stems greater than 0.6 m tall in the understory and all trees in the intermediate and suppressed crown classes. Similarly, Motsinger (2006), working in Mississippi River bottomland forests in Missouri, increased understory light levels to 16 percent of full sunlight and improved the survival and growth of pin oak (*Q. palustris* Muenchh.) reproduction. In Mississippi, Lockhart and others (2000) increased understory light to 30 percent of full sunlight in bottomland pine-hardwood forests by removing trees in the intermediate and suppressed crown classes, thus improving the growth of cherrybark oak (*Q. pagoda* Raf.) advance reproduction. The higher light levels achieved in this study may be largely a result of the abundance of pine in the overstory, and light's increased ability to penetrate pine foliage and crowns compared to a pure hardwood canopy of similar stand stocking. In a diversity of forest types across eastern North America, it has been demonstrated that removal of midstory woody stems benefits oak reproduction by increasing light, but light intensity after these treatments, in general, are still significantly below the light levels associated with maximum net photosynthesis and growth.

Overstory Harvesting

In general, more significant reductions in overstory canopy density are necessary to increase understory light to 30 to 70 percent of full sunlight, which is required for maximum net photosynthesis and growth in northern red oak advance reproduction. Regeneration methods that substantially reduce overstory stocking are recommended for oak regeneration (Hannah 1988, Gardiner and Yeiser 2006, Johnson and others 1986, Loftis 1990a, 1993, Sander and Graney 1993, Smith 1993, Spetich and others 2002, Weigel and Johnson 1998a). Oak reproduction grows rapidly in group selection openings that are at least as wide as the height of the adjacent forest (Marquis 1965, Minckler 1961). The clearcutting method has been most successful in regenerating oaks in xeric ecosystems where large advance reproduction has been able to accumulate before harvesting (Johnson and others 2002). However, releasing small oak reproduction to full sunlight by clearcut or large group selection harvests on high quality sites without concurrent competition control has not been successful because oak seedlings are quickly suppressed by either well-established shade tolerant advance reproduction or fast-growing, shade intolerant species (Beck and Hooper 1986, Brose and Van Lear 1998, Canham 1988, 1989, Jensen and Parker 1998, Loftis 1983, 1990b, McQuilkin 1975, Orwig and Abrams 1994, Ross and others 1986, Smith 1981, Weigel and Parker 1997).

In a diversity of forest types, shelterwood harvesting that reduces stand stocking by more than 40 percent, or basal area by more than 50 percent, or crown cover by more than 30 percent can provide 35 to 50 percent of full sunlight to oak reproduction provided the midstory canopy is also removed (Gardiner and Yeiser 2006, Godman and Tubbs 1973, Leak and Tubbs 1983, Lorimer and others 1994, Parker and Dey in press, Sander 1979, Schlesinger and others 1993). As mentioned earlier, maintenance of higher residual shelterwood density and continuous canopy cover in the main overstory helps to control shade intolerant competitors (Loftis 1990b, Schlesinger and others 1993). Schlesinger and others (1993) found that control of taller understory woody competitors in the Missouri Ozark Highlands is not necessary to promote oak advance reproduction growth on low quality sites (i.e., site index 18 m black oak) where oak reproduction did best growing under a 40 percent stocked shelterwood. In contrast, oak reproduction on more productive sites (i.e., site index 21 m black oak) did best when stocking was left at 60 percent and woody understory competition was controlled. These Missouri Ozark ecosystems are outside of the range of yellow-poplar and really do not have a comparable aggressive shade intolerant competitor. Final removal of the shelterwood is triggered when the needed stocking of large oak advance reproduction is present, and depends on management goals for oak stocking at stand maturity. Dominance probability models such as those presented by Loftis (1990b) or Spetich and others (2002) are helpful in assessing the regeneration potential of oak advance reproduction, and timing the shelterwood removal.

Oak Recruitment into the Overstory

For the first few years after a disturbance, tens of thousands of small seedlings and sprouts per hectare compete for the free growing space, but the winners of this competition become apparent in just a few years. As stand age increases after clearcutting, density of all trees, as well as number of dominant and codominant trees, decreases (fig. 6). In central Appalachia, there may be as many as 7,400 trees

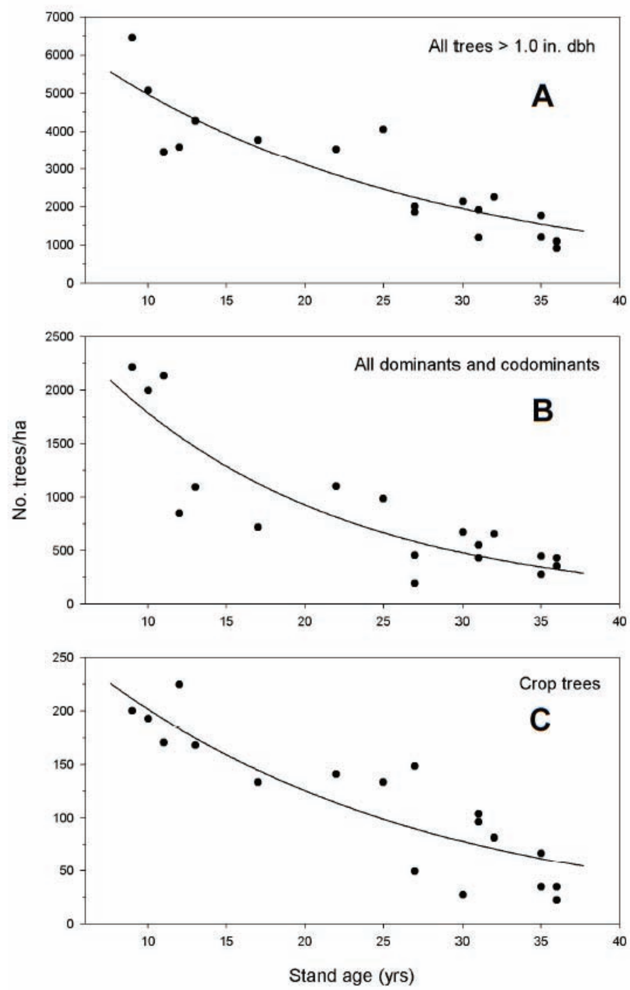

Figure 6— Number of trees per hectare of (a) all trees >2.5 cm d.b.h., (b) all dominant and codominant trees, and (c) only crop trees in 18, even-aged, mixed hardwood stands on northern red oak site index 20 to 21 m in the central Appalachians. Circles are observed densities and lines are regression estimates of average density by stand age (from Miller et al. 2007).

per hectare in a 10- to 15-year-old mixed hardwood forest clearcut (site index 20 to 21 m northern red oak), where the overstory trees are 9 to 12 m tall at canopy closure (fig. 6a) (Miller and others 2007). Although a young stand contains thousands of trees per hectare, only the canopy dominant and codominant stems can persist for many years and provide long term benefits. For example, a typical 15-year-old stand averages 3,700 trees per hectare (≥ 2.5 cm d.b.h.) (fig. 6a), of which only one-third of the trees are in competitive positions in the canopy (fig. 6b).

Without management intervention, suppressed and intermediate northern red oaks have low probability of becoming more dominant as stands progress toward maturity. Ward and Stephens (1994) studied stand dynamics in second growth northern red oak-mixed hardwood forests in Connecticut as the stands developed from age 25 to 85. Northern red oak trees were distributed among all crown classes at stand age 25 with the majority of the oaks in the intermediate and larger classes. However, the only northern red oaks to persist to stand age 85 were those that were dominant or codominant. They reported that about 75 percent of the dominant northern red oak at age 25 remained dominant after 30 years, but only about 25 percent of the codominant oak remained in the codominant class by age 55. Most of the codominant trees dropped into the lower crown classes or died. Between stand age 25 and 55 most of the intermediate and suppressed oaks died. Northern red oak trees that express dominance early are the only trees that are able to persist as dominants into maturity. Thus, stands must be thinned at an early age to favor oak so that managers have flexibility in meeting their stand stocking objectives. This is more imperative on high quality sites, or where oak is competing with fast-growing species such as yellow-poplar.

The probability of oaks recruiting into the overstory as canopy codominants and dominants is modified by site quality. As forests progress through the stand initiation stage to the stem exclusion stage (Oliver and Larson 1996), oaks have a higher chance of becoming dominant on low to average quality sites than on higher quality sites (Johnson and others 2002). Hilt (1985) observed in 29 Ohio Valley clearcuts of oak-dominated forests that the proportion of oak in the reproduction was similar for the first 14 years regardless of site quality, and that 20 to 30 percent of all trees were oaks. After stand age 15, the proportion of oak increased dramatically to 60 percent of all trees on sites of relatively low quality (15 to 18 m site index). They were one-third of stand density on medium quality sites (18 to 21 m site index), but dropped to about 10 percent of stand density on high quality sites (21 to 24 m site index). Stocking of dominant, competitive oak can be increased even on high quality sites by thinning (cleaning) to release potential oak crop trees.

Crop Tree Thinning

Crop trees are defined by management objectives. The tree's market value, wildlife value, esthetic value, or "diversity" value determines whether or not it meets management objectives. Species and quality are key characteristics in identifying potential crop trees. Northern red oak that exhibit good bole quality (i.e., straight, branch-free boles, no cankers, no low forks, good attachment to stumps) and good vigor (i.e., healthy crown and bark development, etc.) are desirable crop trees. Only a small percentage of dominant and codominant trees qualify as crop trees (fig. 6c). A 15-year-old stand that contains about 1,200 dominant or codominant trees per hectare may have only 160 potential crop trees (Miller and others 2007).

It is also important to recognize that the number of potential crop trees declines as the stand ages (fig. 6c). Trees continue to compete for growing space and some overstory trees lose their canopy status and even die as the stand matures. Each year some potential crop trees succumb during the natural thinning process. In the absence of crop tree release treatments, stands older than 25 years often contain less than 100 crop trees per hectare (fig. 6c). In any given stand, the management objectives can be even more restrictive than the general characteristics described here. As a result, the crop tree densities in Figure 6c probably overestimate the number of potential crop trees in most stands.

The number of crop trees released within a typical hardwood stand varies depending on site quality, species composition, stand age, management objectives, and cost of pre-commercial operations. As a result, it is important to understand how hardwood stands develop and change over time so that release treatments can be applied effectively to achieve management objectives. When the stand is mature, the main canopy will contain 148 to 173 dominant and codominant trees per hectare, so there is no need to release more than this density when the stand is young. In most cases, however, the release treatment will involve far fewer trees (fig. 6c) due to their actual availability.

Guidelines for spacing of crop trees are less precise. Hardwood stands that form from natural regeneration exhibit random patterns in their distribution of potential crop trees. It is not unusual to find a clump of three or four potential crop trees growing in close proximity or scattered 15 to 30 m apart. A general guideline is to focus on finding the best available crop trees, regardless of spacing, and provide them with an adequate release. Avoid releasing trees that do not qualify as crop trees just for the sake of achieving an even distribution, as this approach may not be an efficient use of resources. In rare cases where crop trees are abundant and dispersed throughout the stand, seeking an even distribution of crop trees is acceptable, so long as each crop tree receives an adequate release.

In young hardwood stands, the best time to apply release treatments is when the canopy begins to close and continues for about 10 to 15 years after canopy closure. The stand age at canopy closure varies with site quality. On high-quality sites, where abundant resources accelerate stand development, canopy closure can occur about age 8 to 10 years. On poorer sites, where fewer species are competitive and stand development is somewhat slower, canopy closure can occur about age 13 to 15 years. Keep in mind that young stands contain more potential crop trees compared to older stands (fig. 6c) and thus opportunities to improve long-term species composition of the overstory are greatest in young stands that are less than 25 to 30 years old because the number of crop trees per hectare tends to diminish with stand age.

In older hardwood stands approaching large pole or small sawtimber size, there are still opportunities to culture dozens of crop trees per hectare to improve vigor, growth, and spacing of desirable trees as the stand matures. Beyond age 25 or 30 years, the number of potential crop trees can range from 62 to 99 trees per hectare, but this number will decline in coming years if such trees are not favored with crown release.

In sawtimber stands, crown release can be applied as a commercial operation, thus yielding timber sale revenue to offset other management costs while favoring selected crop trees. The problem with delaying the release until operations are commercial is that the number of remaining crop trees is relatively low in older stands and the opportunity to enhance oak stocking has long been lost. The overstory in commercial stands may include 148 to 173 trees per hectare, but usually less than half of them are desirable crop trees if the

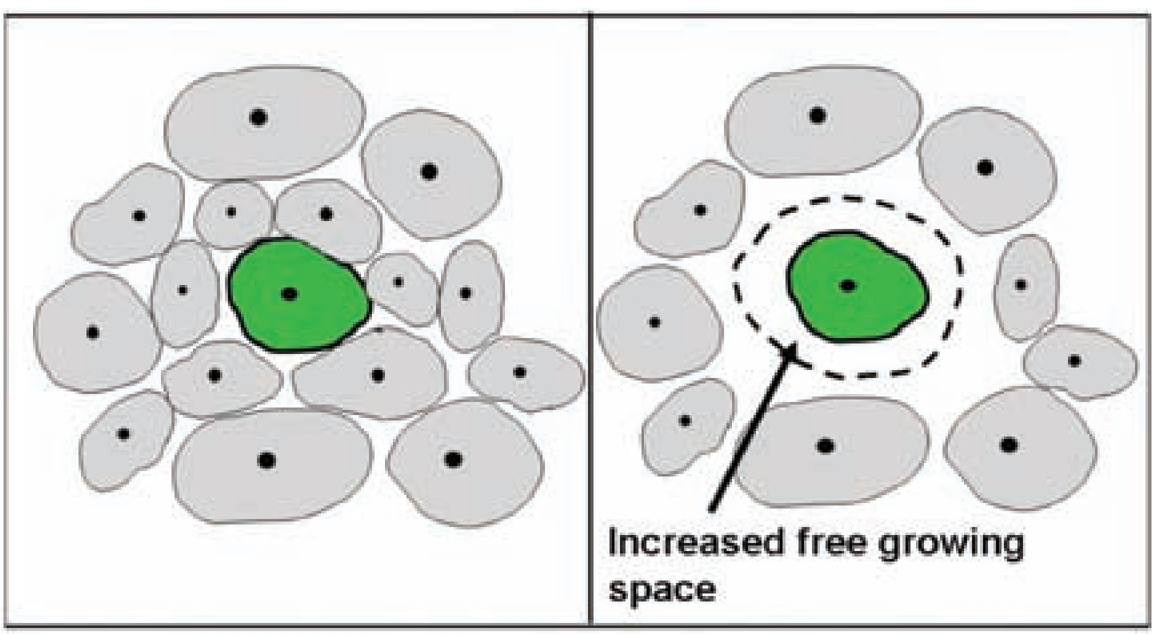

Figure 7— A crop tree crown (dark) shown from above the forest canopy. The left diagram represents a crop tree crown before release with 6 adjacent competitors. The right diagram illustrates the free growing space available when a crown-touching release is applied to remove competing trees from all sides of crop tree. (Diagram produced by J.W. Stringer and published originally in Miller and Stringer 2007).

stand has not been managed before. Ideally, release treatments should be applied when the stand is younger to retain as many crop trees, and as many oak crop trees in the overstory as the stand matures.

Cleanings can be used to deaden or remove trees that are limiting the horizontal crown expansion of an oak crop tree, thus increasing its free growing space and improving its survival probability. Generally, a crown-touching release is used to eliminate adjacent competing trees whose crowns touch that of the crop tree (fig. 7). The increase in growing space provides for an increase in direct sunlight and below-ground resources available to the crop tree. The crop tree can then develop more leaf area and a larger crown, thus increasing photosynthesis, stem diameter growth, and root growth. Improved vigor and crown size also has the potential to improve mast production of individual trees.

Thinning treatments can be used to provide various degrees of release based on the proportion of the crown that is left free-to-grow. In most cases, it is beneficial to retain trees in the overtopped and intermediate crown classes adjacent to crop trees. Such trees might be important for wildlife and aesthetics. They can also protect timber crop trees by partially shading their boles and protecting them from sunlight that can trigger epicormic branching.

Thinning Oak Stump Sprouts

Early thinning around oak sprout clumps can increase oak stocking at the end of the stem exclusion stage of stand development, especially on high quality sites. Weigel and others (2006) found an inverse relationship between oak stump sprout dominance probability and site quality within the range of 18 to 22 m site index for northern red oak in southern Indiana. Favoring oak reproduction through precommercial thinning gives managers greater densities of more highly competitive oak reproduction from which to work with to attain desired stocking levels at stand maturity. It is a better to have too much oak, rather than not enough, when sustaining oak is a primary management objective.

To maximize the development and competitiveness of oak stump sprout reproduction, it is helpful to thin around individual stump sprout clumps, thus releasing all the sprouts arising from a single stump. Thinning should be done early, i.e., by stand age 5. The number of oak stump sprout clumps to release follows the same recommendations given above for crop trees, and is ultimately determined by their availability, spatial distribution in the forest, and the desired level of oak stocking at stand maturity. Not all oak stumps produce sprouts following harvesting; their sprouting capacity depends, in part, on the species, and tree size and age (Dey and others 1996, Johnson and others 2002). In general, northern red oak sprouting potential declines with increasing tree diameter and age (Weigel and Peng 2002, Weigel and others 2006). Initial sprouting frequency of northern red oak stumps increases with increasing site quality, but long-term survival and dominance of oak stump sprouts declines with increasing site quality due to the greater intensity of competing vegetation than is found on lower quality sites (Weigel and Peng 2002, Weigel and others 2006). Release of desirable oak stump sprout reproduction may not be enough to meet oak stocking goals, especially when regenerating older stands that have lower stump sprouting potential. Therefore, additional crop tree release of young oak that regenerated as advance reproduction may be necessary.

Early thinning (e.g., at age 5) of oak sprout clumps to a single stem can significantly increase tree diameter for species in the sections Quercus and Lobatae (Lowell and others 1987, 1989, Dwyer and others 1993). Johnson and Rogers (1980, 1984) estimated that 25th-year diameter of northern red oak stump sprouts can be dramatically increased by thinning the sprout clump to a single dominant stem. Thinning oak stump sprouts at any age between five and 15 years significantly increased 25th-year diameter (fig. 8).

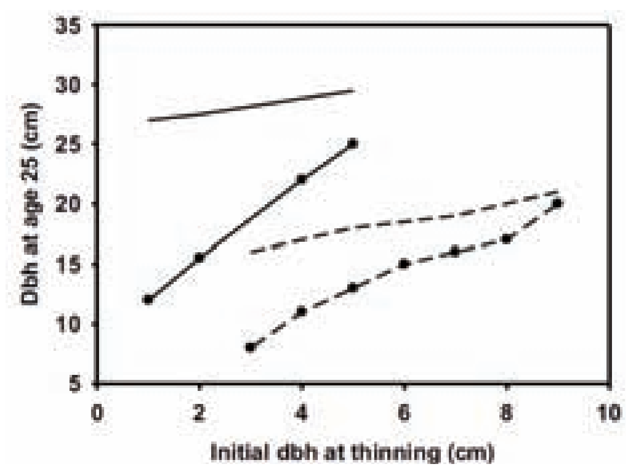

Figure 8— Comparison of estimated 25th-year diameters of northern red oak stump sprouts that were thinned to a single stem (lines without symbols) or not (lines with symbols) at ages 5 years (solid lines), or 15 years (dashed lines) (Johnson and Rogers 1980, 1984).

Initially smaller diameter oak sprouts showed the greatest increase in diameter by stand age 25. The earlier the thinning, the greater was the expected increase in 25th-year diameters. Groninger and others (1998) thinned 12-year-old chestnut oak (*Q. montana* Willd.) and scarlet oak (*Q. coccinea* Muenchh.) stump sprout clumps in Virginia clearcuts by reducing sprouts to a single dominant stem per stump and removing surrounding competitors that had crowns touching the oak crop tree. They found that thinning by chainsaw or herbicide (stem spray or stem injection with 2,4-D or triclopyr): (1) effectively controlled competing stems both within and external to the oak stump sprout clump; (2) significantly increased crop tree diameter after five years; (3) did not affect total height; and (4) herbicides did not cause unusual mortality of crop trees.

CONCLUSION

Sustaining oak stocking requires managing forest regeneration and recruitment processes to enhance oak regeneration potential and its long-term competitiveness through to stand maturity. There are three defining stages in stand development relevant to sustaining northern red oak. In the initial mature forest stage, beginning several years or more before the parent trees are removed, sources of oak reproduction must be present in sufficient numbers, size, and distribution to compete with other vegetation on the site. Oak regeneration potential at this point is largely determined by the probable contributions expected from advance reproduction and stump sprouts. In the stand initiation stage, oaks must be released by a timely and adequate reduction in the canopy such that residual trees do not interfere with the oaks as they compete for dominant or codominant status in the young stand. Once the canopy of the young stand closes, a third critical stage begins during which the young oaks in the upper canopy of the new stand must maintain their status for decades until the stand is mature. Again, upper-canopy oaks can be sustained for long periods by providing a timely and adequate crown release in the young stand.

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Restoring East-Side Ponderosa Pine Ecosystems at the Blacks Mountain Experimental Forest: A Case Study

Jianwei Zhang and Martin W. Ritchie

ABSTRACT

The ecological research project of interior ponderosa pine forests at the Blacks Mountain Experimental Forest in northeastern California was initiated by an interdisciplinary team of scientists in the early 1990s. The objectives of this study were to determine the effects of stand structure, and prescribed fire on vegetation growth, resilience, and sustainability of ponderosa pine ecosystems. Two stand structures were created with mechanical thinning on 12 units of about 100 hectares each. The low structural diversity (LoD) treatment has an even spaced, continuous canopy with few snags and few large canopy gaps. In contrast, the high structural diversity (HiD) includes many large, old trees with multiple canopy layers, abundant large diameter snags, many small canopy gaps, and some dense clumps of smaller trees. Each structural treatment was randomly assigned in each of three blocks. Each combination (unit) was split into two halves and prescribed fire was applied to one of the split plots after the completion of thinning. Because of the complexity of treatments and large experimental units, thinning and prescribed burns took 5 years to complete. Accordingly, our first post-treatment measurements were staged one year after implementation of prescribed fire. The second post-treatment measurements were completed five years after the initial measurement. In the short term, we found that (1) tree growth and understory woody plant cover significantly responded to structural diversity, but not prescribed fire; (i) the HiD plots carried much more basal area and stems than low structural diversity (LoD) plots, (ii) both individual-tree and stand-level growth increment was higher in the LoD plots than in the HiD plots, and (iii) understory woody plant species grew back in five years with higher percentage cover in the LoD than in the HiD, (2) prescribed fire effect was significant for dead woody materials on the forest floor and bark beetle colonization; (i) debris was reduced significantly more with prescribed fire and (ii) bark beetles tended to attack trees weakened by the prescribed fire, (3) by comparing adjacent untreated stands, our HiD treatment appeared to improve health of the largest trees (DBH > 60 cm) in the stand, and (4) no significant interaction was found between structural diversity and prescribed fire. These results suggest that ponderosa pine forests can be treated to enhance stand growth and health without sacrificing understory vegetation diversity and stand productivity in the short-term.

Keywords: interdisciplinary research, ponderosa pine forest, thinning, prescribed fire, stand dynamics, understory shrubs

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INTRODUCTION

In recent decades the Forest Service has applied a multiple-use philosophy in managing the national forests and grasslands (Wilkinson and Anderson 1987). This has not only focused on the production of valuable commodities such as wood, fiber, food, water, and minerals for human use, but also emphasized recreational, spiritual, aesthetic, and educational experiences on the public lands (Kessler et al 1992). Effective management can be helped by interdisciplinary research that addresses multiple factors (Oliver 2000).

Recommendations from the National Research Council (1990) and the Forest Service Strategic Plan for Research (1990) led the Pacific Southwest Research Station to establish an interdisciplinary team to consider research needs in interior pine forests. The team began planning a large-scale ecological research project at Blacks Mountain Experimental Forest in northeastern California in 1991. Experimental design, National Environmental Policy Act (EPA) documentation, and timber sale preparation took some five years; treatments were initiated in 1996. Treatment installation was completed in October of 2000 when the prescribed burning treatment was applied to the last block.

The objectives of this long-term study are to increase our understanding of the effects of forest structural complexity and prescribed fire on health and vigor of an interior ponderosa pine ecosystem and to determine how these ecosystems can be managed for sustained resource values (Oliver 2000; Oliver and Powers 1998). Blacks Mountain Experimental Forest was chosen for this project because of the existing forest conditions with a representative late seral structure still extant. Designated in 1934 as an Experimental Forest, it is one of the few forests in the United States with more than 50 years of records on forest structure. The forest has an extensive, well-designed road and compartment system.

In this report, we will review some short-term results from this study from the initial 5-year post-treatment period. These results are primarily from manuscripts submitted to a special issue of Canadian Journal of Forest Research entitled “Interdisciplinary Research on Interior Pine Forests at Blacks Mountain Experimental Forest” submitted for publication in 2008. We appreciate all authors who allowed us to use their unpublished data.

SITE DESCRIPTION

Blacks Mountain Experimental Forest is located approximately 35 km northeast of Mount Lassen in northeastern California (40°40’ N latitude, 121°10’ W longitude). The forest occupies 3,715 ha on the Lassen National Forest; elevational range is between 1,700 m and 2,100 m. Soils are Typic Argixerolls, 1-3 m deep over basalt with mesic soil temperature regimes at lower elevations. Andic Argixerolls with frigid soil temperature regimes predominate at higher elevations. The climate is characterized by warm dry summers and cold wet winters. Annual precipitation is approximately 460 mm and falls primarily as snow from October through May. Mean air temperatures usually range from -9 to 29°C.

The overstory vegetation on the experimental forest is dominated by ponderosa pine (***Pinus ponderosa*** Dougl. ***ex*** Laws.) with varying amounts of white fir (***Abies concolor*** (Gord. & Glend.) Lindl. ***ex*** Hildebr.), incense-cedar (***Calocedrus decurrens*** (Torr.) Florin), and Jeffrey pine ***(Pinus jeffreyi*** Grev. & Balf.) (Oliver 2000). The understory contains abundant regeneration of both ponderosa pine and white fir. The five shrub species accounting for most of the ground cover are: greenleaf manzanita (***Arctostaphylos patula*** Greene), prostrate ceanothus (***Ceanothus prostratus*** Benth.), snowbrush (***Ceanothus velutinus*** Dougl. ex Hook.), antelope bitterbrush (***Purshia tridentata*** (Pursh) DC), and creeping snowberry (***Symphoricarpos mollis*** Nutt.).

EXPERIMENTAL TREATMENTS

The study covers 1,226 ha of the experimental forest and 111 ha of the adjacent national forest. Silvicultural treatments and grazing have been imposed under a completely randomized split-plot design (Figure 1), with combinations of structural diversity (high vs. low) of the residual stand and grazing (grazed vs. ungrazed) as the main plot factors and with (F) and without (NoF) prescribed fire as the subplot factor (Oliver and Powers 1998, Oliver 2000). Whole-plot experimental units ranged from 77 to 144 ha (μ = 111 ha) and were blocked by proportion of species composition rather than geographic proximity because white fir and incense-cedar become increasingly abundant at the higher elevations in the forests. The two structural treatments were randomly assigned to six experimental units each. The high structural diversity (HiD) treatment was designed to leave all

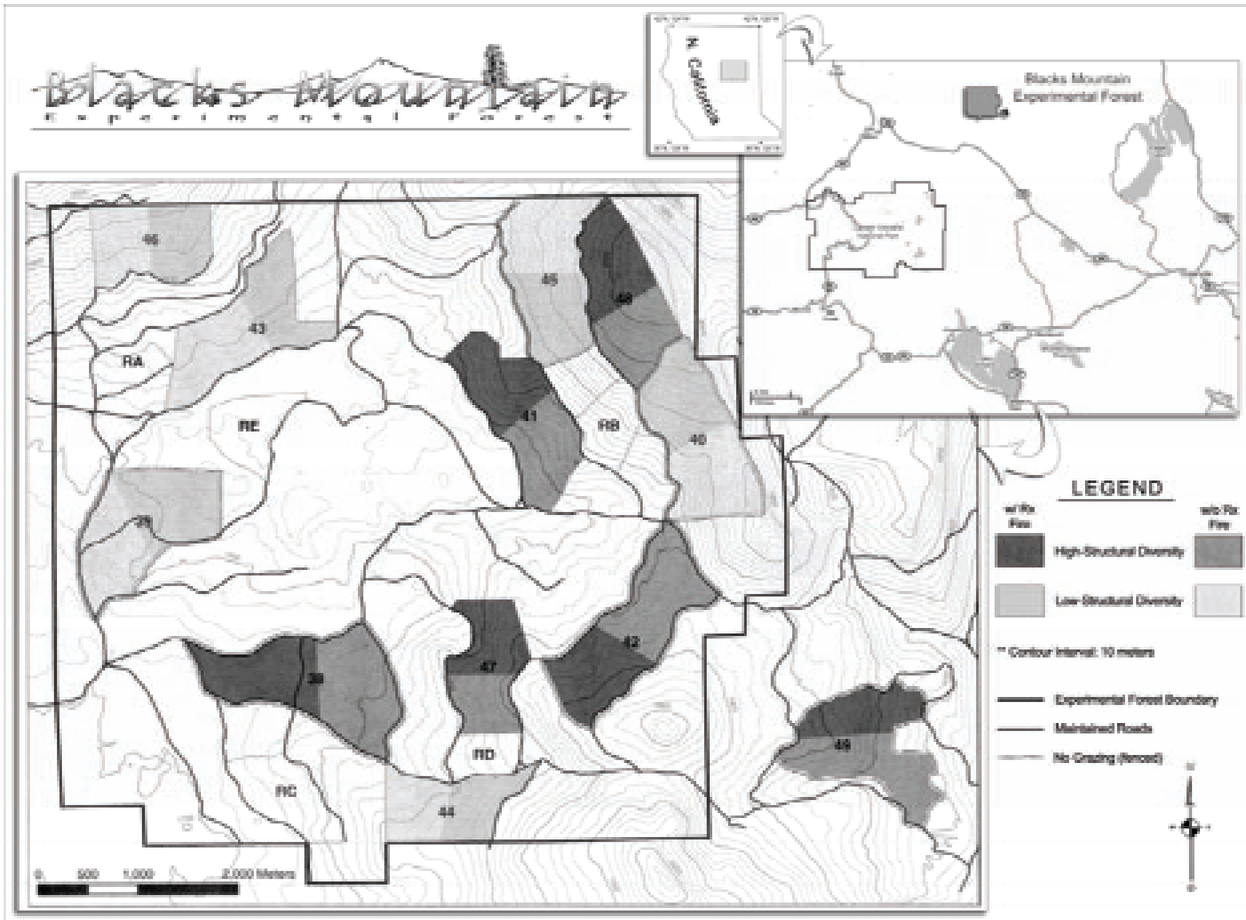

Figure 1— Location of the Blacks Mountain Experimental Forest and layout of the Blacks Mountain Ecological Project in northeastern California, USA.

large, old dominant trees, abundant snags, multiple canopy layers with dense clumps of smaller trees, and many canopy gaps and forest floor openings (Oliver and Powers 1998, Oliver 2000). The low structural diversity (LoD) treatment produced a single canopy layer of well-spaced co-dominant trees, few snags, and few large canopy gaps and forest floor openings (Figure 2). Five Research Natural Areas, which are not included in this study, are well-distributed within Blacks Mountain Experimental Forest (Figure 1) and could provide information on untreated stands.

Because of large plot size, treatment implementation took several years. Three individual blocks, each with two HiD units and two LoD units were created in 1996, 1997, and 1998 respectively. The prescribed fire was planned for the fall of the following year. Because the conditions needed to meet the burning prescription could not be met in 1998, prescribed fire for blocks two and three were delayed to the fall of 1999 and 2000 respectively. Duff litter around existing snags was raked prior to application of prescribed fire.

Six of twelve units were fenced to exclude grazing in the summer of the following year when the structural diversity treatment was applied. After treatment installation, an interdisciplinary team of scientists observed that our grazing treatment has not been effective because grazing intensity is largely a function of proximity to water, which limits most grazing to block one. In addition, a preliminary analysis indicated that there was no significant grazing effect for all variables measured. Therefore, the grazing treatment will not be considered in the analysis, thus doubling the replications within each block.

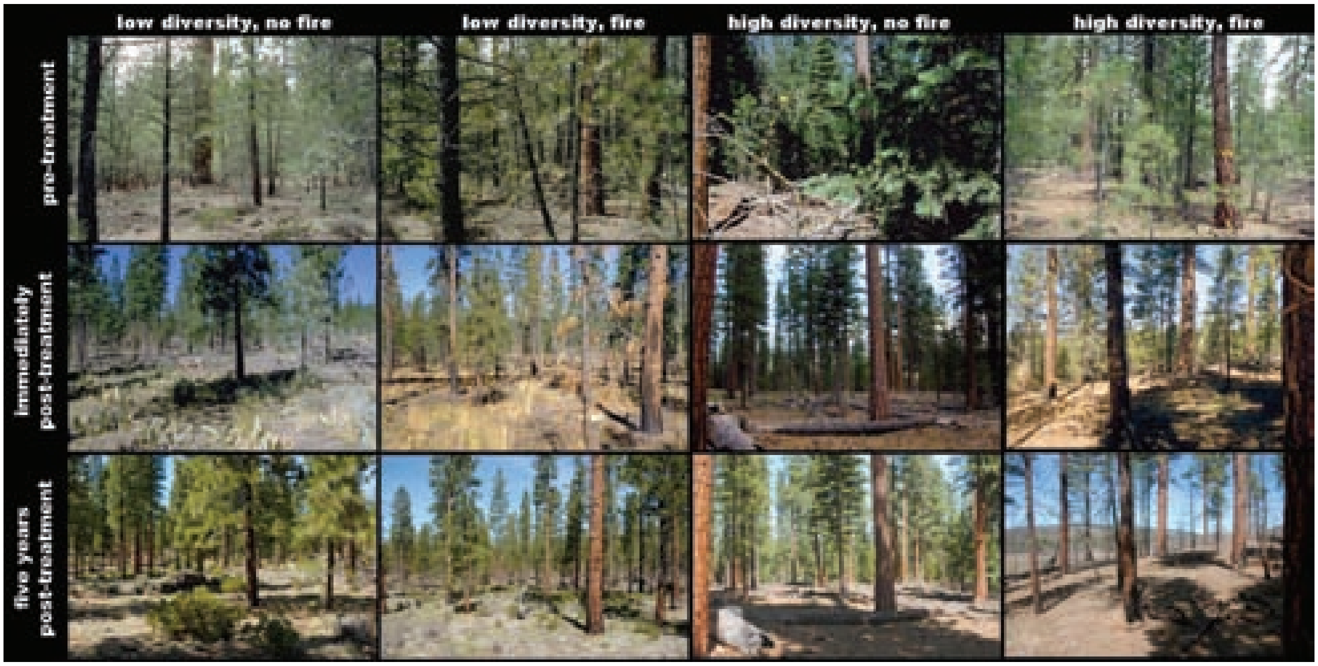

Figure 2— Stand structure and species composition on low structural diversity without prescribed fire, low structural diversity with prescribed fire, high structural diversity without prescribed fire, and high structural diversity with prescribed fire in the pre-treatment, immediate post-treatment, and five years after post-treatment at the Blacks Mountain Ecological Project in northeastern California, USA.

SAMPLING PROCEDURES

Throughout each of the experimental units, a permanently monumented grid on 100-m centers was installed (Oliver 2000). The monuments were located by conventional survey methods. Actual location of each monument was within 6 inches. Monuments are 46-cm rebar with an aluminum cap stamped with the main plot number and Universal Transverse Mercator (UTM) coordinate. All measurements were referenced to these grid points. Among variables that have been measured, trees were most intensively measured; the detailed methods were described in Zhang et al (2008). For example, a pre-treatment tree inventory (DBH > 9.1 cm) was conducted in 1994, with a 25 % grid point sampling intensity. The purpose of the pre-treatment inventory was to collect the baseline data that was utilized to designate the block and unit layout. The post-treatment inventories were more intensive with 50% of the grid points systematically sampled in the year following the prescribed fire application (1998, 2000, and 2001). Every tree with DBH > 9.1 cm DBH has been tagged. As part of the study, the plots are scheduled to be re-measured every five years after the first post-treatment inventory. The second post-treatment re-measurements were conducted in 2003, 2005, and 2006 by blocks.

Understory woody plants, species number and cover percentage were also measured during the post-treatment survey (Zhang et al. 2008). Dead woody materials were measured prior to and post treatment (Brown 1974; F.C.C. Uzoh and C.N. Skinner, unpublished data). A census of bark beetle-induced mortality was established (Fettig et al. 2008). Because there have been many more measurements conducted in the study, we are only focusing on a few variables that have been thoroughly analyzed.

TREATMENT EFFECTS ON FOREST COMPONENTS

Dead woody fuels on the forest floor

Dead woody materials were sampled with the planar intercept method before the treatment was installed in 1994 and one year following treatment. Volume (m^3 ha^{-1}) of sound and decayed woods were separately calculated and converted to dry weight (Mg ha^{-1}) using method of van Wagtendonk et al (1996) (F.C.C. Uzoh and C.N. Skinner, unpublished data). Results show that prescribed fire has significantly reduced both sound and rotten woody materials on the forest floor (Figure 3). Effect of structural diversity was not significant. Although woody materials

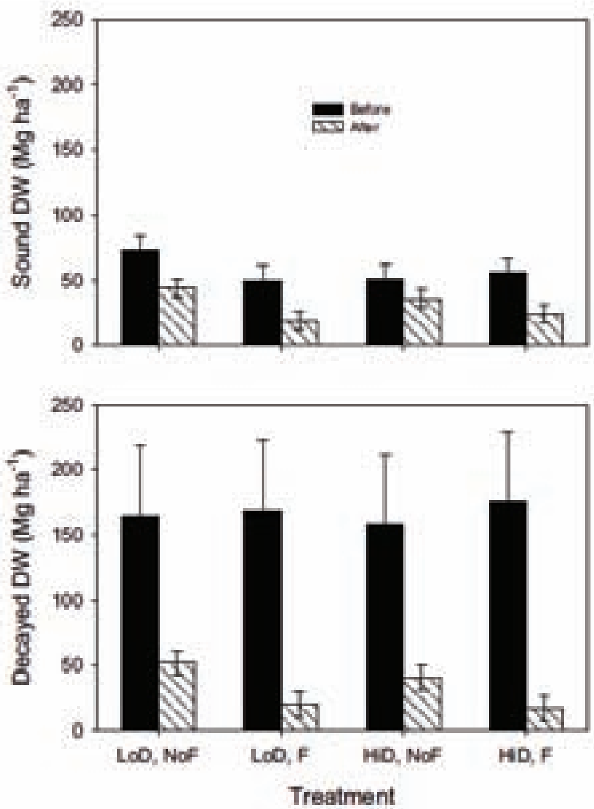

Figure 3— Mean dry weight (±1SE) of sound and decayed dead woody materials (F.C.C. Uzoh and C.N. Skinner, unpublished data) before and after treatments were installed at the Blacks Mountain Ecological Research Project in northeastern California. LoD = low structural diversity, NoF = without prescribed fire, F = with prescribed fire, and HiD = high structural diversity.

with advanced decay were more prevalent than sound woody materials prior to treatments, the amounts were comparable after treatment due to breaking up of larger materials by harvesting machines (Weatherspoon 1983).

Understory woody plants

Using the planar-intercept method on a 100 m transect associated with sampled grid points for the post-treatment measurements, the first post-treatment inventory did not reveal any significant differences in shrub cover or number of species with respect to structural diversity or prescribed fire (Figure 4). Five years later, shrub cover was significantly higher on LoD units, but number of species did not vary with structural diversity. A potential explanation is that tree canopy cover in LoD treatment is much lower than that in HiD treatment (Vaughn and Ritchie 2005). Shrubs may have responded to this with increased growth with more sunlight and perhaps more water available under the more open canopy. However, these treatments did not alter the seed bank or propagules of the existing species before treatment and nor did they appear to create favorable conditions for exotic species.

Stand dynamics

The thinning treatment produced two very different stand structures. The average values of LoD and HiD treatments were 10 vs. 25 m^2 ha^{-1} for BA (Figure 5) and 282 vs. 513 trees ha^{-1} for number of trees (Figure 6), respectively. Based on the initial post-treatment measurements, the upper limit for diameter in LoD units was 60 cm and BA was concentrated between 20 and 40 cm DBH classes. In contrast, HiD treatment maintained trees across a broader range of DBH classes; BA was concentrated between 20 and 80 cm DBH classes (Zhang et al. 2008).

Based on the second post-treatment measurements, structural-diversity effect was still significant in stand BA and density; the average values of LoD and HiD treatments were 11 vs. 25 m^2 ha^{-1} for BA and 258 vs. 487 trees ha^{-1} for

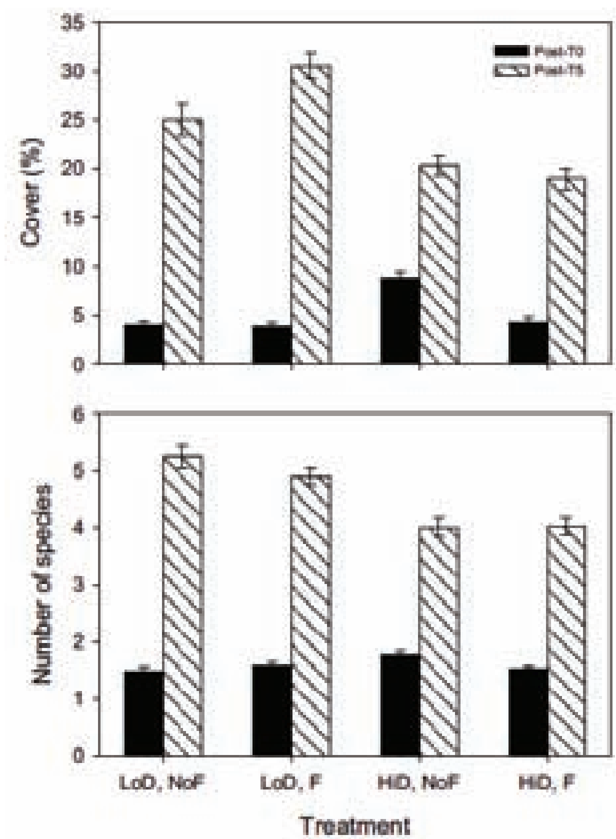

Figure 4— Overall shrub cover (%) and number of species (μ±1se) grown in the treatments at the Blacks Mountain Ecological Research Project measured immediately (Post-T0) and five years (Post-T5) after treatments were installed (Zhang et al. 2008). Abbreviation of x-axis label is in the legend of Figure 3.

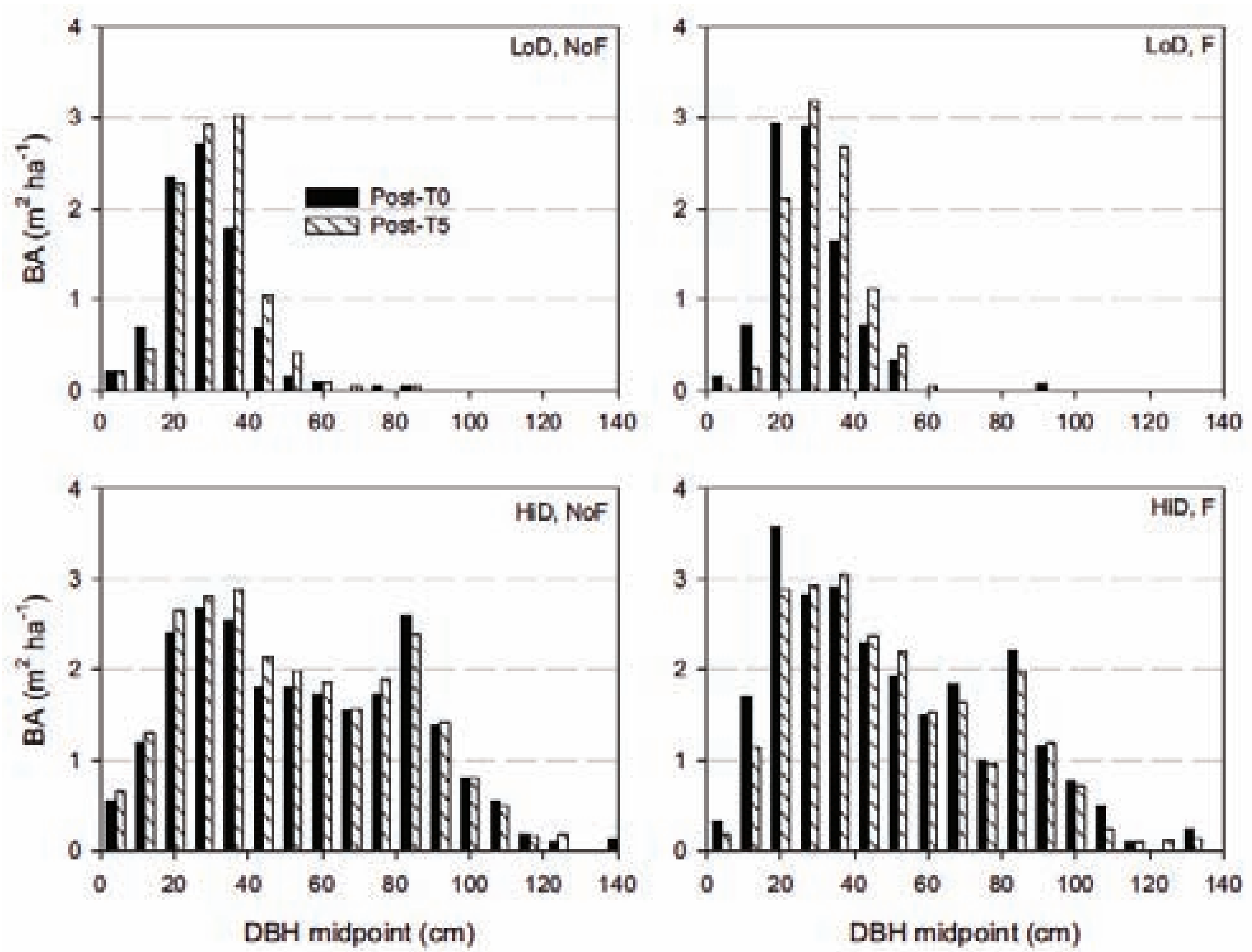

Figure 5 - Diameter-class basal area distributions for trees grown in the LoD (upper two) and in the HiD (lower two) without prescribed fire (left two) and with prescribed fire (right two) at the Blacks Mountain Ecological Research Project measured immediately (Post-T0) and five years (Post–T5) after the treatments were installed.

number of trees, respectively. Prescribed-fire effect was also significant for trees ha^{-1}. Differences were mainly related to mortality caused by prescribed fire and bark beetles. The mortality tended to occur in the smaller DBH classes (<20 cm) (Figures 5 & 6). During the five year period, mean trees ha-1 increased 11% (32 tree ha^{-1}) in the LoD without prescribed fire and decreased 28% (79 tree ha^{-1}) in the LoD with prescribed fire. Similarly, we found a 13% (67 tree ha^{-1}) increase in trees ha^{-1} in the HiD without prescribed fire and a 29% (145 tree ha^{-1}) decrease in the HiD with prescribed fire between the first and second post-treatment measurements. The results demonstrated that prescribed fire damaged or killed some trees. Many trees apparently alive in the first post-treatment measurements died during the next 5 years. Therefore, mortality assessments should consider both primary (directly killed by fire) and secondary (weakened by fire and killed by insects or disease) mortality after the prescribed fire to obtain accurate estimates (McHugh & Kolb 2003).

Stand diameter (QMD) grew significantly more in LoD treatment (0.65 ± 0.02 cm yr^{-1}) than in the HiD treatment (0.33 ± 0.01 cm yr^{-1}) (Figure 7). Effect of structural diversity was not significant for PAI BA, although PAI BA was slightly higher for the LoD treatment (0.38 ± 0.02 m^2 ha^{-1} yr^{-1}) than for the HiD treatment (0.34 ± 0.02 m^2 ha^{-1} yr^{-1}). Trees surviving fire grew as much as trees without fire, confirming that low intensity surface fires may not affect tree growth (Busse et al. 2000; Ritchie and Harcksen 2005).

Stand productivity based on living trees measured at both times may be underestimated for the LoD plots rela-

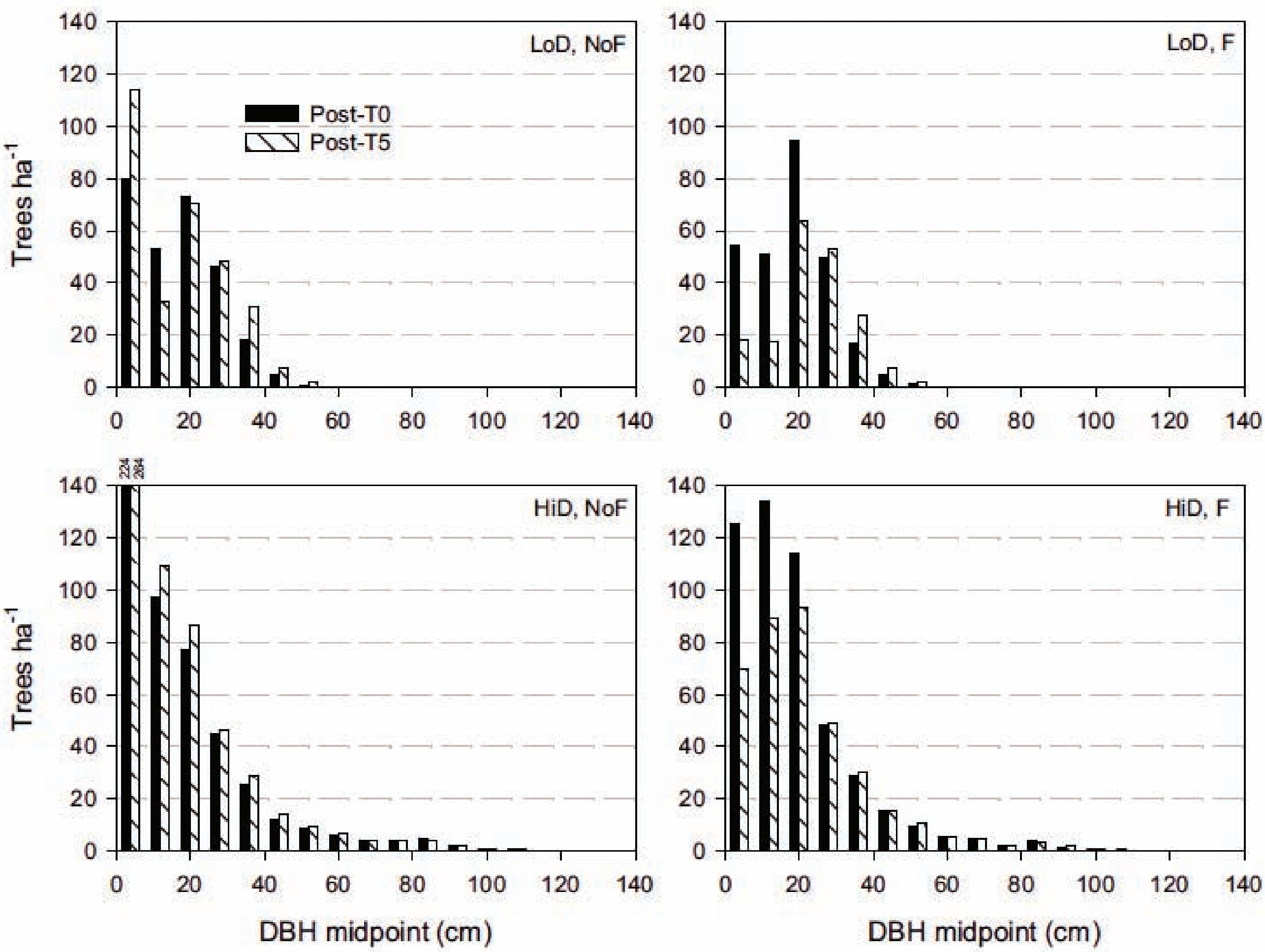

Figure 6 - Diameter-class frequency distributions for trees grown in the LoD (upper two) and in the HiD (lower two) without prescribed fire (left two) and with prescribed fire (right two) at the Blacks Mountain Ecological Research Project measured immediately (Post-T0) and five years (Post–T5) after the treatments were installed.

tive to the HiD plots if the Langsaeter's growth curve was considered (sensu Daniel et al. 1979, p. 318). When an old stand was thinned, annual increment in the earlier periods was strongly related to the initial stand density as found on other long-term density studies in ponderosa pine (Cochran and Barrett 1993, Oliver 1997, Zhang et al. 2006). The trend would not change until the onset of inter-tree competition occurs again (Smith et al. 1997, Zhang et al. 2006). In this regard, it is perhaps surprising that the LoD plots, which carry less than half the BA of the HiD plots, showed such elevated growth levels. If we considered growth proportion by dividing by initial BA to compare both treatments, the LoD would have showed even higher percentage growth (0.38/10 = 3.8% yr^{-1}) than the HiD would (0.34/25 = 1.4% yr^{-1}).

Large trees in the HiD

One of the objectives in restoring ponderosa pine ecosystems is to improve the health of large dominant trees by minimizing environmental stresses. Risk rating is one metric for tree health (Keen 1943; Salman and Bongberg 1942). Evaluating risk rating for the large trees (DBH > 60 cm) in the HiD units, Ritchie et al. (2008) found far fewer of these trees were rated as "high-risk" than the large trees grown on adjacent untreated Research Natural Areas (Figure 8), although the number of large trees was approximately the same on both HiD and RNA. This trend held in both burned and unburned units. Both treatments appear to improve the health of these large dominant trees.

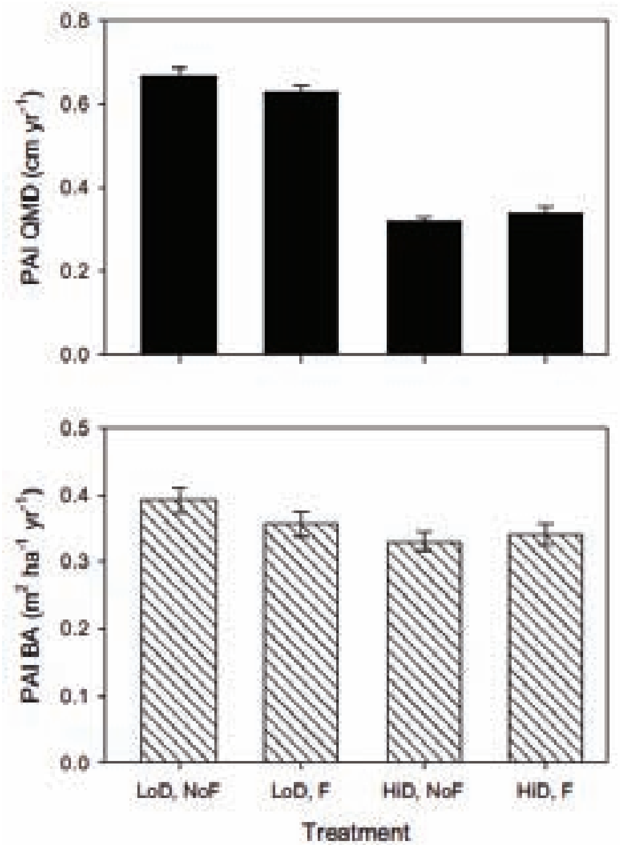

Figure 7— Average (μ±1se) periodic annual increment (PAI) for quadratic mean diameter (QMD) and basal area (BA) for different treatments calculated from first post-treatment measurements and second post-treatment measurements (redrawn from Zhang et al. 2008). Abbreviation of x-axis label is in the legend of Figure 3.

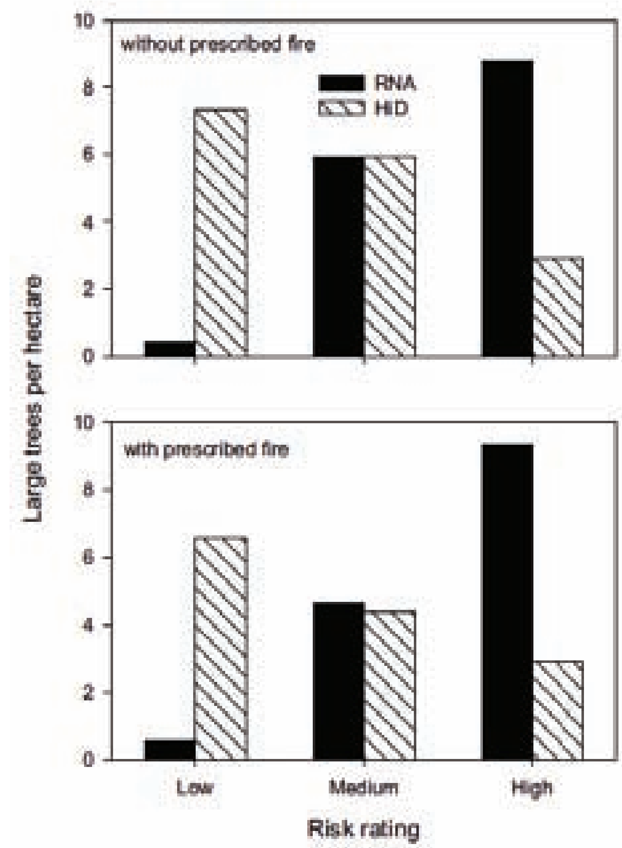

Figure 8— Large trees (DBH > 60 cm) per hectare sorted by risk rating observed from the high structural diversity (HiD) treatment and adjacent Research Natural Area (RNA) without and with prescribed fire at the Blacks Mountain Experimental Forest. Figures were drawn based on data of Ritchie et al. (2008).

Bark beetle colonization

Bark beetles attack significantly more trees on plots in prescribed fire splits than on plots without prescribed fire, regardless of structural diversity (Figure 9). About 86% of bark beetle-caused mortality occurred in the prescribed fire splits, mainly in smaller diameter classes (Fettig et al. 2008). This result may explain the significantly different mortality between the first and second post-treatment measurements within the prescribed fire split. Therefore, secondary mortality may be expected even when initial fire-caused mortality is low because prescribed fire weakens trees that are susceptible to the bark beetle attack. It is not known if this same phenomenon will be observed with later applications of prescribed fire in stands with more vigorous trees and healthier crowns.

IMPLICATIONS

Across northeastern California, many crowded ponderosa pine forests are threatened by severe, stand replacing fire and bark beetle outbreaks. Forest managers are very interested in using silvicultural tools to restore resilience in these ecosystems. Because the problems have resulted from decades of fire exclusion combined with inadequate stand density management in this fire-adapted ecosystem, they cannot be solved overnight. This project demonstrates a good start, by successfully using thinning and/or prescribed fire to treat these forests. The treated stands contain much fewer high-risk trees than untreated stands, suggesting that the health of the treated stands has been improved within five years. Based on results of dead woody materials and shrub cover for the initial five years, it appears that the dead woody fuels on forest floor only respond to prescribed fire (Figure 3) while the growth of shrubs is significantly associated with over-story structural diversity but not

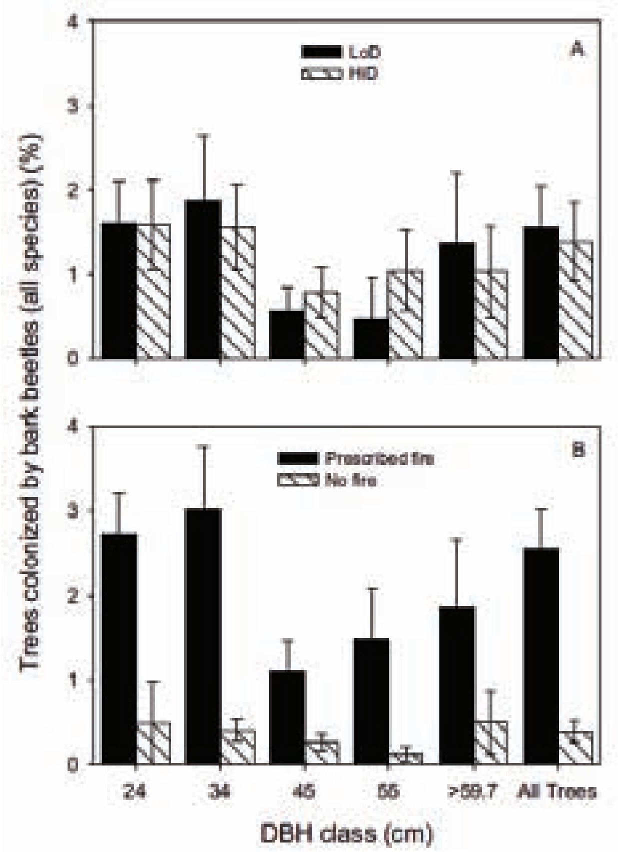

Figure 9— Mean (μ±1se) percentage of tree colonized by bark beetles (all species combined) by diameter class for all trees by structure treatment (A) and with and without prescribed fire (B) at the Blacks Mountain Ecological Research Project (redrawn from Fettig et al. 2008).

with prescribed fire (Figure 4). These results may explain the findings by Ritchie et al. (2007) that a combination of thinning and prescribed fire is more effective than thinning only at reducing wildfire severity.

The stand structures considered in this study are representative of two contrasting management priorities. The LoD stand is consistent with a timber production emphasis – maintaining thrifty fast-growing trees — yielding a much higher rate of growth both for individual trees and at the stand level. The HiD stand is consistent with an emphasis on maintenance of a range of tree sizes that may be more consistent with historic (pre-settlement) conditions. This increase in structural diversity comes at a significant loss in productivity. This cost may be offset by other amenities (e.g. wildlife habitat, visual resources) that they may provide. With either type of stand structure, use of prescribed fire may be beneficial in maintaining the health and vigor of these stands in the long term. However its application may reduce the occurrence of snags and woody debris, and both of these contribute to wildlife habitat.

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Silvicultural Strategies for Restoring Missouri Ozark Ecosystems on the Mark Twain National Forest

Paul Nelson and Charly Studyvin

ABSTRACT

The 1.5 million acre Mark Twain National Forest (MTNF) spans thirteen ecological subsections stretching over 200 miles of the Ozark Highlands in Missouri. Historically, this landscape contained a diverse assemblage of savanna, woodland, forest, glade, fen, spring, stream and cave natural communities. However, following European settlement, decades of domestic livestock overgrazing, logging, conversion to agriculture and fire suppression led to degradation and loss of these natural communities. This degradation resulted in loss of biodiversity, altered successional pathways, soil loss, reduced forest productivity, oak decline and fuel buildups. Today, ecosystems and the disturbance processes that they depend upon are out of character relative to historic reference conditions across most of the MTNF. The 2005 Forest Plan allocates 29 percent of the forest, or 438,000 acres, to Management Prescriptions 1.1 and 1.2, which emphasize the restoration of globally significant natural communities. A basic restoration premise is to restore the structural vegetative condition and emulate historical disturbance processes and functions under which natural communities evolved and were adapted. Implementation strategies and silvicultural methods are discussed.

Keywords: Ozark Highlands, ecosystem restoration, natural communities, Mark Twain National Forest

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INTRODUCTION

The 1.5 million acre Mark Twain National Forest (MTNF) encompasses 9 separate units that span thirteen ecological subsections stretching over 200 miles of the Ozark Highlands. It contains unique, diverse, ecologically complex and globally distinct biological areas. Historically, this landscape contained a diverse assemblage of savanna, woodland, forest, glade, fen, spring, stream and cave natural communities that provided habitat for nearly 2,000 vascular plant species and nearly 700 vertebrate animal species including over 150 endemics (Ozark Ecoregional Assessment Team, 2003). Some 12,500 years ago, Native Americans occupied Missouri's landscape, and influenced the composition, structure and distribution of natural communities (see Wettstaed in Nelson, 2005). The vegetation at the time of European settlement was for the most part composed of essentially indigenous plants and animals that co-evolved and adapted to centuries of aboriginal disturbance regimes, especially fire (Guyette 1991, Nelson 2005). Wind and storm events, and native grazing ungulates such as deer, elk, and woodland bison also influenced the distribution, productivity, patterns and composition of vegetation making up some 65 distinct natural communities on the Forest.

By the early 1800s, Native Americans and their practices of landscape burning, hunting-gathering and farming from small villages had been largely eliminated from the Ozark region. Increasing settlement by Europeans brought industrial logging (resulting in the nearly complete unregulated highgrading of virgin timber), lead mining, charcoal production, subsistence farming, and open range grazing by domestic livestock (Smith 2003). Significant soil erosion problems occurred after the timber boom, mainly as a result of poor farming practices, overgrazing and annual burning. By the 1930s, much of the most productive soil resource was literally downstream due to severe sheet and gully erosion. Prior to European settlement the stream channels were narrow and deep, but over the years the river channels became wider, shallower and unstable, and large gravel bars developed along the stream beds (Smith 2003). The landscape the settlers depended on was exhausted by the early 1900s, and eventually some of these self-reliant folk were forced to turn to the government for assistance (Smith 2003).

The U.S. Forest Service started purchasing property in Missouri in the fall of 1933, and began the process of landscape stabilization by planting trees, instituting a grazing policy, and excluding fire. Missouri's fire prevention program was especially aggressive as state and federal agencies were convinced that burning by locals was destroying timber and wildlife habitat, and was contributing to soil erosion. However, these early efforts at management could not alter the fact that decades of abuse and disruption of disturbance regimes led to the degradation of natural communities. This degradation resulted in loss of biodiversity, unnatural succession, soil erosion, reduced productivity of timber products, oak decline and fuel buildups (USDA Forest Service 2005a). On the MTNF, eight decades of management since the 1930s have gone a long way to stabilize the landscape and soil resource, and to reestablish a productive and sustainable forest condition. But these forests are substantially different from the ecosystems that existed prior to European settlement, largely because key disturbance processes, notably fire, have been effectively removed from the landscape. As a result, ecosystems that are adapted to fire, and the important flora and fauna they contain, are dramatically underrepresented in the modern forest landscape.

In 2002 the MTNF began the process of revising the 1986 Forest Plan. Among the revision topics identified as a need for change was ecological sustainability and ecosystem health. The MTNF identified opportunity areas to conserve the best, most viable arrays of ecosystems, plants and animals. Opportunity areas were delineated using information from the following sources:

- Ozarks Ecoregional Conservation Assessment (TNC 2003)
- The Terrestrial Natural Communities of Missouri (Nelson 2005)
- The Missouri Natural Areas Program
- Partners In Flight; Ozark-Ouachita Physiographic Region
- Missouri Resource Assessment Partnership
- The Missouri Grasslands Coalition

The MTNF worked closely with the Missouri Chapter of The Nature Conservancy (TNC) to gather information on conservation areas and discuss the results of the Ozarks Ecoregional Conservation Assessment (OECA) (Ozarks

Ecoregional Assessment Team 2003). This assessment identified globally significant species, natural communities, and ecological systems in the Ozark ecoregion, and helped determine the spatial configuration that would most efficiently conserve viable examples of globally significant biodiversity features. The MTNF used the OECA to focus conservation planning and management efforts on specific opportunity areas on Forest lands, also known as "portfolio sites." As a result of these discussions and planning, the Forest Planning Team, and appropriate ranger district personnel, identified 19 separate areas to be managed for ecosystem restoration (see Appendix A of the 2005 Management Plan for the MTNF).

These 19 areas, comprising 29% of the forest (438,000 acres), are in Management Prescriptions 1.1 and 1.2, which emphasize restoration of globally significant natural communities. The plan includes specific objectives for restoration work on 100,000 to 150,000 acres during the planning period (ten years). These objectives were formulated by identifying minimum/maximum viability targets for natural community types. Natural community types include prairies, glades, savannas, open woodlands, closed woodlands, forests and fens. A basic management premise is to restore the structural vegetative conditions and maintain (emulate) the historical disturbance processes and functions under which natural communities evolved and were adapted. Fire disturbance is emphasized because of its historical role in the region, and its influence on natural community structure and species composition.

NATURAL COMMUNITY TYPES ON THE MTNF

In Missouri state and federal land managers have been classifying, inventorying, describing and restoring Ozark ecosystems as Natural Areas for 30 years (Nelson 2005). Missouri Natural Areas, and select locations in state parks and the MTNF, have been managed for their natural qualities for at least 25 years by emulating historic disturbance processes. These sites serve as reference benchmarks, exhibiting ecological conditions that existed prior to European settlement. Using these benchmarks as desired conditions, prescriptions can be written to move the current conditions of the project area towards those desired conditions. Project monitoring also relies on these reference sites to compare species richness and other ecological parameters.

Using the reference sites and other literature on the structure and dynamics of forest ecosystems, the MTNF developed a set of parameters, which were included in the forest plan, to describe the desired condition of these communities of interest.

- A Forest is a relatively large area (typically over 10 acres) dominated by trees forming a closed canopy and interspersed with multilayered shade-tolerant sub-canopy trees, shrubs, vines, ferns, and herbs. Most forests occurred on steep north-facing slopes, deep coves and in broader river valleys thereby limiting the effects of historic fires.
- Woodlands are highly variable natural communities with a canopy of trees ranging from 30 to 80 percent closure with a sparse understory (or midstory) and a dense ground flora rich in forbs, grasses and sedges. Two types of woodlands, open and closed, are managed on the MTNF. The gently to moderately dissected plains and hills of the Ozarks were conducive to the frequent spread of historical fires on which woodlands depend (Nelson 2005).
- Savannas are fire-dependent grasslands interspersed with open-grown scattered oak trees, groupings of trees of various age, and shrubs. The tree canopy cover is generally less than 30 percent. Broad, gently dissected plains facilitated more intensive and frequent fires, which resulted in the development of savanna landscapes.
- Prairies are fire dependent natural communities dominated by perennial grasses and forbs with scattered shrubs and very few trees (less than 10 percent cover). Intense, frequent surface fires propagated across the nearly level to gently rolling Missouri plains thereby resulting in the development of treeless prairies.
- Glades are open, rocky barren areas dominated by drought-adapted forbs, warm season grasses and a specialized fauna. They occur as openings primarily on southwest-facing slopes across the otherwise wooded Ozarks landscape. Trees and shrubs occur on glades, but are not dominant unless overgrazing and/or disruption of natural fire regimes have resulted in invasion by woody species like red cedar.

Presently, many natural communities on the MTNF are out of character (see Table 1); basal areas are too high and the tree canopies are closed with associated dense shading,

Table 1. Range of Ecological parameters for respective natural communities in Management Prescriptions 1.1 and 1.2

Natural Community Types	% canopy	Basal Area	Understory	Aspect, Slope, roughness	Shrub layer	Structural age/growth stages	Ground organic layer	Percent ground cover	Patch size
Prairie	<10	NA	NA	All aspects gentle slopes,	Sparse	Grassland with f few scattered shrubs and trees	Grass, sedge, and forb cover	90-100	10 to 200 acres
Savanna	10-30	<30	Scattered oaks and Shrubs	Broad ridges, all aspects, Gentle slopes	Dense, mostly scattered oaks and other shrubs	Shrub oak/pine covering 10-25% of area	Grass, sedge and forb cover	90-100, grasses dominant	50 to over 1,000 acres
Open woodland	30-50	30-50	Mixed shrubs, early seral	Southwest-facing to upper ridges, gentle to steep, plains and hills	Dense, mostly scattered oaks and various shrubs	Shrub oak/pine covering 10-25% of area, even-aged stands	Grass, sedge, and forb cover, little accumulated leaf litter	60-80 grasses dominant	100 to over 1,000 acres
Closed Woodland	50-80	50-90	Early- to mid-seral trees	Upper ridges to s base of north-facing slopes, gentle to steep, hills and break	Sparse, mostly scattered oak and various shrubs	Shrub oak/pine in 5-10% of area, even-aged stands	Shallow leaf litter, mixed grasses sedges and herbs	80-100	100 to over 1,000 acres
Upland forest	80-100	80-100	Shade-tolerant shrubs and small trees	Generally north-facing slopes, steep to very steep, hills and breaks	Sparse, scattered, vines present	Oak and mixed species of variable age, small isolated gaps 1-5 acres	Moderately deep leaf litter	50-70	10 to 100 acres
Bottomland Forest	80-100	90-100	Shade-tolerant shrubs and small trees	North-facing slopes, very steep or broad-level floodplains, hills and breaks	Sparse, vines present	Multi-layered, uneven-aged, few gaps	Deep leaf litter, ephemeral herbs	50-70	10 to 500 acres
Fen	<10	NA	NA	Toe slopes, ravines and floodplains	Dense to sparse or none	Shrub thickets to open herb/sedge meadows	Shallow marly to deep muck	90-100	<100 sq ft. to 15 acres
Glade	<10	NA	Small shrubs and trees, rock outcrops	Generally southwest-facing but all aspects on igneous and White River, steep to very steep, hills and breaks	Variable	Variable exposed	Sparse to dense grass cover, mineral soil often	30-90 grasses dominant	½ to 300 acres

heavy fuel loading from deep leaf litter, and sparse ground cover. As well as being out of character with respect to the range of ecological parameters, many areas are also out of character with respect to overstory species. An estimated 4.2 million acres of shortleaf pine forest and 2.4 million acres of mixed pine-oak forest existed prior to 1880 (Liming 1946). Based on 2003 Forestry Inventory and Analysis (FIA) data across 29 counties covering the MTNF, the extent of shortleaf pine and mixed pine-oak is 545,929 acres--which equals 8% of the pre 1880 estimate. Much of the MTNF that was historically populated by shortleaf pine or shortleaf pine/white oak mixtures is now dominated by trees of the red oak group--mostly black and scarlet oak (USDA Forest Service, 2005a).

RAPID ECOLOGICAL ASSESSMENTS

Although there are 438,000 acres covered by MP 1.1 and 1.2 where restoration is the main emphasis, it may not be possible or even desirable to prescribe restoration work for all of those acres. It is especially important to focus our efforts on those ecological areas where restoration goals are most likely to succeed. The MTNF uses "rapid ecological assessments" (REAs) to assist in identifying those areas to best do the work. The REA evaluates the ecological condition, or health, of the landscape by evaluating species richness and restorability, concentrations of habitat variety, and how close the area is to the desired condition parameters for respective natural communities. The REA process uses a combined GIS and field evaluation approach conducted by the Forest Ecologist, Forest Botanist and respective ranger district staff to rapidly evaluate (within 2-3 months) larger management areas in MP 1.1 and 1.2. Because regular and frequent prescribed burning will be required to restore most natural community types, the REA also considers the feasibility of conducting large landscape-scale prescribed burns. The REA results in a report that lists high priority sites (future project areas) with restorable natural communities, sensitive areas, threats and risks, and operational management recommendations. Once identified by the REA, projects can be planned in those high priority areas that have the most opportunity to accomplish restoration work using timber sales, and the Knutson-Vandenberg (KV) funds that those sales generate—which promotes opportunities for far greater habitat restoration than could be accomplished with appropriated dollars alone (Guldin and Graham 2007).

MAPPING OF NATURAL COMMUNITY TYPE AND HISTORIC VEGETATION

A process to develop a detailed natural community map for respective projects in MP 1.1 and 1.2 is outlined in the implementation guide for the 2005 forest plan. This process combines GIS mapping, modeling of the landscape, GLO notes and field ground truthing. Part of this analysis process also includes comparison of the natural community types with current vegetation conditions and knowledge about a local ecoregion's historic disturbance processes. Information and data that can be used include: land survey records, direct evidence of remnant shortleaf pine stumps from the initial cut of virgin timber in the late 1800s to early 1900s, studies/inventories of natural areas, writings of early travelers, county histories, naturalist accounts (early botanists, birders, etc.), archaeological/paleontological studies, and various terrain/historic vegetation models. Armed with this information, plus a knowledge of the description for the respective natural communities (Nelson 2005), resource managers can make informed decisions about on-the-ground natural community mapping.

Forest staff also use "indicator plant species" to help determine where the respective boundaries for various natural community types fall on the ground. While plant species occurrences vary widely across ecological subsections of the Missouri Ozark Highlands, certain plant indicator species are surprisingly faithful to a respective natural community type. For example, the presence of cream white indigo (*Baptisia bracteata*), turbinate aster (*Symphytrichum turbinellum*), little bluestem (*Schizachyrium scoparium*), lowbush blueberry (*Vaccineum pallidum*) and bristly sunflower (*Helianthus hirsutus*) suggests an open woodland that would be dominated by shortleaf pine (*Pinus echinata*), post oak (*Quercus stellata*) and black hickory (*Carya texana*); similarly, plants such as the elm-leaved goldenrod (*Solidago ulmifolia*), tick trefoil (*Desmodium nudiflorum*), Boscii's panic grass (*Dichanthelium boscii*) or woodland brome (*Bromus pubescens*) indicate a closed woodland that would be dominated by white oak (*Quercus alba*) and mockernut hickory (*Carya tomentosa*).

SILVICULTURAL STRATEGIES

A primary focus of natural community restoration is emulating critical disturbance processes on a landscape scale. The 2005 Forest Plan for the MTNF provides quantifiable objectives for natural community types that are distributed across eight ecological subsections. Thus, landscapes in need of treatment across these subsections may range from several hundred to several thousand acres. How well natural community remnants recover (assuming we already have sufficient biotic components) depends on whether managers can restore desired physical/structural conditions and distribution patterns, reverse historic damage and then replicate historic disturbance processes across areas large enough to be ecologically and functionally meaningful when restored (see McCarty in Nelson 2005). The goal of restoring landscapes is met by executing silvicultural prescriptions in each of the stands that comprise the landscape under restoration.

The silvicultural prescription to apply in a given stand depends on three key sources of information: the natural community map for the region, the desired condition for the given natural community (Table 1), and the existing condition of the given stand determined from stand examinations and inventory data. Knowing these three elements allows the silviculturist to prescribe treatments to move the area from the existing condition towards the desired condition. Simply put, most management practices fall into some combination of thinning, regeneration, planting, prescribed burning and perhaps control of exotic species. Important measures of success in restoring natural communities include species richness (especially those plant species found in the ground layer), plant and animal indicators (particularly animals adapted to variable structural conditions), overstory composition, and absence of exotic species.

A real challenge to silviculturists is whether it is more important to restore those overstory species (in the case of shortleaf pine) now removed from most woodlands, or the more sensitive diversity of ground cover plant species, many of which are species of conservation concern. The answer is that both are important and the approach depends on the overall ecological conditions of the area to be treated.

Prescribed Burning

Effective application of prescribed burning is critical in restoring natural communities, and maintaining them once they are restored. "An irrefutable body of evidence exists that the biological landscape of the Ozarks reflects the effects of millennia of frequent, low intensity, dormant season fires set by humans" (Ozarks Eco-regional Assessment Team 2003). Many other documents cited in the new Forest Plan (US Forest Service 2005a 2005b) and Nelson (2005) strongly substantiate this evidence. Fire remains vital to the form and function of all types of savannas, woodlands, glades, fens and prairies on the MTNF. The absence of fires is one of the most significant change agents of modern times, second only to the outright destruction of native vegetation due to overgrazing, habitat conversion to other uses and exotic species invasion. Further, there is no similar or substitute treatment that can replicate the complicated chemical, biophysical and ecological effects of the burning from which ecosystems and plants/animals have evolved and adapted over millennia. However, it is not feasible nor possible to apply the burning process to all ecosystems on the MTNF. It is no accident that planning for ecosystem restoration on MTNF involved prioritization of applying prescribed fire to Management Prescriptions 1.1 and 1.2, and to those high priority areas identified in the Rapid Ecological Assessment process.

The reintroduction of fire requires an understanding of fire frequency and interval, fire intensity, and seasonal and spatial patterns of fire (see discussion by McCarty in Nelson 2005 regarding Missouri fire regimes). The progression of desired change and effects of fire application for restoring woodlands is well documented in Missouri (Nelson 2005). This information can be used to develop and implement an ecologically-based prescribed burn program to restore and maintain fire-adapted Ozark woodlands, savannas and glades (Table 2). A major challenge to resource managers practicing ecosystem restoration is how to model fire in a way that mimics the different roles that an average or extreme fire event played historically. This becomes especially difficult in a rural environment now fragmented by homes, farms and livestock; thus, it becomes more challenging to plan and execute the types of prescribed burns that will achieve conditions worthy of restoring and sustaining an ecosystem while protecting other values (timber, wildlife, soils, air quality, homes, etc.).

Restoration Thinning

Among the first treatments required in savanna and woodland restoration is a reduction in stem density in both the overstory and the midstory. The definition of restoration thinning is "thinning to a specified basal area to restore the natural community type." The basal area specified (see Table 1) for savannas and woodlands is much lower than for a regular commercial thinning, and often would fall below the lower threshold for optimal utilization of the site from a traditional timber management perspective. The advantage of a low residual basal area is that resources become available for understory flora, a key component of composition and structure of desired woodlands and savannas.

However, it may be prudent to accomplish this reduction in stem density with a two-step process. Reducing the basal area to restoration levels in a single thinning can often result in unacceptable logging damage in the residual stand, and may predispose the stand to other kinds of damaging agents such as wind-throw and/or the spread of undesirable invasive plants. In addition, excessively heavy thinning can create unintended developmental dynamics such as a flush of hardwood growth from advanced regeneration and sprouts that would overwhelm the establishment of desirable ground flora (where problematic, the advanced regeneration of undesirable sprouting can be controlled through various means including herbicide application to stump sprouts, repeated prescribed burning and/or hardwood understory reduction treatments). Also, initial burns in thinned areas with accumulated deep leaf litter may be intense enough to reduce the basal area further than desired.

If it exists on the site, the desired historic vegetation (shortleaf pine, white oak) can be favored. Along with the thinning, a burning schedule should be established to reduce the leaf litter and encourage establishment of grasses, sedges and herbs. As an example, consider an open woodland site with a basal area of 120-150 ft^2/ac. The restoration prescription suggests an initial thinning of sawtimber to a residual basal area of 70-80 ft^2/ac of sawtimber followed by a series of three prescribed fires at 3-4 year intervals. The second thinning in about ten years would then further reduce the basal area to about 30-50 ft^2/ac.

Table 2. Estimated fire frequency by natural community based on 30 years of applied ecosystem restoration.

Natural Community	Restoration Fire Landform	Maintenance Fire Frequency	Frequency
Prairie	Broad, level Plains	1-3 years	2-4 years
Savanna	Rolling Plains (lower intensity)	1-4 years	3-5 years
Open woodland	Dissected Plains and hills	1-3 years	3-7 years
Closed woodland	Hills and Breaks	2-3 years	3-10 years
Forest	Breaks, Floodplains	5-25 years	10-25 years
Glade	Dissected hills	2-3 years	4-10 years

The thinning and burning prescription is simple to implement if the trees being thinned can be sold commercially, and if KV funds can then be collected and used to accomplish the burning. However, if the trees are not marketable, funding the thinning and the burning can be much more difficult. Options might include using fuels dollars to execute prescribed burning that may accomplish some thinning, as well as removing the leaf litter; doing non-commercial thinning and prescribed burning using KV funds generated from a timber sale in other stands within the sale area; or using appropriated funds. We also continue to explore the feasibility of marketing small diameter, low quality hardwoods for biofuels and other products.

Oak Decline Salvage

Black, red and scarlet oak now dominate many former open woodlands once dominated by shortleaf pine, post oak and white oak (Law et. al 2002). This out-of-character dominance has contributed to an increase in oak decline, a complex condition caused by the combination of degraded soils, removal of those dominant overstory trees better suited to dry open woodlands (Kabrick et. Al 2002), and removal of herbaceous ground cover by historic overgrazing. Many of these black and scarlet oak stands are also overstocked and reaching the end of their life span, creating conditions ripe for oak decline. An estimated 400,000 acres of the MTNF are currently at moderate to severe risk of oak decline.

Stands that show evidence of moderate to severe oak decline may be salvaged to recover economic value that would otherwise be lost, and to help move the forest towards the desired ecological condition for the given natural community in which the stand is located. Prescriptions vary depending on the percentage of black and scarlet oak in the stand. Salvage and sanitation cutting in stands with lower percentages of black and scarlet oak may result in residual basal areas that approach those desired in restoration thinnings or shelterwoods. Harvesting in stands with larger percentages of black and scarlet oak to be salvaged may mimic seed trees or clearcuts. Reforestation work such as artificial regeneration with shortleaf pine, or release of appropriate advanced shortleaf pine and/or oak regeneration may be required. Note that salvage activities are not subject to limits on the maximum size of temporary openings that restrict even-aged regeneration harvests.

Regeneration

The rotation age in MP 1.1 and 1.2 is increased from that in the general forest area—from 70 to 80 years for red, black and scarlet oak, from 70 to 100 years for shortleaf pine, and from 90 to 120 years for white oak. The 2005 Forest Plan projects 34,500 acres of reproduction cutting during the 10-year planning period intended to redress the imbalance of age classes found on the MTNF. Disturbance ecology and fire scar data suggest that those natural communities most affected by moderate to high intensity fire and wind damage regenerate themselves in uniform age cohorts that can be recreated using even-aged silvicultural methods. Missouri's variable topography coupled with cyclic severe droughts likely created a landscape of variable even and uneven-aged natural communities. Fire scar studies across Missouri showed even-age cohorts for post oak and some shortleaf pine stands likely resulting from more intense or mixed severity fires that occurred during 10-20 year drought cycles (Guyette 2007). The standards and guidelines in MP 1.1 and 1.2 allow the use of even-aged silvicultural methods to create forest openings up to 500 acres in size on the Houston/Rolla and Ava/Cassville/Willow Springs districts, primarily because the interpretation of general land office survey notes suggest patterns of larger historical barrens. Accordingly, most regeneration in savannas and woodlands is projected to be obtained using even-aged silvicultural prescriptions. Clearcutting, with scattered relict tree and reserve tree retention, will be used in natural community restoration, and planting to restore shortleaf pine will be required in many areas. Most of the uneven-aged silvicultural prescriptions employed will be practiced in upland forest natural communities.

SUMMARY AND CONCLUSIONS

In the restoration of natural communities, silvicultural prescriptions are intended to move stands from their existing condition toward a desired future condition that better achieves important ecological objectives. The silvicultural techniques practiced are similar to those routinely used in traditional forest management. However, rather than maximizing production of outputs such as timber products or specific wildlife habitats, the emphasis is on regulating structure and composition, and emulating historic disturbance process that create diverse, healthy, natural communities. Moving towards the desired condition may take 15 to 25 years for ground cover and more than 100 years on a landscape scale for the composition and structure of respective canopy characteristics. The ecosystem management approach should create diverse, productive, and healthy ecosystems, and perhaps capture 80% of Ozark biodiversity, including most species of conservation concern. Production of timber and other goods and services may be a by-product of management activities, but should not be considered an end result.

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Integrating Land and Resource Management Plans and Applied Large-Scale Research on Two National Forests

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ABSTRACT

Researchers working out of the Southern and Northern Research Stations have partnered with two National Forests to conduct two large-scale studies designed to assess the effectiveness of silvicultural techniques used to restore and maintain upland oak (*Quercus* spp.)-dominated ecosystems in the Cumberland Plateau Region of the southeastern United States. We based both large-scale studies on approved Land and Resource Management Plans (LRMP), but the projects being studied have been implemented under separate authorities. On the Bankhead National Forest, the District Ranger and staff have worked with a formal forest liaison board to gain interest and acceptance of the new Forest Health and Restoration Project to be implemented under traditional Forest Service authorities. The impetus for the Daniel Boone National Forest study was also consistent with the LRMP developed in collaboration with interested publics, but the study projects were implemented using authorities granted by the Healthy Forest Restoration Act of 2003. In both studies, researchers are assessing response to silvicultural techniques used to control species composition and alter habitat for wildlife and restoration purposes, including intermediate treatments (e.g., thinning and prescribed burning) and regeneration harvests (e.g., shelterwood). Forest managers and researchers are interested in modeling changes in overstory cover and composition, fuel loading, residual tree health and vigor, available light, and understory regeneration. In addition, the response of the herbaceous component, the response of the avian and herpetological populations, and the response of habitats will be assessed using a multidisciplinary approach. We will use results from these studies to help forest managers monitor and predict forest response and to develop habitat and regeneration models for each forest system. We discuss the logistics, challenges, and triumphs of implementing such large-scale projects, and joint efforts for science delivery.

Keywords: silviculture, oak regeneration, national forests, fire

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INTRODUCTION

Increased human population densities, loss of the American chestnut (*Castanea dentata* L.), and a variety of land-use changes have caused current forest ecosystem composition and structure in the southeastern United States to vary widely from that thought to exist in pre-Columbian times (Levitt 2002). Attempts to restore ecosystems in the region to a former state or to manage these ecosystems in their current state will require a thorough understanding of how various environmental factors affect ecosystem characteristics.

Oak-dominated forests are the most abundant cover types in the Southern Appalachian Highlands hardwood region (Johnson et al. 2002). Silvicultural practices to regenerate hardwoods have been the subject of much research (Loftis and McGee 1993; Johnson et al. 2002; Hannah 1987; Roach and Gingrich 1968; Sander 1977, 1988). The clearcutting method, variations of the shelterwood method, and uneven-aged methods can be applied to regenerate upland hardwood stands in both research and operational settings, but these approaches have not been uniformly successful in restoring native oak species, particularly on more mesic sites (Beck 1988).

Prescribed burning and thinning are intermediate treatments that when applied in oak-dominated stands can improve forest health, increase the abundance and size of oak regeneration, and control species composition. A more site-specific method to increase oak regeneration potential is the use of herbicides to alter the light environment in the understory to enhance the survival and growth of oak seedlings (Loftis 1990). However, prescriptions for herbicide use were developed specifically for use on sites of higher than average productivity, and herbicide use on National Forests is often administratively restricted. Generally, a combination of various techniques (herbicide treatments, prescribed fire, thinning, and shelterwoods) applied at intervals (depending on vegetation response and site characteristics) may have the most profound effect on community structure and composition (Franklin et al. 2003; Loewenstein and Davidson 2002; Lorimer 1992; Nowak et al. 2002). However, the interactions of these techniques are largely unknown.

The main problems limiting the adoption and use of silvicultural techniques by National Forest System (NFS) managers are administrative constraints, social influences on management decisions, and imperfect transfer of the knowledge from researchers to forest managers. Researchers can assist NFS managers by conducting collaborative research with them and by sharing up-to-date technology. If researchers and NFS managers jointly test the effectiveness of new or modified silvicultural techniques on National Forest lands, it will likely enhance the ability of NFS managers to predict the outcomes of management practices and efficiently meet their forest management objectives.

We implemented two large-scale studies on the Daniel Boone and William B. Bankhead National Forest in the Cumberland Plateau Region of Kentucky and Alabama. The overall goal of the studies was to determine the effects of various silvicultural techniques on restoring or maintaining the oak component of the forest. The studies are still ongoing, but in the process of establishing the studies we learned many practical lessons about conducting large-scale, long-term research projects on National Forest lands.

The first study we discuss was implemented on the William B. Bankhead National Forest (BNF) in north-central Alabama. Decline and death of southern yellow pines due to the southern pine beetle (*Dendroctonus frontalis* Zimmermann) has provided impetus for this study, which examines the effectiveness of the BNF's current management techniques, as detailed in their latest Forest Health and Restoration Project (FHRP) (USDA Forest Service 2003), in restoring the oak component. The second study we discuss was implemented on the Daniel Boone National Forest (DBNF) in south central Kentucky. The forest may soon be affected by gypsy moth and the often-related disease complex, oak decline, and these impacts would reduce the ability of managers to provide expected benefits derived from the forest. This study was implemented under the auspices of the Healthy Forest Restoration Act of 2003 (HFRA) and was designed to examine the effectiveness of various silvicultural techniques in restoring and maintaining the oak component and improving forest health conditions on the DBNF. The use of these silvicultural techniques is described in the DBNF's LRMP (USDA Forest Service 2004).

The purpose of this paper is to discuss the research goals and general experimental design of each study, the challenges and successes of study implementation, and lessons learned through participation in cooperative research projects of the NFS and Research branches of the Forest Service. We hope to provide general guidelines for other large-scale replicated research projects implemented on NFS lands.

METHODS

Study Areas

William B. Bankhead National Forest

The BNF, established by proclamation in 1914, has a long history of repeated logging and of soil erosion caused by poor farming practices during the Depression era. The 180,000-acre BNF is in the Strongly Dissected Plateau sub-region of the Southern Cumberland Plateau, within the southern Appalachian Highlands (Smalley 1982). In 1975, the 12,000-acre Sipsey Wilderness Area in the northwestern portion of the forest was established to preserve a natural forest environment. Today the Sipsey Wilderness totals 25,202 acres. The forest contains the Black Warrior Wildlife Management Area and the Sipsey Wild and Scenic River. Base age 50 site indices for loblolly pine (*Pinus taeda* L.), red oaks (*Quercus rubra* L., *Q. velutina* Lamarck, *Q. coccinea* Muench., *Q. falcata* Michx.), and white oaks (*Q. alba* L., *Q. prinus* L.) are 75 ft, 65 ft, and 65 ft, respectively (Smalley 1982).

Daniel Boone National Forest

Two separate tracts were merged and proclaimed as the DBNF in 1966. The area proclaimed as the Cumberland National Forest in 1937, a narrow strip along the western edge of the Cumberland Plateau, contains the study area. The rugged eastern area is maturely dissected and characterized by narrow, winding ridgetops, steep sideslopes, and narrow winding valleys (Smalley 1986). Within the forest are 18,000 acres of designated Wilderness and 19 miles of Wild and Scenic Rivers. Base age 50 site indices for shortleaf pine (*Pinus echinata* Mill.), red oaks (*Quercus rubra* L., *Q. velutina* Lamarck, *Q. coccinea* Muench.) and white oaks (*Q. alba* L., *Q. prinus* L.) are 60 ft, 55 ft and 60 ft, respectively (Smalley 1986).

The study was implemented on the Cold Hill Area of the London Ranger District of the DBNF. The treatment units are located on the Central Escarpment (221 Hb) landtype association, which is transitional between the Highland Rim and the Cumberland Plateau (Taylor et al. 1997; USDA Forest Service 2004).

Study Design

William B. Bankhead National Forest

The BNF study employs a randomized complete block design with a 3 by 3 factorial treatment arrangement and four replications of each treatment. The treatments are three residual basal area treatments (50 ft^2a^{-1}, 75 ft^2a^{-1}, and an untreated control) with three fire frequencies (frequent burns every 3-5 years, infrequent burns every 8-10 years, and an unburned control; Table 1). Each treatment is replicated 4 times, for a total of 36 treatment units. Treatments are representative of management practices described in the BNF's FHRP for restoring oak forests and woodlands.

Table 1--Disturbance treatments for silviculture research on the Bankhead National Forest, Alabama

	DISTURBANCE TREATMENTS
1	No Burn and No Stand Density Reduction
2	No Burn and 75 ft^2a^{-1} Residual Stand Density
3	No Burn and 50 ft^2a^{-1} Residual Stand Density
4	Infrequent Burn (8-10 yrs) and No Stand Density Reduction
5	Infrequent Burn (8-10 yrs) and 75 ft^2a^{-1} Residual Stand Density
6	Infrequent Burn (8-10 yrs) and 50 ft^2a^{-1} Residual Stand Density
7	Frequent Burn (3-5 yrs) and No Stand Density Reduction
8	Frequent Burn (3-5 yrs) and 75 ft^2a^{-1} Residual Stand Density
9	Frequent Burn (3-5 yrs) and 50 ft^2a^{-1} Residual Stand Density

Criteria for stand selection were based on species composition, stand size, and stand age. Treatment units for the study were located on upland sites currently supporting 20 to 35-year-old loblolly pine plantations with a significant hardwood component in the understory. Treatment units were at least 22 acres in size with basal areas ranging from 80 to 140 ft^2a^{-1}.

Prescribed burning was conducted during the dormant season (December-February) using backing fires and strip headfires to ensure that only surface fire occurred. Commercial thinning was conducted by marking from below smaller trees or trees that appeared diseased or damaged. Hardwoods were preferentially retained. Stand density reduction treatments were completed prior to the initiation of the burning treatments (thinning conducted from June through November).

Daniel Boone National Forest

The study design for the DBNF study was a randomized complete block design with a 2 by 5 factorial treatment arrangement with two site types (dry-mesic and dry-xeric) and five disturbance treatments (shelterwood with reserves, oak shelterwood, B-line thinning, oak woodland, and a control; Table 2). The treatments were replicated three times, for a total of 30 treatment units.

Criteria for stand selection and treatment assignment were based on several factors including administrative constraints, proximity to road infrastructure, stand history, ownership boundaries, topography, soil type, and species composition. All treatment units were located on broad ridges, were dominated by oak species, had basal areas ranging from 70 to 150 ft^2a^{-1}, and were occupied by stands between 70 and 150 years old.

The shelterwood with reserves treatment (1) will leave trees that will promote good forest health conditions and improve habitat for wildlife and plant species that benefit from open, low basal area forest conditions. In the oak shelterwood treatment (2), triclopyr ester will be applied as a thinline basal bark treatment to trees less than 3 inches dbh, of undesirable species. Trees greater than 3 inches dbh in the mid- and understories will be treated with stem injection. Undesirable tree species include red maple, which has little benefit to wildlife, and trees with unhealthy stems and/or crowns. Thinning to B-line (3), based on the Gingrich (1967) stocking chart, will reduce tree density and allow residual trees to take advantage of improved growing conditions. The oak woodland treatment (4) will be conducted by first thinning to 30-50 ft^2a^{-1} and then conducting prescribed burning every 3-5 years. White oaks will be favored as residual trees to increase hard mast production. Enhancement of spatial and vertical heterogeneity will be an objective of the treatment. The final treatment (5), a control, will not receive a silvicultural treatment, and will be used as a basis of comparisons and evaluations.

Table 2—Disturbance treatments for silviculture research on the Daniel Boone National Forest, Kentucky

	DISTURBANCE TREATMENTS (X= dry-xeric; M= dry-mesic)
1	X- No Burn and No Stand Density Reduction
2	X- Shelterwood with Reserves (10-15 ft^2a^{-1}) Residual Stand Density
3	X- Oak Shelterwood (60-75 ft^2a^{-1}) Residual Stand Density
4	X- B-Line Thinning following Gingrich's Stocking Chart
5	X- Oak Woodland (30-50 ft^2a^{-1}) Residual Stand Density and Frequent Burn (3-5 yrs)
6	M- No Burn and No Stand Density Reduction
7	M- Shelterwood with Reserves (10-15 ft^2a^{-1})Residual Stand Density
8	M- Oak Shelterwood (60-75 ft^2a^{-1} Residual Stand Density
9	M- B-Line Thinning following Gingrich's Stocking Chart
10	M-Oak Woodland (30-50 ft^2a^{-1})Residual Stand Density and Frequent Burn (3-5)yrs

IMPLEMENTATION

We established five 0.2-acre vegetation measurement plots (BNF) or twenty 0.1-acre vegetation measurement plots (DBNF) in each treatment unit and measured plots prior to and just after treatment implementation. All plot centers are permanently marked with rebar, flagging, and GPS coordinates. We permanently tagged all trees 1.5 inches and greater diameter at breast height (dbh) with aluminum tags. We measured and recorded tree species, dbh, crown condition, tree grade, canopy cover, and tree height. In each plot, we also created a 0.01-acre plot where we enumerated regeneration (trees < 1.5 inches dbh) by species and height class. We tagged 5 representative seedlings per regeneration plot and recorded species and measured height and basal diameter (immediately above the root collar). In the BNF study, we sampled fuel loading using line transects and employed electronic recording devices and temperature sensitive paints to quantify fire behavior during burns. We revisit plots once a year near the end of the growing season to document recruitment and tree growth.

Stand selection and data collection for the BNF began in the summer of 2004, and to date, three of the four replications of thinning and burning treatments have been implemented. The final replication will be completed by the winter of 2008. Stand selection and data collection for the DBNF began in the summer of 2005 and treatments should be implemented by summer of 2008.

DISCUSSION

The goal of both studies is to increase the knowledge of silvicultural prescriptions used by forest managers to create and sustain desired habitat conditions. These two studies represent unique opportunities not only to increase the knowledge base, but also to increase collaboration and trust between the two branches of the agency. Researchers learn about how NFS operations such as harvesting and burning are conducted. NFS personnel have personal access to up-to-date research to assist in answering questions and concerns about management strategies. The synergy created by these two projects outweighs the constraints presented.

Although the studies were only recently initiated, we have already gained a great deal of practical experience about coordinating large, long-term research projects between the research and land management branches of the Forest Service. First, we learned it can be done successfully- at least through study establishment in our case. We also learned that it isn't always easy, and sometimes things that seem obvious to one group are not at all obvious to others.

Both the NFS and Research branch personnel need to be aware of and prepared for the long time commitment needed to install silvicultural research studies implemented at the stand level. We quickly learned that installation of large-scale, replicated studies requires a significant amount of coordination and communication between NFS managers and researchers. Having designated leaders from both branches facilitates this coordination and communication. The land management plans are an excellent reference for researchers as they develop study plans. NFS must provide the initial leadership in locating potential study stands and in providing the researchers with appropriate information, such as stand history, stand characteristics, location, and any special or limiting circumstances. In our case the NFS managers are solely responsible for all the coordination and implementation of the harvesting and burning. The DBNF personnel were trained by researchers to assist with data collection, and the research work units provided the Forest funding to support these activities. All data on the BNF were collected by the research teams.

Site selection is one of the most important phases of any large-scale research project. The easiest and best procedure for choosing appropriate stands is to have a NFS representative who is intimately knowledgeable with the Forest vegetation and management history work directly with the researchers. We found that the Continuous Inventory of Stand Conditions (CISC) database maintained by each National Forest was not adequate to guide decisions on study site selection. The CISC database was either in a format not conducive to use for analysis, or it lacked information researchers needed. Consequently, substantial amounts of time were spent conducting field reconnaissance and gathering new data for potential study sites. A large time commitment and effort by all parties should be expected and detailed at the inception of the study.

Clear lines of communication are essential. Data collection for these two studies was intensive and expensive, and communication about timing of treatment implementation is important. Researchers should know when treatments are going to be implemented so they can properly plan collection of pre-and post-treatment data. Researchers should communicate with managers the cost and importance of the data being collected so all can feel a sense of ownership and responsibility. We found it helpful to hold meetings once every 3 to 6 months to share findings to date and discuss logistics of treatment implementation and data collection.

Researchers and NFS staff often struggled with basic communication because the two groups used certain terms differently and because areas were not always referred to by names that were understood by all parties. For example, the researchers used "stand" to refer to the treatment units; the NFS staff referred to these same areas as "units" within compartments. We developed a glossary that defined terms and outlined the names of places within the study areas, including the 'local' names of the prescribed burn units. It is imperative that all involved practice patience and take care to explain terms.

Other complicating factors were the administrative and social constraints on management in National Forests that led to non-random assignment of treatments to selected stands. For example, it was postulated that a certain threatened plant existed in several of the study stands after stands had already been randomly assigned to a treatment. Consequently, the study design was altered post-hoc so the threatened plant would not be disturbed. That had the potential to violate underlying statistical assumptions about randomization of experimental treatment. If the possibility that this plant was present was explained prior to stand selection, some stands could have been removed from consideration for the study. However, the presence of the plant represented a fortuitous chance to study a threatened species, and the design could have been altered to test what habitat conditions the plant required. Other constraints in treatment assignment to stands included proximity to roads and houses for treatments that included commercial harvesting and prescribed burning. Researchers should be aware that treatment assignments to stands available for research on NFS lands may not be totally random, and the potential impact of that must be addressed or reconciled prior to study installation.

Communication between researchers and NFS personnel on potential timing of prescribed burns is crucial, particularly when data must be collected during the burn. The NFS personnel should provide the researchers with information on ideal burning conditions and the researchers should be "on-call" to collect burn data with less than 24 hours notice. Research personnel should obtain proper permission to allow their staff to work before or after regular work duty hours to facilitate collection of data about prescribed burns. Another constraint to data collection was that research personnel were not allowed on burn units without proper safety training and testing. It would behoove researchers who may be actively engaged in fire research to obtain all necessary training, certificates, and equipment needed to work on prescribed burning projects.

Faculty and students from Alabama A&M University are conducting additional research on the BNF plots. This research is in the areas of avian and herpetofaunal ecology, herbaceous characterizations, remotely sensed data analysis, and harvesting operations. The DBNF study also includes additional research conducted by collaborators, including faculty and students from The University of Kentucky, Eastern Kentucky University, and The University of Tennessee. These additional research projects will examine bat ecology, evaluate microclimatic attributes, and establish stand disturbance histories using dendrochronology. Collaborative research greatly benefits all involved parties, but can also greatly complicate design and implementation. We found that we needed to give extra attention to the collaborative partners to ensure that they were cognizant of on-the-ground procedures, such as road and gate usage and safety regulations.

SCIENCE DELIVERY

We found that the science delivery process begins immediately. We have already conducted field tours on both Forests. These tours provide a terrific opportunity for the NFS and Research teams to highlight their partnership, and the breadth and depth of information provided by both teams is impressive. Presentations developed by the research teams have been provided to NFS personnel for additional science delivery. Results are being presented at research conferences and at public and private meetings of lay audiences. Peer-reviewed publications will highlight research results and acknowledge the effort committed by both branches of the agency.

CONCLUSIONS AND RECOMMENDATIONS

We recommend that the following measures be taken when large-scale silvicultural research projects at the stand level are to be implemented on NFS land:

1. Hold frequent meetings and conference calls between researchers and NFS managers, particularly at the beginning of study implementation. Researchers should explain the timing and costs of planned data collection. Managers should explain logistics problems and arrangements, and constraints on implementation of proposed treatments. Take detailed notes and distribute them widely.
2. Both researchers and NFS managers should be clear at the outset about the time required to implement proposed projects. Because these types of studies are generally long-term, the NFS personnel should be prepared to protect and maintain study sites for at least 10 years and to clearly document that necessity through the inevitable personnel changes that will occur over the life of the study.
3. Designate a project leader and alternative from NSF and research. The NFS leader should be knowledgeable about Forest tree species, history, and administrative and social constraints that could affect project implementation. The research leader should be knowledgeable about experimental design, timing and costs of data collection.
4. Researchers should use the LRMP as a starting point in developing study plans. This will ensure that proposed treatments are consistent with the goals and objectives of the Land Management Plan.
5. Clarify terms used frequently, such as stand, unit, and plot. Create a comprehensive study plan with a project glossary that clearly defines both research and NFS terminology used to describe areas in study.
6. Research personnel should obtain required safety training, certificates, and equipment if they are to be on prescribed burn units during burning.
7. Research personnel should obtain proper permission to allow staff to work before or after regular duty hours when necessary (e.g. for collection of prescribed burning data).
8. Researchers should be aware that stands available for treatment may be limited and that assignment of treatments to stands may not be random, which can bias research results. Thus, they may need to revise the study or interpretation of the results accordingly.

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The Potential Role of Red Alder to Increase Structural and Biological Complexity in Even-Aged Hemlock-Spruce Stands of Southeast Alaska

Robert L. Deal and James M. Russell

ABSTRACT

Stand dynamics were evaluated in mixed red alder-conifer forests of southeast Alaska by assessing stand development, tree regeneration, total basal area, and understory biomass in 40-year-old red alder-conifer stands that developed following logging. Overall, these mixed red alder-conifer stands provided more heterogeneous structures, multiple canopy layers, and greater diversity and abundance of understory plants than typically found in similar-aged conifer stands. However, the success of alder plantations and planned silvicultural systems that include mixed alder-conifer forests in southeast Alaska are unknown. Red alder has never been planted in the region and its occurrence has been confined to areas associated with landslides, logging and riparian zones. In 2003, an operational-scale treatment that included alder plantations was replicated at 23 sites within the Tongass National Forest. The alder treatments included a control with no alder planted, a low density planting of 50 tph (trees per hectare) and a high density planting of 200 tph in naturally regenerating 0-5 year old conifer stands. Alder survival of these stands was assessed at 1 and 3 years after planting. Early survival of planted alder seedlings was excellent, but 3-year survival has been much lower and site specific. These plantations will need to be monitored and assessed to determine the long-term potential of mixed alder-conifer stands for improving and restoring forest ecosystems in southeast Alaska. Overall, red alder may serve as an effective tool for increasing biodiversity and improving terrestrial and aquatic habitat when included in regenerating forests following timber harvesting in the region.

Keywords: Stand structure, red alder, Sitka spruce, western hemlock, understory plants, wildlife habitat

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INTRODUCTION

Southeast Alaska is a coastal temperate rainforest region with steeply rising coastal mountains and densely forested islands. These rain forests are dominated by conifer species such as western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) and Sitka spruce (*Picea sitchensis* (Bong.) Carr.), and are characterized by simple species composition with complex age and size structures. Abundant annual precipitation occurs throughout the year along with occasional hurricane force winds. The significance of this climate for the forest is that moisture is generally not a limiting factor for tree regeneration, wildfire is rare, and tree wind-throw is common (Harris, 1989; Nowacki and Kramer, 1998; Kramer et al., 2001). The region's forests contain multi-aged stands created by high-frequency, small-scale natural disturbances such as wind-throw, landslides and endemic tree disease (Deal et al., 1991; Kramer et al., 2001, Hennon and McClellan, 2003). These multi-aged stands contain complex forest structures with many forest canopy layers, abundant understory vegetation, large trees, snags, woody debris, and other important ecological characteristics of old-growth forests (Franklin et al., 1981; Alaback and Juday, 1989; Franklin and Spies, 1991). The understory vegetation is also important as food for herbivores and as cover for ground foraging and nesting birds and small mammals.

Silvicultural systems using clearcutting and natural regeneration have been the dominant timber management practice in southeast Alaska forests since the 1950s (Harris and Farr, 1974). Post-harvest conifer regeneration is frequently abundant (>10,000 trees/ha) with the development of a dense new cohort of western hemlock and Sitka spruce trees. The forest canopy closes in 20-30 years followed by a dense, long-lasting stage of stem exclusion (Alaback, 1982; Deal et al., 1991). Canopy closure eliminates most herbs and shrubs (Alaback, 1982) and attempts to reestablish understory plants through thinning dense young-growth stands has led mostly to conifer regeneration (Deal and Farr, 1994) with little success in herbaceous plant colonization (Tappeiner and Alaback, 1989). These dense young-growth stands have relatively uniform tree height and diameter distributions, and notably lack the multi-layered, diverse forest structures and shrub/herb layers found in old-growth stands (Alaback, 1984; Deal, 2001; Hennon and McClellan, 2003).

This long-lasting stage of stem exclusion has significant implications for understory plants and wildlife such as Sitka black-tailed deer (*Odocoileus hemionus sitkensis* Cowan) that depend on these plants as forage (Wallmo and Schoen, 1980; Schoen et al., 1988; Hanley, 1993). For the first 15 to 20 years after clearcutting, these young-growth stands provide greater understory plant biomass than old-growth stands (Alaback, 1982); however, snow accumulation makes them much less useful for deer habitat in the winter (Kirchhoff and Schoen, 1987; Schoen and Kirchhoff, 1990). Following canopy closure, the resulting dense stands are recognized as having broadly negative consequences for wildlife habitat (Wallmo and Schoen, 1980; Hanley, 1993; Dellasalla et al., 1996; Schoen et al., 1988). Therefore, there is increasing interest in developing forest management practices that maintain or enhance biodiversity and assure long-term sustainability of wildlife and other forest resources.

Red alder (*Alnus rubra* Bong.) is the most common hardwood tree in southeast Alaska, and is frequently found along beaches and streams, on avalanche tracks, landslides and logging skid trails, and as a pioneer species with Sitka spruce (Harris and Farr 1974). In Alaska, pure stands of red alder are rare and alder occurs primarily in mixed stands with shade tolerant conifers including Sitka spruce, western hemlock and western red cedar (*Thuja plicata* Donn ex D. Don). Recent studies of mixed alder-conifer stands in southeast Alaska indicate different successional pathways following clearcutting than the development patterns of pure conifer stands. These mixed alder-conifer stands generally appear to have lower tree stocking and stand density than pure conifer stands of a similar age (40-50 years), with more open forest canopies and more heterogeneous stand structures (Deal, 1997; Deal et al., 2004). Other studies in mixed alder-conifer stands have reported increases in plant species richness and highly productive understory vegetation with biomass similar to that of old-growth stands of the region (Hanley and Hoel, 1996; Hanley and Barnard, 1998; Hanley et al., 2006). Habitat quality for small mammals in even-aged alder-conifer stands may be equal to that of old-growth forests (Hanley, 1996; Hanley and Barnard, 1999). Red alder may convey additional benefits in riparian forests. Riparian forests with some red alder appear to produce more prey biomass and food for fishes than conifer riparian forests (Wipfli, 1997; Piccolo and Wipfli, 2002). If similar processes occur in upland forests, the presence of red alder may increase invertebrate production, providing more food for animals such as birds, small mammals, and fish.

The objective of this paper is to synthesize current information on management strategies for mixed red alder-conifer forests. First, we summarize results from a large recent study of mixed alder-conifer stands in the region, and evaluate the relationship of alder with stand structure, understory plant diversity and biomass, and the potential of alder in conifer-dominated forests for improving wildlife habitat. Second, we assess survival of red alder in young plantations and discuss some management options for increasing stand structural diversity and enhancing biodiversity in young forests in southeast Alaska.

METHODS

Mixed red alder-conifer stands

To examine the relationship between red alder composition and stand structural characteristics, we assessed several mixed alder conifer young-growth stands that naturally regenerated following harvesting. This study was conducted in alluvial valleys in the Maybeso and Harris watersheds, on Prince of Wales Island, Alaska. Nine widely separated stands were selected in the summer of 2000 for intensive sampling. These stands had been clear-cut harvested using cable logging systems about 40 years ago. The amount of alder as a proportion of total stand basal area was the main criterion for stand selection, with an increasing proportion of alder sampled across a compositional gradient ranging from 0 to 86% red alder (Table 1). Stands were selected in upland areas with non-fish bearing headwater streams and site selection included the following criteria:

a) stands that were logged and naturally regenerated;

b) average stand size of 5-10 ha;

c) elevation less than 150 m;

d) moderate slopes of 10-40 percent;

e) no intermediate management activities (e.g., no thinning, planting or alder girdling).

A systematic grid of 20 variable-radius plots was installed throughout each stand to determine average stand composition and to ensure that sites provided a wide range of alder-conifer mixtures.

Fixed-area plots were installed at five randomly selected variable-radius plot locations in each stand. A total of 45 fixed-area plots in the nine stands were used to measure stand density, tree size distribution, patterns of tree mortality, and tree height growth patterns. Detailed sampling methodology is described in Deal et al. (2004). The variable radius plots were used to sample understory vegetation in each of the 9 stands (Wipfli et al., 2002). Total above-ground biomass was measured for herbaceous plants and methodology as described by Hanley et al. (2006). Linear regression analyses were used to test the relations between the proportion of red alder basal area and (1) understory biomass and (2) carrying capacity for deer (Hanley et al., 2006).

Red alder plantations

Following the preliminary results of the previously described study on mixed red alder-conifer stands, the Pacific Northwest Research Station and the Tongass National Forest, designed and implemented a series of operational-scale tests of young-growth management options known as the Tongass-Wide Young-Growth Studies (TWYGS) . One of the treatments in TWYGS included low density and high density planting of red alder seedlings in 0 to 5 year old stands following harvesting (McClellan, 2007). The 0-5 year age class was chosen because conifer regeneration should be small enough to allow for the successful planting and survival of red alder seedlings. The objective of this alder study was to examine red alder seedling survival in conifer plantations. Also, because this was the first large scale planting of red alder in the region, early survival and growth of red alder seedlings could be assessed to determine establishment success of alder in sites where it may not be common. We assessed first and third year survival of red alder over a large geographic range and related survival with site and soil characteristics to determine alder establishment success.

Twenty three sites were selected from a pool of potential sites and selection criteria included a wide geographic range throughout the Tongass National Forest. Operational size treatments with experimental units was generally at least ten acres, elevation less than 350 m, with no intermediate treatments such as weeding or conifer planting. Experiments used a randomized complete block design, with experimental blocks laid out within a single timber-harvest unit, which was divided into three experimental units. In total, 11 sites were established in the southern region of the Tongass National Forest (Ketchikan), 9 sites in the central (Stikine), and 3 sites in the northern region (Chatham).

For the red alder plantations, three treatments were installed that included red alder planted at a low density of 50 tph (trees per hectare), a high density of 200 tph, and a control, where no red alder were planted. Complete sets of treatments were replicated at 23 sites. The red alder planting stock was grown from seed collected in southeast Alaska and contractors planted the trees in recently harvested areas (0 to 5 years old) during the spring of 2003. Survival was assessed from stake-row survival of 100 seedlings in each treatment. Survival was assessed in 2004 and 2006 or one year and three years after seedlings were planted. Elevation, aspect, slope, site index and soil information was determined in the office using Geographic Information System (GIS) map data. Data were analyzed using SAS to determine relationships among first and third year red alder survival for the low and high density planting treatments, and for site class, elevation, slope, aspect and soil drainage. Differences among survival of treatment classes and first and third year survival were determined using Duncan class analyses.

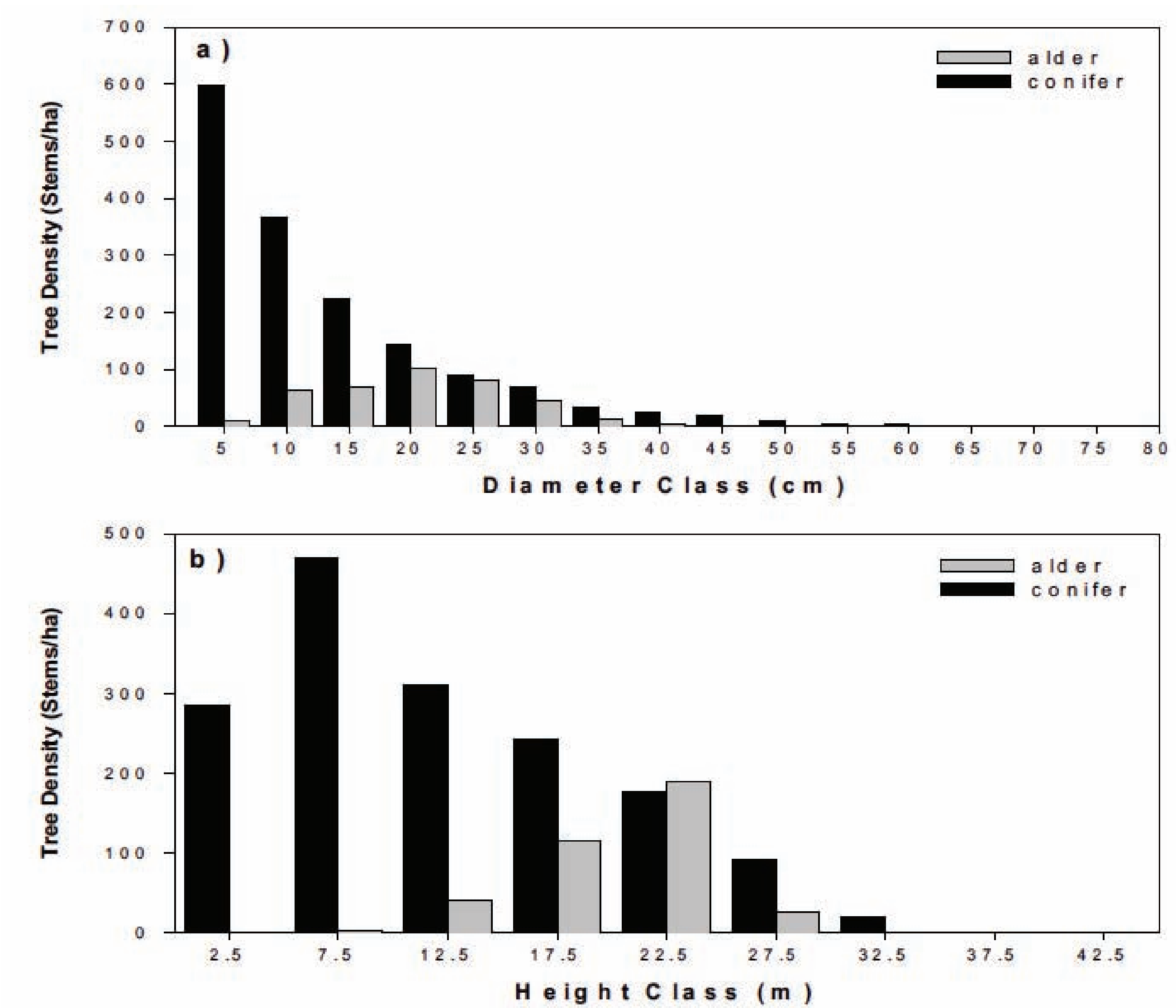


Figure 1— Average tree diameter distribution (a) and average tree height distribution (b) for alder and conifer trees at all nine research sites (modified from Deal et al., 2004).

RESULTS

Density and tree size of mixed red alder-conifer stands

Diameter distributions of trees differed between conifers and alders. Conifers had many more trees in the smallest size classes with decreasing numbers of trees in progressively larger sizes following a reverse J-shaped diameter distribution (Fig. 1a). In contrast, the diameter distribution of alders were bell shaped with most trees in a narrow diameter (20-30 cm) and overstory height (20-25 m) range (Fig. 1a & 1b). The conifers were unevenly distributed with a few taller and larger-diameter trees and several hundred trees in the smallest diameter classes. The mixed alder-conifer stands had more complex diameter distributions than pure or conifer-dominated stands. The alder in these stands provided a different tree size cohort than the conifer-dominated stands that contained numerous small diameter trees. These mixed alder-conifer stands created a multi-layered forest canopy with a few dominant overstory conifers, a mid-canopy level of red alder and a lower canopy level of small diameter conifers (Fig. 1b).

Tree species composition was related to differences in stand basal area and density of trees (Table 1). Analysis of alder composition among the nine stands showed a significant decrease in total stand basal area with increasing proportion of alder basal area ($p = 0.013$; Deal et al., 2004). Tree density, however, was not closely associated with alder composition; a slight and non-significant decline in density occurred with increasing proportion of alder basal area (Table 1; $p = 0.298$). While alder diameters were relatively similar among all of the sites, the average diameter of all conifers significantly decreased with increasing proportion of alder basal area ($p = 0.029$). There was also no significant decease in size of the largest trees with increasing proportion of alder in the stand; the largest diameter conifers appeared to be relatively independent of alder composition with large diameter trees found across a wide range of alder-conifer mixtures. These large conifers could provide an important structural component for riparian zone management to produce large wood for in-stream habitat.

Plant diversity and abundance of mixed red alder-conifer stands

The species richness of these mixed alder-conifer stands was high with 112 species of understory plants including 49 herbs and 15 shrubs. The lowest species richness was in the pure conifer stand; the greatest species rich stands were those stands with more even mixtures of alders and conifers. Understory canopy coverage appeared to be closely related to the amount of alder in the stand, and understory plant cover increased with increasing proportion of alder basal area. Analysis of cover for vascular plants and non-vascular plants (bryophytes) showed different relationships. Non-vascular plant cover was similar for all of the sites, varying from about 35-45%. There was no significant relationship ($r^2 = 0.088$, $p = 0.291$) between the cover of non-vascular plants and the proportion of alder basal area (Fig. 2). In contrast, a strong correlation ($r^2 = 0.701$, $p = 0.005$) occurred between increasing vascular plant cover and increasing proportion of alder basal area (Fig. 2).

Both shrub and herbaceous biomass increased significantly with increasing percentage red alder (Hanley et al., 2006). Total aboveground biomass of understory vegetation

Table 1. Stand density, species composition and soil characteristics of nine sites listed with an increasing proportion of red alder basal area.

Site	Time Since Cutting	Basal Area		Species Composition				Soil Characteristics		
		Stand	Alder	Alder	Spruce	Hemlock	Other	Geomorphic Position	Geomorphic Deposit	Soil Type
	(years)	($m^2 ha^{-1}$)	(% BA)	(trees ha^{-1})	(trees ha^{-1})	(trees ha^{-1})	(trees ha^{-1})			
Upper Good Example	42	59.6	0	0	493	819	0	Mountain Flank	Colluvial/Alluvial	Karta/Tolstoi
Cedar 1&2	40	50.6	3	24	705	2533	488	Mountain Flank	Colluvial	Tolstoi/Rock outcro
Lower Morning	42	48.0	16	348	601	753	468	Mountain Flank	Colluvial/Alluvial	Tolstoi
Lower Good Example	42	60.8	18	451	843	689	27	Mountain Base	Colluvial/Alluvial	Tolstoi/Karta
Mile 22	40	54.4	28	392	265	1415	0	Mountain Flank	Alluvial	Tolstoi/Karta
Big Spruce	40	47.8	33	291	303	672	57	Mountain Flank	Colluvial/Alluvial	Tolstoi/Rock outcro
Group Photo	38	45.5	39	560	519	801	0	Mountain Base	Alluvial	Tolstoi/Karta
Alluvial Fan	42	46.0	64	680	645	333	27	Mountain Base	Alluvial	Tolstoi/Karta
Brushy	40	38.0	86	708	275	389	240	Mountain Flank	Colluvial/Alluvial	Tolstoi/Karta

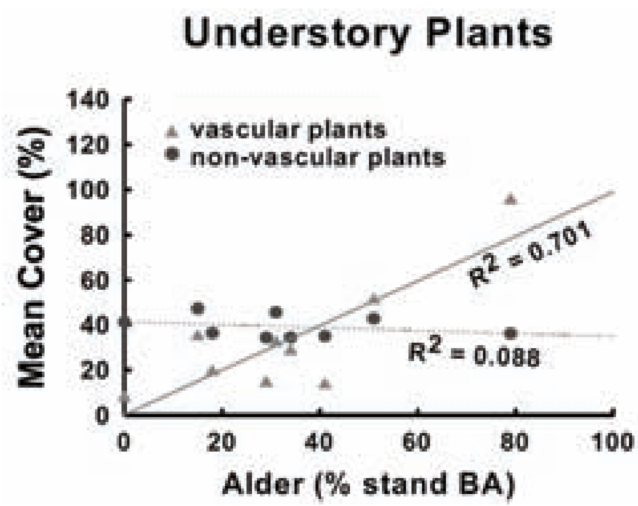


Figure 2— The mean canopy cover of vascular and non-vascular plants as a function of the proportion of red alder basal area (BA) for all nine research sites.

ranged from 10 to 616 kg ha-1 across the 9 stands and was highly correlated with percentage red alder basal area in the stand (Fig. 3a, $r^2 = 0.743$, $p < 0.001$). This correlation of herbaceous biomass with increasing percentage red alder was consistent throughout the range of stands studied. The significant increase in plant biomass during the summer had pronounced positive impacts on the available forage for deer. Carrying capacity for deer in summer increased significantly (Fig. 3b, $r^2 = 0.846$, $p < 0.001$) from 0.05 to 122.18 deer days per hectare with increasing percentage red alder (Hanley et al., 2006). However, there was no significant relation between carrying capacity and percentage red alder in winter (Fig. 3b, $r^2 = 0.246$, $p > 0.10$,). This is probably related to loss of available understory biomass during the winter in these younger forests.

Red alder seedling survival in plantations

First year survival of seedlings was high with an average of 97% survival in the low density treatment, 96% survival in the high density treatment and an overall survival of 96% (Table 2). Third year survival declined markedly with an average of 60% survival in the low density treatment and 74% survival in the high density treatment (Table 2). Overall, third year survival averaged only 68% and was significantly lower than first year survival ($p < 0.001$). Third year survival was higher in the high density treatment and

Table 2. Red alder seedling survival at 23 sites by planting density after one year and three years. Site index and soil characteristics were determined from GIS mapping units.

Site	Alder Seedling Survival				Spruce Site Index	Soil Characteristics			
	First Year Survival (%)		Third Year Survival (%)		50 yr (m)	Geomorphic Position	Geomorphic Deposit	Soil Type	Drainage
	50 tph	200 tph	50 tpa	200 tpa					
Thorne Bay 3	100	100	90	95	22	Smooth hillslope	Till/Colluv./Resid.	McGilvery	Well
Thorne Bay 2	100	100	60	85	24	Smooth hillslope	Till/Colluv./Resid.	Tolstoi-McGilvery	Well
Thorne Bay 1	100	100	60	100	30	Smooth hillslope	Till/Colluv./Resid.	Karta-Tolstoi	Moderate-well
Thorne Bay S.	100	100	20	40	23	Rolling hills	Resid./Colluv.	Wadleigh-Kogish	Poor
Gutchi	90	100	65	60	15	Rolling hills	Organic/Till	Maybeso-Kaikli	Poor
Naukati	95	95	75	85	27	Rolling hills	Colluv./Resid.	Ulloa-Sarkar	Well
Shaheen	100	100	65	90	28	Smooth mtnslopes	Till	Karta	Moderate-well
Yatuk	95	90	65	90	30	Smooth hillslope	Till/Colluv./Resid.	Karta silt loam	Moderate-well
Chin Point 3	100	96	60	72	23	Smooth hillslope	Colluv./Resid.	Kupreanof-McGilvery	Well
Chin Point 2	100	90	33	40	26	Smooth mtnslopes	Colluv./Organic	Kupreanof-McGilvery	Well
Chin Point 1	100	94	40	38	30	Smooth mtnslopes	Colluv./Resid.	Vixen-Traitors	Moderate-well
George Inlet	100	94	60	60	30	Smooth hillslope	Colluv./Resid.	Vixen sandy loam	Moderate-well
Newlywed	NA	100	NA	100	16	Smooth hillslope	Till	Nakwasina	Poor
Nemo	NA	100	NA	96	23	Smooth mtnslopes	Colluv./Resid.	Kupreanof-Mosman	Well
Portage Bay	100	96	57	76	26	Smooth hillslopes	Colluv./Resid.	Kupreanof-Mosman	Well
Big John	92	84	75	60	26	Smooth hillslope	Colluv./Resid.	Kupreanof-Mosman	Well
Duncan Canal	92	96	60	68	30	Smooth hillslope	Till/Colluv.	Karta silt loam	Well
Appleton	92	88	55	60	24	Broken mtnslopes	Till	Yakobi mucky loam	Poor
Baranof	96	100	70	53	25	Broken mtnslopes	Till	Yakobi mucky loam	Poor
Humpback	95	80	75	75	26	Broken mtnslopes	Colluvium	Mitkof loam	Somewhat-poor
Etolin 2	NA	100	NA	76	25	Smooth mtnslopes	Colluv./Till	Mitkof/Mosman	Well
Etolin 3	NA	100	NA	84	18	Smooth mtnslopes	Residuum	St. Nicholas loam	Poor
Etolin 4	NA	100	NA	92	24	Smooth mtnslopes	Colluv./Resid.	Kupreanof-Mosman	Well

survival was significantly higher compared with the low density treatment (p =0.019).

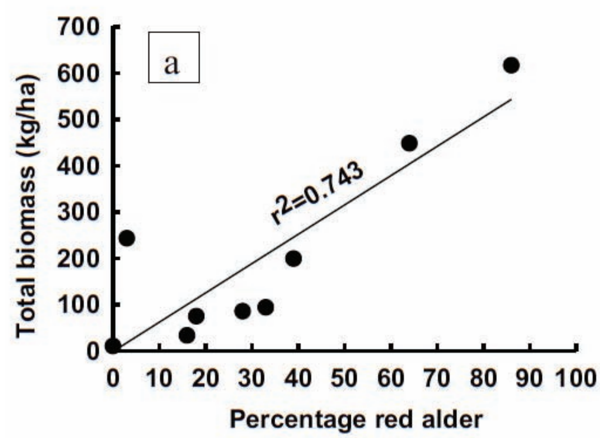
Site index, aspect, elevation and slope were not strongly related with alder seedling survival for any of the treatments and none of these variables were significantly correlated with seedling survival. Soil drainage appeared to be related to third year survival and some of the poorly drained sites had low survival (Thorne Bay S. and Baranof, Table 1). However, some of the well drained sites also had low survival (Chin 1 and Chin 2) as well as very high survival (Thorne Bay 3 and Nemo). Overall, we found no significant relationship between soil characteristics and seedling survival. However, it is important to note that soil types were determined from GIS mapping units and not confirmed in the field, and it is possible that soil type of planting areas were incorrectly identified. Information on elevation, aspect and slope were probably accurate but soil characteristics, soil type and drainage were less certain and need to be confirmed in the field before making any conclusions about soils and seedling survival.

DISCUSSION

Role of alder for increasing stand diversity and understory plant abundance

In younger forests, mixed alder-conifer stands provided different tree size distributions and more complex forest structures than are typically found in pure conifer stands of the same age. Pure conifer young-growth stands in this coastal region are typically very densely stocked (DeMars, 2000) and have relatively uniform tree height and diameter distributions. Our results suggest that red alder can provide different tree size structures with multiple canopy layers in mixtures with conifer stands (Fig. 1). These multiple canopy layers may be the most important structural attribute of mixed alder-conifer stands and may distinguish these stands from predominantly even aged conifer stands of the same age. The presence of alder also did not significantly reduce the size of the largest conifer trees in the stands (Orlikowska et al., 2004; Deal et al., 2004). These conifers will continue to grow in diameter and could provide an important source of large wood for instream fish habitat (Bryant, 1985; Orlikowska et al., 2004).

The mixed alder-conifer younger stands had more complex stand structures than similar aged conifer stands and these structures have important management implications for forest wildlife resources. Researchers have reported broadly negative effects of clearcut logging on wildlife habitat in southeast Alaska (Schoen et al., 1988; Wallmo and Schoen, 1980; Hanley, 1993). The principal problem is that dense conifer regeneration and canopy closure result in an understory with few herbs, few shrubs and little forage for as long as 150 years after initial canopy closure (Alaback, 1982, 1984; Tappeiner and Alaback, 1989). An important finding of this study is the reduced number of small diameter conifers found in the more evenly mixed alder-conifer stands (Deal et al., 2004). The lower stocking of these small diameter trees may allow more light to reach



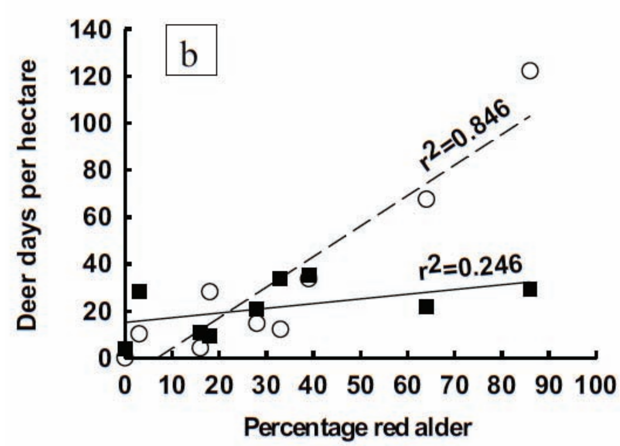


Fig. 3. Total aboveground understory biomass (a) and deer days per hectare (b) of food resources for black-tailed deer during summer (circles) and winter (squares) as a function of the proportion of red alder basal area for the nine research sites (modified from Hanley et al., 2006).

the forest floor that could help maintain understory plants. Recent studies have documented a correlation of herbaceous biomass with increasing percentage red alder throughout the range of stands studied (Hanley et al., 2006). This is important from both a biodiversity and a forest management perspective, because the herb component is difficult to maintain through secondary succession of even-aged stands following clearcutting in southeast Alaska (Alaback, 1982; Hanley, 1993), including precommercial thinning (Doerr and Sandburg, 1986; Deal and Farr, 1994). Summer food value and deer carrying capacity increased with increasing red alder because the quantity of both forbs and shrubs increased with increasing red alder (Fig. 3), however, in winter, most of these red alder-associated species are either senescent (forbs and ferns) or of very low nutritional quality (Hanley and McKendrick, 1983. Therefore, while mixed red alder-conifer stands might improve summer habitat for deer significantly, its potential benefits for winter habitat are somewhat limited.

The effects of mixed red alder-conifer stands for other wildlife species may differ from those for deer. Wildlife-habitat consequences of red alder in young-growth conifer stands differ seasonally and for different wildlife species, depending on the wildlife species' habitat requirements (Hanley et al., 2006). Red alder stands with similar understories to those with the highest percentage red alder stands of this study (Hanley and Hoel, 1996) provided year-around habitat for Keen's mouse that was of equal quality to that of both upland and riparian old-growth forests (Hanley and Barnard, 1999). Other studies conducted in riparian zones and in aquatic ecosystems showed that red alder increased the abundance of food for fish in its immediate vicinity and in downstream reaches (Piccolo and Wipfli, 2002; Wipfli and Musselwhite, 2004). Within red alder stands, vegetation for both invertebrate and vertebrate herbivores was increased by the presence of red alder, an observation consistent with previous work in southeastern Alaska (Hanley, 1996; Hanley and Barnard, 1998; Deal, 1997; Wipfli, 1997; Piccolo and Wipfli, 2002). Streams at the higher end of observed red alder densities delivered about four times more invertebrate biomass than streams canopied predominantly with conifers (Wipfli and Musslewhite, 2004). Also noteworthy was the trophic influence that upland red alder had on more distant habitats. Fish habitats downstream of red alder-dominated reaches were supplied with invertebrates (food for fish) and organic detritus (food for invertebrates) from upstream headwaters (Wipfli and Gregovich, 2002; Piccolo and Wipfli, 2002) and aquatic food webs downstream of red alder received more prey than those below conifer-dominated reaches (Wipfli and Musslewhite, 2004).

Establishment of red alder plantations

In 2001, a large adaptive management study in the region known as the Tongass Wide Young Growth Study (TWYGS) was implemented to assess the effects of different management practices in young-growth stands throughout southeast Alaska including the planting of red alder (McClellan, 2007). Although there are retrospective studies on red alder-conifer stands (Deal, 1997, Hanley and Barnard, 1998; Deal et al., 2004; Hanley et al., 2006), red alder has never been planted in the region, and information on the growth and survival or red alder will be very useful. Assessment of alder plantations showed that early survival was quite high but third year survival declined to 68% and long-term survival of these alder plantations is a concern. However, these results are not particularly surprising considering that plantation sites were selected without consideration of potential establishment problems for red alder. Red alder can be difficult to grow successfully in plantations under better growing conditions in Oregon and Washington and red alder is generally considered less forgiving than conifers when planted on the wrong sites or with improper silvicultural practices (Dobkowski 2006). Many alder plantations initially failed in the Pacific Northwest and it was not until specific site quality information was developed (Harrington 1986) that alder plantations became successful. Although often considered a weed species, alder has specific growing conditions, and poor drainage, frost, drought stress and exposure can negatively affect establishment success (Dobkowski 2006). Harrington (1986) identified 14 soil and site properties important for predicting site index for red alder including geographic position, topographic position, soil moisture, aeration and soil fertility. It is likely that several of the plantation sites in the TWYGS study are poor alder sites and long-term establishment success is questionable. The highly variable third year survival data for these alder plantations is evidence of some potential problems for alder establishment. An important future need is to determine soil and site properties of these plantations and to identify information for better site selection. The preliminary information on alder establishment success will be useful to identify future sites for alder plantations.

In summary, research indicates that inclusion of red alder in conifer-dominated young-growth stands helps maintain understory plants and provides forage for deer and small mammals. Results also show a clear linkage with improved invertebrate diversity in aquatic systems. Well planned silvicultural systems that include a mixture of red alder-compositions could provide trees for timber production and improve other forest resources that are often compromised in pure-conifer young-growth forests in the region.

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