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Management of Riparian Buffers: Upslope Thinning with Downslope Impacts

Kenneth J. Ruzicka Jr., Klaus J. Puettmann, and Deanna H. Olson

We examined the potential of using upslope density management to influence growth and drought tolerance of trees in untreated downslope riparian forests. Increment cores from Douglas-fir trees in three mature stands in western Oregon, USA, were collected and measured. Trees responded to an apparent edge effect up to 15 m downslope of thinned areas but not downslope of gaps. Growth responses in riparian trees were not affected by slope or potential solar radiation (as a function of location and topography). In addition, in a retrospective analysis of tree growth and allocation patterns (represented by the ratio of earlywood to latewood) and climate after treatment over a 12-year period, trees in our study area did not appear to be water limited and did not show a strong correlation with regional drought metrics. We hypothesize that vegetation layers in these riparian forest stands responded differentially to additional resources becoming available as a result of thinning, with overstory trees in riparian areas responding downslope of thinned uplands and subdominant canopy layers responding downslope of gaps. Our study demonstrates that managers can affect riparian forests with upland treatments to a limited spatial extent, which may be the only option in areas where direct riparian management is restricted due to concerns for other ecosystem services.

Keywords: riparian zone, thinning, drought, climate, western Oregon, riparian management

Forested riparian areas serve as a transitional ecotone between upland and aquatic habitats (Gregory et al. 1991) and a discrete habitat type, providing a unique set of ecological functions (Naiman and Decamps 1997). Riparian forests provide habitat for a suite of plant and animal species (Sabo et al. 2005, Richardson and Danehy 2007, Brooks et al. 2012). Riparian areas also provide important ecological subsidies to aquatic habitats, including down wood, sediment, litter, shade, and prey (Naiman et al. 2000, Wipfli et al. 2007). Protecting these important stream-riparian ecological functions and aquatic-riparian sensitive species in managed forest landscapes has been the topic of much research (Olson et al. 2007, Marczak et al. 2010, Richardson et al. 2012).

There is no apparent lack of woody debris in streams from late-successional riparian forests in the Appalachian mountains (Hedman et al. 1996, Keeton et al. 2007) or Iberian peninsula (Diez et al. 2001) or potential snags and down wood in mixed-conifer forests (Romme and Knight 1981). In contrast, researchers in the Pacific Northwest of North America and elsewhere have identified successional trajectories for current second-growth stands that may result in eventual replacement of conifer-dominated riparian areas with hardwood and later shrub- and herbaceous-dominated

communities (Pabst and Spies 1999, Villarin et al. 2009, Goebel et al. 2012). The likely lack of large conifers in future riparian areas in such managed stands has potential negative implications for many riparian functions and processes, including sustained delivery of down wood for fish habitat, stream shading, and nutrient inputs (Gregory 1997, Naiman et al. 2000, Pollock et al. 2012). In the Pacific Northwest, riparian forest compositional and structural complexities, including stem density and basal area, are legacies of historical clearcut harvesting and reforestation activities. Consequently, current dense managed stands are not on a trajectory to develop quickly into “old-growth” riparian areas (Acker et al. 2003), with consequent interruptions of key riparian ecological functions and processes. In response to these concerns, maintenance of riparian conifers and accelerated growth of young riparian conifers are two management goals in the region for riparian forest restoration. In particular, production of large woody debris is related to the successional processes of conifer recruitment and growth via regeneration in riparian areas. Limited conifer regeneration is a concern in many riparian areas in the northwestern United States (Pabst and Spies 1999, Hibbs and Bower 2001).

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Considering the importance of large conifers in riparian areas, the absence of management options in riparian zones with dense forests has raised concerns about future growth and vigor of trees in these areas. Management actions, especially density reductions, could especially benefit trees in high-density patches with delayed size differentiation. Active management in riparian areas could reduce concerns about tree vigor, unstable conditions, and high competition-related mortality and accelerate the availability of large wood in streams (Palik et al. 2012, Zenner et al. 2012). At the same time, reasons for establishment of riparian set-aside zones include the need to protect water and wildlife habitat quality, including maintenance of cool stream temperatures and reducing erosion and the risk of stream sedimentation (e.g., see review in USDA Forest Service and USDI Bureau of Land Management 1993). In all, these myriad issues support both riparian protection and active management, complicating a “one size fits all” strategy for riparian forest management.

The forest management “toolbox” for maintenance or restoration of riparian conditions includes management of adjacent uplands, with edge effects influencing neighboring no-entry forests. For example, clearcuts influence the microclimate of adjacent riparian forested areas (Broszofsky et al. 1997, Davies-Colley et al. 2000, Moore et al. 2005). In addition, variable density thinning has been shown to affect microclimates in adjacent riparian areas (Anderson et al. 2007, Rykken et al. 2007). Microclimate edge effects are influenced by a variety of factors including species, topography, edge influences, and time (Chen et al. 1999).

Because forested riparian areas on small streams are typically on steeper slopes, water movement may be a proximate driver through which alteration of upland conditions can influence tree growth and vigor in downslope riparian areas (Barnard et al. 2010, Govind et al. 2011). Thinning has been shown to increase water availability to remaining trees (Brix and Mitchell 1986, Aussenac and Granier 1988, Bréda et al. 1995). Water availability is also a major concern because climate change models predict increased incidences and severity of drought periods, e.g., in the Pacific Northwest (Mote and Salathé 2010). Thinning has been proposed as a method to reduce the vulnerability of forests to drought (D’Amato et al. 2013). In areas where active riparian management is not allowed, the potential may exist to use thinning in upland areas to indirectly influence responses of trees in riparian areas to future water-availability scenarios. The potential of lateral hillslope flow has been shown to be related to slope steepness and soil depth to bedrock (Hopp and McDonnell 2009) as well as subsurface texture and availability of macropores (Asbjornsen et al. 2011, Bachmair and Weiler 2011). Consequently, the effects of upland forest management on riparian tree growth within protected riparian zones in a context of variable water availability may warrant further consideration.

Furthermore, to address the proximate mechanisms influencing stand development, the availability of resources, especially water, will change physiological processes of trees as they adjust crowns, root systems, leaf morphology, and sapwood area to new conditions (Jozsa and Brix 1989, Aussenac 2000, McDowell et al. 2006). For example, trees with lower water stress have lower sapwood/leaf area ratios (Barnard et al. 2011). Latewood also stores more trunk water and allows easier horizontal water exchange between rings in sapwood (Domec and Gartner 2002). Thus, current water availability has important implications for the potential for trees to adapt physiologically to future droughts (McDowell et al. 2008, Niinemets 2010), and growth allocation, e.g., the ratio between earlywood and

latewood in year ring, can also provide evidence of changing water availability after disturbances such as thinning.

In our study, we test the hypothesis that management in forest uplands can affect riparian tree growth. We hypothesize that this relationship is reflected in the trees’ reaction to water availability, as evidenced by climate-growth relationships and ring-growth allocation to earlywood or latewood. Specifically, we asked:

1. Does manipulating upslope growing conditions affect growth of trees in downslope riparian areas, and, if so, does this effect vary with downhill distance from treatment edge and topographic factors such as slope steepness or light availability?
2. Does reducing upland stem density alter the growing conditions for downslope riparian trees to the point that they are less influenced by drought conditions? That is, does it “decouple” the relationship between annual water availability and tree growth?
3. Does reduced upland stem density affect riparian stem growth characteristics of Douglas-fir, such as the latewood/earlywood ratio, and, if so, what topographic factors influence these relationships?

Methods

Our study uses sites established as part of the Density Management and Riparian Buffer Study of western Oregon, USA (DMS) (Cissel et al. 2006) (Figure 1). The DMS is an operational-scale experiment to investigate the potential of alternative thinning treatments to accelerate late-successional forest structures in young even-aged stands. The Riparian Buffer Study component examines the effects of alternate widths of streamside management zones on aquatic-riparian species and habitat elements. Overall, DMS sites were chosen to be representative of young managed stands on west-side federal forests, from Mount Hood to Coos Bay, Oregon (Cissel et al. 2006). For our study, we chose three DMS study sites (North Soup [NS], OM Hubbard [OM], and Keel Mountain [KM]; elevation 176–783 m) to represent a climatic and latitudinal gradient (Table 1). These sites are dominated by conifer species, primarily Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) with a smaller component of western hemlock (*Tsuga heterophylla* [Raf.] Sarg.) and western redcedar (*Thuja plicata* Donn ex D. Don). Subdominant conifers were more prevalent at the KM site in the western Cascade Range (Figure 1). Soils were primarily humic ultisols and inceptisols with the high infiltration rates typical of western Oregon (Table 1). Western Oregon experiences a Mediterranean climate with cool wet winters and warm dry summers. Average precipitation at our sites ranged from 1,417 to 1,968 mm/year, mostly occurring from November through April. Cissel et al. (2006) provides a complete DMS overview, including individual site histories and descriptions.

The DMS upland treatments were applied over large areas, ranging from 20 to 49 ha at our three sites. DMS treatments and buffer widths were assigned in a randomized block design with some limitations on complete randomization; e.g., stream occurrences often biased delineation of the moderate upslope treatment and the untreated reference area (Olson et al. 2002). These constraints are not expected to lead to biased results for our study because they were based on other site selection criteria and resulted from the variety of sites selected (Dodson et al. 2012). We investigated the treatment with the most drastic tree density reductions in the DMS, which

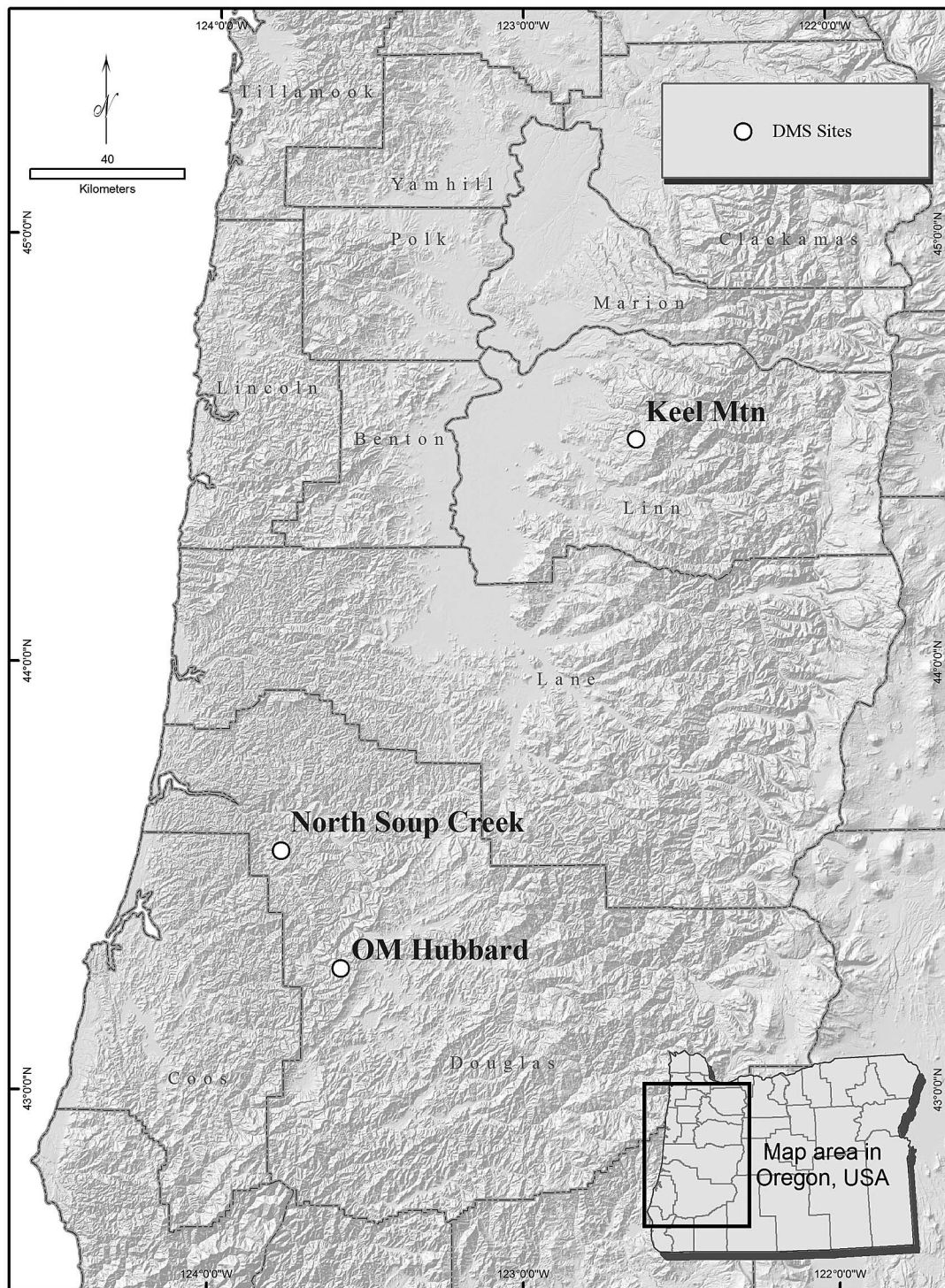


Figure 1. Study sites from the DMS.

were established within the variable density upland treatment (Cissel et al. 2006). We examined two different tree densities within the variable density treatment area: areas thinned to a residual density of 100 trees/ha⁻¹ (“thin” treatment herein); and 0.4-ha circular gaps (“gap” treatment herein) in which all trees were harvested in the gap and the matrix was thinned. There was also an unthinned reference treatment (“control”). The average residual basal area in the second year after treatment was 17.25 m² ha⁻¹ in thin, 53.95 m² ha⁻¹ in control, and 34.73 m² ha⁻¹ in riparian buffers (Anderson 2002). All

thinning was done from below where smaller trees were removed but minority species, mostly hardwoods, were retained for structural and species diversity.

At each site we collected data from overstory trees at plots along 13 preestablished trans-riparian transects (Cissel et al. 2006, Anderson et al. 2007), aligned perpendicular to headwater streams. Riparian buffer widths ranged from 16 to 32 m from the middle of the stream channel. This buffer width was designed to have a 15-m minimum width but could be wider to accommodate local

Table 1. Study site description including physical characteristics and treatment details.

	NS	KM	OM
Latitude	N 43°33'57.0"	44°31'41.0"	43°17'30.0"
Longitude	W 123°46'38.0"	122°37'55.0"	123°35'00.0"
Elevation (m)	176–411	654–756	436–783
Soils (major types)	Absaquil-Blachly-McDuff/Digger-Bohannon-Umpcoos	Kinney/Blachly	Orford/Gustin-Orford
Mean annual precipitation (mm)	1,735	1,968	1,417
Oregon Climate Division	Oregon Coast	Willamette Valley	Southwest Valleys
Site index (m) (King 1966)	40	39	36
Treatment date	August 1998	December 1997	September 1997
Stand age at treatment	48	44	39
Harvesting method	Cable	Ground	Cable + ground

Additional information can be found in Cissel et al. (2006).

conditions of riparian vegetation and topography. In the riparian area, the first and second plot centers were 4.5 and 14 m from the stream center, respectively. In the upland thinned and unthinned treatments, the third plot was 9.1 m upslope of the second riparian plot (22.7 m from stream center), and the fourth and final plot was 18 m upslope of the third plot (41 m from stream center). Four of the “final, 41-m plots” were not in the same headwater drainage as the rest of the respective transect, and no trees were sampled in these plots. One plot did not have any Douglas-fir, resulting in a total of 36 plots. In June to August 2011, we selected and recorded locations of the three codominant and apparently healthy Douglas-firs closest to plot centers. At each tree, we used a laser or acoustic rangefinder depending on understory visibility to measure distance from the treatment-buffer edge, which was defined as closest cut stump. Distance from the ridge top also was measured, and the number of codominant trees within a 12-m circle around each tree was counted. We did not account for slope in these distance measurements to capture the physical length of soil between the edge and ridge top. Trees in riparian buffers were between 2 and 30 m downslope of treatment-buffer edges.

We collected one 7-mm increment core to the pith and three 12-mm cores to capture at least 30 annual rings, covering each cardinal direction side of trees. Each core was sanded until rings and earlywood-latewood boundaries were clearly visible. Tree rings were crossdated visually using pointer years and statistically using the *cross.date* function in the *dplR* program library (Bunn 2010). After crossdating to ensure intra- and intertree accuracy, the four cores from each tree were averaged together by year into a raw ring width chronology for each tree. Raw tree ring widths were mathematically converted to basal area increment (BAI) for earlywood, latewood, and whole ring growth (Phipps 2005) after subtracting the width of the bark to the final diameter (Larsen and Hann 1985). Trees that could not have their ages verified correctly by crossdating were excluded from analysis, resulting in a total of $n = 98$ trees.

To assess relationships between growth and water availability, we used the Palmer Drought Severity Index (PDSI). The PDSI is a general index of drought severity that can be compared across regions because it takes into account precipitation, potential evapotranspiration, and soil moisture (Alley 1984). PDSI data were downloaded from the National Climatic Data Center¹ and divided into early (April–June), late (June–September), and winter (November–March) seasons (Barnard et al. 2012).

We used the hillshade tool in ArcView² to calculate potential solar illumination at ground levels for all trees. This index takes into account the elevation and latitude to calculate the position of the sun as well as mapping and accounting for topographic features that

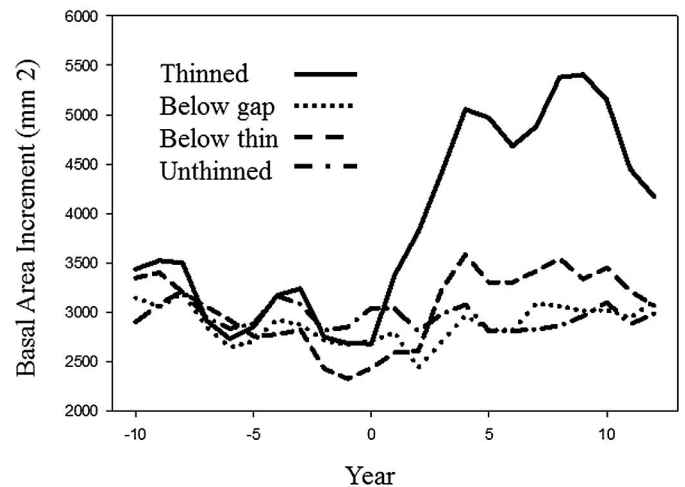


Figure 2. BAI for trees growing in different treated areas in our western Oregon study. Below thin and below gap indicate trees growing in riparian areas downslope of the treatments. Year 0 indicates the year treatments were applied to the upslope.

may shade a given point. We selected June 21, 2010, for calculation of potential solar radiation to capture the sun at its highest position (Suzuki et al. 2008). Our potential solar illumination values ranged from 350,000 to 570,000 W m^{-2} , where smaller values represent north-facing aspects or heavily shaded drainages.

Data Analysis

To examine the effect of upslope thinning on tree growth, we first developed a process to determine the time periods that best represent pre- and posttreatment conditions. We plotted average BAI over time for each treatment to visually assess patterns of change in growth (Figure 2). Trees growing in and downslope of thinned areas grew at increased rates after thinning. In contrast, the BAI of trees in unthinned “control” areas appeared unchanged. The data support the notion of several years of physiological adaptation to new growing conditions (Harrington and Reukema 1983, Aussenac 2000). Based on these results, we determined that stabilized growth reflecting posttreatment conditions was reached 4 years posttreatment and classified “after” treatment as the time period from year 4 after treatment to year 12 after treatment. Accordingly, we classified “before” treatment as the time period from 10 years before treatment to year 0 (treatment year). We then calculated differences between average tree growth rates (after – before) to examine patterns with treatment type. In all models described below, an α level of 0.05 was selected as an indication of significant differences.

To examine the effects of upslope tree density reduction on downslope tree growth (below thin, below gap, and unthinned) in riparian areas, we used a mixed-effects model with a nested random effect structure. Only downslope trees were included in the model ($n = 81$). The model examined treatment effects and influences of topographical features (distance downslope and percent slope) and local conditions (potential solar radiation and neighborhood tree density) on differences in tree BAI (Equation 1)

$$Y_{ijkl} = \alpha_i + \beta_{1-4} + \beta_5 + \gamma_j + \lambda_{jk} + \delta_{jkl} + \varepsilon_{ijklm} \quad (1)$$

where Y_{ijklm} is the mean value of the difference in BAI (after – before), α_i is the fixed effect of the upslope treatment i = below thin, below gap, unthinned, β_{1-4} is the fixed effect of 1–4 = distance downslope from the treatment-buffer edge, solar radiation, percent slope, and neighborhood density, β_5 is the slope of the interaction between distance downslope and percent slope, γ_j is the random effect of the j site (NS, KM, or OM), λ_{jk} is the random effect of the k transect in the i site (1–13), δ_{jkl} is the random effect of the l plot in the k transect at the i site (1–29), ε_{ijklm} is the random effect of the individual tree in the i treatment in the j site in the k transect and l plot (1–81), and $\beta \sim N(0, \sigma\beta)$ and $\text{Cov}(\beta, \beta') = 0$, $\gamma_{ij} \sim N(0, \sigma\gamma_2)$ and $\text{Cov}(\gamma_{ij}, \gamma_{i'j'}) = 0$, ε_{ijklm} is independent.

The model was fit using the lme function from the nlme package (Pinheiro et al. 2011) and computed in R 2.15.2 (R Core Team 2012). We used the function varIdent to allow for variance in sample sizes between treatment groups (Zuur et al. 2009). The assumptions of homogeneous variance and normality were confirmed graphically.

Effects of upslope treatments (below thin, below gap, and unthinned) on relationships between downslope tree growth and PDSI (drought severity index) were tested by comparing parameters in multiple linear regression models that were fitted for BAI growth before and after treatment. This model was used to compare the growth of the entire ring as well as the earlywood and latewood segments individually with the different seasonal drought measures of growing season and early, late, and winter. Statistically different slope parameters after treatment indicate that upslope treatments had resulted in different PDSI-growth relationships compared with pretreatment conditions. To test this, the model was fitted to the data from each treatment (Equation 2)

$$Y_{ijklm} = \beta_0 + \beta_1 I_{Bi} + \beta_2 \beta_i + \beta_3 X_i I_{Bi} + \gamma_j + \lambda_{jk} + \delta_{jkl} + \varepsilon_{ijklm} \quad (2)$$

where Y_{ijklm} is the value of the growth in BAI for tree (1–98), I_{Bi} is an indicator for before and after treatment $I_B = 1$ for before, X_i is the value of the continuous covariate PDSI (entire growing season, spring, summer, and winter tested individually), γ_j is the random effect of the j site (NS, KM, or OM), λ_{jk} is the random effect of the k transect in the i site (1–13), δ_{jkl} is the random effect of the l plot in the k transect at the i site (1–29), and ε_{ijklm} is the random effect of the individual tree in the i treatment in the j site in the k transect and l plot (1–98).

The model was fit using the lme function from the nlme package (Pinheiro et al. 2011) and computed in R 2.15.2 (R Core Team 2012). Growth (BAI) was log-transformed in models for each treatment to meet assumptions of normality and homogeneous variance.

To analyze potential relationships between upslope tree density and physiological changes in growth allocation, we tested whether latewood/earlywood ratios differed between pre- and posttreatment

growth. We calculated ratios of BAI of latewood/earlywood for each year and calculated average ratios for pre- and posttreatment growth (year –10 to 0 and year 4 to 12). We then calculated differences between post- and pretreatment averages to test for differences in temporal growth patterns.

A mixed-effects model was used to test for treatment effects and influences of different topographical features on growth patterns (Equation 3)

$$Y_{ijklm} = \alpha_i + \beta_{1-4} + \beta_5 + \gamma_j + \lambda_{jk} + \delta_{jkl} + \varepsilon_{ijklm} \quad (3)$$

where Y_{ijklm} is the mean value of the difference in latewood/earlywood ratio, α_i is the fixed effect of the upslope treatment i = below thin, below gap, and unthinned, β_{1-4} is the fixed effect of 1–4 = distance downslope, solar radiation, percent slope, and neighborhood density, β_5 is the slope of the interaction between distance downslope and percent slope, γ_j is the random effect of the j site (NS, KM, or OM), λ_{jk} is the random effect of the k transect in the i site (1–13), δ_{jkl} is the random effect of the l plot in the k transect at the i site (1–29), ε_{ijklm} is the random effect of the individual tree in the i treatment in the j site in the k transect and l plot (1–81), and $\beta \sim N(0, \sigma\beta)$ and $\text{Cov}(\beta, \beta') = 0$, $\gamma_{ij} \sim N(0, \sigma\gamma_2)$ and $\text{Cov}(\gamma_{ij}, \gamma_{i'j'}) = 0$, ε_{ijklm} is independent.

The model was fit using the lme function from the nlme package (Pinheiro et al. 2011) and computed in R 2.15.2 (R Core Team 2012). We used the function varIdent to allow for different variances in sample size between treatment groups (Zuur et al. 2009).

The explanatory variables were selected to test our stated hypotheses except neighborhood density (number of trees within 12 m), which was added as a covariate to account for different competitive neighborhoods. The assumptions of homogeneous variance and normality were confirmed graphically for each model fitting.

Results

Our results suggest that upslope thinning increased downslope riparian tree growth whereas upslope gaps did not (Figure 3). Post-thinning, trees growing downslope of thinned areas had an increased basal area growth, which averaged more than double (2,710 mm²) the growth difference of trees below the unthinned areas. In contrast, trees growing downslope of clearcut gaps within a thinned matrix showed no difference between pre- and posttreatment growth. In addition, trees growing within riparian areas downslope of unthinned uplands did not appear to experience a change in growing conditions (Table 2). The growth benefits due to upland thinning decreased with distance downhill from the treatment edge and was limited to trees growing within about 15 m of treatment-buffer edges (Figure 4). Steepness of the topography also affected BAI, where trees on steeper slopes showed a reduction in growth below the unthinned stands compared with trees below the two thinning treatments (Figure 5). However, the significance of slope steepness downslope of the unthinned area was the result of a single outlier in the unthinned riparian area that experienced a significant growth reduction after thinning. When this tree was removed from the analysis, slope steepness did not affect growth of trees in riparian areas. Potential solar radiation, neighborhood density, and the interaction between slope and distance to the treatment-buffer edge did not affect growth of trees in our models.

Our data did not indicate a tree-growth response to water availability during the study period. The slopes of regression lines relating growing season drought severity (PDSI) and BAI were not significantly different from 0 (Figure 6). The regressions did reinforce

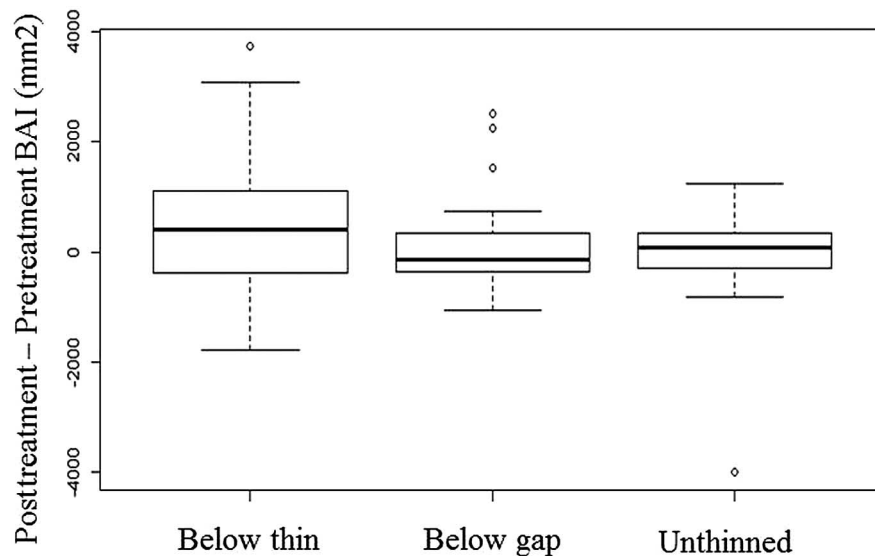


Figure 3. Boxplots of the differences in tree growth between pre- and posttreatment, of trees growing in riparian areas downslope of the treatments (thin and gap) and in the riparian area of unthinned controls in our western Oregon study. The center line represents the median, and the lower and upper edges represent the 25 and 75% quartiles. Whiskers represent the extent of 90 and 10 percentiles with open circles showing trees outside of these values.

Table 2. Effect sizes and 95% CIs for treatments and topographic covariates on the difference in growth for trees growing in riparian areas.

Effect	Mean difference	SE	df	t value	P value*	Lower 95% CI	Upper 95% CI
Below thin	2,710.28	1,307.13	48	2.07	0.04*	82.21	5,338.45
Below gap	-327.71	364.28	8	-0.90	0.39	-1,167.74	512.32
Unthinned	-1,337.96	479.33	8	-2.72	0.02*	-2,442.69	-232.03
Distance	-88.31	32.45	48	-0.56	0.01*	-153.55	-23.06
Radiation	-0.00	0.00	48	-2.27	0.58	-6,390.13	3,607.70
Slope	-17.38	7.66	15	-0.89	0.04*	-33.70	-1.05
Density	-14.57	16.37	48	1.42	0.37	-47.48	18.34

CI, confidence interval.

*Significant results.

the mixed-model analysis that trees growing in thinned and below thinned treatments had increased growth after treatment. Our study showed similar results for seasonal drought patterns, including spring, summer, and winter droughts (data not shown). In general, trees at our study sites did not change growth patterns in response to PDSI (Table 3).

Trees in riparian areas did not change their timing of growth patterns after upslope density reduction. Latewood/earlywood ratios did not appear to differ between pre- and posttreatment conditions (data not shown).

Discussion

Our findings indicate that forest managers can affect tree growth inside untreated riparian buffers by upland forest management, at least within a short distance of the treatment-buffer edge. Previous work in clearcuts indicated that edge effects on tree growth are dependent on interacting factors including distance from the edge, landform, species, and climatic conditions (Cadenasso et al. 1997, Harper et al. 2005, 2007). Our results confirm that earlier findings of the spatial extent of upland tree-to-tree interactions (e.g., 10–20 m for Douglas-fir) (Wimberly and Bare 1996, D'Amato and Puettmann 2004, Puettmann et al. 2009) are also relevant when effects of upland treatments on tree growth in riparian areas are assessed. This suggests that plant interactions, such as competition for water and

growing space, may drive these patterns, as microclimate edge effects may extend further, up to 62 m from the treatment-buffer edge (Chen et al. 1995, Brososke et al. 1997, Anderson et al. 2007). Edge effects on tree diameter growth and understory plant communities have been shown in other studies of variable density thinning including gaps (Fahey and Puettmann 2007, Roberts and Harrington 2008, Dodson et al. 2012).

In contrast to our results, other riparian studies testing the effects of clearcuts on riparian tree growth did not show an edge effect and attributed it to a quick response of the shrub layer and regeneration in the gap to block edge light availability and wind (Hibbs and Bower 2001). In western North Carolina, a shelterwood harvest increased nutrient availability in the harvested areas but not in adjacent unharvested riparian forests. However, the authors found an increase in organic nitrogen in deeper soil layers in the harvested unit, suggesting some movement of nutrients (Knoepp and Clinton 2009). We do not have any direct evidence from our study for specific mechanisms tied to the increase in riparian tree growth. Increased growth may be due to a combination of higher light, water, and nutrient availability (Powers et al. 2009, Lasky et al. 2013), whereby nutrient availability is intrinsically linked to moisture conditions. However, it does not appear that increased availability of moisture to trees is driving the growth response in our study as discussed below. When management and protection of

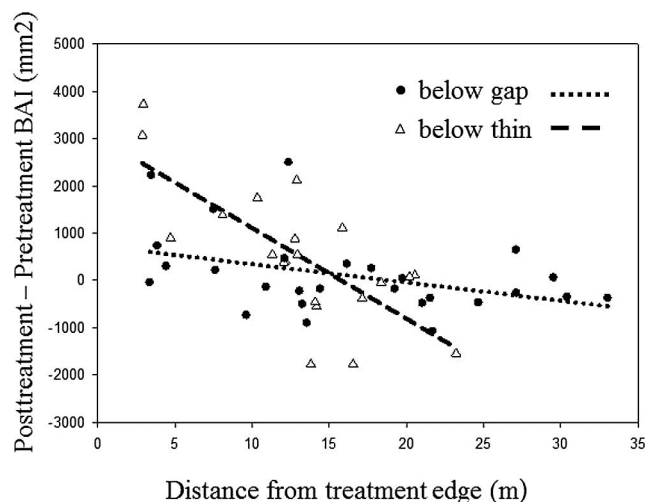


Figure 4. Tree growth difference between pre- and posttreatment and distance downslope of the treatment edge for trees growing in riparian areas downslope of the treatments in our western Oregon study. Distance 0 is the nearest cut stump in the treatment area.

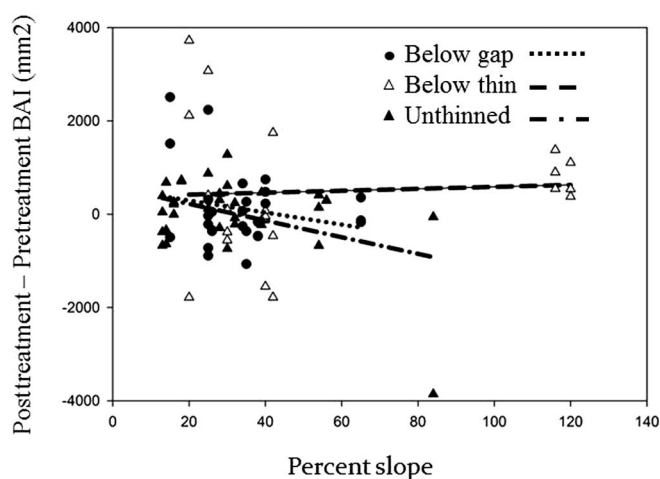


Figure 5. Growth difference and percent slope for trees growing in riparian areas downslope of the treatments in our western Oregon study.

riparian zones are contentious, our results suggest that growth and vigor of trees in riparian buffers can be improved by upslope thinning, albeit to a limited spatial extent (Lee et al. 2004, Rambo and North 2009).

The extent of edge effects from upslope management will be influenced by local vegetation, climatic, and topographic factors (Chen et al. 1999). For example, the understory species composition of riparian areas in the Pacific Northwest varies with local factors such as the distance from a stream, aspect, slope position, and percent slope (Sarr and Hibbs 2007a, 2007b, Sarr et al. 2011). Douglas-fir trees are often more prevalent on steeper slopes in the Pacific Northwest than on gentler sloped floodplains (Hibbs and Bower 2001, Barker et al. 2002). Although topographic factors, such as slope steepness, did not appear to affect tree growth responses to upslope thinning in our study, other studies suggested that management in uplands designed to affect trees in riparian areas needs to account for local topography. Trees adjust their canopy structure and live crown ratios in response to light from different angles, due

to different azimuth, slopes, and edge effects (Muth and Bazzaz 2002, Šálek et al. 2013). Thus, exploration of how canopy structure and tree growth can be manipulated without direct management within riparian areas may provide information on how to create desirable late-successional stand conditions in areas with forest management constraints due to regulatory restrictions (Franklin et al. 2002, Bauhus et al. 2009).

The gaps embedded in variable density thinning treatments in our study have effects on surrounding vegetation different from those of thinning alone, depending on gap size and species (Davis et al. 2007, Fahey and Puettmann 2007, 2008). Gaps can also lead to different microclimate conditions, depending on the prevailing aspect (Gray et al. 2002). However, none of these factors is an obvious candidate to explain why trees downslope of thinned areas responded to a change in upslope densities, whereas trees downslope of gaps did not, even though presumably more resources would be available below gaps than below thinned areas. As reflected in the amount and composition of understory vegetation, gap effects were initially limited to the area inside gaps and did not infiltrate the adjacent forest interior (Fahey and Puettmann 2007, 2008). However, on steep hillsides in riparian areas, the gaps may have benefited smaller adjacent trees in the midstory (Hibbs and Bower 2001, Gray et al. 2012) and also the shrub layers (Sarr and Hibbs 2007b, Montgomery et al. 2010), both of which were not formally sampled in our study. However, personal observations taken during data collection support the fact that understory vegetation in gaps responded with increased growth compared with that in thinned areas (K.J. Ruzicka Jr., Oregon State University, pers. observ., July 14, 2011). Other studies using the DMS sites showed that the amount of understory vegetation was inversely proportional to that of overstory density (Neill and Puettmann 2013). In addition, vegetation in gaps at DMS sites responded quickly in terms of plant cover compared with that in the thinned stand interior (Fahey and Puettmann 2008). Other studies in shelterwoods in Minnesota have found that increased light transmittance through the canopy was correlated with increased shrub cover (Smidt and Puettmann 1998). Studies in western Oregon have shown that management actions initially reduced shrub cover, probably through mechanical breaking, but that smaller shrubs recovered quickly whereas larger shrubs responded more slowly (Berger et al. 2012). Overstory trees are tightly coupled with understory vegetation through a variety of mechanisms such as light transmission, precipitation throughfall patterns, and soil moisture status (Barbier et al. 2008). Overstory structure (Nagaike et al. 1999) and species composition (Berger and Puettmann 2000) is reflected in composition of understory vegetation. The effect of the overstory on the shrub layer varies by time and successional status (McKenzie et al. 2000). Conversely, neighboring shrub cover (nonnitrogen-fixing species) has been linked to reduced nitrogen in overstory tree foliage as well as reduced soil moisture status and temperature reducing photosynthesis (see review by Li et al. 2012). It is possible that subdominant species were situated to take advantage of increased aboveground resources, such as light (Shatford et al. 2009, Comfort et al. 2010), co-opting other belowground resources released by gap creation (Montgomery et al. 2010).

Our study did not show that potential drought effects (within the sampled conditions) were mitigated by reduced stem density. We hypothesized that lateral soil water flow would have potentially led to an increase in available water to downslope riparian trees because riparian trees have been shown to reduce stream flows through evapotranspiration on a diel scale (Barnard et al. 2010). In areas with

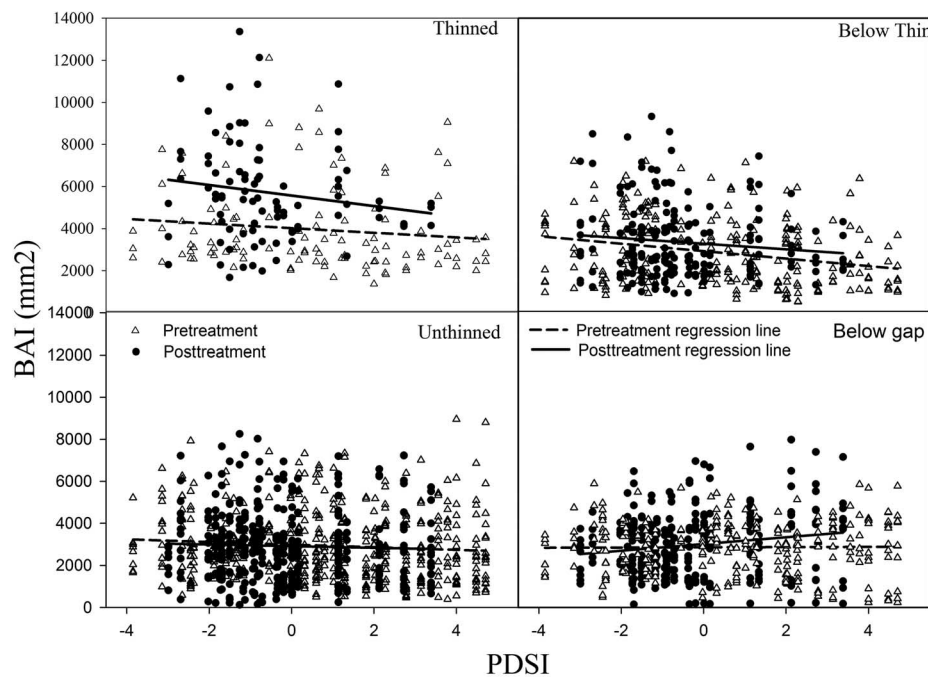


Figure 6. Comparison of regression lines of growth and growing season drought between post- and pretreatment in our western Oregon study.

Table 3. Significant trends for comparison of multiple regression lines using yearly growth and growing season PDSI.

Parameter	Upland thinned	Below thinned	Below gap	Control
Growth ~PDSI	NS (0.1857)	NS (0.7314)	NS (0.5415)	NS (0.2430)
After – before	+ (<0.0001)	+ (0.0001)	NS (0.4543)	NS (0.1744)
Interaction	+ (0.0086)	+ (0.0103)	NS (0.0851)	NS (0.0598)

P values are in parentheses. NS, not significant.

a less expressed summer dry season, e.g., the southeastern United States, harvesting in headwater riparian communities resulted in higher water tables and increased composition of hydrophilic understory vegetation communities (Clinton 2011, Choi et al. 2012). For species that are able to directly access stream water, low stream flows were correlated with reduced tree growth (Coble and Kolb 2012) and fertilized forests area able to reduce streamflow. Riparian vegetation also influences streamflow and nutrient loads in headwater streams. In agricultural landscapes, steeper slopes in riparian buffers were correlated with higher levels of in-stream nitrogen due to increased subsurface flow (Burt et al. 2002, Vidon and Hill 2004a, 2004b).

It is likely that thinning would not alter the relationship between water availability (PDSI) and tree growth if trees are not water stressed sufficiently except in extreme drought conditions (≤ -4 PDSI) (Alley 1984). Thus, they are not tightly coupled to regional patterns in water availability (Niinemets 2010, Barnard et al. 2012). Extreme weather events are important drivers of vegetation mortality (Parmesan et al. 2000, Jentsch et al. 2007, Thompson et al. 2013) as well as long-term drought stress on trees (McDowell 2011). It is possible that the drought events at our study sites during the period investigated were not severe enough to be reflected in the growth of dominant tree species (Lloret et al. 2012). Although the trees at our sites experienced several years with moderate drought with PDSI of approximately -3.9 (1992 and 2001), they were interspersed with very wet years of up to PDSI of 4.7 (1996 and

2006). Prolonged multiyear drought has not been a potential stressor at our sites.

Deciduous *Quercus* and *Fagus* species in Germany as well as mixed Mediterranean tree species in Spain showed decreasing growth sensitivity to drought from xeric to mesic habitats (Pasho et al. 2011, Scharnweber et al. 2011). Growth models of lodgepole pine in southwest Canada have predicted that provenances adapted to temperature and precipitation extremes, high or low, would respond strongly to climate change, whereas trees in moderate environments were more adaptable and did not strongly respond to climatic pressures (McLane et al. 2011). In the Pacific Northwest, growth of conifers, especially Douglas-fir is more strongly correlated with temperature and precipitation at higher altitudes and at the edge of its climatic range than in lower elevation forests (Ettinger et al. 2011). Trees at our sites were growing in moderate climates at low elevation for Douglas-fir, and, thus, we expected that they were adapted to the moderate drought conditions exhibited during our study and could alter other physiological mechanisms to maintain relatively constant growth. It is also possible that trees at our sites were limited by factors other than water (Aussenac 2000, Brooks and Mitchell 2011) and instead responded to the availability of nitrogen (Brooks and Coulombe 2009, Kastendick et al. 2012). Nitrogen is the primary limiting nutrient for Douglas-fir in the Pacific Northwest (Vance et al. 2010).

This argument was also supported by the fact that at our sites trees downslope of treatment did not show changes in temporal

diameter growth patterns, regardless of slope steepness and light intensity. Shifting growth allocation to latewood can be a physiological adaptation by trees to drought through changes in the hydraulic conductance of water transport xylem and lateral water flow processes (Barnard et al. 2011, Wang et al. 2012). If trees had been extremely drought stressed, we would have expected to detect changes in timing of growth allocation (Peñuelas et al. 2011). Further study explicitly examining the water status of these trees is needed to support or conclusively reject our hypothesis.

Conclusion

Riparian areas are an important component of the forested landscape and provide many ecological subsidies to uplands, aquatic environments, and downstream users. Management of riparian forests can be contentious although it may be desirable for specific goals, such as provision of large trees. We found an edge effect below upland thinning treatments that extended up to 15 m into untreated riparian buffers. There was no similar edge effect for trees downslope of gaps. We speculate that this difference may be due to the understory, shrub, and subdominant canopy layers responding more strongly to gap creation than thinning. Our study demonstrates that upland management can be used to influence riparian forests at the upland edge but only to a limited spatial extent. Such management practices may be enough to support the functional goals of riparian buffers such as maintaining potential in-stream coarse woody debris, stream temperature moderation, and nutrient uptake. Maintaining lower tree densities directly above riparian areas may be especially beneficial if other methods to increase tree growth and vigor such as thinning are not allowed directly in riparian management areas. We did not find evidence in our study that upslope thinning decreased the impact of drought in riparian trees, but that may be due to the limited extent of drought over the time frame in our study. However, managing for increased tree vigor seems to be a reasonable goal, considering an uncertain future with global change.

Endnotes

1. For more details, see www.ncdc.noaa.gov/.
2. Please visit Environmental Systems Research Institute website at www.esri.com/software/arcgis/arcgis-for-desktop for more information.

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