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ABUNDANCE OF WESTERN RED-BACKED SALAMANDERS (*PLETHODON VEHICULUM*) IN THE WASHINGTON COAST RANGE AFTER HEADWATER STREAM-BUFFER MANIPULATION

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ABSTRACT—We evaluated the effect of forest riparian alternative tree buffer designs on Western Red-backed Salamanders (*Plethodon vehiculum*) along headwater stream banks in managed forests of the Washington Coast Range. We used pit trap live removals in early autumn to estimate relative abundances of surface-active salamanders before and after 3 levels of riparian buffer retention (strip, patch, and no-buffer clear-cut) with upland regeneration harvest. The study spanned a pre-treatment year, the harvest year after logging, and 2 post-treatment years (post1, post2). We observed reduced average tree canopies and increased average down-wood cover along streams that received the buffer treatments, especially in the cut portion of patch buffer and no-buffer clear-cut treatments. Compared to pre-treatment, mean salamander relative abundance was lower in no-buffer treatments in the harvest year and in the post2 year, but not the post1 year. Weather differences between years likely partially influenced these results. *Plethodon vehiculum* abundance in the no-buffer clear-cut treatment was lower than in both the control units in post2 and patch buffer treatments during the harvest year and post2 year. Retention of existing down-wood and recruitment of post-treatment down-wood may have ameliorated treatment effects on *P. vehiculum* abundances in patch buffers by maintaining microclimates and microhabitats. In the no-buffer clear-cut treatments, however, there was no similarly-moderating influence (no effect) from down-wood except when mediated by higher rainfall and cooler conditions. It appears that maintaining amounts of dead down-wood in no-buffer clear-cut treatments and the cut areas of patch buffers that averaged amounts 3 to 6 times greater than occurred before buffer creation along these headwater streams may help lessen initial treatment effects on these woodland salamanders.

Key words: alternative forestry, amphibians, Douglas-fir forests, forest buffer, Pacific Northwest, riparian buffer, streamside habitat

Plethodon or woodland salamanders have been advocated for monitoring biodiversity and ecosystem integrity following forest disturbance (Welsh and Droege 2001; Welsh and Hodgson 2013). Salamander responses to small stream buffers that retain trees in the Pacific Northwest, however, show varied results. Western Red-backed Salamanders (*Plethodon vehiculum*) in small stream buffers were found to be sensitive to adjacent clear-cuts <5 y old in the Oregon Coast Range (Vesely and McComb 2002); but in a study of experimentally thinned riparian buffers in Washington, Hawkes and Gregory (2012) found increased abundance of *P. vehiculum* within 2-y post-harvest, and again, a decade later. In western Oregon, thinned buffers of varying widths (but ≥6 m) by headwater streams showed no evidence of treatment effects on *P. vehiculum* (Olson and others 2014).

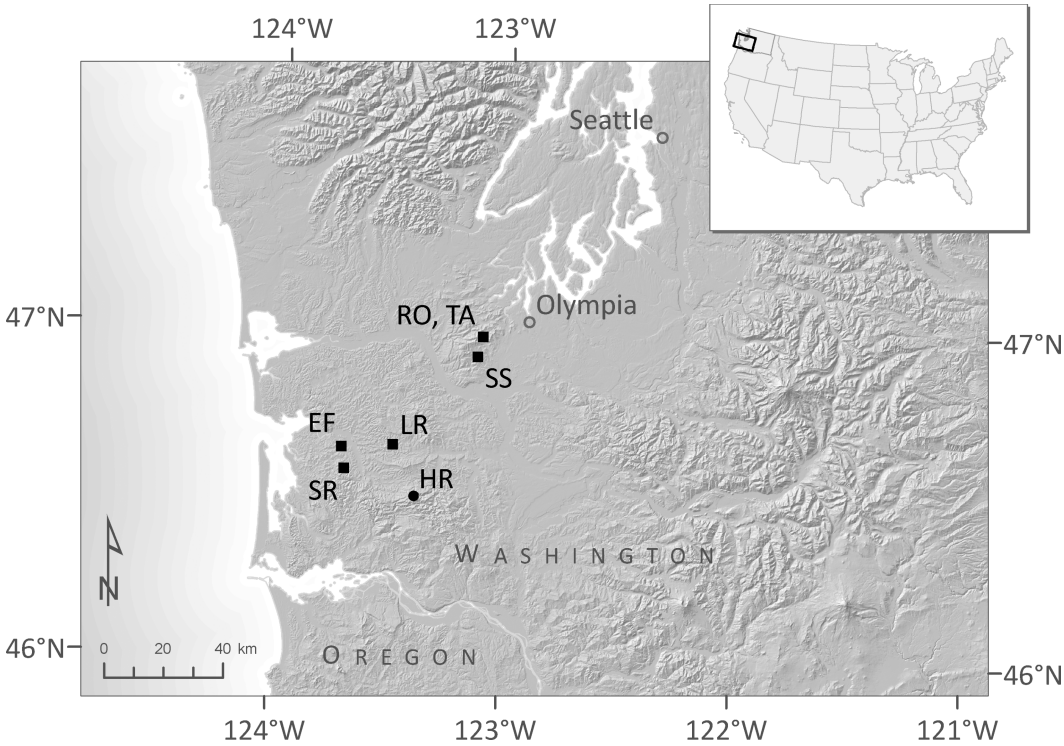

FIGURE 1. Location of 6 riparian buffer study sites and 2 weather reporting stations (Olympia and Huckleberry Ridge [HR]) in western Washington. Site codes: RO = Rotten; TA = Tags; SS = See-saw; LR = Lonely Ridge; EF = Ellsworth Flats; SR = Split Rue.

We assessed the initial response of bank-dwelling *P. vehiculum* to alternative forest buffers along headwater streams in western Washington. Our objective was to determine initial treatment effects of buffer manipulation on *P. vehiculum* abundance and on down-wood as salamander habitat. Our research goal was to increase understanding of the effectiveness of alternative buffering strategies in the conservation and protection of headwater streams during timber harvest.

METHODS

We sampled 6 study sites located across the Willapa and Black Hills, Washington, that ranged from 60 to 400 m in elevation (Fig. 1, Table 1). At all sites, upland forests were naturally-generated 2nd-growth, consisting of Douglas-fir (*Pseudotsuga menziesii*), Western Hemlock (*Tsuga heterophylla*), and Western Red-cedar (*Thuja plicata*); tree ages ranged between 61 and 108 y. We sampled 23 streams with catchments ranging in size between 1.9 and 8.1 ha ($\bar{x}$ = 4.7 m, $s_{\bar{x}}$ = 0.4); length of stream reaches ranged between 111 and 480 m ($\bar{x}$ = 252 m, $s_{\bar{x}}$ = 19); and channel gradient ranged between 11 and 46% ($\bar{x}$ = 29%, $s_{\bar{x}}$ = 2) (Bisson and others 2013). Stream bank-full widths were generally <3 m and flow was mostly perennial, but 1 stream had a well-defined seasonally-dry bed.

We examined 3 riparian buffer designs that were incorporated into the timber harvest prescriptions (Fig. 2, Table 1): (1) continuous strip buffers (n = 7 streams), initially varying in widths between about 12 and 26 m on each side of the stream ($\bar{x}$ = 21 m, $s_{\bar{x}}$ = 2); (2) discontinuous patch buffers, which were <0.6-ha leave tree clumps (patch leave portion) within a clear-cut landscape (patch cut portion) (n = 3 streams), configured to protect headwalls, stream sources, and unstable soils, with widths or diameters similar to strip buffers; and (3) no-buffer clear-cuts (n = 7 streams), that had a few scattered or clumped residual trees (hereafter,

TABLE 1. Study locations (see Figure 1 for code names), riparian buffer assignments, and number of individual stream visits (trap checks) for 23 headwater streams.

	Riparian buffer design (number of streams):			
	Continuous strip buffer	Discontinuous patch buffer	No buffer	Control
LOCATION				
RO	2	0	1	1
TA	1	2	1	1
SS	1	0	1	1
LR	1	1	1	1
EF	1	0	1	1
SR	1	0	2	1
TRAP CHECKS				
Pre-treatment	48	24	48	48
Harvest year	56	24	40	48
Post 1 year	56	24	56	48
Post 2 year	56	24	56	48

we use the term ''buffers'' to refer to the logged streams collectively, including the strip, patch, and no-buffer clear-cut treatments). Streams in uncut forest served as controls (n = 6 streams) (Table 1).

The selection of streams to include in the study and the assignment of harvest treatments were largely guided by forest management operational concerns. They were not random, but were semi-systematic. All control streams were located either adjacent to or in close proximity to the associated treated streams. Site conditions, logistics, and logging operations at individual streams resulted in a wide mix of dimensions and design structure in the strip and patch buffers. Treatments were initiated in September 2003 and ended in early 2005. The study spanned 4 y (2003 to 2006) and comprised: (1) a pre-treatment year (2003); (2) the treatment year (harvest, 2004); (3) post1 (1 y after harvest, 2005); and (4) post2 (2 y after harvest, 2006).

In the Pacific Northwest, pit trapping seems to be a more efficient approach to capturing surface active amphibians than by ground searches or cover boards (Bury and Corn 1988; McDade and Maguire 2005). We trapped *P. vehiculum* in number 10 single-deep cans buried to rim level. We added nearby litter and moss to ''season'' traps. We used inner tapered plastic sleeves to prevent salamanders from escaping, and small sticks to allow small mammals to exit traps (Perkins and Hunter 2002). We also used the sticks to prop open square pressboard can lids for deflecting rain and debris. Openings were 40 to 50 mm above the rims and simulated crevice refugia that might entice salamanders to the traps.

We established 18 trapping stations per side of each stream (n = 36 per stream) for each treatment type and control. Arrays were centered at about midpoint of the entire stream reach between the uppermost point and the downstream confluence. The configuration covered about 85 m of a study reach (Fig. 2). We used single-deep cans with escape sticks because a nearby experiment showed that captures of salamanders were similar between single- and double-deep cans, and with or without sticks (Aubry and Stringer 2000). On each stream side we placed stations about 5 m apart, and alternated their positions between the stream edge and about 2 m away from the bank in a zigzag pattern (Fig. 2).

After logging, trap arrays that were in the discontinuous patch buffers could be in the leave tree portion or in the clear-cut portion of the treatment, or partially in both portions of the patch buffer treatments each year (Fig. 2). In strip buffer treatments, all trap arrays were inside the buffers. Across streams in treatments, logging slash and wind-throw buried some trapping stations or slightly altered portions of individual stream courses. For these, we relocated stations close to their original positions. We checked traps 8 times/stream/year for a 15- or 16-d period. Each check was conducted at 2-d intervals; however, in some years the interval between the 7th and final check at some streams was 1 d due to the logistics of returning salamanders to their capture locations during

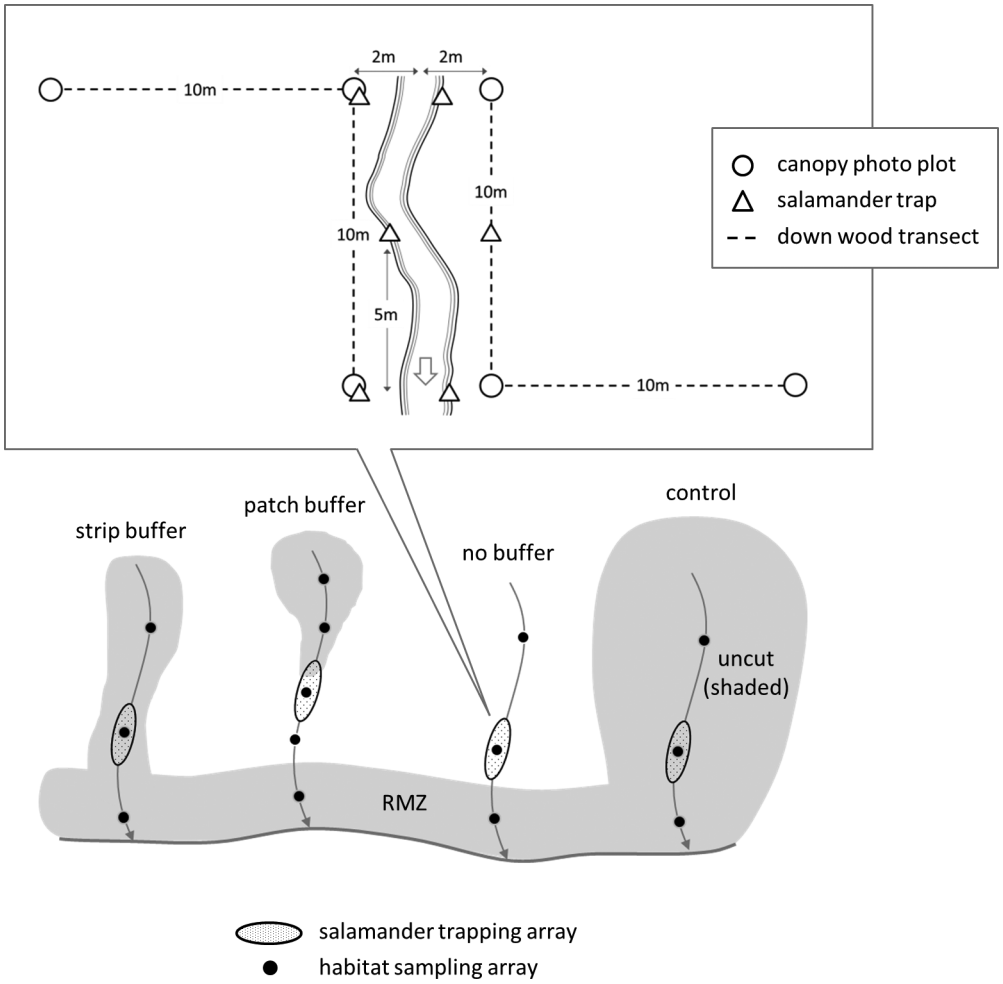

FIGURE 2. Schematic of buffer types with relative locations of salamander and habitat sampling arrays (bottom), and trap station placement, canopy photo points, and down-wood sampling transect layout within habitat sampling arrays (callout). RMZ = riparian management zone buffer.

the final check. In total, we made 687 stream visits with 2-d trap-check intervals, and 17 stream visits with a 1-d interval. We removed all caught amphibians to a refrigerator and returned them back to original capture sites unmarked on the last day of trapping each year. Because of access restrictions, 2 streams partially sampled in pre-treatment (1 strip and 1 no-buffer clear-cut), and 2 streams not trapped in the harvest year (2 no-buffer clear-cuts) were not included in the analysis for those years. All trapping was conducted during the same periods of time each year, ranging between 29 September and 17 October.

To assess habitat effects that may influence *P. vehiculum* abundances, we used pre- and post-treatment measurements of tree canopy cover (%) and down-wood log density (n/ha) (hereafter, down-wood). Wind-thrown trees with green foliage were not tallied as down-wood. These data were collected by random sampling within treatments both inside and outside of trapping arrays, and not directly at established trapping stations (Fig. 2). Canopy point samples and down-wood sampling were conducted in each of two 10- × 10-m plots (= 1 habitat sampling array) on each stream side; down-wood within each array was based upon 20 m of

transect; and riparian management zone buffers were excluded (see Fig. 2). We measured downwood using line-intercept sampling. The diameter of each piece of wood at the point of intersection was ≥10 cm, and the length of each piece was at least 0.3 m (Waddell 2002). We recorded wood species and uniquely marked all dead trees ≥10 cm diameter at breast height. We measured canopy cover using hemispherical photographs from 4 photo points (Fig. 2) that were 1.2 m above ground with a fisheye converter on a digital camera with a 20% mask of the sky (Frazer and others 1999).

We assessed weather conditions for each trapping session for each year. We used average mean daily temperature (average of high and low temperatures over a 24-h period) and 24-h rainfall from data recorded at 2 nearby weather stations, Huckleberry Ridge and the Olympia Airport (Fig. 1).

We made between-year treatment comparisons using paired sample *t*-tests for *P. vehiculum* abundance and habitat variables, and assumption-free Wilcoxon signed-rank related-sample tests for both weather variables. For within-year, between treatment comparisons, we used independent sample *t*-tests for *P. vehiculum* abundance and habitat variables, and Wilcoxon rank-sum tests for weather variables. Paired-sample *t*-tests focus on the differences between compared variables, whereas independent sample *t*-tests focus on the amounts. We computed the relative effect size *r*-statistic for all habitat comparisons, and for statistically significant outcomes of *P. vehiculum* abundance using the benchmarks of Cohen: 0.1 (small effect, explains 1% of the total variance); 0.3 (medium effect, 9% of variance); and 0.5 (large effect, 25% of variance) (Nakagawa and Cuthill 2007; Field 2013). We used $\alpha = 0.1$ for statistical significance because of our small sample size and high variability in the data. In order to compare salamander relative abundances across unequal numbers of buffer treatments, we took the total number of salamander captures at each stream as an average within each treatment. These were salamander captures/stream. To statistically analyze these data, we first normalized individual stream capture data before analysis by taking the $\log_e(y + 1)$, where y = total capture count at each stream (McCune and Grace 2002), and then conducted statistical tests on these transformed data. Canopy cover (%) was square root-transformed. We assessed the variability of the relative abundance of salamanders (transformed) with coefficients of variation ($CV_{\times 100}$). We used IBM SPSS software 21 (2012 version) for data processing and analysis.

Results

Plethodon vehiculum (n = 104) comprised >60% of all salamanders (n = 164) and amphibians (total n = 172) captured. We also captured the Rough-skinned Newt (*Taricha granulosa*), Ensatina (*Ensatina eschscholtzii*), Northwestern Salamander (*Ambystoma gracile*), Coastal Tailed Frog (*Ascaphus truei*), and, in the Willapa Hills only, the Columbia Torrent (*Rhyacotriton kezeri*), Van Dyke's (*Plethodon vandykei*), Dunn's (*P. dunni*) and Cope's Giant (*Dicamptodon copei*) salamanders. The species composition seemed representative for the area.

We observed the following significant differences in *P. vehiculum* relative abundances in the treatment buffers. (1) No-buffer clear-cuts, between years: (a) the mean difference in *P. vehiculum* relative abundance was significantly lower in the harvest year (P = 0.072), and in the post2 year (P = 0.051), compared to the pre-treatment year (Table 2); (b) the mean difference in abundance was significantly lower in the post2 year, compared to the post1 year (P = 0.047); and, (c) the mean differences in *P. vehiculum* abundance in the harvest year was lower than in the post1 year (no statistic) (Table 2, but see Fig. 3). (2) No-buffer clear-cuts, within year and between treatments: *P. vehiculum* mean relative abundance in the post2 year was significantly smaller compared to the patch buffer treatments (P = 0.020), and compared to control streams (P = 0.003) (Table 2, Fig. 3). (3) Strip buffer treatments, between years: the mean difference in *P. vehiculum* relative abundance was significantly lower in both the pre-treatment year (P = 0.086) and the harvest year (P = 0.060) than in the post1 year (Table 2). All effect sizes were large ($r > 0.5$) (Table 2).

In the pre-treatment year, *P. vehiculum* abundances varied 20.0% across all 4 treatment groupings. Across the 3 post-harvest years, we observed the lowest variation in relative abundance in controls ($CV_{\times 100}$ = 28.9%), followed by

TABLE 2. Results of analyses on captures/stream ($\bar{x}$ $\log_e$ [y +1]) of *Plethodon vehiculum*. * = statistically significant; *r*-value shown only if $P \leq 0.1$; sequence of table values = *t*-value (df), *P*-value, *r*-value.

	Between treatments:		
	Strip buffer	Patch buffer	No-buffer
CONTROL			
Pre-treatment	0.72 (10), 0.49	0.78 (7), 0.46	0.28 (10), 0.79
Harvest year	0.21 (11), 0.83	−0.95 (7), 0.38	1.28 (9), 0.23
Post1 year	−0.92 (11), 0.38	0.17 (7), 0.87	−0.57 (11), 0.58
Post2 year	1.31 (11), 0.22	0.24 (7), 0.82	3.80 (11), 0.003, 0.75*
STRIP BUFFER			
Pre-treatment		−0.10 (7), 0.92	0.52 (8.8) 0.62
Harvest year		1.45 (8), 0.19	−1.35 (10), 0.21
Post1 year		−0.91 (8), 0.39	0.60 (9.3), 0.56
Post2 year		0.83 (8), 0.43	−1.28 (12), 0.22
PATCH BUFFER			
Pre-treatment			0.64 (7), 0.54
Harvest year			−3.94 (6), 0.008, 0.85*
Post1 year			0.84 (8), 0.43
Post2 year			−2.90 (8), 0.020, 0.71*
	Between years:		
	Harvest year	Post1 year	Post2 year
PRE-TREATMENT			
Control	1.05 (5), 0.34	0.86 (5), 0.43	0.18 (5), 0.87
Strip buffer	1.09 (5), 0.32	−2.13 (5), 0.086, 0.69*	0.15 (5), 0.88
Patch buffer	−0.84 (2), 0.49	1.00 (2), 0.42	−0.70 (2), 0.56
No-buffer	2.72 (3), 0.072, 0.84*	0.68 (5), 0.53	2.56 (5), 0.051, 0.75*
HARVEST YEAR			
Control		0.10 (5), 0.92	−1.26 (5), 0.26
Strip buffer		−2.32 (6), 0.060, 0.69*	−0.14 (6), 0.90
Patch buffer		1.56 (2), 0.26	1.00 (2), 0.42
No-buffer		—[a]	0.00 (4), 1.00
POST1 YEAR			
Control			−1.18 (5), 0.29
Strip buffer			1.66 (6), 0.15
Patch buffer			−1.82 (2), 0.21
No-buffer			2.50 (6), 0.047, 0.71*

[a] *t*-value could not be computed because the $s_{\bar{x}}$ of the difference was 0.

strip buffers ($CV_{\times 100}$ = 33.8%) and patch buffers ($CV_{\times 100}$ = 34.5%), with more year-to-year fluctuation in the no-buffer clear-cuts ($CV_{\times 100}$ = 88.7%) (Fig. 3).

We found canopy closure (%) significantly reduced in harvested treatments; whereas, down-wood increased in all categories compared to the pre-treatment year and all effect sizes were large ($r > 0.5$) (Table 2). Besides logging slash accumulations inherent in harvest operations, tree blow-down contributed to increased loadings of down-wood and untallied live down-wood. Average amounts of down-wood in the uncut areas of strip buffers, patch buffers, and controls were 1.5 to 1.9 times greater after logging compared to the pre-treatment year. The averaged amounts of down-wood in cut areas along streams (patch-cut portion and no-buffer clear-cut treatments) were 2.6 to 5.7 times greater than in the pre-treatment year (Table 3).

Ambient temperatures in the harvest year ($\bar{x}$ = 12.1°C, $s_{\bar{x}}$ = 0.4) were warmer than in the post1 ($\bar{x}$ = 10.5°C, $s_{\bar{x}}$ = 0.5) (T_{16} = 14, P = 0.016) and post2 years ($\bar{x}$ = 10.9°C, $s_{\bar{x}}$ = 0.4) (T_{16} = 26, P = 0.030). Pre-treatment year temperatures were similar to other years ($\bar{x}$ = 11.9°C, $s_{\bar{x}}$ = 0.8). Rainfall in the pre-treatment year ($\bar{x}$ = 5.3 mm, $s_{\bar{x}}$ = 1.9) was greater than in the post2 year ($\bar{x}$ = 2.1 mm, $s_{\bar{x}}$ = 1.3) (T_{17} = 23, P =

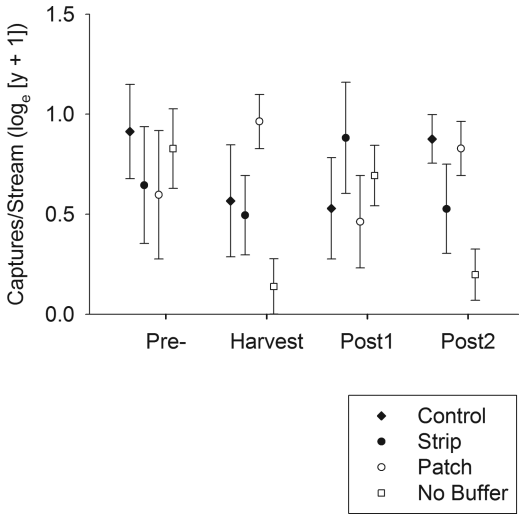

FIGURE 3. Relative abundance of *Plethodon vehiculum* ($\bar{x}$ $\log_e$ [y + 1] ± $s_{\bar{x}}$) in 85-m stream reach trapping arrays in forest headwater stream buffer treatments in the Washington Coast Range in early autumn. Pre- to post2 = 2003 to 2006.

0.064). Rainfall in the post1 year ($\bar{x}$ = 5.5 mm, $s_{\bar{x}}$ = 1.4) was also greater than in the post2 year (T_{17} = 103.5, P = 0.066). Harvest year rainfall was similar to other years ($\bar{x}$ = 4.2 mm, $s_{\bar{x}}$ = 2.4). Rainfall of ≥0.01 mm occurred on 43% of the days in the harvest year, and 41% of the days in the post2 year. In the pre-treatment and post1 years, rain occurred on 72 and 84% of the days, respectively.

DISCUSSION

Within the first 3 y of harvest we observed lower abundances of *P. vehiculum* only in the clear-cut no-buffers. It thus appears that some level of protection was provided by strip and patch buffers. Dupuis and others (1995) and Dupuis and Bunnell (1999) found few *P. vehiculum* in clear-cuts, and Patrick and others (2006) cite several studies with low amphibian abundances in clear-cuts. Clear-cutting and associated reductions of down-wood are believed to be part of the reason for local reductions or extirpations of 7 salamander species in the Pacific Northwest (Bunnell and Houde 2010). Salamander species responses may vary with site conditions retained after harvest, such as canopy cover and down-wood that help maintain important microhabitats and microclimates (see Bunnell and Houde 2010). Landform (for example see Wilkins and Peterson 2000), possibly associated with stream order, may also influence abundances. *Plethodon vehiculum* abundances in no-buffer clear-cut treatments were lower than their abundances in patch treatments in the harvest year and in the post2 year, and were also lower than in the controls in the final year, but no differences in abundances occurred in the post1 year (Fig. 3). Weather conditions, particularly during the transitional period to the rainy season, were likely an important influence on salamander surface activity. When harvests reduce canopy cover, rainfall becomes critical to mediate ambient moisture loss. The shift from no effect in post1 year no-buffer clear-cut treatments to lower abundances in the harvest year and post2 year likely reflected the influence of ''more suitable'' weather conditions in the post1 year trapping session. Higher relative moisture and relatively cooler conditions may have lessened the potential detrimental treatment effects in the clear-cut treatments during

TABLE 3. Measurements of tree canopy and dead down-wood logs ($\bar{x}$ ± $s_{\bar{x}}$) before and after treatment harvests. Statistical results for canopy cover are from analyses on transformed data. Statistically significant if $P \leq 0.1$; $r > 0.5$ = large effect size.

	Control	Strip buffer	Patch leave	Patch cut	No-buffer
Canopy cover (%)					
Pre-treatment	77 ± 1	77 ± 2	78 ± 1	74 ± 0.3	78 ± 1
Post-treatment	77 ± 0.3	66 ± 4	56 ± 7	7 ± 4	9 ± 2
t (df)	−0.15 (5)	3.20 (6)	2.87 (2)	6.31 (2)	12.81 (6)
P, r	0.89, 0.07	0.03, 0.79	0.10, 0.87	0.02, 0.98	<0.001, 0.99
Dead down-wood (n/ha)					
Pre-treatment	731 ± 167	560 ± 206	824 ± 119	466 ± 46	708 ± 222
Post-treatment	1132 ± 333	1065 ± 211	1435 ± 241	2614 ± 515	1822 ± 640
t (df)	−1.85 (5)	−7.60 (6)	−3.87 (2)	−4.06 (2)	−2.41 (6)
P, r	0.12, 0.64	<0.001, 0.96	0.06, 0.94	0.06, 0.80	0.05, 0.70

that period. Lower *P. vehiculum* abundances in the no-buffer clear-cut treatments in the harvest year, particularly, and in the post2 year occurred during drier than average conditions and less frequent rain days. In clear-cuts, rainfall may affect salamander activity more than in mature forests because the infusion of ambient moisture facilitates salamander movement and foraging (Fraser 1976; Johnston and Frid 2002; Perkins and Hunter 2006).

In the Pacific Northwest, it has been suggested that the maintenance of microclimate is key to buffer effectiveness for *P. vehiculum* (Kluber and others 2009; Hawkes and Gregory 2012; Olson and others 2014). Tree canopy cover helps to maintain forest floor moisture and microclimates in riparian forests (Brosofske and others 1997; Anderson and others 2007; Rykken and others 2007). Microclimates in clear-cuts are warmer, drier, and more variable than in mature unmanaged forests due to the unimpeded penetration of solar radiation and wind to the forest floor (Chen and others 1999; Rykken and others 2007). In the absence of frequent rainfall, these conditions result in a drying of upper soil and litter layers forcing *Plethodon* salamanders to retreat to moist refugia to avoid dehydration (Jaeger 1978, 1980; Feder 1983). As a result of harvest, the period of suitable ambient conditions for surface activity is shortened and the capacity of the forest to support salamanders is decreased (Welsh and Droege 2001). In our study, the low surface abundance of salamanders in the no-buffer clear-cut treatment may reflect the displacement of individuals by harvest effects that restrict them to cooler and moister microhabitats and time periods (see Peterman and others 2011).

Studies from the Pacific Northwest demonstrate the importance of canopy closure and decayed logs to salamander persistence (Dupuis and others 1995; Grialou and others 2000; Vesely and McComb 2002). Recruitment of down-wood with partial canopy retention may also provide viable habitat for plethodontid salamanders (Kluber and others 2009). Thus, the anticipated negative effects of logging on *P. vehiculum* may be lessened by retaining pre-harvest down-wood, with new down-wood, as a result of harvest operations and wind-throw. Down-wood helps ameliorate the effects of forest thinning along headwater streams because it helps retain cool, moist conditions, which provide refugia to terrestrial amphibians (Rundio and Olson 2007). Logs of wide size range, as well as soils also protect plethodontid salamanders against thermal extremes reached by air temperatures in thinned stands with limited overstory (Kluber and others 2008, 2009). In our post-treatment patch buffers there were large reductions of canopy closure in patch-cuts with concurrent large additions of down-wood logs (Table 3). Together, with lesser canopy reduction and increases in down-wood in the uncut patches, there was no overall effect on *P. vehiculum* abundances in patch buffers, which seems consistent with the above-cited studies. In no-buffer clear-cuts, there was no consistent similarly-moderating influence (no effect) on abundances from retention of large amounts of down-wood except when mediated by higher rainfall and cooler conditions. Therefore, it appears that maintaining amounts of dead down-wood in no-buffer clear-cut treatments and in the cut areas of patch buffers that averaged 3 to 6 times greater than occurred before buffer creation along these headwater streams (Table 3) may help lessen initial treatment effects on these woodland salamanders.

The *CVs* of *P. vehiculum* abundance may reflect a continuity of abundance by buffer type, even though wide variation occurred in some individual treatments. The low variation in post-harvest strip buffers may reflect the positive effectiveness of this treatment across a wide range of buffer width dimensions. Initial post-treatment conditions in strip buffers varied from relatively low shading in narrower widths to relatively dense shading in the widest buffers. Canopy cover in strip buffers after treatment ranged between 68 and 100% of pre-treatment cover ($\bar{x}$ = 85%, $s_{\bar{x}}$ = 4). Tree wind-throw caused by windstorms also altered strip and patch buffers causing dynamic inter-year structural changes. These changes likely contributed to the high within-year variation in *P. vehiculum* abundances across individual strip buffers (error bars, Fig. 3). In a long term study of amphibians in Oregon, Olson and others (2014) surmised that mixed widths of thinned headwater stream buffers may be beneficial from a management perspective to hedge the uncertainties of ecological risks to riparian vertebrates. Our data tacitly support this idea

because *P. vehiculum* abundances in the strip and patch buffers were similar across years and similar to controls, except for marginal differences in strip buffers between the pre-treatment and the post 1 years (Table 2). The discontinuous patch buffers may be even more dimensionally and structurally diverse than the strip buffers, manifesting a wide range of site conditions with high tree-fall. The tree-clumped patches in the clear-cut landscape were maintained to protect fragile sites, often having seeps. Wet soils create diverse microhabitats and are more preferred by some salamanders than adjacent streamside habitats (Grover and Wilbur 2002), and inherently attract a diverse ground-dwelling fauna, while being excluded from harvest. The patch buffers, therefore, may also provide a hedge to ecological uncertainties and risks to riparian vertebrates as surmised for the strip buffers.

Although we observed differential effects of alternative buffer designs on abundances of *P. vehiculum*, the consistent implementation of our experimental approach among sites was challenging (see Bisson and others 2013). This may limit the scope of inference of our findings because: (1) treatments were not assigned randomly and the design was imbalanced owing to operational constraints; (2) the implementation of buffers among years introduced potential time effects among sites (such an effect may also confound our single pre-treatment year of data collection); and (3) large variation in the ultimate dimensional design of strip and patch buffers may have obscured some differences among treatments (Wilk and others 2010; Raphael and Wilk 2013). Log cover along small streams following harvest is influenced by amount and pattern of tree-fall (Grizzel and Wolff 1998). In our experience, there was also a strong element of observer bias in the down-wood data in sampling plots due to the presence of live wind-thrown trees (which were not tallied as down-wood) and logging-induced debris; each added to the variability in down-wood measurements. Overall, forest-floor structure was therefore underestimated for actual down-wood (dead and alive) due to unrecorded live logs. Despite these challenges, our results seem to support the results of other studies and provide a basis for comparisons that seem biologically meaningful (see Johnson 1999), but may not necessarily provide evidence for extrapolating conclusions to areas beyond the scope of this study. Our findings do, however, supplement and highlight the need for additional data to help inform and support headwater stream riparian habitat conservation planning in the Washington Coast Range.

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