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Amphibian distributions in riparian and upslope areas and their habitat associations on managed forest landscapes in the Oregon Coast Range

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ABSTRACT

Over the past 50 years, forested landscapes of the Pacific Northwest have become increasingly patchy, dominated by early successional forests. Several amphibian species associated with forested headwater systems have emerged as management concerns, especially after clearcutting. Given that headwater streams comprise a large portion of the length of flowing waterways in western Oregon forests, there is a need to better understand how forest management affects headwater forest taxa and their habitats. Mitigation strategies include alternatives to clearcutting, such as harvests that remove only part of the canopy and maintenance of riparian buffer strips. Our study investigates effects of upland forest thinning coupled with riparian buffer treatments on riparian and upland headwater forest amphibians, habitat attributes, and species-habitat associations. Amphibian captures and habitat variables were examined 5–6 years post-thinning within forest stands subject to streamside-retention buffers and variable-width buffers, as well as unthinned reference stands. We found no treatments effects, however, our results suggest that ground surface conditions (e.g., amount of rocky or fine substrate) play a role in determining the response of riparian and upland amphibians to forest thinning along headwater streams. Distance from stream was associated with amphibian abundance, hence retention of riparian buffers is likely important in maintaining microclimates and microhabitats needed for amphibians and other taxa. Moderate thinning and preservation of conditions in riparian and nearby upland areas by way of variable-width and streamside-retention buffers may be sufficient to maintain suitable habitat and microclimatic conditions vital to amphibian assemblages in managed headwater forests.

1. Introduction

Between 1950 and 1995 approximately 80–90% of old-growth forest stands in Oregon and Washington were harvested (Spies and Franklin, 1988; Smith et al., 1998) resulting in a patchy landscape dominated by early successional forest (Biek et al., 2002; Franklin et al., 2002). In the Coast Range of western Oregon, headwater drainages comprise a large proportion of the forested landscape (Bury, 1988). Hill-slopes and streams in these forested headwater systems are tightly coupled (Gomi et al., 2002), resulting in steep, spatially compressed habitats (Sheridan and Olson, 2003). Because of this, a need exists to better understand how forest management activities affect headwater forest taxa and their habitats in these compacted riparian systems (Bury, 1988; Meyer and Wallace, 2001; Sheridan and Olson, 2003). A variety of approaches to managing forests along headwater streams have been applied ranging from no protection to a one or two site-potential tree height buffer (USDA and USDI, 1994; Sheridan and Olson, 2003; Olson et al., 2007). Although effects of thinning and buffer retention on species and habitats found in headwater forests are likely less than clearcut timber harvest, the efficacy of buffers for protecting headwater riparian and upland species against disturbances experienced during thinning has yet to be quantified (Olson and Rugger, 2007).

In the Pacific Northwest, several amphibian species are associated with forested headwater systems (Sheridan and Olson, 2003; Olson and Weaver, 2007) including stream-associated species, such as tailed frogs (*Ascaphus truei*; Olson et al., 2000, 2007; Bisson et al., 2002; Raphael et al., 2002; Stoddard and Hayes, 2005) and torrent salamanders (*Rhyacotriton* spp.; Olson et al., 2000, 2007; Bisson et al., 2002; Raphael et al., 2002; Russell et al., 2004a; Stoddard and Hayes, 2005), and terrestrial species, such as western red-backed salamanders (*Plethodon vehiculum*; Wilkins and Peterson, 2000; Sheridan and Olson, 2003) and Dunn's

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salamanders (*Plethodon dunni*; Bury et al., 1991; Wilkins and Peterson, 2000; Sheridan and Olson, 2003). Amphibians may play important roles in functions of headwater ecosystems by providing a central link in food webs as both predators and prey (Davic and Welsh, 2004). In particular, amphibians with life history functions potentially requiring both terrestrial and aquatic habitats, such as *A. truei*, *Rhyacotriton* spp., and *Dicamptodon tenebrosus* (Olson et al., 2007), may be pivotal in the exchange of nutrients between streams and uplands in headwater forests (Davic and Welsh, 2004). Disruption of these processes by forest management activities, such as reduction of forest canopy, disturbance of substrates, and altered microclimates, could affect biological processes involving aquatic and terrestrial headwater fauna, in turn resulting in negative impacts on downstream systems (Gomi et al., 2002).

Many plethodontid salamanders are long-lived (e.g., *P. vehiculum* may live up to 10 years) and do not reproduce annually (Ovaska and Davis, 2005). Therefore, effects of silvicultural treatments on some amphibian populations may not be fully realized for many years after timber harvest (Ash, 1988; Petranka et al., 1993). Although there is a lack of long-term data on thinning effects on terrestrial amphibians (Heyer et al., 1994; Perkins and Hunter, 2006), short-term effects of thinning are beginning to emerge. In Virginia, Harpole and Haas (1999) found that salamander relative abundance was significantly lower after partial cutting. Knapp et al. (2003) had similar findings in Virginia (same sites used by Harpole and Haas, 1999) and West Virginia. In western Maine, Perkins and Hunter (2006) found that, although partial harvests along headwater streams had the least affect on amphibians, harvest effects were seen and recommended that riparian buffers may help maintain populations. In southwestern Washington, Grialou et al. (2000) found that although species presence was not affected by thinning, capture rates were reduced. Two years post-thinning, Rundio and Olson (2007) found a negative effect on terrestrial amphibian abundance in response to thinning with riparian buffers at one of two case study sites in western Oregon. In moderately and heavily thinned stands in western Oregon, Suzuki (2000) found short-term (2 years post-thinning) declines in total amphibian captures. During the following year amphibian captures continued to decrease in the heavily thinned stands, but recovered to pre-treatment levels in moderately thinned stands. As an alternative to clearcutting, thinning treatments that remove only part of the canopy and maintain riparian buffer strips may help sustain microclimate (Anderson et al., 2007) and habitat conditions suitable for amphibians (Olson and Rugger, 2007), potentially allowing for a quicker recovery from disturbances experienced during timber harvest (Harpole and Haas, 1999; Ford et al., 2002; Russell et al., 2002, 2004b; Vesely and McComb, 2002; Perkins and Hunter, 2006).

Our study is one of the first to investigate effects of upland forest thinning coupled with riparian buffer treatments on headwater forest amphibians. Our primary objectives for this study were to: (1) examine effects of upland thinning and riparian buffers on terrestrial amphibian abundance and distribution of habitat attributes, accounting for distance from stream; and (2) explore amphibian-habitat associations on managed landscapes. The first objective is particularly relevant as alternative riparian buffer widths are considered for forested headwaters.

We predicted that areas retaining greater canopy cover and experiencing fewer disturbances from thinning operations would result in more favorable microhabitat conditions for terrestrial amphibians (e.g., moss and litter cover, rocky substrates with interstitial spacing, undisturbed downed wood). Therefore, we expected amphibian captures to be greater in the undisturbed areas of our study sites where canopy cover was retained (e.g., wider stream buffers and unthinned areas) compared to narrower buffers and thinned uplands.

2. Methods

Our study area was located in the central Oregon Coast Range within the western hemlock (*Tsuga heterophylla*) vegetation zone, characterized by wet, mild maritime conditions (Franklin and Dyrness, 1988). Three sites were selected from U.S. Bureau of Land Management and U.S. Forest Service lands (Fig. 1). Criteria for site selection included location in the Oregon Coast Range; implementation of thinning and riparian buffer treatments, generally as per the U.S. Bureau of Land Management Density Management Study protocol (Cissel et al., 2006); a minimum of 50 m of upland perpendicular to streams before reaching a ridgeline or entering into the next sub-drainage; and a minimum of 100 m of riparian and upslope area parallel to streams. Two study sites were managed by the Bureau of Land Management (Green Peak, BLM, Salem District; Benton Co., OR; 44°22′00″N, 123°27′30″W, and Ten High, BLM, Eugene District; Lane Co., Benton Co., OR; 44°16′50″N, 123°31′06″W) and one study site was managed by the U.S. Forest Service (Schooner Creek, USFS, Siuslaw National Forest; Lincoln

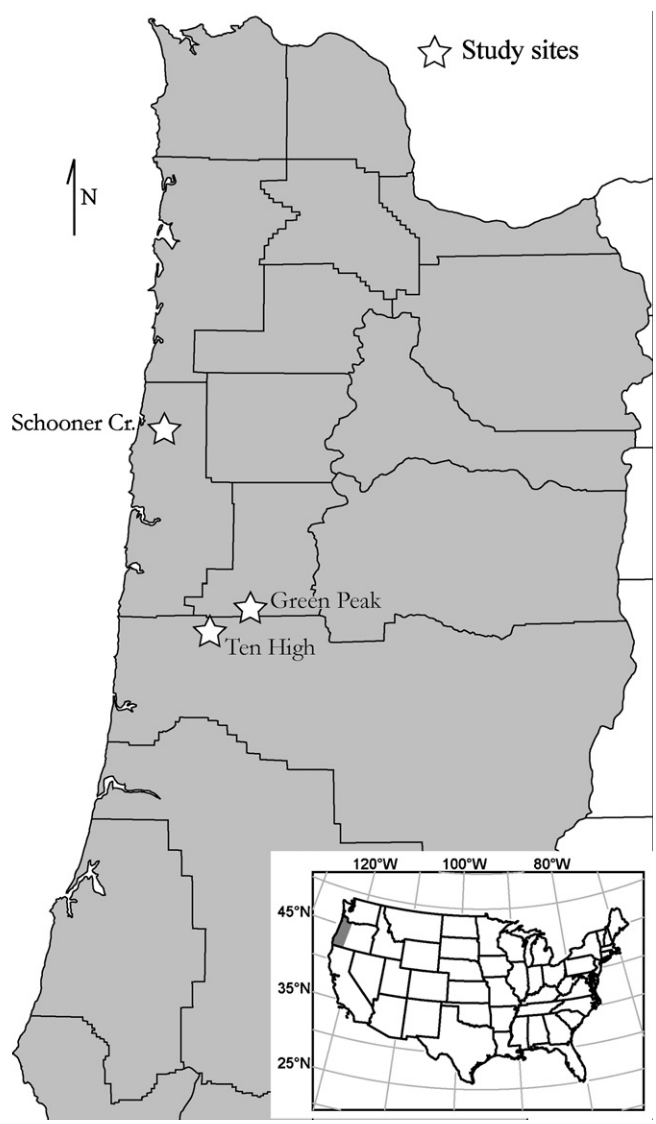

Fig. 1. Location of study sites within western Oregon.

Co., OR; 44°55′45″N, 123°59′18″W). Elevation of the sites ranged from 384 to 870 m.

Each site consisted of two streams with riparian buffer treatments and thinning in the uplands on both sides of the streams, as well as one reference stream with no upslope thinning. The stream reaches were perennial (with the exception of the streamside-retention buffer stream at Ten High, which was intermittent), ranging in width from 0.5 to 1.5 m, and non-fishbearing. Sites ranged from 12–24 ha in size and consisted of previously unthinned 40–60 year old second-growth stands dominated by Douglas-fir (*Pseudotsuga menziesii*), naturally regenerated after clearcut harvests. In 1999 and 2000, thinning occurred at sites as part of a study examining approaches to develop late-successional habitat, such as accelerating development of understory and midstory canopies and increasing spatial heterogeneity of trees and understory vegetation (Cissel et al., 2006). All sites received density management prescriptions, which reduced tree density from 600 trees per hectare (tph) to 200 tph. An unthinned reference stand was retained at each site.

Our study was conducted along streamside-retention and variable-width buffers within thinned treatments (Cissel et al., 2006). The streamside-retention buffers were 6 m wide, and were designed to retain trees along stream banks that likely contributed to bank stability and allowed for overhead shading of streams by their crowns extending over the channel. To ensure a higher degree of stream and riparian shading, as well as litter and wood inputs, the variable-width buffers had a minimum slope distance of 15 m from stream edges on both sides of the stream. Widths were increased for unique riparian vegetation, as well as breaks in slope character such as steep slopes, slumps, and surface seeps. In unthinned reference stands, no harvesting was conducted upslope or adjacent to streams.

During the spring of 2005, amphibian and habitat sampling transects (hereafter bands) were established at all sites, at four distances from each stream, each band extending parallel to streams (Fig. 2). Bands ranged from 100 to 360 m in length, depending on the amount of suitable upland available. Within bands, amphibians and habitat conditions were sampled by randomly placing five, 5 × 10 m sub-sample units for each of the 4 distance categories (resulting in 20 sub-sample units per treatment, 60 per site, 180 total sub-sample units; Fig. 2). Sub-sample units within a band were a minimum of 10 m apart.

Amphibian sampling was limited to one site visit during one sampling season between 4 April and 7 June 2005. Sampling was area-constrained (Olson, 1999) and followed a 1-m-wide zigzag path within sub-sample units (approximately 24 m^2 per sub-sample unit). All moveable cover objects (e.g., rocks, small pieces of wood, moss) were lifted and replaced, any moveable downed wood was turned, decaying logs were dismantled, but not totally destroyed (Olson, 1999), litter was searched, and substrates were

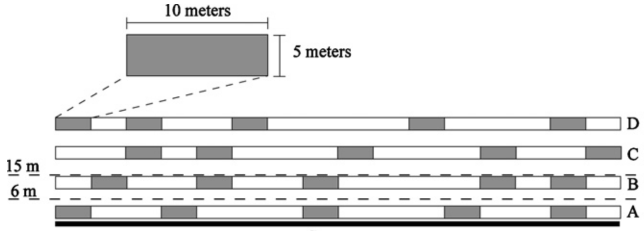

Fig. 2. Schematic diagram of sub-sample units (shaded areas) within four bands (A, B, C, D) aligned along headwater streams in western Oregon. A-band is 0–5 m from stream, B-band is 10–15 m from stream, C-band is 20–25 m from stream, D-band is 30–35 m from stream. Dotted line at 6 m from stream indicates streamside-retention buffer width. Dotted line at 15 m from stream indicates minimum width of variable-width buffer.

Table 1
Habitat variables for which percent cover was collected at our western Oregon study sites

Variable	Description
Fine substrate	Substrate <3 cm diameter
Rocky substrate (coarse substrate)	Substrate >3 cm diameter
Litter and duff	Twigs, dead foliage, branches <10 cm diameter, organic detritus
Shrub cover	Woody plants <3 m in height
Forbs cover	Herbaceous plants (including graminoids)
Moss	Bryophytes
Miscellaneous wood	Chips, chunks, slabs, stumps, loose bark on ground
Downed wood	Downed wood >10 cm diameter and >1 m in length
Midstory cover	Foliage of trees <10 m in height
Canopy cover	Foliage of trees >10 m in height

Visual estimation of percent cover within sub-sample unit was used for all habitat variables except canopy cover, which was measured using a spherical densiometer.

searched to maximum depth of 20 cm using a hand tool. No more than 5 min were spent searching any cover object. Bark and dismantled logs were replaced as best as possible. When amphibians were captured, species was recorded, as well as cover object and substrate association (Bury and Corn, 1988). To estimate amphibian occurrence per distance from stream, amphibian captures in the five sub-sample units within each band were averaged within treatments.

Ten habitat variables were measured or estimated within each sub-sample unit (Table 1). Visual estimates were used to determine percent cover for 9 habitat variables. Percent canopy cover for overstory species was measured at the center of each sub-sample unit using a spherical densiometer (Lemmon, 1956). Percent cover of habitat variables within bands was aggregated by averaging values collected across sub-sample units. All estimates of cover were rounded to the nearest 5%.

We tested for differences in amphibian captures relative to distance from stream and treatment using ANOVA with repeated measures of distance (PROC MIXED) in SAS v. 9.1 statistical software (SAS Institute, 2004). Captures of all amphibian species and captures of the most abundant amphibian species (species with captures >50) were modeled as a randomized complete block (by site) with three treatments (streamside-retention and variable-width buffers, and unthinned reference). Distance from stream was treated as a repeated measure factor with four levels (i.e., bands). The Tukey–Kramer adjustment was applied to accommodate multiple comparisons. A treatment × distance interaction was used to determine if effects of distance from stream was similar among treatments. After viewing residual plots, logarithmic transformations were performed on amphibian captures to meet model assumptions of normality and constant variance. We analyzed whether distance and treatment affected distributions of habitat variables using the same approach. Logarithmic transformations were also performed on habitat variables. Because the number of replications for this study was relatively small ($n = 3$), we considered $P < 0.10$ as statistically significant in all analyses to reduce the chance of committing a type II error (deMaynadier and Hunter, 1995, p. 247; Steidel et al., 1997).

Electivity indices (D) were used to gain insight into how amphibians were using habitat variables at our sites (Afonso and Eterovick, 2007). We used the method of Jacobs (1974):

$$D = \frac{R_i - P_i}{[(R_i + P_i) - (2R_iP_i)]}$$

where R_i = proportion of habitat type 'i' available and where P_i = proportion of habitat 'i' amphibians were associated with at the time of capture. The range of D varies from +1 (indicating

Table 2
Microhabitats and percent available versus amphibian use (number of captures associated with each microhabitat as cover over animal or substrate under animal) at our western Oregon study sites for animals with captures >50

Microhabitat	% Available	Cover				Substrate			
		P. vehiculum		*E. eschscholtzii*		*P. vehiculum*		*E. eschscholtzii*	
		Used	D	Used	D	Used	D	Used	D
Fines	80	9	−0.453	4	−0.499	43	0.314	28	0.526
Rock	20	41	0.849	5	0.339	33	0.731	3	−0.075
Litter	75	8	−0.472	8	−0.117	10	−0.518	7	−0.362
Moss	74	22	0.079	20	0.468	1	−0.943	0	−1.000
Vegetation	24	4	−0.233	1	−0.529	0	−1.000	1	−0.631
Misc. wood	13	7	0.361	9	0.724	10	0.400	9	0.645
Large downed wood	15	13	0.569	5	0.465	8	0.223	4	0.227
Surface visible	100	1	−0.944	0	−1.000	n/a	n/a	n/a	n/a

D = electivity for microhabitat types for *P. vehiculum* and *E. eschscholtzii*. The range of *D* varies from +1 (indicating complete selection or preference for habitat '*i*'), through 0 (indicating that habitat is chosen at random) to −1 (indicating that habitat '*i*' is present but not used). Surface visible is only applicable to the cover options and was used for animals captured in the open (no cover). Percent available exceeds 100% because of the complex, layered nature of microhabitats available to amphibians at our study sites.

complete selection or preference for habitat '*i*'), through 0 (indicating that habitat is chosen at random) to −1 (indicating that habitat '*i*' is present but not used). We calculated D for species with captures >50 (Table 2). Because of the complex, layered nature of microhabitats available to amphibians (e.g., moss on downed wood on fine substrates) we used an additive approach to habitat availability, thus total coverage of habitat variables was allowed to be >100%.

3. Results

3.1. Animal analyses

We captured 225 amphibians of 7 species. Western red-backed salamander (*P. vehiculum*, $n = 105$) and ensatina (*Ensatina eschscholtzii*, $n = 52$) were the most abundant species. We encountered few captures of 5 species: Dunn's salamander (*P. dunni*, $n = 18$); coastal giant salamander (*D. tenebrosus*, $n = 7$); tailed frog (*A. truei*, $n = 6$); rough-skinned newt (*Taricha granulosa*, $n = 4$); and southern torrent salamander (*Rhyacotriton variegatus*, $n = 1$). Additionally, 32 juvenile *Plethodon* spp. (comprised of *P. vehiculum* and *P. dunni*) were captured but were too small (<35 mm SVL) to positively identify to species. The more abundant amphibians were terrestrially associated species, whereas less abundant amphibians were generally aquatic-associated species.

ANOVA revealed an effect on overall amphibian captures with distance from stream ($F = 4.23$; df = 3, 18; $P = 0.019$). However, buffer width did not influence this pattern ($F = 1.83$; df = 2, 4; $P = 0.27$). Amphibian captures were 50% greater in the 0–5 m bands compared to the 20–25 m bands ($P = 0.014$) and 30% greater in the 0–5 m bands compared to the 30–35 m bands ($P = 0.09$). There were no significant differences in captures between the 0–5 and 10–15 m bands ($P = 0.30$). There were no effects of distance × treatment interactions observed in captures ($F = 1.50$; df = 6, 18; $P = 0.23$). Buffer width did not appear to influence captures of *P. vehiculum* ($F = 0.55$; df = 2, 4; $P = 0.61$) or *E. eschscholtzii* ($F = 1.04$; df = 2, 4; $P = 0.43$). However, captures differed with distance for *P. vehiculum* ($F = 2.92$; df = 3, 18; $P = 0.062$). *P. vehiculum* captures were 30% greater in the 0–5 m bands compared to the 10–15 m bands ($P = 0.06$). Distance effects were also observed with *E. eschscholtzii* captures ($F = 5.83$; df = 3, 18; $P = 0.006$). Both the 10–15 and 30–35 m bands had approximately 20% more *E. eschscholtzii* captures bands relative to the 0–5 m band ($P = 0.016$; $P = 0.02$, respectively), whereas there were 22% more *E. eschscholtzii* captures in the 10–15 m band ($P = 0.08$) than in the 20–25 m band. However, no differences in captures were seen between the 0–5 and 20–25 m bands ($P = 0.83$) or between the 20–25 and the 30–35 m bands ($P = 0.10$). No treatment × distance interaction was observed for either *P. vehiculum* ($F = 1.70$; df = 6, 18; $P = 0.17$) or *E. eschscholtzii* ($F = 1.15$; df = 6, 18; $P = 0.37$)

Electivity analyses provided insight into amphibian-habitat relationships for the more common species encountered during our study (Table 2). *P. vehiculum* preferentially used rock for cover and substrate. Large downed wood and miscellaneous wood also had relatively high electivity values as cover for *P. vehiculum*, whereas values for fines and litter were lower (Table 2). *E. eschscholtzii* preferred moss for cover and fines for substrate. However, electivity values for miscellaneous wood (e.g., chips, chunks, slabs) as cover were also high for *E. eschscholtzii* (Table 2).

3.2. Habitat analyses

In our analysis of distance from stream and treatment effects on habitat distribution, ANOVA revealed treatment × distance interactions with percent canopy cover ($F = 11.19$; df = 6, 18; $P < 0.001$). As might be expected, there were no differences in canopy cover between treatments in the 0–5 m bands. However, within the 10–15, 20–25, and 30–35 m bands canopy cover was greater along reference and variable-width buffer streams than along streamside-retention buffer streams (Table 3).

Percent cover of habitat attributes was generally similar among treatments, with the exception of moss ($F = 8.68$; df = 2, 4; $P = 0.03$). ANOVA indicated sub-sample units along variable-width buffer streams contained greater moss coverage than sub-sample units along reference streams (Table 3). Differences in percent cover of downed wood ($F = 8.58$; df = 3, 18; $P = 0.0009$), litter ($F = 5.13$; df = 3, 18; $P = 0.009$), and forbs ($F = 6.77$; df = 3, 18; $P = 0.003$) were observed with distance from stream. Downed wood abundance was greatest in the 0–5 m bands compared to the other three bands, showing that as distance from stream increases, abundance of downed wood decreases (Table 3). Litter coverage in the 0–5 m bands was greater compared to both the 10–15 m bands and the 30–35 m bands (Table 3); no difference in litter coverage was observed between the 0–5 and 20–25 m bands. Coverage of forbs was greater in the 0–5 m bands compared to the 30–35 m bands and greater in the 10–15 m bands than in the 30–35 m bands (Table 3), whereas no differences in coverage were detected between the 0–5, 10–15 and 20–25 m bands.

4. Discussion

We found no significant differences in amphibian captures between thinning and riparian buffer treatments 5–6 years post-

Table 3
Significant differences in least squared means (ANOVA, $P \leq 0.10$) in percent cover of habitat variables at our western Oregon study sites between: (1) buffer treatment and distance from stream (canopy cover); (2) distance from stream (downed wood, litter, forbs); and (3) buffer treatment (moss)

Habitat variable	Contrast (% cover)	Difference	P-value	95% CI	Comments
Canopy cover	Streamside 10–15 m (80) vs. Reference 10–15 m (100)	−20.00	0.049	−39.99, −0.006	20% less canopy cover along streamside-retention buffer stream at 10–15 m
	Streamside 20–25 m (65) vs. Variable 20–25 m (86.67)	−21.67	0.027	−41.66, −1.67	25% less canopy cover along streamside-retention buffer stream at 20–25 m
	Streamside 20–25 m (65) vs. Reference 20–25 m (100)	−35.00	0.0002	−54.99, −15.00	35% less canopy cover along streamside-retention buffer stream at 20–25 m
	Streamside 30–35 m (55) vs. Variable 30–35 m (83.33)	−28.33	0.002	−48.32, −8.34	34% less canopy cover along streamside-retention buffer stream at 30–35 m
	Streamside 30–35 m (55) vs. Variable 30–35 m (98.33)	−43.33	<0.0001	−63.32, −23.34	44% less canopy cover along streamside-retention buffer at 30–35 m
Downed wood	0–5 m (20.56) vs. 10–15 m (12.22)	8.33	0.032	0.59, 16.08	40% more downed wood at 0–5 m
	0–5 m (20.56) vs. 20–25 m (9.44)	11.11	0.004	3.37, 18.86	54% more downed wood at 0–5 m
	0–5 m (20.56) vs. 30–35 m (7.78)	12.78	0.001	5.03, 20.52	62% more downed wood at 0–5 m
Litter	0–5 m (71.67) vs. 10–15 m (80)	−8.33	0.021	−15.60, −1.07	12% less litter at 0–5 m
	0–5 m (71.67) vs. 30–35 m (80.56)	−8.89	0.014	−16.15, −1.62	12% less litter at 0–5 m
Forbs	0–5 m (45) vs. 30–35 m (20)	25.00	0.0003	8.98, 41.02	55% more forbs at 0–5 m
	10–15 m (36.67) vs. 30–35 m (20)	16.67	0.009	0.64, 32.69	45% more forbs at 10–15 m
Moss	Variable (83.75) vs. Reference (55)	28.75	0.036	2.88, 54.61	34% more moss in variable buffer treatment

"Difference" between contrasts in mean percent cover values is shown.

harvest, however, the availability of rocky substrates may have helped ameliorate potential negative effects of harvest. Rundio and Olson (2007) proposed that ground surface conditions may play such a role when they found short-term effects of thinning, 1–2 years post-harvest, on salamanders at only one of their two sites. One of their sites contained a large amount of downed wood, which they suggested ameliorated effects of thinning. In our study, *P. vehiculum* was the most abundant species captured during our study and these salamanders have been shown to have an affinity toward rocky substrates (Dumas, 1956; Keen, 1985; Ovaska and Gregory, 1989; Corn and Bury, 1991), which was supported by our results. It has been shown that substrates in thinned forests can retain cool, moist conditions (Anderson et al., 2007; Kluber, 2007), likely providing suitable refugia for terrestrial amphibians.

The importance of ground surface conditions may lead to factors influencing results of thinning and buffer effects studies on amphibians and may be responsible for the variation in results of other studies. The suite of past responses range from: (1) no reduction in captures (Karraker and Welsh, 2006); (2) short-term decline in captures (Harpole and Haas, 1999; Grialou et al., 2000); (3) increased decline in captures as thinning intensities increased (Suzuki, 2000); (4) little or no effect on amphibian assemblages in upland sites (Perkins and Hunter, 2006); and (5) site-specific patterns driven by available habitat (McKenny et al., 2006; Olson et al., 2006; Rundio and Olson, 2007).

Our results further support the current understanding of Pacific Northwest salamanders, in that *P. vehiculum* numbers vary between riparian and upslope areas (Gomez and Anthony, 1996), but sometimes decrease in numbers at greater distances upslope from stream edges (Vesely and McComb, 2002; Rundio and Olson, 2007). *P. vehiculum* may rely on the cooler temperatures and higher relative humidity as well as physical habitat characteristics (e.g., downed wood) of near-stream environments for physiological and/or ecological functions (Petranka et al., 1993; Dupuis et al., 1995). On the other hand *E. eschscholtzii* are more often detected in upland areas compared to riparian areas (Gomez and Anthony, 1996; Vesely and McComb, 2002; Rundio and Olson, 2007). We suggest that the failure to detect statistically significant differences in amphibian captures between buffer treatments does not detract from the importance of maintaining intact riparian corridors along headwater streams. Over half (60%) of our amphibian captures occurred within 15 m of the stream. As an amphibian conservation strategy, moderate thinning (e.g., 200 tph) coupled with buffers following topographic slope breaks (Anderson et al., 2007) and including vital habitat (e.g., rocky outcrops, moss cover, seeps, downed wood, unique riparian vegetation) may be able to provide suitable protection for amphibian populations in managed headwater forests in western Oregon. However, amphibians associated with more upland areas (such as *E. eschscholtzii* and *Aneides ferreus*) may need additional conservation considerations.

Furthermore, although amphibian-habitat associations in our study were relatively consistent with our current understanding, our results revealed some notable differences, particularly for *E. eschscholtzii*. It has been well documented that *E. eschscholtzii* often are associated with downed wood (Blaustein et al., 1995; Butts and McComb, 2000; Biek et al., 2002). However, percent cover of downed wood at our sites was greater near streams, whereas *E. eschscholtzii* occurrence was greater upslope. There have been variable results regarding *E. eschscholtzii* use of moss as habitat, ranging from moss use when downed wood volume is low (Rundio and Olson, 2007) to negative associations with the presence of moss (Gilbert and Allwine, 1991). We found *E. eschscholtzii* used a variety of habitats ranging from fine substrates, rocky substrates, litter, moss and wood, but were most often captured with moss as cover. It is possible that their apparent flexibility in habitat use may afford them greater resiliency to disturbances such as timber harvest.

Interpretation of our results should consider study limitations, including the small number of study sites used, with only three replications of each buffer treatment. Although our results are directly applicable to these sites only, forest stand conditions at these locations are reflective of managed stands of the Oregon Coast Range. Our findings are also limited by short-term data (one sampling visit in one season, 5–6 years post-thinning) which was the extent of our capability due to limited resources: (1) our 4-person crew conducted continuous intensive sampling throughout the spring wet season; and (2) our destructive sampling approach and other longer term studies at each site restricted the area available to conduct repeated sampling at our sites. We chose to have replication across space (sites) rather than time within a single case study site. Finally, as with many amphibian studies, detectability is a concern (e.g., MacKenzie et al., 2003; Bailey et al., 2004). Our sampling was conducted during the spring rainy season in an attempt to capitalize on the increased surface activity of amphibians during this time (Ovaska and Gregory, 1989; Dupuis et al., 1995). Timing of surveys was further synchronized within study sites to minimize temporal variation in site-level detectability. Nevertheless, a statistical treatment effect may have been masked by either detectability or the relatively low abundances in our sample. It is important to note that the two main species in our sample are the most common terrestrial salamanders in the western Pacific Northwest (Corn and Bury, 1991; Jones et al., 2005). A biodiversity objective of anthropogenic disturbances such as forest management is to maintain the distributions of rare species and reduce the likelihood of downward population trends towards the listing as threatened and endangered (e.g., USDA and USDI, 2008). This objective appears to have been met at our study sites either by implementation of a benign disturbance or via interactions of site conditions that ameliorated disturbance effects. Our common species remained common post-harvest. However, potential adverse treatment effects on uncommon species could not be addressed by this study. *A. truei* and *R. variegatus* are species of concern occurring at our sites, but are stream breeders and were not well represented in our upland sampling.

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