Discerning responses of down wood and understory vegetation abundance to riparian buffer width and thinning treatments: an equivalence–inequivalence approach

Paul D. Anderson and Mark A. Meleason

Abstract: The combined effectiveness of thinning and riparian buffers for increasing structural complexity while maintaining riparian function in second-growth forests is not well documented. We surveyed down wood and vegetation cover along transects hom stream center, through buffers ranging from <5 to 150 m width into thinned stands, patch openings, or unthinned stands of 40- to 65-year-old Douglas-fir (*Pseudotsugo menziesii* (Mirb.) Franco) forests in western Oregon, USA. Small-wood cover became more homogeneous among stream reaches within 5 years following thinning, primarily due to decreases for reaches having the greatest pretreatment abundance. Mean shrub cover converged, predominantly because of decreases in patch openings. Herbaceous cover increased, particularly in patch openings. Relative to unthinned stands, herbaceous cover was similar in wide buffers and increased in the narrowest buffers and in narrow buffers adjacent to patch openings. Moss cover tended to increase in thinned areas and decrease in patch openings. Both conventional point-null hypothesis tests and inequivalence tests suggested that wood and vegetation responses within buffers of >15 m width were insensitive to the treatments. However, inherently conservative equivalence tests infrequently inferred similarity between thinned stands or buffers and untreated stands. Difficulties defining ecologically important effect size can limit the inferential utility of equivalence-inequivalance testing.

Résumé : L'efficacité combinée de l'éclaircie et de lisières boisées riveraines en vue d'augmenter la complexité structurelle dans des forêts de seconde venue tout en maintenant leurs fonctions riveraines n'est pas bien documentée. Nous avons échantillonné le bois mort au sol et le couvert végétal le long de transects établis à partir du milieu de ruisseaux vers des lisières boisées riveraines dont la largeur variait de <5 à 150 m et jusque dans des peuplements éclaircis, des trouées ou des peuplements non éclaircis de Douglas de Menzies (*Pseudotsuga menziesii* (Mirb.) Franco) de 40–65 ans dans l'ouest de l'Oregon, aux États-Unis. Le couvert de jeunes arbres est devenu plus homogène entre les biefs durant les cinq années suivant l'éclaircie, essentiellement à la suite de la diminution du nombre de tiges dans les biefs qui étaient denses avant d'être traités. Le couvert moyen d'arbustes est devenu plus uniforme, principalement à cause de la diminution du nombre de trouées. Le couvert herbacé a augmenté, en particulier dans les trouées. Le couvert herbacé des lisières boisées les plus larges était semblable à celui des peuplements non éclaircis, mais il devenait plus important dans les lisières boisées les plus étroites et dans les lisières boisées étroites adjacentes à des trouées. Le couvert muscinal tendait à augmenter dans les sites éclaircis et à décroître dans les trouées. Des tests classiques d'hypothèses nulles réalisées selon les méthodes statistiques fréquentistes et des tests de non-équivalence selon l'approche bayésienne ont tous suggéré que les réponses du bois mort et du couvert végétal dans les lisières boisées de 15 m de largeur et plus n'étaient pas influencées par les traitements. En revanche, des tests d'équivalence plutôt conservateurs ont suggéré à quelques reprises qu'il y avait des similarités entre les peuplements éclaircis ou les lisières boisées et les peuplements non traités. Des difficultés liées à la définition de l'ampleur que devrait avoir un effet pour être écologiquement important peuvent limiter l'utilité des tests d'équivalence ou de non-équivalence pour tirer des conclusions.

[Traduit par la Rédaction]

Introduction

Forest management can result in landscapes of relatively young and structurally uniform forests typically having low abundance of down wood and little understory vegetation (Wilson and Puettmann 2007). Silvicultural practices such as variable density thinning are potential means to increase stand-level structural and compositional heterogeneity in these types of forests (Bailey and Tappeiner 1998; Bauhus et al. 2009). Down wood and understory vegetation are important characteristics associated with old-growth forest structure (Bauhus et al. 2009) that also serve important riparian functions in headwater forests (Gomi et al. 2001; Piccolo and Wipfli 2002). Maintenance or restoration of riparian function is often a concomitant objective in the restoration of young forests, particularly in headwater landscapes. As a result, forest practices commonly include streamside

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vegetative buffers to mitigate potential harvest impacts on riparian habitats and processes (Lee et al. 2004). Management of second-growth forests to meet joint riparian and upland objectives requires a better understanding of the effectiveness of thinning and buffer practices on the development of structural attributes such as down wood and understory vegetation.

Thinning to increase structural complexity is advocated as a potential restorative practice in relatively uniform, second-growth forests (Bauhus et al. 2009). An obvious effect of thinning is the near-term increase in light available to understory and residual overstory vegetation. However, thinning can also influence many other forest microclimate parameters by altering energy balance, radiation regime, airflow, and soil water regimes (Chen et al. 1999; Anderson et al. 2007). Depending on the intensity and spatial configuration, thinning is one approach to moving a stand from the stem-exclusion phase (Oliver and Larson 1996), either temporarily or permanently, to a more structurally heterogeneous, dynamic stage of development.

Thinning provides an input of down wood through harvesting practices that leave branches and unmerchantable stems as slash and disturbance of understory vegetation during removal of harvested trees. Wood recruitment can also result from windthrow along newly created harvest unit edges (Rowan et al. 2003). The width of riparian management zones and the standing stock of wood, which is a function of forest age, within those zones influences the availability of wood to streams (Meleason et al. 2003). Following clear-cutting, windthrow may be weakly correlated to buffer width (Ruel et al. 2001), as local topography and wind direction may be more influential. However, the propensity for windthrow of "soft" edges created by thinning adjacent to riparian buffers would likely be less than that of "hard" edges associated with clearcut harvesting. Upland thinning can also influence the source and character of large wood delivered to streams through periodic mass-wasting events (Czarnomski et al. 2008).

Although recent studies have documented understory vegetation responses to thinning (see summaries in Wilson and Puettmann 2007; Bauhus et al. 2009), the responses of riparian vegetation communities to forest management are not well documented (Pabst and Spies 1999). Riparian areas in forested landscapes support unique vegetative communities that tend to be more herbaceous than upland habitats but also support many shrub species (Minore and Weatherly 1994). Topographic position with respect to environmental gradients from stream channel to ridge top may be a local driver of riparian vegetation composition (Sarr and Hibbs 2007; Pabst and Spies 1999) with many species expressing affinities to either streamside or hillslope positions. Thus, restorative thinning treatments may have differential effects depending on topographic position. Width of buffers will also likely influence the riparian vegetation responses to adjacent harvest. Palik et al. (2003) attributed increased aspen regeneration, growth, and biomass production in riparian buffers adjacent to clearcuts to edge effects of increased light and air and soil temperatures. Hibbs and Bower (2001) found little evidence of vegetation community differences between narrow streamside buffer strips adjacent to clearcut harvest and similar near-stream zones in unharvested reaches in the Oregon Coast Range. Development of effective riparian management strategies will require additional understanding of the responses of buffer vegetation to buffer width and intensity of upslope harvest.

A purpose of riparian buffers is to mitigate potential impacts of thinning on riparian and stream systems. Given a mitigation objective, the question of interest is not whether conditions in riparian areas of thinned stands with buffers differ from those characteristic of unthinned stands but, rather, whether conditions in buffered riparian areas are similar to those of riparian areas in unthinned stands. In conventional hypothesis testing, failure to reject the point-null hypothesis of no difference among means does not necessarily mean that the means are the same (e.g., "Absence of evidence is not evidence of absence;" Sagan 1995, cited in Parkhurst 2001), and thus, the point-null test could never be used to determine if two riparian treatments are the same. Given a sufficiently large sample, the point-null hypothesis can be rejected, but the difference in means could be so small that it may not be ecologically relevant. In contrast to conventional hypothesis testing focused on discerning statistical differences among treatments, equivalence-inequivalence tests based on an a priori definition of minimum significant difference can determine whether treatments are similar (equivalent) or dissimilar (inequivalent) or if insufficient information exists to conclude either way.

One objective in this study was to determine the influence of riparian buffer width on the understory vegetation and down wood responses to density management both within the unthinned buffer and in the adjacent, thinned stand. We hypothesized down wood abundance to be greater in harvested stands relative to unthinned stands because of harvest damage and unmerchantable residues (slash) left on the site. In adjacent riparian buffers, we expected the abundance of down wood to be greater in narrow buffers than in wide buffers because of greater edge to area ratio for narrow buffers and because narrow buffers occupied lower slope and flood plain positions likely to accumulate down wood migrating from upslope positions. With decreased overstory cover and increased sidelight along buffer or patch opening edges, we expected understory vegetation abundance to increase with thinning - the increase being greater in narrow buffers than in wide buffers adjacent to thinned stands and greater in buffers adjacent to patch openings than in buffers adjacent to thinned stands. In upslope treated areas, we expected understory vegetation abundance to increase more in patch openings than in thinned stands.

Given the practical constraints limiting sample size and the high variability among experimental units, a second objective of this study was to explore the use of equivalence-inequivalence tests as an alternative statistical method for testing the degree of similarity between treated stands. To our knowledge, this is the first use of equivalence-inequivalence tests to compare buffer treatments. This approach provides a means to test whether the difference between two buffer treatments is negligible in at least a practical sense, and ultimately in terms of ecological importance (e.g., Manly 2001; Dixon and Pechmann 2005; Froese and Robinson 2007).

Table 1. Locations and characteristics of study sites.

Site attribute	Bottomline	Green Peak	Keel Mountain	O.M. Hubbard	North Soup Creek
Physiographic province	Coast Range	Coast Range	Cascade Range	Coast Range	Coast Range
Latitude (°N)	43.77	44.37	44.53	43.29	43.57
Longitude (°W)	123.23	123.46	122.63	123.58	123.78
Elevation (m)	235–365	485–760	605–760	425–765	180–365
Range of aspect					
Unthinned unit	E–SE	NE–SE	SW–NW	N–NE	NW–N
Thinned unit	NW–N	NE–SE	SW–W	N–E	W–N
Site potential tree height (m)	72	75	82	61	68
Stand origin date	1939	1933–1935	1949–1953	1953–1955	~1948
Regeneration method	Natural	Natural	Natural	Planted	Natural
Year of treatment	1997	1999	1998	1997	1998
Year of sampling					
Pretreatment (P_0)	1996	1998	1997	1996	1998
First posttreatment (P_1)	1999	2000	2000	1998	2001
Second posttreatment (P_2)	2002	2001	2003	2002	2003

Note: Physiographic provinces are from Franklin and Dyrness (1988), and site information is from Cissel et al. (2006).

Methods

Sites

The investigation, which is a component of the US Bureau of Land Management (BLM) density management and riparian buffer study (Cissel et al. 2006), was conducted in second-growth stands varying in age from 43 to 65 years and dominated by Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) at five locations in the Coast Range and Cascade Range of westem Oregon (Table 1). The general climate is maritime with wet, mild winters and cool, relatively dry summers (Franklin and Dyrness 1988). Across locations, mean annual temperature ranges from 11.0 °C to 13.9 °C; mean annual precipitation ranges from 138 to 222 cm. Prior to the study, these dense, even-aged stands were structurally uniform as is typical of stands in a stem-exclusion phase (Oliver and Larson 1996) of development. Although dominated by Douglas-fir, there were minor components of westem hemlock (*Tsuga heterophylla* (Raf.) Sarg.), bigleaf maple (*Acer macrophyllum* Pursh), and red alder (*Alnus rubra* Bong.). Vine maple (*Acer circinatum* Pursh), Oregon grape (*Mahonia nervosa* Pursh), salal (*Gaultheria shallon* Pursh), and red huckleberry (*Vaccinium parvifolium* Smith) were locally the most abundant shrub species; swordfern (*Potystichum munitum* (Kaulf.) Presl) was frequently the most abundant herbaceous species, although brackenfern (*Pteridium aquilinum* (L.) Kuhn.) and redwood sorrel (*Oxalis oregana* Nutt.) were the most abundant herbs at some locations.

Streams within the study units were typically constrained topographically and of first and second hydrologic order (Olson and Rugger 2007; Anderson et al. 2007). The mean active channel width was 1.1 m (range 0.2-3.7 m), and mean depth was <10 cm. Nearly 70% of the reaches were summer intermittent with a wetted surface over >80% of their length. Sampled reaches were 60-410 m in length (mean 165 m) as defined by internodal distances in the stream network or by transitions between adjacent density management units.

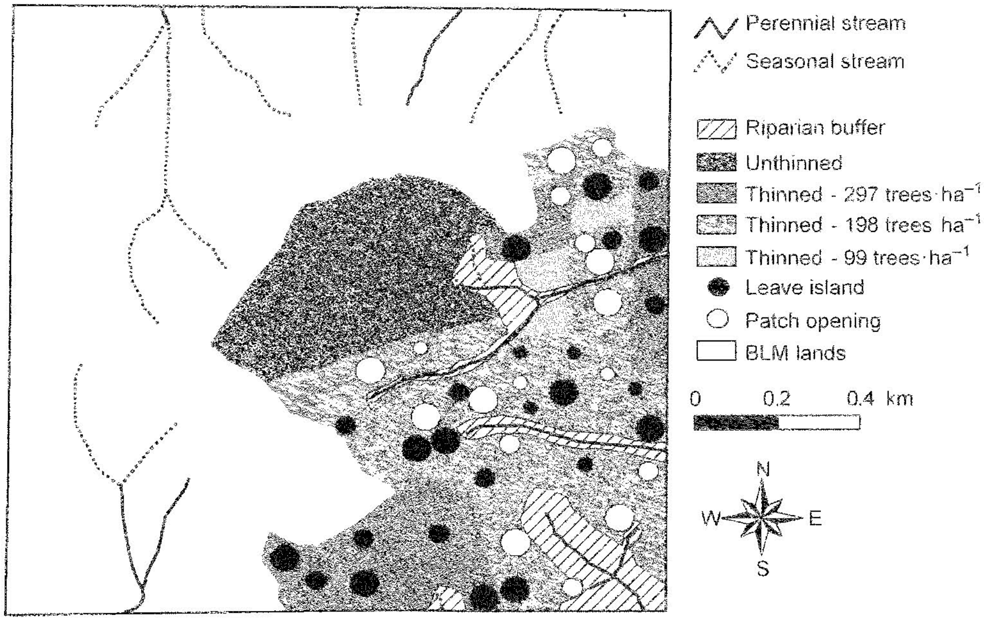
Treatments

Forest stands ranging in size from 16 to 46 ha were either thinned to moderate density (198 trees ha^{-1}) or left unthinned (UT; 500-865 trees·ha^{-1}). Thinnings targeted removal of trees that were of small diameter or that had defects. When present, species other than Douglas-fir were preferentially retained to provide tree species diversity. To enhance within-stand heterogeneity in thinned units, patch openings were created on 10% of the area and clusters of uncut trees (leave islands) were retained on 10% of the area (Fig. 1). Patch openings and leave islands were present in three sizes: 0.1, 0.2, and 0.4 ha, Of these, only the 0.4 ha patches are addressed by the analyses reported here.

Streams within treated units were bounded on both sides by riparian buffers, unthinned strips of forest adjacent to the stream channel, averaging 69 m (one site potential tree height, E1), 22 m (variable width, VB), or 9 m (streamside retention, SR) width as measured from stream center (Fig. 1). The B1 buffers, which ranged from 53 to 73 min width, were representative of riparian reserves specified for nonfish-bearing streams under interim guidelines of the Northwest Forest Plan (USDA and USDI 1994). Variable-width buffers had a minimum width of 12 m from stream center and a maximum width of 32 m from stream center based on features differentiating riparian and upland habitat, such as the transitions from channel to upland topography (slope break) or from riparian to upland vegetation. Streamside retention buffers consisted of all trees having a portion of their crown extending directly over the stream. Ranging from <5 to 14 m in width, SR buffers were presumed to provide greater bank stability and stream shading than a stand thinned to streamside but, perhaps, less moderation of microclimate within the stream and riparian zones relative to VB and B 1 buffers.

Six combinations of buffer width and adjacent upslope density management were evaluated: one site potential tree-height buffer adjacent to thinning (B1–T); one site potential tree height buffer adjacent to a 0.4 ha patch opening (B1-P); variable-width buffer adjacent to thinning (VB-T); variable-

Fig. 1. Treatment layout at the Green Peak density management and riparian buffer study site. Shades of gray show thinning intensity (residual trees per hectare). Hatched areas along streams are buffer treatments, including streamside retention, variable width, and one-site potential tree height buffers. BLM, US Bureau of Land Management.



width buffer adjacent to a 0.4 ha patch opening (VB-P); streamside retention buffer adjacent to thinning (SR-T); and unthinned (UT) serving as a reference. A shortage of suitable stream reaches at some sites prevented complete representation of all six treatments at all sites (Table 2). Furthermore, at four of five sites, the 198 trees·ha^{-1} thinning treatment was nonrandomly assigned to the experimental treatment unit having the highest density of streams to accommodate as many of the buffer treatments as possible (Cissel et al. 2006).

As previously reported (Anderson et al. 2007, fig. 3), treatment mean postthinning stand basal area ranged from 28.0 m^2·ha^{-1} in the SR-T treatment buffer to 51.0 m^2·ha^{-1} in the nominal buffer zone of the UT treatment (treatment effect: p = 0.145). Mean upslope posttreatment basal areas were 0.8 m^2·ha^{-1} for both the VB-P and B1-P patch openings; mean upslope basal areas in the SR-T, VB-T, B1-T, and DT treatments were 27.2, 30.4, 30.7, and 48.6 m^2·ha^{-1}, respectively.

Sampling design

Down wood and vegetation responses to the treatments were characterized through repeated sampling. Transects extended laterally from stream center through the riparian buffer and into the adjacent upslope density-management treatment. There was typically one reach per treatment per site and one or two transects per reach (Table 2). Transects were located near the longitudinal midpoints of the reaches. On some reaches, paired transects were installed to sample opposite sides of the stream. In total, data from 40 transects distributed among 26 reaches across five sites were used in the analyses (Table 2).

Transect lengths ranged from 41 to 151 m (mean 94 m) depending on the combination of buffer treatment and the potential dominant tree height. Where distance from stream to ridgeline permitted, transects extended a mnnmum of 65 m beyond the buffer into the adjacent upslope treatment. Given the variation in stand density arising from patch openings and leave islands embedded within the thinned units, edge effects were likely in sampling of thinned stands and patch openings.

Each transect was anchored with a sample point at stream center. The nearest streamside sample point was typically located 4.6 m slope distance (e.g., parallel to the ground surface) from the stream center; however, in a few instances, it was located at 6.1 or 9.1 m slope distance because of local variation in channel morphology. From the first streamside point to 22.9 m distance from stream center, sample points were spaced at intervals of 9.1 m slope distance. Beyond 22.9 m from stream center, sample points were spaced at intervals of 18.2 m slope distance. Sample point spacing was closer near stream to better sample variability associated with transitions in biotic and abiotic characteristics from channel to hillslope topography. Each sample point served as a location for vegetation and down wood measurements. Stand density was determined at each sample point with the exception of stream center.

Measurements

Abundance of down wood stratified into fine, small, and large size classes (<5.0, 5.0-29.9, and >30.0 cm diameter, respectively) was measured as percent cover. Percent wood cover correlates with the abundance of small vertebrates and their food sources (Carey and Johnson 1995; Bate et al. 2004) and has an advantage over other abundance metrics in precision of estimation (Bate et al. 2004). Cover was determined by measuring total length of cover for each stratum along two permanently marked 6 m transects centered on each sample point. The two transects were perpendicularly oriented, with one having been established along a random azimuth from plot center.

Table 2. Number of study reaches and number of transects per reach (given in parentheses) sampled by site and treatment.

Treatment	Green Peak	Keel Mountain	Bottomline	O.M. Hubbard	North Soup Creek	Treatment total
UT	1 (2)	1 (2)	1 (1)	1 (1)	1 (2)	5 (8)
B1-T	1 (2)	1 (1)	1 (1)	0	1 (2)	4 (6)
B1-P	1 (1)	1 (1)	1 (1)	0	1 (1)	4 (4)
VB-T	1 (2)	1 (2)	1 (1)	1 (1)	1 (1)	5 (7)
VB-P	1 (2)	1 (1)	1 (1)	1 (2)	1 (2)	5 (8)
SR-T	1 (2)	1 (2)	0	0	1 (3)	3 (7)
Site total	6 (11)	6 (9)	5 (5)	3 (4)	6 (11)	26 (40)

Note: Treatments are as follows: UT, unthinned stands; B1-T, one site-potential tree height buffer adjacent to a 198 trees·ha^{-1} thinning; B1-P, one site-potential tree height buffer adjacent to a 0.4 ha patch opening; VB-T, variable width buffer adjacent to a 198 trees·ha^{-1} thinning; VB-P, variable width buffer adjacent to a 0.4 ha patch opening; and SR-T, streamside retention buffer adjacent to a 198 trees·ha^{-1} thinning.

Cover and composition of understory vegetation were measured at each sample point based on the same 6 m transects used to quantify down wood cover. Understory vegetation was stratified into shrub, herbaceous, and moss classes. Fern species were included as a component of the herbaceous strata. In addition to percent cover, the dominant shrub species and the dominant herbaceous species within each subplot were recorded.

Measurements occurred prior to thinning (P_0) and twice - 1 or 2 years (P_1) and 3-5 years (P_2) - postthinning (Table 1).

Analysis

The study consisted of six treatments applied to individual stream reaches (treatment units) at five sites in a randomized complete-block design with missing observations (n = 26 reaches instead of n = 30 reaches for a balanced design). The five sites were considered representative of BLM lands in western Oregon, and therefore, site was included as a random effect in all statistical models (as in Anderson et al. 2007). Reach was nested within site. Reaches were stratified into buffer and upslope zones; because of the restricted randomization, the data were analyzed separately. Buffer zone and upslope zone responses were compared graphically (sensu Milliken and Johnson 1984, p. 352). It is important to note that B1 buffers extended beyond the riparian area into adjacent upland, terrestrial habitat. Similarly, SR buffers did not encompass the entire riparian area and therefore, the treated area adjacent to these buffers encompassed both riparian and upland terrestrial habitat. Thus, we distinguish "upslope zone" from upland and "buffer zone" from riparian to account for the confounding of topography within the treatment units.

Evaluations of down wood and understory vegetation responses were based on five dependent variables, all of which were measured with very few missing data. Down wood load response was based on percent cover of small (5-29.9 cm diameter) and large (>30 cm diameter) wood; understory vegetation response to the treatments was characterized by percent cover of shrubs, herbs, and moss. Our initial screening of these data included Pearson correlation and multivariate analysis of Valiance (MANOV A) analyses to examine the multivariate relationships. The five dependent variables were only marginally correlated (e.g., mean Pearson r value = 0.18), which supports our reporting the univariate response to the treatments. As noted by Tabachnick and Fidell (1996), the main advantage of a MANOV A for this case would be to control for a family-wise type 1 error, which we addressed using Bonferroni correction and the false discovery rate.

We generated inferences regarding buffer performance using linear mixed model analyses, equivalence-inequivalence tests, and two post hoc comparisons - Bonferroni correction and false discovery rate. The linear mixed models, equivalence-inequivalence tests, and post hoc p value adjustments were conducted at α = 0.1. This liberal significance threshold is reasonable given the inherent variation within and among forest stands and the logistical and financial limitations to replication associated with operational-scale management experiments (Monserud 2002). The weight of evidence in determining whether two treatments differed for a given dependent variable was the result of both of the equivalence-inequivalence tests and the two post hoc comparison methods.

Linear mixed models

We fitted two sets of linear mixed models in SAS MIXED procedure (Littell et al. 1996). In the first set, a linear mixed model using the reach-level response for each of the five dependent variables was fitted for buffer and upslope zones at P_0 and P_2 (20 models total). For the second set, a similar linear mixed model was fitted for buffer and upslope zones using the rate of change dependent variables (10 models total). For all linear mixed models, treatment was a fixed effect and site was a random effect.

Post hoc comparisons

Post hoc treatment comparisons used the least square means and standard errors estimated from the linear mixed models. All pairwise comparisons from the six treatments (15 comparisons total) were included because all were considered potentially meaningful in assessing relative treatment performance. We accounted for overall type 1 error rate using two methods. The first method was the Bonferroni correction, which controls for the risk of making at least one type 1 error (rejecting the null hypothesis when it is true) (McBride 2005). The second method was false discovery rate (FDR) which controls for the overall proportion of alpha-level rejections that could be in error - "the proportion of false discoveries" (Benjamini and Hochberg 1995, p. 292). The FDR is a more reasonable approach than the Bonferroni correction in discerning differences for studies with numerous comparisons (McBride 2005), as shown in recent water quality and genetics research (e.g., Weller et al. 1998; McBride 2005). The result of the FDR is typically between the Bonferroni method and unadjusted *p* values (McBride 2005).

In the FDR procedure adopted (Benjamini and Hochberg 1995; McBride 2005), we ranked the *P* values (P_i) for all the test comparisons $H_1, H_2, \ldots, H_m$ in ascending order (e.g., $P_1 < P_2 < \ldots < P_m$). Next, *k* was defined as the highest rank where $P_1 < q^*i/m$, where q^* was set to *a*, i was the ascending rank, and *m* was the total number of comparisons $(a = 0.1$ and $m = 15$ for this study). Finally, all H_i from $i = 1$ to $i = k$ were considered "statistical discoveries" (Soric 1989).

Equivalence and inequivalence tests

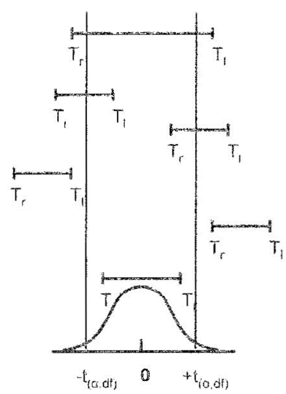
We subjected each post hoc comparison to both equivalence and inequivalence tests (Parkhurst 2001; Dixon and Pechmann 2005; McBride 2005). A definition of "equivalence interval" is required to perform these tests. Ideally, the width of the equivalence interval would be the maximum difference between two populations that was ecologically insignificant. However, operationally, we can only approximate the ecologically important difference, and thus, we prefer the term "practical" difference. In addition, we prefer to base our conclusions on the performance of several equivalence interval widths.

We adapted the equivalence-inequivalence test procedure and nomenclature of McBride (2005). The equivalence hypothesis is H_e: $8_L < 8 < 8_L$, where 8_L is the lower bound and 8_u is the upper bound, and asserts that the difference between population means *(8)* is within the equivalence interval (8_L to 8_u); rejection of H_e: infers the alternative hypothesis of inequivalence K_e: $8_L < 8$ or $8 > 8_u$ (McBride 2005). The inequivalence hypothesis is H_i: $8 < 8_L$ or $8 > 8_u$ and states that the difference between the population means is outside of the equivalence interval, and its rejection infers the alternative hypothesis of equivalence K_i: $8_L < 8 < 8_u$ (McBride 2005). Note that we used the nomenclature presented in McBride (2005) to emphasize the difference with the point-null test and its nomenclature (H_0 and H_a). The hypotheses are evaluated using the two one-sided test method, which tests both parts of the hypothesis separately. Thus, rejection of the equivalence test's hypothesis is evidence for inequivalence and rejection of the inequivalence test's hypothesis is evidence for equivalence (Fig. 2). Failure to reject either H_e or H_i indicates insufficient evidence to support either equivalence or inequivalence (McBride 1999; Parkhurst 2001). The inequivalence hypothesis test provides a "proof of safety" approach because equivalence is inferred only if the there is sufficient evidence to reject inequivalence (McBride 2005).

Equivalence testing is not a power analysis. Although both the power analysis and equivalence test require an a priori definition of a critical effect size, the two methods differ in intent and in application. Power analysis is a prospective test best used in the planning stage of an experiment to determine the number of samples required to regularly detect (e.g., 80% of the time) a specified difference (Parkhurst 2001; Dixon and Pechmann 2005; McBride 2005). It uses estimates of the mean and standard deviation of the populations to provide information for the future performance of a test in determining a population effect. In contrast, the equivalence test applies to a specific set of data. Although not recommended (Hoenig and Heisey 2001), power analysis following a point-null test has been used to determine the probability of making a type II error. Even though both the post hoc power analysis and equivalence testing use a "negligible difference," the detection regions differ depending on the sample standard error for a given test (McBride 2005, fig. 5.13).

Fig. 2. Interpretation for all the possible outcomes of the decision rules applied to test scenarios for the combined equivalence–inequivalence hypothesis tests. These tests involve the comparison of the two test statistics (T_r to T_l), which encompasses the a priori "negligible" difference between the two means, with the Student's *t* distribution value. The results of two one-sided *t* tests determine whether the equivalence test is rejected (evidence for inequivalence), the inequivalence test is rejected (evidence of equivalence), or neither test rejected (inconclusive).

Test Scenarios



Outcome and Interpretation

H_e	H_i	Conclusion
Accept	Reject	Evidence of Equivalence
Accept	Accept	Inconclusive
Accept	Accept	Inconclusive
Reject	Accept	Evidence of Inequivalence
Reject	Accept	Evidence of Inequivalence
Accept	Accept	Inconclusive

Inequivalence test:
reject H_i if $T_r \le -t_{(\alpha,df)}$ *and* $T_l \ge t_{(\alpha,df)}$

Equivalence test:
reject H_e if $T_r > t_{(\alpha,df)}$ *or* $T_l < -t_{(\alpha,df)}$

Through preliminary application of these tests, we discovered that the outcome can be influenced by which of the two treatments being compared is designated as the reference (see Tables 3 and 4 for the complete list of comparisons). Because there was no clear reference in our comparisons, we modified the protocol by substituting the mean of the two treatment means as the reference mean. In this modified version of the test, the outcome was not dependent upon which of the two treatment means was designated as the reference. We defined 8_L and 8_u of the equivalence interval as $8_L = -c \times \underline{X_1X_2}$ and $8_u = c \times \underline{X_1X_2}$ where c is a proportional constant (0.1, 0.2, or 0.5 were used in these tests) and $\underline{X_1X_2}$ is the mean of the absolute values of two treatment means. The test statistics were defined as $T_1 = (d - 8_L)/SE$ (d) and $T_1 = (d + 8_U)/SE$ (d), where T_r and T_1 were the right and left test statistics and SE (d) is the standard error of the difference between the two sample means (d). The decision rule for the equivalence hypothesis is reject H_e if $T_1 < -(t\,a,$ df) or $T_r > t(a,$ df), where $t(a,$ df) is the Student's *t* distribution at size *a* with df degrees of freedom. Rejection of either test was evidence of inequivalence (Fig. 2). The decision

Table 3. Equivalence–inequivalence test results at α = 0.1 for the preharvest sample (P_0) and the second postharvest measure (P_2) on the difference in least square means for the buffer zone and upslope zone.

		Small wood		Large wood		Shrub cover		Herb cover		Moss cover	
Treatment	Reference	P_0	P_2	P_0	P_2	P_0	P_2	P_0	P_2	P_0	P_2
Buffer zone											
B1-P	UT						IIn		nnE		nnE
B1-T	UT					nnE	nnE		nnE		
SR-T	UT	IIn		Inn		IIn		IIn*	III*,†	III	
VB-P	UT			IIn*			nnE	IIn*	Inn*		nnE
VB-T	UT							Inn			Inn
B1-P	B1-T						IIn		nnE		
SR-T	B1-T	III*		IIn		III	Inn	IIn*	III*,†	III	
VB-P	B1-T			III*	IIn		nnE	IIn	Inn		
VB-T	B1-T							IIn*			
SR-T	B1-P	IIn						Inn	III*,†		
VB-P	B1-P			IIn			IIn		Inn*		nnE
VB-T	B1-P										Inn
SR-T	VB-T	IIn		III*					IIn	IIn	
VB-P	VB-T		nnE	III*,†	IIn			nnE	nnE		Inn
SR-T	VB-P	III*,†				Inn		nnE	IIn	Inn	
Upslope zone											
B1-P	UT	nnE			Inn	IIn	IIn		IIn*		
B1-T	UT						nnE				
SR-T	UT									nnE	Inn
VB-P	UT		nnE			Inn	nnE		IIn	nnE	
VB-T	UT		nnE							nnE	nnE
B1-P	B1-T		nnE			Inn	IIn		III*,†		IIn
SR-T	B1-T						Inn				nnE
VB-P	B1-T					Inn	nnE	IIn	IIn*	nnE	Inn
VB-T	B1-T		nnE						Inn		nnE
B1-P	VB-T		nnE						Inn		IIn
SR-T	VB-T								nnE	nnE	
VB-P	VB-T		nnE	nnE		IIn*				nnE	
B1-P	SR-T								Inn		III*,†
VB-P	SR-T					IIn				nnE	IIn
VB-P	B1-P			Inn	Inn	III*,†	IIn	IIn	nnE		

Note: Three tests were conducted for each comparison using a minimum important effect size of 10%, 20%, and 50% of the reference for each comparison (I, equivalence test was rejected, and the inequivalence test was not; E, equivalence test was not rejected, and the inequivalence test was rejected; n, neither test was rejected because of insufficient evidence; e.g., III, I was found at each of the three levels). Blank entries indicate insufficient evidence existed to reject either test at all three levels. See Fig. 2 and the text for details of the tests.

*Values are within the false discovery threshold.

†Values are within the Bonferroni multiple comparison threshold.

rule for the inequivalence hypothesis is reject H_i if $T_l \geq t(\alpha, df)$ and $T_r \leq -t(\alpha, df)$. Rejection of both tests was evidence of equivalence (Fig. 2).

In this study, equivalence-inequivalence tests were conducted at three interval levels: 10%, 20%, and 50% of the mean of the absolute values of the two treatment means. Based on our professional judgment, intervals defined by 20% may be equivalent. Testing at higher and lower equivalence intervals provided a measure of sensitivity. The 10%-50% range in equivalence intervals was arbitrary; however, our intent was to encompass an ecologically meaningful difference.

Results

Small and large wood

There were substantial pretreatment differences in small and large wood abundance within the buffer zone (Table 3, Fig. 3). This was due to relatively high levels of small and large wood in treatment units assigned the narrowest buffers (SR-T) and large wood in treatment units assigned narrow buffers adjacent to patch cuts (VB-P). In contrast, pretreatment small and large wood cover within the upslope zone was similar across treatments.

Posttreatment differences in small and large wood mean abundance in the buffer zone were less apparent (Fig. 3). Rate of change between P_1 and P_2 for small and large wood was more complex in the upland zone. Although the rate of change was mostly within ±1% of 0 overall (Table 4, Fig. 4), the rate of change for small wood tended to decrease more in the narrower treatments.

Based on the equivalence-inequivalence tests, strong evidence for inequivalence was found between SR-T, which had the greatest cover, and both VB-P and B1-T for small

Table 4. Equivalence–inequivalence test results at $\alpha = 0.1$ for the difference in least squares mean rate of change between the first and second postharvest samples (P_1 and P_2) for the buffer zone and upslope zone.

Treatment	Reference	Small wood	Large wood	Shrub cover	Herb cover	Moss cover
Buffer zone						
B1-P	UT					
B1-T	UT					
SR-T	UT			IIn	IIn	
VB-P	UT					
VB-T	UT				IIn	
B1-P	B1-T					
SR-T	B1-T				IIn	
VB-P	B1-T					
VB-T	B1-T					
SR-T	B1-P			III	III	
VB-P	B1-P			IIn	Inn	
VB-T	B1-P					
SR-T	VB-T	Inn			III*,†	
VB-P	VB-T				III	
SR-T	VB-P					
Upslope zone						
B1-P	UT		Inn	III	III*,†	
B1-T	UT			IIn		Inn
SR-T	UT			IIn		IIn
VB-P	UT	IIn		III	Inn	IIn
VB-T	UT	Inn				
B1-P	B1-T		III		IIn*	
SR-T	B1-T		III			
VB-P	B1-T					
VB-T	B1-T					
B1-P	VB-T		IIn	IIn	IIn*	
SR-T	VB-T		IIn			
VB-P	VB-T			Inn		
B1-P	SR-T				III*	
VB-P	SR-T		III			
VB-P	B1-P		III		IIn	

Note: See Table 2 for a description of the treatment and reference names and Table 3 for a description of the symbols used.

*Values are within the false discovery threshold.

†Values are within the Bonferroni multiple comparison threshold.

wood in the buffer at P_0 (Table 3, Fig. 3). There was evidence for equivalence at the broadest definition (50% minimum effect size) for six of the comparisons for small wood in the upland at P_2. For most of the other comparisons, there was insufficient evidence to reject either test. For the rate of change comparisons, sufficient evidence of difference was found for only large wood in the upland zone; all contrasts between treatments with positive slopes (B1-T, VB-T, and VB-P) and those with negative slopes (SR-T and B1-P) were determined inequivalent at some level (Table 4, Fig. 4).

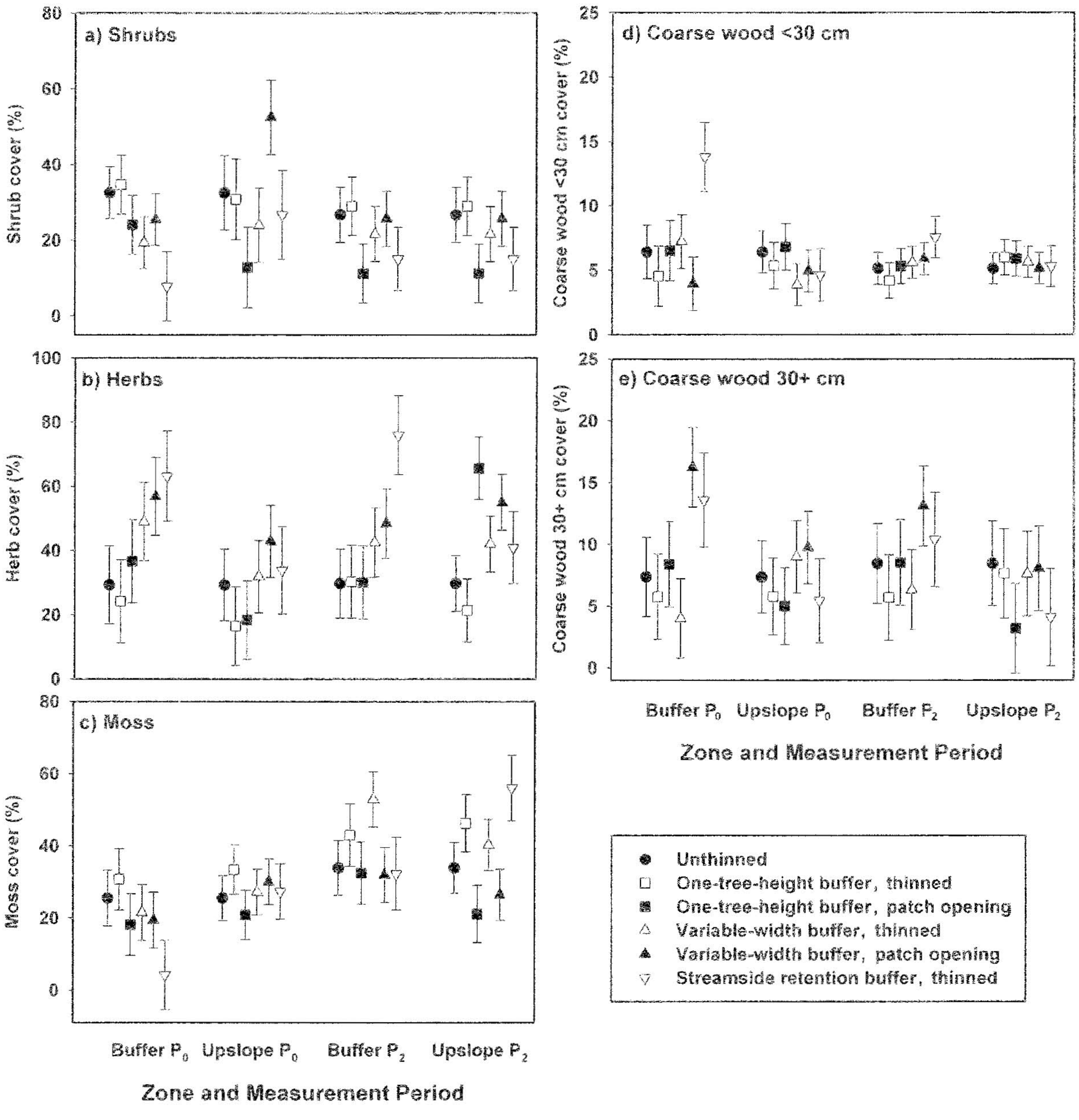
Understory vegetation

By visual inspection, pretreatment mean covers arrayed by decreasing buffer width were suggestive of linear trends for increasing herbs and decreasing shrubs and moss in the buffer zone (Fig. 3). The strong differences in herb cover were between the widest (UT and B1-T) and narrowest (SR-T, VB-P, and VB-T) butlers (Tables 3 and 4, Fig. 3). In the upslope zone, cover for the three understory variables was relatively uniform across treatments at P_0, with the exception of shrub cover, where VB-P and B1-P strongly differed (Table 3, Fig. 3).

At P_2, treatment differences in mean herb cover in the buffer zone were similar to those at P_0, whereas moss cover generally increased in the buffer zone of treated units (Fig. 3). In the upland zone at P_2, differences in shrub cover were not apparent, but increasing cover resulted in emerging differences for herbs (e.g., B1-T and B1-P) and moss (e.g., B1-P and SR-T) (Table 3, Fig. 3). The greatest rate of change over the entire study was increasing herb cover in the upslope zone of B1-P, which strongly contrasted with all treatments except VB-P (Table 4, Fig. 4). Rate of change of herb cover was also greatest in the SR-T butler zone and contrasted strongly with VB-T (Table 4, Fig. 4).

The equivalence-inequivalence tests support the results

Fig. 3. Treatment least square means for (*a*) shrub, (*b*) herbaceous, (*c*) moss, (*d*) small wood, and (*e*) large wood cover at the pretreatment measurement (P_0) and second posttreatment measurement (P_2). Treatment means are arrayed from widest (unthinned) to narrowest (streamside retention buffer, thinned) buffer zone widths. Error bars are SEs.



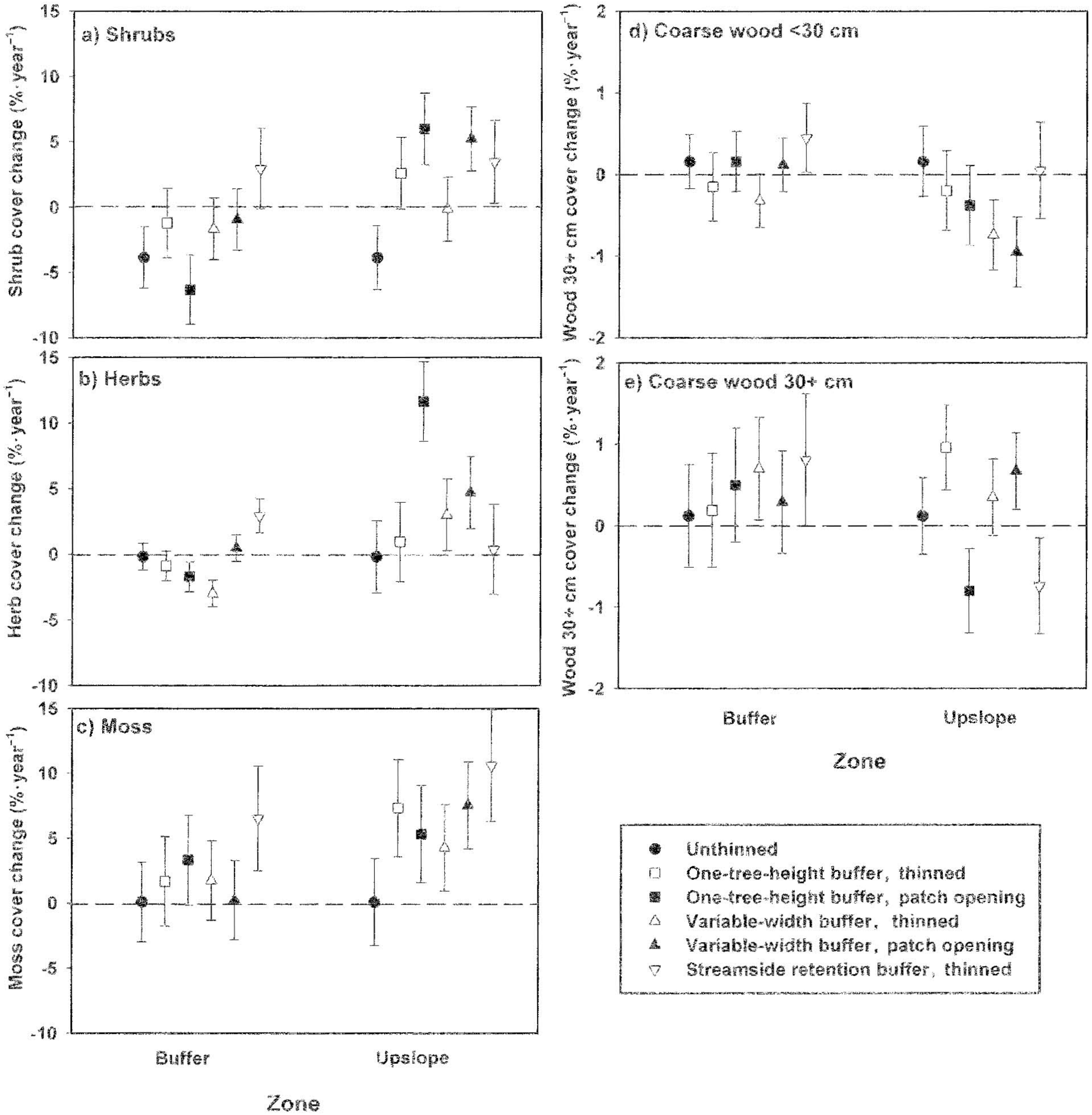
described above and provide additional insights as well. The linear trends visually identified for the three variables in the buffer zone at P_0 were determined to be inequivalent, at least at the extremes of buffer width, and all involved a comparison with SR-T. Herb cover had the greatest number of comparisons that were inequivalent and tended to be with SR-T in the buffer zone and B1-P in the upslope zone. Evidence of equivalence at the broadest definition occurred at P_2 for shrub cover among the same three treatments (UT, B1-T, and VP-P) in the buffer and upslope zones (Table 3, Fig. 3). For moss cover, the greatest numbers of inequivalent comparisons were at P_0 in the buffer zone and P_2 in the upslope zone - both driven mainly by comparisons with extreme covers for SR-T. The greatest numbers of equivalent comparisons (at the broadest level) for moss occurred at P_2 in the buffer zone and P_0 in the upslope zone. Herb cover had the greatest number of inequivalent comparisons for the rate of change variable - driven by SR-T in the buffer and B1-P in the upslope zone (Table 4, Fig. 4).

Discussion

Down wood and vegetation responses to thinning and buffer width

The lack of substantial changes in down wood for either the thinned matrix or patch openings is consistent with at

Fig. 4. Treatment least squares means for (*a*) shrub, (*b*) herbaceous, (*c*) moss, (*d*) small wood, and (*e*) large wood cover for the rate of change in cover between the first (P_1) and second (P_2) posttreatment measurements. Treatment means are arrayed from widest (unthinned) to narrowest (streamside retention buffer, thinned) buffer zone widths. Error bars are SEs.



least one other variable-retention thinning study in the Pacific Northwest. For thinning treatments that included fine-scale features such as gaps and leave islands, Roberts et al. (2007) observed that wind damage affected approximately 1.9% of trees in treated units, relative to only approximately 0.5% of trees in unthinned units, with occurrence being highly variable. Although damage adjacent to gaps tended to be greater than that within the thinned matrix, damage was more common for gaps placed in topographically vulnerable settings (Roberts et al. 2007). Our findings based on patches that were approximately 1.5 times larger in diameter support the conclusion of Roberts et al. (2007) that small gaps are relatively resistant to edge-related windthrow.

The tendency for convergence (from P_0 to P_2) of mean small wood abundance across treatments was strongly associated with decreased cover in buffers of the SR-T treatment (Fig. *3d*). As tree crowns stabilized following thinning, recruitment into the buffers may have declined with the effect being more pronounced for narrow buffers. There was an apparent decrease in the potential for recruitment of small wood from uplsope sources in most of the treatments (Fig. *4d*), with the extent of decrease possibly related to greater increased edge to buffer ratios for narrower buffers.

Concurrently, near-stream small wood may have been undergoing downstream transport; thus, where buffers are narrow, a greater proportion of the buffer was influenced by peak flows and subject to small wood removal. In-stream surveys of wood >10 cm diameter also indicated a postharvest increase in the minimum piece density (Olson and Rugger 2007). In contrast, the greater variability and inconsistent postharvest rates of change observed for large wood cover in the upslope zone (Fig. 4e) could reflect predominance of stochastic recruitment processes such as windthrow. Unfortunately, specific recruitment processes were unidentified for down wood observed in om study.

A likely driver of the observed vegetation responses to thinning and buffer width was canopy light transmittance. Increased herb and moss abundances in thinned areas and patch openings of the upslope zone or in narrow buffers adjacent to patch openings corresponded to previously reported treatment differences in overstory canopy cover and air temperature (Anderson et al. 2007). It was shown that mean maximum air temperatures were elevated in SR buffers (by 4°C) and within upslope patch openings (by 6-9 °C) relative to unthinned stands (see Anderson et al. 2007, fig. 6a). These increased temperatures were associated with similar patterns of canopy openness (see Anderson et al. 2007, fig. 3b). As edge to area ratio increased, mean canopy light transmittance for the buffer increased - a greater portion of the buffer width experienced increased levels of understory light associated with the adjacent harvest. Increased levels of light should promote increased vegetation abundance. However, the response is likely to vary by vegetation taxon: ferns and forbs respond more rapidly than shrubs. This is due to reproductive life history, fecundity, and growth rate. Although initially increasing in cover, the abundance of ferns and forbs may later decrease if shrub cover increases to the point of shading the shorter ferns and forbs (Wilson and Puettmann 2007), A factor confounding anticipated vegetation cover response among buffers is that a greater proportion of the area within narrow buffers is riparian; a streamside retention buffer is almost exclusively riparian in contrast to a one tree height buffer comprising riparian area plus a large proportion of terrestrial terrain. Furthermore, variations in channel aspect will also influence the effect of canopy openness on understory light and air temperature (Heithecker and Halpern 2006).

Although we report vegetation responses generalized by lifeform, it is important to recognize that our results may be species dependent. *Polystichum munitum* (Kaulf.) C. Presl commonly occurred as a predominant herbaceous layer of buffer zones prior to treatment (Appendix A). Following treatment, *Polystichum munitum* cover increased in some treatments, but its initial abundance likely limited the potential for an overall herb response to increased resources or for the increase or colonization by other herbaceous species. In contrast, for the upslope zone, it was less common for *Polystichum munitum* to be the most abundant herbaceous species pretreatment. Following treatment in the upslope zone, substantial increases in herb cover were associated with increasing occurrence of *Pteridium aquilinum* (L.) Kuhn or other herbaceous species as being locally most abundant (Appendix A). Among shrubs, the most consistent responder in harvested areas was *Rubus ursinus* Cham. & Schltdl., which increased in frequency of dominance, particularly in patch openings (Appendix B),

Desiccation-prone bryophytes can be sensitive bioindicators of microclimate alterations associated with harvest edges (Hylander 2005). However, some species of moss can be early colonizers of areas disturbed by harvest (Dynesius and Hylander 2007). In the temperate Douglas-fir-western hemlock forest of the western Cascades, Nelson and Halpern (2005) observed substantial decreases in moss cover 1 year following overstory removal. This apparent contrast to the increased moss cover we observed for some upslope treatments (Table 4, Fig. 4) may be due to different species compositions or to thinning resulting in less severe microclimate conditions than complete overstory removal.

An important question is whether the abundance of wood and vegetation in buffers responds differently to adjacent thinning or patch openings. Patch cutting adjacent to a buffer may favor tree fall into the buffer because of windthrow at an abrupt edge, In contrast, less intensive thinning would result in a more diffuse edge and less propensity for windthrow. As with the vegetation response, these generalizations are subject to the confounding influence of local topography on wind direction, turbulence, and velocity. Patch openings create substantially stronger light conditions than thinning (Anderson et al. 2007; Fahey and Puettmann 2008) and, therefore, could accentuate buffer vegetation responses adjacent to patch openings, relative to thinning.

We have limited information from which to infer the ecological implications of the observed down wood and vegetation responses. Sampling some of the same sites and reaches as those included in our study, Olson and Rugger (2007) did not observe any deletrious effects of upslope thinning on the instream or stream bank occurrence of 13 amphibian species. Two species, rough-skinned newts (*Taricha granulosa* Skilton) and coastal giant slamanders (*Dicamptodon tenebrosus* Baird & Girard), demonstrated increased abundance in buffers relative to riparian areas of untreated reaches, but the analysis did not detect differential effects of buffer width (Olson and Rugger 2007). However, in a limited case study at two of the sites, surveys of amphibian abundance suggested associations with the abundance of down wood and riparian buffers (Rundio and Olson 2007), because down wood provided microhabitats that met critical temperature and humidity requirements for burrowing salamanders (Kluber et al. 2009).

Detection of equivalence and inequivalence

A key issue in the application of equivalence-inequivalence testing is the establishment of minimum differences of practical significance. This can be difficult, particularly because species or taxa respond differently to similar environmental or habitat conditions and perturbations. We assumed that a practical difference existed between the narrow (10% equivalence interval) and broad (50% equivalence interval) definitions of equivalence interval. The narrower the definition of the equivalence interval, the more likely the equivalence test will be rejected (evidence of inequivalence). Thus, rejection of an equivalence test at the 50% level- and thus at the 10% and 20% levels - is strong evidence that the two buffer treatments differed. For all pairwise comparisons indicated as different using the Bonferroni threshold, the equivalence test at the *50%* level was also rejected. Likewise, rejection of an inequivalence test at the 10% level - and thus at the 20% and *50%* levels - is strong evidence of equivalence. No post hoc comparison was "equivalent" at all three levels of the equivalence interval definition. In fact, rejection of the inequivalence hypothesis occurred only at the least restrictive definition of the equivalence interval *(50%).* In all of these cases, tests at the 10% and 20% level failed to reject both the equivalence and inequivalence hypotheses. This is the weakest test where equivalence is inferred.

Existing decision-support tools and published literature were useful to varying extents in defining minimum practical differences for down wood and understory vegetation. The DecAid Advisor (Mellon et al. 2002; http://www.fs.fed.us/r6/nr/wildlife/decaid/) is a decision-support tool developed to inform resource managers of characteristic abundances of snags and down wood in Pacific Northwest forests as coarsely stratified by wildlife habitat type, successional stage of forest development, and disturbance history. In stands of small- to medium-sized trees of westside lowland conifer-hardwood forests of the Oregon coast and western side of the Cascade Range, typical down wood abundances *(>12:5* cm diameter) under natural conditions range from negligible to approximately 16% cover, with approximately *50%* of sites having < 4% cover. Mean percent cover observed pre- and post-treatment in this study were similar to these regional values (Figs. *3d* and *3e).* Our down wood estimates, which were summed across both small and large diameter classes, tend to retlect abundances associated with the upper 25th percentile of the regional distribution of down wood for unharvested forests. The slightly greater down wood abundances of unthinned stands relative to treated stands were indistinguishable in terms of regional reference distributions.

In addition to providing descriptions of regional down wood distributions, DecAid also provides summaries of wildlife usage in relation to down wood percent cover. Cumulative species curves derived from a metaanalysis of western Oregon and Washington wildlife studies illustrate the marginal benefit, or additional habitat service, associated with increasing down wood cover. With a 50% tolerance interval (down wood cover that encompasses the abundance conditions associated with *50%* of species observances), an increase in down wood from 4% to 6% cover is associated with an increase in habitat use from 2 to 12 species of wildlife. Similarly, an increase in down wood from 8% to 10% cover is associated with an increase in use from 18 to 25 species of wildlife. Given the total number of species considered in the functions, a 1% increase in down wood cover is associated with an additional 2%-12% increase in the number of species potentially served.

Based on the species - clown wood cover relationships summarized in DecAid, for a *50%* tolerance level, it is reasonable to consider a 1% increase in down wood cover to be of practical significance. Although it follows that a 1% change in down wood cover is a reasonable, minimum practical difference for equivalence-inequivalence testing, a 1% difference may be difficult to resolve with precision from survey data (see Figs. *2d* and *2e).* Sampling precision may limit our ability to estimate precisely a marginal gain in habitat associated with a forest manipulation or disturbance. Furthermore, such a dynamic relationship defines an extremely narrow equivalence region. If changes in percent down wood cover are measured very precisely, most harvest units would likely accrue the minimum practical difference in down wood cover, leading to the rejection of equivalence (reject H_e). If measured imprecisely, the width of the critical zone, T_r to T_1 narrows, and the test statistics would be less likely to span the range of the Student's critical values *(-t* to *t)* and, therefore, more likely result in an inconclusive determination (see Fig. 2). Thus, perhaps paradoxically, for those sampled attributes in which small changes can have large ecological effect, increased precision of estimation can enhance the ability to infer differences, but it may also result in less likelihood for inferring equivalence. This precautionary characteristic of equivalence testing has been termed "proof of safety" by McBride *(2008)* and has been particularly important to applications in clinical drug trials,

Whereas DecAid provides a means to develop ecological criteria for defining significant effect sizes, it does not directly address wood loads associated with streams. Fox and Bolton (2007) compiled a similar database characterizing the distributions of instream wood abundance by stream width and forest type for unmanaged forest basins in western Washington. Although they suggest using these stream wood distributions, which are essentially the observed natural range of variability, to establish stream restoration targets for salmonid habitat, they acknowledge that the specific quantities and volumes of instream wood needed for salmonid production are unknown.

Other examples of quantitative linkages between riparian vegetation characteristics and ecological functionality exist. Waterhouse et al. (2002) quantified the conditional probability of male winter wren *(Troglodytes troglodytes* L.) song perch use as an asymptotic function increasing with mean distance to overstory trees and as a linear function decreasing with percent moss cover. Peterman and Semlitsch (2009) developed models relating salamander larval abundance to percent sedimentation and buffer width. These functions could be used in calculating marginal changes in habitat-use associated with a change in structural condition and, thus, provide criteria for establishing equivalence intervals. However, individual species response functions may not prove broadly applicable. For example, it has been demonstrated that thinning of young Douglas-fir stands can increase the abundance and diversity of nonconiferous understory vegetation which is importanr to a broad array of wildlife species (Hagar 2007; Wilson and Puettmann 2007). Yet various species groups will use habitats at various spatial scales (Wilson and Puettmann 2007) or will be keyed to different elements of the vegetation community (Hagar 2007) with the net result that, although many species may benefit, some species may decline in response to thinning (Hagar et al. 2004).

Conclusion

In this study, we have demonstrated that thinning of young second-growth Douglas-fir forests, as practiced in the BLM density management study, can lead to subtle changes in understory vegetation cover and composition. In the treated upland, these responses tended to be most prevalent in patch openings. Vegetation responses within unthinned buffers were most prevalent when buffers were narrow, particularly when located directly adjacent to patch openings. Thinning tended to make small down wood cover more homogenous across reaches. Equivalence-inequivalence testing was useful in assessing the significance of these responses. Equivalence was more difficult to discern than inequivalence. However, definition of equivalence intervals was somewhat subjective. To construct truly useful tests of ecological importance, we need to develop better functional understandings of the changes in ecological processes or habitat characteristics associated with changes in the covers of understory vegetation or down wood. Improved functional understanding will help us discern the ecological significance of subtle forest responses to management treatments.

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Appendices

Appendices A and B begin on the following page.

Appendix A

Table A1. Shrub composition in buffer and upslope zones by buffer type and upslope treatment prior to and following treatment.

	Buffer zone						Upslope zone					
Shrub type*	Buffer treatment	Sample size (n)	P_0 shrubs (%)	P_1 shrubs (%)	P_2 shrubs (%)	p	Upslope treatment	Sample size (n)	P_0 shrubs (%)	P_1 shrubs (%)	P_2 shrubs (%)	p
None	UT	14	14.3	7.1	20.0	0.795	UT	40	10.0	7.5	10.0	1.000
ACCI	UT		21.4	28.6	20.0	1.000	UT		22.5	27.5	22.5	1.000
MANE	UT		7.1	21.4	6.7	1.000	UT		10.0	12.5	15.0	0.616
GASH	UT		14.3	7.1	6.7	0.549	UT		20.0	15.0	17.5	0.884
RUUR	UT		0.0	0.0	0.0	nd	UT		0.0	0.0	2.5	0.667
VAPA	UT		28.6	28.6	13.3	0.390	UT		15.0	17.5	17.5	0.882
Other	UT		14.3	7.1	33.3	0.241	UT		22.5	20.0	15.0	0.408
None	B1	46	15.2	17.4	17.4	0.890	T	93	22.6	12.9	1.1	<0.001
ACCI	B1		30.4	28.3	19.6	0.287	T		26.9	23.7	21.5	0.441
MANE	B1		17.4	17.4	21.7	0.691	T		12.9	16.1	15.1	0.757
GASH	B1		8.7	8.7	6.5	0.849	T		8.6	5.4	5.4	0.458
RUUR	B1		0.0	2.2	4.3	0.243	T		3.2	9.7	11.8	0.045
VAPA	B1		15.2	10.9	10.9	0.637	T		8.6	8.6	10.8	0.707
Other	B1		13.0	15.2	19.6	0.479	T		17.2	23.7	34.4	0.009
None	VB	29	10.3	13.8	13.8	0.711	P	49	14.3	8.2	0.0	0.011
ACCI	VB		24.1	20.7	13.8	0.429	P		30.6	22.4	14.3	0.070
MANE	VB		17.2	13.8	17.2	0.863	P		18.4	20.4	12.2	0.504
GASH	VB		17.2	20.7	13.8	0.869	P		14.3	10.2	6.1	0.245
RUUR	VB		0.0	0.0	6.9	0.228	P		0.0	26.5	30.6	<0.001
VAPA	VB		13.8	20.7	13.8	0.863	P		6.1	8.2	4.1	0.836
Other	VB		17.2	10.3	20.7	0.735	P		16.3	4.1	32.7	0.047
None	SR	5	20.0	40.0	20.0	1.000						
ACCI	SR		0.0	0.0	0.0	nd						
MANE	SR		40.0	40.0	20.0	0.749						
GASH	SR		20.0	0.0	40.0	0.703						
VAPA	SR		20.0	0.0	0.0	nd						
RUUR	SR		0.0	0.0	0.0	nd						
Other	SR		0.0	20.0	20.0	0.667						

Note: Values are percentages of sample locations (n) dominated by a species at P_0, P_1, and P_2, where dominance is defined as a species having the greatest cover among all species present. Buffer and upslope treatments are defined in Table 2. Probability is an indicator of significant change in percent dominance by a given species over the P_0 to P_2 measurement period as determined by the Cochran–Armitage trend test. The p values are unadjusted for the overall number of comparisons. nd, not determined.

*ACCI, *Acer circinatum*; MANE, *Mahonia nervosa*; GASH, *Gaultheria shallon*; VAPA, *Vaccinium parviflora*; RUUR, *Rubus ursinus*; other, species other than those listed; none, no shrub component.

Appendix B

Table B1. Herbaceous species composition in buffer and upslope zones by buffer type and upslope treatment prior to and following treatment.

	Buffer zone						Upslope zone					
Herb type*	Buffer treatment	Sample size (n)	P_0 herbs (%)	P_1 herbs (%)	P_2 herbs (%)	p	Upslope treatment	Sample size (n)	P_0 herbs (%)	P_1 herbs (%)	P_2 herbs (%)	p
None	UT	14	28.6	7.1	7.1	0.116	UT	40	45.0	12.5	10.0	<0.001
OXOR	UT		0.0	0.0	0.0	nd	UT		0.0	0.0	2.5	0.667
POMU	UT		71.4	71.4	78.6	1.000	UT		52.5	75.0	70.0	0.127
PTAQ	UT		0.0	0.0	0.0	nd	UT		0.0	0.0	0.0	nd
Other	UT		0.0	21.4	14.3	0.187	UT		2.5	12.5	17.5	0.047
None	B1	46	15.2	0.0	0.0	<0.001	T	93	25.8	5.4	1.1	<0.001
OXOR	B1		0.0	0.0	0.0	nd	T		7.5	15.1	9.7	0.723
POMU	B1		80.4	91.3	93.5	0.073	T		55.9	52.7	63.4	0.336
PTAQ	B1		2.2	4.3	4.3	0.787	T		3.2	8.6	10.8	0.070
Other	B1		2.2	4.3	2.2	1.000	T		7.5	18.3	15.1	0.165
None	VB	29	13.8	0.0	0.0	0.097	P	49	32.7	0.0	0.0	<0.001
OXOR	VB		0.0	0.0	0.0	nd	P		6.1	8.2	6.1	1.000
POMU	VB		86.2	93.1	93.1	0.850	P		55.1	36.7	44.9	0.363
PTAQ	VB		0.0	6.9	3.4	0.742	P		0.0	4.1	16.3	0.002
Other	VB		0.0	0.0	3.4	0.075	P		6.1	51.0	32.7	0.006
None	SR	5	0.0	0.0	0.0	nd						
OXOR	SR		40.0	40.0	20.0	0.750						
POMU	SR		60.0	60.0	80.0	0.750						
PTAQ	SR		0.0	0.0	0.0	nd						
Other	SR		0.0	0.0	0.0	nd						

Note: Values are percentages of sample locations (n) dominated by a species at P_0, P_1, and P_2, where dominance is defined as a species having the greatest cover among all species present. Buffer and upslope treatments are defined in Table 2. Probability is an indicator of significant change in percent dominance by a given species over the P_0 to P_2 measurement period as determined by the Cochran–Armitage trend test. The p values are unadjusted for the overall number of comparisons. nd, not determined.

*OXOR, *Oxalis oregana*; POMU, *Polystichum munitum*; PTAQ, *Pteridium aquilinum*; other, species other than those listed; none, no herbaceous component.