Preliminary Study of the Effects of Headwater Riparian Reserves with Upslope Thinning on Stream Habitats and Amphibians in Western Oregon

Deanna H. Olson and Cynthia Rugger

Abstract: We conducted a preliminary examination of the responses of stream amphibians and instream habitat conditions to alternative riparian buffer zones with forest thinning upslope. Pre and posttreatment surveys were carried out on 68 headwater stream reaches (including 23 unthinned reference reaches) at 11 sites in western Oregon. Streams were in managed conifer stands, 40 to 80 years old, where the thinning treatment reduced stands from 600 trees per hectare (tph) to 200 tph. Treatments consisted of four widths of riparian buffers approximately 6, 15, 70, and 145 m on each side of streams. Over three study years, 3,131 individuals of 13 species were detected. For the more common instream and bank species analyzed, capture rates persisted posttreatment with no negative treatment effect from thinning with any of the buffer widths. More animals were detected after thinning in treatment reaches compared to reference reaches for rough-skinned newts (*Taricha granulosa* Skilton) occurring on stream banks, and for instream coastal giant salamanders (*Dicamptodon tenebrosus* Baird and Girard). Treatment effects on instream habitat parameters were not detected. Interannual variation was evident for western red-backed salamanders (*Plethodon vehiculum* Cooper), and several habitat conditions including pool-riffle ratio, stream spatial intermittency, stream width, and down wood. Overall, riparian buffers with moderate upslope thinning (200 tph) seemed to have retained the aquatic vertebrate community along channels among sites in the first 2 years posttreatment; however, several limitations of the study reduce the inference of the findings, and these preliminary results are best interpreted as hypotheses for further investigation. FOR. SCI. 53(2):331–342.

Keywords: forest density management, headwater streams, riparian buffer zone, salamanders, stream bank

RIPARIAN PROTECTION BUFFERS generally have been applied along streams to retain ecological values including species, habitats, and functions. In forests, these buffers are strips of streamside and adjacent upslope vegetation with restricted or no timber management. In the moist coniferous forest landscape of the Pacific Northwest of North America, the extent of streamside riparian buffers is dependent on the presence of fish, stream size, and landownership (e.g., Olson et al. 2007). At one extreme, buffers measuring up to two site-potential tree heights along each side of streams have been the default widths for perennial streams with fish on US federal lands, with these buffers being potentially narrowed or managed within following watershed analysis of local conditions (USDA and USDI 1994a, b). This width takes into consideration several ecological functions and processes of adjacent riparian areas relative to both instream fish habitat conditions, including stream shading, microclimate retention, and down wood and litter inputs (USDA and USDI 1994b, Naiman et al. 2000), and upland nonaquatic objectives such as improving dispersal corridors for late-successional or old-growth forest-associated terrestrial species of concern (USDA and USDI 1994b, 1996). In headwaters, riparian management policies for stream reaches without fish have had variable direction, ranging from no specific protection to default buffers of one site-potential tree height that can be refined or managed within once watershed analysis is completed (see Sheridan and Olson 2003). Variation in headwater riparian management approaches likely reflects uncertainty regarding the ecological values warranting protection in these uppermost stream reaches, and differing priorities relative to the protection of known values.

Species associated with headwater forests are emerging as a biotic value of significant concern relative to riparian management approaches. Headwater fauna may have an important role in structuring forest communities. Wipfli and Gregovich (2002) found that headwater arthropods are prey contributing significantly to downstream fish production. Progar and Moldenke (2002) found fewer arthropod species but greater abundances in ephemeral headwaters of western Oregon compared to downstream perennial reaches, and suggested these communities are strongly predation-structured by resident amphibians. Welsh et al. (2005) also speculated that animals in headmost stream reaches of

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northwestern California were responding to predators, in particular escaping fish and larger salamander predators in downstream perennial reaches. These may be well-founded speculations given over a third of the native amphibians of the Pacific Northwest region potentially reside in forested headwaters (Jones et al. 2005). Several regional taxa appear to have particular associations with these uppermost stream and riparian zones (e.g., *Rhyacotriton* species [torrent salamanders] and *Plethodon dunni* Bishop [Dunn's salamander]: Sheridan and Olson 2003, Olson and Weaver 2007; *Dicamptodon tenebrosus* Baird and Girard [coastal giant salamander] and *Aneides flavipunctatus* Strauch [black salamander] in California: Welsh et al. 2005). It is likely that headwater-occurring amphibians play important ecological roles in these headwater forests (i.e., functional roles as those described by Marcot and Vander Heyden 2001; stream amphibians as environmental indicators: Davic and Welsh 2004). In particular, being both terrestrial and aquatic, their role in providing reciprocal subsidies among channels, riparian, and upslope habitats may be unique, analogous to those described for invertebrates by Baxter et al. (2005).

Only a few studies have addressed the efficacy of streamside buffers in retaining headwater habitats and fauna in managed Pacific Northwest forests. Historically, few designated buffers have been left along headwater streams, restricting opportunities for retrospective studies (Richardson et al. 2005). However, studies are now emerging that have examined the effects of retained streamside vegetation of different buffer widths with upslope clearcut logging, and buffers appear to benefit many stream-dependent amphibian species. In western Oregon, Vesely (1996) found similar abundances of terrestrially occurring torrent salamanders, ensatina (*Ensatina eschscholtzii* Gray), Dunn's salamanders, and western red-backed salamanders (*Plethodon vehiculum* Cooper) between unmanaged forest and riparian buffer strips along first- to third-order streams; yet he found amphibian species richness was lower in buffers than unmanaged forests, and richness was correlated with buffer strip width. In western Washington, Kelsey (1995) found more Pacific giant salamanders (*Dicamptodon* species) and coastal tailed frogs (*Ascaphus truei* Stejneger) in buffered streams compared to those without riparian buffers. Dupuis and Steventon (1999) found larval tailed frog densities were lower in streams logged without buffers than streams with buffers (5–60 m wide) or old-growth forest. Johnston and Frid (2002) found giant salamander movements were similar in riparian buffer strips compared to forested sites, and movements were reduced in clearcut sites without stream buffers; hence, buffers appeared to mitigate effects on near-stream activities in this species. Raphael et al. (2002, see also Bisson et al. 2002) found Van Dyke's salamander (*Plethodon vandykei* Van Denburgh) only at old-growth forest sites and managed sites with stream buffers, and at no sites that had been previously logged without a buffer. Stoddard and Hayes (2005) found wider buffers increased the odds of finding stream amphibians (i.e., *Dicamptodon, Ascaphus, Rhyacotriton*), and recommended >46 m-wide forested bands along streams with high-quality habitats where they are a conservation priority.

Alternative silvicultural approaches such as forest thinning and mosaic designs of joint tree cutting and retention within stands are gaining broader use in the federal forests of the Pacific Northwest. The effects on headwater species and habitats of such forest density management are likely less than clearcut timber harvests, but this is not yet quantified. In upslope forest, both negative effects (e.g., Harpole and Hass 1999, Grialou et al. 2000, Knapp et al. 2003, Morneault et al. 2004) and variable effects (positive, negative and/or no effects: Suzuki 2000, Bartman et al. 2001, MacCracken 2005, Rundio and Olson 2007) of forest thinning have been reported relative to terrestrial salamander (i.e., family Plethodontidae) abundances. Such mixed effects may be expected relative to amphibians occurring along or within streams within an upslope matrix of thinned forest, depending on whether key species' habitats are affected.

Our main study objective was to conduct a preliminary examination of the effects of upslope density management with streamside buffers of alternative widths on instream and bank-associated amphibians in managed forest headwaters of western Oregon. We conducted a before-after-control study of four riparian buffer widths (approximately 6, 15, 70, and 145 m, Olson et al. 2002, Cissel et al. 2006). These were established as stream geometries allowed along 45 treatment reaches with 23 unthinned reference reaches at 11 study sites. Amphibian species detected in sufficient numbers were analyzed for treatment effects within the first 2 years posttreatment. We also examined treatment effects on a subset of habitat attributes, and examined interannual variation in both animal detections and stream habitat attributes.

Our predicted treatment effects vary with species, their life history and use of stream or upland habitats, and likely effects on their diverse habitat conditions by the treatments. Importantly, if the combined buffer and thinning treatments are benign relative to stream and stream bank amphibians and their critical habitats, we expect no effects. In particular, if instream or bank habitats are not altered, torrent and Dunn's salamanders are likely to be unaffected because they may be restricted to stream and near-stream areas (Jones et al. 2005, Olson et al. 2007). In contrast, we predict reduced species captures posttreatment if critical microhabitat conditions are altered, such as an altered stream hydrology that reduces streamflow. Instream species potentially affected include coastal giant salamanders and tailed frogs; these species potentially disperse into the uplands and may be directly affected by the thinning (Jones et al. 2005, Olson et al. 2007).

Methods

Our riparian buffer study was conducted in managed forest stands of the *Tsuga heterophylla* [Raf.] Sarg. (western hemlock) vegetation zone (Franklin and Dyrness 1988) of western Oregon (Figure 1, Table 1). Eight sites were part of the Bureau of Land Management (BLM) Density Management and Riparian Buffer Study (Cissel et al. 2004, 2006), and three sites were administered by the USDA Forest Service, Siuslaw National Forest (Table 1). The overarching

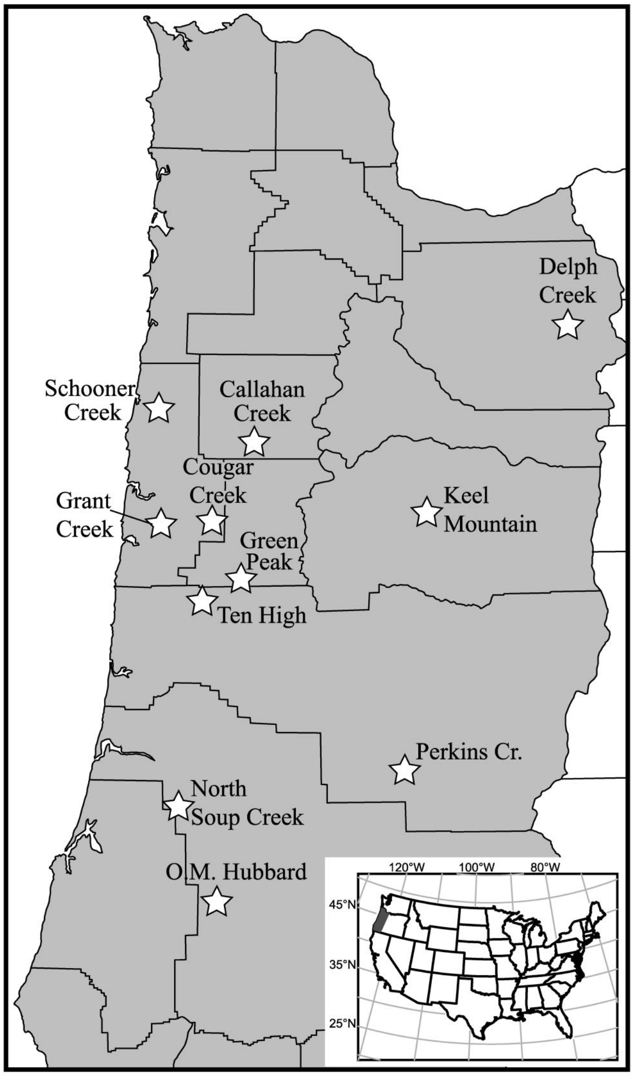

Figure 1. Locations of our 11 riparian buffer study sites in western Oregon, USA.

study objective for the BLM Density Management Study was to examine alternative silvicultural approaches to accelerate the development of late-successional characteristics in managed forests. For this purpose, study sites with forest structure representative of low-elevation forests on US federal lands in western Oregon were selected on four BLM districts (Coos Bay, Roseburg, Eugene, Salem), providing a geographic spread of locations (Cissel et al. 2004, 2006). Similar criteria were used to select sites on USDA Forest Service lands within this geographic range. Hence, due to nonrandom site selection our study has inference only to these sites and not the broader area, although findings may have implications to similar forests in the western Pacific Northwest forest landscape.

Forest stands at sites were of two general age classes, 30 to 50 years (nine sites) and 70 to 80 years (two sites, Callahan Creek and Perkins Creek). Sites had been previously clearcut without stream buffers, and naturally regenerated to 430 to 600 trees per hectare (tph). One of the older sites, Perkins Creek, was thinned 20 years earlier, at about age 50 years, to 250 tph. A more comprehensive description of sites and treatments can be found in Cissel et al. (2006); sites are also described further in Anderson et al. (2007), Olson and Weaver (2007), and Rundio and Olson (2007).

Density management timber harvest occurred at all 11 sites, during which stream reaches within treatments were protected by one of four riparian buffer widths. Per site, the harvest usually occurred within a 2-year time window, and these were staggered among sites, 1997 to 2000. Thinning reduced overstory tree density to about 200 tph at all but one older site; Perkins Creek was thinned to 100 to 150 tph. At the six younger BLM sites, the thinning was a mosaic of "matrix" thinned to this level, and circular clearcut gaps and leave islands of three sizes (0.1, 0.2, and 0.4 ha). Callahan Creek and the Forest Service sites were thinned to 200 tph without gaps and islands. An unthinned reference treatment was designated at all sites.

Forty-five first- and second-order stream reaches within the thinning treatment had designated riparian buffers measuring 6, 15, 70, and 145 m on each side of streams (Table 1). The narrowest buffer, "streamside retention," was designed to retain all streamside trees with branches extending over the stream bank, and was an indicator of root distribution. It was hoped that this narrowest buffer would maintain stream bank stability. For the "variable-width" buffer, 15 m was the minimum width on each side of the stream and it was enlarged in areas to conform to local topography (e.g., slope breaks). Widening also occurred in steep areas to mitigate landslide potential, or in areas with unique vegetation (e.g., a wolf tree) or site conditions (e.g., seepy headwalls). The 70 and 145 m buffers matched the US federal guidance of one and two site-potential tree heights for default riparian reserves before watershed analysis under the Northwest Forest Plan (USDA and USDI 1994b); these are termed one-tree and two-tree buffers here. Twenty-three unthinned reference stream reaches also were included in our study, for a total of 68 stream reaches. Characterization of stream reach attributes was conducted pre and posttreatment, and is described below.

Depending on local stream network geometry, two to four different buffer widths were applied per site (Table 1). At some sites, more than one replicate of a single buffer width was applied. Upslope of buffers, perpendicular to streams, we sought distances of at least 60 m for the forest thinning treatment. Up and downstream reach boundaries usually were determined by tributary junctions, stream length, or end of water flow. A minimum reach length measured two site-potential tree heights, 110 to 150 m. Buffer configurations ranged from those positioned in a longitudinal series along a single long stream reach (Figure 2A, B) to one where the headmost stream reach was buffered by a single buffer and separated from other reaches by a tributary junction (Figure 2C, D), and sometimes also an intervening stream distance and higher-order stream segment (Figure 2E). In any case, complete independence of reaches could not be practically applied at the site scale; water either flowed from one reach into another or reaches were linked by one or more tributary junctions.

Outside the study area, upstream and downstream factors also had potential to influence study reaches (e.g., physical habitat conditions, animal dispersal). Due to amphibian life histories with possible terrestrial dispersal capabilities, it is

Table 1. Number of reach-replicates by site and riparian buffer width

	No. Reaches					
Site	Streamside retention (6 m)	Variable width (15 m min.)	One-tree height (70 m)	Two-tree height (145 m)	Unthinned control	Total
Delph Creek	1	1	0	0	1	3
Green Peak	1	1	1	0	2	5
Keel Mountain	2	1	1	1	3	8
North Soup Creek	1	1	1	0	2	5
O.M. Hubbard	1	2	0	0	2	5
Ten High	2	3	1	0	2	8
Cougar (USFS)	1	1	1	1	2	6
Grant (USFS)	2	1	0	0	3	6
Schooner (USFS)	1	4	0	0	1	6
Callahan Creek	3	2	1	1	1	8
Perkins	2	2	0	0	4	8
Total No. Reaches	17	19	6	3	23	68

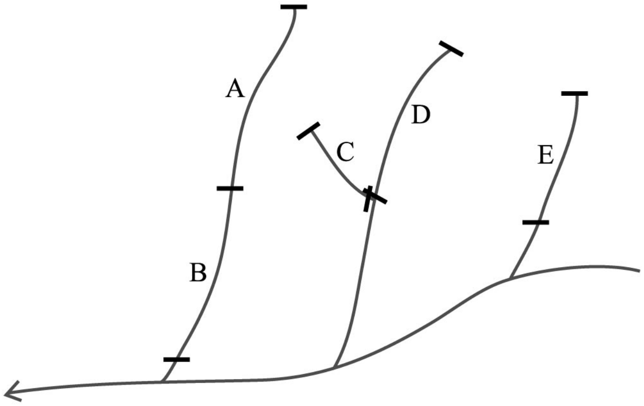

Figure 2. Schematic diagram of stream reach geometries represented in our riparian buffer study. Reaches A–E are distributed in three parallel headwater subdrainages at a site. Reaches A and B represent a longitudinal series along a continuous stream channel (reach endpoints are denoted by lines perpendicular to the channel, reach length criteria used). Reaches C and D are headmost channels where reach C flows into the stream with reach D, and the tributary junction is the downstream delineation point. Reach E is a headmost channel connected to other reaches at the site by intervening tributary junctions.

likely many populations also were connected over land between parallel stream basins. The neighborhood around the sites was a mix of stand types and management activities, including private industrial forests with recent or ongoing clearcut harvests and actively managed federal landscapes. These were larger-scale effects that were neither controlled nor examined here.

To the extent possible, streams were randomly assigned buffer widths. However, space limitations restricted the placement of wider buffers. Due to sites being in headwaters with dendritic stream networks, there was room for a single reach of the two-tree buffer at three study sites, and the one-tree buffer at six sites (Table 1).

Surveys for Habitat and Animals

Habitat and animal surveys were conducted during 1 year pretreatment, 1995–99, and in each of the first 2 years posttreatment per site, 1998–2002. Data were collected in both spring (wet season) and summer (dry season); however, most data examined here are from spring surveys. At higher elevation sites, data were collected after snowmelt when temperatures were above freezing.

Habitat.—Habitats of stream channels were typed from downstream to upstream using a modified Hankin and Reeves (1988) approach. A longitudinal sequence of habitat units was designated. Habitat units were typed as slow water (pool; flat water surface with little visible surface flow) or fast water (riffle; also included steps, cascades, glides, and runs) using a visual estimate of the dominance of the flow regime across the wetted width of the stream channel. During habitat typing, units were designated as sampleable or not for animals based on access and visibility to surveyors (e.g., if abundant down wood prohibited visibility of a unit, it was deemed unsampleable). Per unit (pool or riffle), we recorded unit size (length, wetted width, maximum depth), dominant substrate composition (visual estimate; six categories by particle diameter: fines, ≤2 mm; small gravel, 3–10 mm; large gravel, 11–100 mm; cobble, 101–300 mm; boulder, >300 mm; bedrock), number of down wood pieces (five diameter categories for pieces ≥1 m in length: 0.10–0.19 m; 0.20–0.29 m; 0.30–0.39; 0.40–0.49; ≥0.50), and stream gradient (visual estimate conducted by a single person to reduce observer bias; four categories: flat, moderate, moderately steep, steep). Due to the subjective nature of substrate and gradient data, these attributes were used only to generally describe site conditions, and were not incorporated into analyses.

Physical habitat parameters were aggregated at the reach level to characterize study streams and explore possible relations to buffer treatments or interannual variation (Table 2). Four reaches were excluded because of missing data (mostly dry reaches where stream habitat data were not collected; $n = 64$). For reach depth and width, reach averages weighted by habitat unit lengths were calculated. A dominant reach substrate category was derived by summing stream unit lengths per substrate category, calculating percentage contribution to total reach length, and selecting the category with the resulting maximum. Gradient information was based on habitat unit gradient classifications in the pretreatment survey. Reach-level gradients were determined as the gradient category accounting for the largest percentage of reach length. Pool-riffle ratios were computed as the length of reach classified as pools divided by the length of

Table 2. Reach-level physical habitat parameters analyzed for treatment effects and inter-annual variation

Parameter	n	Treatment effect	Interannual variation
Stream width	64	KW AOV1, KW AOV3	WSR**
Pool/riffle ratio	64	KW AOV1, KW AOV3	WSR**
Flow fragmentation (no. wet segments)	64	KW AOV1, KW AOV3	WSR*
Downed wood	42	KW AOV1, KW AOV3	WSR**
Hydrotype	65	Not analyzed	Chi-square

Tests for treatment effects (Kruskal-Wallis one-way analysis of variance on post minus pretreatment differences) were conducted with data from all treatment buffers combined (KW AOV1) and with data from three buffer categories (streamside retention, 6-m buffers; variable width, 30-m minimum buffers; one and two site-potential tree heights, >70-m buffers; KW AOV3). Interannual variation analysis (Wilcoxon signed rank test, WSR) was intended to detect any changes between pretreatment and posttreatment stream surveys for both reference and treatment reaches combined. *$P < 0.10$, **$P < 0.05$, n = no. treatment reaches analyzed.

reach classified as fast water. An index of stream fragmentation (continuity of water flow, no. of wet segments) was derived by counting the number of contiguous segments of wetted stream interspersed between dry stream beds.

The fragmentation data also were used to classify reach hydrotypes, based on seasonal continuity of streamflow. For this classification, "wet" reaches had no dry segments within the reach, "dry" reaches had no flow but evidence of scour and deposition, and "intermittent" reaches had dry segments within an otherwise flowing reach. An annual reach hydrotype was then defined as a pair (summer and spring) of flow characterizations. These data were drawn from habitat surveys on the same 68 reaches as those used for other analyses with the following exceptions: (1) summer survey data were included for hydrotype characterizations, where only spring data were used relative to all other data; and (2) three reaches were excluded from hydrotype analyses due to data collection errors ($n = 65$).

Unfortunately, down wood data collection methods were inconsistent among some sites and years. Compatible data for 42 of the 64 reaches (two-thirds) were used to characterize site conditions: four sites were excluded (Callahan Creek, Cougar, Grant, and O.M. Hubbard). Stream unit counts were summed for each of the five wood diameter classes to yield reach size distributions. Also, reach wood densities were calculated as total (sum of diameter classes) wood pieces per meter of reach length.

Amphibians.—Animals were subsampled in 10 units per reach, in each of the 68 reaches of our study. Units within 15 m of reach boundaries were not sampled for fauna. Habitat-typed units were sampled in proportion to their frequency in the remaining reach (e.g., if 40% of units were pools, we sampled four pools). The first unit was randomly chosen from among the first five units at the downstream end of the reach. The remaining sampled units were systematically arrayed along the reach such that every nth unit was sampled where n = no. units by type (pool or riffle) divided by total number of remaining sampleable units in that reach.

Three types of animal surveys were conducted. In wetted stream channels, either electrofishing or hand sampling was conducted. Although not reported here, we also collected data on fish in these systems. For this reason, electrofishing was used in streams with fish where hand sampling was ineffective ($n = 14$ streams). Although it is likely that amphibians were not completely sampled by electrofishing because of their occurrence within substrates, we applied equal effort among units and reaches and retained consistent sampling methods for a given reach across annual surveys. However, to avoid bias because of sampling method, we conducted analyses of treatment effects and interannual variation only with the 54 reaches that were hand sampled. We report amphibian captures by electrofishing only to provide a more comprehensive description of amphibians detected at our study sites. Hand sampling was done in reaches where fish had not been seen. Per unit, hand sampling was conducted from a downstream to upstream direction, and involved looking under all movable substrates and cover items, which were carefully replaced.

Bank sampling was conducted on each bank alongside each wetted channel unit sampled ($n = 68$ reaches). Banks were searched using a timed area search within 2 m of wetted channels. Surveyors looked in, on, and under surface cover for 5 min. per bank (10 min. total per unit). Search areas were flexibly designated such that surveyors could move a bit upstream or downstream if their bank adjacent to instream units was impractical to sample (bedrock, large bole of live tree). Survey areas were visually estimated, but generally were about 2 m wide by 4 m long. Down wood, litter, moss, and substrate were examined in layers and replaced to minimize disturbance effects.

Animals were identified to species and replaced. Summary statistics were prepared to identify species presence and their capture rates for the 68 study reaches. Total counts per species for all reaches and survey years were summed to give the total number of observations; bank and instream data were kept separate. The total counts were then divided by the sample area sums to calculate animal capture rates for only those reach samples where the species was actually observed.

Analyses

We used only 1 year of posttreatment survey data in our analyses of treatment effects and interannual variation. Although we conducted surveys in both years 1 and 2 posttreatment, these sampling time frames overlapped. Because of timber harvest activities occurring in adjacent areas to our treatment units, it was not safe to gain access to some sites immediately postharvest. Year 1 postharvest data were collected from zero to 17 months after the thinning had been completed in our study unit. Then, year 2 posttreatment surveys occurred 1 year after the first year's surveys, 11 to 29 months postharvest. To reduce the time differential in posttreatment data, we selected one of the two survey years to use in analyses. For this purpose we selected the posttreatment surveys with the most comparable dates among sites, where we chose either year 1 (five sites with later dates of survey) or year 2 (six sites with earlier dates of survey) data per site to reduce the time difference among sites. This "most-comparable year" analysis used data 7 to 18 months postharvest. Unless otherwise specified, we report only most-comparable year posttreatment results.

Animal counts and habitat parameters were examined for interannual variation and for riparian buffer treatment effects. Raw data were aggregated to reach level values, as previously described, and generally did not conform to a normal distribution. Animal capture rates, in particular, were strongly skewed to the right and changes in detections between survey years contained both zero and negative values. The presence of zero counts also would have created incomplete and unbalanced study designs for conventional analysis of variance (ANOVA) analyses. For this reason, nonparametric tests of paired pre and posttreatment reach values were considered appropriate. Consequently, we report median rather than mean values when describing results. All procedures were performed using Statistix software (version 8.1, Analytical Software, 2006).

Specifically, our analyses addressed the questions (1) Do treatment reach differences differ significantly from reference reaches (treatment effects analysis)?; and (2) Do pretreatment reaches show the same frequency distributions as posttreatment reaches (analysis of interannual variation)? Animal captures (animals m^{-2}) and four habitat parameters (Table 2) were examined for buffer treatment effects with a Kruskal-Wallis one-way analysis of variance on attribute differences (post minus pretreatment) to produce P values using a chi-squared approximation. The null hypothesis was that both groups have the same distribution. We conducted two versions of tests for riparian buffer treatment effects, comparing reference reaches with two different groupings of buffer types. One group had three buffer designations (here termed KW AOV3; streamside, variable, and a combined one-tree and two-tree "wide" category of buffer widths), and the other group consisted of all buffer designations combined into a single "treatment" group (KW AOV1). Animal detections (animals m^{-2}) and five habitat parameters (Table 2) were tested for interannual variation using the Wilcoxon Signed Rank Test (WSR, two-tailed P values for normal approximation with continuity correction). The null hypothesis was pretreatment frequency distribution equals posttreatment distribution. This analysis on interannual variation included reference reaches posttreatment, hence does not indicate findings specific to the buffer widths implemented. Comparison of reach hydrotype classifications in pre versus posttreatment years was conducted by chi-square analysis.

For tests on species captures, source data consisted of bank sampling data from all 68 reaches, and instream data from 54 reaches. The 14 reaches sampled by electrofishing were excluded from instream analyses to avoid introducing potential methodological bias into results. Species examined were restricted to the most abundant species (proportion of total animal count $\geq$5%) on reaches with animals present (i.e., pre plus posttreatment counts $>$0). In addition, a total count for all amphibians overall reaches was tested. Four taxa were included from bank data: Dunn's salamander, western red-backed salamander, torrent salamanders (southern and Cascades torrent salamanders combined because of their complementary distribution among sites; *Rhyacotriton variegatus* Stebbins and Low and *R. cascadae* Good and Wake, respectively), and rough-skinned newt (*Taricha granulosa* Skilton). Instream samples used data for three species: coastal giant salamander, torrent salamanders (again, both species combined), and rough-skinned newt.

Results

Overall, $>$3,000 individuals of 13 amphibian species were detected at our study reaches in combined pre and posttreatment surveys. Seven species were captured more often (Table 3) and six species were less frequently detected: clouded salamanders (*Aneides ferreus* Cope, n = 5

Table 3. Total species observations (n), capture rates (animals m^{-2}), and number of study sites at which each more common amphibian occurred in our western Oregon study of headwater streams

	Stream bank			Instream		
Species	n	Capture rates	No. sites	n	Capture rates	No. sites
Coastal tailed frog, *Ascaphus truei* Stejneger	21	0.014	7	213	0.082	8
Coastal giant salamander, *Dicamptodon tenebrosus* Baird and Girard	24	0.015	9	1082	0.163	11
Rough-skinned newt, *Taricha granulosa* Skilton	72	0.016	8	43	0.064	8
Torrent salamanders (two *Rhyacotriton* species): cascade torrent (*R. cascadae* Good and Wake) and southern torrent (*R. variegatus* Stebbins and Lowe)	81	0.022	10	491	0.314	9
Western red-backed salamander, *Plethodon vehiculum* Cooper	337	0.032	9	8	0.032	4
Dunn's salamander, *Plethodon dunni* Bishop	648	0.057	10	24	0.026	8

Data are all amphibian observations pooled for bank and instream samples, one pretreatment and two posttreatment years, both hand and electroshock stream samples. Capture rates are for only those reaches where the animal was actually observed.

detections overall), ensatina ($n = 32$), Oregon slender salamanders (*Batrachoseps wrighti* Bishop, $n = 17$), Pacific tree frogs (*Pseudacris regilla* Baird and Girard, $n = 10$), northern red-legged frogs (*Rana aurora* Baird and Girard, $n = 15$), and northwestern salamanders (*Ambystoma gracile* Baird, $n = 1$). Only one species, the coastal giant salamander, was detected at all 11 study sites. Along banks, all 13 species were found and total counts ranged from 1 to 648 total animals per species. Dunn's salamanders occurred at the highest capture rates (0.057 animals m^{-2}). Nine instream amphibians were detected and some of these species were found in high numbers (e.g., 1,082 coastal giant salamander observations, and torrent salamander capture rate in occupied reaches was 0.314 m^{-2}).

Headwater streams in our study were small, fast-flowing perennial and intermittent channels (Table 4, additional site attributes available in Cissel et al. 2006). Study reaches were generally shallow (median across years, 0.06 m) and narrow (median, 0.79 m). Fast water habitat tended to predominate as evidenced by an overall median pool-riffle ratio of 0.15. Streams were largely continuous, with flow fragmentation measures having a median value of 2 (number of wetted segments per reach). Five reach-level hydrotypes were present (Table 5) with intermittent flows being most common (55% of reaches for both survey years combined). Down wood counts per reach ranged from 57 to 497 (0.26 to 1.65 pieces per meter reach length) pretreatment, and 122 to 684 (0.42 to 1.68 pieces per meter reach length) posttreatment. Down wood was dominated by small wood with over half of the pieces <0.3 m diameter. Dominant substrates consisted mainly of fines (≤2 mm diameter, 40% of reaches across years; between years, 14–25% of reaches were dominated by small gravel; 17–28% of reaches were dominated by large gravel). Reach gradients were mostly moderate (8% flat, 56% moderate, 19% moderately steep, and 17% steep).

No negative treatment effects were detected for any species in either instream or bank samples, however, two species increased in captures in treatment reaches. A single bank species, the rough-skinned newt, showed a treatment effect associated with the pooled buffer widths (Figure 3A, KW AOV1, $P = 0.09$). For this species, reference reaches experienced a decline in capture rates (−0.008), while reaches with buffer treatments experienced an increase (+0.007). A similar treatment effect was detected for instream coastal giant salamander captures (Figure 3B, KW AOV1, $P = 0.04$). Once more, reference reaches experienced a decline in capture rates (−0.043), and buffer-treated reaches exhibited an increase (+0.034). For this species, the other test of buffer treatments using three buffer categories in the analysis was nearly significant (KW AOV3, $P = 0.109$). No treatment effects were seen for the four habitat attributes examined (KW AOV1 and AOV3 tests, $P > 0.10$, Table 2).

We found interannual variation in captures for one species and all species combined along banks (Figure 3A, B). There was a decrease in bank-sampled western red-backed salamanders between sample years (Figure 3A: WSR, $P = 0.0002$). The magnitude of the decreased capture rates between survey years was about 0.015 animals m^{-2}. However, no riparian buffer treatment effect was significant for this species. The reduction of this species posttreatment was probably the major contributor to the general decline in total bank amphibian captures across all reaches (Figure 3A, WSR, $P = 0.02$); again, with no significant treatment effects for all bank-occurring species combined.

Interannual variation was detected for four of five habitat parameters analyzed (Table 2). A decrease between survey years was found for pool-riffle ratios (WSR, $P = 0.0005$, decrease of about a third), and reach widths (WSR, $P = 0.01$, difference ~0.15 m). Flow fragmentation increased showing a small (0.7) but significant rise in the number of wetted segments per reach between years (WSR, $P = 0.07$). The increased flow fragmentation during posttreatment surveys is supported by a trend for increased reach intermittency; 9 of 65 (14%) total reaches (6 of 42 treatment reaches, 14%) in Table 5 occur above the diagonal, indicating they changed from a wetter channel hydrotype to a drier category posttreatment. When analyzed with a chi-square test, hydrotype showed no interannual variation ($P > 0.10$). While the size distribution of down wood pieces stayed fairly constant over the duration of the survey years, we found higher down wood densities in postharvest years (WSR, $P = 0.00$). The magnitude of the increase was about 0.3 pieces per meter reach length.

Discussion

Sixty-eight headwater stream reaches at 11 managed forest study sites in western Oregon were predominantly spatially intermittent channels that overall exhibited a rich vertebrate fauna. No adverse effect following thinning with alternative buffer widths was observed for the more common species or instream habitat conditions analyzed. These buffers with thinning may be a benign disturbance relative to headwater amphibians and habitats. However, some

Table 4. Reach-level hydrological habitat characteristics from 64 reaches

Variable	Parameter	Pretreatment	Posttreatment
Depth (m)	Median	0.07	0.06
	Minimum, maximum	0.00, 0.20	0.00, 0.28
Pool/riffle ratio	Median	0.21	0.14
	Minimum, maximum	0.00, 1.97	0.00, 1.57
Width (m)	Median	0.90	0.72
	Minimum, maximum	0.08, 3.61	0.83, 3.71
No. wet segments	Median	2	2
	Minimum, maximum	1, 20	1, 35

Table 5. Pre and posttreatment reach-level hydrotype classifications of 65 streams examined in our riparian buffer study

	Posttreatment hydrotype						
Pretreatment hydrotype	WW	WI	WD	II	ID	DD	Total
WW	12 (10)	5 (2)	0	1 (1)	0	0	18
WI	0	6 (4)	0	3 (3)	0	0	9
WD	0	0	1 (1)	0	0	0	1
II	0	2 (2)	0	32 (17)	1	0	35
ID	0	0	0	0	0	0	0
DD	0	0	0	0	0	2 (2)	2
Total	12	13	1	36	1	2	65

Parentheses indicate the 42 treatment reaches in our sample. Stream flow regime: W = wet, I = spatially intermittent, D = dry; first letter is spring designation, second letter is summer designation. Shading along diagonal shows the majority of streams did not have a change in their hydrotype classification.

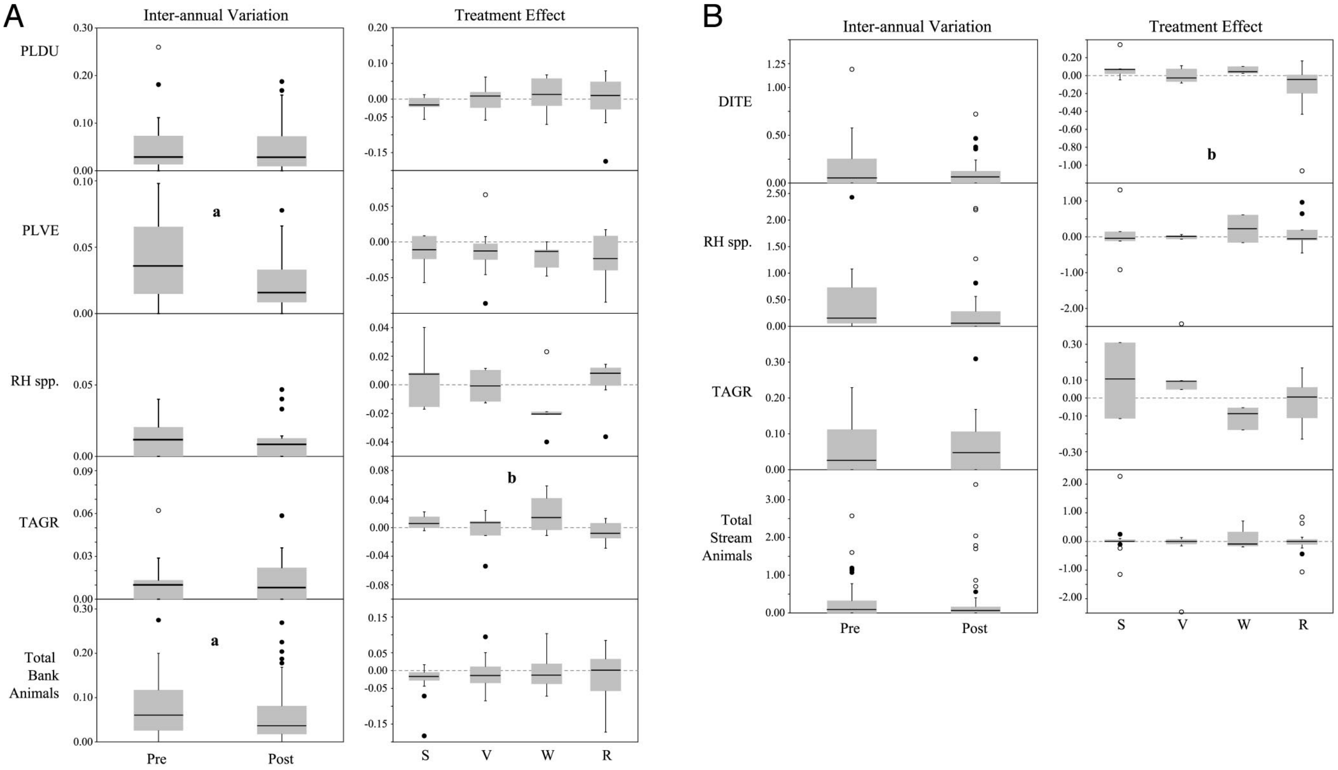

Figure 3. Capture rates (animals m^{-2}) of bank (A) and stream (B) amphibians analyzed for posttreatment trends and treatment effects. Interannual variation (left column) shows capture rates from pre and posttreatment survey years for reference and treatment reaches combined (analyzed by Wilcoxon Signed Rank tests). Treatment effects (right column) show differences in survey years (post minus pretreatment) by groupings of reference and three buffer types (analyzed by Kruskal-Wallis "AOV3"). Similar data for the analysis of reference versus all treatments combined (Kruskal-Wallis "AOV1") are not shown. Box and whisker plots include horizontal line (median), box (1st to 3rd quartile), whiskers (1st or 3rd quartile ± 1.5 × interquartile range [3rd minus 1st quartile]), closed circles (near outliers), and open circles (far outliers, >3 × interquartile range). a = significant interannual variation; b = significant treatment effect (AOV1); PLDU = Dunn's salamander, PLVE = western red-backed salamander; RH spp. = torrent salamander species; TAGR = rough-skinned newt; DITE = coastal giant salamanders; S = reaches with streamside retention buffer widths (6 m); V = reaches with variable width buffers (15 m); W = wide buffers including data from combined reaches with one- and two site-potential tree heights (≥70 m); R = unthinned reference reaches.

trends were detected among survey years that could not be attributed to treatment effects. Although our findings are tempered by constraints of our data and design, as explained in Methods and below, implications of our findings should be considered potentially relevant as hypotheses to pursue further from northwestern California to British Columbia because many taxa and site conditions may be similar across this broader forested landscape.

Headwater channels in our study appeared to serve as a nexus of amphibian assemblages. We report observations of over 3,000 individuals of 13 species throughout our pre- and 2-year posttreatment sampling period. Some taxa achieved capture rates >0.3 animals m^{-2}, although half the species were found in low numbers. All basic amphibian assemblages were represented in and along our headwater channels, with comparable species richness (four species) observed among pond-breeders, terrestrial-breeders (five species if you include Dunn's salamander in this group), and stream-breeders (four species, Jones et al. 2005). This is an interesting mix given that banks were searched only as far as 2 meters from wetted channels. These observations are consistent with Sheridan and Olson's (2003) contention that headwaters are spatially compressed systems, such that riparian zones do not extend far upslope. Emerging results of near-stream microclimates in headwater systems also support a narrow zone of relatively stable cool, moist conditions (Anderson et al. 2007, Rykken et al. 2007). As such, these strips appear to be suitable for several amphibians, perhaps serving as linear habitats for foraging and refuge. Headwater streams also may function as dispersal barriers or corridors for terrestrial breeders, which could explain their occurrences in these narrow strips. For example, if terrestrial breeders do not tend to cross open water, they may turn and traverse streamside zones, analogous to following fence lines that act as barriers for larger mammals.

Eight of 13 (61%) amphibian species we observed have status of concern somewhere in their range, and five amphibians we detected are Oregon State-sensitive species (Corkran and Thoms 1996, Jones et al. 2005). Of these five concern-species in our study area, southern and Cascades torrent salamanders and coastal tailed frogs may have specific headwater associations (e.g., Jones et al. 2005). Torrent salamanders were sometimes observed in higher numbers in our study, and may be of most concern in these particular systems due to their associations with spatially intermittent channels (Olson and Weaver 2007).

As a measure of retaining ecological integrity, managing for species persistence in forest landscapes or at the scale of managed stands might be considered an overarching biodiversity objective. In our study, the more abundant amphibians, regardless of concern status, were most often detected both pre and posttreatment; thus they persisted in reaches after thinning with our various stream buffer widths.

The changes in species' detections we found (two positive treatment effects, two cases of interannual variation) could result from alternative proximate mechanisms that were not directly examined in our study. These mechanisms include detectability, migration, and survival: (1) there may be detectability issues relative to observer bias; for example, posttreatment survey crews may have been more effective at capturing animals due to factors such as expertise or greater visibility with increased light from the adjacent harvest; (2) there may be detectability issues relative to animal behavior; for example, the surface activity of animals may have changed between survey years; (3) animals may have moved into or away from streams and banks from surrounding areas; and (4) there may have been altered survival posttreatment. The latter two alternatives may be tied to a change in surface cover, such as down wood, which was observed to increase in our study over time, or a change in site productivity and invertebrate prey; for example, increasing from increased light levels (e.g., Murphy et al. 1981). Increased growth rates, an indicator of survivorship, were reported for cutthroat trout (*Oncorhynchus clarki* Richardson) in response to logging and associated increases in light levels, invertebrate drift, and fish foraging efficiency (e.g., Wilzbach et al. 1986).

Species detectability is a growing concern for studies attempting to monitor population abundances over time (e.g., MacKenzie et al. 2003, Bailey et al. 2004). In particular, amphibian detectability can be influenced by their seasonal and diel patterns of activity, ties to weather patterns, and their cryptic tendencies. Although few studies of amphibians have quantified species-specific detectability relative to the various survey methods used, survey methods typically aim to reduce detectability concerns by focusing efforts during restricted times of optimum environmental conditions, when animals are more reliably surface-active or visible (e.g., in the Pacific Northwest, for pond-breeding amphibians see Olson et al. 1997, for terrestrial-breeding salamanders see Olson 1999). For stream amphibians in our study, although some relevant stream conditions such as microclimates may be relatively constant at headwater reaches (Anderson et al. 2007, Rykken et al. 2007), neither detectability nor variation in species occurrences over time is known. Relative to our sampling methodology these topics are under investigation to advance inventory and monitoring approaches. Nevertheless, here, our findings are constrained by a lack of understanding of species detectability during a single representative survey per year, although we have used only spring wet-season data in our analyses to reduce potential seasonal variation and did not compare data collected by hand sampling with data collected by electroshocking. Consequently, we discuss our results in terms of detections and captures rather than abundances and densities; we cannot distinguish the proximate mechanisms contributing to detection patterns, and urge caution relative to the inference of our preliminary findings which stem from the first years after thinning. Our results are best considered as hypotheses warranting further testing.

We report some patterns of changed instream habitat conditions between pre and posttreatment survey years, and these were unrelated to the buffer widths with thinning. Posttreatment, pool-riffle ratios decreased, stream widths decreased, spatial intermittency of water flow increased, and down wood pieces increased, yet it is unclear how these instream elements might relate to the amphibian patterns. Olson and Weaver (2007) examined species-habitat relationships at these sites and reaches during pretreatment surveys, and found torrent salamanders were particularly associated with spatially intermittent stream segments in headwaters. Thus, the greater fragmentation observed here posttreatment could favor this taxon; we did not observe a change in detections of these animals, however. Conversely, Olson and Weaver (2007) found coastal giant salamanders and fishes were tied to perennial stream channels, so this assemblage could be reduced from channels that become fragmented. Our initial results showed an increase in coastal giant salamanders in response to treatments. We are monitoring instream and bank species and habitats in posttreatment year 5 to follow these trends and assess corresponding species patterns. A closer inspection of bank habitat conditions may be warranted to explain our bank-species trends.

Western red-backed salamanders are not restricted to stream bank habitats and occur from streamsides into upslope forests (Jones et al. 2005, Rundio and Olson 2007). Although it is possible that our trend for lower captures of this salamander posttreatment could be a signal of a populationwide response to the site-level disturbances, our understanding of their relatively limited mobility argues against this. In a mark-recapture study in British Columbia, their movements were on the order of meters (e.g., Ovaska and Davis 2005). However, movement patterns in Oregon are unknown and they may have wider home ranges, and they may move more in response to a disturbance. Alternatively, it is possible that the reduced numbers of western red-backed salamanders may correspond to the upslope study by Rundio and Olson (2007). They reported a reduction in western red-backed salamander captures in upslope thinned areas at one of two sites examined, which were sites also examined by us in this article (Green Peak and Keel Mountain). They suggested that site conditions could ameliorate treatment effects and cited differences between these two sites that might be relevant for ground-dwelling salamanders (e.g., legacy large down wood). If a site-specific negative effect were to occur within our stream or bank data, it would likely be occluded in our overall analysis. Unfortunately, examining each site as a case study was not practical with our stream and bank data because of lack of buffer replication within sites and low or variable capture rates among reaches within sites.

Additional caveats should be considered when evaluating our findings. Few treatment effects may have resulted from other factors, including lack of reach independence, relatively small spatial scale of study reaches and the study site, and up and/or downstream effects from off-site occurrences. These might affect reach-level analyses if animals disperse among reaches or in/out of sites. Pond-breeders such as rough-skinned newts appear relatively mobile such that changes in their numbers may reflect longer distance movements (Jones et al. 2005). Few studies address movements of headwater-associated amphibians. Sagar (2004) found juvenile coastal giant salamanders moved very little in small streams, averaging about 3 m in summer and 15 m over winter. Such limited movement supports independence of our reaches relative to this life history stage. Many terrestrial-breeding salamanders similarly may have limited dispersal; for example, as mentioned above, western red-backed salamanders have been reported to move only a few meters (Ovaska and Davis 2005). Also, Olson and Weaver (2007) reported that species assemblages could differ among adjacent stream reaches within sites, for the same sites and stream reaches examined here. In these cases, animals among reaches may not be interacting over short time frames, such as the few years of our study, yet both Bisson et al. (2002) and Stoddard and Hayes (2005) detected longer-term landscape level effects on stream animal abundance. The temporal or spatial scale of our study may not have been large enough to detect such effects.

The idea that thinning to 200 tph was a relatively mild disturbance relative to species has been supported by other studies conducted at subsets of sites examined here. One study, at three of these sites (Green Peak, Delph Creek, Keel Mountain) and one other site not examined here, compared upslope biodiversity between thinned and unthinned areas, as well as the upslope leave islands (Wessell 2005). She found only nine of 112 (8%) faunal measures (abundances of species, functional groups, diversity measures) differed ($P < 0.10$) between thinned and unthinned upslope areas, with two positive and seven negative thinning effects observed. Second, Anderson et al. (2007) found temperatures and relative humidities at streams with the various buffer widths and upland thinning were similar to unthinned reference reaches. Similarly, at other sites, Rykken et al. (2007) found streamside microclimates did not differ between intact forests and clearcuts with riparian buffers. Their data support a strong "stream effect" of cool, moist microclimates radiating upslope, and these zones are well retained with buffers adjacent to clearcuts. However, it should be noted that short-term species effects may be sublethal and not be detected in the first 2 years, the time frame of this study (e.g., reduced reproduction and recruitment of young), yet may appear after a lag time. To hedge these types of uncertainties as well as the issues of our design and data constraints (e.g., limited inference), where retention of headwater amphibians is a priority, it may be prudent to deploy a mix of riparian buffer strategies within subdrainages with density management timber harvest upslope.

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